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Risk Assessment of Spent Foundry Sands In Soil-Related Applications

Evaluating Silica-based Spent Foundry Sand From Iron, Steel, and Aluminum Foundries

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List of Abbreviations

Abbreviation	Definition
2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
3MRA	Multipathway, Multimedia, Multireceptor Risk Assessment Modeling System
ADD	average daily dose
AGS	aluminum green sand
AMC	antecedent moisture class
ARS	Agricultural Research Service
ASE	accelerated solvent extractor
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
AWC	available water capacity
BGS	brass green sand
BTEX	benzene, toluene, ethylbenzene, and xylenes
CASRN	Chemical Abstract Service Registry Number
CEC	cation exchange capacity
CERCLIS	Comprehensive Environmental Response, Compensation, and Liability Information System
COC	constituent of concern
CONUS	contiguous United States
CPP	chemical properties processor
CR	consumption rate
CSEFH	Child Specific Exposure Factors Handbook
CSF	cancer slope factor
CV	coefficient of variation
DAF	dilution attenuation factor
DHA	dehydrogenase activity
DMG	dry matter growth
DTPA	diethylenetriamine pentaacetic acid
DW	dry weight
DWEL	Drinking Water Equivalent Level
Eco-SSL	ecological soil screening level
EFH	Exposure Factors Handbook
EI	erosivity index
EP	extraction procedure
EPA	U.S. Environmental Protection Agency
EPACMTP	EPA's Composite Model for Leachate Migration with Transformation Products
ET	Evapotranspiration

Abbreviation	Definition
EXAFS	extended X-ray absorption fine structure spectroscopy
FC	field capacity
foc	fraction organic carbon
GC-MS	gas chromatography-mass spectrometry
GIRAS	Geographic Information Retrieval and Analysis System
GIS	geographic information system
GSCM	Generic Soil Column Model
HGDB	Hydrogeologic DataBase for Modeling
HH-SSL	human health soil screening level
HMTA	hexamethylenetetramine
Hp	hepta
HQ	hazard quotient
HS-SPME	headspace solid-phase microextraction
Hx	hexa
ICP-AES	inductively coupled plasma-atomic emission spectrometry
ICP-MS	inductively coupled plasma-mass spectrometry
IGS	iron green sand
IRIS	Integrated Risk Information System
ISC3	Industrial Source Complex Model version 3
ISCST3	Industrial Source Complex-Short Term Model version 3
IUPAC	International Union of Pure and Applied Chemistry
IWEM	Industrial Waste Management Evaluation Model
LADD	lifetime average daily dose
LDH	layered double hydroxide
LOEL	lowest observable effects level
LOQ	limit of quantitation
MAF	moisture adjustment factors
MCL	Maximum Contaminant Level
MDI	diphenylmethane-4,4-diisocyanate
MDL	method detection limit
MRL	Minimum Risk Level
NAAQS	National Ambient Air Quality Standard
NAPL	nonaqueous phase liquid
NBS	steel phenolic urethane no-bake sand
NIST	National Institute of Standards and Technology
NOEL	no observable effects level
NOM	natural organic matter
NPDWS	National Primary Drinking Water Standards
NRC	U.S. Nuclear Regulatory Commission

Abbreviation	Definition
NSDWS	National Secondary Drinking Water Standard
NWS	National Weather Service
O	octa
OC	organic carbon
ORCR	Office of Resource Conservation and Recovery
OSHA	Occupational Safety and Health Administration
OSU	The Ohio State University
OSWER	Office of Solid Waste and Emergency Response
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PDF	probability distribution function
Pe	penta
PET	potential evapotranspiration
PM_{10}	particulate matter with a mean aerodynamic diameter of 10 microns or less
ppm	parts per million
PPRTV	Provisional Peer-Reviewed Toxicity Value
PQL	practical quantitation limit
PVC	polyvinyl chloride
RCRA	Resource Conservation and Recovery Act
RDMG	relative dry matter growth (relative to controls)
REL	Reference Exposure Level
RfC	Reference Concentration
RfD	Reference Dose
RSD	relative standard deviations
SAB	Science Advisory Board
SAMSON	Solar and Meteorological Surface Observation Network
SCRAM	Support Center for Regulatory Air Models
SCS	Soil Conservation Service
Sdev	standard deviation
SFS	spent foundry sand
SMFC	Soil moisture field capacity
SMS	spent mushroom substrate
SMWP	Soil moisture wilting point
SOM	soil organic matter
SPLP	synthetic precipitation leaching procedure
SSL	Soil Screening Level
STATSGO	State Soil Geographic Database

Abbreviation	Definition
T	Tetra
TCLP	Toxicity Characteristic Leaching Procedure
TEF	toxic equivalency factor
TEQ	toxic equivalency value
UAC	unitized air concentration
USDA	U.S. Department of Agriculture
USLE	Universal Soil Loss Equation
WHO	World Health Organization
WMU	waste management unit
WP	wilting point
WW	wet weight

Executive Summary

<u>Purpose</u>: To provide states with a sound scientific basis from which to evaluate the health risks to human and ecological receptors associated with the beneficial use of silica-based spent foundry sand (SFS) from iron, steel, and aluminum foundries in soil-related applications.

<u>Within the scope and limitations of this evaluation, the following conclusions were drawn</u>:

- Metals found in SFS are present at concentrations similar to background in U.S. and Canadian soils.
- The conclusions of this report apply to silica-based SFS from iron, steel, and aluminum foundries.
- The evidence demonstrates that the evaluated uses of silica-based SFS produced by iron, steel, and aluminum foundries (i.e., used in manufactured soil, in soil-less potting media, and in road subbase) were found to be protective of human health and ecological receptors.

Roughly 2.6 million tons of SFS is beneficially used each year outside of the foundries, of which 14% is used in soil-related applications (USEPA, 2008c). In 2002, the U.S. Department of Agriculture's Agricultural Research Service (USDA-ARS) implemented the Foundry Sand Initiative under National Program 206 (Manure and Byproduct Utilization; renamed since to NP 214 - Agricultural and Industrial Byproducts) to address agricultural and horticultural uses of SFS. A collaborative effort was initiated to evaluate the potential risks of using SFS in soil-related applications and to encourage this beneficial use if found to be protective of human health and the environment. USDA-ARS, The Ohio State University (OSU), and the U.S Environmental Protection Agency (EPA) formed an expert team of agronomists, soil scientists, and environmental health risk assessors to develop an SFS-specific risk assessment. The overall goals for this document were to:

- Review the available information on SFS in soil-related applications
- Identify likely exposure pathways and receptors associated with various use scenarios
- Use a combination of screening and modeling methods to determine whether the proposed unencapsulated uses of SFS are protective of human health and the environment
- Discuss the findings within the context of certain overarching concepts (e.g., the complexities of soil chemistry) and provide conclusions.

Reviewing Available Information: SFS Characterization

Forty-three samples of spent molding and core sand from U.S. foundries were collected and analyzed by USDA-ARS and OSU. Other materials, such as broken or unused cores, or floor sweepings from core room operations, were not examined in this evaluation. The characteristics of the samples taken are as follows:

- **Metal cast:** 4 aluminum sands, 31 iron sands, 6 steel sands, and 2 non-leaded brass sands

  Only nonhazardous SFSs are within the scope of this evaluation. Sands from brass and bronze foundries that use lead are frequently hazardous waste because they leach lead at levels above the federal regulatory limit (see 40 CFR 261.24). Therefore, sands from leaded brass and bronze foundries were not collected, and such sands were not evaluated in this study.
- **Mineral type:** 41 silica sands and 2 olivine sands
- **Binder type of molding sand:** 36 green sands and 7 chemically bound sands.

USDA collected the initial 43 samples in June 2005. To test variation over time, USDA trained foundry personnel in proper collection techniques, and most foundries collected and sent USDA two additional sample sets, in September 2005 and July 2006.[1] USDA conducted total constituent testing on all samples for elements (metals and metalloids), polycyclic aromatic hydrocarbons (PAHs), and phenolics. Ten of the June 2005 samples were also analyzed for dibenzodioxins, dibenzofurans, and dioxin-like polychlorinated biphenyls (PCBs).

OSU also conducted total constituent testing on the initial 43 samples for elements. The test method that OSU used had a lower detection limit than the method used by USDA, and was therefore able to more accurately estimate concentrations at the lower end of the range.

To characterize the leaching behavior of trace elements, USDA conducted leach tests on SFS using the toxicity characteristic leaching procedure (TCLP), the synthetic precipitation leaching procedure (SPLP), and the American Society for Testing and Materials (ASTM) International method D 3897.[2] The conditions simulated by SPLP (leaching from soil due to acid rain) and the ASTM method (material's native leaching potential) are more relevant than TCLP (highly acidic leaching in a municipal waste landfill) for evaluating the conditions considered in this report. Therefore, TCLP leach data were only used in this evaluation if SPLP or ASTM leach data were not available.

To assess plant uptake of trace metals, USDA grew spinach, radishes, and perennial ryegrass in a 50% SFS mixture with added nutrients. Spinach and radish experienced typical levels of elements. Ryegrass, on the other hand, was found to be iron deficient and contained elevated but nontoxic concentrations of boron, manganese, and molybdenum.

USDA also assessed the potential of SFS to impact soil invertebrates. This was done in a 28-day experiment where earthworms were placed in blends of 10%, 30%, and 50% SFS. The worms did not exhibit higher levels of any elements, except in the samples from the two non-leaded brass foundries.

Data were identified from industry, academia, and the peer-reviewed literature. However, based on the number, geographic distribution and types of sampled foundries and SFS, and the breadth of aspects studied, as well as the types of analytical methods used and the level of QA/QC built into the studies, the USDA and OSU datasets are considered the most complete and

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[1] 38 foundries (88%) sent samples in September 2005, and 37 foundries (86%) sent samples in July 2006. 79% of foundries sent samples on both dates.

[2] TCLP (U.S. EPA SW-846, method 1311, U.S. EPA, 2007a)
SPLP (U.S. EPA SW-846, method 1312, U.S. EPA, 2007a)
ASTM (ASTM International, 2004)

scientifically robust. The risk assessment therefore used the OSU totals dataset because it more accurately represented the low end of concentration ranges, and the USDA leachate data.

The existing data on non-leaded brass sands and olivine sands demonstrated levels of copper, lead, nickel, and zinc that were both potentially phytotoxic and much higher than the other 39 SFSs, but insufficient samples existed to characterize constituent concentration variability. Therefore, while descriptions of non-leaded brass sands and olivine sands are retained for completeness, they are not evaluated in the risk assessment, and any risk assessment findings apply only to silica-based SFS from iron, steel and aluminum foundries.

Identifying Likely Exposure Pathways/Receptors: Conceptual Model

The purpose of this analysis is to evaluate whether the use of silica-based SFSs from iron, steel, and aluminum foundries will be protective of human and ecological receptors in the United States if the SFSs are used in manufactured soils, soil-less potting media, or road subbase. This evaluation defines "protective" as a reasonably maximally exposed individual incurring no more than a 10^{-5} excess risk of cancer, or for noncancer effects, exposures to ensure that the effects would not be expected over a lifetime, for both human and ecological receptors.

As discussed in Chapter 3, the quantitative evaluation focused on the use of SFS in manufactured soils (comprised of 50% SFS, by weight), because potential exposure to human and ecological receptors from constituents of concern was judged to be higher than potential exposures in the other two uses. Therefore, if the potential for adverse effects to human and ecological receptors from SFS-manufactured soils was found to be protective, then the other two uses would also be protective.

The exposure scenarios that were judged to have the greatest potential for human and ecological exposure from the use of SFS in manufactured soils included residents living near commercial blending facilities,[3] home gardeners that use SFS-manufactured soils, and ecological receptors that come in contact with these home gardens. The conceptual models developed for these scenarios describe potential exposures to adult and child receptors through three basic pathways: (1) *groundwater pathway* - the ingestion and dermal exposure to groundwater contaminated by the leaching of SFS constituents; (2) *ambient air pathway* - the inhalation of SFS emitted from soil blending operations; and (3) *soil pathway* – the incidental ingestion and dermal exposure to SFS-manufactured soil, as well as ingestion of fruits and vegetables grown in the soil. The conceptual models included exposures to ecological receptors through direct contact with SFS-manufactured soil.

Screening and Modeling

Analytical data were available for 25 metals, 16 PAHs, 17 phenolics, and 20 dioxins and dioxin-like compounds. In Phase I (screening), the SFS data and available screening criteria (e.g., available health benchmarks, media-specific screening levels) and models were used to determine which constituents, if any, required further evaluation. Phase II (risk modeling) used constituent-, regional- and site-specific data to address the variability in home garden conditions across the country.

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[3] Commercial soil blending facilities use construction equipment, such as a front-end loader, to combine large volumes of the various mineral and organic components to manufacture soil.

Each of the three pathways identified above was evaluated individually. In addition, the soil pathway evaluation used screening levels that also addressed inhalation exposures. The exposure scenarios and pathway evaluations were developed to produce conservative risk estimates; that is, the methodology was designed to estimate risk from reasonable maximum exposure, to ensure that the analysis included an ample margin of safety. This approach ensures that the results of this analysis can be used to determine if soil-related uses of SFS are protective of human health and the environment. The risk assessment provides decision makers with information on the potential for adverse effects to the reasonably maximally exposed individuals and ecological receptors that could come in contact with SFS.

Phase I Results

All PAHs, phenolics, and dioxin and dioxin-like compounds were screened out of all three pathways, and therefore required no further evaluation. Inhalation screening eliminated all SFS constituents (i.e., including the metals) from further evaluation; the inhalation pathway itself therefore required no further evaluation. Dermal screening of soil and groundwater exposure likewise found that all evaluated constituents were well below a level of concern, and dermal exposure was also eliminated from further evaluation. However, based on groundwater ingestion screening, soil multi-pathway exposure screening and ecological screening, 11 metals were retained for further evaluation in the risk modeling phase. **Table ES-1** lists the metals retained for risk modeling.

Table ES-1: Phase I Results – SFS Constituents Requiring Further Evaluation

	Human	Ecological
Groundwater Pathway		
	Antimony (Sb) Arsenic (As) Beryllium (Be) Cadmium (Cd) Lead (Pb)	Not evaluated
Inhalation		
	All constituents below a level of concern. No need for further inhalation evaluation	Not evaluated
Soil/Produce		
	Arsenic (As) Cobalt (Co) Iron (Fe)	Antimony (Sb) Chromium (Cr) Copper (Cu) Manganese (Mn) Nickel (Ni)

Phase II Results

The SFS concentrations of all eleven modeled constituents fell below their respective human and ecological modeled SFS-specific screening levels.

Table ES-2 summarizes the analytical and background soil information on metal constituents in SFS.[4] Human health SSLs and Eco-SSLs are provided. In addition, the table provides the modeled screening values for the specific home gardener scenario developed in this report, as well as modeled screening values based on median and high-end consumption by the general public. As shown in this table, there is substantial evidence that the metal constituents found in SFS are present at concentrations that are very similar to those found in native soils.

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[4] Table ES-2 lists only metals because all organics were screened out early in the analysis. Discussions and results of the screening of organics can be found in Chapter 4.

Table ES-2. Comparing SFS Concentrations to Various Screening Values (mg kg^{-1}dry weight, unless otherwise noted)

	Silica-based Iron, Steel, and Aluminum Sands[a]				Human Screening Values				Eco Screening Values			U.S. and Canadian Surface Soils[b]		
						Modeled Consumption Rates[c]								
Elements	**Max**	**95%-ile**	**Median**	**SFS-Manuf. Soil**	**SSL[d]**	**Home Gardener**	**Gen. Pop. Median**	**Gen. Pop. High**	**Eco-SSLs[e]**	**Modeled (SFS-specific)**	**USDA[f]**	**Max**	**95%-ile**	**Median**
Al (g kg^{-1})	11.7	11.2	5.56	5.6	77	--	--	--	ND	--	--	87.3	74.6	47.4
As	7.79	6.44	1.05	3.22	6.7[g]	8.0	30	9.1	18	40	--	18.0	12.0	5.0
B	59.4	20.2	10.0	10.1	16,000	--	--	--	ND	--	--	ND	ND	ND
Ba	141	17.7	5.00	8.85	15,000	--	--	--	330	--	--	1800	840	526
Be	0.60	0.38	0.15	0.19	160	--	--	--	21	--	--	4.0	2.3	1.3
Cd	0.36	0.20	0.05	0.10	70	--	--	--	0.36	--	--	5.2	0.6	0.2
Co	6.62	5.99	0.88	3.00	23	22	58	21	13	--	--	143.4	17.6	7.1
Cr (III)	115	109	4.93	54.5	120,000	--	--	--	34	510	--	5320	70.0	27.0
Cu	137	107	6.22	53.5	3,100	--	--	--	49	159	200	81.9	30.1	12.7
Fe (g kg^{-1})	64.4	57.1	4.26	28.9	55	160	230	150	ND	--	--	87.7	42.6	19.2
Mn	707	670	54.5	335	1,800	--	--	--	220	1000	--	3,120	1,630	490
Mo	22.9	21.8	0.50	10.9	390	--	--	--	ND	--	--	21.0	2.16	0.82
Ni	117	102	3.46	51.0	1,500	--	--	--	38	290	200	2,314	37.5	13.8
Pb	22.9	15.3	3.74	7.65	400	--	--	--	56	--	--	244.6	38.0	19.2
Sb	1.71	1.23	0.17	0.62	31	--	--	--	0.27	4.1	--	2.3	1.39	0.60
Se	0.44	0.20	0.20	0.10	390	--	--	--	0.52	--	--	2.3	1.0	0.3
Tl	0.10	0.09	0.04	0.05	0.78	--	--	--	ND	--	--	1.8	0.7	0.5
V	11.3	9.90	2.88	4.95	390	--	--	--	280	--	--	380	119	55
Zn	245	72.1	5.00	36.1	23,000	--	--	--	79	--	300	377	103	56

-- = No modeled value was generated because constituent was screened out of further study in an earlier stage of the evaluation. If a constituent screened out based on human health SSL and had no Eco-SSL, the constituent was considered to have screened out for both human and eco.

ND = No Data.

[a] Source: Dayton et al. (2010).

[b] Source: Smith et al. (2005).

[c] See **Chapter 5** for a detailed discussion of how the modeled values were generated.

[d] Concentrations of SFS constituents in manufactured soil (a 1:1 blend) were compared to an order-of-magnitude below the SSLs listed here, as discussed in **Chapter 4, Section 4.4.3**. Values are from EPA Regional Screening Tables (http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm). Unless otherwise noted, all values are based on noncarcinogenic impacts.

[e] Concentrations of SFS constituents in manufactured soil (a 1:1 blend) were compared to the Eco-SSLs, as discussed in **Chapter 4, Section 4.4.3**.

[f] See **Appendix C** for an explanation of USDA Phytotoxicity Screening Values for copper, nickel, and zinc.

[g] Based on carcinogenic risk, set at the standard EPA Office of Resource Conservation and Recovery risk target level of 1E-05.

Risk Characterization

Risk characterization summarizes the various lines of evidence presented earlier in the evaluation and discusses them within the context of the conservative nature of screening risk assessment and the complexities of soil chemistry. First, the context was set by reviewing the high-level risk questions that the evaluation was designed to address, and by introducing overarching concepts while reviewing constituent-specific information. Second, constituent-specific information was reviewed and conclusions drawn.

When reviewing the various lines of evidence, it is important to keep in mind the key risk assessment questions that this evaluation was designed to answer:

- Will the addition of SFSs to soil result in an increase in the constituent concentrations in soil relative to background levels, and how should the results of the risk assessment be interpreted across varied national soils?
- How do constituent forms found in the SFS matrix behave with respect to bioaccessibility and bioavailability, and how does that affect potential risks?
- How will the behavior of individual constituents in manufactured soil, such as the soil-plant barrier, impact the potential for exposure through the food chain pathway and, ultimately, the potential for adverse human health and ecological effects?
- How do the risk assessment results compare to levels required to maintain nutritional health in plants and animals?

When reviewing the various lines of evidence, there are also a number of other overarching concepts to consider:

- **Background Concentrations.** Comparing the 95th percentile metal concentrations in U.S. and Canadian soils to silica-based U.S. iron, steel, and aluminum SFSs reveals that the concentrations of most trace metals in SFSs are below background concentrations in U.S. and Canadian soils.
- **Chemical Reactions in Soil.** Metals reaching soils in elemental forms will oxidize rapidly depending on the redox characteristics of the metal and the soil. Sorption is a chemical process that buffers the partitioning of trace metals between solid and liquid phases in soils and byproducts. Metal cations can sorb onto the metal oxides referred to above, as well as onto soil organic matter.
- **Soil-Plant Barrier.** Soil chemical processes may limit the availability of metals for uptake, while phytotoxicity limits the chances that contaminated plants will be consumed (i.e., plant death acts as a barrier to contamination up the food chain).
- **Interactions among Constituents.** The presence (or absence) of some metals may affect the toxicity of other metals. For example, copper-deficiency-stressed animals are more sensitive to dietary zinc than animals fed with copper-adequate diets. Also, increased zinc in forage diets strongly inhibits cadmium absorption and reduces liver and kidney cadmium concentrations in cattle.
- **SFS use as a manufactured soil component**. The evaluation considered a high end use: a 20 cm layer of manufactured soil containing 50% SFS (dry weight) in the blend. Blends are much more likely to include 10% or less SFS (dry weight).

Considering all of the above, and based on the evidence, most constituents were well below SSLs and Eco-SSLs. Those that required further, more refined study were found to be below levels of concern.

Conclusions

This assessment, driven by conservative assumptions and risk screening models, indicates that the silica-based SFSs from iron, steel, and aluminum foundries evaluated in this report do not pose risks of concern to human health or ecological receptors when used in manufactured soils. Among other lines of evidence, the constituent concentrations in SFS-manufactured soils are at or below relevant regulatory and health-based benchmarks for human and ecological receptors. Because human and ecological exposure potential is lower for use in soil-less potting media or road subbase than it is for use in manufactured soil, we similarly conclude that these SFSs do not pose risks of concern when used in soil-less potting media, or road subbase.

Any conclusions drawn by this risk assessment should be understood within the limitations and scope of the evaluation, including the following:

- Only silica-based SFS from iron, steel and aluminum foundries are evaluated. In contrast, SFS from leaded brass and bronze foundries often qualify as RCRA hazardous waste. Also, there weren’t sufficient data to characterize SFS from non-leaded brass foundries and SFS containing olivine sand, and therefore these SFSs are not evaluated in this risk assessment.
- In addition to SFS, foundries can generate numerous other wastes (e.g., unused and broken cores, core room sweepings, cupola slag, scrubber sludge, baghouse dust, shotblast fines). This assessment, however, applies only to SFS as defined in the assessment: molding and core sands that have been subjected to the metalcasting process to such an extent that they can no longer be used to manufacture molds and cores. To the extent that other foundry wastes are mixed with SFS, the conclusions drawn by this assessment may not be applicable.
- Samples from 39 foundries (totals and pore water data from 39 samples, and leachate data from 108 samples) were used to represent silica-based SFS from all iron, steel, and aluminum foundries in the U.S. Because the foundries were not chosen randomly, there is uncertainty regarding whether the data are statistically representative of SFS from all iron, steel, and aluminum foundries. However, these foundries were specifically selected to ensure that the full range of constituents and their concentrations were adequately represented, and the analytical data from these samples are the best available for characterizing SFS constituents.
- Analytical data were available for 25 metals, 16 PAHs, 17 phenolics, and 20 dioxins and dioxin-like compounds. USDA analyzed for organic compounds that are major binder components (i.e., phenolics) or might be generated during thermal degradation of chemical binders and other organic additives (i.e., PAHs, dioxins, furans), because these constituents present the greatest hazard if at elevated levels in the environment. Review of the scientific literature for evidence of additional organic compounds present in SFS indicated that they were well below levels of concern.
- Screening and modeling evaluated those constituents for which toxicity benchmarks exist.

- Evaluated beneficial uses include manufactured soil, soil-less growth media and road subbase. The home garden using SFS-manufactured soil was modeled because it demonstrated the greatest potential for exposure.

The beneficial use of spent foundry sand, when conducted in an environmentally sound manner, can contribute significant environmental and economic benefits. These benefits can include reduced energy use, water consumption, and greenhouse gas emissions. An EPA analysis indicates current reuses in road base and manufactured soil result in energy savings of 43 billion BTUs per year, 7.8 million gallons of water, and prevention of more than 4,000 tons of greenhouse gas emissions.

Based on the conclusions of the risk assessment conducted for the specific SFSs applications as stated above, and the available environmental and economic benefits, the EPA and USDA support the beneficial use of silica-based SFS specifically from iron, steel and aluminum foundry operations when used in manufactured soils and soil-less potting media, and roadway construction as subbase. Consistent with the assumptions, limitations, and scope of this analysis, the beneficial uses of SFSs also provide significant opportunities to advance Sustainable Materials Management (SMM) (http://www.epa.gov/smm).

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1. Introduction

Industrial and municipal byproducts were once traditionally viewed as wastes, but their application to soils is now being practiced in the United States and many countries around the world. A number of industrial byproducts have proven beneficial uses in agronomic settings, including byproducts from coal combustion, fertilizer production, construction, and incineration (Stout et al., 1988; Korcak, 1995; Wright et al., 1998; Clark et al., 1999), and paper manufacturing (Beyer and Mueller, 1995; Phillips et al., 1997; Aitken et al., 1998; Simard et al., 1998; Zibilske et al., 2000). Many of these byproducts can provide nutrients to crops or improve the physical and chemical properties of soil. Because the beneficial use[5] of these materials has been shown to improve physical, chemical, and biological properties of soils, there currently exists a demand for the approved use of these byproducts as low-cost soil amendments, as well as for other uses (e.g., road construction). Of these byproducts, spent foundry sand (SFS) has emerged as a material that may be currently underutilized in the production of manufactured soils and other soil-related applications.

Foundries purchase virgin sand to create metalcasting molds and cores. The sand is reused numerous times within the foundry itself. However, mechanical abrasion during the mold-making process and sand reclamation, and exposure to high casting temperatures causes the sand grains to eventually fracture. The fracturing changes the shape of the sand grains, rendering them unsuitable for continued use in the foundry. The resulting residuals are generally managed as a waste or beneficially used. A single foundry can generate numerous wastes, including spent molding and core sands, unused and broken cores, core sand waste, core room sweepings, cupola slag, scrubber sludge, baghouse dust, and shotblast fines. However, only spent molding and core sands from ferrous and nonferrous foundries were considered in this assessment. That is, for the purpose of this assessment, SFS will be used to indicate molding and core sands that have been subjected to the metalcasting process to such an extent that they can no longer be used to manufacture molds and cores. While not all molds contain cores (e.g., solid casting), molds that do contain cores generally produce a commingled waste. Therefore, SFS should also be considered a byproduct that contains only spent molding sand, or spent molding and core sand. Core butts, which are pieces of core that did not break down to grain size after the casting process, were not considered in this evaluation.

Approximately 2.6 million tons of the SFS produced annually are beneficially used outside of the foundries, of which 14% is used in soil-related applications (USEPA, 2008c). Spent foundry sand has been used as a substitute for virgin sand in certain markets. These markets generally can be divided into three groups:

- *Highway and Construction Uses* – SFSs have been shown to perform well in bases and subbases under roadways, paved surfaces and structures. In pavement surfaces, SFSs are also used in hot mix asphalt and in portland cement concrete products.

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[5] The term “beneficial use,” as defined in this document, is the reuse of an industrial material in a product that provides a functional benefit; that may replace a product made from virgin raw materials, thus conserving natural resources that would otherwise need to be obtained through practices such as extraction; and that meets relevant product specifications and regulatory standards.

- *Aggregate Substitutes* – SFSs substitute for other fine aggregates in products that are bound together in some manner. Such products include: portland cement, ready mix concrete, pre-cast concrete, bricks, blocks and pavers, grouts and mortars, ceramic tiles and other manufactured products where sand is a raw material.
- *Manufactured Soils* – Nurseries and landscaping companies are manufacturing soils by blending SFSs with low-grade soils and organic materials.

Spent foundry sands are potentially useful in manufactured soils because of their uniformity, consistency, and dark color in the case of green sands. The sands can be blended with soils and/or organic amendments (e.g., peat, composted yard waste, manures, biosolids) to develop manufactured soils suitable for horticultural, landscaping, and turfgrass applications (Jing and Barnes, 1993; Nayström et al., 2004; Lindsay and Logan, 2005). A high sand content (as much as 50% by weight) is required in manufactured soils to reduce compaction and increase water movement, especially in high foot traffic soils such as golf putting greens and athletic fields (Swartz and Kardos, 1963; Brown and Duble, 1975; Davis, 1978; Taylor and Blake, 1979; Baker, 1983). A laboratory study by McCoy (1998) demonstrated that progressive increases in the sand content of silt loam and loam soils while maintaining a low organic matter content greatly improved the quality of soil with respect to compaction properties and water movement. In addition, SFSs have also been successfully used in non-agricultural applications, for example, highway subbases, structural fills, flowable fills, cement, concrete, pipe bedding, and backfill (Naik et al., 1994; Leidel et al., 1994; FIRST, 2004; Abichou et al., 2004; Guney et al., 2006; Deng and Tikalsky, 2008). Spent foundry sands may also be useful as a low-cost reactive medium to remove trace elements and organics from contaminated water (Lee et al., 2004a, b; Lee and Benson, 2004).

While SFSs satisfy the engineering and other performance specifications for many of the above-mentioned applications, their use has been constrained in many states, especially as an ingredient in manufactured soils and for land application. The unencapsulated[6] use of SFS is of particular concern to many states because the application to land poses the highest potential for human and ecological exposure to chemical constituents found in the material. To address this concern for SFS and other byproducts, a number of states have established beneficial use programs for industrial materials. With the increase in environmental, legislative, and economic activities that are favorable to beneficial use of industrial byproducts, more states are beginning to develop such beneficial use programs. States are generally receptive to beneficial use proposals from industry that are backed by sound science, but frequently lack the necessary resources to determine whether or not the proposed use could pose significant risks to human health and the environment. Questions also persist among regulators and scientists as to whether the levels of trace elements and organic compounds in industrial materials will cause adverse effects to ecosystems or humans. Consequently, the availability of an evaluation based on sound science would be enormously helpful to states that are just beginning to develop programs to evaluate the beneficial use of SFS (Kauffmann et al., 1996), and for states with existing

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[6] Unencapsulated use is sometimes also referred to as unconsolidated or unbound use and means that the material is not bound chemically or physically within a matrix such as cement or asphalt.

programs, such a risk assessment could serve as a confirmation of current methods or a template to further refine and improve current methods used in evaluating beneficial use proposals.

Developed through collaboration between the U.S. Environmental Protection Agency (EPA), the U.S. Department of Agriculture-Agricultural Research Service (USDA-ARS), and The Ohio State University (OSU), this report characterizes the potential for adverse human health and ecological effects associated with the beneficial use of SFS in soil-related applications. By combining the results of current scientific research on SFS and metal and organic behavior in soils with the results of risk screening modeling, this report is intended to provide states with a sound scientific basis with which to evaluate the potential risks to human health and the environment associated with the beneficial use of SFS in soil-related applications.

This chapter presents (1) the purpose, (2) the major features of the report, and (3) a “roadmap” to this report that summarizes the major components of the SFS evaluation.

1.1 Purpose

In 2002, the USDA-ARS implemented the Foundry Sand Initiative under National Program 206 (Manure and Byproduct Utilization) to address agricultural and horticultural uses of SFS. Prior to the inception of this initiative, there was limited information on the use of SFS in manufactured soils, although sands are commonly used as an ingredient in a variety of soil-related applications. The USDA-ARS supports research to address the increasing national need for manufactured soils, particularly for use in disturbed and degraded environments and agricultural applications. A multiyear research project was conducted to characterize inorganic and organic constituents of environmental concern in SFSs and to assess the potential mobility and uptake of these constituents by environmental receptors. Research results were published as peer-reviewed scientific articles, which are available in the public domain (Dungan 2006; Dungan and Dees, 2006, 2007, and 2009; Dungan and Reeves, 2005 and 2007; Dungan et al., 2006 and 2009 and Dayton et al., 2010). In an effort to address the potential risks of using SFS in soil-related applications, the USDA-ARS and EPA formed an expert team of agronomists, soil scientists, and environmental health risk assessors to develop a SFS-specific risk assessment. The main purpose of this work was to determine whether or not SFSs pose unacceptable risks to human health or the environment when used in manufactured soils. The risk management criteria used in this evaluation stipulate that the estimated risks to human or ecological receptors exposed to SFS chemical constituents in manufactured soils should not exceed a target cancer risk and noncancer hazard as defined below:

- For carcinogenic (cancer-causing) constituents, the target cancer risk is defined as an excess lifetime cancer risk above 1 chance in 100,000 (i.e., 10^{-5}).
- For constituents that cause noncancer health effects, the target hazard level is defined as a ratio of the estimated exposure level to a reference level—the hazard quotient (HQ)—of 1.
- For noncancer effects to ecological receptors (e.g., plants, animals, soil invertebrates), the target hazard level is defined as the ratio of the predicted exposure level to a chosen environmental quality criterion or allowable medium concentration.

Thus, the question to be answered by this evaluation may be stated as follows: is the use of silica-based iron, steel, and aluminum SFSs in manufactured soils protective of human and ecological receptors in the United States where this material is used? This evaluation defines the term “protective” in terms of Y excess risk of cancer (i.e., $\leq 10^{-5}$) for human receptors and Z hazard (i.e., ≤ 1) for noncancer endpoints for both human and ecological receptors. The SFS evaluation uses a lines-of-evidence approach to draw conclusions, taking advantage of a significant body of research on SFS and constituent behavior in soils, as well as risk screening modeling.

In pointing out that the SFS evaluation uses a lines-of-evidence approach, it is useful to consider exactly what that means. As detailed in Chapter 2, the constituents of potential concern in SFS include metals, metalloids, and a number of organics, including polycyclic aromatic hydrocarbons (PAHs), phenolics, dibenzodioxins, dibenzofurans, and dioxin-like compounds. With respect to the presence of metals and metalloids (hereafter simply referred to as metals), the assessment considers a number of different issues that EPA has identified in the *Metals Framework for Risk Assessment* (U.S. EPA, 2007b), including:

- Will the addition of SFS to soil result in an increase in the metal concentrations in soil relative to background levels, and how should the results of the risk assessment be interpreted across varied national soils?
- How do metal species found in the SFS matrix behave with respect to bioaccessibility and bioavailability? What soil properties are most important to consider in evaluating the metal behavior and toxicity (e.g., pH is often referred to as the master soil variable for metals)?
- How will the behavior of individual metals in manufactured soil, such as the soil-plant barrier, impact the potential for exposure through the food chain pathway and, ultimately, the potential for adverse human health and ecological effects?
- How do the risk assessment results compare to levels required to maintain nutritional health in plants and animals? Do issues of essentiality suggest that the predicted risks to plants and animals overestimate the potential for adverse effects?
- How do the interactions among metals in the SFS matrix influence the mobility and toxicity of metals? If used as a component of manufactured soils, would a decrease or increase in toxicity be expected?

Each of these questions is important in assessing the potential risks posed by metal constituents in SFS-manufactured soils, because the properties of this material may increase or decrease the risk to human health and the environment. Therefore, the lines-of-evidence approach taken in this risk assessment brings recent study information on SFS and metal constituents—including both qualitative and quantitative information—to address these questions and to ensure that the risk characterization presents a comprehensive view of the potential for adverse effects.

1.2 Major Features of the SFS Evaluation

The problem formulation chapter (**Chapter 3**) and the analysis chapters (**Chapters 4** and **5**) provide a detailed description of the conceptual approach, as well as the models and data used in considering the potential risks associated with SFS constituents in manufactured soil. The following list of features provides a broad sense of the SFS evaluation:

- The point of exposure is assumed to be the point of application. That is, the exposure scenarios focus primarily on the potential risks associated with exposure at the point of SFS application. Thus, the SFS risk assessment is based on conservative assumptions regarding exposure (e.g., the drinking water well is immediately adjacent to the use location).
- The recent research conducted for this evaluation includes an analysis of the constituent concentrations found in SFS, leaching potential, plant uptake, and toxicity to soil invertebrates. Data include both constituent-specific information as well as studies on SFS as a material (e.g., soil invertebrate toxicity). Taken together, this body of data represents the best available characterization of SFS and its constituents.
- The risk assessment draws upon a number of different sources of information in developing conclusions regarding the potential risks to human health and the environment. The information developed and presented in this report includes
  - Qualitative (e.g., descriptions of how the soil-plant barrier renders certain exposure pathways incomplete for certain SFS constituents)
  - Semi-quantitative (e.g., comparisons of SFS constituent concentrations to environmental quality criteria)
  - Quantitative (e.g., quantitative estimates derived using risk assessment screening models to evaluate the inhalation, groundwater ingestion, and plant ingestion pathways).
- A tiered risk assessment approach was used to identify constituents and exposure pathways of concern; the information produced at each step was used to identify the constituents to be included in the following step.
- The EPA model SCREEN3 (U.S. EPA, 1995b) was used in screening-level modeling of the inhalation pathway to develop conservative estimates of exposure concentrations for comparison with EPA inhalation benchmarks.
- EPA’s Industrial Waste Management Evaluation Model (IWEM; U.S. EPA, 2002a, 2002b) was used in screening-level modeling of the groundwater ingestion pathway to develop conservative estimates of groundwater exposure concentrations for use in standard risk equations.
- EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP; U.S. EPA, 2003f, g, h; 1997a) was used in refined probabilistic groundwater modeling of arsenic. Drinking water well exposure concentrations were developed for use in standard risk equations.
- The EPA model (with minor modifications) that is currently used to support EPA’s 2004, 2005, and 2006 biosolids risk assessments under section 503 of the Clean Water Act (U.S. EPA, 2002e) was used for selected constituents, screening-level probabilistic modeling of the direct ingestion of soil and the ingestion of home-grown produce.
- The risk characterization addresses the potential for adverse effects to both human and ecological receptors for exposure scenarios involving direct contact with and use of manufactured soils containing SFS.

- The risk characterization uses recent research (largely conducted by USDA-ARS and OSU) to interpret the screening-level estimates of risk, making full use of a wealth of information describing and sometimes quantifying the behavior of chemical constituents in soil, as well as exhaustive analytical data on constituent concentrations and leach tests on SFS.

1.3 Roadmap to this Report

As shown in **Figure 1-1**, the SFS assessment framework is comprised of five key components: (1) SFS Characterization; (2) Problem Formulation; (3) Analysis; (4) Risk Characterization; and (5) Conclusions. Information gathered during the SFS characterization is used to support the risk assessment, which is performed under the Problem Formulation, Analysis, and Risk Characterization phases shown in Figure 1-1. The Analysis applied a phased approach where Phase I identified SFS constituents and pathways of potential interest, and Phase II applied a probabilistic screening approach to further evaluate those constituents and pathways that did not pass the Phase I screen. As illustrated in Figure 1-1, the information collected during the SFS Characterization (which included scientific research on the SFS constituents) was critically important to the Risk Characterization; in conjunction with the risk modeling results, the information on SFS and its constituents was synthesized to develop conclusions regarding the potential health and ecological risks associated with soil-related SFS use. In summary, the chapter organization is as follows:

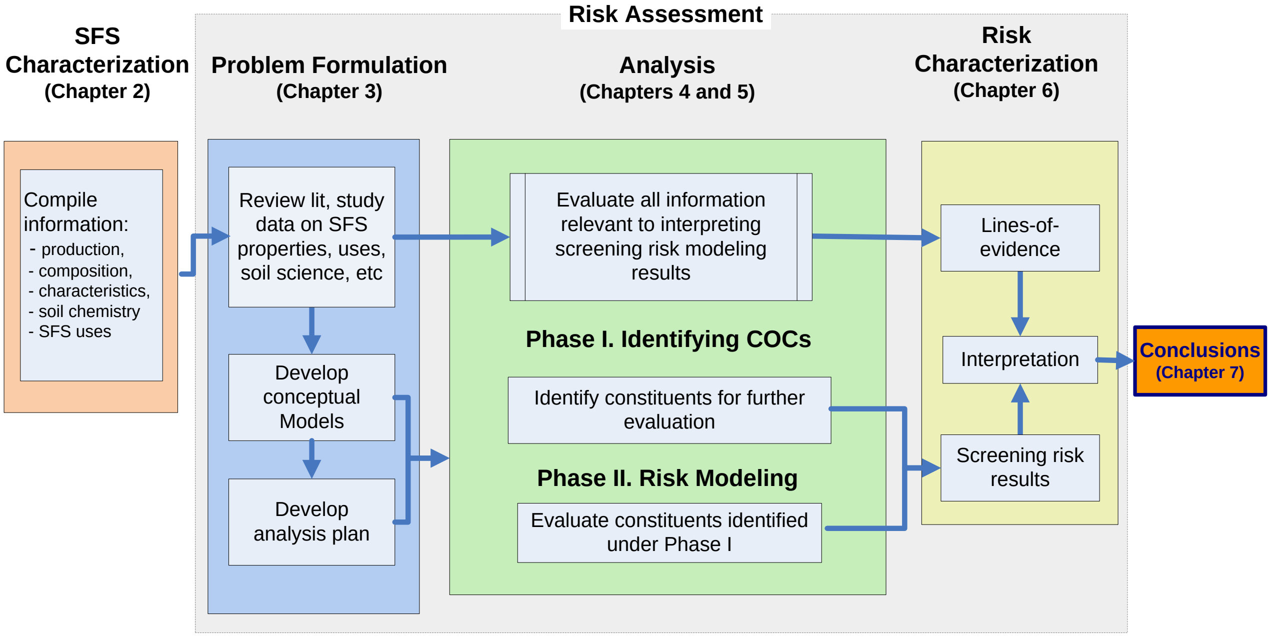

Figure 1-1. Framework for the SFS assessment.

- **Chapter 2—Background and Characteristics of Spent Foundry Sands.** Summarizes information on the sources and types of foundry sands, provides data on the physical and chemical properties of U.S. iron, steel, and aluminum SFSs, and provides data on the uptake of metals by plants and earthworms, and the impact of those metals on soil microorganisms. Chapter 2 also provides additional information, especially on metal constituents, relevant to the potential exposure pathways of interest.

- **Chapter 3—Problem Formulation.** Defines the scope of this risk assessment, presents the conceptual models that illustrate the sources, exposure pathways, and receptors of interest, and summarizes the analysis plan developed to characterize the potential for adverse health and ecological effects associated with constituent releases from SFS in manufactured soils.
- **Chapter 4—Analysis Phase I: Identification of COCs for Modeling.** Describes the rationale for selecting the constituents of concern (COCs) for the groundwater, inhalation, and soil pathway modeling. This chapter presents the comparison of constituent concentrations in SFS with screening criteria for groundwater, air, and soil exposures, respectively. The screening results identified the COCs and exposure pathways for probabilistic risk modeling.
- **Chapter 5—Analysis Phase II: Risk Modeling of COCs.** Describes the probabilistic screening and refined modeling of the groundwater and soil pathways for the home gardener scenario. This chapter presents the methodology and inputs/outputs for each part of the modeling and discusses the results of the model simulations.
- **Chapter 6—Risk Characterization.** Presents the lines-of-evidence interpretation of the potential for adverse health and ecological effects (1) for SFS as a material used in manufactured soils, (2) by constituent category such as PAHs and dioxins, and (3) by constituent for the majority of metals found in SFS. This chapter pulls together the information and risk modeling results from the previous chapters, and incorporates critical research on areas such as the soil-plant barrier that are essential to the interpretation of the risk assessment results. In addition, this chapter discusses key sources of uncertainty in the characterization of risk.
- **Chapter 7—Conclusions and Recommendations.** Distills the findings from the risk characterization into a concise summary to be used in interpreting the results of this risk assessment for the purposes of decision making regarding the beneficial uses of SFS addressed by the assessment.

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2. Background and Characteristics of Spent Foundry Sand

The overall goals for this report are to (1) evaluate all available information on the beneficial use of SFS in the various use scenarios addressed in this assessment, (2) identify likely exposure pathways and receptors associated with those use scenarios, and (3) determine whether the unencapsulated use of SFS in those beneficial scenarios have the potential to cause adverse health or ecological effects. With these goals in mind, this chapter presents information on the production, composition, characteristics, and uses of SFS.

2.1 Foundry Sand Characteristics

Sand is used by the foundry industry to create metalcasting molds and cores. The sand has the ability to absorb and transmit heat because it allows gases generated during casting to pass between the sand grains. The most commonly used sand is silica sand (silicon dioxide, SiO_2) because of its wide availability and relatively low cost. Several other sands are used for specialty casting because of the specific properties related to limited expansion upon heating (e.g., chromite, olivine, zircon, and staurolite). While thermal expansion is an important physical property that must be considered before selecting a sand, other important physical properties are grain shape, grain fineness, permeability, and density. Specifically:

- Sand grain shapes can be classified as round, subangular, angular, and compound. Round sand is superior for green sand systems (see discussion on green sands in **Chapter 2.2.1**, below), while subangular sand with obtuse angles is the most common type of silica sand. Angular sands have grains with edges that form acute angles, and compound sands have grains that are fused together; both angular and compound sands are poor sands for making castings.
- Grain fineness is based on the average sand-grain size. Steel castings typically use very coarse sand, while nonferrous castings (e.g., aluminum, brass, bronze) use finer sand.
- Permeability is a measure of how fast gases will pass through the mold. If the gases do not freely pass through the sand, then the resulting pressure buildup may crack the mold. On the other hand, if the gases pass too quickly, then the molten metal may penetrate the voids, causing a very rough casting.
- Higher sand density is desirable because high-density sands will absorb heat faster and result in fewer surface defects. A smaller coefficient of thermal expansion is also preferred. High-quality silica sand has about a 1.8% thermal expansion from ambient temperature up through casting temperatures of 1,540–1,590°C (2,800–2,900°F). This is an important consideration when trying to hold dimensional tolerances.

2.2 Molding and Core Sands

2.2.1 Green Sands

Green sand is the most widely used in the molding process. The main components of green sand systems are sand, sodium and/or calcium bentonite clay, and carbonaceous additives (e.g., bituminous coal, gilsonite, cellulose). Green sands are named not because of their color, but because the sand mixture contains water and provides "green strength." Green strength is the ability of an incompletely cured material to be handled without distortion. Green sands contain about 85–99% sand and up to 10% clay and 5% carbonaceous material. Sodium and calcium bentonite are hydrous alumina silicates, which provide cohesion and plasticity in the green state (i.e., wet) and when dried. Sodium bentonite (also called western bentonite) can swell to 10–20 times its original volume when thoroughly wetted and has a burnout temperature of about 1,290°C (2,350°F). Calcium bentonite (also called southern bentonite) is a non-swelling clay. Calcium bentonite's burnout temperature of 1,100°C (1,950°F) results in a sand that is less durable than sodium bentonite.

Bituminous coal (called seacoal by the foundry industry) and gilsonite partially combust in the presence of the molten metal, leading to off-gassing of vapors. Release of the organic vapors from within the mold is necessary to prevent the mold from splitting and causing casting defects. Cellulosic additives (such as wood flour, corn flour, cotton hulls, rice hulls, walnut shells, and pecan shells) absorb the moisture, prevent expansion defects, and can improve the flowability of the sands. The individual sand grains are coated with clay and water through the use of a mulling process.

2.2.2 Chemically Bonded Sands

In addition to clay or other inorganic binders, individual sand grains can also be held together using a variety of organic resins. These resins are used to create molds and cores. Cores are used to create a hollow cavity within a metal casting and are exclusively made using resin-coated sand prepared by a number of different processes. Some of the most commonly used resins/processes are the phenolic urethane coldbox[7] and no-bake; furan no-bake and warmbox; novolac; resole; and sodium silicate.[8]

Phenolic Urethane

All phenolic urethanes are three-part systems consisting of a phenolic resin, polyisocyanate, and a tertiary amine catalyst (Gardziella et al., 2000). The phenolic resin is a phenol-formaldehyde polymer and is adjusted to a specific viscosity with a complex mixture of high-boiling aromatic hydrocarbons. The polyisocyanate used is diphenylmethane-4,4-diisocyanate (MDI) and is similarly diluted with solvents. MDI is produced from aniline and formaldehyde. Additives of a proprietary nature are often added to coldbox formulations to increase moisture resistance, bench life, and core box release. The urethane is formed when the isocyanate group reacts with a hydroxyl group in the phenolic resin (all urethanes share a common functional group, i.e., R-NHC=OO-R). Amine catalysts are used in both coldbox and no-bake core and mold making to accelerate the polyurethane reaction. The tertiary amine catalysts—dimethylethylamine and triethylamine—are used in coldbox systems.

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[7] "Coldbox" is a term used to describe any binder process that uses a gas or vaporized catalyst to cure the resin while at ambient temperature.

[8] In addition to these resins, a new class of sand binder was created by General Motors and is known as GMBOND. This protein-based binder is made from high strength collagens with an additive to promote thermal breakdown of the binder coating. The minimum protein content of the binder is 99.5% and it contains trace quantities of iron oxide, methyl paraben, propyl paraben, benzalkonium chloride, and sodium benzoate. Unlike the thermoset polymers of many binder systems, this protein-based binder system forms a biopolymer crystalline structure.

Furan

In the furan (i.e., heterocyclic organic compound, but not related to dibenzofurans) no-bake process, polymerization occurs when the liquid resin is exposed to an acid catalyst at ambient temperature. While the major component of furan resins is furfuryl alcohol, other additives such as phenol, formaldehyde, urea, 2-furancarboxaldehyde (furfural), and 2,5-bis(hydroxymethyl)furan are often used to improve specific resin properties (Gandini and Belgacem, 1997). The acid catalyst is a combination of acid (phosphoric acid–based or sulfonic acid–based), methanol, and water. Optimum binder concentrations vary from approximately 0.8–1.5% of the sand mixture by weight before metalcasting. The furan warmbox process uses the same equipment and procedures as the no-bake process, except that heat is applied (130–180°C) to aid in resin curing.

Novolac

Novolac oligomers are thermoplastic, brittle, and do not cross-polymerize without the help of a cross-linking agent. The oligomers are produced under reflux at 100°C with a molar ratio of formaldehyde to phenol <1 and the addition of an acid catalyst (e.g., sulfuric acid, oxalic acid). Cross-polymerization or curing of the oligomers occurs when they are heated in the presence of hexamethylenetetramine (HMTA), which decomposes to formaldehyde and ammonia. The shell process is used to produce free-flowing, storable sand that is coated with a novolac-HMTA film (1.6–3.8% based on sand weight before metalcasting), which is then cured on hot pattern plates or in heated coreboxes (180–350°C) to form hollow and solid cores (Gardziella et al., 2000). To reduce brittleness, 1–2% iron oxide is often added to the resin.

Resole

Phenolic resoles are prepared by a reaction of excess formaldehyde with phenol and the addition of a base catalyst (e.g., sodium hydroxide, potassium hydroxide) at temperatures up to 100°C (Gardziella et al., 2000). Curing occurs when the phenolic resoles react with an acid at ambient temperature (no-bake process) or heating to 180–250°C (hotbox process), or from a reaction with an aliphatic ester (ester no-bake process).

Sodium Silicate

Sodium silicate ($Na_2O \cdot SiO_2$) is an inorganic system that can be cured using an organic ester or during gassing with carbon dioxide (CO_2) (Owusu, 1982; Gardziella et al., 2000). In the ester-cured system, the ester is hydrolyzed by alkaline sodium silicate. The acid produced during this reaction then reacts with the sodium silicate to form a gel, which bonds the sand grains. Some typical organic esters used are glycerol diacetate, ethylene glycol diacetate, and glycerol triacetate (Winkler and Bol'shakov, 2000).

2.3 Reclamation and Disposal

Many foundries have invested in sand reclamation systems that can recover up to 90% of the sand used in the casting process (Stevenson, 1996; Zanetti and Fiore, 2002). Used molding and core sands can be reclaimed through mechanical and/or thermal treatment. During mechanical reclamation, the sand is crushed to grain size, then dry abrasion is used to separate the binder from the sand grains. Thermal reclamation is a process where all organic binders and carbonaceous additives are burned off after the sand is pre-crushed. This is a more expensive process than mechanical attrition because it requires high-energy inputs to heat the sand to 500–800°C. Reclaimed sand can be reused a number of times in the casting process; however, because heat and mechanical abrasion eventually render the sand unsuitable for continued use in the foundry, the resulting sand must be managed as a waste or beneficially used outside the foundry. Much of the SFS sent to landfills is used as daily cover, but it is not uncommon for foundries to dispose of their SFS in monofills at the foundry.

2.4 Collection and Analysis of U.S. SFSs

An examination of the peer-reviewed literature on metals and organics in SFS revealed that many peer-reviewed reports on this topic have been published over the last two decades. Because there was great interest in using SFS in geotechnical applications, prior to its use in manufactured soils, the majority of the research addressed the leaching potential of various constituents (Riediker et al., 2000; Ji et al., 2001; Kendall, 2003; Lee and Benson, 2006; Deng and Tikalsky, 2008). The most comprehensive data sets on metals and organics in SFS have been generated by the USDA. The USDA data sets contain information on total and leachable metals (Dungan, 2008; Dungan and Dees, 2009; Dayton et al., 2010), PAHs and phenolics (Dungan, 2006), and dioxins (Dungan et al., 2009). A database was also created by The Pennsylvania State University (Penn State), where industry data on different foundry waste materials were compiled (Tikalsky et al., 2004). This database contains information on total and leachable concentrations of various constituents in foundry byproducts, many of which were not suitable for beneficial use in soil-related applications. While the Penn State database was not used in this risk evaluation as a result of inconsistent analytical data among the foundry byproducts, a comparison of the database with the USDA data set revealed that total and leachable concentrations of organic and inorganic constituents in molding sands were very similar. USDA analyzed for organic compounds that are major binder components (i.e., phenolics) or might be generated during the thermal degradation of chemical binders and other organic additives (i.e., PAHs, dioxins, furans), because these constituents present the greatest hazard if at elevated levels in the environment. Evidence of additional organic compounds present in SFS found them at concentrations well below levels of concern. Therefore, additional organic compounds, beyond those analyzed by the USDA, were not considered in this assessment.

2.4.1 Spent Foundry Sand Collection

In June 2005, September 2005, and July 2006, 43 SFSs (36 green and 7 chemically bonded molding sands) were collected from ferrous and nonferrous foundries located in 12 states (Alabama, Georgia, Iowa, Indiana, Michigan, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Virginia, and Wisconsin). A description of the SFSs can be found in **Table 2-1**. The June 2005 samples were collected as described by Dungan (2006), while the remaining sets were collected by foundry personnel after receiving training on sample collection. Briefly, a clean section of polyvinyl chloride (PVC) pipe was used as a sampling device to collect four samples from each SFS pile. The samples were transferred into 500-mL glass jars with Teflon-lined polypropylene closures and immediately shipped to the laboratory in Styrofoam coolers with ice packs. Upon receipt, the samples were stored at 4^{o}C for no longer than 2 weeks until processed. All SFSs were passed through a 0.5-mm sieve to remove any core butts before being analyzed.

Table 2-1. Description of the U.S. Spent Foundry Sands

Sand	State	Sampling Dates			Metal Poured	Molding Sand	Core Binder System and Process
		6/05	9/05	7/06			
1	PA	x	x	x	Iron	Green sand	PU[b] coldbox, PU no-bake, shell, core oil
2	PA	x			Iron	Green sand[a]	Shell
3	PA	x	x	x	Iron	Green sand	Shell, furan warmbox
4	PA	x	x	x	Aluminum	Green sand	Shell
5	PA	x	x		Iron	Green sand	PU no-bake, shell, sodium silicate
6	PA	x	x	x	Steel	PU no-bake[a]	PU no-bake
7	PA	x	x	x	Iron	Green sand	PU no-bake
8	OH	x	x	x	Iron	Green sand	PU coldbox, PU hotbox
9	OH	x	x	x	Iron	Green sand	PU coldbox, PU hotbox
10	OH	x	x	x	Iron	Green sand	PU coldbox, PU hotbox
11	OH	x	x	x	Iron	Green sand	PU coldbox, PU no-bake, shell
12	IN	x	x	x	Iron	Shell[c]	Shell
13	OH	x	x	x	Iron	Green sand	PU coldbox, PU no-bake, shell
14	OH	x	x	x	Aluminum	Green sand	PU no-bake, shell, core oil
15	IN	x			Iron	Green sand	PU coldbox, shell
16	OH	x	x	x	Iron	Green sand	PU coldbox, PU hotbox
17	OH	x	x	x	Iron	Green sand	PU coldbox, PU hotbox
18	IN	x	x	x	Iron	Green sand	PU coldbox, PU hotbox, shell
19	WI	x	x	x	Iron	Green sand	PU coldbox
20	OH	x	x	x	Aluminum	Green sand	Shell
21	IN	x	x	x	Iron	PU no-bake	PU coldbox, PU no-bake, furan warmbox
22	MI	x	x	x	Iron	Green sand	PU no-bake, shell
23	MI	x	x	x	Iron	Green sand	PU coldbox, shell
24	WI	x	x	x	Iron	Green sand	Shell
25	WI	x	x	x	Iron	Green sand	PU coldbox
26	MI	x	x	x	Iron	Green sand	None
27	OH	x	x	x	Iron	Green sand	PU no-bake, shell
28	TN	x	x	x	Iron	Green sand	None
29	WI	x	x	x	Steel	PU no-bake	PU no-bake
30	WI	x	x	x	Iron	Green sand	PU coldbox, shell
31	TN	x	x	x	Iron	Green sand	Shell, resin/CO_2
32	TN	x			Iron	Green sand	PU coldbox
33	AL	x	x	x	No lead brass	PU no-bake	PU no-bake
34	AL	x	x	x	No lead brass	Green sand	PU no-bake
35	VA	x	x		Iron	Green sand	PU coldbox
36	GA	x	x	x	Iron	Green sand	PU coldbox, shell
37	SC	x	x	x	Iron	Green sand	PU coldbox, shell
38	IA	x		x	Steel	Phenolic ester-cured	PU coldbox, shell, resin/CO_2
39	IA	x		x	Steel	Green sand	PU coldbox, shell, resin/CO_2
40	NC	x	x	x	Iron	Green sand	PU coldbox, shell
41	IN	x	x		Steel	PU no-bake	PU no-bake
42	IN	x	x	x	Iron	Green sand	PU coldbox
43	WI	x	x	x	Steel	Green sand	PU no-bake, shell, core oil, resin/CO_2

[a] Olivine sand utilized [b] PU = phenolic urethane [c] Shell process associated with use of novolac resin

2.4.2 PAHs and Phenolics[9]

An accelerated solvent extractor (ASE 200, Dionex, Sunnyvale, CA) was used to extract the PAHs and phenolics for analysis by gas chromatography-mass spectrometry (GC-MS). Twenty grams of SFS was placed into the center of a 33-mL stainless steel extraction cell, which was then packed at each end with clean Ottawa sand (20–30 mesh, U.S. Silica Corp., Ottawa, IL) to fill the void. If the SFS was moist, anhydrous Na_2SO_4 was mixed with the sand prior to the addition to the cells. The conditions of the ASE were as follows: solvent, dichloromethane/ acetone (1:1); static extraction for 5 min at a pressure of 14 MPa (2,000 psi) and an oven temperature of 100°C; flush volume, 60% of the cell volume; N_2 purge, 1 MPa (150 psi) for 60 s. All extracts were collected in 40-mL vials. Immediately after the extraction, the extracts were evaporated to near dryness under N_2 and then reconstituted with 2 mL of dichloromethane. The method detection limit (MDL) for this data set was calculated by multiplying the standard deviation of replicate standards ($n = 6$) by the Student's t-value at the 99% confidence interval. Calculating the MDL at the 99% confidence interval allows for the possibility that 1% of the samples analyzed, which have a true concentration at the MDL, will be false positives.

2.4.3 Dioxins and Dioxin-like Compounds

The SFSs were processed and analyzed for polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and coplanar polychlorinated biphenyls (PCBs) by EPA Method 1613 (tetra- through octa-chlorinated dioxins and furans by isotope dilution HRGC/HRMS, 1994B) modified to include the coplanar PCBs (IUPAC nos. 77, 126, and 169). Toxic equivalency values (TEQs) were calculated by summing the products of each congener concentration and its World Health Organization (WHO) 2005 toxic equivalency factor (TEF) (Van den Berg et al., 2006).

2.4.4 Trace Elements

USDA-ARS Data Set

The SFSs were digested according to EPA method 3050B. The digests were filtered through Whatman no. 40 paper layered with Whatman 2V fluted filters (Florham Park, NJ). The filtrate was diluted to 100 mL with 0.1 *M* HCl and analyzed by inductively coupled plasma-atomic emission spectrometry (ICP-AES). Blanks and standard reference material 2709 (San Joaquin Soil, National Institute of Standards and Technology [NIST], Gaithersburg, MD) were run regularly to ensure quality control. The limit of quantitation (LOQ) was calculated as 10 or 30 times the standard deviation of digestion blank values ($n = 20$) and was expressed as mass of element per sample dry weight.

Ohio State University Data Set

Elemental concentrations were determined by EPA method 3051A (U.S. EPA, 2007d); a microwave-assisted aqua regia digestion followed by ICP-AES analysis and inductively coupled plasma-mass spectrometry (ICP-MS) for elements below detection by ICP-AES. ICP-AES and ICP-MS analyses for total elemental analysis were carried out according to EPA methods 6010C and 6020A, respectively. Quality control operations included analysis of laboratory control

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[9] See **Section 2.5.3** for a discussion of the selection process for organics.

samples (CRM 059-050; RTC Corporation, Laramie, WY) with each microwave tray, pre-digestion spikes, initial calibration verification, initial calibration blank, continuing calibration verification for every 10 samples, continuing calibration blank for every 10 samples, and low LOQ verification for every 20 samples. All checks were within the quality control limits set in EPA, ILM04.0b (U.S. EPA, 1999a).

2.4.5 Leach Tests

The toxicity characteristic leaching procedure (TCLP) and the synthetic precipitation leaching procedure (SPLP) were conducted according to EPA methods 1311 and 1312, respectively. The water leach test was conducted according to American Society for Testing and Materials (ASTM International) method D 3897 (ASTM International, 2004). All leaching procedures were slightly modified as described by Dungan and Dees (2009). The extracts were analyzed by ICP-AES. Also, Dayton et al. (2010) estimated pore water elemental content on the SFS by equilibrating SFS in a 1:1 SFS:deionized water saturated paste for 24 hours. Extracts were analyzed by ICP-AES. The LOQ was calculated as 10 times the standard deviation of matrix blanks ($n = 10$) and was expressed as mass of element per volume of leaching solution.

2.5 Constituents and Properties of Spent Foundry Sand

2.5.1 Properties Important to Soil Quality and Function

Manufactured soils, such as horticultural potting soils or those made for landscaping, generally contain some low-grade native soil. Soils made for such purposes are created by blending organic and mineral components, such as SFS. For SFSs to be considered for beneficial use as a soil amendment or a component of a soil blend, they must have soil-like qualities, make a contribution to soil quality/fertility, or provide a functional benefit (e.g., acid neutralization, contaminant sorption/binding). SFSs tend to have low fertility, but they often have soil-like qualities that make them attractive as components in a soil blend. Soil quality has been defined as “the capacity of a soil to function, within ecosystem and land-use boundaries, to sustain biological productivity, maintain environmental quality and promote plant and animal health” (Doran and Parkin, 1996). A manufactured soil suitable for plant growth should have desirable chemical (e.g., pH, salinity) and physical (e.g., drainage, texture, water holding capacity) properties. Components used in a manufactured soil are chosen to provide suitable levels of these properties. An added advantage of manufactured soils is that component ratios can be adjusted so a soil blend can be “tailored” to specific uses. For example, in horticultural applications, soils used for market pack containers need to be light and well drained, while soils used for landscaping or container mixes for trees and shrubs need to be heavier and have a good water-holding capacity. To be beneficial, a manufactured soil also must not cause toxicity to plants and biota.

Properties important to soil quality and function were measured in 43 ferrous and nonferrous SFSs to characterize the sands as potential components in manufactured soil blends.

Soil Organic Carbon

Soil organic carbon (OC) typically comprises 0.5–3% by weight of mineral soils (Brady and Weil, 2007), but its importance to soil chemistry and function is greater than these numbers suggest. Soil OC contributes to soil quality in many ways. It increases water-holding capacity and infiltration. It also improves soil structure by promoting soil aggregate formation and serves as a major reservoir for plant nutrients and carbon and energy for soil microorganisms (Brady and Weil, 2007). Soil organic matter has a large surface area (800–900 m^2 g^{-1}) and is rich in reactive functional groups (e.g., carboxyl or phenolic) (McBride, 1994; Bohn et. al., 2001; Sparks, 2003). The ionization of these groups, as mediated by pH, imparts a high pH-dependent cation exchange capacity (CEC, 150–300 $cmol_c$ kg^{-1}) to soil organic matter (Bohn et. al., 2001; Adriano, 2001; Sparks, 2003). Binding of nutrient cations to the exchange sites reduces leaching and provides nutrient storage for plant nutrition. Nutrients in equilibrium with the soil solution are readily resupplied to the solution as plants feed. A more stable form of metal complexation with soil organic matter is through chelation of cationic micronutrients with soil organic matter.

The OC content of the 43 SFSs, measured using dry combustion after acid pretreatment to remove inorganic carbonates, ranged from 0.29–2.99%, with a mean of 1.71%. The SFS OC includes OC additions made to the molding sands (i.e., seacoal, polymers) and is within the typical range for native soils.

Soil Texture

Soil texture is determined by the proportionate content of different sized soil particles. Particle size distribution determines the soil textural class. Knowing a soil’s particle size distribution or textural class provides insight into important aspects of the soils behavior (e.g., water retention, infiltration, bulk density).

Many horticultural manufactured soil blends are composed of high levels of coarse materials (e.g., bark, rice hulls, perlite). These soil blend components are light weight and freely drain, but finer fractions also are needed to increase the water holding capacity and provide plant nutrient storage. Clay-size particles or clay minerals are a highly reactive component of soil characterized by having a particle size <2.0 µm and a large surface area.[10]

A small but important component of many foundry sands is their clay content. Although we refer to SFS as sand, the addition of clay, seacoal, and other carbonaceous additives contribute finer particles that can affect the soil textural class and properties of SFS. The particle size distribution for the 43 sands was determined using the hydrometer method (Gee and Bauder, 1986) and is summarized in **Table 2-2** (a more complete breakdown is provided in **Appendix B, Table B-25**). Sand (0.05–2 mm) was the dominant size fraction, ranging from 76.6–100% with a mean of 91%, while silt size particles (2–50 µm) ranged from 0–16.9%, with a mean of 3.43%, and clay size particles ranged from 0–11.3%, with a mean of 5.54%. Using the USDA Soil Texture Calculator (USDA, 1993), the SFS textural class was calculated based on the particle size distribution. The SFS bulk density was calculated using the Saxton equation (Saxton et al., 1986). Soil texture, in general, ranges from sand (coarse) to clay (fine). Not surprisingly, the

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[10] The reactions between clay minerals are primarily attributed to their cationic exchange capacity (CEC) or ligand exchange (specific adsorption) reactions that occur on non-crystalline or amorphous metal oxide clays, typically of iron or aluminum. The permanent, negatively charged portion of the soil CEC is associated with isomorphic-substituted 2:1 clay minerals, such as smectite and montmorillonite. These clay minerals have a large surface area and high CEC. Montmorillonite, for example, has a surface area of 600–800 m^2 g^{-1} and a CEC of 80–150 $cmol_c$ kg^{-1}. The pH-dependent CEC sites are associated primarily with non-crystalline metal oxide clays. These amorphous metal oxides also have a large surface area. For example, iron and aluminum oxides have a specific surface area of 70–250 and 100–220 m^2 g^{-1}, respectively (Bohn et. al., 2001; Adriano, 2001; Sparks, 2003).

texture of the SFSs ranged from sand to sandy loam and the bulk density ranged from 1.57–1.66 g cm^{-3}, with a mean of 1.64 g cm^{-3}.

Table 2-2. Particle Size Distribution, USDA Textural Class, and Bulk Density for 43 SFSs

	Sand (0.05–2mm) %	Silt (2–50 μm) %	Clay (<2 μm) %	Bulk Density g cm^{-3}
Minimum	76.6	0	0	1.57
Maximum	100	16.9	11.3	1.66
Mean	91	3.43	5.54	1.64

The hydrous metal oxides of aluminum and iron were measured using an acid ammonium oxalate extraction (McKeague and Day, 1996). The aluminum oxide content ranged from 0.072–2.43 g Al kg^{-1}, with a median of 0.386 g Al kg^{-1}, while the iron oxide content ranged from 0.213–32.1 g Fe kg^{-1}, with a median of 1.39 g kg^{-1}. These values are within the typical range for natural soils (Brady and Weil, 2007). The clay/silt component of SFS suggests that they could increase the water-holding capacity of coarse horticultural soil blends, but is not so high as to inhibit drainage. The higher bulk density (see Table 2-2) compared to typical mineral soils (1.25 g cm^{-3}, Brady and Weil, 2007) suggests that SFS alone may be heavy, which could inhibit root penetration. Due to relatively high concentrations of bentonite clays in foundry sands, the use of SFS alone as a potting medium is likely to inhibit root penetration, as they exhibit high rupture strength under dry conditions (de Koff et al., 2008). However, the addition of SFS to potting or landscape media may be beneficial where shrubs or trees are planted and a heavier mix is advantageous.

pH

Soil pH is often called the “master variable.” It has the potential to modify metal/nutrient solubility/availability in several ways. It controls dissolution/precipitation and therefore influences the speciation of minerals. It regulates the ionization of pH-dependent cation exchange sites on organic matter and metal oxide clay minerals. The ionization of pH-dependent functional groups on soil organic matter also affects stable organic complex formation (McBride, 1994; Adriano, 2001; Sparks, 2003).

The pH of the 43 SFSs ranged from 6.67–10.2, with a mean of 8.76. In some instances, the pH of the SFSs was higher than a typical productive soil. Certainly, the pH will moderate upon blending SFS with other components. There would only be a concern if the pH of the final blended soil remained high, as high pH can reduce plant nutrient availability. In addition, the potential for the formation of unstable aluminum species due to high pH is apparent in the pore water soluble aluminum (**Appendix B, Table B-26**), which ranged from 0.1–1,847 mg L^{-1}, with a median of 1.79. High pH can also induce plant deficiencies of metal cation micronutrients, including iron, manganese, copper, and zinc. Iron chlorosis is the visual symptom of iron plant deficiency induced at soil pH >8.5. Blending SFS with organic materials (e.g., compost, biosolids, manure) and/or soil will buffer the soil pH. SFS will likely be combined with organics, soil, and other materials to make topsoil. The pH buffer capacity of the organic and/or soil materials is much greater than SFS. Therefore, the final pH of the manufactured soil will be closer to the pH of the organic and/or soil materials than the original SFS pH. That is, the final pH of the manufactured soil will be more relevant than the original pH of the SFS.

2.5.2 Metals and Metalloids

The sand and other materials used to create metalcasting molds contain natural levels of metals and metalloids (which will collectively be called metals), but metals may also be transferred to the molding sands during the casting process (Dungan et al., 2006). Abundant industry data are available characterizing the leaching of metals in SFS and other foundry wastes. That is, much of the data are not total numbers, but were derived using leaching procedures, such as the TCLP (U.S. EPA SW-846, method 1311, U.S. EPA, 2007a). TCLP concentrations are used to assess risk of metals in landfill leachates, but have limited relevance to risk assessment for surface soil. The following metals (i.e., aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, molybdenum, nickel, selenium, silver, thallium, vanadium, zinc) were specifically targeted for testing in the 43 U.S. SFSs because they are potential contaminants of ground and surface waters and are a toxicity threat to plants, animals, and humans if present at elevated concentrations.

In a study conducted by Dungan and Dees (2009), a totals analysis was conducted for 19 metals in the 43 SFSs listed in Table 2-1. The total metal concentrations in the SFSs, as determined by EPA method 3050B (SW-846), are summarized in **Table 2-3**. Of the 19 metals analyzed for total concentrations, four (antimony, boron, cadmium, and silver) were not detected in any of the SFSs above the LOQ. The LOQ for antimony, boron, cadmium, and silver were 4.5, 19.2, 5.9, and 17.6 mg kg^{-1}, respectively. The remaining metals (aluminum, arsenic, barium, beryllium, chromium, cobalt, copper, iron, lead, magnesium, manganese, molybdenum, nickel, vanadium, and zinc) were detected above the LOQ in some, but not all, of the SFSs.

In the June 2005 set of SFS samples, sand #2 (green sand from an iron foundry) contained the highest total concentrations of beryllium, cobalt, iron, magnesium, manganese, and nickel at 3.1; 95; 44,320; 51,574; 671; and 2,328 mg kg^{-1}, respectively. For the remainder of the sands, beryllium, cobalt, magnesium, and manganese were generally below the LOQ of 1.2, 0.84, 720, and 45 mg kg^{-1}, respectively. Sand #6 contained the second-highest concentration of nickel at 1,022 mg kg^{-1}. It is likely that the nickel in sands #2 and #6 came from the olivine sand that these foundries use, which typically contains about 2,000 mg Ni kg^{-1} (Dungan and Dees, 2009). The mineral olivine is a magnesium iron silicate and contains naturally elevated concentrations of nickel, cobalt, and chromium. Although silica sand is the most abundantly used sand, olivine sands are used by some foundries because they have a lower thermal expansion coefficient, and therefore hold tighter dimensional tolerances. Olivine sands also produce a better cast surface than silica sands.[11] Sand #39 (green sand from a steel foundry) contained nickel at 107 mg kg^{-1}, which was elevated due to the metal alloy, not because they use olivine sands.

Sands #2 and #6 also contained elevated concentrations of chromium at 57 and 149 mg kg^{-1}, respectively. In sand #22 (green sand from an iron foundry), the molybdenum concentration was 9.6 mg kg^{-1}. In all of the other SFSs, chromium was generally well below 50 mg kg^{-1} and molybdenum was less than the LOQ of 4.4 mg kg^{-1}.

Arsenic was detected in all 43 SFSs at concentrations above the LOQ of 0.03 mg kg^{-1}, but no higher than 7.79 mg kg^{-1}. The arsenic results (and chromium results discussed above) are similar to those obtained by Lee and Benson (2006), who analyzed 12 green sands from gray-iron foundries and found respective ranges of 0.002–2.9 and 1.5–66.4 mg kg^{-1}.

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[11] Characterization of sands #2 and #6 are included for completeness; however, they were not evaluated as part of the risk assessment because they contain olivine sand.

Table 2-3. Total Metal Concentrations in the Spent Foundry Sands as Determined by EPA Method 3050B

	Collected June 2005[a], 43 samples ($mg\ kg^{-1}$)				Collected September 2005, 38 samples ($mg\ kg^{-1}$)				Collected July 2006, 37 samples ($mg\ kg^{-1}$)			
Metal	**Min**	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
Ag[c]	<17.6		8.8	0	<17.6		8.8	0	<17.6		8.8	0
Al	<311	10,048	1,853	37	<311	6,940	1,771	33	<311	6,189	1,656	33
As	0.04	4.8	1.0	43	0.13	5.1	1.7	37	0.07	4.9	1.0	37
B[c]	<19.2		9.6	0	<19.2		9.6	0	<19.2		9.6	0
Ba	<8.7	151	23.3	30	<8.7	72.5	19.2	28	<8.7	149	25.3	27
Be	<1.2	3.1	0.8	5	<1.2	3.5	0.72	3	2.47	2.5	0.65	1
Cd[c]	<5.9		3.0	0	<5.9		3.0	0	<5.9		3.0	0
Co	<0.84	95.3	3.7	7	<0.84	9.1	0.77	5	<0.84	9.1	0.88	4
Cr	<1.0	149	11.6	40	<1.0	196	12.	37	<1.0	132	8.8	33
Cu	<23.1	3,318	97.1	9	<23.1	14,360	772	6	<23.1	4,668	148	8
Fe	<352	44,320	5976	42	727	60,020	6,262	38	<352	45,120	4,867	36
Mg	<720	51,574	2,804	11	<720	26,994	1,313	13	<720	16,566	1,285	4
Mn	<45.0	671	96.0	18	<45	920	91.8	16	<45	845	75.9	15
Mo	<4.4	9.6	2.4	2	<4.4	19.8	2.9	3	<4.4	54.6	3.6	1
Ni	<1.2	2,328	85.7	40	<1.2	139	10.9	34	<1.2	189	12.2	31
Pb	<7.7	25.7	5.1	4	<7.7	28.9	5.8	5	<7.7	212	13.6	10
Sb[c]	<4.5		2.3	0	<4.5		2.3	0	<4.5		2.3	0
V	<7.4	9.1	3.8	1	<7.4	19.3	4.1	1	<7.4	9.7	3.9	1
Zn	<33.4	1,640	60.1	5	<33.4	1,732	91.1	4	<33.4	2,829	102	3

< means less than the LOQ.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the LOQ.

[c] All concentrations recorded below the LOQ.

The highest concentrations of copper and zinc at 3,318 and 1,640 mg kg^{-1}, respectively, were found in sand #34, which is a green sand from a non-leaded brass foundry. This is of little surprise, as brass is an alloy of copper and zinc. The lead concentration in sand #34 was only 19 mg kg^{-1}, which is relatively low due to the fact that it was a non-leaded brass foundry. In contrast, sand #33 is a chemically bonded molding sand from the same brass foundry, but it contained considerably less copper and zinc at 70 and 44 mg kg^{-1}, respectively, and lead was <7.7 mg kg^{-1}.

Table 2-3 also shows total element data from samples collected in September 2005 and July 2006 from a subset of the same 43 foundries. Overall, the data show that there is little change in the element concentrations in sands collected from specific foundries over time. Except for sand #6, only the non-leaded brass foundry sands showed a large temporal variation. The nickel concentration in sand #6 decreased from 1,022 to 111 mg kg^{-1} by the third sampling event, while copper in sand #34 increased to 14,200 mg kg^{-1} by the second sampling event, but was lower at 4,670 mg kg^{-1} by the third sampling event. In sand #33 (from the same foundry as sand #34), the copper increased to 14,360 mg kg^{-1} by the second sampling event, but was down to 38.5 mg kg^{-1} by the third sampling event. Although sands #33 and #34 are from a non-leaded brass foundry, lead in sand #34 increased from 19 to 212 mg kg^{-1} by the third sampling event.

Detection limits for some SFS constituents in the USDA dataset are higher than those required for risk assessment (e.g., the detection limit for antimony (4.5 mg kg^{-1}) is higher than the human screening level (3.1 mg kg^{-1}), and the detection limit for cadmium (5.9 mg kg^{-1}) is higher than the ecological screening level (0.36 mg kg^{-1})). For this reason, and for comparative purposes, Dayton et al. (2010) analyzed the 43 SFSs from the June 2005 sampling event using an analytical method able to reach lower detection limits (i.e., EPA method 3051A), and the data are presented in **Table 2-4**. Because of the lower detection limits, total elemental data generated Dayton et al. (2010) were used for analysis in the risk assessment.

The existing data on non-leaded brass sands and olivine sands demonstrated levels of copper, lead, nickel, and zinc that were both potentially phytotoxic and much higher than the other 39 SFSs, but insufficient samples existed to characterize constituent concentration variability in non-leaded brass and olivine sands. Therefore, while descriptions of non-leaded brass sands and olivine sands (i.e., sands #2, #6, #33, and #34) are retained for completeness, they are not evaluated in the risk assessment.

Table 2-4. Metal Concentrations in 39 of 43 Spent Foundry Sands (June 2005 Samples) as Determined by EPA Method 3051A[a]

Metal	Units	Minimum	Maximum	Mean	Median	95%-ile
Al	g kg^{-1}	0.19	11.7	5.14	5.56	11.2
As	mg kg^{-1}	0.13	7.79	1.70	1.05	6.44
B	mg kg^{-1}	<20.0	59.4	11.5	10.0	20.2
Ba	mg kg^{-1}	<10.0	141	8.81	5.00	17.7
Be	mg kg^{-1}	<0.1	0.60	0.17	0.15	0.38
Ca	g kg^{-1}	0.09	44.1	1.89	1.89	3.23
Cd	mg kg^{-1}	<0.04	0.36	0.07	0.051	0.20
Co	mg kg^{-1}	<0.5	6.62	1.26	0.88	5.99

Metal	Units	Minimum	Maximum	Mean	Median	95%-ile
Cr	mg kg^{-1}	<0.5	115	17.6	4.93	109
Cu	mg kg^{-1}	<0.5	137	21.2	6.22	107
Fe	g kg^{-1}	1.28	64.4	9.20	4.26	57.1
K	mg kg^{-1}	<50.0	1,780	388	328	1300
Mg	g kg^{-1}	0.05	3.20	1.26	1.28	3.02
Mn	mg kg^{-1}	5.56	707	112	54.5	670
Mo	mg kg^{-1}	<1.0	22.9	2.98	0.50	21.8
Na	g kg^{-1}	<0.02	1.93	0.93	1.02	1.85
Ni	mg kg^{-1}	1.11	117	15.2	3.46	102
P	mg kg^{-1}	5.41	96.6	51.2	50.9	85.9
Pb	mg kg^{-1}	<1.0	22.9	4.38	3.74	15.3
S	g kg^{-1}	<0.05	2.04	0.62	0.59	1.64
Sb	mg kg^{-1}	<0.04	1.71	0.30	0.17	1.23
Se	mg kg^{-1}	<0.4	0.44	0.21	0.20	0.20
Tl	mg kg^{-1}	<0.04	0.096	0.04	0.04	0.089
V	mg kg^{-1}	<1.0	11.3	3.44	2.88	9.90
Zn	mg kg^{-1}	<10.0	245	20.0	5.00	72.1

Source: Dayton et al. (2010)

[a] Brass green sands and olivine sands (i.e., sands #2, #6, #33, and #34 from Table 2-1) were omitted from calculations; calculations based on setting samples <LOQ at one half that value.

2.5.3 Organics

During the casting process, the molten metal causes thermal decomposition of the carbonaceous additives and resin binders, which results in the formation of potentially hazardous organics which are emitted to the atmosphere and condense in the molding sand. Because of the conditions within a mold during casting, a number of specific organic compounds and classes were targeted for quantitative analysis within the SFSs. PAHs were of particular interest in the SFSs because they are known to form during the incomplete combustion of organic substances. The majority of green sands contain bituminous coal, which is known to produce PAHs during incomplete combustion processes (Mastral et al., 2000). Gilsonite, another molding sand additive, was also tentatively identified as a source of PAHs (Dungan and Reeves, 2007). Phenolic compounds are of interest in SFSs because phenol is used as a major component in many resin binders. The thermal decomposition of phenol-based binders results in the generation of various phenolics (Lytle et al., 1998a,b, Dungan and Reeves, 2005), some of which are on EPA’s priority list, such as 2-methylphenol (*o*-cresol), 3-methylphenol (*m*-cresol), 4-methylphenol (*p*-cresol), and 2,4-dimethylphenol. Dibenzodioxins, dibenzofurans, and dioxin-like PCBs were also quantified in a small subset of the SFSs. Although dioxins and furans are generated during combustion processes, they would not be expected to be present at elevated concentrations in the SFSs because the SFSs contain low levels of chlorine. PCBs are not naturally found in the environment, so the presence of PCBs was unlikely in SFSs unless the sands were accidentally contaminated within the foundry or the source sands were contaminated before or after mining. Benzene, toluene, ethylbenzene, and xylenes (BTEX) were also tentatively identified during the pyrolysis (heating in the absence of oxygen) of green sands and other mold components (Dungan and Reeves, 2005; 2007). While every effort was made to target the widest range of organic constituents that are of concern from an environmental and human health standpoint, it is possible that additional organics were present in the SFSs and not addressed in this risk evaluation. However, evidence of additional organics found them at concentrations well below levels of concern.

In early studies conducted by Gwin et al. (1976), Scott et al. (1976, 1977), and Palmer et al. (1985), some of the most abundant organics emitted from green sand molds were BTEX, phenolics, and PAHs such as acenaphthalene, benzo[a]pyrene, naphthalene, phenanthrene, and pyrene. These organic compounds are a potential threat to the environment and human health (Alberg et al., 2002; Boström et al., 2002; Rana and Verma, 2005; Baird et al., 2007). In green sand molds, volatile organics are generated during the thermal decomposition of carbonaceous additives such as coal, gilsonite, lignite, and cellulose (Dungan and Reeves, 2007; Wang et al., 2007). During the pyrolysis of a green sand at temperatures up to 1,000°C, Dungan and Reeves (2007) tentatively identified substituted benzenes (e.g., BTEX), phenolics, and PAHs (**Appendix B, Figure B-1 and Table B-27**). When novolac, phenolic urethane, and furan resins were pyrolyzed at temperatures up to 1,000°C, similar thermal decomposition products were identified (Lytle et al., 1998a,b; Hetper and Sobera, 1999; Sobera and Hetper, 2003; Dungan and Reeves, 2005).

In a study conducted by Dungan (2006), all samples from the 43 foundries listed in Table 2-3 were analyzed for 15 PAHs and 17 phenolics that are identified as priority pollutants by EPA. Summary concentration information of the PAHs and phenolics in the SFSs are shown in **Tables 2-5 and 2-6**, respectively. Although no published reports are available on BTEX compounds in SFSs, a preliminary scan of the SFSs using headspace solid-phase microextraction (HS-SPME) was conducted. The benzene; toluene; ethylbenzene; *o*- and *m*-xylene; and *p*-xylene concentrations ranged from below the MDL to maximum values of 50.9, 79.2, 32.9, 72.0, and 41.9 μg kg^{-1}, respectively, for the June 2005 samples. In the September 2005 samples, the maximum concentrations were 1,670; 164; 14.5; 16.4; and 16.8 μg kg^{-1}, respectively (R.S. Dungan, unpublished data).

The majority of the PAHs that were present at concentrations above the MDLs were 2-ring and 3-ring PAHs (i.e., acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene). For most of the SFSs, naphthalene was at the highest concentrations, followed by phenanthrene. Three SFSs in particular (sands #6, #33, and #41) had the highest concentrations of naphthalene, which ranged from 28–48 mg kg^{-1}. These sands were from foundries that used both phenolic urethane molding and core sands (i.e., not green sands). The 4-ring PAHs, benz[a]anthracene and chrysene, were at concentrations slightly above the MDL only in SFSs #9, #34, and #40 (and fluoranthene in SFSs 8 and 27). The respective MDLs for benz[a]anthracene, chrysene, and fluoranthene were 0.10, 0.08, and 0.06 mg kg^{-1}. The following 5-ring and 6-ring PAHs were all below the MDLs in every SFS:

- Benzo[b]fluoranthene
- Benzo[k]fluoranthene
- Benzo[g,h,i]perylene
- Benzo[a]pyrene
- Dibenz[a,h]anthracene
- Indeno[1,2,3-cd]pyrene.

Table 2-5. Concentrations of the PAHs in Spent Foundry Sands

Compound	Collected June 2005, 43 Samples[a] (mg kg^{-1})				Collected September 2005, 38 Samples (mg kg^{-1})				Collected July 2006, 37 Samples (mg kg^{-1})			
	Min	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
Acenaphthene	<0.04	11.7	0.39	12	<0.04	0.18	0.04	10	<0.04	0.40	0.05	8
Acenaphthylene	<0.03	0.29	0.06	20	<0.03	0.32	0.06	13	<0.03	0.33	0.05	13
Anthracene	<0.03	0.95	0.32	34	<0.03	0.99	0.41	34	<0.03	0.69	0.19	31
Benz[a]anthracene	<0.10	0.31	0.06	3	<0.10	0.20	0.06	3	<0.10	0.15	0.06	2
Benzo[b]fluoranthene[c]	<0.12		0.06	0	<0.12		0.06	0	<0.12		0.06	0
Benzo[k]fluoranthene[c]	<0.13		0.07	0	<0.13		0.07	0	<0.13		0.07	0
Benzo[g,h,i]perylene[c]	<0.14		0.07	0	<0.14		0.07	0	<0.14		0.07	0
Benzo[a]pyrene[c]	<0.20		0.10	0	<0.20		0.10	0	<0.20		0.10	0
Chrysene	<0.08	0.30	0.05	3	<0.08	0.11	0.04	1	<0.08		0.04	0
Dibenz[a,h]anthracene[c]	<0.16		0.08	0	<0.16	0.17	0.08	1	<0.16		0.08	0
Fluoranthene	<0.06	0.50	0.05	2	<0.06	1.03	0.07	5	<0.06	0.33	0.05	6
Fluorene	<0.04	2.58	0.31	39	<0.04	1.19	0.34	32	<0.04	1.05	0.23	30
Indeno[1,2,3-cd]pyrene[c]	<0.14		0.07	0	<0.14		0.07	0	<0.14		0.07	0
Naphthalene	<0.03	48.1	3.67	40	<0.03	14.6	1.46	35	<0.03	42.2	2.01	34
Phenanthrene	<0.03	2.2	0.62	41	<0.03	1.91	0.73	37	<0.03	1.86	0.49	35
Pyrene	<0.03	0.53	0.14	23	<0.03	0.86	0.17	24	<0.03	0.73	0.11	33

< means less than the MDL.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the MDL.

[c] All concentrations recorded below the MDL.

Table 2-6. Concentrations of Phenolics in Spent Foundry Sands

Compound	Collected June 2005, 43 Samples[a] (mg kg^{-1})				Collected September 2005, 38 Samples (mg kg^{-1})				Collected July 2006, 37 Samples (mg kg^{-1})			
	Min	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
2-sec-Butyl-4,6-dinitrophenol[c]	<0.21		0.11	0	<0.21		0.11	0	<0.21		0.11	0
4-Chloro-3-methylphenol	<0.18	0.82	0.11	2	<0.18	0.45	0.10	1	<0.18		0.09	0
2-Chlorophenol[c]	<0.11		0.06	0	<0.11		0.06	0	<0.11		0.06	0
2,4-Dichlorophenol[c]	<0.13		0.07	0	<0.13		0.07	0	<0.13		0.07	0
2,6-Dichlorophenol[c]	<0.06		0.03	0	<0.06		0.03	0	<0.06		0.03	0
2,4-Dimethylphenol	<0.08	12.3	1.13	27	<0.08	7.45	0.72	24	<0.08	10.9	1.12	25
2,4-Dinitrophenol	<0.24	0.86	0.14	1	<0.24		0.12	0	<0.24		0.12	0
2-Methylphenol	<0.21	14.9	2.19	32	<0.21	9.90	1.29	27	<0.21	10.5	1.85	24
3- and 4-Methylphenol	<0.08	6.11	0.99	30	<0.08	3.98	0.58	33	<0.08	4.70	0.9	27
2-Methyl-4,6-dinitrophenol[c]	<0.16		0.08	0	<0.16		0.08	0	<0.16		0.08	0
2-Nitrophenol[c]	<0.09		0.05	0	<0.09		0.05	0	<0.09		0.05	0
4-Nitrophenol[c]	<0.44		0.22	0	<0.44		0.22	0	<0.44		0.22	0
Pentachlorophenol[c]	<0.24		0.12	0	<0.24		0.12	0	<0.24		0.12	0
Phenol	<0.07	186	11.2	39	<0.07	50.0	4.41	35	<0.07	28.5	4.78	30
2,3,4,6-Tetrachlorophenol[c]	<0.09		0.05	0	<0.09		0.05	0	<0.09		0.05	0
2,4,6-Trichlorophenol[c]	<0.12		0.06	0	<0.12		0.06	0	<0.12		0.06	0
2,4,5-Trichlorophenol[c]	<0.14		0.07	0	<0.14		0.07	0	<0.14		0.07	0

< means less than the MDL.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the MDL.

[c] All concentrations recorded below the MDL.

Sand #12 (iron foundry that used novolac molds and cores) was the only sand where all of the PAHs were below the MDLs. A summary of PAH data from the two additional sampling events—that is, September 2005 and July 2006 can also be found in Table 2-5; the results are markedly similar to those found in the first sampling event. One exception is sand #6, where the naphthalene concentration during the first sampling event was 48.1 mg kg^{-1}, but by the second and third sampling event, it decreased to 8.3 and 0.16 mg kg^{-1}, respectively. The other exception is sand #5, in which the naphthalene concentration increased from 0.41 to 42.2 mg kg^{-1} by the third sampling event. It is possible that the sand storage practices at the foundries account for these differences.

Anthracene, fluorene, naphthalene, and phenanthrene were the most prevalent PAHs, detected in >79% of the SFSs (Dungan, 2006). No discernible trend between the PAH concentration and the type of molding sand, core binder, or metal poured was apparent. It is likely that other variables, such as casting and core size and sand handling and storage, play a role in the amount of organics found in the SFSs. Except for the naphthalene concentrations in SFSs #6, #33, and #41, the results obtained by Dungan (2006) were similar to those obtained by Lee and Benson (2006), who found that naphthalene (0.02–4.6 mg kg^{-1}), phenanthrene (0.08–0.9 mg kg^{-1}), and 2-methylnaphthalene (0.004–9.8 mg kg^{-1}) were generally present at higher concentrations than the other PAHs. PAH-specific data for individual samples are found in **Appendix B, Tables B-4, B-5, and B-6**.

In a study conducted by Ji et al. (2001), naphthalene, 1- and 2- methylnaphthalene, and phenanthrene were also at the highest concentrations in waste green sands from iron, steel, and aluminum foundries. When compared to chemically bonded sands, the PAH concentrations were higher in the green sands. Naphthalene accounted for about 30% of the PAHs found in all of the SFSs.

Of the 17 phenolics analyzed, 11 were at concentrations less than the MDL in all 43 SFSs in the June 2005 sampling event. Phenolics that were quantitatively detected in the majority of the SFSs were phenol, 2-methylphenol, 3- and 4-methylphenol, and 2,4-dimethylphenol. In general, phenol was found at the highest concentration, followed by 2-methylphenol and then 3- and 4-methylphenol and 2,4-dimethylphenol. Phenol was present in samples from 39 of 43 foundries at concentrations ranging from 0.12–186 mg kg^{-1}. Sand #6, from a steel foundry that used both phenolic urethane no-bake molds and cores, contained the highest concentration of phenol. In contrast, sand #29 was from a steel foundry that used the same mold and core binders, but it contained substantially less phenol at 0.36 mg kg^{-1}. The highest concentrations of 2-methylphenol, 3- and 4-methylphenol, and 2,4-dimethylphenol were 14.9 mg kg^{-1} (sand #34), 6.1 mg kg^{-1} (sand #20), and 12.3 mg kg^{-1} (sand #20), respectively. Of the remaining phenolics, only 2,4-dinitrophenol and 4-chloro-3-methylphenol were found at concentrations that slightly exceeded the MDL of 0.24 and 0.18 mg kg^{-1}, respectively, in sands #6, #38, and #41. Phenolic data from the two additional sampling events can also be found in Table 2-6. Constituent-specific data for individual samples are found in **Appendix B, Tables B-7, B-8, and B-9**.

PCDDs, PCDFs, and PCBs are ubiquitous environmental contaminants. They are nonpolar, lipophilic, persistent in the environment, and bioaccumulate in the food chain. Unlike PCBs, PCDDs and PCDFs were never intentionally manufactured, but are largely released into the environment during combustion processes. Ten representative spent sands from iron, aluminum, and steel foundries, shown in **Table 2-7**, were analyzed for PCDD/PCDFs and PCBs

(Dungan et al., 2009). The concentrations of the PCDD/PCDFs and PCBs in the SFSs, expressed as ng kg^{-1}, are presented in **Table 2-8**. Except for 1,2,3,7,8,9-HxCDF, the tetra (T), penta (Pe), hexa (Hx), hepta (Hp) and octa (O) congeners of PCDD and PCDF were found above the MDLs, but not in all SFSs. Concentrations of the PCDD congeners ranged from <0.01–44.8 ng kg^{-1}, with 1,2,3,4,6,7,8,9-OCDD being found at the highest concentration in all of the SFSs. Although the OCDD concentrations were the greatest, based on the TEF, OCDD is considered to be less toxic than 2,3,7,8-TCDD by four orders of magnitude. 2,3,7,8-TCDD, with concentrations ranging from <0.01-0.14 ng kg^{-1}, was detected in only 50% of the SFSs.

Table 2-7. Description of the Spent Foundry Sands Analyzed for PCDDs, PCDFs, and Coplanar PCBs

Sand	Metal Poured	Molding Sand	Core Binder System and Process
4	Aluminum	Green sand	Shell[a]
8	Iron	Green sand	PU[b] coldbox, PU hotbox
12	Iron	Shell	Shell
14	Aluminum	Green sand	PU no-bake, shell, core oil
16	Iron	Green sand	PU coldbox, PU hotbox
20	Aluminum	Green sand	Shell
28	Iron	Green sand	None
29	Steel	PU no-bake	PU no-bake
39	Steel	Green sand	PU coldbox, shell, resin/CO_2
43	Steel	Green sand	PU no-bake, shell, core oil, resin/CO_2

[a] Shell process associated with the use of novolac resin

[b] PU = phenolic urethane

Table 2-8. Concentrations of PCDDs, PCDFs, and Coplanar PCBs and Homolog Totals in the Spent Foundry Sands (*n* =1)

Congener	TEF[a]	Spent Foundry Sand (ng kg^{-1})									
		4	**8**	**12**	**14**	**16**	**20**	**28**	**29**	**39**	**43**
2,3,7,8-TCDD	1	<0.02	0.03	<0.01	<0.02	<0.05	0.02	<0.03	0.02	0.14	0.07
1,2,3,7,8-PeCDD	1	0.03	0.13	0.02	0.03	<0.04	0.07	<0.03	0.15	0.72	0.24
1,2,3,4,7,8-HxCDD	0.1	0.02	0.09	<0.02	0.01	0.02	<0.02	<0.04	0.16	0.58	0.21
1,2,3,6,7,8-HxCDD	0.1	0.05	0.60	0.02	0.02	0.05	0.18	<0.04	0.21	0.81	0.33
1,2,3,7,8,9-HxCDD	0.1	0.03	0.35	<0.02	0.03	0.06	0.13	<0.04	0.15	0.66	0.23
1,2,3,4,6,7,8-HpCDD	0.01	0.38	5.29	0.42	0.15	0.60	0.74	0.21	1.24	5.00	1.62
1,2,3,4,6,7,8,9-OCDD	0.0003	27.8	44.8	2.89	1.60	8.76	5.89	2.95	3.01	12.5	2.42
2,3,7,8-TCDF	0.1	0.03	0.46	0.03	0.03	0.16	0.09	0.01	0.13	1.69	0.45
1,2,3,7,8-PeCDF	0.03	0.03	0.19	0.01	0.01	0.07	0.13	0.02	0.15	1.50	0.46
2,3,4,7,8-PeCDF	0.3	0.04	0.29	0.01	0.01	0.08	0.20	<0.04	0.21	2.61	0.72
1,2,3,4,7,8-HxCDF	0.1	0.06	0.25	0.01	<0.01	0.10	0.18	<0.04	0.18	2.32	0.63
1,2,3,6,7,8-HxCDF	0.1	0.04	0.18	0.01	0.01	0.04	0.15	<0.03	0.15	2.30	0.56
2,3,4,6,7,8-HxCDF	0.1	0.04	0.22	0.01	<0.01	0.02	0.24	<0.03	0.17	2.34	0.55
1,2,3,7,8,9-HxCDF	0.1	<0.02	<0.03	<0.02	<0.01	<0.03	<0.02	<0.03	<0.02	<0.04	<0.02
1,2,3,4,6,7,8-HpCDF	0.1	0.17	1.01	0.13	0.02	0.11	0.48	0.14	0.73	9.93	1.72
1,2,3,4,7,8,9-HpCDF	0.01	0.03	0.11	<0.02	<0.02	0.03	0.06	0.17	0.06	0.50	0.10
1,2,3,4,6,7,8,9-OCDF	0.0003	0.12	1.51	0.48	0.09	0.16	0.36	0.16	0.26	3.10	0.26
PCB-77	0.0001	0.30	47.4	0.43	2.03	7.14	2.13	0.53	0.81	4.35	1.21
PCB-126	0.1	0.12	1.22	0.02	0.06	0.24	0.72	0.01	0.22	1.99	0.38
PCB-169	0.03	0.02	0.09	<0.01	0.02	<0.03	0.06	<0.02	0.05	0.68	0.12

Congener	TEF[a]	Spent Foundry Sand (ng kg^{-1})									
		4	8	12	14	16	20	28	29	39	43
Sum TCDD		0.33	1.41	0.01	0.22	0.58	2.80	0.24	9.78	21.8	9.58
Sum PeCDD		0.33	1.37	0.00	0.17	0.42	1.51	0.83	8.39	20.7	9.70
Sum HxCDD		0.42	5.01	0.07	0.23	0.90	2.24	0.42	8.12	22.7	9.64
Sum HpCDD		0.76	10.3	0.63	0.35	1.48	1.52	0.44	2.71	10.2	3.54
Sum TCDF		0.66	5.10	0.33	0.50	1.59	5.32	0.08	6.06	53.0	16.8
Sum PeCDF		0.55	2.75	0.15	0.16	0.57	2.89	0.21	3.25	32.8	9.31
Sum HxCDF		0.46	2.22	0.14	0.10	0.45	1.52	0.37	1.89	22.1	5.55
Sum HpCDF		0.28	2.07	0.36	0.06	0.25	0.78	0.32	0.94	12.1	2.10

< means less than the MDL.

Source: Dungan et al. (2009).

[a] Values assigned by WHO (Van den Berg et al., 2006).

Table 2-9 shows the PCDD, PCDF, PCB, and total dioxin concentrations corrected for their TEFs and expressed as TEQs. However, because PCB-81 and mono-*ortho*-substituted PCBs were not measured, the PCB contribution to the total TEQ concentration is not known. Total dioxin concentrations ranged from 0.01–3.13 ng TEQ kg^{-1}, with an average concentration of 0.58 ng TEQ kg^{-1}. The highest total dioxin concentration of 3.13 ng TEQ kg^{-1} was found in sand #39, (a green sand from a steel foundry). This concentration is about 100 times lower than the 300 ng TEQ kg^{-1} limit considered by EPA for biosolids (U.S. EPA, 2002e). In sand #39, 23%, 25%, and 22% of the TEQ was attributed to 1,2,3,7,8-PeCDD, 2,3,4,7,8-PeCDF, and HxCDFs, respectively. Only 5% of the TEQ could be attributed 2,3,7,8-TCDD, the most toxic dioxin congener. Other SFSs with higher TEQs were sands #8 and #43 (green sands from iron and steel foundries), at 0.68 and 0.91 ng TEQ kg^{-1}, respectively. In sand #8, 49%, 32%, and 19% of the TEQ was attributed to PCDDs, PCDFs, and PCBs, respectively. In sand #43, 44%, 51%, and 5% of the TEQ was attributed to PCDDs, PCDFs, and PCBs, respectively. In the remaining SFSs, PCDDs and PCDFs accounted for 76 to 94% of the total TEQ.

Table 2-9. Toxicity Equivalents (TEQs) of PCDDs, PCDFs, Coplanar PCBs, and Total Dioxins in the Spent Foundry Sands

	Spent Foundry Sand (ng TEQ kg^{-1})									
	4	**8**	**12**	**14**	**16**	**20**	**28**	**29**	**39**	**43**
PCDDs	0.05	0.33	0.02	0.03	0.02	0.13	0.00[a]	0.23	1.12	0.40
PCDFs	0.03	0.22	0.01	0.01	0.06	0.14	0.01	0.14	1.80	0.47
PCBs	0.01	0.13	0.00	0.01	0.03	0.07	0.00 [a]	0.02	0.22	0.04
Total[a]	0.10	0.68	0.04	0.05	0.11	0.34	0.01	0.40	3.13	0.91

[a] Sufficiently low that it rounds to zero.

[b] Sum of the PCDDs, PCDFs, and PCBs; does not include mono-*ortho*-substituted PCBs.

2.5.4 Constituent Leaching Potential

The amount of any constituent that might be mobilized (leached) from a waste or material depends on the constituent of concern, the matrix of the waste or material, and the environmental conditions under which the waste or material is managed. It is important to have information about the potential for the constituents to leach because leached constituents could be transported to groundwater. Laboratory leaching tests are often used to determine the potential for a given waste material to contaminate groundwater. Over the past two decades, a number of studies have characterized the leaching potential of chemical constituents from SFSs and their impact on the environment (Ham et al., 1981, 1986, 1993; Stanforth et al., 1988; Krueger et al., 1989; Regan et al, 1994; Riediker et al., 2000; Lee and Benson, 2006). Many of these studies used the extraction procedure (EP) toxicity test (U.S. EPA, SW-846 method 1310B), which was later replaced by the TCLP. The TCLP was designed to determine the leachability of 25 organic compounds, 8 trace elements, and 6 pesticides regulated under the Resource Conservation and Recovery Act of 1976 (RCRA).

The main drawback of the TCLP and EP for gathering data to assess SFS soil-related applications is that they simulate leaching in an environment very different from that found in such beneficial use scenarios. For example, the TCLP uses organic acids to simulate the

The TCLP

The TCLP estimates leachate concentrations, which are used by EPA to determine whether a solid waste exhibits the hazardous characteristic of toxicity (Kendall, 2003). A waste exhibits the toxicity characteristic under RCRA if any one of the constituents in the TCLP leachate exceeds its RCRA Toxicity Characteristic regulatory limit. Conversely, if leachate estimates do not exceed the regulatory limits, the waste is not considered to exhibit the characteristic of toxicity and thus, is not a hazardous waste under RCRA. The test was designed to determine the mobility of both inorganic and organic analytes present in liquids, solids, and multiphasic wastes in landfills. The Toxicity Characteristic regulatory levels are 100 times the National Primary Drinking Water Standards (NPDWSs). This factor was established by EPA because it is assumed that constituents in the leachate will be diluted and attenuated as they seep from an unlined landfill.

conditions found within municipal solid waste landfills. A buffered solution of acetic acid is used as the extraction fluid in the TCLP, and the pH of the extraction fluid is 4.93 ± 0.05 (or 2.88 ± 0.05 for highly alkaline wastes). SFS would be used in various surficial applications and would not be exposed to water containing high concentrations of acetic acid or water with such a low pH; thus, TCLP test conditions have limited relevance to determining the acceptability of soil-related beneficial uses of SFS.

Nevertheless, TCLP is often used because (1) it is commercially available and (2) some state beneficial use determination processes require that SFSs be tested using EPA-approved methods for the analysis of solid wastes. The concentrations of 10 elements in TCLP extracts from SFSs are summarized in **Table 2-10** (Dungan and Dees, 2009). Similar TCLP results were obtained for samples that were collected from the same foundries at later dates (also in Table 2-10). Element-specific data for each sample are detailed in **Appendix B**.

Dungan and Dees (2006) used the TCLP to assess the leachability of other elements that are not regulated under RCRA Subtitle C, including antimony, beryllium, copper, nickel, and zinc. In the vast majority of cases, these elements were not detected. A few exceptions did occur where copper, nickel, and zinc were detected in the TCLP extracts. During the first sampling event, both copper and zinc at 3.5 and 37.6 mg L^{-1}, respectively, were at the highest levels in the extract from sand #34 (i.e., non-leaded brass green sand), which also contained the highest total copper and zinc concentrations (see Table 2-3). The TCLP extract from sand #2 (which had the highest total nickel concentration at 2,328 mg kg^{-1}) contained 0.94 mg Ni L^{-1}. However, the TCLP extract from sand #39 contained the highest concentration of nickel at 1.5 mg L^{-1}, although its total nickel concentration was about 22 times lower than that of sand #2. These data appear to support the premise that the total element concentrations should not be used to predict the amount of the element that is likely to leach from the SFS.

To our knowledge, published data do not exist that link the trace element concentrations in TCLP leachates and their relationship to an industrial landfill or beneficial use field results. Ham et al. (1986) found no relationship between the trace element concentrations in laboratory leach extracts and those found in the unsaturated zone, saturated zone, and groundwater at ferrous foundry landfills. As discussed above, the environmental conditions that the TCLP simulates are unlike the conditions in which SFS would be beneficially used in soil-related applications. Therefore, the most appropriate use of TCLP analytical data is to test whether SFSs are hazardous waste under RCRA Subtitle C. As illustrated in **Table 2-11**, based on existing data, SFSs do not exhibit the Toxicity Characteristic.

Table 2-10. Metal Concentrations in the TCLP Extracts from the Spent Foundry Sands

	Collected June 2005. 43 Samples[a] (Concentration, mg L^{-1})				**Collected September 2005. 38 Samples (Concentration, mg L^{-1})**				**Collected July 2006. 37 Samples (Concentration, mg L^{-1})**			
Element	**Min**	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
Ag[c]	<0.04		0.020	0	<0.04		0.020	0	<0.04		0.020	0
As	<0.001	2.40	0.058	24	<0.001	0.019	0.003	25	<0.001	0.017	0.003	23
Ba	<0.86	1.13	0.446	1	<0.86		0.430	0	<0.86		0.430	0
Be	<0.01	0.043	0.007	3	<0.01		0.005	0	<0.01		0.005	0
Cd	<0.01	0.065	0.007	3	<0.01		0.005	0	<0.01	0.064	0.007	1
Cr[b]	<0.46		0.230	0	<0.46		0.230	0	<0.46		0.230	0
Cu	<0.10	3.52	0.193	8	<0.10	43.9	1.23	6	<0.10	5.39	0.194	1
Ni	<0.14	1.50	0.163	9	<0.14	0.298	0.092	6	<0.14	1.71	0.128	4
Pb	<0.05	0.098	0.027	1	<0.05		0.025	0	<0.05	1.13	0.055	1
Sb[c]	<0.02		0.010	0	<0.02		0.010	0	<0.02		0.010	0
Zn	<0.41	37.6	1.16	3	<0.41	40.3	1.47	4	<0.41	42.5	1.49	4

< means less than the LOQ.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the LOQ.

[c] All concentrations recorded below the LOQ.

Table 2-11. Spent Foundry Sands TCLP Extracts Compared to Toxicity Characteristic Regulatory Levels

Element	All Sampling Events, 118 Samples (Concentration, mg L^{-1})			Toxicity Characteristic Regulatory Level
	Min	Max	Mean	
Ag[a]	<0.04			5.0
As	<0.001	2.40	0.02	5.0
Ba	<0.86	1.13	0.44	100.0
Be	<0.01	0.04	0.01	
Cd	<0.01	0.06	0.01	1.0
Cr[a]	<0.46			5.0
Cu	<0.10	43.9	0.53	
Ni	<0.14	1.71	0.13	
Pb	<0.05	1.13	0.03	5.0
Sb[a]	<0.02			
Zn	<0.41	42.5	1.36	

< means less than the LOQ.

[a] All levels recorded below LOQ.

An alternative leaching procedure, the SPLP (SW-846 method 1312) was designed to simulate the leaching of trace elements and organics from wastes or contaminated soils due to acidic rainfall. Because the environmental conditions being mimicked or approximated by the SPLP are more similar to some beneficial use situations than those approximated by the TCLP, SPLP provides a more realistic estimate of trace element and organic mobility under field conditions during precipitation events.[12] Summary SPLP extract data from the 43 SFSs are presented in **Table 2-12**. In every extract, antimony, beryllium, cadmium, chromium, lead, nickel, and silver were below their respective LOQ. Arsenic, barium, copper, and zinc were detected in some of the SPLP extracts. SPLP extracts of SFSs from the second and third sampling events demonstrate similar results (also in Table 2-12). Compared to the TCLP leaching results, which is run at a pH of 4.93 buffered by acetic acid, fewer trace elements were found to be above the LOQ in the SPLP extract, which has an initial pH of 4.2. This can be explained by the fact that the strong mineral acids used to make the SPLP extracting solution provide little buffering capacity. After the extraction, the pH in the SPLP extracts was higher (pH range of 4.8–9.9) than in the TCLP extracts (pH range of 4.6–5.7). Some elements tend to be less soluble at the higher pH range found in the SPLP extracts.

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[12] The SPLP may not be used to assess the Toxicity Characteristic of a solid waste.

Table 2-12. Metal Concentrations in the SPLP Extracts from the Spent Foundry Sands

Element	Collected June 2005, 43 Samples[a] (Concentration, mg L^{-1})				Collected September 2005, 38 Samples (Concentration, mg L^{-1})				Collected July 2006, 37 Samples (Concentration, mg L^{-1})			
	Min	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
Ag[c]	<0.08		0.040	0	<0.08		0.040	0	<0.08		0.040	0
As	<0.001	0.098	0.006	25	<0.001	0.024	0.008	24	<0.001	0.017	0.004	28
Ba	<0.23	0.612	0.161	9	<0.23	0.371	0.129	3	<0.23	0.634	0.154	5
Be[c]	<0.02		0.010	0	<0.02		0.010	0	<0.02		0.010	0
Cd[c]	<0.01		0.005	0	<0.01		0.005	0	<0.01		0.005	0
Cr[c]	<0.01		0.005	0	<0.01		0.005	0	<0.01		0.005	0
Cu	<0.21	0.546	0.115	1	<0.21	0.748	0.122	1	<0.21	1.66	0.147	1
Ni	<0.05	0.238	0.030	1	<0.05	0.089	0.028	3	<0.05	0.070	0.026	1
Pb[b]	<0.08		0.040	0	<0.08		0.040	0	<0.08	0.284	0.047	1
Sb[c]	<0.04		0.020	0	<0.04		0.020	0	<0.04		0.020	0
Zn	<0.18	3.05	0.165	2	<0.18	1.62	0.130	1	<0.18	3.95	0.194	1

< means less than the LOQ.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the LOQ.

[c] All concentrations recorded below the LOQ.

The TCLP and SPLP represent standard tests that are widely used by the EPA and other regulatory agencies to evaluate the potential for constituent release into the subsurface. With few exceptions,[13] the aggressive conditions of the TCLP are thought to provide a *very* conservative screen for leach potential. The scenario that the TCLP mimics, however, is not representative of SFS use in manufactured soil because the level of acidity will overestimate constituent release. In addition, the organic component of manufactured soils (e.g., composts, peat moss, pine bark, biosolids) would likely sorb elements released from the molding sand (Basta et al., 2005; Kumpiene et al., 2008). The SPLP conditions that mimic acid rain are more relevant than TCLP for evaluating the conditions considered in this report.

Dungan and Dees (2009) also performed a shake extraction procedure using deionized water, known as ASTM D 3987 (ASTM International, 2004), on the 43 SFSs at a liquid-to-solid ratio of 1:20 (see **Table 2-13**). A comparison of the ASTM and TCLP results reveals that fewer elements were above the LOQ in the water extracts; also, concentrations were generally lower in the water extracts than concentrations from the TCLP. As discussed above, these results indicate that pH is a factor affecting the leaching of elements from the SFSs. As with the non-buffered SPLP extracting solution, the water used for the ASTM procedure is non-buffered. The pH of the extracts from the ASTM procedure ranged from 4.7 to as high as 9.9, which explains why the results are similar to those from the SPLP. In the water extracts from all SFSs, the concentrations of silver, barium, beryllium, cadmium, lead, and antimony were below their respective LOQ. The only water extracts that contained copper and zinc at concentrations that were one to two orders of magnitude higher than the LOQ were from sands #33 and #34. The copper and zinc concentrations in the extract from sand #33 were 1.1 and 1.0 mg L^{-1}, while in sand #34, they were 0.3 and 1.3 mg L^{-1}, respectively. With respect to arsenic in the water extracts, 21 of 43 sands were below the LOQ. The water extract from sand #5 (green sand from an iron foundry with 0.65 mg arsenic kg^{-1}) had the highest concentration of arsenic at 0.018 mg L^{-1}. Sand #27 (another green sand from an iron foundry), however, with the highest total concentration of arsenic at 3.0 mg kg^{-1}, leached <0.003 mg arsenic L^{-1}. In a study by Lee and Benson (2006), arsenic in water extracts from 12 green sands ranged from 0.003 to 0.008 mg L^{-1}. Water extract data from the second and third sampling events can also be found in Table 2-13. As with the TCLP and SPLP results, the ASTM extract data from the subsequent sample sets were very similar to data from the first set.

For most elements, pore water concentrations (**Appendix B, Table B-26**) were low, and for many sands were below detection limits. However, plant nutrients are evident in SFS pore water. The 39 SFSs (brass and olivine sands were omitted) have median soluble concentrations of the macro nutrients calcium, magnesium, potassium, phosphorus, and sulfur of 32.5, 13.5, 27.3, 0.39, and 125 mg kg^{-1}, respectively, and median concentrations of the soluble micro nutrients boron, iron, manganese, zinc, copper, and molybdenum of 0.53, 1.14, 0.09, 0.05, 0.01, and 0.11 mg kg^{-1}, respectively. Only pore water aluminum is occasionally elevated, ranging from <0.2–1,847 mg Al kg^{-1}, with a median of 3.89 mg Al kg^{-1}. However, despite this large range, 33.3% of SFS pore waters were below the aluminum detection limit of 0.2 mg kg^{-1}. Not all aluminum species are phytotoxic, and it is unlikely that the soluble aluminum found in the raw SFS will remain stable in solution for long once blended with other soil components (Kinraide, 1991).

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[13] Recent research indicates that the TCLP may not provide an adequately conservative test for arsenic in mature landfills characterized by alkaline pH, low redox potential, biological activity, long retention time, and organic composition of mature landfills (e.g., Ghosh et al., 2004).

Table 2-13. Metal Concentrations in Water Extracts from the Spent Foundry Sands

	Collected June 2005. 43 Samples[a] (Concentration, mg L^{-1})				**Collected September 2005. 38 Samples (Concentration, mg L^{-1})**				**Collected July 2006. 37 Samples (Concentration, mg L^{-1})**			
Element	**Min**	**Max**	**Mean[b]**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**	**Min**	**Max**	**Mean**	**No. of Detects**
Ag[c]	<0.05		0.030	0	<0.05		0.030	0	<0.05		0.030	0
As	<0.003	0.018	0.005	23	<0.003	0.024	0.008	24	<0.003	0.017	0.005	24
Ba[c]	<0.24		0.120	0	<0.24		0.120	0	<0.24		0.120	0
Be[c]	<0.01		0.005	0	<0.01		0.005	0	<0.01		0.005	0
Cd[c]	<0.01		0.005	0	<0.01		0.005	0	<0.01		0.005	0
Cr[c]	<0.02		0.010	0	<0.02		0.010	0	<0.02		0.010	0
Cu	<0.07	1.06	0.070	2	<0.07	0.218	0.045	2	<0.07	0.080	0.041	1
Ni	<0.05	0.046	0.026	1	<0.05		0.026	0	<0.05		0.026	0
Pb[c]	<0.11		0.055	0	<0.11		0.055	0	<0.11		0.055	0
Sb[c]	<0.04		0.020	0	<0.04		0.020	0	<0.04		0.020	0
Zn	<0.22	1.34	0.159	2	<0.22		0.110	0	<0.22	1.57	0.150	1

< means less than the LOQ.

[a] Source: Dungan (2008) and Dungan and Dees (2009).

[b] Mean calculated with all non-detects set at one half the LOQ.

[c] All concentrations recorded below the LOQ.

2.5.5 Plant Uptake of Trace Metals from Spent Foundry Sands

To date, only a few studies on the growth of plants in SFSs have been reported. In a laboratory study conducted by Dungan and Dees (2007), high purity silica sand was mixed with 50% SFS (dry weight). Spinach (*Spinacia oleracea*, cv. Bloomsdale), radish (*Raphanus sativus*, cv. Cherry Belle), and perennial ryegrass (*Lolium perenne*, cv. Pizzazz) were grown with added soluble fertilizers (i.e., Hoagland's solution) to assess the phytoavailability of aluminum, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, molybdenum, nickel, vanadium, and zinc. The SFSs used in this study were from two aluminum, two iron, and two steel foundries. Plastic pots were used and filled with 1,500 g of the foundry sand blend. There were four replicates of each treatment, plus a control. The sand blends were adjusted to pH 6 with a dilute solution of H_2SO_4, because the pH of foundry sands tends to be slightly alkaline. After germination, the spinach and radish seedlings were thinned to three plants per pot. The ryegrass was planted with 1 g of seed per pot. The pots were watered with 150 mL of full-strength Hoagland's solution, alternating with the same volume of deionized water. Plastic saucers were used at the bottom of each pot so that the applied volume of deionized water and nutrient solution was allowed to be taken up. The pots were kept in a growth chamber at 20 ± 2°C, 50% humidity, and under a light-dark cycle of 16 hours light and 8 hours darkness. Radish globes and leaves were harvested at 27 days, and the spinach leaves with stems were harvested at 39 days. The perennial ryegrass was harvested three times, at 27, 57 and 87 days, by collecting all of the top growth when it reached a height of about 15 cm. After harvest, all plant parts were thoroughly rinsed with deionized water and then dried to constant weight at 65^{o}C. The plant samples were digested to determine total metals following the method of Kukier et al. (2004).

Although there were differences in the amounts of trace metals accumulated by the various plant species, excessive amounts of trace metals (i.e., above the amount necessary for proper plant nutrition and health) were not taken up, regardless of the SFS treatment (see **Appendix B, Tables B-20, B-21, and B-22**). For the spinach and radish, boron, copper, iron, manganese, and zinc were found to be within or close to the sufficiency range for agronomic crops. In the ryegrass cuttings at 27, 57, and 87 days, copper and zinc were within sufficiency ranges, but plants were iron deficient and contained elevated nontoxic concentrations of boron, manganese, and molybdenum.

To evaluate the transmission of nutrients and trace metals from SFS into plant tissue, Romaine lettuce (*Lactuca sativa*, cv. Parris Island Cos*)* was grown in 100% of a subset of 10 SFSs and a silica sand (play sand) control. Prior to planting, the SFS pH was reduced to a target pH of 7.5 ± 0.5 using 3 applications of a 2% acetic acid solution, with wetting and drying cycles between applications. Pots were prepared with 1 kg of pH-adjusted SFS or silica sand, the top 1.3 cm of which was amended with vermiculite to facilitate germination. To ensure nutrient sufficiency, each pot was amended with Miracle-Gro® (15% N + 30% P_2O_5 + 15% K_2O) to supply nitrogen, phosphorus, and potassium at 200, 230, and 190 mg kg^{-1}, respectively, in a split application. An additional 100 mg N kg^{-1} was added as NH_4NO_3. Twenty lettuce seeds were planted per pot. Three replicates of each SFS and the silica sand control were grown in a completely randomized design. Plants were grown in a controlled environment growth chamber with 18 hours of light per day, light temperatures of 20^{o}C, and dark temperatures of 18.5°C. Pots were thinned to four lettuce plants per pot (if more than four plants were present) at 14 days.

Lettuce was harvested after 40 days, rinsed in deionized water, and dried at 70°C for 48 hours and crushed by hand. The dried material was weighed to determine dry matter growth (DMG). Dry lettuce tissue (0.25 g) was predigested for 4 hours in 10 mL of aqua regia. Predigested samples were digested at 140°C for 4 hours, or until clear. Filtered (0.45 μm) solutions were analyzed by ICP-AES.

By growing lettuce in 100% sand and not allowing the pots to drain, there was no opportunity for dilution of either the plant nutrients or other trace metals. However, the poor physical properties of the sand, due to high bulk density, made germination difficult. Germination ranged from 23.3–100% with a mean of 67%. The only pots that had full germination were the silica sand control pots. However, for lettuce grown in SFS, DMG relative to that in the control pots (RDMG) ranged from 9.6–226%, with a mean of 110%. The SFS with low RDMG was also low for germination, so there were fewer plants. Generally, despite a slow start, lettuce grown in the SFS performed well. The average plant tissue concentration of the plant macronutrients nitrogen, phosphorus, potassium, and sulfur were all within the nutrient sufficiency levels, although calcium and magnesium were slightly low. For the micronutrients boron, copper, iron, manganese, molybdenum, and zinc, the tissue concentrations were all adequate. Arsenic tissue concentrations were below 1 mg kg^{-1}, except in the control sand, where they were 1.43 mg kg^{-1}, which is within the typical range for arsenic in plant tissue. Similarly, other trace metals found in SFS tissue were within or below the levels typically found in plant tissue.

In a greenhouse study conducted by Hindman et al. (2008), SFSs from two iron foundries and one aluminum foundry were blended with either yard trimmings compost, spent mushroom substrate (SMS), or biosolids compost, and a silt loam soil at a dry weight ratio of 6.5:1.5:2.0 (SFS: compost: subsoil). All manufactured soils were characterized as sandy loams. Each of the manufactured soils was initially amended with inorganic fertilizer and seeded with annual ryegrass, which was harvested seven times. The grass cuttings were analyzed for aluminum, boron, calcium, cadmium, copper, iron, potassium, magnesium, manganese, molybdenum, sodium, nickel, phosphorus, lead, sulfur, and zinc. The ryegrass yields in the manufactured soils exceeded the growth in natural topsoil, which was likely the result of the more available nitrogen. Among the manufactured soils, the SMS plus biosolids compost showed larger yields than blends containing yard compost. There was no evidence of trace metal deficiencies or toxicities in ryegrass on the manufactured soils. Ryegrass tissue analyses indicated that most tissue trace metal concentrations were lower or the same as the control and that most tissue nutrient concentrations fell within the sufficiency range.

2.5.6 Potential to Impact Soil Biota

Microorganisms

Bacteria are the most numerous organisms in soils, and are important because they are involved in essential processes, such as cycling of nutrients, biodegradation of organic pollutants, formation of humus, and the stabilization of soil structure. Inputs of toxic elements can alter the biological activity of soil microorganisms, sometimes causing a severe ecosystem disturbance. Affected soils often exhibit decreased microbial diversity, microbial biomass and enzyme activities, and lower respiration rates per unit biomass. An increasing body of evidence suggests that microorganisms are more sensitive to heavy metal pollution than the faunal or floral community growing on the same soil (Giller et al., 1998). However, a summary of the effects of trace elements on soil microorganisms from laboratory and field studies shows enormous differences between studies as to which element concentrations are toxic (Bååth, 1989). In particular, the addition of metal salts during short-term (acute toxicity) laboratory experiments is a poor predictor of long-term (chronic toxicity) effects on microbial biomass and activity (Renella et al., 2002). Further complications arise as pollution in field studies generally involves multiple elements, while laboratory studies often focus on a single element, making it difficult to draw conclusions regarding the toxicity of element combinations. Perhaps because of these difficulties, no advanced risk assessment schemes or regulatory policies have dealt with impacts on microorganisms in environmental risk assessments (Giller et al., 1999). Despite these obstacles, soil microorganisms are being examined as indicators of adverse effects of trace element pollution, which could potentially be used to define critical trace element loadings for soil protection (Chapman, 1999). Some measures used to investigate the response of soil microorganisms to trace element pollution are enzyme activity, microbial biomass, respiration rate, carbon mineralization, nitrogen fixation, and fatty acid composition (Rother et al. 1983; Ibekwe et al., 1995; Aceves et al., 1999; Lee et al., 2002; Broos et al., 2004; Zhang et al., 2006; Vásquez-Murrieta et al., 2006).

Leguminous plants are important in maintaining soil fertility because they contain within their root nodules symbiotic bacteria capable of fixing atmospheric nitrogen. Within soils, free-living associative and asymbiotic nitrogen-fixing microorganisms also play an important role, but generally fix less nitrogen (Stevenson, 1982). To date, many laboratory and field studies have investigated the impacts of trace elements on legumes and nitrogen-fixing bacteria (Rother et al., 1983; McGrath et al., 1988; Giller et al., 1986; Ibekwe et al., 1995, 1997; Smith, 1997; Lakzian et al., 2002; Broos et al., 2004, 2005). In an early experiment, Rother et al. (1983) reported only minor decreases in nitrogenase activity, plant size, and nodulation of white clover (*Trifolium repens*) growing on mine spoils containing up to 216 mg Cd kg^{-1}; 30,000 mg Pb kg^{-1}; and 20,000 mg Zn kg^{-1}. Rhizobia from other legume species have not been found to be inhibited by soil element concentrations below those which cause significant phytotoxicity (Heckman et al., 1986; Kinkle et al., 1987; Angle and Chaney, 1991; Angle et al., 1988; El-Aziz et al., 1991).

Although no specific studies have been conducted to assess the impacts of trace elements in SFSs on rhizobia, the results from the above-mentioned studies do not implicate SFS as having possible adverse effects on soil microbes, except for brass or other spent sands where trace element concentrations are up to a few orders of magnitude higher than element concentrations in native background soils. With the exception of a few SFSs where the concentrations of copper, nickel, and/or zinc are strongly elevated, minimal impacts on rhizobia can be expected to occur in SFS-amended soils. Due to the naturally low trace element concentrations in most ferrous and aluminum foundry sands (see Table 2-3), manufactured soils and agricultural soils amended with these SFSs will not reach element levels required to cause adverse effects on soil microbes. Furthermore, compared to the results obtained by Broos et al. (2005), all of the SFSs from iron, steel, and aluminum foundries contained cadmium at <5.9 mg kg^{-1} and zinc no higher than 352 mg kg^{-1} (**Appendix B, Table B-24**).

Dehydrogenases are intracellular enzymes involved in microbial respiratory metabolism (von Mersi and Schinner, 1991). The dehydrogenase activity (DHA) assay is a sensitive technique that has been used to assess microbial activities in soil amended with organic residues, composted municipal solid wastes, and biosolids (Obbard et al., 1994; Albiach et al., 2000;

García-Gil et al., 2000; Yang et al., 2003). In a study conducted by Dungan et al. (2006), the DHA of a sandy loam soil amended with green sands or chemically bonded sands at 10, 30, and 50% (dry weight) was determined. The green sands were obtained from iron, aluminum, and brass foundries, and the chemically bonded sands were made with phenol-formaldehyde or furfuryl alcohol based resins. Overall, the addition of these sands resulted in a decrease in the DHA that lasted throughout the 12-week experimental period (see **Figures 2-1 and 2-2**). This effect was largely determined to be a result of blending the sand into the soil, which subsequently reduced the total microbial population in the sample, and thus, resulted in decreased DHA. When plain silica sand with very low trace element levels was added to the soil at the same application rates, there was a decrease in the DHA as the blending ratio increased, which also lasted throughout the 12-week period. A brass green sand that contained high concentrations of copper, lead, and zinc at 8,496; 943; and 4,596 mg kg^{-1}, respectively, severely impacted the DHA. By week 12, no DHA was detected in the 30% and 50% treatments. In contrast, the DHA in soil amended with an aluminum green sand was 2.1 times higher (all blending ratios), on average, at week 4, and 1.4 times greater (30% and 50% treatments only) than the controls by week 12. In core sand–amended soil, the DHA results were similar to soils amended with aluminum and iron green sands. Increased activity in some treatments may be a result of the soil microorganisms utilizing the core resins as a carbon source.

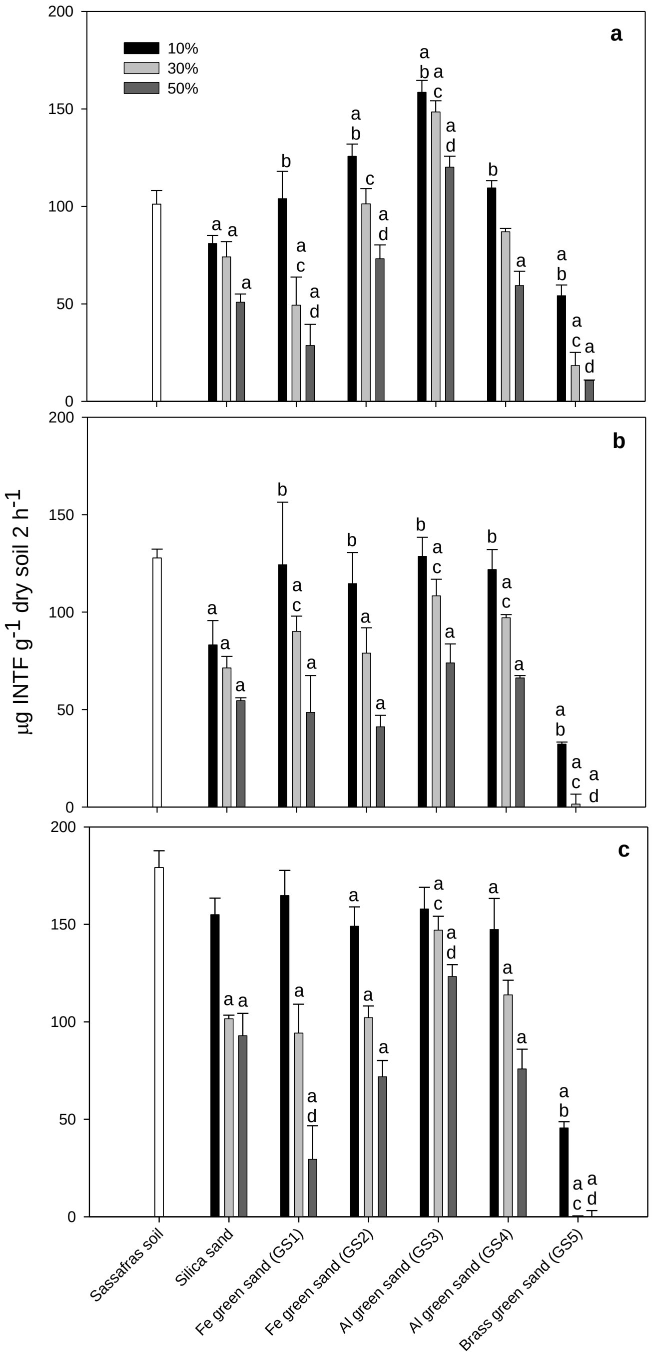

Figure 2-1. Dehydrogenase activities at (a) week 4, (b) week 8, and (c) week 12 in Sassafras sandy loam soil amended with 10%, 30%, and 50% (dry wt.) spent green sand from iron, aluminum, or brass foundries.

Error bars represent the standard deviation of triplicate samples. Treatments with letter *a* were significantly different (p <0.05) from the soil only control, while those with a letter *b*, *c*, or *d* were significantly different (p <0.05) from the respective silica sand treated soil.

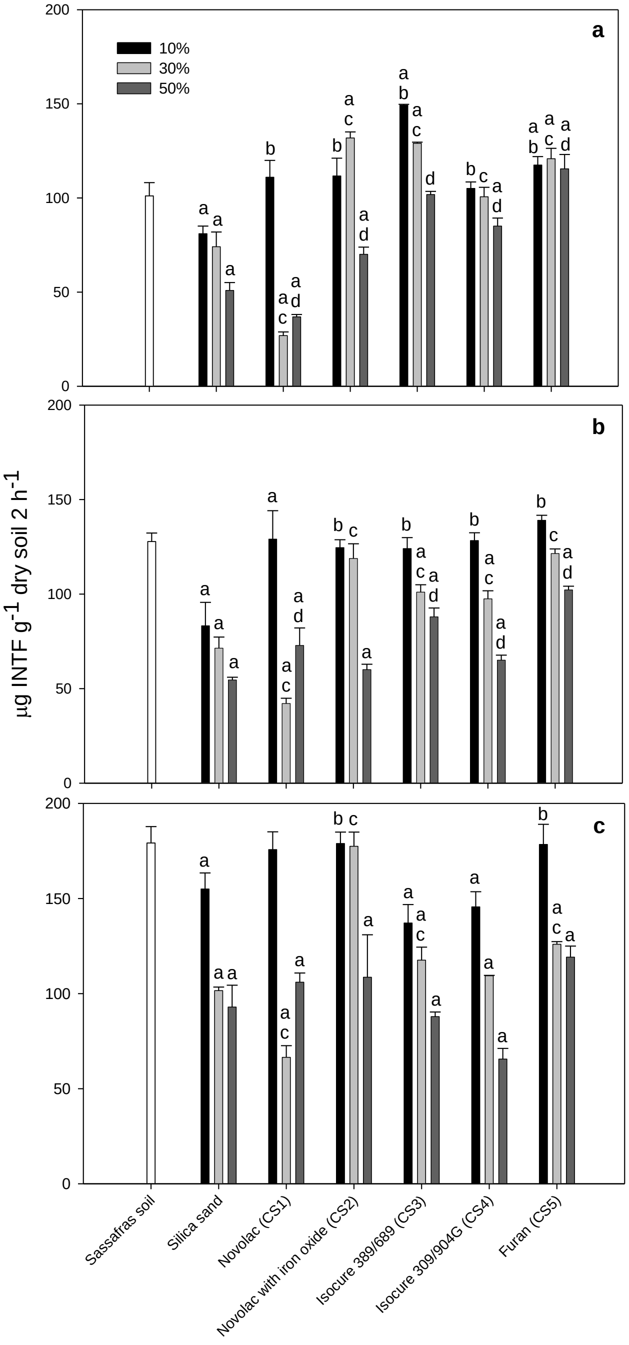

Figure 2-2. Dehydrogenase activities at (a) week 4, (b) week 8, and (c) week 12 in Sassafras sandy loam soil amended with 10%, 30%, and 50% (dry wt.) fresh core sand made with either phenol-formaldehyde, phenolic urethane, or furfuryl alcohol based resins.

Treatments with letter *a* were significantly different (p <0.05) from the soil only control, while those with a letter *b*, *c*, or *d* were significantly different (p <0.05) from the respective silica sand treated soil.

Earthworms

Earthworms play a beneficial role in the development of soil structure and fertility by enhancing the decomposition and mixing of organic matter and improving water infiltration and aeration (Lee, 1985). Earthworm activities are important in native grassland and woodland soils, as well as agricultural soils; however, earthworms have difficulty performing essential functions when they are exposed to harmful soil concentrations of trace elements (Edwards and Bohlen, 1996). Earthworms bioaccumulate some trace elements in their tissues as a result of oral (i.e., ingestion of large quantities of soil) and dermal routes of exposure (Helmke et al., 1979; Morgan and Morgan, 1999). As a result, earthworms living in some contaminated soils present a significant element-transfer risk to animals whose diet consists largely of earthworms (e.g., shrews, moles, badgers). If earthworms do survive in element-contaminated soils, it is more pertinent to examine the element risk to the earthworm-consuming animals than to assess the toxicity to the earthworms themselves (Chaney and Ryan, 1993; Brown et al., 2002). The accumulation of cadmium, lead, and zinc in moles has been shown to reflect the bioavailability of these elements to earthworms (Ma, 1987). In acidic sandy soils, cadmium accumulated in the earthworms to a considerable extent, and critical concentrations of cadmium toxicity in moles can be exceeded even when the soil cadmium concentration is relatively low. Earthworms and moles also accumulated much more lead from the contaminated acidic sandy soils than from soils that have been limed (Ma, 1987), demonstrating the importance of soil pH on element bioavailability to earthworms.

Many earthworm studies have been conducted to determine the effects of trace elements on survival, growth, cocoon production, litter breakdown, and the bioaccumulation of elements (Anderson, 1979; Hartenstein et al., 1980; Beyer et al., 1982, 1987; Ma, 1982, 1984; Khalil et al., 1996; Spurgeon and Hopkin, 1996; Morgan and Morgan, 1988, 1999; Posthuma et al., 1997; Conder and Lanno, 2000; Dai et al., 2004). A potential shortcoming of some of these studies is that they examined the effect of added metal salts (Ma, 1982, 1984; Khalil et al., 1996; Posthuma et al., 1997; Conder and Lanno, 2000), rather than contaminated field soils nearer equilibrium. When metal salts are added to soils (i.e., metal-spiking studies), they become more acidic with increasing metal rate as protons are displaced. Trace elements applied as salts are generally more bioavailable than those from mineralized or environmentally contaminated soils (Basta et al., 2005). When Ma (1984) corrected the acidity of copper salt amended soils, the high earthworm toxicity observed at low pH was reversed. Due to long-term soil-ageing processes, trace element availability generally decreases with time (Ford et al., 1997; Trivedi and Axe, 2000; Lock and Janssen, 2001). However, depending on the element and pH of the system, aging will not necessarily result in decreased element bioavailability (Lock and Janssen, 2003).

There is a relatively large amount of data on the concentration of trace elements in earthworms from biosolid-amended soils, smelter-contaminated soils, and mine spoils. In most reports, earthworms were not found to bioconcentrate lead and zinc, but earthworms have been found to bioconcentrate cadmium (Pietz et al., 1984; Beyer and Stafford, 1993). Cadmium concentrations in earthworms are generally greater than soil concentrations, while lead concentrations in earthworms are generally similar to or lower than soil concentrations. Beyer et al. (1990) examined the ratio of chromium in earthworms to that in soil of dredged material deposit sites and found no evidence of chromium accumulation. Helmke et al. (1979) found that chromium measured in earthworms was related to residual soil contamination. Many of these studies generally report the element concentrations in earthworms after the internal soil has been removed (i.e., purged or depurated). However, there is little evidence to suggest that earthworm consumers can avoid ingestion of the internal soil. From a risk perspective, it may be more appropriate to consider the element-transfer risk posed by earthworms that have not been purged (Brown et al., 2002) as approximately 35% of the unpurged earthworm dry weight is soil (Beyer and Stafford, 1993).

Dungan and Dees (2006) conducted a 28-day experiment with *Eisenia fetida* (a red worm adapted to manure or composts) to assess the bioavailability of trace elements in iron, aluminum, steel, and brass SFSs. The soil blends contained 10%, 30%, and 50% SFS on a dry-weight basis. After 28 days, the number of viable adult earthworms across all treatments and blending ratios was not significantly different from the control, except in blends containing 30% and 50% SFS from a brass foundry (see **Figure 2-3**). The high earthworm mortality in the brass sand blends correlated well with the high total and diethylenetriamine pentaacetic acid (DTPA)–extractable concentrations of copper, lead, and zinc (see **Table 2-14**). The DTPA procedure is widely used to determine plant available micronutrients in soils (Lindsay and Norvell, 1978) and has also been used to assess the accumulation of trace elements by earthworms (Dai et al., 2004). Trace element concentrations in the tissues of purged earthworms from iron, aluminum, and steel SFS blends did not exceed those in the control. The copper and zinc concentrations in worm tissue from the 10% brass blend were about 10 and 2 times higher than the control, respectively. Because of the high copper, lead, and zinc concentrations (i.e., above those found in background soils) in many brass molding sands, they should not be considered for beneficial use in manufactured soils or other unencapsulated uses.

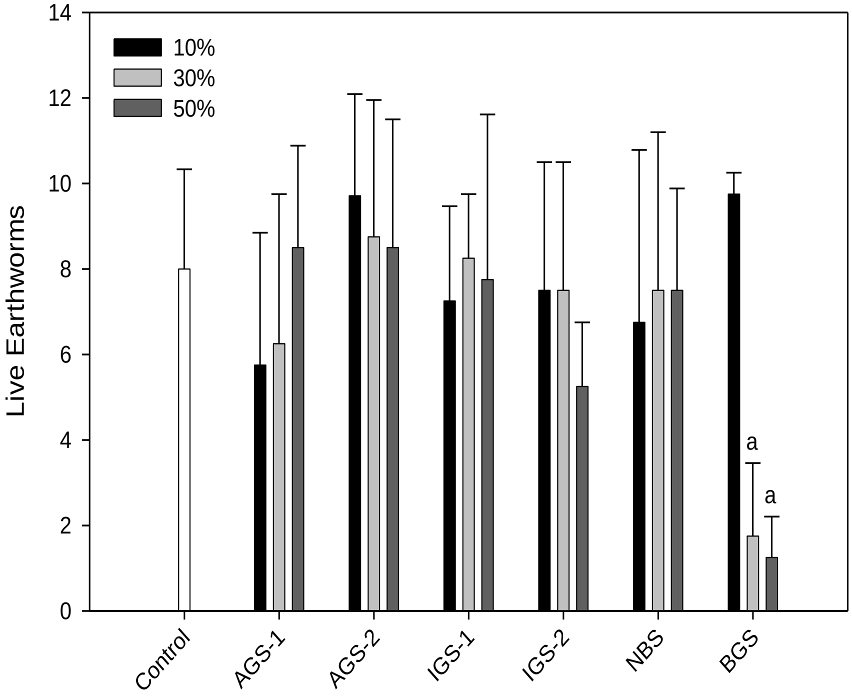

Figure 2-3. Adult earthworm survival after 28 days in the SFS blends.

Treatments marked with a letter are significantly different than the control (*p* <0.05, Holm-Sidak method). Error bars represent the standard deviation of four replicates (eight replicates in the case of the control). AGS = aluminum green sand; IGS = iron green sand; NBS = steel phenolic urethane no-bake sand; BGS = brass green sand.

Table 2-14. Total and DTPA-Extractable Metal Concentrations in the Brass Green Sand Blends

Blending Ratio	Copper			Lead			Zinc		
	Total[a]	DTPA (1:5)[a]	%[b]	Total	DTPA (1:5)	%	Total	DTPA (1:5)	%
10%	812.0	154.8	19.1	87.2	31.8	36.4	438.4	72.7	16.6
30%	2198.7	494.7	22.5	243.4	135.2	55.5	1186.4	194.7	16.4
50%	3713.3	884.5	23.8	386.2	216.7	56.1	1975.3	320.0	16.2

[a] mg kg^{-1}

[b] Percent of total metal that was DTPA extractable.

PAHs are common xenobiotic compounds in soils and are persistent because of their low mobility and resistance to degradation. Because PAHs are hydrophobic in nature, they tend to associate with soil organic matter and mineral fractions (Semple et al., 2003). The lipophilic nature of PAHs can result in the bioaccumulation of these chemicals by soil biota, such as earthworms (Krauss et al., 2000; Tang et al., 2002; Jager et al., 2003). As with trace element contaminants, the bioaccumulation of PAHs and other persistent lipophilic compounds (e.g., PCBs) by earthworms presents a potential risk to earthworm-consuming animals. However, as the soil-PAH contact time increases, there is a corresponding decrease in the extractability of the PAHs in the soil, and their bioavailability to earthworms also decreases with time (Kelsey and Alexander, 1997; Johnson et al., 2002). Johnson et al. (2002) found that tissue concentrations of pyrene and benz[a]anthracene in earthworms declined by 58% and 43%, respectively, after spiked soils were incubated for 240 days. In general, the extractability (via chemical extraction procedures) and bioavailability of xenobiotics in soils, composts, and biosolids has been found to decline substantially within months after application (Hatzinger and Alexander, 1995; Wang et al., 1995; Puglisi et al., 2007). This process is known as “aging” and results from the slow diffusion of xenobiotics to microsites or adsorption deeper into lipophilic soil organic matter particles (Alexander, 1995). Even low molecular weight xenobiotics can become aged and less bioavailable over time in soils (Frink and Bugbee, 1989; Guo et al., 2003). PAHs and phenolics are present in SFSs below background soil concentrations (Dungan, 2006), and because of the aging process, it is likely that these compounds will present a minimal risk to earthworms and higher organisms. Thus, as long as SFSs are managed appropriately, the concentrations of most organic compounds of concern will remain low and sensible land application of byproducts will result in minimal risk to animals, humans, and the environment from organics (Kester et al., 2005; Overcash et al., 2005).

3. Problem Formulation

The overall goals for this assessment are to (1) evaluate all available information on behavior of SFS in soils; (2) identify likely exposure pathways and receptors associated with various unencapsulated SFS use scenarios; and (3) determine whether the proposed SFS uses have the potential to cause adverse health or ecological effects (defined in this assessment as above 10^{-5} risk for cancer, and an HQ of 1 for noncancer and ecological effects). With these goals in mind, this chapter presents

- A description of the overall **scope** of the risk assessment, including the types, relevant characteristics, and proposed uses of SFS
- **Conceptual models** illustrating the environmental behavior and potential exposure pathways relevant to constituent releases from SFS in three soil-related applications
- The **analysis plan** developed to identify COCs and screen for potential risks associated with SFS use in manufactured soils, soil-less media, and road subbase.

3.1 Scope of the SFS Risk Screening

Chapter 2 presented the body of data used in this analysis. This is the most rigorous and consistent body of data available characterizing SFS and its constituents to date. The scope of this screening risk assessment focuses on specific “unencapsulated” uses of SFS. Unencapsulated uses present the highest potential for release of a material and its constituents because the material is not chemically or physically bound. Below is a summary of the types of SFS, constituents in SFS, and beneficial uses that are included in the scope of this analysis, as well as other information about the scope.

3.1.1 Types of SFSs

As described in **Chapter 2**, there are many different types of SFS. The assessment categorized SFSs according to three characteristics: the type of metal cast (e.g., aluminum, iron, brass), the mineral type of the virgin sand (e.g., silica, olivine), and the type of binder used (e.g., clay, chemical binders). Samples from 43 U.S. foundries were collected by USDA-ARS and industry, and analyzed by USDA-ARS. The characteristics of these samples are as follows:

- **Metal cast type:** 4 aluminum, 31 iron, 6 steel, and 2 non-leaded brass sands[14]
- **Mineral type:** 41 silica sands and 2 olivine sands
- **Binder type of molding sand:** 36 green sands and 7 chemically bound sands.

After a thorough review of the analytical data, described in **Chapter 2**, it was determined that the remainder of this evaluation would focus on silica-based SFSs from iron, steel, and aluminum foundries. Therefore, non-leaded brass sands and olivine sands would not be included in this analysis. One of the two non-leaded brass sand samples had high levels of copper and zinc

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[14] Sands from brass and bronze foundries that use lead are frequently hazardous waste because they leach lead at levels above the federal regulatory limit (see 40 CFR 261.24). Only nonhazardous SFSs are included in the scope of this evaluation. Therefore, sands from leaded brass and bronze foundries were not collected, and such sands were not evaluated in this study.

(3,318 and 1,640 mg kg^{-1}, respectively). Additionally, both olivine sand samples had high levels of nickel (2,328 and 1,022 mg kg^{-1}). As discussed in **Chapter 2, Section 2.5.2**, the nickel in those sand samples did not come from the foundry operations; rather, the mineral olivine is a magnesium iron silicate and contains naturally elevated concentrations of nickel, cobalt, and chromium. It is important to note that the olivine sands were not included in this assessment because there was limited constituent-specific data on those sand types. Further assessment of olivine sands from aluminum, iron, and steel foundries could be performed to determine the risk associated with the use of these sands in unencapsulated applications, and their exclusion from this assessment should not be interpreted to mean that olivine sands could not be considered or approved for such uses, where analytical data indicate they are acceptable.

3.1.2 SFS Characteristics

Both the chemical and physical characteristics of SFS are relevant to effects associated with their use. The sand, clay, and silt content of the SFS affect the potential for particle emissions and leaching. Smaller particle sizes (i.e., higher silt content and lower sand content) result in greater potential for particle emissions (because the individual particles are more readily released into the air) and for leaching (because a greater surface area of each particle is exposed to the precipitation and groundwater that leaches the constituents from the particle). As shown in Table 2-2, the silt content of SFS ranges from 0–16.9%, whereas the sand content ranges from 76.6–100%. The particle size information was used in the inhalation pathway screening assessment to calculate emission rates for SFS.

As discussed in **Chapter 2, Section 2.5.1**, leaching potential is affected by pH, especially for metals. For most metals, higher leaching occurs at the extreme ends of the acid/alkaline spectrum and lower leaching occurs when the leachate is neutral. However, other variables, such as redox potential, can significantly alter the leaching behavior of some metals (e.g., arsenic). Agricultural and horticultural uses of SFS generally require that the soil remain near neutral pH to promote healthy plant growth. Of the various types of leaching data presented in Chapter 2 (i.e., TCLP, SPLP, ASTM D3987, and pore water), this evaluation primarily used SPLP and ASTM data. SPLP simulates leaching due to acid rain, and is run at an unbuffered pH of 4.2. ASTM method estimates leaching at the material’s natural pH, which for SFS ranged from 6.67–10.2. These tests were performed on each SFS sample to empirically estimate the leaching potential. Leaching data are described in **Chapter 2, Section 2.5.4**, and presented in Tables 2-12 and 2-13. These data were used in this assessment to evaluate the groundwater and produce consumption pathways. In addition, TCLP data, estimated under very acidic conditions, were used when neither SPLP nor ASTM data were available (see **Chapter 4, section 4.2.1**). Finally, pore water data were used in refined ecological exposure modeling (see **Chapter 5, Section 5.3.8**).

The total concentrations of constituents were important inputs into both the screening process and the predictive risk modeling. Used initially to identify constituents for evaluation, total concentrations were also used to assess the inhalation pathway, the groundwater ingestion pathway, and the soil pathways (i.e., the ingestion of soil and home grown produce and dermal contact with soil). In addition, total concentrations were used in evaluating the potential for adverse effects to ecological receptors. Total concentration data for metals used in this evaluation are described in **Chapter 2, Section 2.5.2**, and presented in Table 2-4, and total concentrations of organics used in this evaluation are described in **Chapter 2, Section 2.5.3**, and presented in

Tables 2-5, 2-6, and 2-8. With the exception of arsenic exposure through incidental soil ingestion, the conservative assumption made in this screening risk assessment is that 100% of the total concentration of each constituent is biologically available to the receptors. This is a conservative assumption because, as discussed in **Chapter 2, Section 2.5.6**, metals exist in soils in solid phases, not as the more bioavailable soluble salts, and the metals become less bioavailable over time as soil organic components age. When assessing exposures to arsenic in soil, U.S. EPA (2012b) recommends applying a default relative bioavailability (RBA) value of 60% when a site-specific value in unavailable. This assessment used the recommended default value.

3.1.3 Beneficial Uses of SFS

In general, SFS can be used as an effective replacement for virgin sand in many geotechnical and agricultural applications. This evaluation focused on the following potential unencapsulated beneficial uses of SFS:

- Roadway construction as subbase
- Soil-less potting media for horticultural purposes
- Mineral component of manufactured soils.

Road subbase, soil-less potting media, and manufactured soils are discussed in greater detail below in **Section 3.1.4**.

3.1.4 Conceptual Models

The information on the SFS characteristics and constituents presented in **Chapter 2** was used to develop the conceptual models. The conceptual models describe the sources, exposure pathways, and receptors associated with SFS use in roadway construction, blending operations that produce manufactured soils and soil-less potting media, and use of manufactured soils in home gardens.

Figure 3-1 shows the conceptual model for SFS used as road subbase. Road subbase is a layer of material required in some roadway applications to change the physical characteristics of the land area on which the roadway is to be built so that the pavement is capable of withstanding the stress of vehicle traffic and seasonal changes (e.g., freeze/thaw cycles). The subbase is placed directly onto the subgrade and is covered by the base course, which is the layer in the roadway beneath the pavement. Subbase thickness varies depending on road type, site requirements, and material used, but sand subbase thickness typically ranges from 10–25 cm (i.e., 4-9 inches, U.S. ACE, 1984). Pre-use storage and processing would vary by proposed use, but would likely involve at least some storage in open areas. Rainfall on stored SFS piles or not yet covered subbase could potentially leach constituents that could migrate through the subsurface and contaminate an underlying groundwater aquifer. While possible, constituent releases into surface waterbodies are not likely to be significant because standard road construction practices include engineering controls to prevent significant runoff/erosion[15]. During loading and unloading

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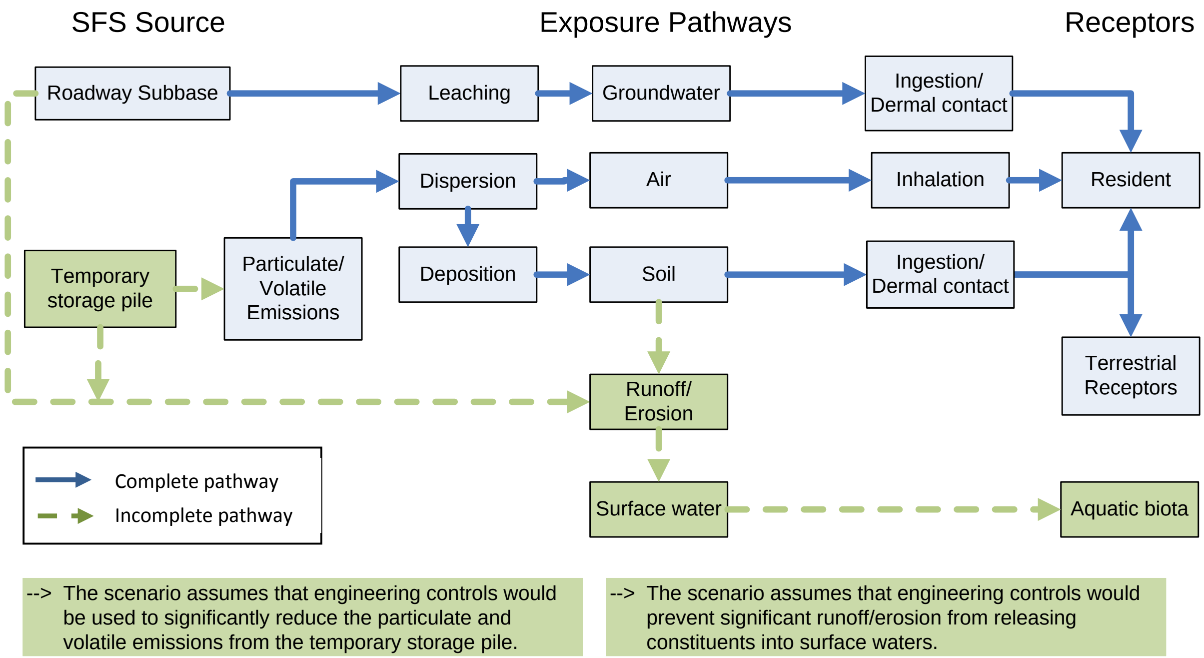
[15] Runoff controls are a legal requirement under the National Pollutant Discharge Elimination System (NPDES) that is part of the Clean Water Act. Most states have been authorized to implement the NPDES storm water program (http://cfpub.epa.gov/npdes/stormwater/authorizationstatus.cfm), although some areas (e.g., tribal lands) remain

operations at roadway construction sites, nearby residents could be exposed via the inhalation of particulate emissions and/or the incidental ingestion of soil following particle deposition; terrestrial receptors (e.g., small mammals, soil invertebrates) could be exposed to chemical constituents in SFS through direct and indirect exposure pathways.



--> The scenario assumes that engineering controls would be used to significantly reduce the particulate and volatile emissions from the temporary storage pile.

--> The scenario assumes that engineering controls would prevent significant runoff/erosion from releasing constituents into surface waters.

Figure 3-1. Conceptual model: the use of SFS in roadway subbase.

Given their inherent properties and low cost, SFS could potentially be of value as feedstock for the blending of soil-less potting media and manufactured soil. Soil-less potting media are generally used by nurseries as temporary growth media while individual plants await sale, whereas manufactured soils more closely mimic native soils, and can be used on a much larger scale as a long-term replacement for degraded native soils. Soil-less potting media and manufactured soil could be mixed at the site of application (e.g., manufactured soil blended at a construction site to landscape degraded topsoil), or mixed at a nursery, landscaping company, or commercial soil-blending operation (hereafter referred to collectively as blending sites). SFS used in these horticultural or agricultural applications is not encapsulated, and piles of SFS feedstock may be uncovered for short periods of time. **Figure 3-2** shows the conceptual model for residents near a blending site. This scenario assumes that SFS would be temporarily stored on site near other media components, along with piles of various blended soil and soil-less potting media.

If uncovered, rainfall on stored SFS and blended piles could potentially leach constituents; if the piles are stored on a pervious surface, these constituents could potentially

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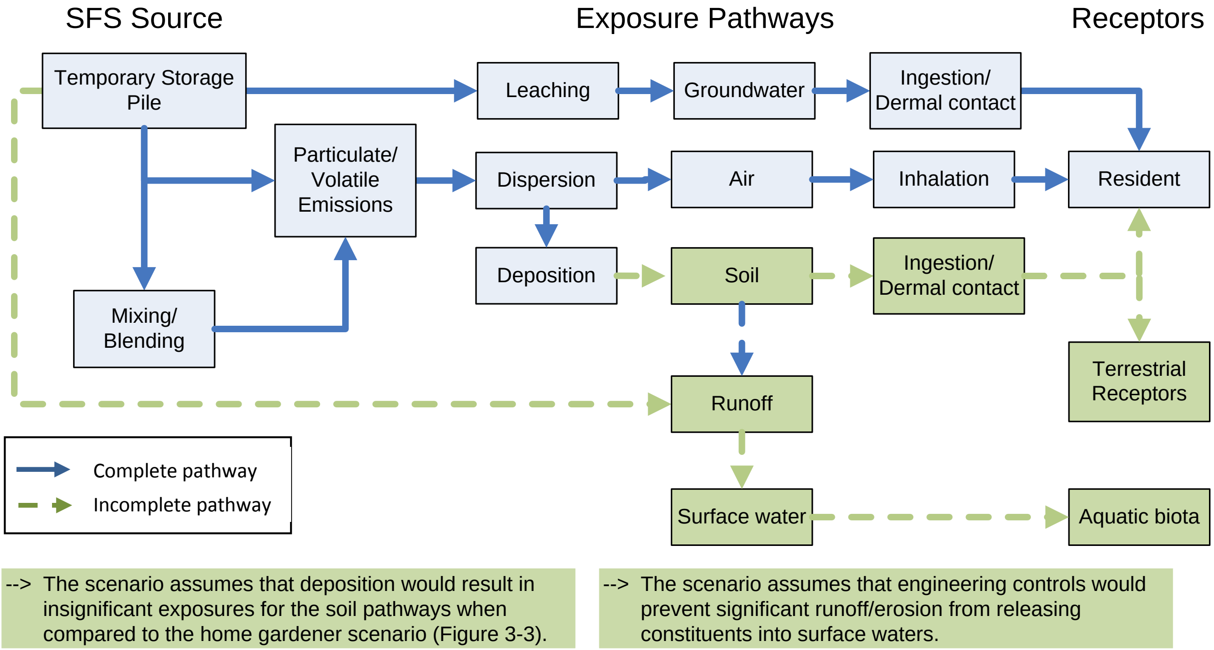
under the direction of EPA. The NPDES regulations establish best management practices (BMPs) for any source of sediment, from sites or operations (e.g., construction, agricultural, or industrial), that might impact surface waters. Many of the BMPs applicable to the control of runoff are similarly used to control fugitive dust emissions as required under the Clean Air Act.

migrate through the subsurface and contaminate an underlying aquifer. In addition, rainfall and windblown erosion could result in some portion of the SFS running off and possibly reaching nearby surface waters, assuming that the blending site did not include any sort of runoff collection system. Storage and blending processes at commercial soil-blending facilities could potentially be conducted on a much larger scale relative to storage and blending soil-less potting media, and cover a wide range of manufactured soil "recipes." During storage, and particularly during the blending process, chemical constituents could volatilize or be released via particulate emissions. Nearby residents could be exposed through the groundwater pathways or the inhalation of ambient air. Terrestrial receptors could be exposed to chemical constituents in SFS through direct and indirect exposure pathways.



--> The scenario assumes that deposition would result in insignificant exposures for the soil pathways when compared to the home gardener scenario (Figure 3-3).

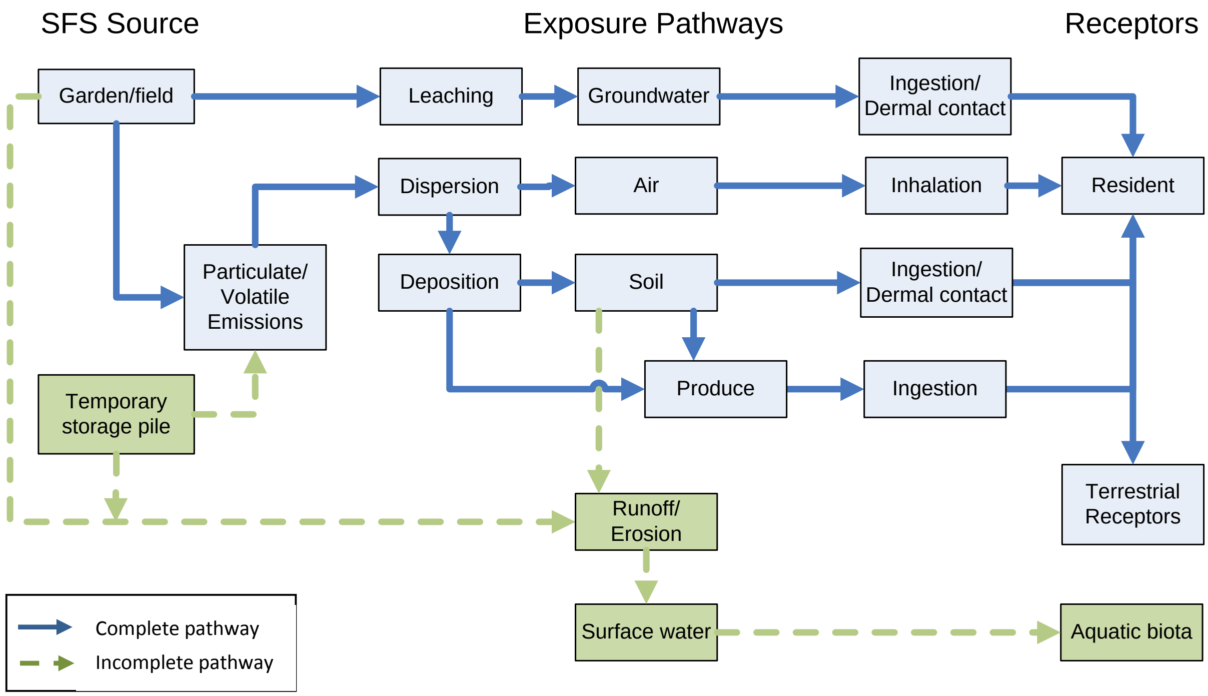
--> The scenario assumes that engineering controls would prevent significant runoff/erosion from releasing constituents into surface waters.

Figure 3-2. Conceptual model: the blending site.

Figure 3-3 shows the conceptual model for the use of SFS-manufactured soil (i.e., blended soils containing SFS) in home gardens. Although SFS-manufactured soil could be used in corporate and residential landscaping (e.g., resurfacing construction sites), the home gardener could potentially receive a much higher exposure to SFS constituents under the following assumptions

- The home gardener incorporates a significant amount of SFS-manufactured soil into the home garden
- The home gardener frequently works in the garden, thereby increasing the opportunities of dermal contact and incidental ingestion of the SFS-manufactured soil, and
- A significant portion of produce consumed by the home gardener would be taken from the garden consisting of SFS-manufactured soil.

Because the SFS-manufactured soil is unencapsulated, direct exposures (e.g., ingestion, dermal contact) could occur, and constituents could leach from the home garden following rainfall events and/or irrigation. Additionally, terrestrial receptors could be exposed to chemical constituents in SFS through direct and indirect exposure pathways.



--> The scenario assumes that manufactured soil is used soon after delivery, so constituent releases from the temporary storage pile are insignificant.

--> The scenario assumes that the home gardener would impose controls to prevent significant runoff/erosion of manufactured soil from the garden.

Figure 3-3. Conceptual model: the use of SFS-manufactured soils in home gardens.

The three conceptual models shown above were used in developing the Analytical Plan discussed in **Section 3.3**.

3.1.5 Assumptions Behind the Risk Screening

The development of these conceptual models included assumptions that influenced the selection of which exposure pathways to evaluate. These assumptions include the following:

- Acute and short-term worker exposures during application would be addressed by existing standards developed by the Occupational Safety and Health Administration (OSHA), and therefore potential worker exposures were not evaluated.
- For the temporary storage and use of SFS, indirect exposure pathways (e.g., air emissions to soil deposition to soil-to-plant uptake to ingestion) would be unlikely to produce significant exposures because
  - there would likely be engineered controls to prevent the loss of valued commodities, such as SFS feedstocks or blended soils,

  - few chemical constituents have been shown to biomagnify in terrestrial food webs,[16]
  - the time to reach steady state with respect to plant and animal concentrations would be insufficient, so bioaccumulation would be limited, and
  - releases during roadway construction using SFS would be temporary and intermittent and, as a result, the potential for exposure would be very limited.

- The potential for exposure via direct contact (e.g., human incidental soil ingestion, ecological exposures) is greater in the home garden scenario than the temporary storage and use at blending facilities because air emissions and deposition from blending facilities or other temporary storage piles are unlikely to result in residential soil concentrations greater than those found in home gardens.

- The potential to contaminate groundwater is greater in the home garden scenario than the other scenarios because (1) the SFS would remain in the garden indefinitely, (2) the SFS is incorporated into the soil rather than sitting on top of the soil, (3) the garden presents a much larger footprint (approximately 405 m^2) than the temporary storage pile (assumed to be 150 m^2 in size), and (4) the soil underlying a garden would likely have a higher hydraulic conductivity than a compacted soil or concrete pad used for the temporary storage of SFS.

- Because SFS and manufactured soils have economic value[17], blending sites would process the SFS as rapidly as possible to generate revenue. This means that (1) the temporary storage pile would remain in place for a relatively short period of time before soil blending, and (2) the storage pile would likely be managed to protect the material's value and workability (e.g., use of a temporary cover to prevent loss due to runoff, and prevent the pile from becoming saturated with water).

- Commercial blending facilities demonstrate the greatest potential for nearby residential inhalation exposures, because they tend to work with larger volumes of feedstock and product (thereby emitting greater volumes of particulates) and conduct operations throughout the year.

- The economics of purchasing, transporting, and applying SFS-manufactured soil would make its large-scale agronomic application untenable – farmers could not afford it.[18] Other potential agronomic uses for SFS (e.g., to improve soil texture) involve application rates that would result in SFS concentrations lower than the assumed 1:1 blend (i.e., the soil is 50% SFS, by weight) in SFS-manufactured soil.

In addition to these overarching assumptions, the risk assessment was predicated on a number of conservative assumptions intended to ensure that the results could be used to support management decisions with a high degree of confidence. That is, the assessment was intentionally designed *not* to underestimate the potential risks to human health and the environment.

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[16] With the exception of certain persistent organic pollutants, such as dioxins and PCBs, we are not aware of any studies demonstrating biomagnification for multiple trophic levels (e.g., from terrestrial soil invertebrates up through top predators).

[17] In 2007 manufactured soil sold for approximately \$21.50 yd^{-3} (cost of product and delivery), which would be about \$22,800 A^{-1} for a 20 cm-deep layer (Kurtz Bros., Inc. 2007).

[18] See previous footnote.

- The exposure scenarios focus on sensitive populations with respect to behaviors that tend to increase exposures. For example, the home gardener scenario represents adults and children that will have a relatively high level of direct contact (e.g., incidental soil ingestion) and indirect contact (e.g., ingestion of home grown produce) when compared to other populations.
- For carcinogenic (i.e., cancer-causing) constituents, the target cancer risk was defined as an excess lifetime cancer risk of 1 chance in 100,000 (i.e., 1E-05).
- For constituents that cause noncancer health effects, the target hazard level was defined as a ratio of predicted intake levels to safe intake levels—the HQ—of 1.
- The Phase II modeling (explained further in **Section 3.2.2**, below) used the upper end of the exposure concentration distribution (i.e., groundwater screening modeling used the 90th percentile receptor well concentration, and refined surface and groundwater modeling used the 90th percentile of the exposure distribution) rather than a central tendency measure.
- Exposure assumptions used in the risk modeling were designed to overestimate, rather than underestimate, potential exposures. For example, the exposure estimates from ingestion of home-grown produce assumed that the receptor consumes a very large amount of produce because the total produce diet is the sum of multiple produce categories (e.g., root vegetables, leafy greens). This implies that (1) all of these categories can be grown in the 0.1 acre garden in the same season, (2) all of these categories are consumed at relatively high rates, and (3) all these categories are consumed year round.
- For effects to ecological receptors (e.g., plants, animals, soil invertebrates), conservative environmental quality criteria (i.e. Eco-SSLs – see section 4.4.3 for more on the conservative nature of these screening levels) were used to define the target levels.
- The home garden was accessible to all residents, including children at all times; and
- The addition of SFS-manufactured soil (containing SFS at 50% of the soil dry weight) to the home garden essentially replaced the existing top 20-cm layer of local soil.

3.2 Analysis Plan

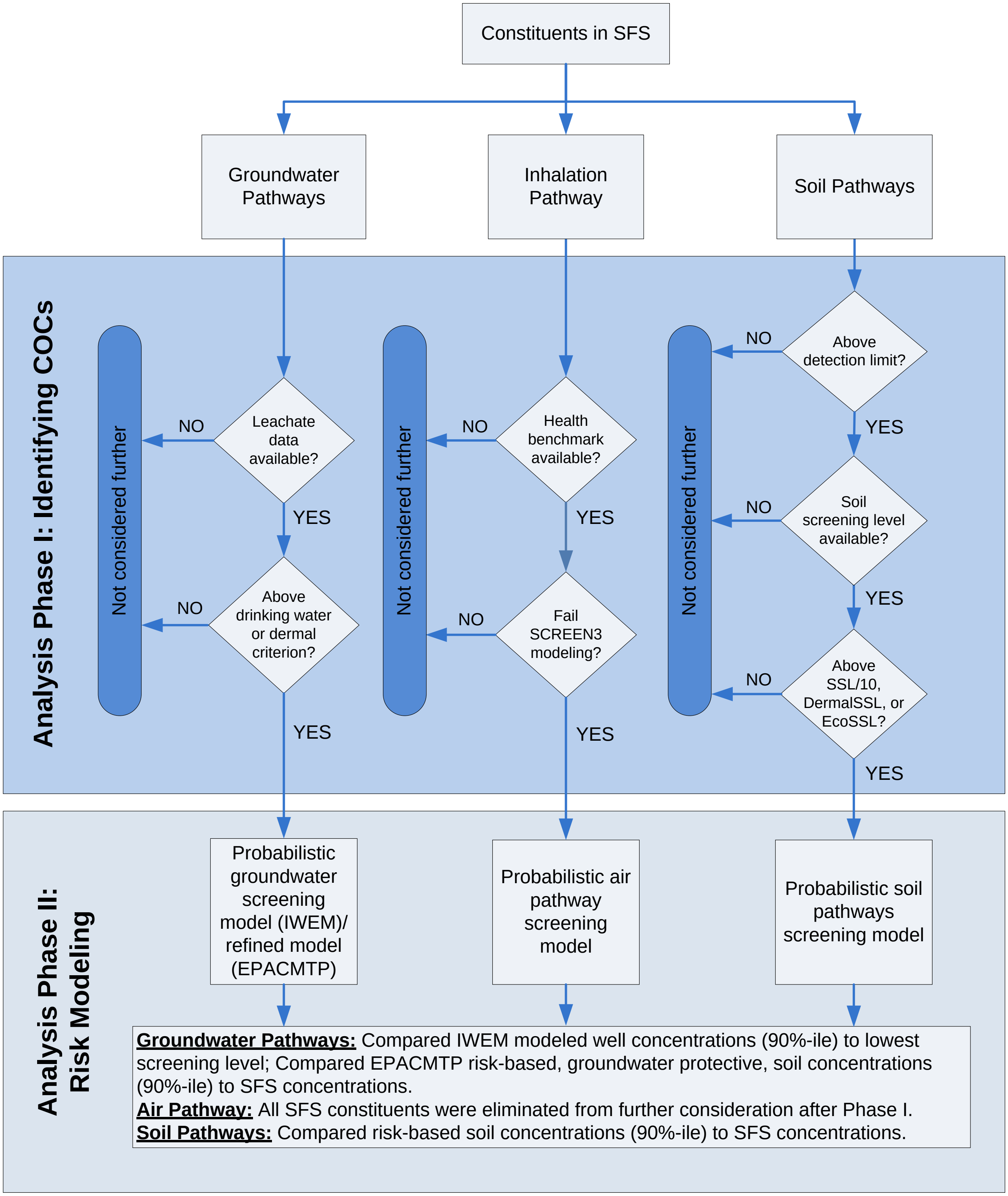
The analysis plan presents the overall approach used to (1) identify which, if any, SFS constituents have the potential to cause adverse health and environmental effects, and (2) model those constituent in the scenarios described in Section 3.1 associated with the greatest potential for exposure to SFS constituents.

Of the exposure scenarios described in **Section 3.1**, it was judged that the home garden scenario involved the greatest potential for exposure to SFS constituents. If risks from the use of SFS-manufactured soil in home gardens was below levels of concern for human health and ecological receptors, then risks from the other uses of SFS addressed by this assessment (i.e., soil-less potting media and road subbase) would also be below levels of concern. The exposure pathways evaluated included in the home garden scenario are (1) the ingestion of and dermal exposure to groundwater contaminated by SFS constituents leaching from SFS-manufactured soil in a home garden; (2) the inhalation of SFS emitted from soil-blending operations; and (3) the incidental ingestion and dermal exposure to SFS-manufactured soil, as well as ingestion of fruits and vegetables grown in SFS-manufactured soil.

As illustrated in **Figure 3-4**, the analysis plan involved a two-phase process for (1) identifying the COCs using a lines-of-evidence approach for the groundwater, inhalation, and soil pathways; and (2) conducting probabilistic risk modeling of any COCs identified for further study. Information gathered in Phase I, as well as the risk modeling results, represent lines of evidence. The risk characterization, presented in **Chapter 6**, integrates these lines of evidence with the substantial body of scientific research on SFSs presented in **Chapter 2** to develop a complete picture of the potential for adverse effects to both human and ecological receptors.

3.2.1 Analysis Phase I: Identifying Constituents of Concern

As illustrated in **Figure 3-4**, Phase I of the analysis was designed to identify the universe of SFS constituents needing more refined study; the COCs. This initial step included a review and synthesis of a wide variety of information on the types of SFS, production processes, properties of constituents in SFS (e.g., total constituent concentrations, leach test data), toxicological studies, and relevant soil science on the uptake and accumulation of chemicals (particularly metals) in plants and animals. Under Phase I, SFS constituents that met relevant pathway-specific screening criteria would need no further evaluation. SFS constituents that did not meet relevant pathway-specific screening criteria, however, would be evaluated further under Phase II.



Groundwater Pathways: Compared IWEM modeled well concentrations (90%-ile) to lowest screening level; Compared EPACMTP risk-based, groundwater protective, soil concentrations (90%-ile) to SFS concentrations.
Air Pathway: All SFS constituents were eliminated from further consideration after Phase I.
Soil Pathways: Compared risk-based soil concentrations (90%-ile) to SFS concentrations.

Figure 3-4. Analysis Plan for the risk assessment of SFS uses in soil-related applications.

Analysis Phase I: Groundwater Pathway

In the home gardening scenario, the only source of drinking water for the home was a well located directly downgradient from the garden. As shown in Figure 3-4, a two-step process was used to identify which SFS constituents, if any, would require further evaluation for the groundwater pathway.

- **Step 1: Leachability of constituents.** Leachability was evaluated based on the availability of leachate data obtained using appropriate test methods (i.e., SPLP or ASTM). If a constituent was not detected in any samples, the constituent was removed from further evaluation.
- **Step 2: Comparison to Drinking water or Dermal criteria.** SFS leachate data were compared directly (i.e. undiluted) to the EPA water quality criteria, including Regional Tapwater Screening Levels, Maximum Contaminant Levels (MCLs), and National Secondary Drinking Water Standards (NSDWS). Water dermal exposure was evaluated by comparing dermal absorbed doses to dermal benchmarks (i.e., oral benchmarks that were adjusted using EPA gastrointestinal absorption factors). If a constituent concentration was at or below the various drinking water criteria and the dermal absorbed dose was at or below the dermal benchmark, the constituent was removed from further evaluation.

COCs that were not removed through this initial two-step screen would be modeled under Phase II of the analysis. A detailed description of the groundwater pathway analysis, including inputs and results, is found in Chapter 4, Section 4.2.

Analysis Phase I: Inhalation Pathway

In the inhalation pathway, a resident living immediately downwind of a soil-blending operation (either at the use site, or a commercial blending operation) was exposed to fugitive dust released via windblown emissions from a storage pile, as well as emissions that occur as the result of loading/unloading operations. As shown in Figure 3-4, a two-step process was used to identify which SFS constituents, if any, would require further evaluation for the inhalation pathway.

- **Step 1: Availability of health benchmarks.** The availability of inhalation benchmarks was determined based on the Office of Solid Waste and Emergency Response (OSWER) toxicity value hierarchy (USEPA, 2003a). Because benchmarks are required for the quantitative evaluation of health effects, those constituents lacking inhalation benchmarks were removed from further inhalation evaluation.
- **Step 2: SCREEN3 Modeling.** SCREEN3 was used to estimate constituent-specific air concentrations associated with loading/unloading activities and windblown emissions. These modeled air concentrations were used to calculate the allowable concentration for each constituent in SFS based on potential risk

OSWER Benchmark Hierarchy

1. EPA's Integrated Risk Information System (IRIS; U.S. EPA, 2012)
2. EPA's Provisional Peer-Reviewed Toxicity Values (PPRTVs)
3. Other toxicity values (e.g., California Environmental Protection Agency [CalEPA] chronic inhalation Reference Exposure Levels [RELs] and cancer potency factors [CalEPA, 2005]; Agency for Toxic Substances and Disease Registry [ATSDR] Minimum Risk Levels [MRLs]).

via the inhalation pathway. The allowable concentration of each constituent in SFS—the health-based screening level for SFS—was intended to be protective of human health for the inhalation pathway. If a constituent concentration was at or below the allowable concentration in SFS, the constituent was removed from further evaluation.

COCs not removed based on available inhalation benchmarks and the SCREEN3 simulation would be modeled under Phase II of the analysis. A detailed description of the Phase I inhalation pathway analysis, including inputs and results, is found in Chapter 4, Section 4.3.

Analysis Phase I: Soil Pathway

In the home gardening scenario described in Section 3.1.4 and illustrated in Figure 3-3, home gardeners (adults or children) could be exposed via two direct pathways and five indirect pathways. Direct pathways included incidental ingestion and dermal contact with soil, and indirect pathways included ingestion of exposed fruits (e.g., strawberries), protected fruits (e.g., oranges), exposed vegetables (e.g., lettuce), protected vegetables (e.g., corn), and root vegetables (e.g., carrots). The home garden was assumed to supply a significant fraction of the home gardener’s produce diet. As shown in Figure 3-4, a three step process was used to identify SFS constituents that may pose risk above levels of concern for the soil pathways.

- **Step 1: Samples above detection limit.** As discussed in **Chapter 2**, numerous SFS samples were collected and analyzed. Analytes not identified in any sample were not evaluated further.
- **Step 2: Availability of Soil Screening Levels.** EPA’s Soil Screening Levels (SSLs) for soil ingestion were available for a large number of SFS constituents. Constituents with soil ingestion SSLs have EPA-approved ingestion benchmarks; therefore, those constituents lacking SSLs, and lacking health benchmarks with which to derive SSLs, were not evaluated further.
- **Step 3: Soil SSL Comparison.** For manufactured soils, concentrations of SFS constituents remaining after Step 2 were compared to human and ecological SSLs. The human health SSL was divided by a factor of 10 to account for Home Gardener indirect exposure pathways (i.e., ingestion of home-grown produce) not already accounted for in the SSL. If the constituent concentration was at or below the Adjusted SSL, Dermal-SSL, and Eco-SSL, then the constituent was not evaluated further.

Detected COCs not removed based on soil screening levels would be modeled under Phase II of the analysis. A more detailed description of the Phase I soil pathway analysis, including inputs and results, is found in Chapter 4, Section 4.4.

3.2.2 Analysis Phase II: Risk Modeling

A national-scale evaluation needs to account for variability in conditions across the country. The Phase II evaluation of SFS constituents used probabilistic modeling to account for national-scale variability. Specifically, Phase II used a Monte Carlo approach to probabilistically model site-specific conditions across the country. Monte Carlo simulation techniques are useful when there is substantial variability in the data and probability distributions[19] can be developed

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[19] A probability distribution for a parameter describes both the range of possible values *and* the likelihood of where in the possible range any single value will be.

for most or all of the input parameters. The Monte Carlo approach essentially performs a series of many site-specific evaluations of randomly selected locations, using input parameter values appropriate for each location. Taken together, the results of these many individual evaluations comprise a distribution of results from across the country. This approach is particularly appropriate for a risk analysis of soil-blending operations and home gardens that can be located across a wide range of environmental conditions.

The Phase II modeling methodology for each pathway is briefly described below. Chapter 5 includes additional introductory information on probabilistic modeling in general, as well as more detailed descriptions of how it was applied to evaluate the home garden scenario.

Groundwater Pathway: EPA’s Industrial Waste Management Model V2.0 (IWEM) and EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) were used to evaluate risks from exposure to groundwater. Both models have undergone external peer review, including the EPACMTP model having been subjected to peer review by the Science Advisory Board (SAB). Modeling performed with each of these models is described below.

Screening Modeling

IWEM provides a flexible basis for considering the potential leaching from SFS in manufactured soils. Detailed information on this model can be found in the IWEM User’s Guide (U.S. EPA, 2002a) and Technical Background Document (U.S. EPA, 2002b).[20] Some modeling input parameter values (e.g., distance from the garden to the drinking water well) were chosen to be conservative (i.e., protective of human health). When data were available, values for other input parameters (e.g., depth to the water table) were chosen from distributions representing variable conditions across the country. The remaining parameters used default values provided in the IWEM User’s Guide (U.S. EPA, 2002a).

Probabilistic modeling calculated groundwater concentrations at a hypothetical receptor well located from 1 to 200 m from the edge of the garden. Using the 95th percentile SFS leachate concentration for each of the COCs,[21] the model estimated groundwater concentrations at the receptor well. The model ran each leachate concentration 10,000 times, varying site conditions based on user inputs. The 90th percentile groundwater well concentration for each constituent was selected from the output distributions. Each constituent-specific concentration was then compared to the lowest of the health benchmarks collected during Phase I (e.g., drinking water MCLs). If the 90th percentile concentration estimate was at or below the benchmark, the leachate concentration was considered protective.

If the 90th percentile concentration estimate from the IWEM model was above the benchmark, more refined probabilistic groundwater modeling was performed using EPACMTP and source model leachate concentrations.

Refined Modeling

Consistent with other EPA national-scale groundwater modeling assessments, probabilistic groundwater modeling was performed using EPACMTP (U.S. EPA, 2003f,g,h;

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[20] Supporting documentation for IWEM, IWAIR, and EPACMTP can be found http://www.epa.gov/waste/nonhaz/industrial/tools/index.htm

[21] This analysis used the higher of the 95th percentile leachate concentrations found by either SPLP or the ASTM leachate methods.

1997a). As described in **Section 5.3,** the refined groundwater modeling was performed concurrently with the probabilistic modeling of the soil pathways by using the source model outputs (i.e., garden leachate fluxes and annual average leachate infiltration rates) as EPACMTP model inputs. Coupling the groundwater and surface pathways in this way both addressed environmental variability (e.g., local meteorological patterns, soil types) and ensured that the groundwater pathway and surface pathway exposure estimates were based on the same environmental conditions. Refined groundwater modeling placed the drinking water receptor well 1 m from the edge of the garden in the centerline of the plume.

The probabilistic simulation produced distributions of risk for the adult and child receptors, which reflect the variability in environmental setting. As described in **Chapter 5**, these distributions were subsequently used to estimate protective target SFS concentrations based on EPA's risk management criteria (e.g., HQ of 1). These target SFS concentrations represent conservative estimates which, if the SFS were a component of manufactured soil, would result in exposures (and risk) via groundwater pathway below the risk management criteria. A SFS constituent concentration at or below the target concentration would be considered protective. Please note that although the groundwater and soil pathways were evaluated concurrently, separate target SFS concentrations were developed for each pathway based on analyses discussed in **Section 5.3.5** and **Appendix J** that indicate that these exposures will not occur within the same timeframe.

A more detailed description of the Phase II groundwater pathway analysis is found in Chapter 5, Sections 5.2 and 5.3.

Inhalation Pathway: The Phase I analysis found that no constituents required further evaluation, and therefore no Phase II inhalation modeling took place. However, for completeness, a description of the Phase II inhalation modeling methodology is included below.

EPA's Industrial Waste Air Model (IWAIR) would have been used to evaluate risks from inhalation. IWAIR was developed to assist facility managers and regulatory agency staff in evaluating inhalation risks for workers and residents in the vicinity of a management unit. Detailed information on this model can be found in the IWAIR User's Guide (U.S. EPA, 2002c) and Technical Background Document (U.S. EPA, 2002d). With a limited amount of blending site-specific information (e.g., pile surface area and height, and constituent-specific emission rates), IWAIR can estimate whether temporary storage piles of SFS and SFS-manufactured soils might pose an unacceptable inhalation risk to human health. IWAIR default dispersion factors address variability in environmental settings across the country. These dispersion factors were developed based on dispersion modeling with the EPA's Industrial Source Complex – Short Term (ISCST3). Modeling was performed for many separate scenarios designed to cover a broad range of unit characteristics, including a range of storage pile surface areas and heights, 6 receptor distances from the unit and 60 meteorological stations, chosen to represent the different climatic and geographical regions of the contiguous 48 states, Hawaii, Puerto Rico, and parts of Alaska. The model would have been run thousands of times based on user inputs. The 90th percentile air concentration for each constituent would be compared to human health benchmarks identified under Phase I. If the 90th percentile concentration estimate was at or below the benchmark value, the SFS concentration would be considered protective.

Soil Pathway: The home-gardener scenario assumed that a raised garden received a single "addition" of SFS-manufactured soil containing 50% SFS by weight,[22] to a depth of 20 cm (a typical tilling depth). Based on this composition, it was further assumed that the basic properties and characteristics of the manufactured soil were similar to those of natural soil in the area.

The risk modeling framework currently used by EPA to support the Part 503 biosolids program was modified and adopted to evaluate soil pathway risks. This framework represents variability in soil and meteorological conditions in areas that produce SFS, as well as variability in consumption rates for fresh fruits and vegetables that are home grown. This risk modeling framework was adapted to capture variability in environmental settings within the context of "economic feasibility areas" for the use of SFS, defined as areas within 50 km of the foundry.[23] Locations within these areas were selected at random; no locations outside of the economic feasibility areas were included in the Monte Carlo simulations. The assumed application site and rates were also modified from the Biosolids framework to reflect home gardening practices rather than farming practices.

The probabilistic simulation produced distributions of risk/hazard for the adult and child receptors, as well as for plants, soil invertebrates and small mammals, which reflect the variability in conditions within the economic feasibility areas. As described in **Chapter 5**, these distributions (and the groundwater pathway distributions discussed above) were developed using an initial "unitized" soil concentration of 1 part per million (ppm) for each constituent. Based on the model's linearity with respect to constituent concentration, the 90th percentile of each constituent-specific unitized risk estimate was scaled to estimate a protective SFS-specific screening level based on EPA's risk management criteria (e.g., HQ of 1). These SFS-specific screening levels represent conservative estimates of the selected SFS constituent concentrations which, if the SFS were used in manufactured soil, would be protective of human health and the environment. An SFS constituent concentration at or below the target SFS screening level would be considered protective.

A more detailed description of the Phase II soil pathway analysis is found in Chapter 5, Section 5.3.

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[22] This is a conservative blend, as most manufactured soil blends would contain 5–10% SFS by weight. See **Chapter 2** for more details on soil blend recipes.

[23] SFS use areas are based on the ZIP codes of the membership of the American Foundry Society as of November 2007. Since we did not know a foundry's exact location within its ZIP Code area, we extended the ZIP Code boundary out 50 km to establish the economic feasibility areas.

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4. Analysis Phase I: Identification of COCs for Modeling

Chapter 3 described the three beneficial use scenarios for SFS under consideration in this assessment, and presented conceptual models for the exposure pathways and receptors for using SFS in roadway subbase, in blending operations, and in home gardening. As shown by the conceptual models, the home gardener scenario includes all of the exposure pathways in the other two scenarios. However, the blending scenario represents the highest potential inhalation exposure among any of the three scenarios. Therefore, the assessment used the home gardening scenario and the blending scenario to represent the exposure pathways that are most likely to present an unacceptable risk to human health and the environment. By focusing attention on the exposure pathways associated with manufactured soils that are potentially of greatest concern, the assessment could confidently identify the COCs (Analysis Phase I) and model only those COCs that might pose unacceptable risks to human health and the environment (Analysis Phase II). This chapter describes the process used to select COCs for further modeling evaluation and, by default, determine whether the exposure pathways are of concern.

4.1 Purpose

The primary purpose of the first phase of the analysis was to identify COCs for additional analysis in the risk modeling phase. If all constituents screened out for a particular exposure pathway, the potential risks for that pathway would no longer need to be evaluated using probabilistic risk models. Because this phase was designed to perform a screening function, a very conservative approach was used to ensure that an ample margin of safety was applied before eliminating a constituent from further consideration. For example, leachate concentrations were compared directly with EPA screening criteria for the protection of drinking water; this assumes that there would be no attenuation or dilution of the leachate and no degradation of organic compounds as they move through the subsurface to the drinking water well. Importantly, the following pathway-specific high-end concentrations provided the basis for the various Phase I analyses performed as described in this section:

- Groundwater pathway: 95th percentile leachate concentrations;
- Inhalation pathway: 95th percentile SFS constituent concentrations;
- Soil pathway: Manufactured soil concentrations ($Conc_{MS}$) reflecting a soil/SFS mixture that contained SFS with 95th percentile constituent concentrations.

As seen in the conceptual models for SFS-manufactured soils (see Figures 3-2 and 3-3), there are three basic media-specific exposure pathways to be evaluated: (1) *groundwater pathway* - the ingestion of, and dermal contact with, groundwater contaminated by the leaching of SFS constituents; (2) *ambient air pathway* - the inhalation of SFS emitted from soil blending operations; and (3) *soil pathway* - dermal contact with, and incidental ingestion of soil, as well as ingestion of fruits and vegetables grown in the SFS-manufactured soil. Although some constituents, such as manganese elicit similar toxicological responses (e.g., neurotoxicity) via different exposure pathways, neither the screening nor the modeling stages of the analysis considered cumulative exposures across these three pathways. Rather, the exposure scenarios and pathway evaluations were developed and parameterized to produce conservative risk estimates. The risk assessment is therefore an efficient approach to providing decision makers with information on the potential for adverse effects to the most highly exposed individuals and ecological receptors that could come in contact with SFS constituents.

4.2 Groundwater Exposure

Given the use of SFS-manufactured soil in a home garden, leaching to groundwater is a potential pathway of concern. Under this pathway, residents could be exposed to SFS constituents through the ingestion of contaminated drinking water or through dermal contact while bathing. Thus, this section: (1) examines the potential for SFS to leach constituents of potential concern; (2) evaluates drinking water ingestion exposure by comparing leachate data to regulatory levels and screening criteria developed to protect water use; and (3) evaluates water dermal exposure by comparing dermal absorbed doses to oral benchmarks adjusted using EPA gastrointestinal absorption factors. If a constituent concentration exceeded one of the drinking water criteria or if a dermal absorbed dose exceeded the adjusted oral benchmark, the constituent was flagged for further evaluation under Phase II.

4.2.1 Leachate Data

The first step in the groundwater analysis was to examine the leachability of SFS constituents. As discussed in **Chapter 2**, Dungan and Dees (2009) used the TCLP, SPLP and ASTM methods to estimate the leaching potential of metals from ferrous and aluminum foundry SFSs. The TCLP method, however, was designed to predict leaching potential under conditions very different from SFS use in manufactured soil or other soil-related applications (see Chapter 2, Section 2.5.4 for a more detailed discussion of the relevance of TCLP data to SFS soil-related applications). Therefore, the conditions reproduced by TCLP are not relevant to the SFS uses evaluated in this assessment.

The SPLP method was designed to mimic leaching from soil due to acid rain conditions, and the ASTM method tests leaching potential at a material's "natural" pH. The conditions reproduced by the SPLP and ASTM methods are more relevant than TCLP for characterizing SFS leaching potential under the conditions evaluated in this report. This part of the evaluation therefore only used SPLP or ASTM leach data.

Table 4-1 presents a summary of the SPLP and ASTM leachate data for the 39 silica-based iron, steel, and aluminum SFSs.

Table 4-1. Leaching Data for Silica-based Iron, Steel, and Aluminum SFSs (mg L^{-1})

	SPLP			ASTM		
Metal	**Maximum**	**95%-ile**	**Median**	**Maximum**	**95%-ile**	**Median**
Ag	<0.08	N/A	N/A	<0.05	N/A	N/A
As	0.098	0.017	0.004	0.024	0.018	0.005
Ba	0.63	0.37	0.12	<0.24	N/A	N/A
Be	<0.02	N/A	N/A	<0.01	N/A	N/A
Cd	<0.01	N/A	N/A	<0.01	N/A	N/A
Cr (III)	<0.01	N/A	N/A	<0.02	N/A	N/A
Cu	<0.21	N/A	N/A	1.1	0.04	0.04
Ni	0.24	0.025	0.025	0.046	0.025	0.025
Pb	<0.08	N/A	N/A	<0.11	N/A	N/A
Sb	<0.04	N/A	N/A	<0.04	N/A	N/A
Zn	<0.18	N/A	N/A	<0.22	N/A	N/A

Data from Dungan (2008) and Dungan and Dees (2009), all three sampling events of 39 silica-based iron, steel, and aluminum SFSs.

4.2.2 Selection of Constituents

Because leachate data for only 11 constituents (i.e., antimony, arsenic, barium, beryllium, cadmium, chromium, copper, lead, nickel, silver, and zinc) are available from Dungan and Dees (2009), these were the constituents of potential concern that were evaluated. A limitation of this data set is that for some constituents, the analytical detection limits were higher than the screening levels (or regulatory levels) to which they were being compared. In addition, this leachate analysis did not include mercury and selenium. Therefore, mercury and selenium were not evaluated quantitatively. However, the leaching potential of mercury and selenium from SFSs is discussed below.

4.2.3 Comparisons to Screening Levels and Regulatory Levels

To evaluate drinking water ingestion exposures, several risk levels were available for comparison to SFS leachate data. EPA's Superfund program developed Tapwater Screening Levels to be protective at 1E-06 cancer level[24] and an HQ of 1 for noncancer risk levels. EPA has also developed National Drinking Water Regulations. These include primary standards such as Maximum Contaminant Limits (MCLs), as well as secondary standards. **Table 4-2** provides the comparison of SFS leachate concentrations to all three screening and regulatory levels.

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[24] This cancer risk target is an order of magnitude lower than the risk target level that the EPA Office of Resource Conservation and Recovery typically uses in risk assessments. As mentioned elsewhere in this report, this evaluation used a risk target of 1E-05 for cancer.

Table 4-2. Leachate Comparisons (mg L^{-1})

Metal	SFS 95%-ile [a]		Screening and Regulatory Levels		
	SPLP	ASTM	Tapwater SL[b]	MCL[c]	NSDWS[d]
Ag	<0.08	<0.05	0.094	N/A	0.1
As	0.017	0.018	0.00052[e]	0.01	N/A
Ba	0.37	<0.24	3.8	2.0	N/A
Be	<0.02	<0.01	0.025	0.004	N/A
Cd	<0.01	<0.01	0.0092	0.005	N/A
Cr (III)	<0.01	<0.02	22	0.1[f]	N/A
Cu	<0.21	0.040	0.8	1.3	1.0
Ni	0.025	0.025	0.39	N/A	N/A
Pb	<0.08	<0.11	N/A	0.015	N/A
Sb	<0.04	<0.04	0.0078	0.006	N/A
Zn	<0.18	<0.22	6.0	N/A	5.0

[a] Data from Table 4-1

[b] Tapwater Screening Levels can be found at http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm

[c] MCLs are primary drinking water standards that can be found at http://water.epa.gov/drink/contaminants/index.cfm#Primary

[d] NSDWSs can be found at http://water.epa.gov/drink/contaminants/index.cfm#Secondary

[e] To be consistent with other ORCR risk assessments, the listed Tapwater SL for arsenic represents the Regional Tapwater SL converted to a 10^{-5} risk level

[f] Based on total Cr

To examine the potential for groundwater dermal exposure, the evaluation performed a screening level dermal assessment based on guidance provided in EPA's *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment)* (U.S. EPA; 2004). The assessment evaluated the SFS COCs identified in Section 4.2.2 using a three step process:

1. Identify COCs for quantitative analysis: Constituents for quantitative analysis were identified using the RAGs Part E Screening Tables, which flag chemicals where the dermal pathway has been estimated to contribute more than 10% of the oral pathway, using conservative residential exposure criteria. The screening tables reflect the comparison of two main household daily uses of water: as a source for drinking and for showering or bathing. This step determined that beryllium, cadmium, chromium (III), and zinc should be quantitatively evaluated for dermal exposure.[25]
2. Calculate dermal absorbed dose (DAD): Adult and child-specific DADs were calculated for beryllium, cadmium, chromium(III), and zinc using the reasonable maximum exposure (RME) scenario for residential settings as defined in U.S. EPA (2004). For the home garden use of SFS-manufactured soil scenario, the evaluation assumed that the adult and child showered or bathed with groundwater concentrations equivalent to

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[25] Lead was not included in U.S. EPA (2004) and sufficient data were not available to quantitatively assess dermal exposures for this constituent. However, the U.S. EPA notes that cutaneous absorption is generally not a significant route of exposure for inorganic lead (http://www.epa.gov/superfund/lead/almfaq.htm#dermal).

leachate concentrations. As discussed in Section 4.2.1, leachate data were available from both the SPLP and ASTM leachate methods. In this assessment, the higher of the two leachate values were used to calculate the DADs. These calculations were performed using the *Inorganic Chemicals in Water* spreadsheet available from U.S. EPA (2004), and using exposure parameter values shown in **Table 4-3**.

Table 4-3. Recommended Dermal Exposure Parameters for RME Residential Scenario

Exposure Parameters	Showering/ Bathing	
Event Frequency (events d^{-1})	1	
Exposure Frequency (days yr^{-1})	350	
Averaging Time (d)	25,550	
	Adult	**Child**
Event Duration (h $event^{-1}$)	0.58	1.0
Exposure Duration (yr)	30	6
Skin Surface Area (cm^2)	18,000	6,600
Body Weight (kg)	70	15

3. Compare DAD to dermal criterion: The resulting DAD estimates were then used to calculate constituent-specific HQs. Methods for estimating dermal risk are based on absorbed dose – the fraction of administered dose that is absorbed into the body. However, oral benchmarks such as RfDs and Slope Factors are typically based on administered dose. Use of oral benchmarks to estimate dermal risk required the adjustment of oral benchmarks using gastrointestinal absorption factors (ABS_{GI}). In accordance with U.S. EPA (2004), the oral reference dose (RfD) for noncarcinogens was multiplied by the constituent-specific ABS_{GI} to estimate a reference dose based on absorbed dose (RfD_{ABS}). The DAD estimates were then divided by the RfD_{ABS}s to calculate the constituent-specific hazard quotients. As seen in **Table 4-4**, the dermal hazard quotients were all below a level of concern (i.e., HQ = 1).

Table 4-4. Comparison of Water Dermal Absorbed Doses (DADs) to Health Benchmarks

	SFS 95%-ile Concentration		**Benchmark**		**DAD**		**Dermal Hazard**	
Constituent	**SPLP (mg L^{-1})**	**ASTM (mg L^{-1})**	**Oral RfD (mg $kg^{-1}d^{-1}$)**	**RfD_{ABS}[a] (mg $kg^{-1}d^{-1}$)**	**Adult DAD (mg $kg^{-1}d^{-1}$)**	**Child DAD (mg $kg^{-1}d^{-1}$)**	**HQ Adult**	**HQ Child**
Be	<0.02	<0.01	2.0E-03	1.4E-05	1.2E-06	7.2E-07	8.6E-02	5.1E-02
Cd	<0.01	<0.01	5.0E-04[b]	1.3E-05	6.2E-07	3.6E-07	4.8E-02	2.8E-02
Cr (III)	<0.01	<0.02	1.5	2.0E-02	1.2E-06	7.2E-07	6.0E-05	3.6E-05
Zn	<0.18	<0.22	0.3	0.3	8.1E-06	4.8E-06	2.7E-05	1.6E-05

[a] U.S. EPA (2004) presents gastrointestinal absorption efficiencies for beryllium (0.7%), cadmium (2.5%), and chromium (III) (1.3%), and recommends an efficiency of 100% for zinc in the absence of a reported value.

[b] Oral RfD (water)

4.2.4 Results

Only samples of arsenic had detectable leachate levels that exceeded the screening or regulatory levels for drinking water. That is, using the SPLP and ASTM leachate methods, several SFSs exceeded the MCL for arsenic (0.01 mg L^{-1}). In addition, the detection limit for arsenic (0.001 mg L^{-1}) was above the Tapwater Screening Level (0.00045 mg L^{-1}).

Analyses for the remaining constituents showed no samples that exceeded the screening or regulatory levels for drinking water. However, while all leachate samples of antimony, beryllium, cadmium, and lead were below their respective detection limits, the detection limits were higher than their respective MCLs. The detection limit for antimony also exceeded its Tapwater Screening Level.

Results from the water dermal screening assessment indicated that none of the constituents needed to be further evaluated for groundwater dermal exposure. As seen in Table 4-4, the dermal hazard quotients were all below a level of concern (i.e., HQ = 1).

With respect to mercury and selenium leachate concentrations, they are also not expected to exceed their regulatory levels based on the following considerations. In a study conducted by Fahnline and Regan (1995), the maximum concentrations of mercury and selenium in TCLP extracts from 50 spent foundry molding sands (from foundries of unknown type) were <0.10 mg L^{-1} and <0.83 mg L^{-1}, respectively. These TCLP data are being used here because no SPLP or ASTM data are available. Also, the TCLP method is likely more aggressive than either the SPLP or ASTM method when testing SFS (see **Chapter 2, Section 2.5.4**, for TCLP, SPLP and ASTM leaching results), such that actual leachate concentrations are unlikely to be greater than those listed in Fahnline and Regan (1995). Also, with respect to selenium, even if one assumes complete leaching of all selenium in the 39 SFSs considered (see **Appendix B**), no sand would exceed the regulatory level of 1.0 mg L^{-1}.

Therefore, as a result of the high detection limits for some constituents, and the exceedances of arsenic described above, the following constituents were retained for Phase II risk modeling (see **Chapter 5**):

- Antimony
- Arsenic
- Beryllium
- Cadmium
- Lead.

All remaining constituents were screened out from the groundwater pathway and were not retained for Phase II modeling.

4.3 Inhalation Exposure

As discussed earlier, SFS can replace mined sand as a mineral component of manufactured soil. It is probable that during storage and mixing, some components of the SFS (e.g., clays) will be emitted into the air and migrate offsite as fugitive dust. Therefore, as shown in the blending site conceptual model (Figure 3-2), nearby residents could be exposed to SFS constituents through the inhalation of this fugitive dust. Manufactured soils can be blended at the site where they will be used, or at a separate commercial blending facility. Residents living near a commercial blending facility would likely be exposed to fugitive dust for longer durations (potentially years) than those living near a site where the soil was blended once and then applied to the land. Activities at a soil-blending facility are also likely to result in higher emission rates and higher potential exposure frequencies than would be expected from gardening activities. This assessment therefore evaluated residential inhalation exposures to fugitive emissions from a soil-blending facility.

4.3.1 Scenario

In this scenario SFS is loaded and unloaded from a storage pile at an active soil blending facility. Soil blending involves using construction equipment, such as a front-end loader, to combine large volumes of the various mineral and organic components. The blending site was assumed to blend SFS-manufactured soil year-round. Some of the information used to develop the exposure scenario was based on the only commercial soil blender that currently uses SFS in soil-blending operations (Bailey, 2007); specifically,

- The amount of SFS managed
- The size of the SFS storage pile
- The distance from the site to the nearest residence.

Within the soil-blending industry this facility is considered quite large. Use of information from this facility (e.g., size of the SFS storage pile) is therefore considered a conservative assumption.

4.3.2 Selection of Constituents of Potential Concern

Constituents were chosen to undergo screening based on the availability of human health benchmarks for inhalation. Because benchmarks are required for the quantitative evaluation of health effects, those without benchmarks were not evaluated here. Cancer and noncancer benchmarks were chosen based on the Office of Solid Waste and Emergency Response (OSWER) toxicity value hierarchy.[26] **Table 4-5** provides the health benchmarks used to calculate the screening criteria for inhalation. The benchmarks in Table 4-5 are based on chronic exposure, 24 h d^{-1}, 365 d yr^{-1}. All 14 of the SFS constituents with inhalation exposure benchmarks (listed in Table 4-5) were screened.

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[26] The hierarchy is listed in the 2003 OSWER Directive 9285.7-53. This directive can be found at http://www.epa.gov/oswer/riskassessment/pdf/hhmemo.pdf.

Table 4-5. Inhalation Human Health Benchmarks

Constituent	Concentration (mg m^{-3})	Non-cancer target organ/ toxicological endpoint
Carcinogenic		
Arsenic [a]	2E-06	- -
Benz[a]anthracene [h]	2E-04	- -
Benzo[b]fluoranthene [h]	2E-04	- -
Benzo[k]fluoranthene [h]	2E-04	- -
Benzo[a]pyrene [h]	2E-05	- -
Beryllium [a]	4E-06	- -
Cadmium [a]	6E-06	- -
Chrysene [h]	1.1E-05	- -
Dibenz[a,h]anthracene [h]	2E-05	- -
Indeno[1,2,3-cd]pyrene [h]	2E-04	- -
Naphthalene [c]	3E-03	- -
Pentachlorophenol	5E-03	- -
2,3,7,8-TCDD [d,f]	1E-09	- -
2,4,6-Trichlorophenol [a]	3E-02	- -
Noncarcinogenic		
Aluminum [b]	5E-03	Neurological
Barium [g]	5E-04	Fetotoxicity
Boron [g]	2E-02	Respiratory system
Cobalt [d]	1E-04	Respiratory system
Manganese [c]	5E-05	Impaired neurobehavioral function
2-Methylphenol [e]	6E-01	nervous system
3- and 4-Methylphenol [e]	6E-01	nervous system
Nickel [e]	5E-05	Respiratory system, hematologic system
Phenol [e]	2E-01	Liver, cardiovascular system, kidney, nervous system
Selenium [e]	2E-02	Liver, cardiovascular system, nervous system

[a] Source: IRIS – Air concentration that would elicit a carcinogenic risk estimate of 1E-05 (U.S. EPA, 2012a)

[b] Source: PPRTVs – RfC for chronic inhalation exposure (U.S. EPA, 2006)

[c] Source: IRIS – RfC (U.S. EPA, 2012a)

[d] Source: ATSDR – MRL (ATSDR, 2007)

[e] Source: CalEPA – REL (CalEPA, 2005)

[f] 2,3,7,8-TCDD is used as the benchmark for the toxicity equivalent of all dioxins, furans, and dioxin-like PCBs

[g] Source: Health Effects Summary Table (HEAST, U.S. EPA, 1997b)

[h] Source: CalEPA – Inhalation Unit Risk (CalEPA, 2009) used in the methodology for generating Regional Screening Levels (the User's Guide is available at http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/index.htm) to estimate an air concentration that would elicit a carcinogenic risk estimate of 1E-05

4.3.3 Deterministic Modeling

To perform a screening assessment for the inhalation pathway, it was necessary to determine whether residents living near the soil-blending site could be exposed via inhalation at levels above the benchmarks in Table 4-5. In general, air exposure for a particular constituent would be the concentration of that constituent in the fugitive dust multiplied by the concentration of fugitive dust in the air:

$$Exposure = [X] \times [FD] \times 10^{-6}$$

Where:

Exposure = Exposure to the constituent (mg m^{-3})
[X] = Concentration of the constituent in fugitive dust (mg kg^{-1})
[FD] = Concentration of fugitive dust in the air (mg m^{-3})
10^{-6} = Conversion factor from mg to kg (kg mg^{-1}).

The SCREEN3 model (U.S. EPA, 1995b) was used to estimate the concentration of fugitive dust in the air near a soil-blending site.[27] SCREEN3 (a screening version of ISC3) is a single source Gaussian plume model that provides maximum ground-level concentrations for point, area, flare, and volume sources. It was developed to provide an easy-to-use method of obtaining pollutant concentration estimates based on *Screening Procedures for Estimating the Air Quality Impact of Stationary Sources* (U.S. EPA, 1992b). SCREEN3 outputs were used in conjunction with the health benchmarks in Table 4-5 to calculate screening levels for each constituent, as follows:

$$SL = \frac{[HB]}{[FD]} \times 10^{6}$$

Where:

SL = Screening level (mg constituent kg^{-1} fugitive dust)
[HB] = Health benchmark (mg m^{-3})
[FD] = Concentration of fugitive dust in the air (mg m^{-3})
10^{6} = Conversion factor from mg to kg (mg kg^{-1}).

The inhalation pathway was evaluated by comparing the calculated screening level for each constituent to the 95th percentile concentration of the constituent in SFS. If the 95th percentile concentrations are less than the screening level concentrations, it is reasonable to assume that the inhalation pathway, when taken in isolation, does not pose risks requiring further analysis and modeling, for the following reasons:

- The health benchmarks used to calculate the screening level are based on the worst-case exposure duration and frequency of 24 h d^{-1}, 365 d yr^{-1}
- The health benchmarks are protective of the general population and sensitive subpopulations
- The SCREEN3 model was implemented based on guidance provided in Section 4.1.2 of the *Workbook of Screening Techniques for Assessing Impacts of Toxic Air Pollutants*

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[27] SCREEN3 is publicly available at http://www.epa.gov/scram001/dispersion_screening.htm.

(U.S. EPA, 1992a) for continuous fugitive/windblown dust emissions. Modeling options were selected to examine the full range of meteorological conditions and wind directions to ensure that the highest maximum concentrations were identified. Outputs from this model are short-term, maximum 1-hour air concentrations. These short-term concentrations were then combined with chronic health benchmarks to develop conservative screening levels. Input parameters for the model (described in the following subsections), including emission factors, were selected to increase potential exposure, and

- The 95th percentile concentrations of constituents in SFS were used instead of median concentrations.

4.3.3.1 Emission Factors

To model the concentration of the SFS in the air, it was necessary to estimate the emission rate for the SFS managed in the soil-blending scenario. Two emission factors were calculated and converted into emission rates: one for loading and unloading the sand onto and off of the storage pile, and the other for windblown emissions. The loading/unloading emission factor was based on AP-42 (Compilation of Air Pollutant Emission Factors) Section 13.2.4 “Aggregate Handling and Storage Piles” (U.S. EPA, 1995a):

$$E = k(0.0016)\frac{\left(\frac{U}{2.2}\right)^{1.3}}{\left(\frac{M}{2}\right)^{1.4}}$$

Where:

E = Emission factor (kg Mg^{-1})
k = Particle size multiplier (dimensionless)
U = Mean wind speed (m s^{-1})
M = Material moisture content (%).

Information from U.S. EPA (1995a) was used to determine the values for k and U. For k, 0.35 was chosen based on an aerodynamic particle size of <10 µm (i.e., clay- and silt-sized fractions). AP-42, Section 13.2.4, reports a range of wind speeds for calculating particulate emissions by batch or continuous drop operations as 0.6–6.7 (m s^{-1}), and 5.4 m s^{-1} was selected to serve as the high-end wind speed to be consistent with wind conditions used to calculate windblown particulate emissions from a storage pile. The material moisture content of 3% was based on Table 1 in *Foundry Sand Facts for Civil Engineers* (FIRST, 2004), assuming that the foundry sand contains some clay-sized particles. The calculated emission factor for loading/unloading was 1.02E-03 kg Mg^{-1}.

Approximately 86,450 tons (78,410 Mg) per year of SFS is used at the active soil-blending site described in this assessment (Bailey, 2007). Based on the mass of sand managed per year, the area of the storage pile (150 m^2), and the assumption that the sand is being loaded/unloaded 4 h d^{-1}, 260 d yr^{-1}, the calculated emission factor (1.02E-03 kg Mg^{-1}) was converted to an emission rate of 1.42E-04 g s^{-1} m^{-2}.

The windblown emission factor was calculated using the equation for "Continuous Fugitive/Windblown Dust Emissions" (U.S. EPA, 1992a):

$$E = 1.9\left(\frac{s}{1.5}\right)\frac{(365 - p)}{235}\left(\frac{w}{15}\right)$$

Where:

E = Emission factor (kg d^{-1} ha^{-1})
s = Material silt content (%)
p = Number of days per year with more than 25 mm of precipitation (dimensionless)
w = Percent of time wind speed exceeds 5.4 m s^{-1} (%).

The material silt content of 12% was based on particle size analysis of the 39 samples of silica-based SFS from iron, steel and aluminum foundries (see **Chapter 2 Section 2.5.1**, and **Appendix B Table B-25**). The default values in U.S. EPA (1992a) of 0 for p and 20% for w were used in calculating this emission factor. The result (31.5 kg d^{-1} ha^{-1}) was converted to g s^{-1} m^{-2}, with a final emission rate of 3.64E-05 g s^{-1} m^{-2}.

4.3.3.2 Other Input Parameters for SCREEN3

In addition to the emission rates, SCREEN3 also required the following input parameters:

- **Source Type:** An area source was chosen because the emissions would be coming off of a storage pile and not from a smokestack or other point source
- **Length, Width, and Height of Storage Pile:** 15 m, 10 m, and 4 m were chosen based on an aerial photograph of the only currently operating facility that uses foundry sand in soil blending operations (Bailey, 2007). Within the soil-blending industry this facility is considered quite large.
- **Receptor Height:** 0 m was chosen to be protective of a child or infant receptor close to the ground
- **Urban or Rural:** Rural was chosen because it is more conservative than the urban option and based on the location of the blending operation in the aerial photograph referenced above
- **Search for Maximum Direction:** A positive response was chosen as a conservative assumption so that the maximum air concentration would be located.

SCREEN3 requires the user to specify the modeling area, defined as the region between two distances from the source, within which to estimate maximum concentrations. For this study, the modeling area was defined as the region from 0 to 1,000 m from the source to ensure that the maximum concentration of airborne SFS would be included in the range. SCREEN3 gives the user the option to specify "discrete" distances, which are specific distances from the source at which to identify maximum concentrations. Because the distance to the nearest resident was estimated to be 500 m, based on the aerial photograph referenced above, SCREEN3 calculated the concentration 500 m away.[28] **Table 4-6** summarizes the SCREEN3 input parameters used.

Table 4-6. Input Parameters for SCREEN3

Parameter Description	Scenario	
	Loading and Unloading	**Windblown Erosion**
Source type	Area	Area
Emission rate (g s^{-1} m^{-2})	1.42E-04	3.64E-05
Height of storage pile (m)	4	4
Length of storage pile (m)	15	15
Width of storage pile (m)	10	10
Receptor height (m)	0	0
Urban or rural	Rural	Rural
Search for maximum direction	Yes	Yes
Choice of meteorology	Full	Full
Automated distance array	Yes	Yes
Minimum distance (m)	0	0
Maximum distance (m)	1,000	1,000
Use discrete distances	Yes	Yes
Distance (m)	500	500

4.3.3.3 SCREEN3 Outputs

Using the inputs listed in Table 4-6, SCREEN3 estimated the concentration of SFS in the air at ground level under both the loading/unloading and windblown erosion scenarios. **Table 4-7** shows both outputs from SCREEN3 at a distance of 500 m. In addition, the estimated concentrations for these two scenarios were summed to provide a total concentration that a receptor might be exposed to. This calculated total concentration was 49.7 µg m^{-3}.

Table 4-7. SCREEN3 Output Summary

Parameter Description	Scenario		
	Loading and Unloading	**Windblown Erosion**	**All Scenarios (Sum Total)**
Concentration at 500 m (µg m^{-3})	39.6	10.2	49.7

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[28] While the assumption of a 500 m distance to the nearest residence is based on empirical evidence, it may not be a conservative assumption. However, a preliminary analysis found that reducing the distance to 100 m would not change the Phase I results: all modeled constituents would pass the screen, and therefore no constituents would require Phase II evaluation.

4.3.4 Results

Neither the loading and unloading scenario nor the windblown erosion scenario estimated levels of particulates higher than the Primary National Ambient Air Quality Standard (NAAQS) for coarse inhalable particulates[29] (150 µg m^{-3}). The combined concentration of both scenarios also fell below the NAAQS for coarse inhalable particulates. However, even when the particulate levels do not exceed their primary air standard, it is still possible that one or more constituents in the fugitive dust could exceed chemical-specific, health-based target levels (see Table 4-5).

As described above, conservative screening concentrations were calculated for each of the constituents in Table 4-5 by dividing the health benchmarks by the total SFS air concentration listed in Table 4-7. Exposure was assumed to be at the total concentration 24 h d^{-1}, 365 d yr^{-1}. **Table 4-8** shows the actual 95th percentile concentrations of constituents in SFS and the calculated conservative screening concentrations for the inhalation pathway.

Table 4-8. Comparison to Screening Values: Inhalation Pathway

SFS Constituent [a, b]	SFS 95%-ile (mg kg^{-1})	Calculated Screening Concentration (mg kg^{-1})
Carcinogens		
Arsenic	6.44	40.2
Benz[a]anthracene	0.13	4,020
Benzo[b]fluoranthene	0.06 [c]	4,020
Benzo[k]fluoranthene	0.07 [c]	4,020
Benzo[a]pyrene	0.10 [c]	402
Beryllium	0.38	80.4
Cadmium	0.20	121
Chrysene	0.04	221
Dibenz[a,h]anthracene	0.08	402
Indeno[1,2,3-cd]pyrene	0.07 [c]	4,020
Naphthalene	3.45	60,300
Pentachlorophenol	0.12	100,500
2,3,7,8-TCDD TEQ [d]	3.13E-6	0.0201
2,4,6-Trichlorophenol	0.06	603,000
Noncarcinogens		
Aluminum	11,200	100,500
Barium	17.7	10,060

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[29] A standard for particulate matter with a mean aerodynamic diameter of 10 microns or less (PM_{10})

SFS Constituent [a, b]	**SFS 95%-ile (mg kg^{-1})**	**Calculated Screening Concentration (mg kg^{-1})**
Boron	20.2	402,000
Cobalt	5.99	2,010
Manganese	670	1,005
2-Methylphenol	8.74	Capped
3- and 4-Methylphenol	3.41	Capped
Nickel	102	1,005
Phenol	20.2	Capped
Selenium	0.20	402,000

[a] Constituent data from silica-based iron, steel, and aluminum sands (Dayton et al., 2010)

[b] PAH and phenolic data from Dungan (2006, 2008)

[c] Not detected in any samples. Value represents one half the detection limit

[d] Due to the small size of the dataset for dioxins and dioxin-like compounds, the maximum value for 2,3,7,8 TCDD TEQ was used, rather than the 95th percentile.

Capped = Screening modeling estimates indicated risks below levels of concern at concentrations above 1E06 mg kg^{-1} (i.e., SFS could be comprised entirely of this constituent and still not cause risk).

None of the constituent concentrations in SFS exceeded their respective screening levels. Therefore, no SFS constituents required further evaluation and Phase II risk modeling for the inhalation pathway was not performed.

4.4 Soil Pathways Exposure

When SFS-manufactured soil is used in a home garden, potential exposure pathways include incidental ingestion of soils, dermal contact with soils, and the ingestion of produce grown in the home garden. The three-step process used to identify COCs for the soil pathways included the following:

1. Remove SFS constituents that were not detected in any samples
2. Remove SFS constituents with no human health benchmarks
3. Remove SFS constituents by comparing the constituent concentrations to (a) adjusted SSLs for the ingestion pathways (use of adjusted SSLs is discussed in Section 4.4.3), (b) DermalSSLs for soil dermal exposure, and (c) Eco-SSLs.

Although Dungan and Dees (2009) examined total metals, data from Dayton et al. (2010) were used because their analytical methods had lower detection limits. Data from Dungan and Dees (2009) were used to screen PAHs and phenolics, and data from Dungan et al. (2009) were used to screen dioxins and dioxin-like compounds.

It is also important to note that different categories of semi-volatiles were handled differently. Specifically, PAHs were each dealt with individually, while dioxins and dioxin-like compounds were dealt with in terms of their toxic equivalence values (TEQs – which estimate toxicity relative to 2,3,7,8-TCDD). Evaluation of dioxins and dioxin-like compounds in terms of their TEQ is an accepted approach that the Agency often uses. Therefore, from this point forward all dioxin-like compounds will be represented by an aggregated toxicity equivalent, or 2,3,7,8-TCDD TEQ.

4.4.1 Remove SFS Constituents that are Nondetects

Although SFS samples were analyzed for numerous constituents of potential concern, not all analytes were necessarily detected in the samples. Therefore, constituents of potential concern that were not identified in any sample were not retained for further evaluation. **Table 4-9** lists all constituents of potential concern, identifying those that were not detected in any sample.

As shown in Table 4-9, all metals were detected in at least one sample, and were therefore retained for further screening. Of the PAHs, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[g,h,i]perylene, benzo[a]pyrene, and indeno[1,2,3-cd]pyrene were not detected in any of the samples and were dropped from further study. Most phenolics also were not detected in any of the samples and were also dropped from further study. Only 4-chloro-3-dinitrophenol, 2,4-dichlorophenol, 2,4-dinitrophenol, 2-methylphenol, 3- and 4-methylphenol, and phenol were detected in at least one sample, and were therefore retained for further screening. Finally, 1,2,3,7,8,9-HCDF was not detected in any of the samples, and was therefore dropped from further study; all other dioxins and dioxin-like compounds were retained for further screening.

Table 4-9. Constituents Detected in at Least One Sample

Constituent	Detect ✓=Yes ×=No	Constituent	Detect ✓=Yes ×=No	Constituent	Detect ✓=Yes ×=No
Al	✓	Sb	✓	2,4-Dimethylphenol	✓
As	✓	Se	✓	2,4-Dinitrophenol	×
B	✓	Tl	✓	Fluoranthene	✓
Ba	✓	V	✓	Fluorene	✓
Be	✓	Zn	✓	Indeno[1,2,3-cd]pyrene	×
Ca	✓	Acenaphthene	✓	2-Methylphenol	✓
Cd	✓	Acenaphthylene	✓	3- and 4-Methylphenol	✓
Co	✓	Anthracene	✓	2-Methyl-4,6-dinitrophenol	×
Cr	✓	Benz[a]anthracene	✓	Naphthalene	✓
Cu	✓	Benzo[b]fluoranthene	×	2-Nitrophenol	×
Fe	✓	Benzo[k]fluoranthene	×	4-Nitrophenol	×
K	✓	Benzo[g,h,i]perylene	×	Pentachlorophenol	×
Mg	✓	Benzo[a]pyrene	×	Phenanthrene	✓
Mn	✓	2-sec-Butyl-4,6-dinitrophenol	×	Phenol	✓
Mo	✓	Chrysene	✓	Pyrene	✓
Na	✓	4-Chloro-3-methylphenol	✓	2,3,7,8-TCDD TEQ	✓[a]
Ni	✓	2-Chlorophenol	×	2,3,4,6-Tetrachlorophenol	×
P	✓	Dibenz[a,h]anthracene	✓	2,4,6-Trichlorophenol	×
Pb	✓	2,4-Dichlorophenol	×	2,4,5-Trichlorophenol	×
S	✓	2,6-Dichlorophenol	×		

[a] All dioxin-like compounds except for 1,2,3,7,8,9-HxCDF were detected.

4.4.2 Remove SFS Constituents without Benchmarks

Health benchmarks are required to quantify potential health risks, and the screening criteria developed by EPA require an EPA-approved health benchmark. The SSLs developed by EPA (U.S. EPA, 1996) to be protective of the soil ingestion pathway are based on EPA-approved health benchmarks, as well as conservative exposure assumptions. **Table 4-10** lists SSLs for constituents of potential concern or indicates that no benchmark exists for generating SSLs.[30]

Of the constituents of potential concern remaining after the first step, there were no health benchmarks for calcium, magnesium, phosphorus, potassium, sodium, and sulfur, all six of which are also essential plant nutrients. Therefore, these constituents were removed from further quantitative evaluation. Eighteen metals, 9 PAHs, 20 dioxins and dioxin-like compounds, and 5 phenolics remained after the first two steps in the screening process for soil pathways.

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[30] SSLs are not national cleanup standards, nor do they define “unacceptable” levels of contaminants in soil. They were designed as tools for the Superfund program to quickly identify sites that no longer need federal attention. Because of this, soil concentrations above SSLs do not in and of themselves denote a problem, only that further study may be warranted. More information on SSLs can be found at http://rais.ornl.gov/calc_start.shtml.

Table 4-10. Residential Soil Screening Levels (mg kg^{-1})[a]

	Carcinogenic SSL [b]				**Noncarcinogenic SSL**			
	Pathways included in the Screening Level			**Residential SSL**	**Pathways included in the Screening Level**			**Residential SSL**
Analyte	**Ingestion**	**Dermal**	**Inhalation**		**Ingestion**	**Dermal**	**Inhalation**	
Aluminum				N/A	X		X	7.7E+04
Arsenic	X	X	X	6.7E+00	X	X	X	3.4E+01
Antimony				N/A	X			3.1E+01
Barium				N/A	X		X	1.5E+04
Beryllium			X	1.6E+03	X		X	1.6E+02
Boron				N/A	X		X	1.6E+04
Cadmium			X	2.1E+03	X	X	X	7.0E+01
Calcium	No Benchmark				No Benchmark			
Chromium(III)				N/A	X			1.2E+05
Cobalt			X	4.2E+02	X		X	2.3E+01
Copper				N/A	X			3.1E+03
Iron				N/A	X			5.5E+04
Lead [c]				N/A				4.0E+02
Magnesium	No Benchmark				No Benchmark			
Manganese				N/A	X		X	1.8E+03
Molybdenum				N/A	X			3.9E+02
Nickel			X	1.5E+04	X		X	1.5E+03
Phosphorus	No Benchmark				No Benchmark			
Potassium	No Benchmark				No Benchmark			
Selenium				N/A	X		X	3.9E+02
Sodium	No Benchmark				No Benchmark			
Sulfur	No Benchmark				No Benchmark			
Thallium				N/A	X			7.8E-01
Vanadium				N/A	X		X	3.9E+02
Zinc				N/A	X			2.3E+04
Acenaphthene				N/A	X	X		3.5E+03
Acenaphthylene	No Benchmark				No Benchmark			
Anthracene				N/A	X	X		1.7E+04
Benz[a]anthracene	X	X	X	1.5E-01				N/A
Chrysene	X	X	X	1.5E+01				N/A
4-Chloro-3-methylphenol				N/A	X	X		6.2E+03
Dibenz[a,h]anthracene	X	X	X	1.5E-02				N/A
2,4-Dimethylphenol				N/A	X	X		1.2E+03
Fluoranthene				N/A	X	X		2.3E+03
Fluorene				N/A	X	X		2.3E+03
2-Methylphenol				N/A	X	X	X	3.1E+03
3- and 4-Methylphenol				N/A	X	X	X	3.1E+03
Naphthalene			X	3.8E+00	X	X	X	1.3E+02
Phenanthrene	No Benchmark				No Benchmark			
Phenol				N/A	X	X	X	1.8E+04
Pyrene				N/A	X	X		1.7E+03
2,3,7,8 TCDD (TEQ)	X	X	X	4.9E-06	X	X	X	5.1E-05

N/A = Not Available

[a] SOURCE: EPA (2009)

[b] Cancer values are based on 10^{-5} risk level

[c] The health benchmark for lead was being revised while this evaluation was conducted.

4.4.3 Remove SFS Constituents by Comparing to SSLs and Eco-SSLs

The home gardener scenario represents a sensitive population because of the assumption that individuals live near their garden and grow food using SFS-manufactured soils. These individuals are potentially more exposed to SFS constituents than the general population. As shown in the conceptual model (see Figure 3-3), the exposure pathways of concern include the incidental ingestion of soil, dermal contact with soil, and the ingestion of exposed fruits (e.g., strawberries), protected fruits (e.g., oranges), exposed vegetables (e.g., lettuce), protected vegetables (e.g., corn), and root vegetables (e.g., carrots).

Comparing the soil concentrations to EPA's Residential SSLs is a common technique to identify COCs for exposure via soil ingestion, dermal exposure to soil, or inhalation of fugitive dust in residential (as opposed to industrial) exposure scenarios (U.S. EPA, 2002c). Residential SSLs are also available, on a constituent-specific basis, which address cumulative exposures from two or more of the above-referenced exposure pathways. Table 4-10 lists the exposure pathways addressed by the Residential SSLs for the remaining SFS constituents. Residential SSLs are screening values for soil, regardless of the source of the contamination; in addition, the Residential SSLs do not consider exposure via ingestion of produce grown on the soil. Therefore, knowing that Residential SSLs are conservative screening levels for soil ingestion (and in some instances dermal and inhalation exposures), the Residential SSLs were divided by a factor of 10 to account for indirect exposure associated with the ingestion of produce grown in SFS-manufactured soil. Work by U.S. EPA (1993) on biosolids strongly suggests that the soil ingestion pathway is the dominant exposure pathway when compared to the ingestion of plant or animal products grown on amended soil. Based on EPA's insights on biosolids-amended soil, the adjustment factor of 10 was used to provide a reasonably conservative adjustment to the Residential SSLs. Thus, this screening step was only satisfied if the blended soil concentration ($\mathrm{Conc}_{\mathrm{MS}}$) was below the Adjusted SSL (i.e., an order of magnitude below the respective Residential SSL). If the $\mathrm{Conc}_{\mathrm{MS}}$ for a constituent was below the Adjusted SSL, the constituent was removed from further evaluation of the soil pathways. Constituent concentrations in SFS-manufactured soil were calculated as follows:

$$Conc_{MS} = Conc_{FS} \times FracMS_{FS}$$

Where:

$\mathrm{Conc}_{\mathrm{MS}}$ = Concentration of the constituent in SFS-manufactured soil (mg kg^{-1})

$\mathrm{Conc}_{\mathrm{FS}}$ = 95th percentile constituent concentration in SFS (mg kg^{-1})

$\mathrm{FracMS}_{\mathrm{FS}}$ = SFS fraction of manufactured soil (dimensionless). Under this assessment, set to 0.5 representing 50% SFS.

This equation assumes that the SFS is the sole source of the constituent in the manufactured soil (i.e., background concentrations are not considered).[31]

As discussed above and listed in Table 4-10, many of the Residential SSLs used in the assessment address dermal exposure. However, to further evaluate direct dermal contact with

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[31] Failure to be screened out by this very conservative approach does not imply that the constituent presents a risk, but rather that for the purposes of this assessment, the constituent was included in a more refined evaluation discussed in **Chapter 5**.

SFS-manufactured soils, a screening assessment compared SFS-manufactured soil concentrations to dermal soil screening levels (dermal SSLs). For this conservative assessment, the SFS-manufactured soil concentrations were calculated based on the 95th percentile SFS concentrations shown in **Table 4-11**. The SFS-manufactured soil concentrations were then compared to U.S. EPA's standardized risk-based dermal SSLs to determine if the pathway should be further evaluated.

Dermal SSLs were obtained from the U.S. EPA's Mid Atlantic Risk Assessment website (U.S. EPA 2009). This website provides tables of screening levels for various exposure scenarios, including a residential soil scenario. The residential soil scenario table presents both the dermal screening levels and the toxicity values used in the derivation of these levels. Those COCs for which both noncancer (i.e., RfD) and cancer oral benchmarks (i.e., cancer slope factor, or CSF) were available, two dermal SSLs were provided, one for each endpoint. The noncancer SSL is based on a hazard quotient of 1 and the carcinogenic SSL is based on a cancer risk of 1E-05.[32] For those COCs with both noncancer and cancer risk-based SSLs, the SFS-manufactured soil concentration was compared to the lower of the two SSLs. The calculation of dermal SSLs also requires the input of a dermal absorption fraction from soils (ABS) and a gastrointestinal absorption factor (ABS_{GI}). The ABS factors are included in the soil dermal calculations to account for uncertainty due to different soil types and other variable conditions. The ABS_{GI} values are used to adjust the oral benchmarks which are usually based on administered dose and include GI absorption. Table 4-11 presents the dermal SSLs, the associated benchmarks, and ABS values. With the exception of cadmium, an ABS_{GI} factor of 1 (i.e., 100%) is applied for all of the COCs shown in this table. The ABS_{GI} value applied for cadmium was 0.025 or 2.5%, as recommended by U.S. EPA (2004).

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[32] The carcinogenic SSL presented in the screening level table was based on a cancer risk of 1E-06. For the current assessment, the carcinogenic SSLs were adjusted to reflect the established allowable cancer risk level of 1E-05.

Table 4-11. Comparison to Dermal Soil Screening Levels

Constituent	SFS 95%-ile [e] (mg kg^{-1})	Manuf. Soil Conc. (mg kg^{-1})	Human Health Benchmark RfD (mg kg^{-1} d^{-1}) or CSF (per mg kg^{-1} d^{-1})	Cited Ref [a]	ABS (unitless)	Dermal SSL [b] (mg kg^{-1})
Arsenic (As)	6.44	3.22	1.5E+00 (CSF)	I	0.03	51
Cadmium (Cd)	0.20	0.10	1.0E-3 (RfD)[f]	I	0.001	730
Acenaphthene	0.26	0.13	6.0E-02 (RfD)	I	0.13	13,000
Anthracene	0.87	0.44	3.0E-01 (RfD)	I	0.13	67,000
Benz[a]anthracene	0.13	0.07	7.3E-01 (CSF)	E	0.13	5.7
Chrysene	0.04	0.02	7.3E-03 (CSF)	E	0.13	570
Dibenz[a,h]anthracene	0.08	0.04	7.3E+00 (CSF)	E	0.13	0.57
2,4-Dimethylphenol	4.38	2.19	2.0E-02 (RfD)	I	0.1	5,800
Fluoranthene	0.18	0.09	4.0E-02 (RfD)	I	0.13	8,900
Fluorene	0.71	0.36	4.0E-02 (RfD)	I	0.13	8,900
2-Methylphenol[c]	8.74	4.37	5.0E-02 (RfD)	I	0.1	15,000
3- and 4-Methylphenol[d]	3.41	1.71	5.0E-02 (RfD)	I	0.1	15,000
Naphthalene	3.45	1.73	2.0E-02 (RfD)	I	0.13	4,500
Phenol	20.2	10.1	3.0E-01 (RfD)	I	0.1	87,000
Pyrene	0.47	0.24	3.0E-02 (RfD)	I	0.13	6,700
2,3,7,8-TCDD TEQ [e]	3.13E-06	1.57E-06	1.3E+05 (CSF)	Cal EPA	0.03	5.80E-04

I=IRIS; E = (EPA/ORD) Environmental Criteria and Assessment Office

[a] Reference: Cited in U.S. EPA Mid Atlantic Risk Assessment Generic Tables for Residential Soil Scenario.

[b] Dermal SSLs based on oral cancer slope factors (CSFs) reflect a cancer risk of 1E-05; noncancer SSLs based on RfDs reflect a hazard quotient of 1.

[c] Synonym: o-Cresol.

[d] RfD and Dermal SSL for 3-Methylphenol (m-Cresol) applied; IRIS reports RfD for 4-methylphenol (p-Cresol) withdrawn.

[e] Maximum concentration applied instead of 95th percentile due to small sample size.

[f] Oral RfD (food)

The ecological risk screening focused on receptors that are in direct contact with the SFS-manufactured soil, and the potential for food web exposures specific to the garden. To screen SFS constituents for potential ecological impacts, constituent concentrations in SFS-manufactured soil ($Conc_{MS}$) were compared to the Eco-SSLs for plants, soil invertebrates, or mammals,[33] whichever was lowest. **Table 4-12** shows the ecological screening criteria used in this assessment. Constituents with $Conc_{MS}$ levels below their respective Eco-SSL passed the screen, and therefore were removed from further evaluation.

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[33] Like their human toxicity counterparts, Eco-SSLs are very conservative screening values. Eco-SSLs were designed to overestimate potential impacts to ecological receptors. For example, the most bioavailable forms of a constituent are chosen to estimate exposure.

Table 4-12. Ecological Screening Criteria Used in the Analysis [a]

Constituent	Eco-SSL for Terrestrial Plants (mg kg^{-1} soil)	Eco-SSL for Soil Invertebrates (mg kg^{-1} soil)	Eco-SSL for Mammals (mg kg^{-1} soil)
As	18	—	46
Ba	—	330	2000
Be	—	40	21
Cd	32	140	0.36
Co	13	—	230
Cr(III)	—	—	34
Cu	70	80	49
Mn	220	450	4,000
Ni	38	280	130
Pb	120	1,700	56
Sb	—	78	0.27
Se	0.52	4.1	0.63
V	—	—	280
Zn	160	120	79
Low Molecular Weight PAHs [b] - Total	—	29	100
High Molecular Weight PAHs [c] - Total	—	18	1.1

[a] Eco-SSLs are available at http://www.epa.gov/ecotox/ecossl/

[b] PAHs composed of fewer than four condensed aromatic ring structures (EPA, 2007e)

[c] PAHs composed of four or more condensed aromatic ring structures (EPA, 2007e)

Table 4-13 compares the constituent concentrations in SFS-manufactured soil ($Conc_{MS}$) to human and ecological SSLs.

Table 4-13. Comparing SFS-manufactured Soil to Human and Ecological SSLs

(mg kg^{-1} unless otherwise noted)

Constituent	SFS 95%-ile	Conc_{MS}	Adjusted SSL	Dermal SSL	Passes the Human Health Screen? (✓= Yes)	Eco-SSL	Passes the Eco Screen? (✓= Yes)
Al (g kg^{-1})	11.2	5.60	7.7		✓		
As	6.44	3.22	0.67	51	No	18	✓
B	20.2	10.1	1,600		✓		
Ba	17.7	8.85	1,500		✓	330	✓
Be	0.38	0.19	16		✓	21	✓
Cd	0.20	0.10	7.0	730	✓	0.36	✓
Co	5.99	3.00	2.3		No	13	✓
Cr (III)	109	54.5	1.2E+04		✓	34	No
Cu	107	53.5	310		✓	49	No
Fe (g kg^{-1})	57.1	28.9	5.5		No		
Mn	670	335	1,800		✓	220	No
Mo	21.8	10.9	39		✓		
Ni	102	51.0	150		✓	38	No
Pb	15.3	7.65	40		✓	56	✓
Sb	1.23	0.62	3.1		✓	0.27	No
Se	0.20	0.10	39		✓	0.52	✓
Tl	0.09	0.05	0.078		✓		
V	9.90	4.95	39		✓	280	✓
Zn	72.1	36.1	2,300		✓	79	✓
Low Molecular Weight PAHs [a] - Total	7.59	3.79				29	✓
Acenaphthene	0.34	0.17	350	1.3E+04	✓		
Acenaphthylene	0.20	0.10					
Anthracene	0.88	0.44	1,700	6.7E+04	✓		
Fluorene	0.73	0.37	230	8,900	✓		
Naphthalene	3.89	1.94	3.8	4,500	✓		
Phenanthrene	1.56	0.78					
High Molecular Weight PAHs [a] - Total	0.95	0.48				1.1	✓
Benz[a]anthracene	0.14	0.07	0.15	5.7	✓		
Chrysene	0.04	0.02	1.5	570	✓		
Dibenz[a,h]anthracene	0.08	0.04	0.15	0.57	✓		
Fluoranthene	0.21	0.10	230	8,900	✓		
Pyrene	0.48	0.24	170	6,700	✓		

Constituent	SFS 95%-ile	Conc$_{MS}$	Adjusted SSL	Dermal SSL	Passes the Human Health Screen? (✓= Yes)	Eco-SSL	Passes the Eco Screen? (✓= Yes)
4-Chloro-3-methylphenol	0.09	0.05	620		✓		
2,4-Dimethylphenol	5.60	2.80	120	5,800	✓		
2-Methylphenol	8.76	4.38	310	1.5E+04	✓		
3- and 4-Methylphenol	3.59	1.79	310	1.5E+04	✓		
Phenol	22.1	11.1	1,800	8.7E+04	✓		
2,3,7,8-TCDD TEQ [b]	3.1E-06	1.6E-06	4.9E-06	5.8E-04	✓		

[a] Low Molecular Weight PAHs are composed of fewer than four condensed aromatic ring structures, and High Molecular Weight PAHs are composed of four or more condensed aromatic ring structures (EPA, 2007e).

[b] Maximum concentration applied instead of 95th percentile due to small sample size

4.4.4 Results

The 95th percentile SFS-manufactured soil concentrations of many of the SFS constituents were below their respective Adjusted SSL, dermal SSL and ecological SSL, and therefore required no further evaluation. For example, the SFS-manufactured soil concentrations for all of the phenolics, PAHs, dioxins, and dioxin-like compounds were below the screening criteria. In addition, all constituents with dermal SSLs were below the screening criteria, suggesting that these constituents do not require further evaluation for this pathway. However, the SFS-manufactured soil concentrations of three metals—arsenic, cobalt, and iron—were above the Adjusted SSL for multi-pathway exposures. Also, the SFS-manufactured soil concentrations for five metals - antimony, trivalent chromium, copper, manganese and nickel – were above the Eco-SSL. Based on these findings and constituent-specific information, the following decisions were made:

- Arsenic was retained for further study in Phase II.
- Due to their potential for phytotoxicity, both manganese and nickel were retained for further study in Phase II.
- The SFS-manufactured soil concentrations for antimony, trivalent chromium, and copper were similar to, but above their Eco-SSL's for small insectivorous mammals. Therefore, antimony, chromium (III) and copper were retained for further study in Phase II.
- The SFS-manufactured soil concentrations of cobalt and iron were above their respective Adjusted SSLs. Therefore, cobalt and iron were retained for further study in Phase II.

4.5 Analysis Phase I Results

At the beginning of this evaluation, there were three major media-specific exposure pathways under consideration: (1) *groundwater pathway* - the ingestion and dermal exposure to groundwater contaminated by the leaching of SFS constituents; (2) *ambient air pathway*- the inhalation of SFS emitted from soil-blending operations; and (3) *soil pathway* - the incidental ingestion and dermal exposure to soil, as well as ingestion of fruits and vegetables grown in SFS-manufactured soil. Because all evaluated SFS constituents were removed from further consideration by the inhalation screening, the inhalation pathway itself will not be further evaluated. Under the soil and groundwater dermal screening assessment, all evaluated SFS constituents were well below a level of concern, and dermal exposure likewise will not be further evaluated. However, based on other groundwater and soil evaluation criteria (e.g., Adjusted SSL screen for multi-pathway exposures), eleven metals were retained for further evaluation in the risk modeling phase. **Table 4-14** lists the metals retained for Phase II risk modeling.

Table 4-14: SFS Constituents Retained for Phase II Risk Modeling

Human Risk Modeling	Ecological Risk Modeling
Antimony (groundwater)	Antimony
Arsenic (groundwater and soil/produce)	Chromium III
Beryllium (groundwater)	Copper
Cadmium (groundwater)	Manganese
Cobalt (soil/produce)	Nickel
Iron (soil/produce)	
Lead (groundwater)	

5. Analysis Phase II: Risk Modeling of COCs

Based on the screening evaluations described in **Chapter 4**, five metals (antimony, arsenic, beryllium, cadmium, and lead) were retained for probabilistic modeling of the groundwater pathway, while eight metals (antimony, arsenic, chromium III, cobalt, copper, iron, manganese, and nickel) were retained for probabilistic modeling of the soil pathways. Arsenic, cobalt, and iron were evaluated for human exposures through the soil/produce ingestion pathway, but only arsenic was evaluated under the groundwater pathway. Manganese and nickel in SFS were modeled in the home gardening scenario because of their potential for phytotoxicity. Finally, concentrations of antimony, trivalent chromium, and copper were retained for further study due to the potential to impact small insectivorous mammals as described in Chapter 4.

Probabilistic modeling was conducted to address the variability in conditions across the country. This was done by using metal-, regional- and site-specific data to conduct probabilistic analyses of the remaining constituents of potential concern and exposure pathways.

This chapter is organized as follows:

- **Section 5.1** provides an overview of Phase II probabilistic modeling
- **Section 5.2** explains the screening probabilistic modeling of exposure via groundwater ingestion
- **Section 5.3** describes the more refined probabilistic modeling of exposures via soil, produce consumption, and groundwater ingestion, the results of the modeling, and the derivation of screening levels for the modeled constituents of potential concern in SFS.

5.1 Overview of Phase II Probabilistic Modeling

Figure 5-1 is a simple depiction of how the Monte Carlo probabilistic approach was implemented for the SFS evaluation. It shows how the distributions for input parameters were sampled and used to produce the probability distribution and cumulative distribution function from which specific percentiles (e.g., 90th percentile) can be identified. The example parameters A, B, and C each have their own distributions, which may represent variability or uncertainty or both. For each model run, a single value was sampled from each input distribution regardless of the type of variation (i.e., variability or uncertainty). For each modeling scenario (e.g., adult or child), the simulation produced the probability distribution of risk results, as shown in Figure 5-1 (i.e., the distribution of risk across exposed individuals across all sites represented in the analysis). Lastly, the cumulative distribution function was created, and the specific percentiles (e.g., 90th percentile) were selected and used to characterize risks.

Home garden location was the primary determinant for selecting parameter values that describe the environmental setting. A geographic information system (GIS) sampling procedure was used that correlated location, climate station, and soil type, thus ensuring that feasible combinations were modeled. The rest of the regional data (i.e., long-term climate data and daily meteorological data) were held constant for all sampled locations within a given climate region.

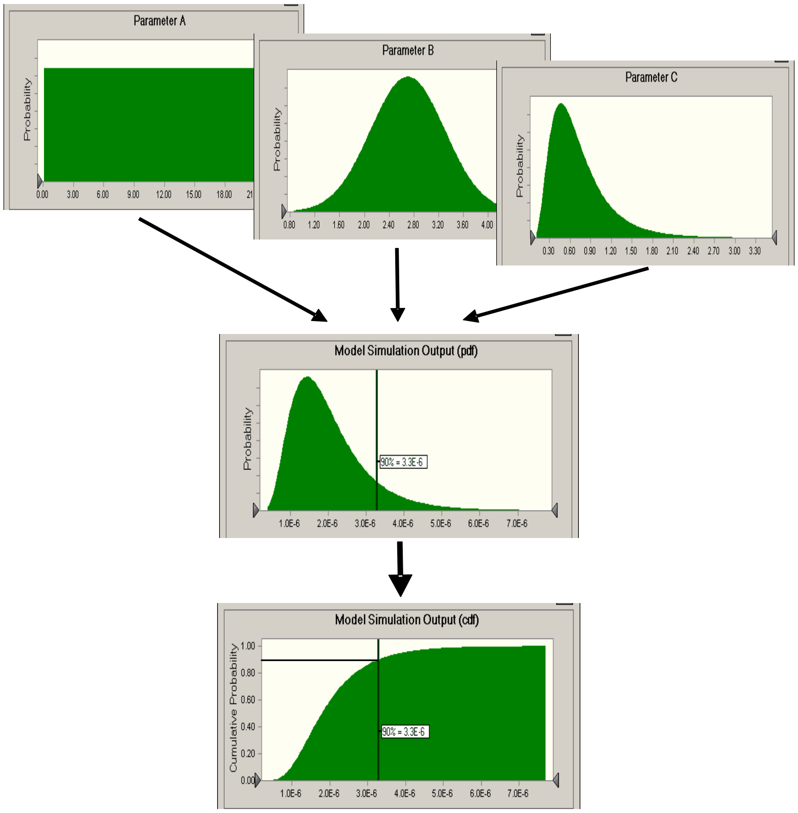

Figure 5-1. How the Monte Carlo approach addresses uncertainty.

5.2 Screening Probabilistic Modeling of the Groundwater Ingestion Pathway

Five constituents (antimony, arsenic, beryllium, cadmium, and lead) were retained for Phase II evaluation. EPA's IWEM probabilistic groundwater screening model was used to evaluate the home garden scenario groundwater pathway. IWEM has undergone extensive peer review, and provides a flexible scenario for considering the potential leaching from the use of SFS in manufactured soils. Detailed information on this model can be found in the *IWEM User's Guide* (U.S. EPA, 2002a) and *Technical Background Document* (U.S. EPA, 2002b).[34]

As a conservative assumption, the 95th percentile SFS leachate concentration for each of the five constituents was used with site-descriptive parameter values. The model ran each constituent 10,000 times for 10,000 years assuming a constant leachate profile from a single application of SFS-manufactured soil, varying site conditions based on original inputs. **Figure 5-2** illustrates a conceptual cross-section of the subsurface modeled in the SFS evaluation. After all runs were completed, the estimated well-water concentration representing the 90th percentile (i.e., higher than 90 percent of the other estimates) was compared to the lowest Phase I screening level (i.e., Tapwater Screening Level, MCL, or National Secondary Drinking Water Standard – see Chapter 4 Section 4.2.3 for more information on these screening levels). If the constituent's

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[34] Supporting documentation for IWEM and EPACMTP can be found at http://www.epa.gov/osw/nonhaz/industrial/tools/

90th percentile well-water concentration estimate was at or below the screening level, then the constituent was not retained for further evaluation.

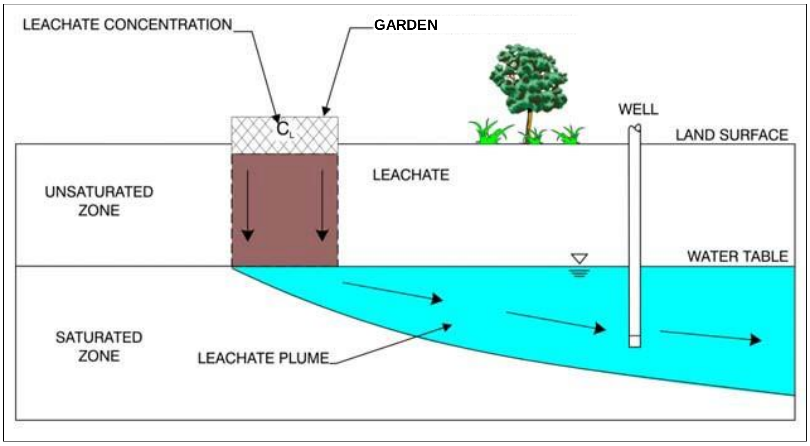

Figure 5-2. Conceptual Cross-Section View of the Modeled Subsurface

5.2.1 Groundwater Model Inputs

Some modeling input parameter values (e.g., distance from the garden to the drinking water well) were chosen to be conservative and to maximize drinking water estimates. Values for some other input parameters (e.g., depth to aquifer) were chosen from distributions representing variable conditions around the country. For the remaining parameters, the default values provided in the IWEM User's Guide (U.S. EPA, 2002a) were used. The model used the following parameters to define the use scenario:

- A 405 m^2 (i.e., 0.1 acres) land application unit (i.e., unconsolidated application to land) was operated for 40 years.[35] An area of 0.1 acres was selected to be conservatively representative of a garden suitable for SFS-manufactured soil use and that is of sufficient size to feed a home gardening family for a year.[36]
- To test the effect of distance from the garden to the drinking water well, separate sets of 10,000 runs were performed for each of the following distances: 1 m, 15 m, 30 m, and 50 m.

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[35] An operating life of 40 years for the land application unit is consistent with the default operating life applied in EPACMTP and in the Multi-media, Multi-pathway, Multi-receptor Risk Analysis (3MRA) modeling system for land application (U.S. EPA, 2003d, g).

[36] A 0.1 acre garden is more than sufficient to support the home gardener scenario that includes an adult and child receptor. The North Carolina State University, Department of Horticultural Science, reports that a garden of 25 ft × 40 ft (approximately 0.02 acres) will produce most of the vegetables needed by 2 people for one year (http://www.ces.ncsu.edu/depts/hort/hil/ag-06.html). Additional references also report garden sizes much smaller than the modeled 0.1 acres. For example, The National Gardening Association reported in 2009 that only 6% of U.S. gardens were larger than 2,000 ft^2 (0.05 acres) (http://www.gardenresearch.com/files/2009-Impact-of-Gardening-in-America-White-Paper.pdf).

- The following subsurface parameters (e.g., groundwater pH, depth to water table) were set to model defaults:[37]
  - Depth to water table: 5.18 m (IWEM default for a shallow aquifer)
  - Aquifer thickness: 10.1 m
  - Hydraulic conductivity: 1.89E+03 m yr^{-1}
  - Regional hydraulic gradient: 0.0057
  - Groundwater pH: 7
  - Chemical-specific decay rate: 0 for metals
  - Soil-water partition coefficient: selected from isotherms generated by the MINTEQA2 geochemical speciation model[38]
- Each model run included a randomly selected well-screen depth, constrained to occur within the aquifer (i.e., between 5.18 m and 15.28 m below the ground surface).
- Other unsaturated zone parameters were varied for each run based on a nationwide distribution of three soil types: sandy loam (15.4%), silt loam (56.6%), or silty clay loam (28%)
- To represent conditions across the country, three climates were modeled: a representative dry climate (Phoenix, AZ), a moderate climate (Indianapolis, IN), and a wet climate (Seattle, WA)
- For arsenic, the higher of the 95th percentile leachate concentrations determined by either the SPLP or ASTM leachate methods (0.018 mg L^{-1}) was modeled. Antimony, beryllium, cadmium, and lead were not detected in any samples, and were therefore modeled at one half their detection limits in accordance with U.S. EPA (1991b). Thus, their modeled leachate values were 0.02, 0.01, 0.005, and 0.055 mg L^{-1}, respectively.

Effect of well distance on drinking water concentration: As illustrated in Figure 5-2, some horizontal distance is required for the constituent plume to mix to the bottom of the aquifer. The horizontal distance required for a constituent to mix to the bottom of the aquifer depends on constituent-specific characteristics (e.g., soil-water partitioning), and therefore the distance will vary by constituent. Constituent concentrations *within the groundwater plume* will be highest directly under and near the garden. Concentrations will decrease as the plume travels horizontally, the constituent mass diluting into an ever larger volume of groundwater.

The random selection of well-screen depth (see bullet 4, above) will, for some model runs, result in the contaminant plume “missing” the well. For instance, if the screen depth illustrated in Figure 5-2 had been chosen to be 15 m (i.e., near the bottom of the aquifer) rather than within the contaminant plume, the plume would have moved above the screen and produced a zero well concentration. Existence of these zero concentrations in the output distribution would skew percentile calculations lower (i.e., a lower value can be above 90% of the other values when some of the other values are zero).

To test the interplay between constituent dilution and the effects of zero concentrations on output percentiles, a complete set of 10,000 model runs was completed for each constituent in

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[37] See U.S. EPA (2002b), section 4.2.3.1 for details on how these defaults were chosen for IWEM.

[38] See U.S. EPA (2002b) section 4.3.4.3.2 for details on how MINTEQA2 was used to produce the isotherms sampled for partition coefficients.

each climate at well distances of 1 m, 15 m, 30 m and 50 m, respectively. If IWEM returned a receptor well concentration of “0” (i.e., below 1.0E-20 mg L^{-1}) at all four distances, then no further modeling of that constituent was performed in that climate. If IWEM returned non-zero values that demonstrated a peak and dilution with greater distance, then no further modeling of that constituent was performed in that climate. If no peak concentration was demonstrated (i.e. modeling at greater distances elicited higher receptor well concentrations), IWEM was run again and re-evaluated at 75 m, 100 m, 150 m, and 200 m, or until a peak was demonstrated.

5.2.2 Groundwater Model Outputs

Table 5-1 lists the groundwater modeling results at the 90th percentile. As shown in the table, the exposure estimates for arsenic in the Wet and Central Tendency climates were above the screening level, and below the screening level for the Dry climate. The exposure estimates for beryllium, cadmium, lead, and antimony were consistently lower than the screening levels in all three climates.

Table 5-1. Tested Leachate Concentrations, Receptor Well Concentrations for the Home Gardener Exposure Scenario, and Screening Levels (mg L^{-1})

Constituent	Tested Leachate Conc.	90th Percentile Modeled Exposure Level [a]						Lowest Screening Level [b]
		1 m	15 m	30 m	50 m	75 m	100 m	
Wet Climate								
As	0.018	**4.9E-03**	**3.4E-03**	**2.5E-03**	**1.8E-03**	NM	NM	4.5E-04[c]
Be	0.01	1.7E-09	3.8E-08	7.2E-07	1.5E-06	1.1E-06	NM	4.0E-03
Cd	0.005	2.3E-03	1.5E-03	1.1E-03	7.3E-04	NM	NM	5.0E-03
Pb	0.055	5.9E-03	3.00-03	1.7E-03	1.1E-03	NM	NM	1.5E-02
Sb	0.02	5.9E-03	4.5E-03	3.2E-03	2.4E-03	NM	NM	6.0E-03
Moderate Climate								
As	0.018	**5.2E-04**	**9.6-04**	**8.9E-04**	**6.8E-04**	NM	NM	4.5E-04[c]
Be	0.01	0	6.9E-14	8.2E-13	2.9E-12	4.1E-12	3.4E-12	4.0E-03
Cd	0.005	2.6E-04	4.3E-04	3.7E-04	2.6E-04	NM	NM	5.0E-03
Pb	0.055	2.0E-03	2.1E-03	1.0E-03	5.3E-04	NM	NM	1.5E-02
Sb	0.02	1.1E-03	1.8E-03	1.7E-03	1.3E-03	NM	NM	6.0E-03
Dry Climate								
As	0.018	0	0	0	0	NM	NM	4.5E-04[c]
Be	0.01	0	0	0	0	NM	NM	4.0E-03
Cd	0.005	0	0	0	0	NM	NM	5.0E-03
Pb	0.055	0	0	0	0	NM	NM	1.5E-02
Sb	0.02	0	0	0	0	NM	NM	6.0E-03

[a] The model reports a “0” level if the 90th percentile modeled well concentration is lower than 1.0E-20 mg L^{-1}.

[b] Unless otherwise noted, MCLs were the lowest screening level.

[c] For arsenic, the Tapwater Screening Level was the lowest screening level. The arsenic Tapwater Screening Level used in this evaluation is based on a 10^{-5} risk level

NM = Not Modeled

5.2.3 Results

The well distance demonstrating peak concentration varied by constituent and climate, but in no case was further than 75 m. In modeling a wet climate, antimony, arsenic, cadmium, and lead demonstrated peak concentrations at a well distance of 1 m, and were therefore not modeled beyond 50 m. Beryllium demonstrated a peak receptor well concentration at a well distance of 50 m.

In modeling the constituents in a moderate climate, antimony, arsenic, cadmium, and lead demonstrated peak concentrations at a well distance of 15 m, and were therefore not modeled beyond 50 m. IWEM estimated a receptor well concentration of zero for beryllium at a 1 m well distance, but ultimately peaked at a distance of 75 m. In modeling the constituents in a dry climate, IWEM estimated receptor well concentrations of zero for all constituents across the first four distances, and therefore no further modeling performed.

The screening probabilistic modeling for groundwater ingestion found that estimated exposures for antimony, beryllium, cadmium, and lead were below drinking water screening levels in all climates and at all well distances. Therefore, no further evaluation of exposure to those constituents via groundwater ingestion was necessary. Estimated exposures for arsenic were consistently above the drinking water screening level in the Wet and Moderate climates, and consistently below the screening level in the Dry climate. Arsenic was therefore retained for more refined study.

5.3 Refined Probabilistic Modeling of the Soil/Produce and Groundwater Ingestion Pathways

As described in **Chapter 4**, four constituents of potential concern required further evaluation of the soil/produce ingestion pathway: arsenic, lead, manganese, and nickel. In addition, as described in Section 5.2, arsenic was retained for refined evaluation of the groundwater pathway. As part of this evaluation, probabilistic modeling of these constituents was performed to derive risk-based modeled screening levels for comparison to SFS constituent concentrations. If the SFS concentrations were below these conservative SFS-specific screening levels, then the beneficial use of SFS as a component of manufactured soil would be considered protective of human health and the environment. The following provides an overview of the process used to derive the modeled screening levels.

Risk distributions were developed using an initial soil concentration of 1 ppm for each constituent; this initial concentration is referred to as a “unitized” concentration in the sense that it does not represent an actual concentration in SFS or soil; rather, it represents an arbitrarily chosen concentration that is used to estimate risk per “unit” of constituent in soil. Consistent with previous EPA risk assessments and based on the model’s linearity with respect to constituent concentration, the 90th percentile of the unitized risk estimates was scaled to estimate protective target SFS constituent concentrations based on EPA’s risk management criteria (e.g., hazard quotient of 1).[39,40] These SFS-specific concentrations (i.e., concentrations in SFS, rather than concentrations in soil) are conservative estimates of the selected SFS constituents that would be protective of human health and the environment if the SFS were used in manufactured soil. The following summarizes the individual steps taken to develop the target SFS concentrations from the unitized risk distributions.

Step 1. Estimate Environmental Releases

Using an initial soil concentration of 1 ppm, the source model was run to simulate the release of constituents to surrounding media from a home garden assumed to receive a single "addition" of SFS-manufactured soil to a depth of 20 cm (a typical tilling depth). As discussed in **Sections 5.3.3 and 5.3.4**, release mechanisms simulated by the model include losses due to leaching, volatile and particle releases to the air, and horizontal movement of pollutants (i.e., runoff and erosion from the garden). The model generates time-series estimates for these releases, as well as estimates for surficial and root zone soil concentrations. For arsenic (i.e., the only SFS constituent requiring refined groundwater modeling), leachate fluxes (g m^{-2} yr^{-1}) estimated by the source model were used by the groundwater fate and transport model to estimate arsenic concentrations at the drinking water receptor well.

Step 2. Calculate Unitized Ratios

Calculating risk from the source modeling outputs involved fate and transport modeling (**Section 5.3.5**, groundwater modeling, and **Section 5.3.6**, food chain modeling), human exposure and health effects modeling (**Sections 5.3.7 and 5.3.8**), and ecological exposure and health effects modeling (**Section 5.3.9**). The probabilistic simulation generated distributions of unitized risks for adult and child home gardeners, as well as for ecological receptors, that reflect the variability in conditions within the economic feasibility areas.

Step 3. Calculate SFS Screening Level

Using 90th percentile unitized risk estimates, and EPA's risk management criteria (e.g., HQ of 1), screening levels were calculated for each constituent. As shown in **Section 5.3.11**, the calculation of SFS screening levels also allows for the adjustment of levels based on the fraction of SFS in manufactured soil. The resulting soil concentrations represent conservative estimates of SFS constituent concentrations considered protective of human health and the environment.

The remainder of this chapter is organized as follows:

- **Section 5.3.1** provides an overview of the risk modeling framework implemented to perform probabilistic modeling.
- **Section 5.3.2** describes the exposure scenario, including conservative screening assumptions, developed for application of SFS in home gardens.

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[39] Similar unitized approaches have been applied under previous U.S. EPA risk assessments. For example, the unitized approach was applied in the Risk-Based Mass Loading Limits for Solvents in Disposed Wipes and Laundry Sludges Managed in Municipal Landfills. This risk assessment and the unitized approach have been extensively reviewed, and the final rule based on this risk assessment, Solvent-Contaminated Wipes, was published July 31, 2013 (U.S. EPA, 2013a)

[40] Appendix J describes the analysis that was performed to confirm that the unitized calculation method was appropriate for the groundwater modeling of arsenic.

- **Section 5.3.3** describes the receptors (both human and ecological) and the exposure pathways by which receptors could potentially be exposed to SFS constituents.
- **Sections 5.3.4 through 5.3.10** describe the models, inputs, and outputs used in the probabilistic screening of health and ecological risk associated with SFS use in home gardens.
- **Section 5.3.11** describes how the human and ecological modeling results were used to calculate SFS-specific screening levels.
- **Section 5.3.12** compares the SFS constituent concentrations to the lowest human health-based SFS-specific screening levels, as well as ecological SFS screening levels**.**

5.3.1 Modeling Framework Overview

Unitized risk distributions were developed for this analysis using a risk modeling framework currently used by EPA to support the Part 503 biosolids program. The risk modeling framework integrates a variety of models and input datasets facilitating site-based and national-level exposure and risk assessments. The SFS assessment modified and adopted the system to evaluate soil/produce and groundwater ingestion risks associated with the use of SFS in manufactured soils.

Under this assessment, we used a Monte Carlo approach that essentially loops over randomly selected locations within the area of economic feasibility, selecting input parameter values that correspond to each particular location. Within the looping structure, a series of modules are executed in a specific order. The modeling process can be summarized as follows:

- The source models estimate pollutant releases to the environment
- The environmental fate and transport models estimate concentrations in environmental media (e.g., soil, groundwater, ambient air) and in dietary items (e.g., fruits and vegetables)
- The exposure models estimate the pollutant levels to which receptors are exposed
- The human risk model estimates the chemical-specific human health risk, and the ecological effects model estimates chemical-specific hazard quotients.

The major functionality of the models implemented in this risk analysis is described in **Sections 5.3.4** through **5.3.10**.

As illustrated in **Figure 5-3**, the looping structure is comprised of four nested loops: Chemical; RunID; Human Receptor; and Ecological Receptor. The outmost loop is the chemical loop, which allows a Monte Carlo simulation to be performed on a constituent-specific basis. The next loop is the RunID loop, which controls the number of iterations performed in a given simulation and is used as the primary index to input datasets, including site location. As shown in Figure 5-3, the source, media, and food modules are executed for each Monte Carlo iteration. Outputs from the source model are used as inputs to the downstream groundwater, media and food modules to estimate concentrations that receptors can potentially be exposed to.

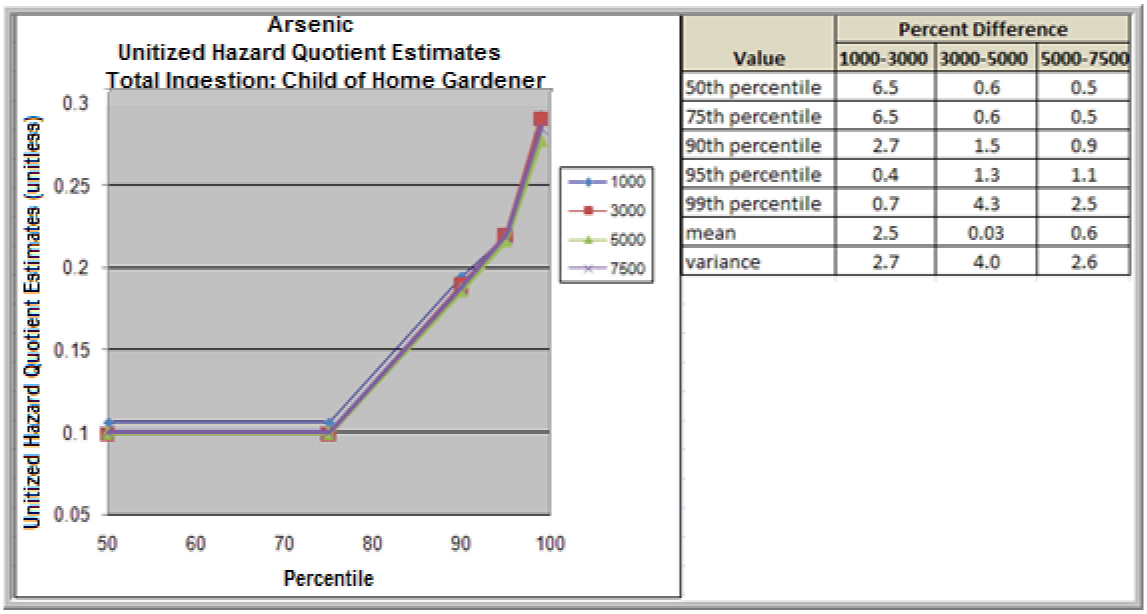
Within the Monte Carlo loop, the next loop in the probabilistic analysis cycles through the different types of receptors. The model considers both adult and child receptors and various ecological receptors. The receptor type determines the exposure factors used. Receptor type and exposure factors were not specific to location; as a result, any receptor (human or ecological) could be present at any location with any applicable exposure parameter values. Receptor-specific exposure factors for humans include exposure duration, the receptor's age when exposure begins, dietary consumption rates, and individual body weight. A set of adult and child exposure parameters was chosen for each iteration. Exposure parameters were not correlated with each other or with geographic locations. Ecological exposure parameters included the receptor-specific health benchmarks. More detailed descriptions of human and ecological exposure modeling are found in **Sections 5.3.7** and **5.3.8**, respectively.

```
SourceID = Home Garden
  Chemical (CAS) Loop
    RunID Loop (Monte Carlo iterations)
        Call Source Module: calculate emission rates; soil concentrations and losses
            due to leaching, runoff, and erosion
        Call Media Module: calculate groundwater and air concentrations
        Call Food Module: calculate concentrations for food items
      Human Receptor Loop (adult, child)
        Select pathways and exposure data based on human receptor type
        For Adult Receptor
            Calculate intake over exposure duration
        For Child Receptor
          Cohorts Loop (ages child through age cohorts)
              Calculate cohort intake
          Next Cohort
              Calculate intake over exposure duration
        Call Human Risk Module: calculate risk based on human health benchmarks
      Next Human Receptor
      Ecological Receptor Loop
        Select pathways and ecological exposure data based on ecological receptor type
        Call Ecological Exposure Module and calculate ratios of media concentrations to
            ecological concentration benchmarks
      Next Ecological Receptor
    Next RunID
  Next Chemical
```

Figure 5-3. Basic Monte Carlo looping structure for the home garden.

The Monte Carlo simulation represents a set of individual model realizations, with each realization defined in terms of a unique set of values for the input parameters required by the model. The approach is implemented by creating input files prior to the assessment that include data that are randomly selected based on the regional setting and scenario selected for each iteration. Chemical-specific data are generally constant across all iterations and are not correlated with other input parameters. The SFS-manufactured soil concentration was also held constant under this assessment to allow the calculation of the unitized risk estimates. The input of the fixed initial soil concentration of 1 ppm wet weight (i.e., unit concentration) into this linear system allowed for the development of unitized risk estimates that reflect national variability. The unitized approach was ideal for the SFS analysis since it provided the flexibility to generate distributions of unitized risk estimates that could be scaled to calculate screening concentrations using a variety of recipes for SFS-manufactured soils.

Under the SFS analysis, 7,500 Monte Carlos iterations were executed. To ensure the stability of the model results and determine the appropriate number of Monte Carlo simulations, the model was run for 4 different sets of iterations: 1,000; 3,000; 5,000; and 7,500 iterations. Tolerance criteria were established at 5%; that is, the model would be considered to be stable if the mean, variance, and the 50th and 90th percentile results did not change by more than 5%. Based on previous experience, the model was expected to converge in less than 5,000 iterations. The results of the stability test are shown in **Figure 5-4**. The table shown in the figure presents the absolute percent changes between samples. As demonstrated by this figure, the model is stable before 5,000 iterations for the mean, variance, and at the 50th and 90th percentiles, and extending the simulation to 10,000 iterations was considered unnecessary.



Value	Percent Difference		
	1000-3000	3000-5000	5000-7500
50th percentile	6.5	0.6	0.5
75th percentile	6.5	0.6	0.5
90th percentile	2.7	1.5	0.9
95th percentile	0.4	1.3	1.1
99th percentile	0.7	4.3	2.5
mean	2.5	0.03	0.6
variance	2.7	4.0	2.6

Figure 5-4. Model stability.

5.3.2 Exposure Scenario—Use of SFS in Home Gardens

The modeled use of SFS in home gardens assumed that a portion of a residential yard is used for home gardening: either the yard itself is tilled or raised beds are constructed. A single application of 20 cm (approximately 8 inches) of SFS-manufactured soil is spread in the residential construction area as topsoil, or a single application of 20 cm of SFS-manufactured soil is used in the construction of raised gardening beds. SFS is generated across the United States; therefore, the evaluation used a regional approach to capture the variability across site conditions. The modeling framework used regional climate and soil data to estimate constituent-specific releases and to predict their fate and transport in the environment. For example, the source model used soil data and daily precipitation data to estimate events such as runoff, erosion, and leaching.

The SFS was assumed to be used within 50 km of the foundry (EPA, 2008c).[41] This approach thereby focused the evaluation on climate and soil conditions relevant to where SFSs might reasonably be used as a component of manufactured soil. **Figure 5-5** shows the areas included in the assessment.

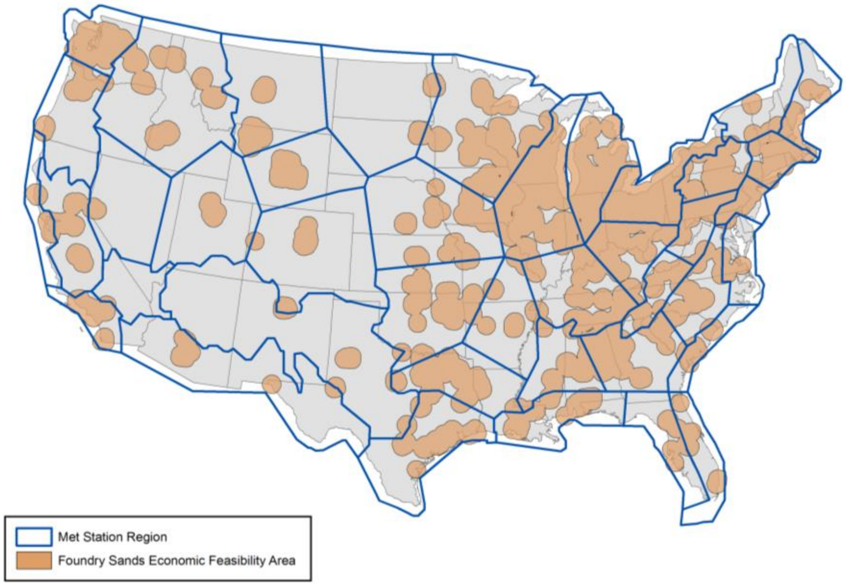


Figure 5-5. Meteorological regions and SFS use areas.

The scenario consists of the following elements:

- Regional data for 41 climate regions. Climate regions were shaped such that climate data from a single location would represent any location within the region, taking into account geographic boundaries, such as mountains, and other parameters that differentiate meteorological conditions (e.g., temperature and wind speed) as described in **Appendix D**.
- Locations of foundries in the United States, in the form of ZIP Code boundaries extended 50 km.
- Using a geographic information system (GIS), a soil layer was overlaid with the meteorological regions to identify location-specific soil texture and characterize soil parameters as described in **Appendix E**.

5.3.3 Potential Release Pathways and Receptors

Chapter 3 described the conceptual models that define the sources, releases, exposure pathways, and receptors relevant to the use of SFS in manufactured soil. The potential exposure pathways not fully modeled previously—incidental soil ingestion and ingestion of fruits and vegetables grown in SFS-manufactured soil—were modeled in this phase of the evaluation. In addition, the groundwater pathway was further evaluated for arsenic. **Figure 5-6** diagrams the

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[41] SFS use areas are based on the ZIP codes of the membership of the American Foundry Society as of November 2007. Since a foundry's exact location within its ZIP Code area was not provided, the ZIP code boundary was extended by 50km.

portions of the conceptual model (described in the problem formulation) that were addressed by this national-scale modeling. The diagram shows how the data flow from the *source models*, which are used to estimate releases to the environment, to the *environmental fate and transport models*, which are used to estimate concentrations in the soil, leachate, groundwater, eroded soil and air, to the *exposure models*, which are used to estimate concentrations in the food chain and resulting exposures to human and ecological receptors.

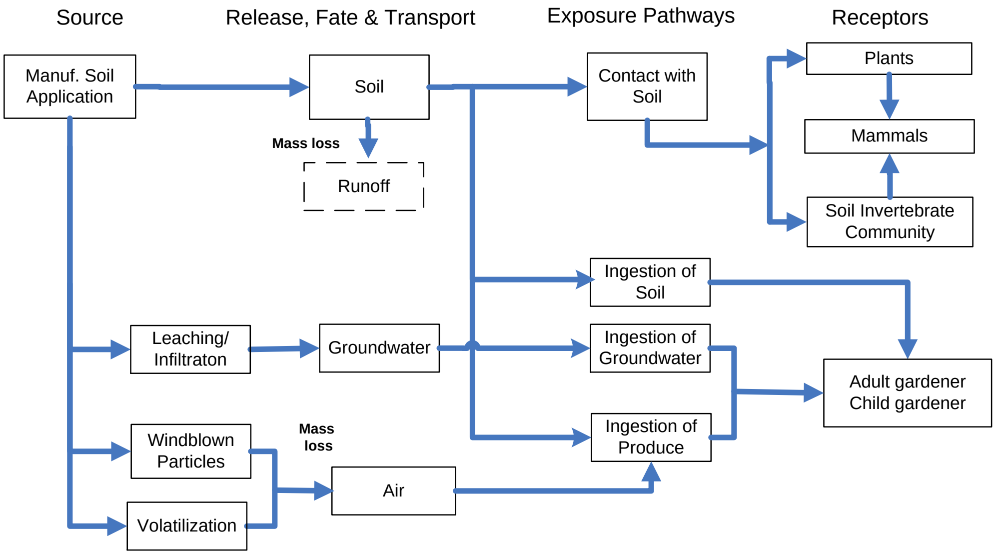

Figure 5-6. Conceptual model for modeling the home gardener.

As shown in Figure 5-6, the human and ecological receptors identified in the conceptual model could be exposed through various pathways. To estimate screening SFS concentrations, human and ecological receptors that would be subject to reasonable maximum exposures were identified. The potentially exposed human receptors are assumed to be members of a family that live and grow food in a garden on property where manufactured soil contains SFS. These individuals would be more highly exposed to SFS than the general population. In addition, the percentage of the gardening receptor’s diet that consists of home-grown produce is assumed to be higher than the percentage for the general population. Throughout the modeling, exposure assumptions were designed to be conservative; that is, they were likely to overestimate, rather than underestimate potential exposures.

The exposure pathways considered for adult and child receptors are summarized in **Table 5-2**. Although these pathways were evaluated concurrently within the modeling framework, analyses were performed as discussed in Section 5.3.5 and Appendix J that indicated that the maximum groundwater and soil/produce pathway exposures would not occur within the same period of time. As a result, separate SFS screening levels were developed for the groundwater and the soil/produce pathways.

Table 5-2. Human Exposure Pathways for SFS-Manufactured Soil in Home Gardens

Receptor	**Ingestion of Ground-water**	**Ingestion of Soil**	**Ingestion of Exposed Fruits** (e.g., apples)	**Ingestion of Protected Fruits** (e.g., oranges)	**Ingestion of Exposed Vegetables** (e.g., lettuce)	**Ingestion of Protected Vegetables** (e.g., corn)	**Ingestion of Root Vegetables** (e.g., carrots)
Resident Adult Gardener	✓	✓	✓	✓	✓	✓	✓
Resident Child	✓	✓	✓	✓	✓	✓	✓

5.3.4 Source Modeling

This section provides an overview of the source model and modeling approach, and identifies model inputs and outputs.

5.3.4.1 Conceptual Source Model

The source model used in this assessment was the land application unit model developed for ORCR as part of the 3MRA modeling system (U.S. EPA, 2003c). The land application unit model was developed to estimate annual average surface soil constituent concentrations and constituent mass release rates to the air, downslope land, and groundwater. The model simulates the vertical movement of pollutants within the agricultural land (releases through leaching to groundwater), volatile and particle releases to the air, and horizontal movement of pollutants (runoff and erosion from the agricultural land across any buffer area to a nearby waterbody). The model considers losses from the agricultural land due to hydrolysis and biodegradation, as well as leaching, volatilization, and particle emissions due to tilling (mixing) operations and wind erosion.

The model has been extensively peer reviewed and has been used to support several risk assessments conducted for EPA’s ORCR and Office of Water. Although the source model was initially developed to assess hazardous wastes, it has been used to support regulatory risk assessments, including the 2003 and 2013 biosolids exposure and hazard assessments. Under these national assessments, biosolids were assumed to be applied to agricultural fields used to grow crops or used as pastureland. Under the SFS assessment, the crop modeling scenario was adopted and modified to assess human and ecological impacts associated with the application of SFS-manufactured soil in residential gardens. The following highlight areas where the current screening approach deviated from the biosolids methodology:

- A “soil replacement” assumption was applied instead of the “soil amendment” assumption in biosolids. The soil replacement scenario definition represents a reasonably conservative description regarding the use of SFS in manufactured soil.
- In the biosolids analyses, farm areas are varied stochastically by sampling from a distribution using data from Hoppe et al. (2001) that spans a range from 45 – 73 hectares (i.e., 111 to 180 acres). Because residential gardens are significantly smaller, the modeled application area was reduced to better reflect actual gardening practices. The garden was modeled as a 405 m^2 (i.e., 0.1 acres) area consistent with the IWEM modeling discussed in Section 5.2.1.

- Choices of garden location, meteorological data, and soil data were constrained to fall within the SFS economic feasibility areas. A discussion of this approach is provided in **Chapter 3**, Problem Formulation.
- The current screening level ecological assessment focused on direct contact with the soil. To evaluate potential ecological risks associated with the SFS constituents, EPA's Eco-SSLs for soil invertebrates, terrestrial plants, and small insectivorous mammals were identified and compared to predicted maximum predicted soil concentrations.

Under the soil replacement scenario, it was assumed that SFS-manufactured soil is applied one time, evenly across an area used for home gardening to a depth of 20 cm. Based on the assumed composition of SFS-manufactured soil, it was also assumed that the properties and characteristics of the SFS-manufactured soil would mimic those of natural soil in the area. That is, the SFS-manufactured soil used in the garden will be similar to the local native soil (which is a function of the garden location).

Consistent with the 2013 biosolids exposure and hazard assessments, the source model was coupled with EPACMTP to evaluate impacts to the groundwater pathway. The leachate fluxes (g m^{-2} yr^{-1}) and infiltration water fluxes (m d^{-1}) estimated by the source model were subsequently used as input to EPACMTP to estimate arsenic concentrations at the receptor well.

5.3.4.2 Source Model Inputs

The source model requires numerous input parameters, including location-specific parameters, constituent-specific parameters, and parameters that describe the garden's dimensions and operating practices. The following identifies key inputs and describes the approach used in characterizing the parameters; additional details on the source model mass-balance governing equations and parameter inputs are provided in **Appendix G**, Home Garden Source Model, and **Appendix F**, Chemical Data:

- **Constituent Concentrations.** Constituent concentrations were fixed to a unit concentration of 1 mg kg^{-1}. In applying a unitized concentration approach, the resulting constituent-specific hazard estimates were used to estimate concentrations in SFS-manufactured soil that could be applied without exceeding the hazard criterion adopted for this analysis. The criterion for this analysis was a Unitized Dose Ratio (UDR) of 1 for cancer and noncancer effects[42] at the 90th percentile of the hazard probability distribution.[43] A detailed discussion of the UDR is found in **Section 5.3.9.1**.
- **Chemical properties.** The model requires the input of several parameters, such as diffusivity in air and water. The chemical-specific properties used in this assessment are presented in **Appendix F**. The primary data source for these parameters is the Superfund Chemical Data Matrix (SCDM; U.S. EPA, 2008b), because it is peer reviewed and contains all of the constituents evaluated. Other sources include the Hazardous

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[42] In this evaluation, UDR refers to the generic ratio of estimated exposure divided by health benchmark, regardless of the type of adverse effect (i.e., cancer or noncancer) the benchmark is based on.

[43] EPA's *Guidance for Risk Characterization* (U.S. EPA, 1995c) defines the risk criterion for the hazard-based calculation to be protective of 90% of hypothetically exposed individuals, stating that "For the Agency's purposes, high end risk descriptors are plausible estimates of the individual risk for those persons at the upper end of the risk distribution," or conceptually, individuals with "exposure above about the 90th percentile of the population distribution."

Substances Data Bank (HSDB) (NLM, 2006) and the Merck Index (Budavari, 1989). Distributions for soil water partition coefficients (Kd) were derived from U.S. EPA (2005a).

- **Area of the garden.** As discussed in Section 5.2.1, the size of the garden was assumed to be 405 m^2 (i.e., 0.1 acres). An area of 0.1 acres was selected to be conservatively representative of a typical residential garden that is of sufficient size to feed a home gardening family for a year.
- **Characteristics of the SFS-manufactured soil (e.g., percent solids, bulk density, fraction organic carbon).** Properties and characteristics of the SFS-manufactured soil were assumed to mimic those of natural soil in the area. Because soil characteristics vary spatially, it was necessary to assign gardens to specific locations. With the added consideration of economic feasibility areas, the approach applied in making these assignments was consistent with the approach used in the biosolids assessments. Considering the joint probability of occurrence, gardens were assigned to one of 41 climate regions. Using a geographic information system (GIS), a soil layer was overlaid with the climatic regions to identify the predominant soil texture for the top 20 cm of soil. Specific soil parameters, such as bulk density and fraction of organic carbon, were characterized based on the selected soil type. The percent solid of the SFS-manufactured soil was calculated based on soil moisture at field capacity and soil bulk density.
- **Climate conditions at the garden site.** Gardens were assigned to one of the 41 climate regions. As discussed in **Appendix D**, a representative meteorological station and data set was selected for each climate. This data set was assumed to be representative of the conditions throughout the entire region.
- **Tilling depth.** The soil mixing depth for the garden was set to a default value of 20 cm to reflect tilling conditions. This value is consistent with the recommended default value for tilled soil in EPA’s *Human Health Risk Assessment Protocol* (U.S. EPA, 2005b).

5.3.4.3 Source Model Outputs

The outputs of the source model include the following:

- Annual average constituent concentration in the surface of the garden soil
- Annual average constituent concentration in the root zone of the garden soil
- Annual emission of volatile constituents from the surface of the garden soil
- Annual emission of constituents sorbed to particles from the surface of the garden soil due to tilling and wind erosion
- Daily concentrations and mass of soil eroded from the garden soil
- Daily concentrations and volume of runoff from the garden (used in calculating the load to the buffer)
- Daily concentrations and volume of runoff from the buffer area
- Annual infiltration rate of water from the garden
- Annual leachate flux of constituents from the garden.

5.3.5 Fate and Transport: Refined Groundwater Modeling

Refined probabilistic groundwater modeling used the EPACMTP (U.S. EPA, 2003f, g, h; 1997a). Consistent with other EPA national-scale assessments, EPACMTP inputs included distributions of leachate fluxes and infiltration rates from the home garden source model, rather than the single, 95th percentile leachate concentration used in screening probabilistic modeling. Coupling the source and groundwater modeling in this way captures national variability in conditions through the use of location-specific climate and soil distributions, as well as constituent-specific input parameters (e.g., soil Kd distributions) to estimate constituent-specific releases and to probabilistically predict their fate and transport in the environment.

EPACMTP accounts for advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and transformation processes via chemical hydrolysis. In this analysis, sorption of arsenic being leached from SFS-manufactured soil into the unsaturated and saturated zones was modeled using soil-water partitioning coefficients (Kd values) selected from nonlinear sorption isotherms generated from the equilibrium geochemical speciation model MINTEQA2 (U.S. EPA, 1991). As discussed in Appendix J, maintaining linearity with respect to sorption was critical to supporting the appropriateness of applying the unitized approach to estimate SFS Screening Levels. Kd selection was therefore monitored during the EPACMTP simulations, ensuring that the assumption of linearity was valid.

The groundwater concentrations are used in estimating drinking water exposures as shown in the equations presented in **Appendix H**.

5.3.5.1 Conceptual Groundwater Model

The groundwater pathway was modeled to estimate receptor well concentrations that result from a predicted release of arsenic from SFS-manufactured soil used in a home garden. The release of a constituent occurs by leachate, containing the constituent, percolating through the soils into the subsurface as a result of precipitation water infiltrating through the SFS-manufactured soil. The released constituent is transported via aqueous-phase migration through the unsaturated zone (the soil layer beneath the garden and above the aquifer) to the underlying saturated zone (i.e., groundwater), and then downgradient in the groundwater to a hypothetical residential drinking water well (the “receptor well”) located near the home garden.

Receptor Well Location

One of the key inputs for EPACMTP is the receptor well location. EPACMTP estimates the exposure concentration at the intake point of a hypothetical residential drinking water well located at a specified distance from the downgradient edge of the source area and at a specified depth below the water table. For this analysis, modeling simulated groundwater impacts to a well assumed to be placed in the centerline of the plume at a fixed distance of 1 m from the edge of the garden. The depth of the well was varied uniformly throughout the aquifer thickness, to a maximum of 10 m, whichever was less. That is, the well depth was never allowed to exceed 10 m below the water table. This limitation for well depth, used in several previous EPA analyses, is applied primarily for two reasons: (1) to be representative of typical residential well scenarios where wells are generally shallow because of the higher cost of drilling a deeper well and (2) to produce a conservative estimate of risk (because the infiltration rate is generally lower than the groundwater seepage velocity, groundwater plumes tend to be relatively shallow).

Key Assumptions

The groundwater modeling approach included the following key assumptions. More comprehensive documentation of the EPACMTP model and associated assumptions are available in the *EPACMTP Technical Background Document* (U.S. EPA, 2003f).

- The model assumes that the vertical migration is 1-D and that transverse dispersion is negligible in the unsaturated zone.
- The model assumes linear and non-linear equilibrium sorption and homogeneous aquifer conditions. However, as discussed in **Appendix J**, linearity has been demonstrated for the SFS arsenic leachate concentrations modeled in this analysis.
- The model assumes that receptors use the uppermost aquifer, rather than a deeper aquifer, as their drinking water source. This assumption could overestimate risks in cases in which the uppermost aquifer is not used.
- The model assumes that long-term average conditions are sufficient and that shorter frequency fluctuations (e.g., in rainfall/infiltration) are insignificant in estimating long-term risk.

Preferential flow in karst aquifers or in fractures was not considered, although such conditions are known to exist over broad areas. Preferential flow can allow contamination to migrate faster and at a higher concentration than in a standard porous medium. However, the contamination typically does not spread over such a broad area. As a result, the modeling may under- or overestimate the concentrations in groundwater.

5.3.5.2 Groundwater Model Inputs

EPACMTP requires a number of input parameters. Provided below is a summary of the key types of EPACMTP inputs and how they were parameterized in the SFS evaluation.

The leachate fluxes ($g\ m^{-2} \cdot yr$) estimated by the home garden source model were used as inputs to EPACMTP to estimate arsenic concentrations at the receptor well. All leachate fluxes from the source model were applied uniformly over the footprint of the garden, immediately below the garden.

To model the unsaturated zone, EPACMTP requires inputs for the following soil-related hydrological parameters: saturated hydraulic conductivity, van Genuchten soil moisture parameters, residual and saturated water contents, percent organic matter, and soil bulk density. Values for these parameters vary, and EPACMTP includes distributions of appropriate values organized by soil texture. EPACMTP requires a site-specific soil texture be input in order to determine which soil-related hydrologic parameter distributions will supply the unsaturated zone model input parameters. A pre-sampled distribution of saturated hydraulic conductivity (a particularly important variable) was shared by the home garden source model and the unsaturated zone model.

Similarly, the hydrogeological setting assigned to each garden was used to select appropriate aquifer conditions from EPACMTP’s Hydrogeologic DataBase (HGDB). Given an aquifer code setting for a garden site, a correlated sample of key aquifer model input parameters (hydraulic conductivity, hydraulic gradient, depth to the water table, and saturated thickness) was selected from a population of samples taken from similar hydrogeological settings. Details of the data used to parameterize the unsaturated zone and the development and use of the HGDB are given in the *EPACMTP Parameters/Data Background Document* (U.S. EPA, 2003b).

Recharge is water percolating through the soil to the aquifer outside the footprint of the garden. EPACMTP selects a recharge rate using a meteorological station assignment (based on the geographic location and topography of a garden setting) and by the garden's associated soil texture. Using the soil texture and station assignment, a recharge rate is selected from a database of the Hydrologic Evaluation of Landfill Performance (HELP) model–derived recharge rates for climate stations across the country and for various soil textures. Further details about how these rates are determined and other options for determining recharge rates outside of the EPACMTP model can be found in the *EPACMTP Parameters/Data Background Document* (U.S. EPA, 2003b). A few required inputs are based upon established empirical distributions and are described in the *EPACMTP Parameters/Data Background Document* (U.S. EPA, 2003b).

5.3.5.3 Groundwater Model Outputs

EPACMTP's outputs (i.e. predictions of the contaminant concentrations arriving at a downgradient receptor location) are time-dependent; they can vary over time. The model can calculate both the peak concentration arriving at the well and maximum time-averaged concentrations. The SFS Evaluation used maximum time-averaged concentrations (based on the exposure duration for each receptor type) to develop human risk estimates.

In some cases, it may take a long time for the plume to reach the receptor well, and the maximum groundwater exposure may not occur until a very long time after the application. This time delay may be on the order of thousands of years. If the model predicts that the maximum exposure will not have occurred after 10,000 years, the actual receptor concentration at 10,000 years will be used in the risk calculations.

An analysis was performed to evaluate anticipated arrival times to determine if the exposure through the soil/produce pathway would overlap with exposure through the groundwater pathway. To determine the approximate timeframe when the peak groundwater exposure might occur, estimates were made of the time at which the contaminant plume would arrive at the receptor well and the time when the contaminant plume would finish passing the well. Arrival of peak concentrations would only occur somewhere within this time period. These estimates were based upon two additional outputs from the unsaturated zone transport simulation: 1) first arrival time of leachate at the water table and 2) cessation time of leachate arrival at the water table. Retardation effects were used to account for horizontal travel to the receptor well. The results of this analysis are summarized in **Table 5-3**.

Table 5-3. EPACMTP Arrival Times of Arsenic Plume at the Receptor Well

	Arrival Time Zone (year)	
Percentile	**Beginning**	**End**
90 %	29	200
80 %	61	200
70 %	100	202
60 %	150	220
50 %	201	272
40 %	203	345
30 %	207	457
20 %	229	663
10 %	398	1,112

Based on the analysis (see Appendix J for more details), it is unlikely that peak soil/produce pathway exposures and peak groundwater exposures will occur within the same timeframe. For example, the earliest estimated timeframe for groundwater arrival of arsenic from the garden spanned from 29 to almost 400 years following the application of the SFS. It is therefore likely that the peak well concentrations will not occur until well past the timeframe for peak soil/produce pathway exposures, and perhaps even past the timeframe of residency (i.e., exposure duration of the gardeners who originally applied the SFS-manufactured soil). Therefore, separate screening levels were developed for the groundwater and soil/produce pathways.

5.3.6 Fate and Transport: Produce Modeling

The food chain model calculates constituent concentrations in food items using soil concentrations and emissions predicted by the source model and using air concentrations and deposition rates from the dispersion model. Constituents pass from contaminated soil and air through the food chain to the gardening family. For example, constituents in air may be deposited on plants growing in the garden. Simultaneously, these plants may take up constituents from the soil and accumulate constituents from both routes in the fruits and vegetables consumed by the receptors.

This section presents the methodology used to calculate constituent concentrations in the aboveground and belowground produce grown in the residential garden.

5.3.6.1 Conceptual Produce Model

The human food chain model is designed to predict the accumulation of a constituent in the edible parts of food crops eaten by the human receptor. Edible crops include exposed and protected fruits, exposed and protected vegetables, and root vegetables. The term “exposed” refers to the fact that the edible portion of the produce is exposed to the atmosphere. The term “protected” refers to the fact that the edible portion of the produce is shielded from the atmosphere (i.e., not impacted by air-to-plant transfer and particle deposition). Examples of the categories include tomatoes (exposed vegetable), corn (protected vegetable), apples (exposed fruit), oranges (protected fruit), and potatoes (root vegetables). The equations used to calculate the food chain concentrations of constituents are presented in **Appendix H**.

5.3.6.2 Produce Model Inputs

The inputs to the food chain model are vegetation-specific properties, soil and air concentrations, deposition rates, and other chemical-specific properties, such as bio-uptake factors. Estimation of soil concentrations is discussed in **Section 5.3.4**. The following identifies the additional input parameters that are needed to calculate constituent concentrations in aboveground and belowground (i.e., root vegetables) produce:

- **Aboveground produce.** Concentrations in aboveground produce consider impacts due to air-to-plant transfer, root uptake, and particle deposition. Exposed fruits and vegetables are susceptible to contamination through all three mechanisms, while protected vegetation is assumed to be impacted only through root uptake. The vegetation-specific parameters used in calculating these impacts are presented in **Appendix H**. The air-to-plant and root uptake factors for each constituent are identified in **Appendix F**.
- **Belowground produce.** Concentrations in belowground produce consider impacts due to root uptake, which is calculated for metals using chemical-specific soil-to-plant bioconcentration factors. These chemical-specific factors are presented in **Appendix F**.
- **Conversion to Wet Weight (WW).** The implemented equations predict aboveground and belowground concentrations on a dry weight basis. The model must convert these values to a wet-weight basis for use in the downstream exposure model, which applies wet-weight consumption rates. As shown in **Appendix H**, this conversion is made using plant-specific moisture adjustment factors (MAFs) (i.e., percent moisture). These factors, which vary by vegetation type, are identified in **Appendix H**.

5.3.6.3 Produce Model Outputs

The food chain model outputs constituent-specific concentrations in exposed and protected fruits, exposed and protected vegetables, and root vegetables. These concentrations serve as input to the exposure model, where they are combined with human consumption rates and other exposure factors to calculate an individual’s ingested dose.

5.3.7 Human Exposure Modeling

The predicted constituent concentrations in soil, drinking water, and food chain items are used to estimate human exposures. This section describes the human exposure modeling that was performed to estimate exposure based on the potential dose ingested. **Appendix H** presents the equations used to calculate dose for each pathway and for total ingestion.

5.3.7.1 Human Exposure Conceptual Model

Exposure through the ingestion route was estimated by multiplying the concentration of the constituent in the soil, drinking water, or food item by the consumption rate of the individual. This is the average daily dose (ADD) for an individual. Calculation of a lifetime average daily dose (LADD) for constituents with cancer endpoints also considers the individual’s exposure duration, averaging across an assumed lifetime (70 yr), and exposure frequency (350 d yr^{-1}). **Appendix H** presents the equations used to calculate ADD and LADD.

Exposure modeling relies heavily on default assumptions concerning population activity patterns, mobility, dietary habits, body weights, and other factors. The following highlights the key assumptions that were applied in estimating the level of constituents that the hypothetical home gardener and child were exposed to via ingestion of soil and homegrown aboveground and belowground produce.

- Both the adult and child members of the family were exposed to constituents through the application of SFS-manufactured soil to their own home garden. The exposure period for the receptors was constrained to begin at the time of application of the soil to the garden.
- The adult was 20 years old when exposure began, and the child was 1 year of age when exposure began. Application of these start ages maintains the conservative nature of this screening assessment. Infant exposure (i.e., 0 to 1 year of age) via the breastmilk pathway was not evaluated under this modeling scenario given that none of the metals included in the probabilistic modeling phase have been identified in current studies as being of significant concern via the breastmilk pathway.
- Receptors both lived and worked at the exposure location. This assumption may overestimate exposure, because individuals may live at the exposure location, but commute to work (or school or daycare) outside of the study area, or commute to areas within the study area where SFS-manufactured soil had not been used.
- In the case of incidental soil ingestion, the EPA's default relative bioavailability (RBA) value of 60% (U.S. EPA, 2012b) was used to adjust the distribution of arsenic concentration in soil for the exposure modeling. All other constituents were assumed to be 100% bioavailable.

5.3.7.2 Human Exposure Model Inputs

The inputs to the exposure model are human exposure factors and soil, drinking water, and food concentrations. Estimation of soil, drinking water, and food item concentrations is discussed in **Sections 5.3.4, 5.3.5 and 5.3.6**, respectively. The key human exposure factors used as inputs to the analysis include the following:

- Averaging time for carcinogens
- Exposure duration
- Exposure frequency
- Ingestion rate for soil
- Ingestion rate for drinking water
- Consumption rates for exposed vegetables, protected vegetables, exposed fruit, protected fruit, root vegetables
- Fraction food preparation loss for exposed vegetables, protected vegetables, exposed fruit, protected fruit, root vegetables.

These exposure factors were used to calculate the dose for the soil and produce ingestion pathways. The primary data source of human exposure model inputs used in this analysis was EPA's *Exposure Factors Handbook* (EFH; U.S. EPA, 2011) and *Child-Specific Exposure Factors Handbook* (CSEFH; U.S. EPA, 2008a). These references summarize data on human behaviors and characteristics related to human exposure from relevant key studies and provide recommendations and associated confidence estimates on the values of the exposure factors. These data were carefully reviewed and evaluated for quality before being included in the EFH and CSEFH. EPA's evaluation criteria included peer review, reproducibility, pertinence to the United States, currency, adequacy of the data collection period, validity of the approach, representativeness of the population, characterization of the variability, lack of bias in study design, and measurement error (U.S. EPA, 2011). **Table 5-4** characterizes the distributions of consumption rates for produce items and drinking water, as well as the distributions of body weights and exposure durations used in this analysis. **Table 5-5** identifies the exposure parameters, including soil ingestion, that were fixed at constant values in this analysis.

Table 5-4. Produce and Drinking Water Consumption Rate (CR), Body Weight, and Exposure Duration Distributions for the Home Gardener

Age	Distribution Type	Mean (or Shape)[a]	Std Dev (or Scale)[a]	Minimum	Maximum	Reference [a]
Exposed Fruit (g [WW] kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Gamma	1.43E+00	1.58E+00	0.00E+00	1.60E+01	U.S. EPA (2011); Table 13-58
Child 6–11yrs	Lognormal	2.78E+00	5.12E+00	0.00E+00	3.60E+01	
Child 12–19 yrs	Lognormal	1.54E+00	2.44E+00	0.00E+00	1.80E+01	
Adult (20–69 yrs)	Lognormal	1.57E+00	2.3E+00	0.00E+00	1.29E+01	
Exposed Vegetables (g [WW] kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Gamma	9.70E-01	2.62E+00	0.00E+00	2.10E+01	U.S. EPA (2011); Table l 13-60
Child 6–11yrs	Lognormal	1.64E+00	3.95E+00	0.00E+00	2.70E+01	
Child 12–19 yrs	Gamma	9.10E-01	1.19E+00	0.00E+00	1.10E+01	
Adult (20–69 yrs)	Weibull	1.57E+00	1.76E+00	0.00E+00	1.03E+01	
Protected Fruit (g [WW] kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Gamma	7.37E-01	1.59E+01	0.00E+00	4.50E+01	U.S. EPA (2011); Table 13-59
Child 6–11yrs	Gamma	7.37E-01	8.15E+00	0.00E+00	2.60E+01	
Child 12–19 yrs	Gamma	7.36E-01	3.56E+00	0.00E+00	3.80E+01	
Adult (20–69 yrs)	Lognormal	6.63E+00	1.57E+01	0.00E+00	4.73E+01	
Protected Vegetables (g [WW] kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Lognormal	1.88E+00	1.98E+00	0.00E+00	1.60E+01	U.S. EPA (2011); Table 13-61
Child 6–11yrs	Lognormal	1.07E+00	1.04E+00	0.00E+00	8.00E+00	
Child 12–19 yrs	Lognormal	7.70E-01	6.90E-01	0.00E+00	6.00E+00	
Adult (20–69 yrs)	Lognormal	1.01E+00	1.19E+00	0.00E+00	6.49E+00	
Root Vegetables (g [WW] kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Lognormal	2.31E+00	6.05E+00	0.00E+00	4.10E+01	U.S. EPA (2011); Table 13-62
Child 6–11yrs	Weibull	6.80E-01	1.06E+00	0.00E+00	1.50E+01	
Child 12–19 yrs	Weibull	8.40E-01	9.10E-01	0.00E+00	9.00E+00	
Adult (20–69 yrs)	Weibull	1.15E+00	1.32E+00	0.00E+00	7.47E+00	

Age	Distribution Type	Mean (or Shape)[a]	Std Dev (or Scale)[a]	Minimum	Maximum	Reference [a]
Drinking Water Ingestion (mL kg^{-1} body weight d^{-1})						
Child 1–5 yrs	Weibull	1.15E+00	2.56E+01	2.23E-03	1.86E+02	U.S. EPA (2008a); Table 3-19
Child 6–11yrs	Weibull	1.14E+00	1.75E+01	2.23E-03	1.86E+02	
Child 12–19 yrs	Weibull	1.08E+00	1.14E+01	2.23E-03	1.86E+02	
Adult (20–69 yrs)	Weibull	1.16E+00	1.66E+01	1.00E-02	1.26E+02	U.S. EPA (2011) Table 3-38
Body Weight (kg)						
Child 1–5 yrs	Lognormal	1.55E+01	2.05E+00	4.00E+00	5.00E+01	U.S. EPA (2011); Table 8-3
Child 6–11yrs	Lognormal	3.07E+01	5.96E+00	6.00E+00	2.00E+02	
Child 12–19 yrs	Lognormal	5.82E+01	1.02E+01	1.30E+01	3.00E+02	
Adult (20–69 yrs)	Lognormal	7.12E+01	1.33E+01	1.50E+01	3.00E+02	
Exposure Duration (yr)[b]						
Child (1-19 yrs)	Weibull	1.32E+00	7.06E+00	1.00E+00	3.80E+01	U.S. EPA (2011); Table 16-109
Adult (20–69 yrs)	Weibull	1.34E+00	1.74E+01	1.00E+00	5.00E+01[b]	

[a] Shape and scale are presented for Gamma and Weibull distributions.

[b] Exposure duration was capped at 50 years so it would never exceed the 70-year lifetime assumption implicit in the averaging time used, given the starting age of 20 years.

Table 5-5. Summary of Exposure Parameters with Fixed Values Used in Probabilistic Analysis

Parameter	Units	Constant Values	Reference
Averaging time for carcinogens	yr	7.00E+01	U.S. EPA (1991a)
Exposure frequency	d yr^{-1}	3.50E+02	U.S. EPA (1991a)
Fraction food preparation loss: exposed fruit	Fraction	2.10E-01	U.S. EPA (2011); Table 13-69
Fraction food preparation loss: exposed vegetables	Fraction	1.61E-01	
Fraction food preparation loss: protected fruit	Fraction	2.90E-01	
Fraction food preparation loss: protected vegetables	Fraction	1.30E-01	
Fraction food preparation loss: root vegetables	Fraction	5.30E-02	
Fraction contaminated: drinking water	Fraction	1.00E+00	U.S. EPA Policy
Fraction contaminated: soil	Fraction	1.00E+00	U.S. EPA Policy
Ingestion rate: soil (adult)	mg d^{-1}	5.00E+01	U.S. EPA (1997c); Table 5-1
Ingestion rate: soil (child 1, child 2, child 3)	mg d^{-1}	1.00E+02	

The conservative nature of the distributions used to estimate home gardener adult and child consumption rates could result in overly conservative consumption rates of home-grown produce. Two additional sets of runs were therefore added for comparison: one using point estimates of 50th percentile annual produce consumption rates for the general population, multiplied by 50% to account for crop growth periods and climate limitations to crop harvest periods (reducing the effective consumption rate to home-grown produce); and a set of runs using the 90th percentile annual produce consumption rates for the general population, similarly multiplied by 50%. All other distributions and constant values were the same. Thus, the three sets of runs are as follows:

- **Set 1:** Home gardener, modeled distributions of consumption rates (for home gardeners) —the produce consumption rates specific to home-grown produce;
- **Set 2:** General population, 50th percentile (for the general population) consumption rates —the median produce consumption rates for the general population were multiplied by 0.5 to derive a value specific to home-grown produce; and
- **Set 3:** General population, 90th percentile (for the general population) consumption rates —the high produce consumption rates for the general population were multiplied by 0.5 to derive a value specific to home-grown produce.

Table 5-6 identifies the 90th percentile home gardener produce consumption rates, and the general population median and high produce consumption rates that were used in the additional runs. Evaluation of the groundwater pathway did not require the development of different drinking water consumption rate datasets for each population type; it was assumed that the general population receptor and the home gardener receptors would ingest drinking water at consistent rates.

Table 5-6. Summary of Produce Consumption Rates (CR)
(g [WW] produce kg^{-1} body weight d^{-1})

	Home Gardener Estimates	General Population Estimates [a]	
Age	**90%-ile**	**Median**	**High**
Exposed Fruit			
Child 1–5 yrs	5.41	1.95	10.62
Child 6–11 yrs	6.98	1.10	3.15
Child 12–19 yrs	3.41	0.44*	1.45
Adult (20–69 yrs)	5.00	0.32*	1.06
Exposed Vegetables			
Child 1–5 yrs	6.43	0.32	2.48
Child 6–11 yrs	3.22	0.30	1.70
Child 12–19 yrs	2.35	0.27	1.25
Adult (20–69 yrs)	6.01	0.45	1.63
Protected Fruit			
Child 1–5 yrs	13.00	2.70	7.19
Child 6–11 yrs	6.92	0.17	4.05
Child 12–19 yrs	7.44	1.80	2.70
Adult (20–69 yrs)	15.00	0.93	2.09
Protected Vegetables			
Child 1–5 yrs	3.05	0.63*	1.93
Child 6–11 yrs	2.14	0.39*	1.30
Child 12–19 yrs	1.85	0.23*	0.75
Adult (20–69 yrs)	3.55	0.27*	0.85
Root Vegetables			
Child 1–5 yrs	5.72	0.72	3.01
Child 6–11 yrs	3.83	0.50	2.10
Child 12–19 yrs	2.26	0.41	1.50
Adult (20–69 yrs)	3.11	0.35	1.29

SOURCE: Values derived from EPA's *Exposure Factors Handbook* (U.S. EPA, 2011).

[a] The listed general population values are the general population consumption rates listed in U.S. EPA (2011) multiplied by 0.5 to derive a value specific to home-grown produce.

* Based on mean values.

5.3.7.3 Human Exposure Model Outputs

The outputs from the exposure model are receptor- and pathway-specific ADDs for constituents with noncancer endpoints, and LADDs for constituents with cancer endpoints. As discussed in Section 5.3.1, each model run generated an ADD/LADD for each of the exposure pathways (i.e., separate ADDs/LADDs for exposure from ingestion of soil, exposed fruits, exposed vegetables, etc). Each model run also combined the pathway-specific ADDs/LADDs into a "Total Ingestion" ADD/LADD.

Running the model probabilistically generated distributions of exposure values for each pathway, as well as a distribution of Total Ingestion values. **Table 5-7 thru 5-10** list pathway-specific and Total Ingestion values taken from example runs that generated the 50th and 90th percentile Total Ingestion values.

Table 5-7. Example 50th Percentile Adult Unitized Doses for SFS-Manufactured Soil Constituents—Total Ingestion Pathway (mg kg^{-1} d^{-1})

				General Population			
		Home Gardener		Median Consumption Rates		High Consumption Rates	
Constituent	**Pathway**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD**
Cancer							
As	Soil	4772	2.0E-08	8883	3.6E-08	7041	4.8E-08
	Protected Veg		1.1E-07		1.1E-08		3.6E-08
	Exposed Veg		9.2E-08		3.7E-08		1.4E-07
	Protected Fruit		5.8E-08		8.1E-09		3.8E-08
	Exposed Fruit		7.5E-08		9.1E-09		3.3E-08
	Root Veg		2.5E-08		2.3E-08		8.7E-08
	Total Ingestion		3.7E-07		1.2E-07		3.8E-07
	Groundwater	PI					
Noncancer							
Co	Soil	569	3.6E-07	5410	3.4E-07	509	2.3E-07
	Protected Veg		8.0E-07		3.1E-07		9.8E-07
	Exposed Veg		2.6E-06		5.9E-07		2.2E-06
	Protected Fruit		6.2E-07		2.2E-07		1.0E-06
	Exposed Fruit		8.4E-07		2.5E-07		8.4E-07
	Root Veg		2.7E-06		7.8E-07		3.0E-06
	Total Ingestion		7.9E-06		2.5E-06		8.2E-06
Fe	Soil	959	5.7E-07	1301	2.3E-07	7952	3.4E-07
	Protected Veg		1.3E-07		4.1E-08		1.2E-07
	Exposed Veg		4.5E-07		1.1E-07		3.7E-07
	Protected Fruit		5.3E-08		3.0E-08		1.3E-07
	Exposed Fruit		2.7E-07		3.4E-08		1.0E-07
	Root Veg		1.0E-07		1.5E-07		5.0E-07
	Total Ingestion		1.6E-06		5.9E-07		1.6E-06

PI = Pathway incomplete (constituent does not reach receptor well during simulation)

Table 5-8. Example 50th Percentile Child Unitized Doses for SFS-Manufactured Soil Constituents—Total Ingestion Pathway (mg kg^{-1} d^{-1})

Constituent	Pathway	Home Gardener		General Population			
				Median Consumption Rates		High Consumption Rates	
		RunID	ADD/ LADD	RunID	ADD/ LADD	RunID	ADD/ LADD
Cancer							
As	Soil	5114	1.7E-07	5208	1.8E-07	2701	1.7E-07
	Protected Veg		6.9E-08		1.6E-08		5.0E-08
	Exposed Veg		4.9E-08		1.6E-08		1.3E-07
	Protected Fruit		1.9E-07		1.5E-08		7.9E-08
	Exposed Fruit		3.2E-08		3.5E-08		1.8E-07
	Root Veg		2.9E-08		3.0E-08		1.3E-07
	Total Ingestion		5.4E-07		2.9E-07		7.4E-07
	Groundwater	PI					
Noncancer							
Co	Soil	495	5.4E-06	3059	3.5E-06	9733	4.3E-06
	Protected Veg		6.8E-07		6.9E-07		1.9E-06
	Exposed Veg		7.5E-06		4.0E-07		2.9E-06
	Protected Fruit		4.8E-07		6.3E-07		3.1E-06
	Exposed Fruit		1.7E-06		1.5E-06		7.6E-06
	Root Veg		8.3E-07		1.6E-06		6.0E-06
	Total Ingestion		1.7E-05		8.2E-06		2.6E-05
Fe	Soil	7672	3.7E-06	6883	3.0E-06	2508	2.8E-06
	Protected Veg		1.4E-07		9.3E-08		3.1E-07
	Exposed Veg		4.1E-07		8.5E-08		6.4E-07
	Protected Fruit		3.6E-07		7.8E-08		4.9E-07
	Exposed Fruit		3.8E-07		2.5E-07		1.2E-06
	Root Veg		5.6E-07		3.0E-07		1.3E-06
	Total Ingestion		5.5E-06		3.8E-06		6.8E-06

PI = Pathway incomplete (constituent does not reach receptor well during simulation)

Table 5-9. Example 90th Percentile Adult Unitized Doses for SFS-Manufactured Soil Constituents—Total Ingestion Pathway (mg kg^{-1} d^{-1})

		Home Gardener		General Population			
				Median Consumption Rates		High Consumption Rates	
Constituent	**Pathway**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD**
Cancer							
As	Soil	7831	1.6E-07	1770	8.5E-08	3447	6.6E-08
	Protected Veg		7.7E-08		5.2E-08		1.7E-07
	Exposed Veg		7.2E-07		1.7E-07		6.5E-07
	Protected Fruit		1.3E-07		3.8E-08		1.8E-07
	Exposed Fruit		8.7E-08		4.2E-08		1.5E-07
	Root Veg		6.7E-07		1.1E-07		4.1E-07
	Total Ingestion		1.8E-06		5.0E-07		1.6E-06
	Groundwater	9716	2.1E-07	Same as Home Gardener			
Noncancer							
Co	Soil	5661	9.4E-08	5260	6.8E-07	9534	4.9E-07
	Protected Veg		4.8E-07		3.2E-07		1.0E-06
	Exposed Veg		1.9E-08		6.1E-07		2.3E-06
	Protected Fruit		1.5E-05		2.3E-07		1.1E-06
	Exposed Fruit		1.1E-06		2.6E-07		9.5E-07
	Root Veg		9.4E-07		8.2E-07		3.1E-06
	Total Ingestion		1.8E-05		2.9E-06		8.9E-06
Fe	Soil	9766	4.4E-07	5677	5.7E-07	4181	5.5E-07
	Protected Veg		1.4E-07		4.7E-08		1.4E-07
	Exposed Veg		2.3E-06		1.4E-07		4.9E-07
	Protected Fruit		2.0E-07		3.4E-08		1.5E-07
	Exposed Fruit		1.1E-07		5.7E-08		1.8E-07
	Root Veg		7.8E-09		1.7E-07		6.0E-07
	Total Ingestion		3.2E-06		1.0E-06		2.1E-06

Table 5-10. Example 90th Percentile Child Unitized Doses for SFS-Manufactured Soil Constituents—Total Ingestion Pathway (mg kg^{-1} d^{-1})

				General Population			
		Home Gardener		**Median Consumption Rates**		**High Consumption Rates**	
Constituent	**Pathway**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD**	**RunID**	**ADD/ LADD)**
Cancer							
As	Soil	4734	3.2E-07	2116	2.2E-07	1692	1.4E-07
	Protected Veg		1.0E-07		3.7E-08		1.0E-07
	Exposed Veg		1.6E-07		6.1E-08		2.8E-07
	Protected Fruit		3.1E-07		3.0E-08		2.2E-07
	Exposed Fruit		1.8E-07		6.6E-08		2.5E-07
	Root Veg		6.5E-08		7.8E-08		2.8E-07
	Total Ingestion		1.1E-06		5.0E-07		1.3E-06
	Groundwater	4302	2.5E-07	Same as Home Gardener			
Noncancer							
Co	Soil	5049	6.8E-06	8674	6.8E-06	4005	6.2E-06
	Protected Veg		4.6E-06		6.7E-07		2.3E-06
	Exposed Veg		3.3E-06		4.3E-07		3.4E-06
	Protected Fruit		6.7E-07		5.1E-07		3.7E-06
	Exposed Fruit		2.9E-06		1.4E-06		9.0E-06
	Root Veg		1.2E-05		1.6E-06		7.1E-06
	Total Ingestion		3.1E-05		1.1E-05		3.2E-05
Fe	Soil	3020	4.0E-06	4792	6.0E-06	7537	6.1E-06
	Protected Veg		3.1E-07		1.0E-07		3.3E-07
	Exposed Veg		4.0E-06		8.5E-08		6.9E-07
	Protected Fruit		6.1E-08		9.6E-08		5.3E-07
	Exposed Fruit		5.4E-07		2.3E-07		1.3E-06
	Root Veg		6.0E-07		3.3E-07		1.4E-06
	Total Ingestion		9.6E-06		6.8E-06		1.0E-05

It is important to note that the pathway-specific values listed in Tables 5-7 thru 5-10 are those which, when totaled, result in the 50th (or 90th) percentile Total Ingestion ADD/LADD. Each pathway-specific value is not necessarily the 50th (or 90th) percentile value for that individual pathway. For example, in the distribution of child Total Ingestion LADDs for arsenic based on home gardener ingestion rates, the 50th percentile value (i.e., the Total Ingestion LADD at the exact center of the distribution) was generated in model run 5114 (see Table 5-8). This Total Ingestion LADD includes an LADD of 6.9E-08 mg kg^{-1} d^{-1} from ingestion of protected vegetables. However, in the distribution of child LADDs for arsenic specific to ingestion of protected produce, the 50th percentile LADD of 1.1E-08 mg kg^{-1} d^{-1} was generated in model run 8883. Pathway-specific 50th and 90th percentile ADDs/LADDs for adult and child receptors (including the probabilistic runs that generated them) are listed in Appendix K, Tables K-1 through K-4. Example Total Ingestion 50th and 90th percentile ADDs/LADDs for adult and child receptors, including their respective pathway-specific contributions and the probabilistic runs that generated them, are listed in Appendix K, Tables K-5 through K-8.

These ADDs/LADDs are used as input to the human health effects model, as discussed in **Section 5.3.10**.

5.3.8 Ecological Exposure Modeling

The following sections describe the ecological exposure modeling. **Section 5.3.8.1** provides an overview of the conceptual model, including the basic approach and assumptions. **Section 5.3.8.2** discusses the input parameters and values used in this risk analysis. **Section 5.3.8.3** discusses the model outputs.

5.3.8.1 Ecological Conceptual Exposure Model

As described in **Section 5.3.3**, ecological receptors could be exposed to SFS constituents via direct contact with soil. Depending on the receptor (i.e., plants, soil invertebrates, or small mammals), ecological exposure was estimated by adjusting the concentration of the constituent in soil to reflect the phyto-available fraction or the receptor's home range.

Exposure modeling relies heavily on default assumptions concerning population activity patterns, mobility, dietary habits, body weights, and other factors. For example, Phase I screening assumed that 100% of SFS-bound metals were bioavailable to ecological receptors. This assumes that SFS-bound metals are equally available to biological systems as soluble metal salts added to soils in laboratory studies. Phase I screening also assumed that animals received 100% of their diet from the home garden; they do not forage or feed beyond the boundaries of the garden. Both of these assumptions are upper bound estimates that are reasonable for a screening analysis.

One function of refined probabilistic modeling is to replace upper bound estimates with more realistic conservative inputs. The key assumptions that were applied in refined ecological exposure modeling include:

- Plants were grown in the home garden, and therefore 100% of the soil they were exposed to was SFS-manufactured soil. However, soil concentrations were adjusted to reflect the soluble, and therefore phyto-available, fraction of SFS constituents (see Section 5.3.8.2 for a more detailed discussion of this assumption).
- Soil invertebrates spend their entire lives in home garden soils.
- As a highly exposed species, the short-tailed shrew was the surrogate species used to derive the Eco-SSL for mammals, and evaluated for potential adverse impacts. Constituent soil concentrations were adjusted to reflect the fraction of shrew diet to come from the garden (see Section 5.3.8.2 for a more detailed discussion of this assumption).

5.3.8.2 Ecological Exposure Model Inputs

The inputs to the ecological exposure model are soil concentrations and ecological exposure factors. Estimation of soil concentrations is discussed in Section 5.3.4. The key ecological exposure factors used as inputs to the analysis include the following factors.

Plant Toxicity

Manganese and nickel were retained for further study in Phase II due to the potential for phyto-toxicity. Because the toxicity of metals is dependent on the soluble soil fraction, the risk posed to terrestrial plants will be directly related to the amount of metal that can desorb from

SFS particles and become available in the soluble fraction. In her review of plant responses to metal toxicity, Reichman (2002) noted that:

> *The total metal concentration of a soil includes all fractions of a metal, from the readily available to the highly unavailable. Other soil factors, such as pH, organic matter, clay and redox conditions, determine the proportion of total metal which is in the soil solution. Hence, while total metal provides the maximum pool of metal in the soil, other factors have a greater importance in determining how much of this soil pool will be available to plants (Wolt, 1994). In addition, researchers have found that while total metal correlates with bioavailable soil pools of metal, it is inadequate by itself to reflect bioavailability (Lexmond, 1980; Sauve et al., 1996; McBride et al., 1997; Sauve et al., 1997; Peijnenburg et al., 2000).*

Lacking empirical data on the soluble fraction of metals in SFS-amended soil, this evaluation used SFS sample-specific pore water concentrations as a surrogate to develop estimates of the soluble (and therefore bioavailable) fraction in soil. This approach defines the constituent-specific bioavailable fractions as the ratio of SFS sample-specific pore water concentrations to corresponding total concentrations (see Appendix B Tables B-26 and B-19). The empirical distributions of the "pore water/total" ratios establishes a reasonable range for the bioavailable fraction. The 95th percentile of the ratio range (i.e., an estimate of the bioavailable fraction that is higher than 95 percent of other estimates) was used as a reasonably conservative estimate of the bioavailable fraction. Therefore, the maximum soil concentrations for manganese and nickel would be adjusted by a fraction of 0.10 and 0.07, respectively. In effect, this adjustment estimates that the majority of manganese and nickel is in a solid form unavailable for plant uptake. That is, only a fraction of the metals found in SFS-amended soil behaves similarly to the metals added in spiked soil studies (e.g., soluble metal salts).

Dietary Exposure to Mammals

Antimony, chromium, and copper were retained for further study in Phase II due to the potential for toxicity to small insectivorous mammals (based on studies for the short tailed shrew). The area of the home garden (i.e. 405 m^2) may be substantially less than the home range for the shrew. In developing the ecological risk assessment methodology for 3MRA, EPA determined that it was reasonable to prorate exposures based on a comparison between the "habitat" (i.e., the area in which the material is managed – the home garden in the SFS evaluation), and the median home range for the animal so that dietary exposure was not grossly overestimated. This methodology was reviewed and approved by EPA's Science Advisory Board in 2003, as a reasonable method to account for the spatial heterogeneity in animals' use of feeding and foraging areas.[44] The same method is used in this risk assessment to avoid the unrealistic and overly conservative assumption that 100% of the shrew diet comes from the home garden.

Information on home ranges of species was reviewed for northern, southern, Adirondack, Sherman's, and Elliot's short-tailed shrews (ADCNR, 2008; FFWCC, 2013; Getz and McGuire, 2008; KBS, 2014; MNHP, 2014; Saunders, 1988; U.S. EPA, 1993 and 2002; VDGIF, 2014). The short-tailed shrew diet consists primarily of insects, earthworms, slugs, and snails, while plants,

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[44] The SAB review report is available at http://yosemite.epa.gov/sab/sabproduct.nsf/95eac6037dbee075852573a00075f732/99390efbfc255ae885256ffe00579745/$FILE/SAB-05-003_unsigned.pdf

fungi, millipedes, centipedes, arachnids, and small mammals also are consumed (U.S. EPA, 1993b). The literature on short-tailed shrews noted that these animals can be found in a wide variety of habitats, although areas with litter/grass cover (e.g., forest, wetlands) and high moisture levels are clearly preferred (Miller and Getz, 1977; van Zyll de Jong, 1983). A variety of factors that influence the home range and habitat preference for short-tailed shrews were identified; for example, the availability of prey, season, and reproductive status were shown to influence movement and home ranges for short-tailed shrews in east-central Illinois (Getz and McGuire, 2008). **Figure 5-7** presents the median home range values identified in that review, ranging from 0.06 to 6.2 acres with a median (of the medians) of 2.4 acres (9700 m^2), and a 10th percentile value of 0.7 acres (2800 m^2). The variability in results shown in Figure 5-7 suggests that the species, as well as the geographical location, has a significant influence on the home range and movement (a surrogate for foraging behavior) for the short-tailed shrew.

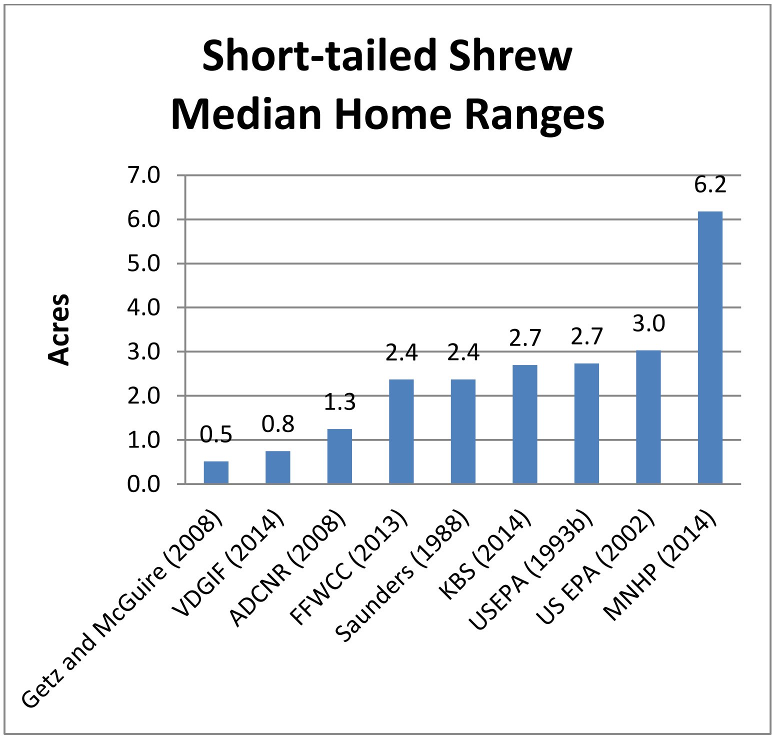

Figure 5-7. Analysis of Home Range Sizes for the Short Tailed Shrew.

Comparing the home garden area of 0.1 acres (405 m^2) to the 10th percentile value for home ranges shown in Figure 5-7, 0.7 acres (2800 m^2) attributes roughly 15% of the short-tailed shrew diet to the home garden. As a consequence, a fraction of 0.15 was assumed for all three COCs to reflect the percentage of diet likely to come from the home garden.

5.3.8.3 Ecological Exposure Model Outputs

The outputs from the ecological exposure model are distributions of predicted receptor- and constituent-specific soil concentrations adjusted to reflect bioavailability and mammal home range. **Table 5-11** lists the 50th and 90th modeled soil concentrations, adjustment factors, and adjusted soil concentrations.

Table 5-11. 50th and 90th Percentile Ecological Exposure Model Outputs for SFS-Manufactured Soil Constituents

	Terrestrial Plants			Soil Invertebrates			Mammals		
Constituent	**Modeled Soil Conc. (mg kg^{-1})**	**Adjustment Factor (unitless)**	**Adjusted Soil Conc. (mg kg^{-1})**	**Modeled soil conc. (mg kg^{-1})**	**Adjustment Factor (unitless)**	**Adjusted Soil Conc. (mg kg^{-1})**	**Modeled soil conc. (mg kg^{-1})**	**Adjustment Factor (unitless)**	**Adjusted Soil Conc. (mg kg^{-1})**
50th percentile									
Cr (III)	—	—	—	—	—	—	0.94	0.15	0.14
Cu	0.90	NA	0.90	0.90	NA	0.90	0.90	0.15	0.13
Mn	0.93	0.10	0.093	0.93	NA	0.93	0.93	NA	0.93
Ni	0.92	0.07	0.064	0.92	NA	0.92	0.92	NA	0.92
Sb	—	—	—	0.82	NA	0.82	0.82	0.15	0.12
90th percentile									
Cr (III)	—	—	—	—	—	—	0.98	0.15	0.15
Cu	0.97	NA	0.97	0.97	NA	0.97	0.97	0.15	0.15
Mn	0.97	0.10	0.097	0.97	NA	0.97	0.97	NA	0.97
Ni	0.97	0.07	0.068	0.97	NA	0.97	0.97	NA	0.97
Sb	—	—	—	0.96	NA	0.96	0.96	0.15	0.14

The adjusted soil concentrations are used as input to the ecological effects model described in **Section 5.3.10**.

5.3.9 Human Health Effects Modeling

This section presents the human health benchmarks and the modeling approach used to estimate potential health hazards. **Section 5.3.9.1** provides an overview of the conceptual model, including the basic approach and assumptions. **Section 5.3.9.2** discusses the input parameters and values used in this hazard analysis. **Section 5.3.9.3** discusses the model outputs. The hazard equations used in the human health effects modeling are presented in **Appendix H**.

5.3.9.1 Human Health Effects Conceptual Model

Human health effects modeling was performed to estimate cancer and noncancer health impacts due to ingestion of soil and home-grown produce. A chemical constituent's ability to cause an adverse health effect depends on the toxicity of the particular constituent, the route of exposure, the duration and intensity of exposure, and the resulting dose that an individual receives. The human health benchmarks used in this assessment were compared to the ADD for noncarcinogens or the LADD for carcinogens. For constituents with noncancer endpoints, the health benchmark was the RfD. For constituents with cancer endpoints, the health benchmark was the dose that yields a cancer risk level of 10^{-5} (1 in 100,000) over a lifetime (calculated as 10^{-5}/oral cancer slope factor [CSF]). The ratio of the ADD or LADD to the health benchmark (shown below) is referred to as a Unitized Dose Ratio (UDR) and was used to establish a threshold of concern for a specific health effect. The level of concern established by EPA for this analysis is a UDR of 1.

$$Unitized\ Dose\ Ratio = \frac{ADD\ or\ LADD}{Health\ Benchmark}$$

where

Unitized Dose Ratio = Comparison of exposure dose to benchmark dose (unitless)

For noncarcinogens:

ADD = Average daily dose (mg kg^{-1} d^{-1})

Health Benchmark = RfD (mg kg^{-1} d^{-1}).

For carcinogens:

LADD = Lifetime average daily dose (mg kg^{-1} d^{-1})

Health Benchmark = Cancer risk level of 10^{-5}/oral CSF (mg kg^{-1} d^{-1}).

Although some constituents such as manganese elicit similar toxicological responses (e.g., neurotoxicity) via different exposure pathways, the modeling stages of the analysis did not consider cumulative exposures or impacts. The exposure scenarios and pathway evaluations were developed and parameterized to produce conservative risk estimates; that is, the methodology was designed to overestimate the actual risk to ensure that an ample margin of safety was built into the analysis.

5.3.9.2 Human Health Model Inputs

Inputs to the human health effects model include estimates of toxicity (the human health benchmarks) and exposure doses. The estimation of exposure dose is discussed in **Section 5.3.7**. The human health benchmarks used as input to the model are discussed below.

Human health benchmarks for chronic exposures were used in this analysis to characterize the potential cancer and noncancer hazards associated with the use of SFS-manufactured soil in residential gardens. Oral CSFs and RfDs were used to estimate the cancer and noncancer hazards from oral exposures, respectively.

The CSF is an upper-bound estimate (approximating a 95% confidence limit) of the increased human cancer risk from a lifetime of exposure to an agent. This estimate is usually expressed in units of proportion (of a population) affected per mg of agent per kg body weight per day (per (mg kg^{-1} d^{-1})). Unlike RfDs, CSFs relate levels of exposure to a probability of developing cancer.

The RfD is the primary benchmark used to evaluate noncarcinogenic hazards posed by environmental exposures to chemical constituents. The RfD is an estimate of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious noncancer effects during a lifetime (U.S. EPA, 2012a). However, an average lifetime exposure above the RfD does not imply that an adverse health effect would necessarily occur.

The chronic human health benchmarks used in the Phase II analyses are summarized in **Table 5-12**. This table provides the constituent's name, Chemical Abstract Service Registry Number (CASRN), RfD (in units of mg kg^{-1} d^{-1}) and oral CSF (CSFo) [per (mg kg^{-1} d^{-1})], if available. Health benchmarks for arsenic are from EPA's Integrated Risk Information System (IRIS, U.S. EPA, 2012a), which is EPA's official electronic repository of chronic human health benchmarks and represents EPA-wide consensus on critical human health effects associated with exposure to chemical constituents released into the environment (U.S. EPA, 2012a). Benchmarks in IRIS have been extensively reviewed, and each file contains descriptive and quantitative information on potential health effects associated with the benchmark and other studies evaluated during its derivation.

The health benchmarks for cobalt and iron are Provisional Peer-Reviewed Toxicity Values (PPRTVs). The second tier of human health toxicity values in the OSWER toxicity value hierarchy (USEPA, 2003a), PPRTVs are derived when such values are not available in IRIS. PPRTVs are derived after a review of the relevant scientific literature using the methods, data sources and guidance for value derivation used by the EPA IRIS Program. All PPRTVs receive internal review by EPA scientists and external peer review by independent scientific experts. PPRTVs differ in part from IRIS values in that PPRTVs do not receive the multi-program consensus review provided for IRIS values. This is because IRIS values are generally intended to be used in all EPA programs, while PPRTVs are developed specifically for the Superfund and RCRA programs.

Table 5-12. Human Health Benchmarks Used in Phase II Analysis

Constituent	CASRN	RfD (mg kg^{-1} d^{-1})	CSF (per mg kg^{-1} d^{-1})
As [a]	7440382	0.0003	1.5
Co [b]	7440484	0.0003	—
Fe [b]	7439896	0.7	—

[a] SOURCE: IRIS (U.S. EPA, 2012a)

[b] SOURCE: PPRTV (U.S. EPA, 2014)

5.3.9.3 Health Model Outputs

The human health effect model generated a distribution of Unitized Dose Ratio estimates (UDRs) for adult and child receptors and each exposure pathway, as well as aggregates for the soil exposure pathways (titled “Total Ingestion” reflecting exposures through incidental soil and ingestion of produce). Analyses discussed in **Section 5.3.5.3** and **Appendix J** indicate that exposures via groundwater will not occur within the same timeframe as exposures via soil pathways. Consequently, UDRs for soil and groundwater pathways were not combined. Rather, the individual, pathway-specific UDRs were used to develop separate pathway-specific SFS screening levels.

As discussed in **Section 5.3.9.1**, the UDRs represent a ratio of the ADD (or LADD) and the health benchmarks listed in Table 5-12. Any UDR less than one equates to estimates below the health benchmark. As discussed in **Section 5.3.7.2**, three separate sets of model runs were performed: the first set produced home gardener exposure estimates using consumption rates based on distributions from the EFH (U.S. EPA, 2011) and CSEFH (U.S. EPA, 2008a); sets 2 and 3 produced exposure estimates based on constant values for general population median and high-end annual consumption rates assuming that no more than 50% of the produce consumed was grown on the home garden. The 50th and 90th percentile UDRs from the two sets of general population runs were then compared to the 50th and 90th percentile UDRs from the set of home gardener runs. **Table 5-13** lists the 50th and 90th percentile Total Ingestion ADD/LADDs, health benchmarks, and UDRs for each adult receptor (home gardener, general population median consumption rate, and general population high consumption rate). **Table 5-14** lists parallel information for the child receptor. Detailed 50th and 90th percentile values for adult and child receptors for all pathways are listed in **Appendix K**. Tables 5-13 and 5-14 also present information on arsenic exposure for the groundwater pathway. However, in the case of the 50th percentile groundwater UDR a value of “PI” is reported indicating that the constituent did not reach the receptor well during the simulation.

The UDRs in Tables 5-13 and 5-14 were used to estimate SFS-specific screening concentrations, as discussed in **Section 5.3.11**.

Table 5-13. 50th and 90th Percentile Adult Unitized Dose Ratios for SFS-Manufactured Soil Constituents

						General Population					
			Home Gardener			**Median Consumption Rates**			**High Consumption Rates**		
Constituent	**Health Benchmark** [a]	**Pathway**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**
50th Percentile											
Cancer											
As	6.67E-06 (CSF based)	Soil/Produce	4772	3.7E-07	0.056	8883	1.2E-07	0.019	7041	3.8E-07	0.057
		Groundwater	PI								
Noncancer											
Co	0.0003 (RfD)	Soil/Produce	569	7.9E-06	0.026	5410	2.5E-06	0.0083	509	8.2E-06	0.027
Fe	0.7 (RfD)	Soil/Produce	959	1.6E-06	2.2E-6	1301	5.9E-07	8.5E-7	7952	1.6E-06	2.2E-6
90th Percentile											
Cancer											
As	6.67E-06 (CSF based)	Soil/Produce	7831	1.8E-06	0.28	1770	5.0E-07	0.074	3447	1.6E-06	0.24
		Groundwater	9716	2.1E-07	0.031	Same as Gardener					
Noncancer											
Co	0.0003 (RfD)	Soil/Produce	5661	1.8E-05	0.058	5260	2.9E-06	0.0097	9534	8.9E-06	0.030
Fe	0.7 (RfD)	Soil/Produce	9766	3.2E-06	4.6E-06	5677	1.0E-06	1.4E-06	4181	2.1E-06	3.0E-6

[a] Health Benchmark = RfD (mg kg^{-1} d^{-1}) for noncancer risk and 10^{-5}/oral CSF (per mg kg^{-1} d^{-1}) for cancer risk.
PI = Pathway incomplete (constituent does not reach receptor well during simulation)

Table 5-14. 50th and 90th Percentile Child Unitized Dose Ratios for SFS-Manufactured Soil Constituents

			Home Gardener			General Population					
						50%-ile Consumption Rate			90%-ile Consumption Rate		
Constituent	**Health Benchmark [a]**	**Pathway**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**	**RunID**	**ADD/ LADD (mg kg^{-1} BW d^{-1})**	**Unitized Dose Ratio (unitless)**
50th Percentile											
Cancer											
As	6.67E-06 (CSF based)	Soil/Produce	5114	5.4E-07	0.081	5208	2.9E-07	0.044	2701	7.4E-07	0.11
		Groundwater	PI								
Noncancer											
Co	0.0003 (RfD)	Soil/Produce	495	1.7E-05	0.055	3059	8.2E-06	0.027	9733	2.6E-05	0.086
Fe	0.7 (RfD)	Soil/Produce	7672	5.5E-06	7.9E-06	6883	3.8E-06	5.4E-6	2508	6.8E-06	9.7E-06
90th Percentile											
Cancer											
As	6.67E-06 (CSF based)	Soil/Produce	4734	1.1E-06	0.17	2116	5.0E-07	0.075	1692	1.3E-06	0.19
		Groundwater	4302	2.5E-07	0.037	Same as Gardener					
Noncancer											
Co	0.0003 (RfD)	Soil/Produce	5049	3.1E-05	0.10	8674	1.1E-05	0.038	4005	3.2E-05	0.11
Fe	0.7 (RfD)	Soil/Produce	3020	9.6E-06	1.4E-5	4792	6.8E-06	9.7E-06	7537	1.0E-05	1.5E-05

[a] Health Benchmark = RfD (mg kg^{-1} d^{-1}) for noncancer risk and 10^{-5}/oral CSF (per mg kg^{-1} d^{-1}) for cancer risk.
PI = Pathway incomplete (constituent does not reach receptor well during simulation)

In all cases, UDRs based on home gardener consumption rates were higher than estimates based on the general population median consumption rates, for both the adult and child. On the other hand, at the 50th percentile of all model runs – as summarized in Tables 5-13 and 5-14 – UDRs based on home gardener consumption rates were often lower than UDRs based on general population high consumption rates, for both the adult and child. This is likely because home gardener consumption rates varied with each model run (i.e. the consumption rate probability distributions in Table 5-4 were sampled for each run, generating run-specific consumption rates) and reflect consumption rates from across the entire range, whereas the general population consumption rates were constrained at the high end of the range.

At the 90th percentile of all model runs, home gardener UDRs were almost always higher than general population high consumption rate UDRs for both adult and child. For arsenic, the home gardener child UDR was slightly lower than the general population high consumption rate child UDR (i.e., 0.17 and 0.19, respectively).

5.3.10 Ecological Effects Modeling

Based on the conceptual model used for SFS in manufactured soil identified in **Chapter 3** and depicted in Figure 5-6, this assessment evaluated the potential for adverse impacts to plants, animals and soil invertebrates from the use of SFS in manufactured soil.

5.3.10.1 Conceptual Ecological Effects Model

This screening ecological assessment evaluated only direct contact with soil. Ecological risk was expressed in terms of risk ratios. Risk ratios were calculated as the ratio of the maximum soil concentration to the relevant SSL. For example, the risk ratio for soil invertebrates was calculated as the ratio of the soil concentration to the soil invertebrate SSL.

5.3.10.2 Ecological Effects Model Inputs

The inputs to the ecological effects model for direct contact are surficial soil concentrations and ecological health benchmarks. Estimation of soil concentrations is discussed in **Section 5.3.4**. **Table 5-15** presents EPA's Ecological SSLs (Eco-SSLs)[45] that were used, with maximum soil concentrations, to calculate the constituent-specific HQs for terrestrial plants and soil invertebrates.

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[45] Developed by EPA's Superfund program, Eco-SSLs are concentrations of contaminants in soil that are protective of ecological receptors that commonly come into contact with soil or ingest biota that live in or on soil. These values can be used to identify those contaminants of potential concern in soils requiring further evaluation in a baseline ecological risk assessment. Although these very conservative screening levels were developed specifically to be used during the Superfund ecological risk assessment process, EPA envisions that any federal, state, tribal, or private environmental assessment can use these values to screen soil contaminants to determine if additional ecological study is warranted (U.S. EPA, 2005c).

Table 5-15. Eco-SSLs Used in Phase II Analysis (mg kg^{-1} soil)

Constituent	CASRN	Terrestrial Plants	Soil Invertebrates	Mammals
Cr(III)	16065831	—	—	34
Cu	7440508	70	80	49
Mn	7439965	220	450	4000
Ni	7440020	38	280	130
Sb	7440360	—	78	0.27

5.3.10.3 Ecological Effects Model Outputs

The ecological effects model generates distributions of constituent-specific Unit Dose Ratios. As discussed in **Section 5.3.10.1**, these values represent a ratio of the modeled exposure value and the ecological health benchmarks listed in Table 5-15. Any UDR less than one equates to exposure estimate below the benchmark. As listed in **Table 5-16** and discussed in **Section 5.3.11**, values representing the 50th and 90th percentiles of these UDR distributions were used to estimate risk-based SFS-specific ecological screening concentrations.

Table 5-16. 50th and 90th Percentile Ecological Unitized Dose Ratios for SFS-Manufactured Soil Constituents

	Terrestrial Plants			Soil Invertebrates			Mammals		
Constituent	Adjusted Soil Conc. (mg kg^{-1})	Eco-SSL (mg kg^{-1})	UDR (unitless)	Adjusted Soil Conc. (mg kg^{-1})	Eco-SSL (mg kg^{-1})	UDR (unitless)	Adjusted Soil Conc. (mg kg^{-1})	Eco-SSL (mg kg^{-1})	UDR (unitless)
50th percentile									
Cr (III)	—	—	—	—	—	—	0.14	34	0.0041
Cu	0.90	70	0.013	0.90	80	0.011	0.13	49	0.0027
Mn	0.093	220	0.00042	0.93	450	0.0021	0.93	4000	0.00023
Ni	0.064	38	0.0017	0.92	280	0.0033	0.92	130	0.0071
Sb	—	—	—	0.82	78	0.010	0.12	0.27	0.45
90th percentile									
Cr (III)	—	—	—	—	—	—	0.15	34	0.0043
Cu	0.97	70	0.014	0.97	80	0.012	0.15	49	0.0030
Mn	0.097	220	.00044	0.97	450	0.0022	0.97	4000	0.00024
Ni	0.068	38	0.0018	0.97	280	0.0035	0.97	130	0.0075
Sb	—	—	—	0.96	78	0.012	0.14	0.27	0.53

5.3.11 Calculating Modeled SFS-Specific Screening Levels

Health model outputs compare health benchmarks to exposure estimates assuming a starting constituent concentration in SFS-manufactured soil of 1 mg constituent in one kilogram of soil on a wet weight basis. The home garden conceptual model assumes a soil recipe that includes 50% SFS. Therefore, SFS-manufactured soil UDRs listed in Tables 5-13 and 5-14 were converted to modeled SFS-specific screening concentrations using the following equation:

$$Screening\ Conc_{SFS} = \frac{2}{Unit\ Dose\ Ratio} \times 1.1$$

where:

Screening $Conc_{SFS}$ = Concentration of the constituent in SFS unlikely to cause adverse effect (mg kg^{-1} SFS dry weight)

Unit Dose Ratio = exposure dose to health benchmark (unitless, based on a starting soil concentration in mg kg^{-1} wet weight)

2 = 1/SFS fraction of manufactured soil (unitless).

1.1 = Factor for converting from wet weight to dry weight reflecting average modeled solids content of 90 percent (10 percent moisture) (unitless).[46]

Table 5-17 lists the SFS screening values protective of human health. These values represent the concentration of the constituent that could be found in SFS and not exceed the health benchmark.

Table 5-17. Modeled SFS-specific Screening Levels for the Home Garden Scenario
(mg kg^{-1} SFS)

	Adult			Child		
		General Population			General Population	
Constituent	**Home Gardener**	**Median Consumption Rates**	**High Consumption Rates**	**Home Gardener**	**Median Consumption Rates**	**High Consumption Rates**
Soil/Produce Pathway						
As	8.0	30	9.1	13	30	12
Co	38	230	74	22	58	21
Fe (g kg^{-1})	480	Capped	730	160	230	150
Groundwater Pathway						
As	71			59		

Capped = Calculated SFS-specific screening level would allow SFS to be 100% Fe, so value capped.

Table 5-18 lists the SFS screening values protective of ecological receptors. Appendix L presents the 50th and 90th percentile values and their corresponding soil concentrations. These values represent constituent concentrations that could be found in SFS and not exceed the ecological health benchmark.

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[46] As required by the source model, chemical-specific concentrations are input on a wet weight basis as a mass concentration. Noting that the SFS concentrations are similarly mass concentration–based, except that they are expressed on a *dry* weight basis, it is necessary to account for the modeled solids content.

Table 5-18. Modeled SFS-specific Ecological Screening Levels for the Home Garden Scenario (mg kg^{-1} SFS)

Constituent	Terrestrial Plants		Soil Invertebrates		Mammals	
	50%-ile	90%-ile	50%-ile	90%-ile	50%-ile	90%-ile
Cr(III)	—	—	—	—	530	510
Cu	170	160	200	180	800	740
Mn	5200	5000	1,100	1,000	9,500	9,000
Ni	1300	1200	670	630	310	290
Sb	—	—	210	179	4.8	4.1

5.3.12 Results: Comparing Screening Values to SFS Constituent Concentrations

Table 5-19 compares SFS constituent concentrations to the lowest human health–based SFS-specific screening values, as well as the ecological SFS-specific screening values, derived in **Section 5.3.11**. For each constituent, the human health–based value is the lower of the adult or child screening values. Likewise, the listed ecological health-based value is the lowest of the plant, soil invertebrates, or mammal screening values.

Table 5-19. Comparing SFS Constituent Concentrations to Modeled SFS-Specific Screening Levels (mg kg^{-1} SFS)

Constituent	SFS 95%-ile Concentration	Modeled SFS-Specific Screening Levels			
		Home Gardener	General Population		Ecological
			Median Consumption Rates	High Consumption Rates	
As	6.44	8.0	30	9.1	- -
Co	5.99	22	58	21	- -
Cr	109	- -	- -	- -	510
Cu	107	- -	- -	- -	160
Fe (g kg^{-1})	57.1	160	230	150	- -
Mn	670	- -	- -	- -	1,000
Ni	102	- -	- -	- -	290
Sb	1.23	- -	- -	- -	4.1

- - = Constituent was screened out in Phase I and did not require modeling for this receptor.

The SFS concentrations of all eight modeled constituents fell below their respective human and ecological modeled SFS-specific screening levels.

Modeling results are specific to the assumptions used in the modeling, and should be understood within the context of the complexity of the environmental conditions they represent. **Chapter 6** discusses the various lines of evidence described in the report, including the modeling results presented above and the information provided by them, as well as the uncertainties in and limitations of the analysis.

6. Risk Characterization

Chapter 2 introduced the current state of research on the origins, characteristics, and behavior of SFS in soil. **Chapters 3**, **4**, and **5** detailed the screening and modeling steps undertaken to assess the potential for human and ecological health impacts from soil-related uses of SFS: the results from quantitative evaluation of SFS-manufactured soil in home gardens would also apply to SFS use in soil-less potting media and use in road subbase. The results of these various efforts represent lines of evidence.

EPA's *Risk Characterization Handbook* (U.S. EPA, 2000) states that a risk characterization "integrates information from the preceding components… and synthesizes an overall conclusion about risk that is complete, informative, and useful for decision makers." This chapter provides the risk characterization for the evaluation. This chapter first discusses overarching concepts, such as the conservative nature of the risk screen used and the complexities of soil science. This information is then integrated with the results of the risk evaluation to provide a summary of the potential for human health and environmental impacts.

As discussed in **Chapter 2** of this report, generating industries, consumers, and regulatory agencies need to be confident that the scientific basis for making beneficial use decisions on SFS provides a high degree of certainty that potential risks to human health and the environment have been thoroughly evaluated. To address this need, the human health risk analysis was specifically designed to focus on the upper end of the distribution of risk to individuals that could potentially be exposed to SFS constituents because they (1) live near soil manufacturing facilities that include SFS among their soil recipes; (2) live near roadway construction projects that use unencapsulated SFS as a subbase for roads; or (3) use manufactured soil containing SFS in home gardens. In the *Guidance for Risk Characterization* developed by EPA's Science Policy Council (U.S. EPA, 1995c), EPA defined the high end of the risk distribution as being at or above the 90th percentile risk or hazard estimate generated during the Monte Carlo simulation.

Similarly, the ecological risk analysis focused on receptors that are in direct contact with the soil media and the potential for food web exposures specific to the area of use. This is particularly conservative because small perturbations and stresses to a field that represents a small fraction of the landscape may not be significant from either an ecological or societal perspective. Therefore, the portion of this report that addresses the potential for adverse effects to ecological receptors is also conservative and should be considered as a high-end approach analogous to the human health risk analysis.

With the conservative nature of the analysis in mind, **Section 6.1** provides an overview of the risk characterization by describing how a lines-of-evidence approach has been used to organize the information on modeling and scientific research.

6.1 Overview of the Risk Characterization

The goal of this evaluation was to determine whether SFS used in certain soil-related applications will be protective of human health and the environment. This assessment defines "protective" in terms of specific cancer risk (not to exceed an incremental risk of 10^{-5}, or 1 in 100,000) and noncancer risk for human and ecological receptors (not to exceed a threshold dose or concentration). There are many ways to present information relevant to these goals, all of which would satisfy the requirements of a risk characterization. However, given the complexity of risk-related issues surrounding the use of SFS (e.g., the relevance of comparing background metal content of soil to SFS metal content), as well as the need to integrate the screening modeling results with research on SFSs, soil chemistry, and toxicity, the most effective way to create transparency in this section was to begin at a high level by laying out a series of risk assessment questions, and then work through the analysis, ultimately presenting this information at the constituent level of detail. As discussed later in this chapter, the use of available scientific research on SFS and SFS constituent behavior and toxicology is critical to the interpretation of the screening modeling results. All models are simplifications of reality, and although they are extremely useful tools for predictive risk assessment, the modeling results should be considered in conjunction with the science of chemical behavior in the environment as it relates to exposure and, ultimately, risk. The remainder of this chapter is organized as follows:

- **Section 6.2, Key Risk Assessment Questions.** This section presents key risk assessment questions that pertain to certain soil-related beneficial uses of SFS. These questions are presented and discussed at a level that is intended to be accessible to risk managers, and they provide the context for the entire risk characterization. These questions may be tracked through all of the subsequent sections of the risk characterization.
- **Section 6.3, Overarching Concepts**. This evaluation is unique in that it deals with the beneficial use of a material and needs to address several technical issues. Because these issues are important to the interpretation of the risk modeling results and affect more than one SFS constituent, this section describes these concepts as a prelude to the more detailed elements of the risk characterization that follow.
- **Section 6.4, SFS Product Risks.** This section reviews the qualitative and semi-quantitative information on SFS as a material that may be beneficially used. It is important to understand what is known and what issues should be considered when interpreting the scientific research and screening-level modeling results.
- **Section 6.5, PAHs, Dioxins, Furans, and Dioxin-like PCBs in SFS.** PAHs, dioxins, furans, and dioxin-like PCBs constitute major groups of chemical constituents that have been quantified above detection limits in SFS. In some cases, these constituents have been addressed in risk assessments of other materials, such as dioxins in biosolids. The results of these risk assessments are clearly relevant to the interpretation of information specific to SFS; however, differences in exposure scenarios, modeling assumptions, the constituent-specific matrix, and other determinants of risk should be carefully considered when comparing the results of a risk assessment of those other materials to the SFS risk assessment. Therefore, this section will consider both the interpretation of other risk assessments, as well as the information and screening results developed in this report.
- **Section 6.6, Phenolics in SFS.** Although most phenolics were below detection limits, some have been found above detection limits in SFS (e.g., phenol, 2,4- dimethylphenol, 2-methylphenol). These compounds were evaluated as part of this risk assessment. This section presents the risk assessment modeling results and discusses the potential for adverse effects on human health and the environment associated with phenolics above detection limits in SFS.

- **Section 6.7, Metals and Metalloids in SFS.** Because of their persistence and potential toxicity in the environment, metals represent a critical group of chemical constituents found in SFS. A wide range of metals have been found above detection limits, and given the complexity of metals' behavior in soil systems and critical science-policy issues (e.g., background soil levels), this section presents a detailed lines-of-evidence determination for each metal constituent of concern. In addition to presenting the modeling results for various exposure pathways and scenarios, this section integrates scientific research on metals' behavior and toxicity and discusses whether this information (1) indicates that the results are conservative, and (2) suggests that an exposure pathway could *not* be completed at levels of concern because of natural obstacles, such as the soil-plant barrier.
- **Section 6.8, Uncertainty Characterization.** This section presents and discusses the data gaps and major sources of uncertainty in this risk assessment, focusing again on the overall goal to ensure that soil-related applications of SFS will not pose risks to human health and ecological receptors above levels of concern. Therefore, this section does *not* provide detailed information on modeling; that aspect of the risk assessment was designed to be conservative, and the bias inherent in data inputs and scenario assumptions is in the direction of overestimating risk. This section discusses the uncertainties from a decision-maker's perspective; that is, it examines whether or not the uncertainties in this risk assessment either (1) support or discourage the use of SFS in soil-related activities, or (2) require additional research to improve the quality of the information.

6.2 Key Risk Assessment Questions

To ensure that this report provides a high level of confidence, it is important to articulate the key risk assessment questions that this analysis was designed to address:

- Will the addition of SFSs to soil result in an increase in the constituent concentrations in soil relative to background levels, and how should the results of the risk assessment be interpreted across varied national soils?
- How do constituent forms found in the SFS matrix behave with respect to bioaccessibility and bioavailability, and how does that affect potential risks?
- How will the behaviors of individual constituents in SFS-manufactured soil, such as the soil-plant barrier, impact the potential for exposure through the food chain pathway and, ultimately, the potential for adverse human health and ecological effects?
- How do the risk assessment results compare to levels required to maintain nutritional health in plants and animals? Do issues of essentiality suggest that the predicted risks to plants and animals overestimate the potential for adverse effects?

6.3 Overarching Concepts

6.3.1 Background Concentrations

The components used to create metalcasting molds are not anthropogenically derived, but are obtained from the natural environment. Sands are either mined from terrestrial or aquatic (e.g., lakebeds) environments, while phyllosilicate clays (bentonites) are mined from terrestrial environments. A typical green sand contains as much as 90% sand, 5–10% clay, 5%

carbonaceous material (e.g., seacoal, cellulose), and 2–5% water by weight. These mold components, like soils, contain a variety of trace metals at concentrations found in native soils.

Soils themselves contain metals because they are composed of weathered rock and minerals (e.g., sand, clay) and decomposed plant and animal debris. However, metal levels in some soils can be elevated through human activities and even natural processes (Adriano, 2001; He et al., 2005). Good examples of natural element mineralization of soil are found in California's central valley, where soils are enriched with selenium due to a high-selenium parent material (Dungan and Frankenberger, 1999); or in northern California, where soils contain nickel levels as high as 1,000–2,000 mg kg^{-1}, because the parent material is serpentine, a mineral with high natural levels of nickel. As discussed in **Appendices A** and **C**, risks to plants and grazing livestock from most trace metals in soil are low. Serpentine soils with high nickel concentrations (as much as 50 times greater than other background soils) are rarely phytotoxic if the pH does not fall below 6 (Kukier and Chaney, 2004). Even at these extreme soil nickel concentrations, natural flora and fauna thrive without detriment.

Comparing metal concentrations in background soils and silica-based U.S. iron, steel, and aluminum SFSs (see **Table 7-1**) reveals that the concentrations of most metals and metalloids in SFS fall below those in most background U.S. and Canadian soils. However, the 95th percentile concentrations of arsenic, chromium, copper, manganese, molybdenum, and nickel in SFS exceed the median soil background concentrations for these metals. This does not, however, by itself mean that SFS should not be used as a soil amendment or component in a manufactured soil, as other lines of evidence (e.g., comparison to human and eco screening values) may mitigate concern. Based on the total metal data for silica-based iron, steel, and aluminum SFSs reported here, applications of most SFSs to average U.S. soils will not cause significant increases in the total soil metal concentrations.

6.3.2 Chemical Reactions in Soil

Soils contain metals at concentrations dependent on the parent material from which the soil is derived (Kabata-Pendias, 2001). Metals may also reach soils as components of fertilizers, manures, byproducts, and aerosols, and hence may exist in varied chemical forms. If metals reach soils in elemental forms, they will oxidize rapidly depending on the redox characteristics of the metal and the soil. For example, silver, gold, and even copper are found in a metallic form in some reducing soils, but copper and silver are usually oxidized in aerobic soils over time. Some are oxidized rapidly, but a few persist for long periods depending on the particle size of the metal that reached the soil (smaller particles have higher surface area and react more rapidly) or the redox status of the soil. Flooded soils (e.g., peat soils) may provide a reducing soil environment, which will allow metallic or metal sulfide particles to persist for long periods.

The soluble cation and oxyanion forms of trace metals in aerobic soils are potentially more mobile, and thus potentially more bioavailable than the elemental forms of the trace metals, so a risk assessment for the aerobic soil forms is appropriate. In a normal aerobic soil, most metals are present as hydrated or complexed cations or anions controlled by their chemistry in equilibrium with the ions bound to the soil surfaces or precipitated as minerals in the soil (Langmuir et al., 2005), such as Zn^{2+}, Cu^{2+}, Ni^{2+}, Pb^{2+}, Cd^{2+}, MoO_4^{2-}, SeO_4^{2-}, and $H_2PO_4^-$. Many ions remain in the cation form regardless of soil redox conditions: Li^+, K^+, Na^+, Rb^+, Cs^+ (alkali cations), Be^{2+}, Mg^{2+}, Ca^{2+}, Sr^{2+}, Ba^{2+} (alkaline earth cations), and select trace elements, including

Zn^{2+}, Cu^{2+}, Ni^{2+}, Pb^{2+}, Cd^{2+}, and MoO_4^{2-}. Similarly, many anions occur as halides (F^-, Cl^-, Br^-, I^-) in terrestrial soils (Bohn et al., 2001).

Flooding a soil (e.g., rice paddies) causes the redox potential to decrease as the soil becomes reducing, as little oxygen dissolves in water and soil organisms consume the oxygen. The soil pores become filled with water or gases are formed in the soil under anaerobic conditions. With the reducing environment, some metalloids are reduced to chemical forms different than those found in normal aerobic soils. In particular, As(V) as arsenate (AsO_4^{3-}) is reduced to the more mobile As(III) as arsenite (AsO_3^{3-}), which increases the arsenic in the soil solution. This is important in the case for phytotoxicity of arsenic; flooded rice is the crop plant found to be most sensitive to excessive soil arsenic. The higher concentration of AsO_3^{3-} in flooded soils compared to AsO_4^{3-} in aerobic soil allows much easier plant uptake and injury from the soil arsenic. Uptake of some other ions may be increased in reducing soils, but the potential for toxicity of other metals is not increased by reducing conditions as found with arsenic.

Sorption is a chemical process that buffers the partitioning of trace metals between solid and liquid phases in soils and byproducts. Iron, aluminum, and manganese oxide soil minerals are important sinks for trace metals in soil and byproduct-amended soils (Essington and Mattigod, 1991; Lombi et al., 2002; Hettiarachchi et al., 2003). Trace metal sorption by the oxide surface is a pH-dependent process; protons compete with cations for sorption. The adsorption of metal cations by the oxide surfaces increases to almost 100% with increasing pH (McKenzie, 1980). In contrast, oxyanion adsorption generally decreases with increasing pH.

Trace metal cations can also sorb to soil organic matter (SOM) and other forms of humified natural organic matter (NOM). Strong adsorption by NOM in byproducts (through the formation of metal chelates) reduces solubility of several trace metals in soil (Adriano, 2001). Sorption of trace metals by SOM or NOM increases with pH because protons compete less well with increasing pH. Trace metal sorption by NOM is reduced less at lower pH than is trace metal ion sorption on iron and manganese oxides.

Trace metal cations also form sparingly soluble precipitates with phosphate, sulfides, and other anions (Lindsay, 2001; Langmuir et al., 2005). Trace metal precipitation is highly pH dependent and increases with pH for many trace metal cations. Arsenate and other trace metal oxyanions can form insoluble precipitates with multivalent cations, including aluminum, calcium, and iron. Trace metal precipitation affects the amount of trace metal in solution (i.e., availability and mobility).

6.3.3 Soil-Plant Barrier

The potential risk that diverse trace metals in soils pose to the feed- and food-chain has been thoroughly examined over the last several decades. One purpose of that investigation has been to understand the risk from application of biosolids, livestock manure, and other trace metal contamination sources to soil. During this period, the soil-plant barrier concept was introduced to communicate how metal addition rate and chemistry, soil chemistry, and plant chemistry affected the risk to plants and animals from metals in soil amendments (Chaney, 1980; 1983; Langmuir et al., 2005). The soil barrier protects by way of soil chemical processes that limit the availability of metals for uptake, while senescence due to phytotoxicity further reduces the chances that excessively contaminated plants will be consumed (i.e., plant barrier). This concept is based on much experience in veterinary toxicology and agronomy. Reactions and processes that influence the soil-plant barrier include the following:

- Solid adsorbent sources (e.g., iron, aluminum, and manganese oxyhydroxides and organic matter) in soil amendments have adsorptive surfaces that influence soil chemistry
- Adsorption or precipitation of metals in soils or in roots limit uptake-translocation of most metals to shoots
- The phytotoxicity of some elements (e.g., aluminum, arsenic, boron, chromium III, copper, fluorine, manganese, nickel, zinc) limits the concentrations of these metals in plant shoots to levels chronically tolerated by livestock and humans
- The food-chain transfer of an element may not constitute a risk, but the direct ingestion of the contaminated soil may cause risk from arsenic, fluorine, lead, and some other elements under poor management conditions if the soils are highly contaminated
- The soil-plant barrier does not restrict the transfer of soil selenium, molybdenum, and cobalt well enough to protect all animals (selenium, molybdenum) or ruminant livestock (cobalt), or cadmium to subsistence rice consumers or cadmium in the absence of the usual 100-fold greater concentrations of zinc than the concentrations of cadmium.

A summary of the trace metal tolerances by plants and livestock is presented in **Appendix A, Table A-1**. It should be noted that the National Research Council (NRC, 1980) committee, which identified the maximum levels of trace metals in feeds tolerated by domestic livestock, based its conclusions on data from toxicological-type feeding studies in which soluble trace metal salts had been mixed with practical or purified diets to examine the animals’ response to the dietary metals. If soil or some soil amendment is incorporated into the diet, metal solubility and bioavailability are much smaller than in the tests relied on by the NRC (1980). For example, it has been noted that until soil exceeds about 300 mg Pb kg^{-1}, animals show no increased body lead burden from ingesting the soil (Chaney and Ryan, 1993). Other metals in equilibrium with poorly soluble minerals or strongly adsorbed in ingested soils are often much less bioavailable than they would be if they were added to the diet as soluble salts.

6.3.4 Interactions Among Constituents

The toxicity to animals of biosolids or manure-applied metals is an example of how the interaction between metals affects their toxicity. Specifically, copper deficiency–stressed animals are more sensitive to dietary zinc than animals fed with copper-adequate diets. Biosolids-fertilized crops are not low in copper, reducing animal sensitivity to zinc levels (Chaney, 1983).[47] Similarly, copper toxicity to sensitive ruminant animals is substantially reduced by increased dietary levels of cadmium, iron, molybdenum, zinc, and SO_4^{2-}, or sorbents such as SOM. In contrast with the predicted toxicity from copper in ingested swine manure or biosolids, reduced liver copper concentrations have been found in cattle or sheep that ingested biosolids, unless the ingested biosolids exceeded about 1,000 mg Cu kg^{-1} (Chaney and Ryan, 1993). Similarly, zinc in plants inhibits the absorption of cadmium by animals, as plant sulfate inhibits

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[47] Chaney (1983) also found that zinc phytotoxicity further protects livestock (including the most sensitive ruminants) against excessive zinc in forages: Plant senescence from phytotoxicity reduces the chances that excessively contaminated plants will be consumed by animals.

absorption of plant selenium. Interactions that reduce risk are evident in many trace element issues.

Interactions can also limit toxicity and risk. For example, cadmium bioavailability is strongly affected by the presence of normal background levels of zinc in soils (100- to 200-fold cadmium level); zinc inhibits the binding of cadmium by soil, but also inhibits cadmium uptake by roots, cadmium transport to shoots, and cadmium transport to storage tissues. Furthermore, zinc in foods significantly reduces cadmium absorption by animals (Chaney et al., 2004). Increased zinc levels in spinach and lettuce reduced the absorption of cadmium in these leafy vegetables by Japanese quail (McKenna et al., 1992a). Also, increased zinc in forage diets strongly inhibited cadmium absorption and reduced liver and kidney cadmium concentrations in cattle (Stuczynski et al., 2007).

6.3.5 Highly Exposed Populations

Risk assessment for wildlife is similar to that of livestock; because of their limited range, the diets of some species (e.g., plants, soil invertebrates, small mammals) can originate entirely from the soil or plants grown on a site. Because these species have higher exposures than most wildlife, they are used as the highly exposed populations. In cases involving wildlife in unmanaged ecosystems, maximal plant residues may exceed those allowed on managed farmland—wildlife may eat sick plants that would not be harvested by a commercial grower. Evaluation of the literature on wildlife exposure to trace metal–contaminated soils indicates that animals that consume earthworms are the highly exposed populations (Brown et al., 2002).

Cadmium has received much study because of extensive human cadmium disease in nations where subsistence rice farmers consume locally grown rice for their lifetime (Chaney et al., 2004). The disease results from chronic exposure to food-borne cadmium. Basic studies on the bioavailability of food cadmium have indicated that rice promotes cadmium absorption by inducing iron and zinc deficiency in the subsistence rice farm families because of the very low levels and low bioavailability of iron and zinc in polished rice (Reeves and Chaney, 2002). A diet deficient in iron and zinc causes much more of the cadmium to be absorbed than in other diets tested (Reeves and Chaney, 2004). Several epidemiological studies have found no evidence of human cadmium disease from garden foods grown on Zn+Cd rich smelter or mine waste contaminated garden soils (Chaney et al., 2004).

Cobalt is another unusual case in that ruminant livestock are at risk from dietary cobalt at much lower crop cobalt levels. Cobalt is essential for vitamin B12 synthesis by rumen bacteria. Crops can accumulate at least 25 mg Co kg^{-1} dry weight before even sensitive crops are injured by the absorbed cobalt, but ruminants can tolerate no more than about 10 mg Co kg^{-1} dry weight (DW) diets (Keener et al., 1949; Becker and Smith, 1951; Corrier et al., 1986; NRC, 1980). In practice, no case of cobalt toxicity has been reported, apparently because excessive levels of cobalt in soil are rare. It remains theoretically possible for cobalt in soil to poison ruminants. In the case of serpentine soils geochemically enriched with both nickel and cobalt, the nickel inhibits the uptake of cobalt and the soil properties limit the uptake of both nickel and cobalt, and the potential adverse effects of cobalt to plants or animals have never been observed.

6.4 Spent Foundry Sand Product Risks

Spent foundry sand has been found to be useful in making fertile soil mixtures for many agricultural and horticultural uses. The present evaluation considered a high-end use: a 20-cm layer of manufactured soil containing 50% SFS by dry weight in the blend. Such blends often contain soil, composts, manure, and other ingredients that provide a rooting mixture for diverse plants. These soils are used for yards, gardens, institutional lawns, and other instances where existing soils have been disturbed or have very low fertility and fail to support plant growth.

Uses of SFS in manufactured soils are mostly at lower rates than the rates assumed in the present risk assessment. Evaluation of SFS alone (i.e., not blended with organic additives) as a replacement soil was considered, but research has shown that for many SFSs, this is not feasible. SFS without treatment tends to form a cemented solid material (De Koff et al., 2008). Often this is due to the presence of sodium bentonite in the SFS, which causes the cementation reaction and “sealing” of the soil (Dungan et al., 2007). This can be corrected through the addition of soluble calcium salts. The usefulness of SFS alone is also restricted by its limited particle size. Soil-related beneficial uses of SFS generally use SFS as a small fraction of a mixed soil. Under the expected conditions (i.e., SFS as a component of manufactured soil), no risks were identified in the literature.

Under aerobic conditions, long-term exposures to metals in SFS-manufactured soil will continue to be low as it weathers. Over time, the sand and clays present in SFS are reduced in size by physical processes and/or dissolution, while organic byproducts will be broken down to elemental forms, mainly through biological processes. The trace metals in a SFS-manufactured soil are not normally bioavailable, as they are bound within the matrix of minerals or sorbed to organic matter or metal oxides. Even exposing pure iron, steel, and aluminum SFSs to acid conditions (e.g., TCLP, SPLP) did not cause significant quantities of trace metals to be released into leachates. Given the pH range of SFS (neutral to slightly alkaline), the presence of aluminum, iron, and manganese will decrease the availability of trace metal cations due to the adsorption on oxide surfaces. Metal oxides, such as iron and manganese, are important in regulating the partitioning of trace metals between solution and solid phases in soils (Basta et al., 2005). Trace metal cations and oxyanions, which are generally more mobile and bioavailable than elemental forms, can also be expected to sorb to organic matter and form insoluble precipitates. Because an SFS-manufactured soil will become more “soil-like” with time, elements released due to weathering and mineralization are likely to behave like those in native soils.

6.5 PAHs, Dioxins, Furans, and Dioxin-Like PCBs in SFS

6.5.1 PAHs

Chapter 2 points out that the majority of the PAHs that were found at concentrations above detection limits were the 2- and 3-ring PAHs (i.e., acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene). Anthracene, fluorene, naphthalene, and phenanthrene were the most prevalent PAHs, detected in >79% of the SFSs (Dungan, 2006). Also detected above the MDLs, though in only a few sands, were benz[a]anthracene, chrysene, fluoranthene, and pyrene.

The 95th percentile concentrations for 11 PAHs in SFS were compared to (1) the Residential SSLs adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), (2) the inhalation screening level concentrations for benz[a]anthracene, chrysene, dibenz[a,h]anthracene, and naphthalene, the only PAHs for which inhalation health benchmarks were available, and (3) Eco-SSLs for total Low Molecular Weight PAHs and total High Molecular Weight PAHs (see Section 4.4.3 for a discussion of these categories). As seen in **Table 6-1**, in all cases, the 95th percentile constituent concentrations in SFS were below the corresponding Adjusted SSL, with most cases, the 95th percentile constituent concentrations being orders of magnitude below the corresponding Adjusted SSL. When aggregated by molecular weight category, the 95th percentile constituent concentrations of Low and High molecular weight PAHs were similarly below their respective Eco-SSLs.

Based on this comparison, the presence of these PAH compounds in SFS are unlikely to cause adverse human or ecological health impacts at levels of concern when SFS is used in SFS-manufactured soils, soil-less potting media, or road base..

Table 6-1. Comparison of PAH Concentrations in SFS to Screening Criteria (mg kg^{-1})

Constituent	SFS 95%-ile	$Conc_{MS}$	Adjusted SSL	Inhalation Screening Level	Eco-SSL
Low Molecular Weight PAHs [a] - Total	7.59	3.79	N/A	N/A	29
Acenaphthene	0.34	0.17	350	N/A	N/A
Acenaphthylene	0.20	0.10	N/A	N/A	N/A
Anthracene	0.88	0.44	1,700	N/A	N/A
Fluorene	0.73	0.37	230	N/A	N/A
Naphthalene	3.89	1.94	3.8	60,300	N/A
Phenanthrene	1.56	0.78	N/A	N/A	N/A
High Molecular Weight PAHs [a] - Total	0.95	0.48	N/A	N/A	1.1
Benz[a]anthracene	0.14	0.07	0.15	4,020	N/A
Chrysene	0.04	0.02	1.5	221	N/A
Dibenz[a,h]anthracene	0.08	0.04	0.15	402	N/A
Fluoranthene	0.21	0.10	230	N/A	N/A
Pyrene	0.48	0.24	170	N/A	N/A

N/A = no benchmark available.

[a] Low Molecular Weight PAHs are composed of fewer than four condensed aromatic ring structures, and High Molecular Weight PAHs are composed of four or more condensed aromatic ring structures (EPA, 2007e).

6.5.2 PCDDs, PCDFs, and Dioxin-like PCBs

As described in **Chapter 2**, except for 1,2,3,7,8,9-HxCDF, most PCDD and PCDF congeners were detected, but not in all SFSs. Concentrations of the PCDD congeners ranged from <0.01–44.8 ng kg^{-1}, with 1,2,3,4,6,7,8,9-OCDD being found at the highest concentration in all of the SFSs. Expressed in terms of TEQs, the total dioxin concentrations ranged from 0.01–3.13 ng TEQ kg^{-1}, with an average concentration of 0.58 ng TEQ kg^{-1}. However, because PCB-81 and mono-*ortho*-substituted PCBs were not measured, the PCB contribution to the total TEQ concentration is not known. Nevertheless, the highest total dioxin concentration (expressed as a toxic equivalency value) of 3.13 ng TEQ kg^{-1} is about 100 times lower than the 300 ng TEQ kg^{-1} limit developed by EPA for biosolids (U.S. EPA, 2002e). The biosolids matrix has a significantly higher organic carbon content relative to the SFSs; however, SFS-manufactured soils will presumably also contain organic amendments and nutrients at levels that are beneficial to the soil.

The maximum concentration for total TEQs[48] was compared to (1) the Residential SSL adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), and (2) the inhalation screening level concentration derived for the manufacturing scenario. No ecological health benchmarks were available for PCDDs, PCDFs and dioxin-like PCBs; therefore the potential for adverse ecological impacts from exposure to these SFS constituents was not evaluated. As seen in **Table 6-2**, the maximum total TEQ was at least an order of magnitude below the soil and inhalation screening levels. Also, the concentrations of TCDD-TEQ in SFS were below background levels in U.S. agricultural soils, and well below levels in urban soils (Rogowski and Yake, 2005; Andersson and Ottesen, 2008). Furthermore, the highest total dioxin concentration was about 100 times lower than the biosolids limit. Based on the above information, exposure to levels of PCDDs, PCDFs, and dioxin-like PCBs found in SFS is unlikely to cause adverse human health impacts when SFS is used in SFS-manufactured soils, soil-less potting media, or road base.

Table 6-2. Comparison of Total Dioxin TEQ Concentrations in SFS to Screening Criteria
(mg TEQ kg^{-1})

PCDDs, PCDFs, and Co-planar PCBs	Maximum SFS Concentration	$Conc_{MS}$	Adjusted SSL	Inhalation Screening Level
Total dioxin TEQ	3.1E-06	1.6E-06	4.9E-06	2.01E-02

6.6 Phenolics in SFS

As discussed in **Chapter 2**, the phenolics that were detected in the majority of the SFSs included phenol, 2-methylphenol, 3- and 4-methylphenol, and 2,4-dimethylphenol. In general, phenol was found at the highest concentration, followed by 2-methylphenol and then 3- and 4-methylphenol and 2,4-dimethylphenol. Phenol was present in 35 of the 39 silica-based samples from iron, steel, and aluminum foundries at concentrations ranging from 0.11–46.1 mg kg^{-1}.

The 95th percentile concentrations for these five phenolics in SFS were compared to (1) the human health SSLs for soil ingestion, and (2) the inhalation screening level concentrations for the three compounds for which inhalation health benchmarks were available. No ecological health benchmarks were available for the phenolic compounds found in SFS; therefore the potential for adverse ecological impacts from exposure to phenolics in SFS was not evaluated. As shown in **Table 6-3**, high-end phenolic concentrations in SFS are multiple orders of magnitude below ingestion SSLs. Concentrations of phenolics in SFS were also orders of magnitude below inhalation screening levels for those constituents with available inhalation health benchmarks.

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[48] Due to a small data set (10 data points), it was decided to use the maximum value rather than the 95th percentile.

Table 6-3. Comparison of Phenolic Concentrations in SFS to Screening Criteria
(mg kg-1)

SFS Constituent	95%-ile SFS Concentration	$Conc_{MS}$	Adjusted SSL	Inhalation Screening Level
4-Chloro-3-methylphenol	0.09	0.05	620	N/A
2,4-Dimethylphenol	5.60	2.80	120	N/A
2-Methylphenol	8.76	4.38	310	Capped
3- and 4-Methylphenol	3.59	1.79	310	Capped
Phenol	22.1	11.1	1,800	Capped

N/A = no benchmark available.

Capped = Screening modeling estimates indicated risks below levels of concern at concentrations above 1,000,000 mg kg^{-1} (i.e., SFS could be comprised entirely of this constituent and still not cause risk).

Based on the above information, concentrations of these phenolic compounds in SFSs are unlikely to cause adverse impacts to human health when SFS is used in SFS-manufactured soils, soil-less potting media, or road base.

6.7 Metals and Metalloids in SFS

This section brings together previously presented information related to metals in SFS, their behavior in soil, and results of screening and unitized risk-related modeling. Subsections for the eight metals that were considered in the home gardener scenario screening (antimony, arsenic, chromium (III), cobalt, copper, iron, manganese, and nickel) summarize information comparing metal concentrations in SFS to screening criteria and modeling results to evaluate the potential for adverse human health and ecological effects. Constituent-specific total concentrations data for each sample can be found in **Appendix B, Table B-19**. Specific subsections for each metal compare background concentrations in native soils to concentrations in SFS to illustrate the similarity to native soils, as appropriate. Each subsection then describes other factors that will affect the metal's mobility in soil, bioavailability to plants, and toxicity to plants. These factors include processes that affect the dynamics of metal behavior associated with SFS soil applications (e.g., sorption mechanisms), as well as metal-specific characteristics that will limit or prevent certain exposure pathways from being completed (e.g., the soil-plant barrier). Lastly, a lines-of-evidence section integrates this information and presents conclusions regarding the potential risk associated with each of the eight metals evaluated in Phase II.

In addition to these detailed sections, information on other metal and metalloid constituents found in SFS are summarized, essentially distilling all of the information presented earlier in the report into a concise discussion of risk conclusions.

6.7.1 Antimony

The total antimony concentrations (see Table 2-4) in silica-based iron, steel, and aluminum SFSs collected in June 2005 ranged from a minimum of <0.04 mg kg^{-1} to a maximum of 1.71 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 1.23 mg Sb kg^{-1}. Using the SPLP and water extraction, the antimony results were all below the detection limit of 0.04 mg L^{-1} (Dungan and Dees, 2009) (Table 2-12, Table 2-13). Sample-specific SPLP and water extract leachate data can be found in **Appendix B, Tables B-13 through B-18**.

6.7.1.1 Comparison to Screening Levels

The relevant screening levels include Eco-SSLs[49], the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), the tapwater screening level (Tapwater SL), and the MCL for drinking water. These screening levels typically reflect study data on highly bioavailable forms of antimony:

- Eco-SSL (soil invertebrates): 78 mg kg^{-1} soil
- Eco-SSL (mammals): 0.27 mg kg^{-1} soil
- Adjusted SSL (noncancer): 3.1 mg kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)
- Tapwater SL (noncancer): 0.0078 mg L^{-1}
- MCL: 0.006 mg L^{-1}

Comparing the 95th percentile total concentration of antimony in SFS to the SSLs shows that, in a 1:1 manufactured soil blend (i.e., 50% SFS and 50% organic components, by weight), the concentration of antimony in manufactured soil is below the Eco-SSL for soil invertebrates, but exceeds the Eco-SSL for small insectivorous mammals. The 95th percentile antimony concentration is well below the corresponding Adjusted SSL; at a 50% blend, even the maximum concentration of antimony in SFS-manufactured soil would be below the Adjusted SSL. There were no samples above the detection limit for the SPLP and water extraction tests. Although the lack of detections suggests that antimony is unlikely to leach from SFS-manufactured soils at levels of concern, the detection limits are above the Tapwater SL and MCL for antimony.

6.7.1.2 Modeling Results

Based on the comparison with screening levels, the groundwater ingestion pathway and ecological exposure were further evaluated. The groundwater ingestion pathway evaluation used one half the analytical method detection limit (0.02 mg L^{-1}). The 90th percentile risk screening results for dry climate were virtually zero (see **Chapter 5, Section 5.2.2**). The peak 90th percentile risk screening results for central tendency and wet climates were 1.8E-3 and 5.9E-3 mg L^{-1}, respectively, both below the Tapwater SL and MCL (7.8E-3 mg L^{-1} and 6.0E-3 mg L^{-1}, respectively).

The 95th percentile antimony concentration in SFS-manufactured soil (0.62 mg kg^{-1} DW) was above the Eco-SSL for small mammals (0.27 mg kg^{-1} DW). Therefore, there was an evaluation of the critical assumptions associated with the ecological hazard screen. One such assumption was that 100% of the small mammal diet originated from the raised home garden (e.g., for antimony, the shrew was the target species). As discussed in **Section 5.3.8.2**, the percentage of the diet attributable to the home garden was adjusted to better reflect the behavior of the shrew and provide a more realistic scenario for the usage of the home garden as part of the shrew habitat. This refined ecological modeling estimated that up to a concentration of 4.1 mg antimony kg^{-1} SFS (i.e., three times the 95th percentile antimony concentration in SFS), the

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[49] The Eco-SSL development process includes a number of very conservative modeling assumptions (e.g., metal exists in most toxic form or highly bioavailable form, high food ingestion rate, high soil ingestion rate). Soil concentrations above Eco-SSLs are not necessarily of concern, but need further study; constituents with soil concentrations below Eco-SSLs need no further study.

potential for adverse ecological effects would be below levels of concern. This suggests that adverse ecological effects from antimony in SFS are unlikely for the home gardener scenario.

6.7.1.3 Soil Background Concentrations

Background concentrations of antimony in U.S. and Canadian soils range from 0.14–2.3 mg kg^{-1}, with a median value of 0.6 mg kg^{-1} (Smith et al., 2005). As illustrated in **Figure 6-1**, the distribution of antimony in U.S. soils is shifted to the right of the distribution of antimony in SFS. With a maximum SFS value of 1.71 mg kg^{-1}, a 95th percentile value of 1.23 mg kg^{-1}, and a median SFS value of 0.17 mg kg^{-1}(Dayton et al., 2010), the majority of SFS-manufactured soils would fall below median soil background concentrations. Therefore, the addition of SFS-manufactured soil is likely to have little effect on the background soil concentrations of antimony and, in many cases, the concentration of antimony in soil may decrease due to dilution.

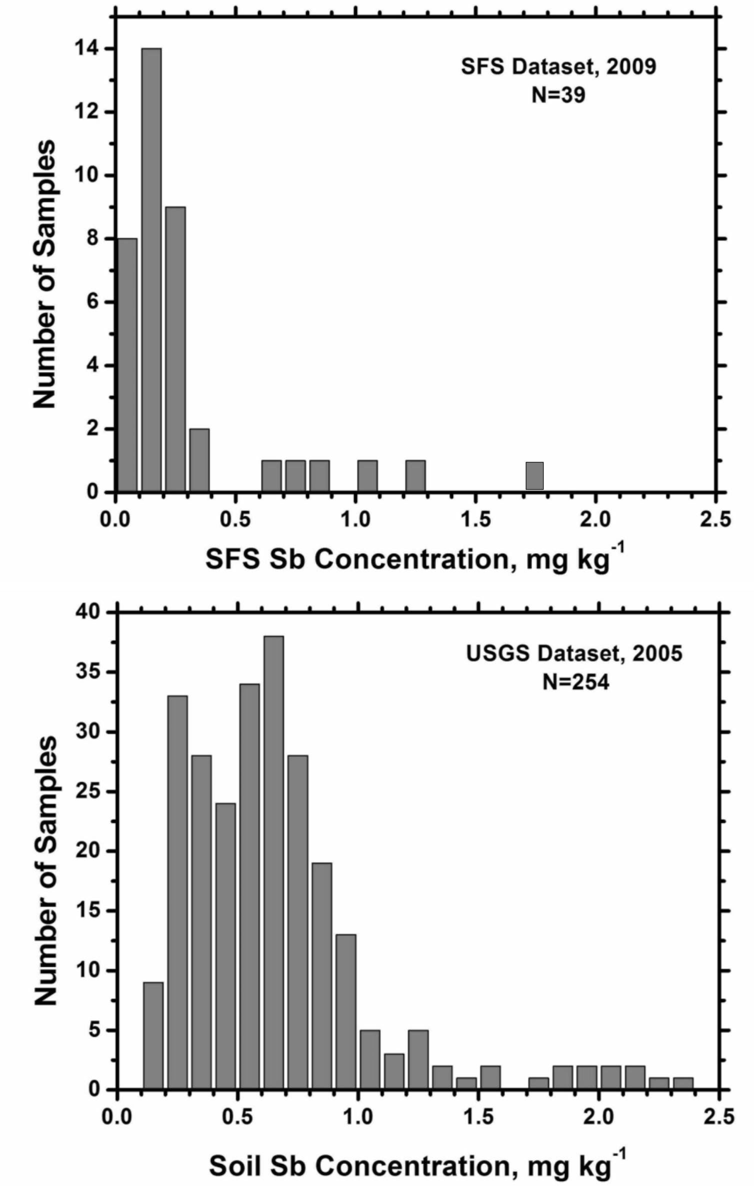

Figure 6-1. Concentration distributions of antimony in SFS (top) and U.S. and Canadian soils (bottom).

6.7.1.4 Additional Factors

Although antimony is not an essential nutrient for plants (e.g. Kabata-Pendias, 2001), it is generally considered to be readily taken up by plants. The few studies that have been published on the phytotoxicity of antimony indicate that antimony is moderately phytotoxic (Pais and Benton Jones, 1997). The lack of reference materials are likely responsible for a lack of sufficient data for EPA to establish an Eco-SSL for terrestrial plants.

6.7.1.5 Lines of Evidence

The 95th percentile concentration of antimony in SFS (1.23 mg kg^{-1}) falls well within the range of typical background concentrations of antimony in U.S. and Canadian soils (Smith et al., 2005). Therefore, the addition of SFS-manufactured soils to native soils (home gardens) would not be expected to result in significant changes with regard to antimony concentrations.

The 90th percentile screening probabilistic modeling results for the groundwater ingestion pathway were virtually zero for the dry climate, and were below the Tapwater SL and MCL (0.0078 mg L^{-1} and 0.006 mg L^{-1}, respectively) for central tendency and wet climates.

The risk screening results for ecological receptors showed that the 95th percentile concentration of antimony in SFS was below the Eco-SSL for soil invertebrate receptors, but exceeds the Eco-SSL for the most sensitive mammalian receptor group, the shrew. Even though the Eco-SSL for mammals (0.27 mg kg^{-1} DW) was below the median background concentration for antimony in the US and Canada (0.6 mg kg^{-1} DW), refined probabilistic modeling was conducted to determine if quantitative estimates of ecological hazard would be above levels of concern. The approach described in **Section 5.3.8** resulted in an SFS-specific ecological screening level for antimony of 4.1 mg kg^{-1} SFS (dry weight), three times the 95th percentile antimony concentration in SFS.

Based on the results of the risk screening and probabilistic screening modeling, and similarity with background concentrations, the levels of antimony in SFS are unlikely to cause adverse effects to human health and ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road base.

6.7.2 Arsenic

The total arsenic concentrations (see Table 2-4) in silica-based SFSs from iron, steel, and aluminum foundries collected in June 2005 (39 detects in 39 samples) ranged from a minimum of 0.13 mg kg^{-1} to a maximum of 7.8 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 6.44 mg kg^{-1} (Dayton et al., 2010). The SPLP leach test data for these same samples (22 of 39 detects) ranged from below the detection limit of 0.001 mg L^{-1} to a maximum of 0.098 mg L^{-1}, with a mean value of 0.007 mg L^{-1}. The concentrations in water extracts from the same samples (23 detects in 39 samples), ranged from below the detection limit of 0.001 mg L^{-1} to a maximum of 0.018 mg L^{-1}, with a mean value of 0.005 mg L^{-1}(Dungan and Dees, 2009). Sample-specific SPLP and water extract leachate data can be found in **Appendix B, Tables B-13 through B-18**.

6.7.2.1 Comparison to Screening Levels

The relevant screening levels include Eco-SSLs, the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), the tapwater screening level (Tapwater SL), and the MCL for drinking water. These screening levels typically reflect study data on highly bioavailable forms of arsenic:

- Eco-SSL (plants): 18 mg kg^{-1} soil
- Eco-SSL (mammals): 45 mg kg^{-1} soil
- Adjusted SSL (cancer): .43 mg kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways, as well as a target risk level of 1E-5)
- Tapwater SL (cancer): 4.5E-4 mg L^{-1}
- MCL: 0.01 mg L^{-1}.

Comparing the 95th percentile total concentration of arsenic in SFS to the SSLs suggests that, in a 1:1 manufactured soil blend (i.e., 50% SFS and 50% organic components, by weight), the concentration of arsenic in soil would be well below any of the identified ecological screening criteria. The 95th percentile arsenic concentration is also below (though not an order of magnitude below) the Adjusted SSL for the soil pathways; in a 50% blend, even the maximum concentration of arsenic from an SFS-manufactured soil would be below the Adjusted SSL. However, the comparison of the SPLP data from the 23 SFS samples that exceeded the detection limit of 0.001 mg L^{-1}, along with the water extract samples, indicates that the 95th percentile arsenic concentrations associated with these tests would exceed both the Tapwater SL and the MCL.

6.7.2.2 Modeling Results

The soil manufacturing scenario (inhalation of fugitive dust emissions by nearby residents) and the home gardener scenario (the groundwater ingestion pathway, and ingestion of soil and home-grown produce) were evaluated. For the inhalation exposure pathway, the screening results indicated that, up to a concentration of 40.2 mg kg^{-1} SFS, the potential for adverse human health impacts from arsenic in SFS-manufactured soil would be below levels of concern.

For the groundwater ingestion pathway, the 90th percentile probabilistic risk screening results were above the lowest screening level (i.e. the Tapwater SL) in the Wet and Central Tendency climates. However, more refined probabilistic modeling of the groundwater pathway found that the risk due to the ingestion of drinking water would be below the levels of concern up to a concentration of 59 mg kg^{-1} SFS.

The soil/produce pathway refined probabilistic results indicated that, up to a concentration of 8.0 mg kg^{-1} SFS, the risk due to the consumption of home-grown fruits and vegetables along with incidental soil ingestion would be below levels of concern.

For the home gardener scenario, separate target SFS screening concentrations were developed for the soil/produce and the groundwater pathways based on analyses that showed that these exposures are not likely to occur within the same timeframe.

6.7.2.3 Soil Background Concentrations

The range of background concentrations of arsenic in U.S. soils is broad, ranging from <0.1–93 mg kg^{-1} (Kabata-Pendias, 2001). The geometric mean of arsenic in surficial soils has been estimated at 5.8 mg kg^{-1} (Shacklette and Boerngen, 1984) and more recent studies on Canadian and U.S. surficial soils estimate that the median concentration of arsenic is 5.0 mg kg^{-1} (Smith et al., 2005). With a maximum SFS value of 7.79 mg kg^{-1}, a 95th percentile value of 6.44 mg kg^{-1}, and a median value of 1.05 mg kg^{-1}, almost all arsenic concentrations in SFS fall below the median soil background concentrations (Dayton et al., 2010). Given the importance of site-specific soil properties—particularly the iron and aluminum content in soil—the comparison between arsenic concentrations in SFS and arsenic background concentrations in soil suggests that arsenic concentrations in SFS overlap with the low end of the background concentration range, with the 95th percentile value in SFS slightly higher than the average soil background level. It is expected that nearly 95% of the SFS samples would have arsenic concentrations that were below the median national background soil arsenic level. **Figure 6-2** demonstrates these points graphically.

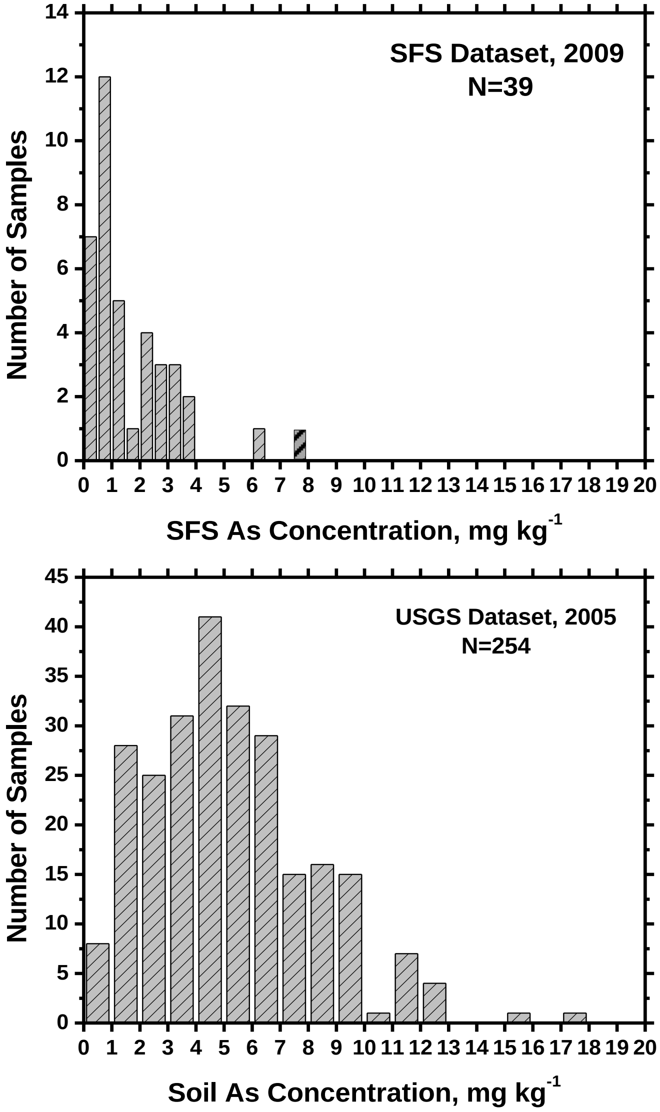

Figure 6-2. Concentration distributions of arsenic in SFS (top) and U.S. and Canadian soils (bottom).

6.7.2.4 Additional Factors

Arsenic is a constituent of most plants, although little is known about its biochemical role (Kabata-Pendias, 2001). The arsenic concentration in plants grown on uncontaminated soils varies from 0.009–1.5 mg kg^{-1} DW, with leafy vegetables falling into the upper end of the range and fruits falling into the lower end of the range. Some authors have shown that the uptake of arsenic depends upon the form of arsenic in the soil; for the radish, the order of uptake is As_{org} >> As(V) > As(III) (Tlustos et al., 1998). There are some reports that indicate the linear uptake of arsenic in soil considers both the soluble and total arsenic forms; however, more recent research has shown that AsO_4^{3-} enters plant roots on the phosphate transporter (Zhao et al., 2009). Although some plant species have been shown to tolerate high levels of arsenic in the tissues, the residue tolerance has generally been established around 2 mg kg^{-1} DW for plant species that are neither highly sensitive nor highly tolerant (Kabata-Pendias, 2001). Phytotoxicity appears to vary with the soil type; "heavy" soils with high organic matter content and vermiculitic clay as the predominant clay tend to significantly reduce the toxicity of arsenic to plants (Woolson et al., 1973).

The chemical reactions of arsenic in soils are thought to be controlled largely by the oxidation state, with the As(V) and As(III) forms dominant at the typical oxidation potential (Eh) and pH ranges of soil. The bioavailability of arsenic in soil is significantly reduced in the presence of hydrated iron and aluminum oxides.[50] A change in the redox potential of the soil to flooded anaerobic conditions results in the greater desorption of As(III), the more highly bioavailable form; flooded arsenic contaminated soils are known to cause arsenic phytotoxicity to rice, but not to other crops. In aerobic soils, As(V) predominates, and solubility can be increased by high additions of phosphate. In short, the chemistry and behavior of arsenic in soil is a highly complex, multivariate phenomenon that depends greatly on soil characteristics, especially soil pH and the redox potential, and the presence of other metals that form arsenical complexes that are generally not available to plants.

Given the complexities of arsenic behavior in soil, an additional analysis was performed that examined the impact of soil water partitioning coefficient (Kd) distributions on SFS screening levels as discussed in **Appendix G, Attachment E.** As described in Section 5.3, the home gardener scenario assumed that the properties and characteristics of the SFS-manufactured soil mimicked those of natural soil in the area. Accordingly, the SFS-specific screening levels were developed based on soil Kd values from U.S. EPA 2005. The resulting screening levels for the soil/produce and groundwater pathways were 8.0 mg As kg^{-1} SFS and 59 mg As kg^{-1} SFS, respectively. Under the Kd analysis, source modeling was also performed with an SFS waste-specific Kd distribution developed using the full set of whole waste/leachate pairs presented in Appendix B (i.e., the SFS total waste concentration was divided by the corresponding leachate concentration). Release estimates developed using the waste-specific Kds represent releases from SFS and so are not likely to accurately reflect releases from SFS-manufactured soil. While not used to generate recommended SFS-specific screening levels, these estimates represent a bounding study. The goal of this effort was to better characterize the uncertainty associated with the SFS arsenic screening levels. **Table 6-4** compares the soil-Kd based SFS Screening Levels and the bounding material-specific Kd screening levels. As seen from this table, the lowest soil-

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[50] To reflect this reduction, the exposure estimates developed for incidental ingestion of soil were adjusted using the EPA's default relative bioavailability (RBA) value of 60% (U.S. EPA, 2012b).

Kd screening level (i.e., 8.0 mg kg^{-1} SFS, for the soil/produce pathway) is nearly identical to the lowest screening level generated using the material-specific Kd distribution (i.e. 7.7 mg kg^{-1} SFS, for the groundwater pathway). The similarity between the recommended screening level and the bounding material-specific estimate fosters a high level of confidence that an SFS-specific screening level generated using the soil Kd distribution will be protective of human health under a range of pathways and environmental conditions.

Table 6-4. Home Gardening 90th Percentile Modeled SFS-specific Screening Levels for Arsenic

Pathway	Arsenic SFS Screening Levels (mg kg^{-1})	
	Based on Soil Kd Distribution	Bounding Estimate: Material-Specific Kd Distribution
Soil/Produce	8.0	9.5
Groundwater	59	7.7

6.7.2.5 Lines of Evidence

Based on the results of the comparison of total arsenic concentrations from SFS to Eco-SSLs, arsenic concentrations in SFS are unlikely to cause adverse health effects to ecological receptors.

For the home gardener scenario, the results of the probabilistic groundwater screening modeling showed that the 90th percentile exposure concentration in water could be above the lowest screening value in the Wet and Central Tendency climates. More refined, yet still conservative groundwater modeling found that the risk due to the ingestion of drinking water would be below the levels of concern up to an SFS arsenic concentration of 59 mg kg^{-1} SFS, which is well above the 95th percentile SFS concentration of 6.44 mg kg^{-1} SFS.

For the ingestion of home-grown produce and the incidental ingestion of soil, the most conservative modeled SFS-specific screening concentration of 8.0 mg kg^{-1} SFS is even above the maximum arsenic concentration in SFS, suggesting that human exposure to arsenic via the ingestion of vegetables and fruit grown in SFS-manufactured soil will be below levels of concern. The conservative nature of the refined modeling (e.g., allowing simultaneous, high consumption rates for multiple produce types) is such that arsenic concentrations in SFS are unlikely to cause adverse health impacts even at produce consumption rates.

The screening modeling analyses also evaluated inhalation risks to receptors living adjacent to a soil manufacturing facility (the most conservative of the inhalation exposure scenarios). This modeling generated allowable arsenic concentrations more than an order of magnitude above the 95th percentile and maximum arsenic concentrations found in SFS samples.

Therefore, because (1) the arsenic concentration in SFS is below all Eco-SSLs; (2) probabilistic modeling found that the potential for adverse health impacts from use of SFS-manufactured soil are below levels of concern in all evaluated exposure pathways; and (3) arsenic concentrations in SFSs are typically below average background soil concentrations, arsenic in silica-based SFS from iron, steel, and aluminum foundries is unlikely to cause adverse effects to human health or ecological receptors when SFS is used in manufactured soil, soil-less potting media, and road base.

6.7.3 Chromium

The total chromium concentrations (see Table 2-4) in silica-based SFSs from iron, steel, and aluminum foundries collected in June 2005 (38 detects in 39 samples) ranged from a minimum of <0.5 mg kg^{-1} to a maximum of 115 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 109 mg kg^{-1} (Dayton et al., 2010). The SPLP and water extract leach test data for these same samples were below the quantitative detection limits of 0.01 mg L^{-1} and 0.02 mg L^{-1}, respectively, for all samples (Dungan and Dees, 2009). Sample-specific SPLP and water extract leachate data can be found in **Appendix B, Tables B-13 through B-18**.

6.7.3.1 Comparison to Screening Levels

The relevant screening levels include Eco-SSLs, the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), the tapwater screening level (Tapwater SL), and the MCL for drinking water. These screening levels typically reflect study data on highly bioavailable forms of chromium (III):

- Eco-SSL (mammals): 34 mg kg^{-1} soil
- Adjusted SSL (noncancer): 12,000 mg kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)
- Tapwater SL (noncancer): 16 mg L^{-1}
- MCL: 0.1 mg L^{-1} (based on total Cr)

Comparing the 95th percentile total concentration of chromium in SFS to the SSLs suggests that, in a 1:1 manufactured soil blend (i.e., 50% SFS and 50% organic components, by weight), the concentration of chromium in SFS-manufactured soil would be above the Eco-SSL for small insectivorous mammals. However, this same concentration is below the Adjusted SSL for soil pathways; in a 50% blend, even the maximum concentration of chromium in SFS-manufactured soil would be below the Adjusted SSL. The SPLP and water extract leach data were all well below the Tapwater SL and MCL screening levels.

6.7.3.2 Modeling Results

The 95th percentile chromium III concentration in SFS-manufactured soil (109 mg kg^{-1} DW) was above the Eco-SSL for small mammals (34 mg kg^{-1} DW). This prompted a refinement of the assumptions associated with the ecological hazard screen. For chromium this involved refining the assumption that 100% of the small mammal diet originated from the home garden (for chromium, the shrew was the target species). As discussed in **Section 5.3.8.2**, the percentage of the diet attributable to the home garden was adjusted to better reflect the behavior of the shrew and provide a more realistic scenario for the usage of the home garden as part of the shrew habitat. Refined ecological modeling estimated that up to a trivalent chromium concentration of 510 mg kg^{-1} SFS (i.e., almost five times the 95th percentile trivalent chromium concentration in SFS), the potential for adverse effects to even the most sensitive ecological receptors would fall below levels of concern. Therefore, adverse ecological effects from chromium in SFS are unlikely for the home gardener scenario.

6.7.3.3 Soil Background Concentrations

The range of background concentrations of chromium in U.S. soils is broad, ranging from 3–5,320 mg kg^{-1}, with a median value of 27 mg kg^{-1}(Smith et al., 2005). As illustrated in **Figure 6-3**, the distribution of chromium concentrations in SFS is similar to that of background soils; however, the median concentrations for SFS is roughly 5 times lower than the median concentration in background soils. Given this comparison, the addition of SFS to soil is not expected to result in a significant change with regards to chromium concentrations.

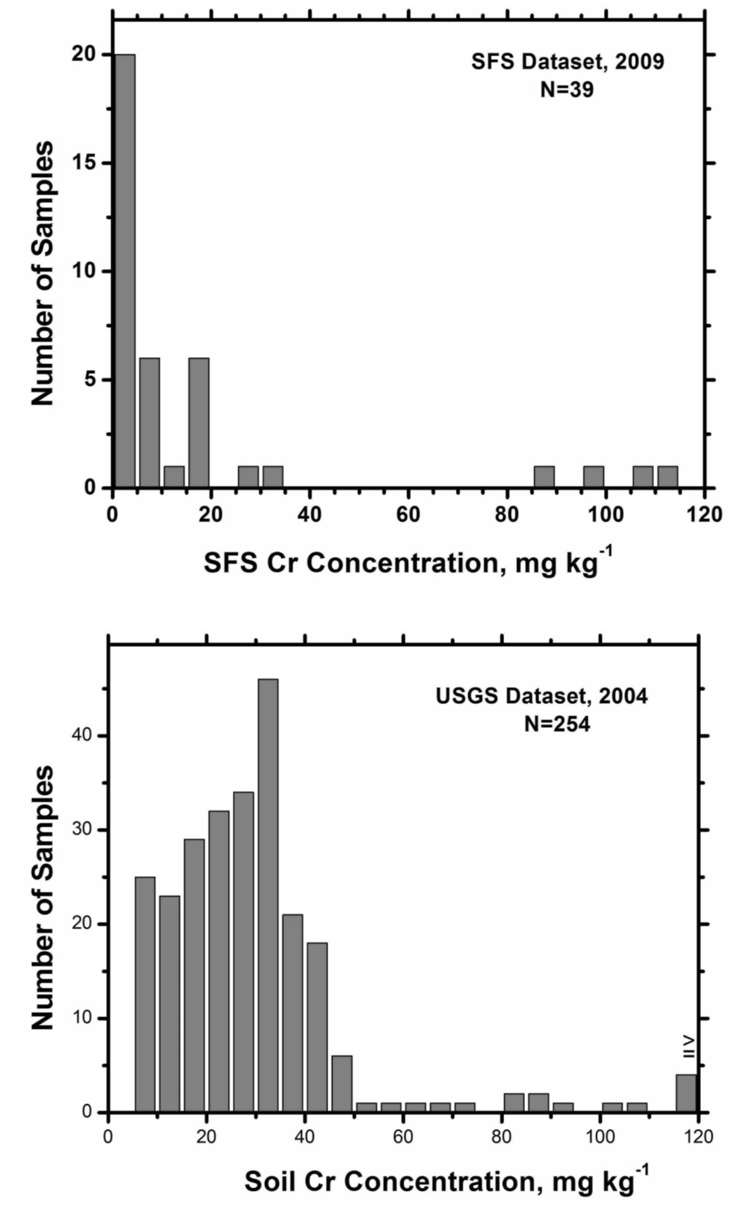

Figure 6-3. Concentration distributions of chromium in SFS (top) and U.S. and Canadian soils (bottom).

6.7.3.4 Additional Factors

Chromium III is not believed to be an essential nutrient for plants, although some studies have reported a stimulatory effect. Chromium is not readily taken up by plants, as there is a relatively low rate of absorption, largely attributed to the mechanism of uptake in plant roots. As with many metals, the content of chromium in plants is dependent on the concentration of soluble chromium in soils, the soil type, and the plant species (Kabata-Pendias, 2001). Pais and Benton Jones (1997) estimated average concentrations of chromium in plants to be 0.02 to 0.2 mg kg^{-1}, with phytotoxic concentrations averaging 10 to 15 mg kg^{-1}, and upper phytotoxic concentrations at > 150 mg kg^{-1} in soil. In terms of edible plants and crop species, average concentrations of total chromium in foods range from 0.05 mg kg^{-1} (apple) to 0.2 mg kg^{-1} (wheat) (Pais and Benton Jones, 1997). As evident from these data, chromium has been reported in varying ranges. However, some studies have documented that concentrations in plants may actually be an artifact of soil contamination issues related to sampling techniques rather than uptake by plants (e.g., Cary and Kubota, 1990; Grubinger et al., 1994; and Cary et al., 1994).

6.7.3.5 Lines of Evidence

The 95th percentile chromium concentration in SFS (109 mg kg^{-1}) falls well within the range of typical background concentrations of chromium for U.S. and Canadian soils (Smith et al., 2005). Therefore, the addition of SFS-manufactured soils to native soils (home gardens) would not be expected to result in significant changes to chromium concentrations.

The evaluation found the 95th percentile concentration of chromium in SFS to be below the health-based benchmarks for human receptors, but exceeded the Eco-SSL for small mammals. However, refined ecological modeling demonstrated, with a high degree of confidence that the risk to the target ecological receptor (shrew) would be below levels of concern. The approach described in **Section 5.3.8** resulted in an SFS-specific ecological screening level for chromium III of 510 mg kg^{-1} DW, more than 100 times higher than the 95th percentile chromium concentration in SFS.

Based on the results of the screening comparison, the refined ecological modeling, and the similarity with background concentrations, chromium levels in in SFS are unlikely to cause adverse effects to human health and ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road base.

6.7.4 Cobalt

The total cobalt concentrations (see Table 2-4) in silica-based SFSs from iron, steel, and aluminum foundries collected in June 2005 (28 detects in 39 samples) ranged from a minimum of <0.5 mg kg^{-1} to a maximum of 6.62 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 5.99 mg kg^{-1} (Dayton et al., 2010). No leach test data were available for cobalt.

6.7.4.1 Comparison to Screening Levels

The relevant screening levels include Eco-SSLs and the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL). These screening levels typically reflect study data on highly bioavailable forms of cobalt:

- Eco-SSL (terrestrial plants): 13 mg kg^{-1} soil
- Eco-SSL (mammals): 230 mg kg^{-1} soil
- Adjusted SSL (noncancer): 2.3 g kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)

Comparing the 95th percentile total concentration of cobalt in SFS (5.99 mg kg^{-1} DW) to the lowest Eco-SSL (13 mg kg^{-1} DW) indicates that the concentration of cobalt in SFS-amended soil would be below the Eco-SSL for terrestrial plants (and substantially below that for mammals). This cobalt concentration in SFS-manufactured soil exceeded the Adjusted SSL for the soil ingestion pathways. No leachate data were available for cobalt in SFS and, therefore, cobalt was not evaluated via the groundwater pathway.

6.7.4.2 Modeling Results

The soil manufacturing scenario (inhalation of fugitive dust emissions by nearby residents), and the home gardener scenario (ingestion of home-grown produce, and incidental ingestion of garden soil) were evaluated. For the inhalation exposure pathway, the screening modeling results indicate that up to a cobalt concentration of 2,010 mg kg^{-1} SFS (i.e., more than 100 times higher than the 95th percentile concentration of cobalt in SFS), the potential for adverse human health impacts would be below levels of concern.

With respect to the home garden scenario, the results of the refined modeling indicate that up to a cobalt concentration of 21 mg kg^{-1} SFS (i.e., over three times the 95th percentile concentration if cobalt in SFS), the use of SFS in manufactured soil is unlikely to cause adverse human health impacts.

6.7.4.3 Soil Background Concentrations

The range of background concentrations of cobalt in U.S. and Canadian soils is broad, ranging from 0.5–143.4 mg kg^{-1}, with a median value of 7.1 mg kg^{-1}(Smith et al., 2005). As illustrated in **Figure 6-4**, the composition of SFS with respect to cobalt appears to be substantially below U.S. soils, suggesting that the addition of SFS to soil would nearly always dilute cobalt levels in native soils.

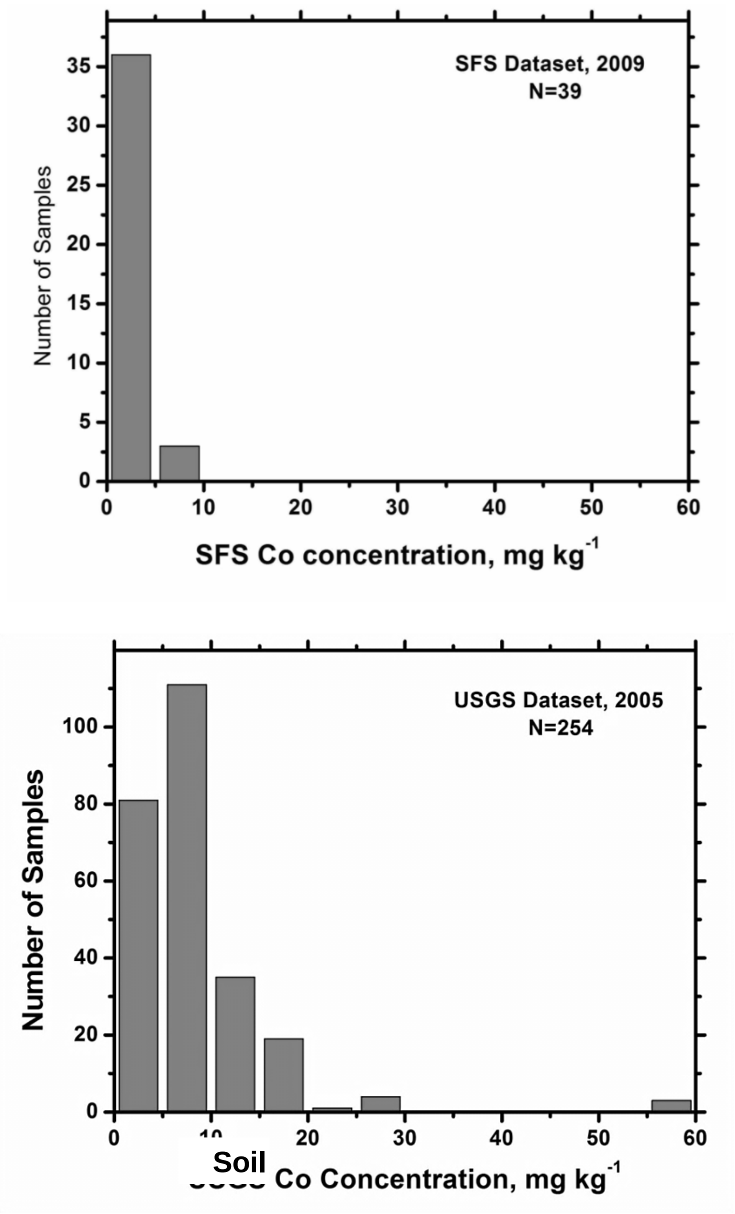

Figure 6-4. Concentration distributions of cobalt in SFS (top) and U.S. and Canadian soils (bottom).

6.7.4.4 Additional Factors

Numerous studies have investigated the interactions between cobalt and various plant species. Although there is evidence that cobalt may enhance plant growth, it is still not clear whether cobalt is an essential element for plants, and there is a substantial amount of data on phytotoxicity of cobalt (Kabata-Pendias, 2001). For instance, excess cobalt has been found to adversely affect plant growth and metabolism. Across several studies, the average concentrations that were found to be phytotoxic ranged from 6 to 143 mg kg^{-1}, depending on plant species (Pais and Benton Jones, 1997). In order to avoid toxicity to animals eating plants contaminated with cobalt, it has been recommended that plants should not contain more than 60 ppm (DW) (Kabata-Pendias, 2001).

Plant uptake of cobalt is highly dependent on environmental conditions, such as soil factors, and varies widely across plant species. For instance, legumes have been shown to accumulate more cobalt than grasses or grain crops. Moreover, soil texture has been cited as one of the most significant parameters controlling cobalt concentrations in plants. Some plants, termed hyperaccumulators, have developed a tolerance mechanism and are able to accumulate high concentrations of cobalt. In terms of edible plants, cobalt content has been shown to vary from 8 ppm (e.g. apples) to 100 ppm (e.g. cabbage) (DW). Studies from different countries report average cobalt concentrations in clover range from 0.10 to 0.57 ppm (DW), while grass concentrations range from 0.03 to 0.27 ppm (Kabata-Pendias, 2001 and references within).

6.7.4.5 Lines of Evidence

The distribution of cobalt concentrations in SFS is below the distribution in native soils; the 95th percentile SFS concentration (5.99 mg kg^{-1}) is below the background concentration median of 7.1 mg kg^{-1}, suggesting that the addition of SFS will tend to dilute rather than increase the level of cobalt in soils.

For the ingestion of home-grown produce and the incidental ingestion of SFS-manufactured soil, the most conservative SFS-specific screening concentration for cobalt (i.e., 21 mg kg^{-1} SFS) is well above the 95th percentile concentration of cobalt in SFS. The conservative nature of the refined screening modeling for these exposure pathways fosters a high level of confidence that an SFS-specific concentration of 21mg kg^{-1} is protective of human health.

Based on the results of the comparison of total cobalt concentrations in SFS with screening criteria, and probabilistic modeling, cobalt concentrations in SFS are unlikely to cause adverse effects to human health and ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road base.

6.7.5 Copper

The total copper concentrations (see Table 2-4) in silica-based iron, steel, and aluminum SFSs collected in June 2005 (39 of 39 detects) ranged from a minimum of <0.5 mg kg^{-1} to a maximum of 137 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 107 mg Cu kg^{-1} (Dayton et al., 2010). The SPLP leach test data for these same SFSs, from all three sampling events (June 2005, September 2005, July 2006) were below the quantitative detection limit of 0.07 mg L^{-1} for all samples. The concentrations in water extracts from the same samples (June 2005 with 2 detects, September 2005 with 0 detects, July 2006 with 1 detect), ranged from <0.07 mg L^{-1} to a maximum of 1.06 mg L^{-1}, with mean values of 0.070, 0.035, and 0.041 mg L^{-1} across the sampling schemes, respectively (Dungan and Dees, 2009). Sample-specific SPLP and water extract leachate data can be found in **Appendix B, Tables B-13 through B-18**.

6.7.5.1 Comparison to Screening Levels

The relevant screening levels include Eco-SSLs, the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), the tapwater screening level (Tapwater SL), and the MCL for drinking water. These screening levels typically reflect study data on highly bioavailable forms of copper (Table 4-12, Table 7-1, and Table 4-2, respectively):

- Eco-SSL (terrestrial plants): 70 mg kg^{-1} soil

- Eco-SSL (soil invertebrates): 80 mg kg^{-1} soil
- Eco-SSL (mammals): 49 mg kg^{-1} soil
- Adjusted SSL (noncancer): 310 mg kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)
- Tapwater SL (noncancer): 0.62 mg L^{-1}
- MCL: 1.3 mg L^{-1}

Comparing the 95th percentile total concentration of copper in SFS to the SSLs indicates that, in a 1:1 manufactured soil blend (i.e., 50% SFS and 50% organic components, by weight), the concentration of copper in SFS-manufactured soil would fall below the Eco-SSLs for terrestrial plants and soil invertebrates, but exceed the Eco-SSL for mammals. The copper concentration in SFS-manufactured soil is well below the corresponding Adjusted SSL for soil pathways; at a 50% blend, even the maximum concentration of copper from an SFS-manufactured soil would be below the Adjusted SSL.

Comparing the 95th percentile leachate concentration of copper in SFS to the Tapwater SL and MCL, the concentration of copper in SFS-manufactured soil is well below both relevant human health water screening levels.

6.7.5.2 Modeling Results

Given the results of the screening comparison for ecological receptors, probabilistic screening modeling was performed and predicted copper exposure concentrations were compared to the Eco-SSLs. As discussed in **Section 5.3.8.2**, the percentage of the diet attributable to the home garden was adjusted to better reflect the behavior of the shrew and provide a more realistic scenario for the usage of the home garden as part of the shrew habitat. The refined ecological modeling results indicate that up to a copper concentration of 160 mg kg^{-1} SFS, the risk posed to ecological receptors would be below levels of concern (see **Table 5-14**). As this is higher than the 95th percentile copper concentration in SFS (i.e., 107 mg kg^{-1} SFS), this indicates that copper found in SFS is below levels of concern for ecological receptors.

6.7.5.3 Soil Background Concentrations

Background concentrations of copper in U.S. and Canadian soils range from 1.7–81.9 mg kg^{-1}, with a median value of 12.7 mg kg^{-1} (Smith et al., 2005). As illustrated in **Figure 6-5**, the distribution of Cu concentrations in background soils is similar to the distribution of concentrations in SFS (e.g., the respective medians are within a factor of 2). However, the tail of the SFS distribution is characterized by higher concentrations than the tail of the distribution for background soils (see **Table 7-1**). Nevertheless, the addition of SFS-manufactured soil would not be expected to result in significant changes in the Cu concentration in native soils.

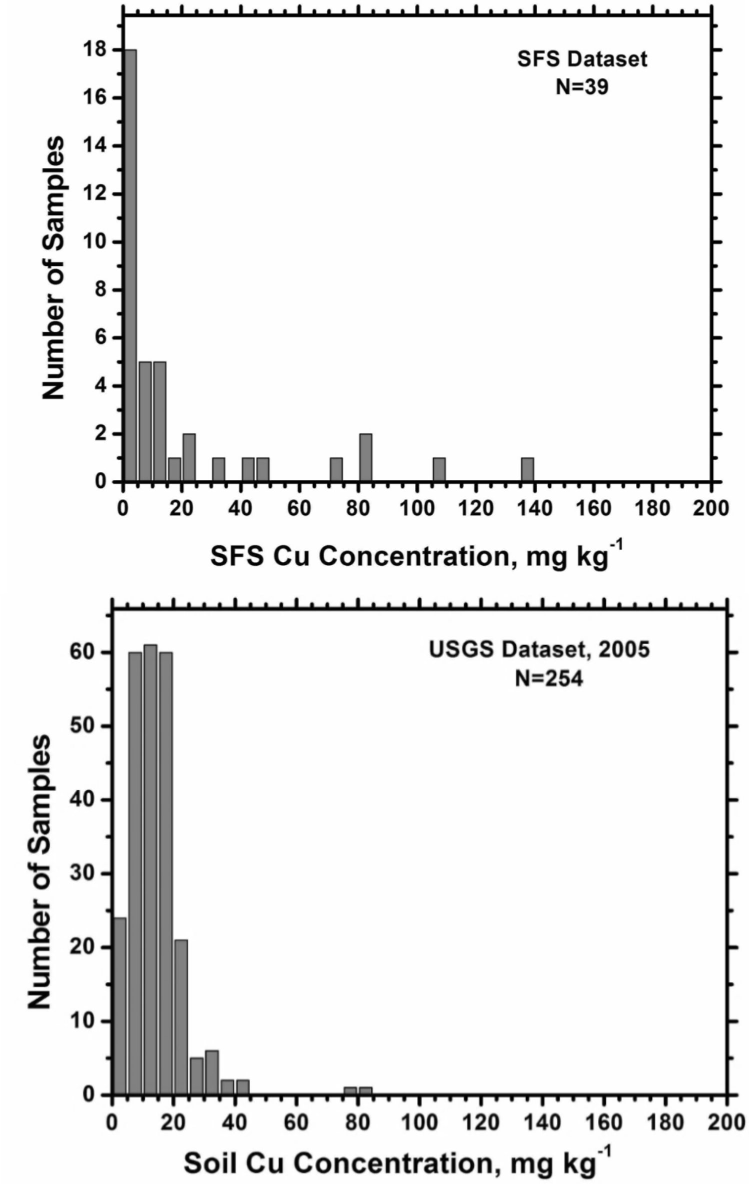

Figure 6-5. Concentration distributions of copper in SFS (top) and U.S. agricultural soils (bottom).

6.7.5.4 Additional Factors

Copper exists normally in soil, primarily as complexed forms of low molecular weight organic compounds, such as humic and fulvic acids (Pais and Benton Jones, 1997). Copper is an essential micronutrient for plants and, under normal conditions, its sufficiency range is 5-30 mg kg^{-1} (DW) (Pais and Benton Jones, 1997). Copper is important for photosynthesis, respiration, carbohydrate distribution, and protein metabolism, as well as nitrogen fixation processes (Kabata-Pendias, 2001). Similar to other metals, there is a variation in tolerance to copper among different plant species. Copper uptake depends mainly on the type of copper species (i.e. the oxide form of copper, largely coming from anthropogenic sources, is more bioavailable than copper coming from pedogenic sources). However, once copper has been absorbed by plant roots, relatively little is expected to be transported to plant tops (Pais and Benton Jones, 1997). In fact, copper has a relatively low mobility inside plant bodies compared to other elements; most of it will remain in the root and leaf tissues until they senesce, and only small amounts may move to young organs.

The distribution of copper inside plants varies, but the general trend is that translocation to leaves and other organs occurs predominantly when there is an abundance of copper available, and the plant is undergoing intensive growth (Kabata-Pendias, 2001). Average concentration ranges of copper in various foods include: vegetables: 0.1 (for celery root) - 3.2 (for garlic cloves) ppm FW; fruits: 0.3 (for grapes) - 4 (avocadoes) ppm FW; cereals: 0.3 (oats, whole grain) – 13 (rye, whole grain) ppm FW; and nuts: 0.2 (fresh coconut meat) – 23.8 (shelled Brazil nuts) (Kabata-Pendias, 2001).

6.7.5.5 Lines of Evidence

The 95th percentile copper concentration in SFS (107 mg kg^{-1}) falls well within the range of typical background concentrations of copper in U.S. and Canadian soils (Smith et al., 2005). Therefore, the addition of SFS-manufactured soils to native soils (home gardens) would not be expected to result in significant changes to copper concentrations.

The screening comparison indicated that copper in SFS-manufactured soil is below levels of concern for human exposures, but exceeded the Eco-SSL for small mammals. Refined ecological modeling demonstrated, with a high degree of confidence that the risk to the target ecological receptor (shrew) would be below levels of concern. The approach described in Section 5.3.8 resulted in an SFS-specific ecological screening level for copper of 159 mg kg^{-1} SFS, which is above the 95th percentile copper concentration in SFS.

Based on the results of the screening comparison for human health, the refined ecological modeling, and the similarity with background concentrations, copper levels in SFS-manufactured soil are unlikely to cause adverse effects to human health or ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road subbase.

6.7.6 Iron

The total iron concentrations (see Table 2-4) in silica-based SFSs from iron, steel, and aluminum foundries collected in June 2005 (39 detects) ranged from a minimum of 1.28 g kg^{-1} to a maximum of 64.4 g kg^{-1} (using EPA method 3051A), with a 95th percentile value of 57.1 g kg^{-1} (Dayton et al., 2010). No leach test data were available for iron.

6.7.6.1 Comparison with Screening Levels

The relevant screening levels include the default Residential soil screening level for the protection of human health, adjusted to also address home gardener produce ingestion pathways (Adjusted SSL). Screening levels typically reflect study data on highly bioavailable forms of iron:

- Adjusted SSL (noncancer): 5.5 g kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)

Comparing the 95th percentile total iron in SFS to the Adjusted SSL indicates that, in a 1:1 manufactured soil blend (i.e., 50% SFS and 50% organic components, by weight), the iron concentration in SFS-manufactured soil would exceed the Adjusted SSL. Iron was therefore evaluated under the Phase II probabilistic risk modeling.

6.7.6.2 Modeling Results

The refined modeling results demonstrate that up to an iron concentration of 150 g kg^{-1} SFS (i.e., almost three times the 95th percentile iron concentration in SFS), adverse human health effects are unlikely to occur.

6.7.6.3 Soil Background Concentrations

The range of iron background concentrations in U.S. and Canadian soils is broad, ranging from 3.8–87.7 mg kg^{-1}, with a median value of 19.2 mg kg^{-1}(Smith et al., 2005). As illustrated in **Figure 6-6**, the iron concentration in SFS would generally be lower than the iron concentration in native soils. The 95th percentile and maximum iron concentrations in SFS are, respectively, both below the corresponding background concentrations, and the median value for SFS is roughly 5 times lower than the median in native soils. This strongly suggests that the addition of SFS-manufactured soils would generally have a diluting effect on the iron concentrations in soil.

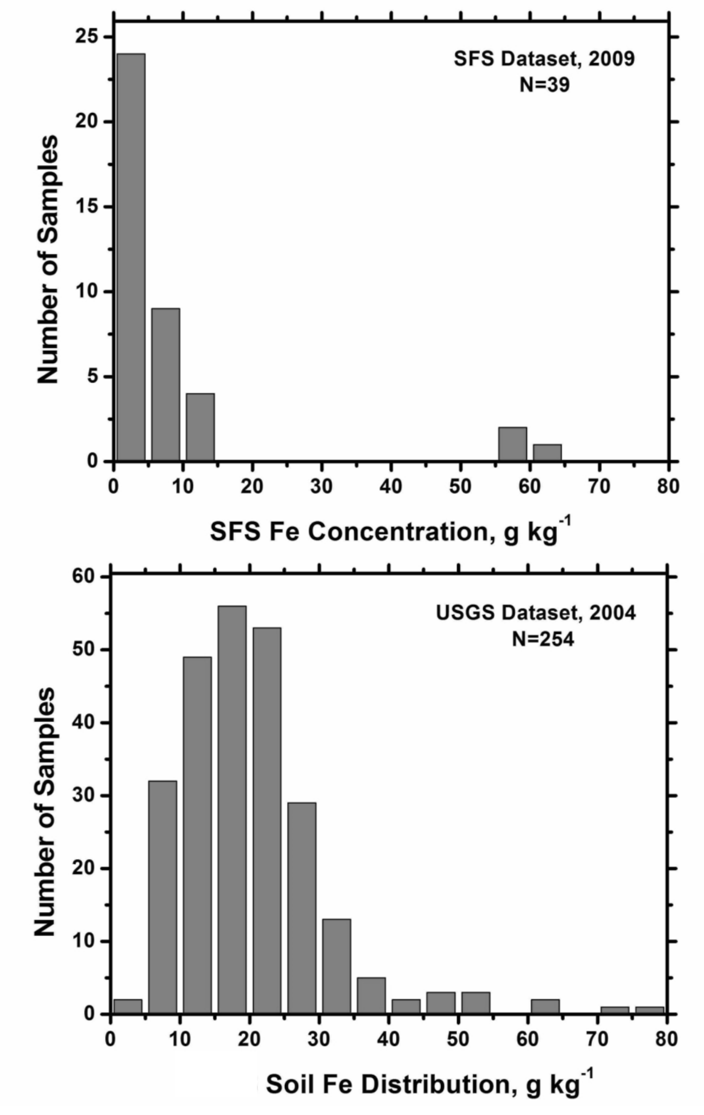

Figure 6-6. Concentration distributions of iron in SFS (top) and U.S. and Canadian soils (bottom).

6.7.6.4 Additional Factors

Iron is an essential micronutrient for all life. The behavior of iron and iron oxides in terrestrial systems is quite complex and specific to the characteristics (e.g., carbon content of soil) of the environment. Iron deficiency in many crops worldwide has led to numerous investigations over the past several decades in order to better understand and mitigate iron deficiencies in important crop plants. Iron deficiency has been associated mostly with alkaline soils, the presence of organic matter, soils with high Zn concentrations, the presence of bicarbonate anion ($HCO^{-}{}_{3}$), and have been noted especially in arid or semi-arid regions (Pais and Benton Jones, 1997; Kabata-Pendias, 2001). As an essential nutrient, iron is readily taken up by plants, usually in the form of the Fe^{2+} cation. When bound to a bio-chelating agent, Fe^{3+} uptake can also take place. The ability for plant roots to reduce Fe^{3+} to Fe^{2+} is one of the most fundamental processes in the absorption of iron in most plants. At excessive soluble iron concentrations, it can be phytotoxic to plants. Phytotoxicity is most likely to occur on strongly acidic soils, on acid sulfate soils, or flooded soils (Kabata-Pendias, 2001).

The normal iron content in plants ranges from 20 to 100 mg kg^{-1}, with a sufficiency range of 5-500 mg kg^{-1} (DW) (Pais and Benton Jones, 1997). Iron content in common foods ranges from approx. 8 to 40 mg kg^{-1} (Pais and Benton Jones, 1997), although higher concentrations in food plants have also been documented (e.g. some grasses and clover with concentrations up to 1000 ppm DW) (Kabata-Pendias, 2001). Kabata-Pendias (2001) summarize concentrations of iron in common food crops, with all values in ppm (FW): vegetables, 3 (celery root) - 31 (spinach); fruits, 1 (apples, honey melon) - 11 (black currant); cereals, 3 (barley pearls) - 37 (rye, whole grain); nuts, 11 (hazelnuts) - 47 (almonds).

6.7.6.5 Lines of Evidence

Iron is well documented as an essential micronutrient for all life, hence the general lack of health and environmental benchmarks for use in the screening comparison. The concentration distribution for iron in SFS indicates that, relative to native soils, SFS would not contribute iron content at a level that would approach phytotoxicity, even for acidic soils. The refined modeling generated SFS-specific screening levels orders of magnitude above concentrations found in SFS. Based on these results, iron levels in SFS soil are unlikely to cause adverse effects to human health or ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road subbase.

6.7.7 Manganese

The total manganese concentrations in silica-based iron, steel, and aluminum SFSs collected in June 2005 (39 of 39 detects) ranged from a minimum of 5.6 mg kg^{-1} to a maximum of 707 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 670 mg kg^{-1} (Dayton et al., 2010). No leach test data were available for manganese.

6.7.7.1 Comparison with Screening Levels

The relevant screening levels include Eco-SSLs, and the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL). These screening levels typically reflect study data on highly bioavailable forms of manganese:

- Eco-SSL (plants): 220 mg kg^{-1} soil
- Eco-SSL (soil invertebrates): 450 mg kg^{-1} soil
- Eco-SSL (mammals): 4,000 mg kg^{-1} soil
- Adjusted SSL (noncancer): 1,800 mg kg^{-1} soil (Residential SSL, adjusted to also address produce ingestion pathways)

Comparing the 95th percentile total concentration of manganese in SFS to the SSLs suggests that in a 1:1 manufactured soil blend, concentrations of manganese in SFS-manufactured soil would be below the Eco-SSLs for soil invertebrates and mammals, but exceed the Eco-SSL for plants. The 95th percentile manganese concentration in SFS-manufactured soil is well below the corresponding Adjusted SSL for the soil pathways; at a 50% blend, even the maximum manganese concentration in SFS-manufactured soil would be below the Adjusted SSL.

6.7.7.2 Modeling Results

Based on the results of the comparison screening levels, the soil manufacturing scenario (inhalation of fugitive dust emissions by nearby residents) and ecological receptors in the home gardener scenario were evaluated. For the inhalation exposure pathway, modeling results indicated that, up to a manganese concentration of 1,005 mg kg^{-1}, the potential for adverse human health effects would be below levels of concern. For the home gardener scenario, the refined ecological modeling results indicated that, up to a manganese concentration of 1,000 mg kg^{-1} SFS, ecological exposures would be below levels of concern.

The 95th percentile manganese concentration in SFS-manufactured soil (335 mg kg^{-1} DW) was above the Eco-SSL for terrestrial plants (220 mg kg^{-1} DW). This prompted an evaluation of the critical assumptions associated with the ecological hazard screen. One such assumption was that 100% of the manganese in SFS-manufactured soil would be available for plant uptake. To better represent the bioavailable fraction of manganese, the total manganese concentration in soil was adjusted by the pore water/total ratio as described in **Section 5.3.8.2**, creating a reasonably conservative estimate for the soil concentration that would be comparable with soil concentrations used in deriving the Eco-SSL for terrestrial plants. The refined ecological modeling results indicate that up to a manganese concentration of 1,000 mg kg^{-1} SFS, the potential for adverse effects to even the most sensitive ecological receptors would be below levels of concern. Therefore, adverse ecological effects from manganese in SFS are unlikely to occur for the home gardener scenario.

6.7.7.3 Soil Background Concentrations

Manganese is one of the most abundant trace elements in the lithosphere; its common range in U.S. soils is 20–3,000 mg kg^{-1} DW, with a mean value of 490 mg kg^{-1} DW (Kabata-Pendias, 2001). Studies on U.S. and Canadian surficial soils estimate that the median concentration of manganese is 490 mg kg^{-1} DW, with a range of 56–3,120 mg kg^{-1} DW (Smith et al., 2005). As illustrated in **Figure 6-7**, the composition of SFS with respect to manganese appears to be very similar to U.S. soils, suggesting that the addition of SFS to soil will not, in general, result in a significant change in soil manganese concentrations. In fact, the beneficial use of SFS would nearly always dilute manganese levels in the amended soils.

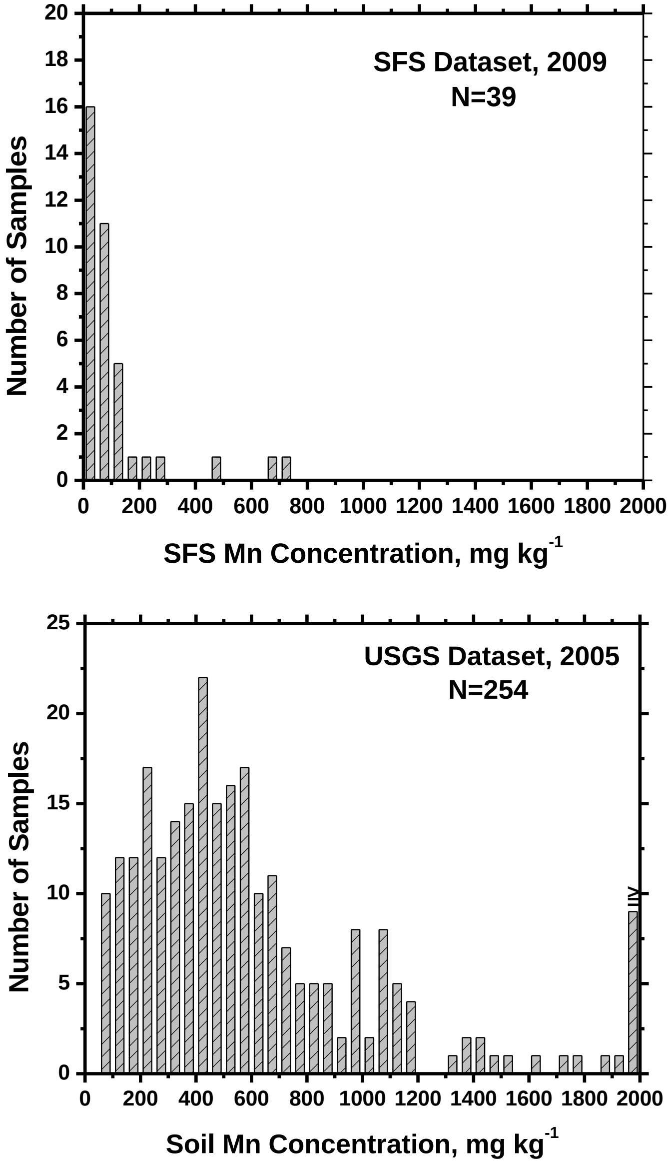

Figure 6-7. Concentration distributions of manganese in SFS (top) and U.S. and Canadian soils (bottom).

6.7.7.4 Additional Factors

All manganese compounds (e.g., MnO_2, Mn(II), Mn(III), and Mn(IV)) are very important soil constituents not only because manganese is essential to plant nutrition, but also because it influences the behavior of several other micronutrients. In addition, manganese affects several critical soil properties that strongly influence the soil redox system (Kabata-Pendias, 2001). The complex behavior of manganese in soil results in the creation of a large number of oxides and hydroxides; the physical features of these manganese compounds (e.g., small size of crystals with high surface area) have important geochemical implications. For example, the oxidation of arsenic, cobalt, chromium, vanadium, and selenium by manganese oxides is likely to be a controlling factor in the redox behavior of these elements in soils (Bartlett, 1986). The strong affinity of these metals to manganese oxides significantly reduces the bioavailability of other trace element metals (i.e., copper, lead, zinc) to plants.

Manganese is readily taken up from the soil and translocated within plants, and there is ample evidence that manganese uptake is metabolically controlled in a way that is similar to other divalent cation species, such as Mg^{2+} and Ca^{2+} (Kabata-Pendias, 2001). Because manganese is easily taken up by plants in its soluble form, the manganese concentrations in plants show a negative relationship with increasing soil pH and a positive relationship with soil organic matter (Kukurenda and Lipski, 1982). An excess of phytoavailable manganese is associated with strongly acid soils ($pH \leq 5.5$) and anaerobic conditions. Plant nutrient sufficient manganese ranges from 30–300 mg kg^{-1} (Kabata-Pendias, 1992). However, even though the manganese deficiency level for most plants ranges from 15–25 mg kg^{-1}, toxicity from manganese is highly variable due to great differences in species sensitivity, as well as the differences in soil characteristics, especially soil pH management (Andersson, 1987). Natural manganese phytotoxicity is one of the reasons that farmers must apply limestone periodically to correct and maintain pH near 6.5. Because the pH of SFS ranges from neutral to slightly alkaline, exceeding the highly conservative Eco-SSL for plants (95th percentile SFS concentration) is not necessarily a valid indicator for adverse effects in plants. In reality, at the typical application rates and pH that would be expected for SFS-manufactured soils used in home gardens, only a fraction of the manganese in SFS would be readily available to plants. Also, as discussed in **Chapter 2**, plant growth studies have found no negative impacts to plants grown in SFS or manufactured soils that include SFS (Dungan and Dees, 2007; Hindman et al., 2008; Dayton et al., 2010).

6.7.7.5 Lines of Evidence

For the home gardener scenario, the 95th percentile and maximum manganese concentrations in SFS-manufactured soil are below the Adjusted SSL for soil pathways. This indicates that manganese concentrations in SFS-manufactured soil are unlikely to cause adverse human health effects.

The results of the refined ecological modeling resulted in SFS-specific ecological screening levels for manganese ranging from 1,000 mg kg^{-1} SFS (90th percentile, soil invertebrates) to 9,500 mg kg^{-1} SFS (50th percentile, mammals). These SFS-specific ecological screening levels are well above even the maximum manganese concentration found in SFS.

Given the similarity between the concentration distribution of manganese in SFS and soil background levels, and no evidence of manganese toxicity in SFS plant growth studies, adding SFS to soil would not increase the likelihood of developing manganese-toxic conditions.

Based on the similarity in concentration distributions for manganese in SFS and background soils, as well as the results of the screening and risk modeling, manganese concentrations in SFS are unlikely to cause adverse effects to human health and ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road subbase.

6.7.8 Nickel

The total nickel concentrations in silica-based iron, steel, and aluminum SFSs collected in June 2005 ranged from a minimum of 1.1 mg kg^{-1} to a maximum of 117 mg kg^{-1} (using EPA method 3051A), with a 95th percentile value of 102 mg Ni kg^{-1} (Dayton et al., 2010). Using the SPLP leaching test, only one sample was above the detection limit of 0.05 mg L^{-1}, with a value of 0.238 mg L^{-1}. The concentrations in water extracts from the same samples (1 detect in 39 samples) were almost all below the detection limit of 0.05 mg L^{-1}; nickel was detected in one sample at the detection limit of 0.05 mg L^{-1} (Dungan and Dees, 2009). Sample-specific SPLP and water extract leachate data can be found in **Appendix B, Tables B-13 through B-18.**

6.7.8.1 Comparison with Screening Levels

The relevant screening levels include Eco-SSLs, the default Residential soil screening level for the protection of human health adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), and the Tapwater SL. The screening levels typically reflect studies based on nickel soluble salts:

- Eco-SSL (plants): 38 mg kg^{-1} soil
- Eco-SSL (soil invertebrates): 280 mg kg^{-1} soil
- Eco-SSL (mammals): 130 mg kg^{-1} soil
- Adjusted SSL (noncancer): 150 mg kg^{-1} soil (soil ingestion SSL, adjusted to also address produce ingestion pathways)
- Tapwater SL (noncancer): 0.3 mg L^{-1}

Comparing the 95th percentile total concentration of nickel in SFS to the SSLs suggests that, in a 1:1 manufactured soil blend the concentration of nickel would fall below the Eco-SSLs for soil invertebrates and mammals, but exceed the Eco-SSL for plants. This same nickel concentration in SFS-manufactured soil would be below the Adjusted SSL. Comparison of the SPLP and water extract data indicates that nickel concentrations associated with these tests would fall below the Tapwater SL.

6.7.8.2 Modeling Results

Based on the results of the comparison with screening levels, the soil manufacturing scenario (inhalation of fugitive dust emissions by nearby residents) and ecological exposure in the home gardener scenario were further evaluated. For the inhalation exposure pathway, the screening results indicate that, up to a nickel concentration of 1,005 mg kg^{-1}, adverse human health effects are unlikely.

As discussed in Section 5.3.8, the phytotoxicity of metals depends on the soluble soil fraction and, therefore, the actual hazard posed to terrestrial plants depends on the amount of metal that can desorb from SFS particles and become available in the soluble fraction. To better represent the bioavailable fraction of nickel, the total nickel concentration in soil was adjusted by the pore water/total ratio as described in Section 5.3.8.2, creating a reasonably conservative estimate for the soil concentration that would be comparable with soil concentrations used in deriving the Eco-SSL for terrestrial plants. The refined ecological modeling results indicate that up to a nickel concentration of 290 mg kg^{-1} SFS (i.e., almost twice the 95th percentile nickel concentration in SFS), adverse impacts to ecological receptors would be unlikely.

6.7.8.3 Soil Background Concentrations

The background concentrations of nickel in soil range from <5–150 mg kg^{-1} soil, with mean values on the order of 15–35 mg kg^{-1} soil across a wide range of U.S. and Canadian soils

(Smith et al., 2005). As illustrated in **Figure 6-8**, average nickel concentrations in SFS are well within this range. The 95th percentile nickel concentration in SFS of 102 mg kg^{-1} falls within this normal background range. Given the importance of site-specific soil properties such as pH level, the comparison between nickel concentrations in SFS and soil background suggests that average concentrations overlap significantly, and that the median concentrations of nickel in SFS are very similar to median concentrations of nickel in native soils.

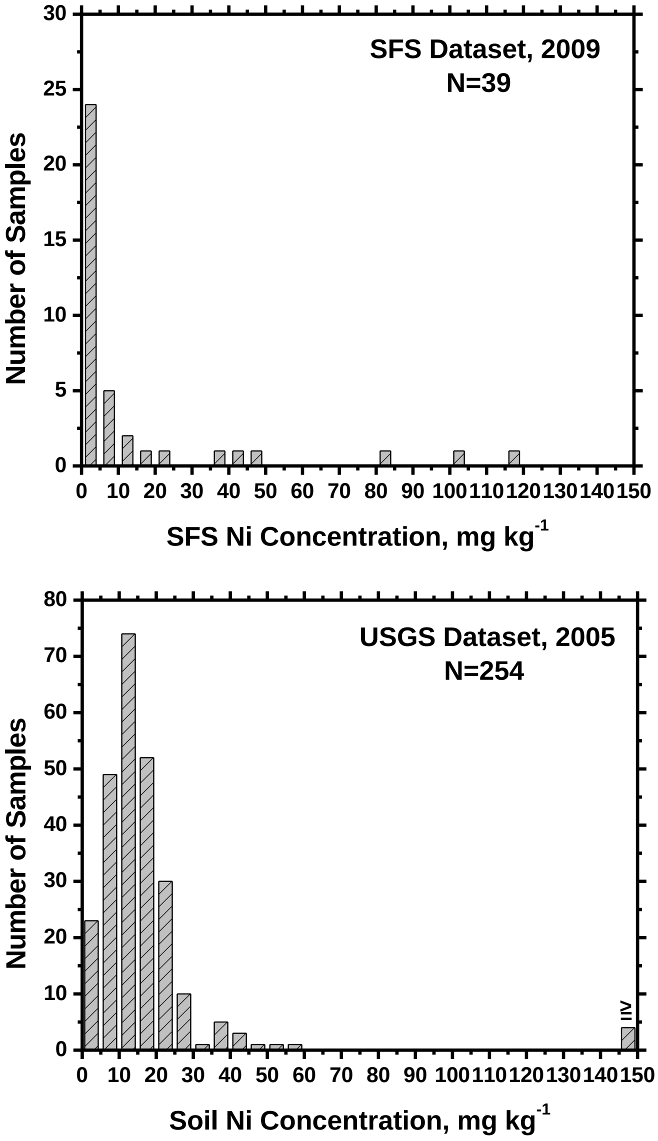

Figure 6-8. Concentration distributions of nickel in SFS (top) and U.S. and Canadian soils (bottom).

6.7.8.4 Additional Factors

Recent research on nickel shows that this metal is an essential nutrient for plants (e.g., Wood et al., 2004). Nickel is readily and rapidly taken up by plants, and up to phytotoxic levels in plant tissue, there is a positive correlation between soluble soil nickel concentrations and plant concentrations of nickel. The soil pH appears to be the controlling factor with regard to nickel mobility, bioavailability, and toxicity; increasing soil pH can significantly reduce the nickel content and reduce the potential for plant toxicity. In soils that are near neutral pH, nickel can undergo rapid reaction to form less soluble and less bioavailable forms. When soluble nickel compounds are mixed with soils, the nickel hydrated cations rapidly enter into the soil chemistry, forming adsorbed forms on iron and manganese oxides and chelated forms with soil organic matter (e.g., Singh and Jeng, 1993). Then other soil minerals dissolve and nickel reacts to form new soil minerals, such as nickel-silicates and nickel-aluminum layered double hydroxides (LDHs - see Appendix A for a more detailed discussion of LDHs). These prevent leaching and strongly limit potential uptake or phytotoxicity of nickel in contaminated soils with 1,000 mg kg^{-1} nickel or higher (Kukier and Chaney, 2004; Siebielec et al., 2007). Therefore, because SFS and manufactured soils are near neutral pH, the bioavailability of nickel is likely to be very low.

Although the transport and storage of nickel seem to be metabolically controlled, nickel is mobile in plants and is likely to be accumulated in both the leaves and seeds (Kabata-Pendias, 2001). The mechanism of nickel toxicity in plants is poorly understood, although restricted growth and injury (e.g., chlorosis) have been observed for decades. In general, concentrations in plants of 10–100 mg kg^{-1} (DW) have been shown to be phytotoxic. Sensitive species are affected at lower foliar concentrations (e.g., 10–30 mg kg^{-1}), while rare nickel hyperaccumulators can contain nickel concentrations well into the thousands of mg kg^{-1}. Typical nickel concentrations in produce (fruits and vegetables) are found in the range of 0.6–3.7 mg kg^{-1} (DW), although plants grown at nickel-contaminated sites may accumulate significantly higher levels of nickel depending on the adaptation of plants, the form of the nickel in the contaminated soils, and other site-specific soil characteristics (especially the pH).

6.7.8.5 Lines of Evidence

The results of the screening comparisons for human health indicate that nickel levels in SFS were below levels of concern for the groundwater pathway and soil/produce pathways. Therefore, nickel concentrations in SFS are unlikely to cause adverse human health effects through dermal contact with or ingestion of groundwater, soil, and home-grown produce.

The inhalation hazard to nearby residents was shown to be well below a level of concern, with modeled inhalation screening concentrations close to 100 times above the 95th percentile nickel concentration in SFS. Therefore, nickel concentrations in SFS are unlikely to cause adverse human health effects through inhalation.

Refined ecological modeling results in SFS-specific ecological screening levels ranging from 290 mg Ni kg^{-1} (90th percentile, mammals) to 5,100 mg Ni kg^{-1} (50th percentile, terrestrial plants). These SFS-specific ecological screening levels are above even the maximum concentration of Ni found in SFS.

Given the similarity between the concentration distribution of nickel in SFS and soil background levels, adding SFS to soil would not significantly alter the nickel content in native soils.

Based on the similarity in concentration distributions for nickel in SFS and background soils, as well as the results of screening comparisons and screening modeling, nickel concentrations in SFS are unlikely to cause adverse effects to human health and ecological receptors when SFS is used in SFS-manufactured soils, soil-less potting media, or road subbase.

6.7.9 Other Metals

6.7.9.1 Lines of Evidence

Appendix A describes a substantial body of research on the behavior of metals in soils with respect to mobility (e.g., sorption and desorption), bioavailability (e.g., metal species, oxides), phytotoxicity (e.g., soil levels that damage plants), and toxicity to animals and soil invertebrates (e.g., nature and severity of potential effects). This information is critical in determining whether or not these other metal constituents in SFS pose a potential risk to human health and the environment when beneficially used in soil-related activities.

To complement the information provided in **Appendix A**, **Table 6-5** presents a summary of the available data on various metals with respect to their potential for release to the environment at levels of concern. The table compares a 1:1 manufactured soil blend using the 95th percentile concentration in SFS with the Residential SSL adjusted to also address home gardener produce ingestion pathways (Adjusted SSL), the inhalation screening level, and the 50th percentile background concentration in soil. This constitutes a conservative comparison because (1) actual soil blends are likely to include less than 10% SFS (Personal communication, USDA/ARS[51]), so the 1:1 blend is highly unlikely, and (2) the SSLs make very conservative assumptions with respect to exposure (e.g., 100% of incidentally ingested soil comes from the SFS-manufactured soil). The concentrations in the 1:1 SFS-soil blend do not exceed the ingestion or inhalation SSLs for any constituent; therefore, it appears highly unlikely that either of these pathways will pose a significant risk to human health. The limited leach test data suggest that the metals that were tested (barium, beryllium, cadmium, lead and zinc) do not pose significant risks via the groundwater pathway and, in fact, only one of the metals (barium) was present above the detection limit in the SPLP leach test.

Finally, comparing the soil blend to the 50th percentile background concentrations suggests that molybdenum is present at levels in SFS that might result in an increase in the soil concentration. However, these concentrations are still well within the range of background levels, and moreover, the research discussed in **Appendix A** strongly suggests that, at the concentrations shown in Table 6-5, the availability and toxicity of molybdenum would be very low under a wide range of soil conditions.

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[51] Personal communication, April 2009, Timothy Taylor, U.S. EPA, with Rufus Chaney, USDA-ARS.

Table 6-5. Summary of Other SFS Metal Concentrations and Relevant Screening Criteria
(mg kg^{-1} unless otherwise noted)

SFS Constituent	95%-ile of 1:1 Soil:SFS blend	95%-ile SPLP/ ASTM	Above Adjusted SSL?	Above Inhalation SSL?	Above Eco-SSL?	Above Ground water Screen?[a]	Above 50%-ile Background?	Above 95%-ile Background?
Al (g kg^{-1})	5.6	—	No	No	—	—	No	No
B	10.1	—	No	—	—	—	NA	NA
Ba	6.9	0.37	No	—	No	No	No	No
Be	0.19	<0.02	No	No	No	BDL	No	No
Cd	0.1	<0.01	No	No	No	BDL	No	No
Mo	10.9	—	No	—	—	—	Yes	Yes
Pb	7.65	<0.11	No	—	No	BDL	No	No
Se	0.10	—	No	—	No	—	No	No
Tl	0.05	—	No	—	No	—	No	No
V	4.95	—	No	—	No	—	No	No
Zn	36.1	<0.22	No	—	No	BDL	No	No

BDL = below detection limit.

NA = not available.

[a] All groundwater screening levels used in this assessment are listed in Table 4-2.

6.8 Uncertainty Characterization

The goal of this report was to bring together risk screening modeling and the best available science to provide industry, consumers, and regulatory agencies with the scientific basis to determine whether certain soil-related beneficial use applications of SFS are appropriate and protective of human health and the environment. This lines of evidence approach, therefore, includes two basic components that will be discussed in this uncertainty characterization: (1) uncertainties associated with the conduct and interpretation of the risk screening modeling, and (2) uncertainties associated with the state-of-the-science research on the behavior of metals and other SFS contaminants in soils.

6.8.1 Risk Screening Modeling

In the *Guidance for Risk Characterization* developed by EPA's Science Policy Council (U.S. EPA, 1995c), EPA defined the high end of the risk distribution as being at or above the 90th percentile risk or hazard estimate generated during Monte Carlo simulation. The high end of the risk distribution for risk screening modeling refers only to hypothetical individuals living within the areas of "economic feasibility" for SFS use that may

- Live near roadways that were constructed with SFS as a component of the subbase
- Live near facilities that manufacture soils and soil-less media by blending SFS with other ingredients
- Incorporate SFS-manufactured soils into their home garden and consume a large fraction of fruits and vegetables from the home garden.

The conceptual model for each of these three scenarios was described in **Chapter 3, Section 3.1.4**. At a minimum, the risk screening modeling was designed to ensure that 90% of the individuals associated with these high-end exposure scenarios would not be exposed to constituents in SFS above the screening levels or benchmarks. In addition, the risk screening modeling also used conservative ecological screening criteria, the Eco-SSLs, to ensure that ecological receptors (e.g., plants, soil invertebrates, and mammals) are not exposed to constituent levels above the criteria levels. However, the receptors considered in this assessment are hypothetical, and the modeling reflects exposures that are almost certain to be well above the 90th percentile of the distribution. In fact, given the conservative nature of the modeling, the modeling results provide bounding estimates of risk that fall at the extreme tail of the distribution. Therefore, this discussion is focused on better understanding the key sources of conservatism in the input data and scenario assumptions that EPA developed to ensure that the modeling results would not underestimate the potential risks associated with SFS. There are considerable uncertainties in the modeling risk estimates. However, these estimates are conservative by design, and the uncertainties in the assumptions and selection of input data bias the risk predictions heavily toward the overestimation of risk.

Roadway Subbase. The use of SFS as a component in roadway subbase was addressed through the evaluation of subbase-relevant exposure pathways (i.e., groundwater ingestion and inhalation of fugitive dust) in a use scenario likely to cause greater exposure – SFS-manufactured soil use in a home garden.[52] Once in place as subbase, the only exposure pathway of potential concern would be leaching of constituents into the subsurface following fracturing of the road surface (allowing rainwater infiltration through the underlying materials) or mounding of a high water table. For almost all constituents, the leach test data (except perhaps that from the ASTM shake method) provide extreme conditions that will not occur under the roadway. Even under these conditions, very few constituents had leach test results above detection limits. For those constituents that demonstrated an ability to leach from SFS, the groundwater screening showed that the potential for these constituents to reach receptors at levels of concern is extremely low. Thus, the demonstration of low leachability even under extreme conditions, along with the conservative groundwater modeling provides a high level of confidence that this pathway will not be of concern.

Similarly, the inhalation screening modeling used a series of conservative assumptions ranging from the emission factors to placing the receptors in the downwind plume of the maximum air concentration. These bounding results demonstrated that the protective concentrations of chemical constituents found in SFS were higher—in many cases orders of magnitude higher—than the actual constituent concentrations found in SFS. Due to the transitory nature of storage piles of SFS during roadway construction, the pathways associated with delivery to nearby streams (after windblown emissions and runoff) were considered to be essentially incomplete. That is, as with other typical roadway construction components, the storage piles are not retained for sufficient periods to result in a significant mass transport to local waterbodies. These materials are valuable, and it was assumed that storage piles would exist for a few days (at most) before being incorporated into the subbase. The relatively large SFS grain size and very low leaching potential of constituents in SFS further supports the

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[52] Though the groundwater modeling was performed for the manufactured soil use scenario rather that road subbase, modeled inputs (e.g., distance to drinking water well) were more conservative than road subbase inputs. The findings from the manufactured soil scenario are therefore also protective of the road subbase use scenario.

contention that (1) very little mass of material could be transported from a storage pile, and (2) the constituents found in the SFS are tightly bound in the sand matrix and not very available at environmental pHs in the aquatic environment.

Manufacture of Blended Soils and Soil-Less Potting Media. Soil blending operations that use SFS pose a potential inhalation risk due to the large volumes of SFS piles that would likely be required to support such operations. As suggested by the conceptual model, leaching of constituents, and inhalation following volatile and particulate emissions, are potential concerns. Thus, the combination of leach test data, inhalation screening, and probabilistic groundwater modeling was used to screen for potential risks. The manufacture of blended soils presents low risks to human health at the 95th percentile constituent concentrations found in SFSs. For this scenario, it was assumed that runoff would not be a pathway of concern because manufacturing facilities would impose basic controls (e.g., berms) to avoid losing valuable ingredients to the soil blending process, and generally states require facilities to institute stormwater controls to prevent significant levels of chemical constituents from being directly discharged into nearby surface waters. Furthermore, it was assumed that deposition from soil-blending emissions would not contribute significantly to the surface soil layer and ecological exposures when compared to SFS use in home gardens. Therefore, given the highly conservative assessment of risks associated with soil manufacturing, the potential for adverse health effects is considered unlikely.

Use of SFS in Home Gardens. The use of SFS-manufactured soils by home gardeners could pose potential risks through inhalation, incidental ingestion of the soil, the consumption of home-grown fruits and vegetables grown in soil containing SFS, or groundwater impacted by garden leachate.

As shown by the comparison of the 95th percentile constituent concentrations in SFS to inhalation screening concentrations for the SFS storage pile (see Table 4-4), the inhalation pathway was screened out by the deterministic modeling of air releases from SFS storage piles. These results also screened out the inhalation pathway for the home garden scenario because they represent a scenario in which SFS-manufactured soil was used in a home garden as top dressing with no mixing or dilution. This is a highly conservative assumption because, in practice, SFS-manufactured soils will be mixed with native soils, thereby diluting the constituent concentrations in the SFS. Thus, comparing the 95th percentile constituent concentrations in SFS with the inhalation pathway screening concentrations demonstrates that the inhalation exposures for the home garden scenario also will be below levels of concern.

Therefore, a screening modeling scenario was developed for the use of SFS-manufactured soils in the home garden that addressed both the incidental ingestion of constituents in SFS, as well as the consumption of contaminated groundwater and produce from the garden. A Monte Carlo simulation was implemented to assess human and ecological exposures under the home gardener scenario. As discussed in **Section 5.1**, the implementation does not distinguish between uncertainty and variability. In essence, input parameters were selected to represent variability (e.g., exposure factors), and in some cases, to also represent the uncertainty in the true parameter value (e.g., soil-specific parameters). Previous chapters of this document describe how input distributions and input values were developed and used to estimate risk. Use of these inputs in a national level assessment may result in an underestimation or overestimation of risk. To ensure that the Monte Carlo simulation was highly conservative and produced a bounding estimate of risk, several assumptions were built into the modeling scenario.

First, the assumption of a 1:1 SFS-soil blend in a single-application "soil replacement" scenario is conservative. It is possible that this blend could be used to replace the topsoil in small home gardens, or that this blend could be used multiple times in smaller volumes to amend existing local soils. However, the amount of SFS required to replace the top soil layer with this blend in a home garden capable of producing a significant proportion of the home-grown diet of fruits and vegetables is quite large. The costs of SFS-manufactured soil for a 0.1-acre garden would be on the order of $2,300 (assuming approximately $21.50 yd^{-3} delivered). In all likelihood, these costs would be prohibitive, and the home gardener would use smaller SFS-manufactured soil application rates or seek alternative methods to improve the physical and chemical properties of the soil for large gardens.

Second, the consumption rates of fruits and vegetables sampled during the probabilistic modeling were based on EPA's *Exposure Factors Handbook* (U.S. EPA, 2011). The distribution for each category of produce (e.g., exposed vegetables) was based on actual survey data; however, these distributions are sampled independently, even though there is very likely a correlation among the consumption of different types of produce. It would be unlikely that a person would consume a high-end amount of root vegetables *and* leafy greens *and* apples that were all grown from the same garden because (1) all types of produce cannot be grown in the same season, (2) there are regional characteristics (e.g., soil type, precipitation) that strongly influence what types of crops can be grown, and (3) there are agronomic limits as to how much produce can be grown, harvested, and consumed that are not reflected in the exposure factor data. Thus, the total ingestion risks tend to overestimate the likely consumption of home-grown produce. For example, in EPA's deterministic risk assessment of chemical pollutants in biosolids conducted in 1993 (U.S. EPA, 1993), the estimated consumption rate of home-grown fruits and vegetables was 105 g (WW) d^{-1} for an average adult (not including tree fruits). In the probabilistic modeling conducted for this assessment, the total consumption rate of home-grown fruits and vegetables for the adult at the 90th percentile risk level was approximately 500 g (WW) d^{-1} for an average adult. Also, it is not possible to harvest most garden crops for more than a short period when the crop is ripe, which considerably limits potential exposure to garden foods. Given the size of the garden required to support such a diet, the costs of delivering SFS-manufactured would likely reduce the actual exposure by several orders of magnitude due to the limited garden area. Thus, the results of the home gardener refined modeling should be considered as an overestimate of the actual risks.

In addition, evaluation of the home gardener groundwater pathway with IWEM and EPACMTP incorporated several conservative assumptions, including the placement of the drinking water receptor well adjacent to the edge of the garden. Considering that the U.S. EPA estimates that only 15% of the U.S. population have their own drinking water sources (U.S. EPA, 2002f) and the fact that modeling identified the 90th percentile groundwater well concentration, the applied approach ensures that the results of this analysis can be used to confidently determine if the applications of SFS will be protective of human health and the environment in the United States.

In summary, the uncertainties associated with the screening and refined risk modeling bias the results to produce overestimates of the potential risks associated with the three exposure scenarios of interest. Although the accuracy of the screening modeling could be increased by making less conservative assumptions and developing additional data inputs for the models, the modeling results are appropriate for their intended purpose—to ensure with a high level of confidence that the risk estimates are health protective. Given the level of conservatism in the modeling assumptions and inputs, the use of SFS in the soil-related applications discussed in this report would not be expected to pose significant risks to human health or the environment.

Human Health Benchmarks. There is uncertainty inherent in the development of the human health benchmarks used in this risk assessment. Uncertainty that is typically associated with human health benchmarks is discussed in detail in EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005c), *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994a), and IRIS (U.S. EPA, 2012a). With regard to the application of human health benchmarks developed by EPA for risk assessment purposes, U.S. EPA (2005c) states that "…the primary goal of EPA actions is protection of human health; accordingly, as an Agency policy, risk assessment procedures, including default options that are used in the absence of scientific data to the contrary, should be health protective." Thus, EPA acknowledges the uncertainty associated with the use of point estimates for human health benchmarks, but also recognizes the Agency's responsibility with regard to the protection of human health in addressing this uncertainty.

Ecological Benchmarks. There is uncertainty inherent in the development of the ecological screening level benchmarks used in this risk assessment (i.e., Eco-SSLs). Like their human toxicity counterparts, Eco-SSLs are conservative screening values. For example, use of conservative modeling assumptions (e.g., metal exists in most toxic form or highly bioavailable form, high food ingestion rate, high soil ingestion rate) in the Eco-SSL derivation process leads to some Eco-SSLs that are below the average background soil concentration. As screening values, users can be confident that if soil concentrations fall below Eco-SSLs, then no further evaluation is necessary.

Eco-SSLs for terrestrial plants, soil invertebrates, and small insectivorous mammals were applied to evaluate exposures to ecological receptors under the home garden scenario. Avian Eco-SSLs were deemed not applicable to the home garden scenario for several reasons. First, it is highly likely that the home gardener will adopt measures (e.g., fencing, netting) that would limit potential exposure for birds. Second, the home ranges for most birds that are either included or represented by the Eco-SSLs are significantly larger than a 0.1 acre (405 m^2) garden. The woodcock, for example is reported in U.S. EPA 1999c as having a mid-point home range of 857,500 m^2. Therefore, the impact attributable to home gardens on reproductive fitness of avian populations is likely to be negligible.

6.8.2 State-of-the-Science on SFS

This report presents a tremendous amount of information on SFS characteristics, uses, and the behavior of SFS constituents in the environment, particularly the metals and metalloids. Where the soil uses are being considered, this information speaks to one important question—namely, is SFS significantly different than native soils.[53] Clearly, the demonstration that SFS is similar in its composition and properties to that of background soil may question the need for risk screening modeling. However, there is variability in the properties of SFS and there is variability in the properties of background soils, and as a result, the use of this information in answering this core question is associated with some level of uncertainty. There are aspects of

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[53] A comparison of other materials used in manufactured soils or road base (including native sand) is also relevant, but beyond the scope of this evaluation.

uncertainty that are not addressed in the screening modeling that can only be considered as they relate to the research and scientific findings presented in this report and appendices. This section addresses several broader aspects of the uncertainty, given the current state-of-the-science, that are highly relevant to the interpretation of potential risk associated with the beneficial use of SFSs in certain soil-related applications. To provide the context for this discussion, four questions are posed that directly relate to the core question in this assessment.

1. ***Are the analytical data on SFS representative of SFSs that will be beneficially used in the soil-related applications addressed in this report?***

The analytical data on total constituent concentrations and leach test data were developed to represent the specific types of sands that have been identified for soil-related beneficial uses. These sands include SFS from iron, steel, and aluminum foundries that were repeatedly used in the molding process; though the initial survey included sampling SFS from nonleaded brass foundries, the risk evaluation did not include SFS from brass or bronze foundries. The data include SFS samples from 39 foundries in 12 states that were specifically selected to ensure that the full range of constituents and concentrations for these types of sands were adequately represented. Given the similarity in molding processes for these types of foundries, both in terms of the input materials used and the reclamation/reuse practices, the analytical data are believed to represent the range of constituent concentrations and the distribution of those concentrations in foundry sand. Nevertheless, it is unknown if the SFS samples from these 39 foundries are statistically representative of SFS from all iron, steel, and aluminum foundries. The related data may, therefore, overestimate or underestimate the range and distribution of SFS constituent concentrations.

2. ***Are the data presented by Smith et al. (2005) representative of background soil concentrations of metals in the areas of economic feasibility?***

The data presented by Smith et al. (2005) represent the USGS's attempt to systematically characterize the background concentrations of metals in the U.S. and Canadian soils. The authors noted that

> "The transects were located to cross multiple climatic, topographic, physiographic, land use, geologic, pedologic, and ecological boundaries. This imposes rigorous field testing of sampling protocols across a wide range of conditions. The generated data will allow estimation of geochemical and microbiological variation at a continental scale." (Smith et al., 2005)

The Smith et al. (2005) data on background concentrations of metals in soil were compared to a variety of other sources of background data for the United States summarized in *Trace Elements in Soils and Plants–Third Edition* (Kabata-Pendias, 2001), as well as in EPA's Attachment 1-4: *Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs)–Review of Background Concentrations for Metals* (U.S. EPA, 2003e). Based on a visual inspection of the data in these respective sources, the data presented by Smith et al. track well with work performed by a number of different sources (e.g., U.S. EPA, 2003e included information developed by states, as well as under the Comprehensive Environmental Response, Compensation, and Liability Information System [CERCLIS]), particularly with respect to the minimum, 95th percentile, and mean values for metal concentrations in soil. The overlap in data on metal concentrations suggests a high level of confidence in the use of the Smith et al. (2005) data to represent background concentrations at a continental scale.

In considering the similarity between SFS and native soils, it is important to recognize that the use of continental or east-west regional data on background soil concentrations represents a source of uncertainty. Because the soil concentrations are variable, it is uncertain whether a specific application of SFS will introduce metals above background levels at a specific location. From a risk assessment standpoint, it was determined that, independent of background levels, the introduction of metals associated with soil-related applications of SFS is unlikely to cause adverse effects to human health and ecological receptors. For the purposes of interpreting the general impacts of soil-related applications with respect to background, the most appropriate comparisons are to (1) consider the entire empirical distributions of metals in SFSs and in native soils, and (2) compare the metal concentrations in SFS products at the high end of the distribution (i.e., 95th percentile) to robust measures of background soil concentrations (i.e., the 50th percentile). The former provides important insights regarding the nature of the respective materials with respect to metals, and the latter provides a statistical indication of the probability of an SFS application exceeding typical background concentrations. Given these comparisons, as well as the results of the conservative risk screening modeling, the uncertainty inherent in using background concentrations at scales above what is expected at local levels is not considered to be significant.

3. ***How will the soil characteristics affect the bioavailability, mobility, and toxicity of metals in the soil-related applications of SFS addressed in this report?***

As discussed throughout this report, the bioavailability of most metals tends to increase with decreasing pH, particularly for acidic soils in the range of pH 4. Given the variability in soil pH, with decreasing pH associated with the use of SFSs in areas that are closer to the East Coast, evaluating the potential impacts of adding SFS to soils at the low end of the pH range is associated with some level of uncertainty. With regard to the leaching potential of metals, the SPLP leach test reflects acid rain conditions, and considering the low levels found, these conditions are not anticipated to significantly alter the leaching potential of metals in SFS. The groundwater pathway screening is sufficiently conservative to state with a high degree of confidence that pH variability will not drive risks due to groundwater ingestion above the levels of concern.

With regard to the home gardener scenario, if SFS-manufactured soils were applied in regions with lower pH *and* assuming that the home gardener did not lime the soil, the uptake and translocation of metals into plants could be increased. Depending on the form of the metal, this could result in higher phytotoxicity or accumulation of metal at higher rates for more tolerant plant species. In addition, the more mobile and toxic metal species may cause adverse effects to invertebrates in the garden soil. Although these effects could occur, the variability in soil pH is not regarded as a significant source of uncertainty for the home gardener scenario for three reasons. First, the size of the garden would have to be relatively large to support the consumption rates used in the evaluation, and as previously discussed, the economics and physical attributes of such large SFS applications would prohibit the blend from reaching 50%. Second, it is reasonable to assume that home gardeners have sufficient experience in cultivating produce to routinely monitor and improve the quality of their soil; this would almost certainly include liming in many of the low pH regions in the east, thereby minimizing the impact of potentially low soil pH on plant health and productivity. Third, in accordance with the soil-plant barrier, soil acidity at pH <5.2 would result in natural soil aluminum phytotoxicity, thereby preventing plant growth and protecting the food chain.

4. ***Are the chemical and physical properties of metals in SFS similar to the chemical and physical properties in native soils?***

With respect to the distribution of metals concentrations in SFS, the data presented in this report indicate that metal concentrations in SFS are generally relatively low compared to the soil background levels at a national scale. However, the concentrations of metals do not, by themselves, indicate whether SFS is similar to soil with respect to how those metals behave. Specifically, the concentration data do not indicate whether the forms of metals in SFS are more mobile, bioavailable, or toxic than those same metals in native soils. Although this is a potential source of uncertainty, three pieces of information strongly suggest that metals in SFS will behave in a very similar manner as metals in native soils.

First, the leach test data on SFS indicate that even under very acidic conditions (representative of a landfill), the metals in SFS demonstrate a very low potential to leach out of this material. Of the very few metals that either demonstrated some leaching potential (arsenic) or had detection limits above the screening criteria (antimony, beryllium, cadmium), the conservative risk screening (e.g., using the 95th percentile leach test concentration) demonstrated that these metals would not pose a significant risk via the groundwater ingestion pathway. Given the similarity between the background concentrations of these metals and the concentrations in SFS, this result indicates that the risks to background concentrations should also be very low.

Second, the most commonly used sand is silica sand (silicon dioxide, SiO_2) because of its wide availability and relatively low cost; this material is a component of native soils.[54] **Section 2.5** describes the "soil-like qualities" of SFS that make this material a valuable soil amendment; these properties include, for example, desirable chemical (e.g., pH, salinity) and physical (e.g., texture, water holding capacity) characteristics that are typical of high-quality soils.

Third, because soil-related applications of SFS are likely to be used in aerobic soils that are typical of home gardens, it is reasonable to assume that the cationic form of many of the trace elements in SFS will be the predominant form. As discussed in **Appendix A**, Section A.1.1.2, complexation of trace metals with amorphous iron and manganese hydrous oxides (both of which are available in SFS) is common in aerobic soils; in addition, the cationic forms of a number of metals in SFS can be expected to sorb to soil organic matter and other forms of humified natural organic matter, reducing the solubility of the metals in the soil. The behavior of metals in SFS added to aerobic soils would, therefore, be expected to be similar to the behavior of metals already present in the soil. Further, the increased availability of iron and manganese in SFS may actually decrease the solubility and availability of trace metals originating from both native soils and SFS due to adsorption on oxides. In consideration of the information on leaching potential, the soil-like qualities of SFS, and the chemical behavior of metals in SFS once added to aerobic soils, it appears very likely that the behavior of metals in SFS would be similar, if not indistinguishable, from the behavior of metals in the native soils to which the SFS is added.

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[54] Sands, including silica sand, are also frequently used in manufactured soil and road subbase.

7. Findings and Conclusions

This report presents an extensive review of information on SFSs, including analytical results for metals and metalloids (including both totals and leach test results), PAHs, phenolics, dibenzodioxins and furans, and dioxin-like PCBs. It also includes deterministic risk screening model results for the inhalation exposure scenario and probabilistic screening and refined model results for the home gardener exposure scenario. Taken together, this information provides the scientific basis for decision makers to determine the appropriate soil-related applications for certain unencapsulated beneficial uses of SFS. The major findings and conclusions from this report as they pertain to silica-based SFSs produced by iron, steel, and aluminum foundries, and their use in manufactured soil, soil-less potting media, and road subbase, are summarized below.

7.1 Beneficial Use of SFS (Chapter 1)

- SFS is a valuable industrial byproduct, and therefore, there are economic and possibly environmental advantages to identifying which soil-related applications are appropriate SFS beneficial uses.
- State regulators need access to sound scientific data and analyses to support the decision-making process regarding the beneficial use of SFS.

7.2 Characterization of SFS (Chapter 2)

- SFS has a number of soil-like qualities that make it an attractive material for use in roadway subbase, soil-less media, and manufactured soils.
- The concentrations of organic constituents and trace elements (including metals and metalloids) are, on average, very low in silica-based SFS produced by iron, steel, or aluminum foundries.
- Published background concentrations of metals in soils provides additional information in evaluating the scientific basis for considering the implications of adding SFS as soil amendments.
- The current data on SFS show that the distributions of metal constituents in silica-based SFS from iron, steel, and aluminum foundries are very similar to the background distributions of metals in native soils.
- The presence of manganese and iron and the neutral pH of SFS strongly suggest that soil-related applications will likely reduce the mobility, bioavailability, and toxicity of metal constituents in SFS and, possibly, metal constituents already in the soil.
- Although applications of SFS in strongly acidic soils (pH <5) could increase the mobility of metals, this increase would mirror the same increase in natural soil. The common agricultural practices of testing pH and liming to ensure good crop growth conditions are expected to preclude highly acidic conditions from occurring.
- Based strictly on a comparison between the SFS and background concentrations of metals, it is unlikely that the addition of silica-based SFS from iron, steel, and aluminum foundries would significantly alter the composition of soil.

7.3 Exposure Scenarios Examined (Chapter 3)

- Three exposure scenarios that reflect the unencapsulated beneficial use considerations for SFS, as well as the potential for complete exposure pathways, included (1) use as subbase in roadway construction, (2) use in soil-less potting media, and (3) blending in manufactured soils.

7.4 Screening of Exposure Pathways (Chapter 4)

- The inhalation pathway screening indicates that even high-end concentrations of the constituents in SFS were well below screening values for all constituents for which inhalation benchmarks were available.
- The groundwater ingestion pathway screening indicates that even high-end concentrations of metal constituents in SFS were below water quality screening criteria for all constituents for which such criteria were available, except antimony, arsenic, beryllium, cadmium, and lead.
- The soil ingestion pathway screening indicates that even high-end concentrations of metal constituents in SFS were below soil screening criteria for all constituents for which such criteria were available, except antimony, arsenic, chromium III, cobalt, copper, iron, manganese, and nickel.

7.5 Modeling of Exposures from Home Gardening (Chapter 5)

- Eight metals (antimony, arsenic, chromium III, cobalt, copper, iron, manganese, and nickel) were evaluated with probabilistic screening modeling and refined modeling. Arsenic, cobalt, and iron were evaluated for human exposures through the soil/produce ingestion pathway but, only arsenic was evaluated under the groundwater pathway. Although concentrations of manganese and nickel in SFS were below their respective human health screening criteria (described in Chapter 4), they were modeled in the home gardening scenario because of their high potential for phytotoxicity. Similarly, concentrations of antimony, trivalent chromium, and copper were below their human health screening levels, but they were retained for further study due to the potential to impact small insectivorous mammals.
- One of the more conservative assumptions for the home gardener soil/produce pathway screening modeling was the addition of exposures across all five produce categories (e.g., exposed vegetables), which results in consumption rates for the home gardener that are well above expected values.
- Investigation of the influence of produce consumption rates suggests that adding across produce categories is likely more appropriate for the median consumption rates for the home gardener, and that the use of values at the tail of the exposure factor distributions is associated with higher levels of uncertainty.
- The refined groundwater modeling used the distribution of the home garden source model outputs (i.e., leachate fluxes and annual average leachate infiltration rates) as input to the groundwater model. Coupling the home garden source and groundwater modeling captured variability in conditions within the SFS economic feasibility areas when predicting SFS constituent fate and transport in the environment. The conservative nature of the assessment was maintained through the placement of the drinking water receptor well 1 m from the edge of the garden in the centerline of the plume.
- Because arsenic has the potential to exhibit nonlinear behavior during transport through the unsaturated zone as simulated by EPACMTP, it was necessary to ensure the appropriateness of applying the unitized approach to the groundwater pathway. As a result, an analysis was performed which demonstrated that arsenic would behave linearly

in the subsurface under anticipated home garden environmental conditions and at concentrations found in SFS samples. **(Appendix J** and Chapter 5**)**.

- An analysis was performed to evaluate anticipated arrival times of peak contaminant concentrations in the receptor drinking water well. Based on the analysis, it is unlikely that peak surface and peak groundwater exposures will occur within the same timeframe. For example, the earliest estimated timeframe for arrival of arsenic in the well spanned from 29 to 200 years following the application of the SFS-manufactured soil. Given this timeframe, it is likely that the peak well concentrations will not occur until well past the receptor's timeframe of residency (i.e., exposure duration). Therefore, surface and subsurface ingestion exposures would not occur together during the same exposure period. (**Appendix J** and Chapter 5).
- The probabilistic modeling for the home gardener scenario demonstrated that, even using consumption rates at the upper end of the distribution, the estimated exposures were below health benchmarks.

7.6 Characterization of Risks Associated With SFS Beneficial Use (Chapter 6)

- The assumption of a 1:1 mix for manufactured soil in the home gardener scenario was a conservative assumption, because this would be cost prohibitive for even small home gardens. A more likely scenario would be a manufactured soil consisting of 5–10% SFS, rather than the 50% SFS modeled here. Therefore, this assumption likely overestimates soil concentrations.
- Evaluating the national-scale beneficial use of SFS in road subbase, soil-less potting media, and manufactured soil includes numerous sources of variability. However, the findings from the available multiple lines of inquiry—such as newly available analytical results for SFS, research on metals behavior in soil (including SFS-specific studies), and risk screening methods (including modeling), all within the context of well-established soil science—when used collectively provide a sound scientific basis for determining appropriate soil-related uses of SFS.
- Given the assumption of high-end concentrations of the metals and other constituents in SFS, and the application of highly conservative screening techniques, risk screening models and refined models, the preponderance of the evidence demonstrates that the evaluated uses of silica-based SFS produced by iron, steel, and aluminum foundries are unlikely to cause adverse effects to human health and ecological receptors.

Table 7-1 provides a useful data summary for regulatory decision makers and other stakeholders; the table presents the analytical and background information on metal constituents in SFS, as well as the HH-SSLs and Eco-SSLs. In addition, the table provides the SFS-specific modeled screening values for the specific home gardener scenario evaluated in this report, as well as modeled screening values based on median and high-end consumption by the general public.[55] As shown in this table, the concentrations of metal constituents found in SFS are below the health-based and ecological screening levels for soil and are present at levels that are similar to those found in native soils.

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[55] **Chapter 5** discusses the rationale for deriving screening levels based on three different consumption rates.

Table 7-1. Comparing SFS Concentrations to Various Screening Values (mg kg^{-1} unless otherwise noted)

	Silica-based Iron, Steel, and Aluminum Sands[a]				Human Screening Values				Eco Screening Values			U.S. and Canadian Surface Soils[b]		
						Modeled Consumption Rates[c]								
Elements	**Max**	**95%-ile**	**Median**	**Manuf. Soil**	**SSL[d]**	**Home Gardener**	**Gen. Pop. Median**	**Gen. Pop. High**	**Eco-SSLs[e]**	**Modeled (SFS-Specific)**	**USDA[f]**	**Max**	**95%-ile**	**Median**
Al (g kg^{-1})	11.7	11.2	5.56	5.6	77	--	--	--	ND	--	--	87.3	74.6	47.4
As	7.79	6.44	1.05	3.22	6.7[g]	8.0	30	9.1	18	40	--	18.0	12.0	5.0
B	59.4	20.2	10.0	10.1	16,000	--	--	--	ND	--	--	ND	ND	ND
Ba	141	17.7	5.00	8.85	15,000	--	--	--	330	--	--	1800	840	526
Be	0.60	0.38	0.15	0.19	160	--	--	--	21	--	--	4.0	2.3	1.3
Cd	0.36	0.20	0.05	0.10	70	--	--	--	0.36	--	--	5.2	0.6	0.2
Co	6.62	5.99	0.88	3.00	23	22	58	21	13	--	--	143.4	17.6	7.1
Cr (III)	115	109	4.93	54.5	120,000	--	--	--	34	510	--	5320	70.0	27.0
Cu	137	107	6.22	53.5	3,100	--	--	--	49	159	200	81.9	30.1	12.7
Fe (g kg^{-1})	64.4	57.1	4.26	28.9	55	160	230	150	ND	--	--	87.7	42.6	19.2
Mn	707	670	54.5	335	1,800	--	--	--	220	1000	--	3,120	1,630	490
Mo	22.9	21.8	0.50	10.9	390	--	--	--	ND	--	--	21.0	2.16	0.82
Ni	117	102	3.46	51.0	1,500	--	--	--	38	290	200	2,314	37.5	13.8
Pb	22.9	15.3	3.74	7.65	400	--	--	--	56	--	--	244.6	38.0	19.2
Sb	1.71	1.23	0.17	0.62	31	--	--	--	0.27	4.1	--	2.3	1.39	0.60
Se	0.44	0.20	0.20	0.10	390	--	--	--	0.52	--	--	2.3	1.0	0.3
Tl	0.10	0.09	0.04	0.05	0.78	--	--	--	ND	--	--	1.8	0.7	0.5
V	11.3	9.90	2.88	4.95	390	--	--	--	280	--	--	380	119	55
Zn	245	72.1	5.00	36.1	23,000	--	--	--	79	--	300	377	103	56

-- = No modeled value was generated because constituent was screened out of further study in an earlier stage of the evaluation. If a constituent screened out based on human health SSL and had no Eco-SSL, the constituent was considered to have screened out for both human and eco.

ND = No Data.

[a] Source: Dayton et al. (2010).

[b] Source: Smith et al. (2005).

[c] See **Chapter 5** for a detailed discussion of how the modeled values were generated.

[d] Concentrations of SFS constituents in manufactured soil (a 1:1 blend) were compared to an order-of-magnitude below the SSLs listed here, as discussed in **Chapter 4, Section 4.4.3**. Values are from EPA Regional Screening Tables (http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/index.htm). Unless otherwise noted, all values are based on noncarcinogenic impacts.

[e] Concentrations of SFS constituents in manufactured soil (a 1:1 blend) were compared to the Eco-SSLs, as discussed in **Chapter 4, Section 4.4.3**.

[f] See **Appendix C** for an explanation of USDA Phytotoxicity Screening Values for copper, nickel, and zinc.

[g] Based on carcinogenic risk, set at the standard EPA Office of Resource Conservation and Recovery risk target level of 1E-05.

8. Agency Policy on the Beneficial Use of Silica-Based Spent Foundry Sands from Iron, Steel and Aluminum Foundries

The beneficial use of SFS, when conducted in an environmentally sound manner, can contribute positive environmental and economic benefits. Environmental benefits can include energy savings, reduced greenhouse gas emissions, and water savings. Economic benefits can include job creation in the beneficial use industry, reduced costs associated with SFS disposal, increased revenue from the sale of SFS, and savings from using SFS in place of more costly materials.

Spent foundry sand has been used as a substitute for virgin sand in certain markets. In this risk assessment, the EPA and USDA have focused on a number of these markets. Approximately 10 million tons of SFS are produced annually, with only 26% of these SFSs being beneficially used beyond the foundry. **Table 8-1** shows the beneficial uses (EPA, 2008c) of SFS that were evaluated in this risk assessment. When beneficially using SFS it is particularly important to check with your State Agency, which may have specific requirements pertaining to such activities.

Table 8-1. Quantity SFS Beneficially used, by Market (tons)

Beneficial Use Market	Quantity Beneficially Used
Road construction (excluding asphalt)	144,288
Top soil mix/horticulture	220,949

SOURCE: U.S. EPA (2008c), Table ES-1

An EPA analysis (EPA, 2008c) provides estimates of the environmental benefits that can be achieved with the beneficial applications that were studied in this risk assessment. The analysis calculated environmental benefits per 1,000 cubic yards of SFSs and then extrapolated these benefits to the total amount of SFSs used in a specific application.

Table 8-2. Primary Environmental Benefits of Beneficial use of SFS, by Market

Avoided Impacts	Road Base Use Extrapolated to 144,288 tons of SFS	Manufactured Soil Use Extrapolated to 220,949 tons of SFS
Energy Consumption (megajoules)	17,800,000	27,900,000
Water consumption (1,000 gallons)	3,000	4,800
CO_2 Emissions (metric tons)	1,500	2,500

SOURCE: U.S. EPA (2008c), Table ES-3

This risk assessment concluded that the beneficial uses of silica-based SFSs from iron, steel and aluminum foundry operations when used in manufactured soils, soil-less potting media and roadway subbase, are protective of human health and the environment. Based on this conclusion, and the available environmental and economic benefits, the EPA and USDA support the beneficial use of silica-based SFSs specifically from iron, steel and aluminum foundry operations when used in manufactured soils, soil-less potting media and roadway subbase. The EPA and USDA believe that these beneficial uses provide significant opportunities to advance Sustainable Materials Management (SMM) (http://www.epa.gov/smm).

Any conclusions drawn by this risk assessment should be understood within the limitations and scope of the evaluation, including the following:

- Only silica-based SFS from iron, steel and aluminum foundries are evaluated. In contrast, SFS from leaded brass and bronze foundries often qualify as RCRA hazardous waste. Also, there weren't sufficient data to characterize SFS from non-leaded brass foundries and SFS containing olivine sand, and therefore these SFSs are not evaluated in this risk assessment.
- In addition to SFS, foundries can generate numerous other wastes (e.g., unused and broken cores, core room sweepings, cupola slag, scrubber sludge, baghouse dust, shotblast fines). This assessment, however, applies only to SFS as defined in the assessment: molding and core sands that have been subjected to the metalcasting process to such an extent that they can no longer be used to manufacture molds and cores. To the extent that other foundry wastes are mixed with SFS, the conclusions drawn by this assessment may not be applicable.
- Samples from 39 foundries (totals and pore water data from 39 samples, and leachate data from 108 samples) were used to represent silica-based SFS from all iron, steel, and aluminum foundries in the U.S. Because the foundries were not chosen randomly, there is uncertainty regarding whether the data are statistically representative of SFS from all iron, steel, and aluminum foundries. However, these foundries were specifically selected to ensure that the full range of constituents and their concentrations were adequately represented, and the analytical data from these samples are the best available for characterizing SFS constituents.
- Analytical data were available for 25 metals, 16 PAHs, 17 phenolics, and 20 dioxins and dioxin-like compounds. USDA analyzed for organic compounds that are major binder components (i.e., phenolics) or might be generated during thermal degradation of chemical binders and other organic additives (i.e., PAHs, dioxins, furans), because these constituents present the greatest hazard if at elevated levels in the environment. Review of the scientific literature for evidence of additional organic compounds present in SFS indicated that they were well below levels of concern.
- Screening and modeling evaluated those constituents for which toxicity benchmarks exist.
- Evaluated beneficial uses include manufactured soil, soil-less growth media and road subbase. The home garden using SFS-manufactured soil was modeled because it demonstrated the greatest potential for exposure.

- The home garden scenario evaluated a single eight-inch deep application of SFS-manufactured soil (comprised of 50% SFS) to a 0.1-acre garden.

  Additional information can be found at the following web-sites:

- EPA's (http://www.epa.gov/solidwaste/conserve/imr/foundry/index.htm),
- American Foundry Society (AFS) (http://www.afsinc.org),
- Industry Recycling Starts Today (FIRST) (http://www.afsinc.org/government/AFSFirst.cfm?ItemNumber=7887&&navItemNumber=528)
- Federal Highway Administration (FHWA) (https://www.fhwa.dot.gov/publications/research/infrastructure/structures/97148/fs1.cfm)

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9. References

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Appendix A

Fundamental Concepts Regarding Trace Elements in Byproduct-Treated Soils

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Appendix A

Fundamental Concepts Regarding Trace Elements in Byproduct-Treated Soils

This section of the report was included to help readers better understand the potential for trace elements in SFS to react in soils and interact in the environment. For decades, researchers have been working to better understand the potential for soil elements to support growth of plants and livestock or to become deficient for or phytotoxic for plants or soil organisms. Much of soil toxicology is based on studies that add soluble metal salts to soils that are cropped immediately to examine adverse effects. But the added metals quickly react with the adsorbent surfaces or precipitate in the soil, greatly reducing phytoavailability. Thus realistic assessment of risk from chronic exposure to trace elements in soils benefits from a deeper understanding of the metal species found in soils and their longer-term behavior.

A.1 Chemical Reactions in Soils

Soils contain all elements at concentrations dependent on the parent rocks from which the soil is derived. Elements may also reach soils as components of fertilizers, manures, byproducts, aerosols, etc., and hence may exist in varied chemical forms. If elements reach soils in elemental forms, they corrode/oxidize depending on the redox characteristics of the element and the soil. For example, Ag and Cu are found in metallic form in some reducing soils, but usually oxidize in aerobic soils over time. Some elements (e.g., metallic Pb, Zn, and Ni) oxidize slowly, while others oxidize more rapidly. A few persist for long periods depending on the particle size of the element that reached the soil (smaller particles have higher surface area and react more rapidly), or redox status of the soil. Flooded peat soils may provide a reducing soil environment that will allow metallic or metal sulfide particles to persist for long periods.

Another aspect of reactions of trace elements in residuals with soils is the unusually low reactivity of some metal oxides such as NiO. This compound was emitted by some Ni refineries and found to persist for decades in aerobic soils (McNear et al., 2007). Studies showed that the dissolution of NiO is inherently slow, with a half-life of 6.5 years at a pH of 6 (Ludwig and Casey, 1996).

For a material such as SFS, the trace elements are present as (1) oxidized equilibrium forms in the input sand and (2) some metallic particles and oxidized forms of the elements used in producing castings at a foundry. Iron and steel may remain partly in the metallic forms for some time, but will eventually oxidize and enter soil equilibria.

For the remaining discussion, we will assume that elements in a residual are the ionized forms in equilibrium with aerobic soils rather than the elemental state which could enter soils from some sources. The ionized forms are more mobile, and thus potentially more toxic than the elemental forms, so risk assessment for the ionized forms is appropriate. In this case, the element will have reacted with redox buffering parts of the soil and with adsorptive or chelation surfaces of the soil. In a normal aerobic soil, most elements are present as hydrated or complexed cations or anions in equilibrium, either bound to the soil surfaces or precipitated as minerals (Langmuir et al., 2004) (e.g., Zn^{2+}, Cu^{2+}, Ni^{2+}, Pb^{2+}, Cd^{2+}, MoO_4^{2-}, SeO_4^{2-}, NO_3^-, SO_4^{2-}, $H_2PO_4^-$). Many ions are so easily oxidized that they remain the cation regardless of soil redox conditions: Li^+, K^+, Na^+, Rb^+, Cs^+ (alkali cations), Be^{2+}, Mg^{2+}, Ca^{2+}, Sr^{2+}, Ba^{2+} (alkaline earth cations). Similarly, reducing soils are not reducing enough to alter the form of halogen ions in soils: F^-, Cl^-, Br^-, I^-, although it is possible to reduce iodide to iodine in strongly reducing soils. Most elements in soils are not transformed to organic compounds with covalent linkage, but those that are transformed have very changed properties.

Mercury is transformed by soil microbes and some plants into Hg^0, HgS, organic matter bound Hg, CH_3-Hg, or even $(CH_3)_2Hg$. The Hg^0 is volatile and can be emitted from the soil; most Hg^0 emission from soils is induced by light striking the surface soil. The methyl-Hg forms are volatile and lipophilic, and can bioaccumulate in organisms. But the fraction of soil Hg in the methylated forms is quite small.

Flooding a soil causes the redox potential to rise as the soil becomes reducing because little O_2 dissolves in water and soil organisms consume the O_2. The soil pores become filled with water or gases formed in the soil under anaerobic conditions. With the reducing environment, some elements are reduced to chemical forms different than those found in normal aerobic soils. In particular, arsenate (AsO_4^{3-}) is reduced to the more soluble and more phytotoxic arsenite (AsO_3^{3-}). This is important because flooded rice is the crop plant found to be most sensitive to excessive soil As; the higher concentration of AsO_3^{3-} in flooded soils compared to AsO_4^{3-} in aerobic soil allows much easier plant uptake and injury from soil As. Uptake of some other elements may also be increased in reducing soils, but without an increased phytotoxicity as demonstrated for As.

Soil Mn is the cation most altered by soil reduction. Mn is usually present in aerobic soils as MnO_2 and not available to plant roots except where roots reduce the MnO_2 to Mn^{2+}. In flooded soils, Mn^{2+} can be greatly increased; Mn^{2+} is not strongly adsorbed by soils and can accumulate to high levels and become phytotoxic to sensitive plant species. Draining the soil allows rapid oxidation of the Mn^{2+} to MnO_2 if the soil pH is higher than 5.5 (the oxidation is catalyzed by soil microbes).

A.1.1 Reactions Over Time

Time is an important variable when assessing soil chemistry and risk from trace element exposure. Most microelements react more strongly with soil over time (Logan and Chaney, 1983; Basta et al., 2005). This is shown by how the plant availability or extractability of an element changes with time after a soluble salt of the element is added to soils. There are several kinds of reactions: hydrolysis (or precipitation), chelation by organic matter, chemisorption on Fe and Mn oxide surfaces, and formation of new solid phases. These reactions are nicely illustrated by the reactions of Ni with mineral and organic soils. When soluble Ni compounds are mixed with soils, the Ni-hydrated cations rapidly form adsorbed forms on Fe and Mn oxides and chelated forms with soil organic matter (SOM). Other soil minerals then dissolve, and Ni reacts to form new soil minerals such as Ni silicates and Ni-Al layered double hydroxides (LDHs).[1] The overall process is illustrated by Singh and Jeng (1993), who tested Ni reactions with soil over time when they grew ryegrass in a greenhouse annually for 3 years in large pots using Ni-salt applications to an acidic sandy soil. Phytotoxicity was not observed at the highest soil nickel application (50 mg Ni kg^{-1}), even though shoot nickel reached nearly 50 mg kg^{-1} dry shoots in

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[1] Ni-Al LDHs were discovered only recently when extended X-ray absorption fine structure spectroscopy (EXAFS) was applied to the reactions of elements with soils (Sparks, 2003).

the first year. In subsequent years, oat shoot nickel declined from 48 mg kg^{-1} in Year 1 to 18 mg kg^{-1} in Year 2 to 8 mg kg^{-1} in Year 3. Such a decline is expected because the freshly added Ni requires time to equilibrate with soil adsorption sites and form more stable Ni silicate and LDH compounds in soil.

Using physical-chemical methods not available earlier (e.g., EXAFS), research conducted during the past decade has provided additional information on how water-soluble Ni^{2+} reacts with soils and with specific minerals. During such reaction/speciation tests, the ionic strength of the soil solution influences the strength of adsorption; high ionic strength inhibits adsorption and could thus promote the formation of inorganic compounds in soil. In one set of experiments, Scheidegger and colleagues (1996a, 1996b, 1997, and 1998), Scheidegger and Sparks (1996), Ford et al. (1999), and Sheinost et al. (1999) added Ni^{2+} to clays and minerals and used Synchrotron radiation after varied amounts of time to examine the formation of LDHs (e.g., nickel aluminum hydroxide) and Ni silicates. The higher ionic strength of these tests (0.1 M KNO_3) led to the formation of LDHs if the clays and minerals released Al, and to the formation of Ni-silicate crystalline materials if the clays and minerals released silicate ions. However, when Elzinga and Sparks (2001) used a lower ionic strength, the relative proportion of adsorption (or chemisorption; specific adsorption) increased, and the formation of surface-induced precipitates decreased.

This work demonstrated important aspects of the reactions of Ni with soils in that slow reactions over time converted added Ni^{2+} to forms of Ni that were much less soluble or phytoavailable. This is further illustrated by Scheckel and Sparks (2001), who examined mineral samples that had been reacting with Ni^{2+} for 1 hour to 2 years. The longer the reaction period, the lower the water solubility or acid extractability of the adsorbed or precipitated insoluble Ni species. For example, after Ni^{2+} equilibrated with several minerals, extractability was as high as 98% for the 1-hour equilibrated materials and as low as 0 for the 2-year equilibrated materials. The increase in stability of the Ni surface precipitates with increasing residence time in their studies was attributed to three aging mechanisms: (1) Al-for-Ni substitution in the octahedral sheets of the brucite-like hydroxide layers, (2) Si-for-nitrate exchange in the inter-layers of the precipitates, and (3) Ostwald ripening of the precipitate phases. We believe these findings are complementary with the work of Bruemmer and colleagues (1988), who found adsorption to strengthen with time of reaction, following a diffusion-type process. The comparatively insoluble chemical forms of Ni formed during the prolonged reactions of Ni^{2+} with soil were simply more ordered Ni silicates and LDHs, not Ni^{2+} adsorbed within nanopores in the surfaces of Fe and Mn oxides.

Ni, Co, and Zn have also been found to form LDH compounds over time after addition to soils or contamination in the field (Ford and Sparks, 2000; Voegelin et al., 2002; Voegelin and Kretzschmar, 2005). At low soil pH, Zn is much less likely to generate LDH forms than Ni, but at neutral pH, the Zn-LDH formed and must contribute to the ability of limestone to remediate Zn-contaminated soils. Voegelin and Kretzschmar (2005) tested formation of mixed LDH with both Zn and Ni and found that the mixed LDH were not as stable (to pH 3 extraction) as Ni-LDH without the presence of high levels of Zn. In any regard, the formation of LDH metal compounds in soils helps explain the very strong difference in response of plants to added soluble metal salts compared to pre-equilibrated metals from different sources. This “metal reacts more strongly with time” response was evident in a study even 30 years ago on the availability of fertilizer Zn added to soils. Based on this study, added Zn becomes less plant available over time and refertilization may be required for soils with the highest ability to inactivate added soluble Zn fertilizers such as $ZnSO_4$ (Boawn, 1974 and 1976).

A.1.2 Sorption in Aerobic Soils

Sorption is a chemical process that buffers the partitioning of trace elements between solid and liquid phases in soils and byproducts. Fe, Al, and Mn oxide soil minerals are important sinks for trace elements in soil and byproduct-amended soil (Essington and Mattigod, 1991; Lombi et al., 2002; Hettiarachchi et al., 2003). Trace element sorption by the oxide surface is a pH-dependent process; protons compete with cations for sorption. The adsorption of metal cations by the oxide surfaces increases to nearly 100% with increasing pH (McKenzie, 1980). In contrast, oxyanion adsorption generally decreases with increasing pH. Differences between adsorption and desorption isotherms typically reveal significant hysteresis (Hettiarachchi et al., 2003), providing evidence that this process is not simply a competitive ion-exchange reaction between metal ions and protons or hydroxyls. Some adsorbed metals are strongly bonded and not readily desorbed from these oxide surfaces. Some research suggests that the increasingly strong sorption and lower phytoavailability results from the trace elements moving to nano-sized pores in Fe and Mn oxides (Bruemmer et al., 1988).

Trace element sorption by oxides shows Fe and Mn oxides have a much greater adsorption capacity compared to Al oxides and clay minerals (Brown and Parks, 2001). Molecular-scale X-ray spectroscopic studies show that the strong bonding of Cu, Co, Cr, Mn, Ni, Cd, Pb, and Zn to these oxide surfaces is due to formation of inner-sphere surface complexes and formation of metal hydroxide precipitate phases (Brown and Parks, 2001; Sparks, 2003). New solids found after trace element ion reactions with soil materials, including metal silicates and mixed double hydroxides with Al, can substantially reduce element solubility and availability (Scheckel and Sparks, 2001). Sorption by Fe and Mn metal oxides is a major mechanism for removal of trace element cations (i.e., Cd, Cr, Cu, Pb, Hg, Ni, Zn) and trace element oxyanions (i.e., AsO_4^{3-}, AsO_3^{3-}, SeO_4^{2-}, SeO_3^{2-}, MoO_4^{2-}, WO_4^{2-}, VO_4^{2-}, CrO_4^{2-}) from aqueous solution (e.g., soil solution) (Stumm, 1992; Sparks, 2003).

Trace element cations also sorb to SOM and other forms of humified natural organic matter (NOM). Strong adsorption to NOM in byproducts by formation of metal chelates reduces the solubility of several trace elements in soil (Adriano, 2001). Sorption of trace elements to SOM or NOM increases with pH because protons compete less well at increasing pH. At lower pH, trace element sorption by NOM is reduced less than is trace element sorption to Fe and Mn oxides.

Trace element cations form sparingly soluble precipitates with phosphate, sulfides, and other anions (Lindsay, 2001; Langmuir et al., 2004). Trace element precipitation is highly pH dependent and increases with pH for many trace element cations. AsO_4^{3-} and other trace element oxyanions can form insoluble precipitates with multivalent cations, including Fe, Al, and Ca. The resulting trace element minerals (i.e., precipitates) may control the amount of trace element in solution (i.e., availability and mobility).

Byproducts typically contain components (NOM; Fe, Mn, and Al oxides; and anions such as phosphate and silicate) that can adsorb or precipitate trace elements. Many types of byproducts (e.g., biosolids, manure, municipal solid waste compost, coal combustion residuals) with a wide range of properties have been applied to agricultural land and have modified the adsorption properties of soils (Power and Dick, 2000; Basta et al., 2005). Trace element phytoavailability is affected by the sorption capacity and properties (e.g., pH, salinity) of both the soil and the byproduct. Sandy soils with low Fe and Mn oxide content and low SOM have low sorption capacities and will have greater trace element phyto- and bioavailabilities than loamy or clayey soils with greater amounts of sorbents (i.e., reactive oxides, SOM), provided the soils have similar pH values. Similarly, byproducts with low Fe and Mn oxide content and low NOM have low trace element sorption capacities and higher potential element availabilities as compared to byproducts with high Fe and Mn oxide and NOM.

Byproduct-soil mixtures would have intermediate sorption properties between that of the soil and byproduct and, perhaps, intermediate phytoavailabilities if other properties (e.g., pH) were similar. As the loading rate of the byproduct increases, the byproduct-soil mixture will be increasingly affected by the sorption properties of the byproduct. Some byproducts have greater amounts of these sorbents than soil and can increase the sorption capacity of soils for trace elements. Added to soil in sufficient amounts, a high-sorbent byproduct can dominate the trace element binding chemistry of the soil-byproduct mixture (Basta et al., 2005; Kukier et al., 2010).

This phenomenon is illustrated in Figure A-1, which shows the results of a Cd phytoavailability bioassay using Romaine lettuce grown on a Christiana fine sandy loam soil with no amendment (control), with 224 t ha^{-1} of a digested biosolid, or with 672 t ha^{-1} of a biosolids compost applied over 25 years before the test was conducted. During this test, all soils were adjusted to a pH of 6.5, and five rates of soluble Cd were applied. Lettuce uptake of Cd was linear, with increasing added soluble Cd, but the slope of this uptake was reduced up to 90% by the historic amendment with high-Fe biosolids or biosolids compost. These two amendments were rich in Fe and phosphate, and it is believed that an Fe-P-NOM complex provides the persistent high Cd binding. It seems likely that inclusion of Fe oxide in organic P-rich byproducts can readily reduce trace element cation phytoavailability (Hettiarchchi et al., 2003; Basta et al., 2005; Kukier et al., 2010).

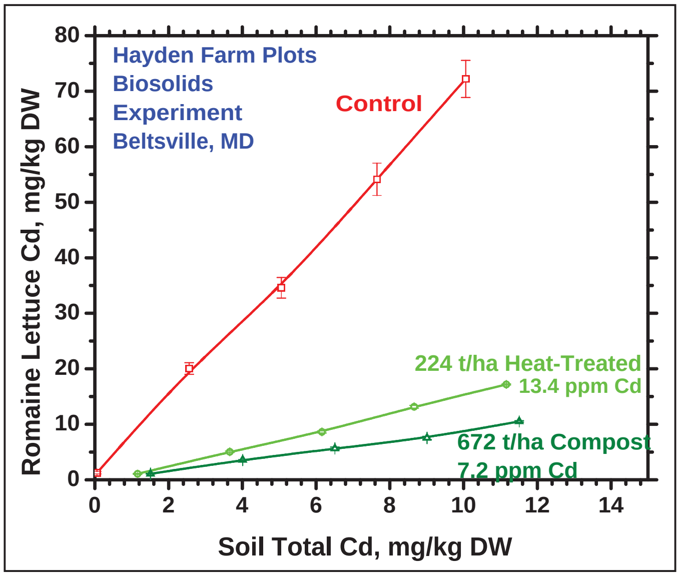

Figure A-1. The effect of historic biosolids applications on the phytoavailability of applied Cd salt to Romaine lettuce.

A.2 Soil-Plant Barrier Limits Risks from Trace Elements in Soils or Soil Amendments

The potential risk that diverse trace elements in soils pose to the feed- and food-chain has been intensively examined during the past 35 years. One purpose of the investigation has been to understand the risk from application of biosolids, livestock manure, and other trace element contamination sources to soil.

During this period, the “Soil-Plant Barrier” concept was introduced to communicate how element addition rate and chemistry, soil chemistry, and plant chemistry affected the risk to plants and animals from elements in soil amendments (Chaney, 1980 and 1983). This concept is based on long experience in veterinary toxicology and agronomy. Reactions and processes related to the Soil-Plant Barrier include the following:

1. Solid adsorbent sources (e.g., Fe, Al, and Mn oxyhydroxides and organic matter) in soil amendments may have adsorptive surfaces that influence soil chemistry.
2. Adsorption or precipitation of elements in soils or in roots limits uptake-translocation of most elements to shoots.
3. The phytotoxicity of Zn, Cu, Ni, Mn, As, B, Al, F, and other elements limits concentrations of these elements in plant shoots to levels chronically tolerated by livestock and humans.
4. Food-chain transfer of an element may not constitute a risk, but the direct ingestion of highly contaminated soil may cause risk from Pb, As, F, and some other elements if the soil is poorly managed.
5. The Soil-Plant Barrier does not restrict transfers of soil Se, Mo, and Co well enough to protect all animals from elements (e.g., Se, Mo) or ruminant livestock (e.g., Co). In addition, the soil-plant barrier does not restrict transfer of Cd in rice and, as a result, subsistence rice consumers may be at risk in situations of moderate Cd contamination because of the physiology of paddy rice and for garden crops where Cd contamination occurs without the usual 100-fold greater Zn contamination.

A summary of trace element tolerances by plants and livestock is presented in **Table A-1**. Please note that the National Research Council (NRC; 1980) committee that identified the maximum levels of trace elements in feeds tolerated by domestic livestock based its conclusions on data from toxicological-type feeding studies in which soluble trace element salts had been mixed with practical or purified diets to examine animal response to the dietary elements. If soil or some soil amendment is incorporated into diets, element solubility and bioavailability very likely are much lower than in the tests relied on by NRC (1980). For example, Chaney and Ryan (1993) noted that animal body Pb burden from ingesting the soil does not increase until the soil Pb concentration exceeds approximately 300 mg Pb kg^{-1}. Other elements, in equilibrium with poorly soluble minerals or strongly adsorbed in ingested soils, are often much less bioavailable than they would be if they were added to the diet as soluble salts.

Table A-1. Maximum Tolerable Levels of Dietary Minerals for Domestic Livestock in Comparison With Levels in Forages

Element	"Soil-Plant Barrier"	Level in Plant Foliage[a] (mg kg^{-1} Dry Foliage)		Maximum Levels Chronically Tolerated[b] (mg kg^{-1} Dry Diet)			
		Normal	**Phytotoxic**	**Cattle**	**Sheep**	**Swine**	**Chicken**
As (inorganic)	Yes	0.01 – 1.0	3 – 10	50	50	50	50
B	Yes	7 – 7.5	75	150	(150)	(150)	(150)
Cd[c]	Fails	0.1 – 1	5 – 700	0.5	0.5	0.5	0.5
Cr^{3+}	Yes	0.1 – 1	20	(3,000)	(3,000)	(3,000)	3,000
Co	Fails	0.01 – 0.3	25 – 100	10	10	10	10
Cu	Likely	3 – 20	25 – 40	100	25	250	300
F	Yes	1 – 5	-	40	60	150	200
Fe	Yes	30 – 300	-	1,000	500	3,000	1,000
Mn	Likely	15 – 150	400 – 2,000	1,000	1,000	400	2,000
Mo	Fails	0.1 – 3	100	10	10	20	100
Ni	Likely	0.1 – 5	50 – 100	50	(50)	(100)	(300)
Pb[c]	Yes	2 – 5	-	30	30	30	30
Se	Fails	0.1 – 2	100	(2)	(2)	2	2
V	Yes	0.1 – 1	10	50	50	(10)	10
Zn	Likely	15 - 150	500 – 1,500	500	300	1,000	1,000

[a] Based on literature summarized in Chaney (1983).

[b] Based on NRC, 1980. Continuous long-term feeding of minerals above the maximum tolerable levels may cause adverse effects. NRC estimated the levels in parentheses by extrapolating between animal species when data were not available for an animal.

[c] NRC based the maximum levels chronically tolerated of Cd or Pb in liver, kidney, and bone in foods for humans rather than simple tolerance by the animals. Because of the simultaneous presence of Zn, Cd in animal tissues is less bioavailable than Cd salts added to diets and the maximum levels chronically tolerated should have been higher than listed.

The chemistry of elements in soils is affected by the presence of ions, which can cause precipitation of the element, organic matter, and sesquioxides, which, in turn, can adsorb elements; redox changes, which affect the chemical species of the elements present; and similar factors. Soils are usually in a relatively restricted pH range of 5.5 to 8 for high-producing soils and as wide ranging as 5 to 9 in nearly all soils in the general environment. Some soil amendments have a pH greater than 8, but soils thus amended absorb atmospheric CO_2, which returns the soil pH to no higher than calcareous soil levels.

Many elements (e.g., Ti, Fe^{3+}, Pb, Hg, Al, Cr^{3+}, Ag, Au, Sn, Zr, and rare earth elements [e.g., Ce] that serve as a label for soil contamination of plants and diets) are so insoluble in aerobic soils between a pH of 5.5 and 8 that they do not cause risk to animals even when soils with relatively high concentrations are ingested by livestock. This is especially well illustrated by Cr uptake by plants growing on high Cr-mineralized serpentine soils (Cary and Kubota, 1990); soil contained more than 10,000 mg Cr kg^{-1}, but all Cr measured in plant samples could be explained by soil particle contamination of the plant sample (based on Ti and other element concentrations). Cr was actually added to diets as a non-absorbed index cation to follow absorption of other nutrients along the gastrointestinal tract or the timing of movement (Irwin and Crampton, 1951; Raleigh et al., 1980). Direct soil ingestion could provide exposure and must be considered separately from plant uptake.

When present at very high concentrations in soil, some elements that are not accumulated by plants to levels of concern for livestock or wildlife (e.g., F, Pb, As, and Zn) may still pose a risk because of absorption from ingested soil. These same elements may comprise risk to earthworm-consuming wildlife (e.g., shrews, moles, badgers, woodcocks) from soil ingestion, but not plant uptake. The earthworm is consumed without depuration of internal soil, exposing the predator to high levels of dietary soil—perhaps 35% of dry weight (Beyer and Stafford, 1993). This high soil ingestion rate makes earthworm-consuming wildlife among the most highly soil-exposed animals, and is an important consideration in risk assessment of soil contaminants such as Cd, which can also bioaccumulate in earthworm tissues.

Freshly applied trace element salts are not in equilibrium with soil and have a greater bioavailability than they would exhibit upon equilibrating with the soil over time. The phytoavailability and bioavailability of trace elements may also be reduced if the metals are adsorbed, chelated, or precipitated before ingestion by children or grazing livestock.

A.3 Toxicity or Prevention of Toxicity by Interaction Among Trace Elements

The toxicity to animals of biosolids- or manure-applied Cu or Zn is an example of the interaction between elements impacting element toxicity. Cu deficiency–stressed animals are more sensitive to dietary Zn than animals fed Cu-adequate diets, but biosolids-fertilized crops are not low in Cu, so ordinarily Zn phytotoxicity protects all livestock against excessive Zn in forages, including the most sensitive ruminants. Similarly, Cu toxicity to sensitive ruminant animals is substantially reduced by increased dietary levels of Zn, Cd, Fe, Mo, and SO_4^{2-} or sorbents such as SOM. In contrast with the predicted increase in liver Cu concentrations and toxicity from Cu in ingested swine manure or biosolids, reduced liver Cu concentrations have been found in cattle or sheep unless the ingested biosolids exceeded approximately 1,000 mg Cu kg^{-1}.

Interactions can also limit toxicity and risk. For example, Cd bioavailability is strongly affected by the presence of the normal geogenic levels of Zn (100- to 200-fold Cd level); Zn inhibits binding of Cd by soil, but also inhibits Cd uptake by roots, Cd transport to shoots and Cd transport to storage tissues. Furthermore, Zn in foods significantly reduces Cd absorption by animals (Chaney et al., 2004). Increased Zn in spinach and lettuce reduced absorption of Cd in these leafy vegetables by Japanese quail (McKenna et al., 1992), and increased Zn in forage diets strongly inhibited Cd absorption and reduced liver and kidney Cd concentration in cattle (Stuczynski et al., 2007).

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Appendix B

Spent Foundry Sand Characterization Data

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Table B-1. Element-Specific Concentrations in Spent Foundry Molding Sands Collected June 2005 (Concentrations in mg kg^{-1})

FIN[a]	Ag	Al	As	B	Ba	Be	Cd	Co	Cr	Cu	Fe	Mg	Mn	Mo	Ni	Pb	Sb	V	Zn
1	*<17.6*	4,379	1.4	*<19.2*	95.0	*<1.2*	*<5.9*	*<0.84*	9.4	97.6	8,914	1,535	145	*<4.4*	6.5	12.7	*<4.5*	*<7.4*	54.8
2	*<17.6*	2,137	0.11	*<19.2*	*<8.7*	3.1	*<5.9*	95.3	57.3	27.0	44,320	51,574	671	*<4.4*	2,328	*<7.7*	*<4.5*	*<7.4*	*<33.4*
3	*<17.6*	1,732	0.46	*<19.2*	28.1	*<1.2*	*<5.9*	*<0.84*	2.1	*<23.1*	1,535	1892	65.0	*<4.4*	2.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
4	*<17.6*	1,983	0.23	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	4.3	*<23.1*	2,020	*<720*	*<45.0*	*<4.4*	3.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
5	*<17.6*	1,896	0.64	*<19.2*	12.2	*<1.2*	*<5.9*	*<0.84*	5.7	*<23.1*	6,354	*<720*	137	*<4.4*	5.3	*<7.7*	*<4.5*	*<7.4*	*<33.4*
6	*<17.6*	996	0.16	*<19.2*	126.2	1.4	*<5.9*	41.4	149	*<23.1*	20,410	46,366	509	*<4.4*	1,022	*<7.7*	*<4.5*	*<7.4*	*<33.4*
7	*<17.6*	*<311*	0.04	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	4.3	*<23.1*	549	*<720*	*<45.0*	*<4.4*	2.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
8	*<17.6*	805	0.77	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	7.1	*<23.1*	5,709	*<720*	73.6	*<4.4*	3.5	*<7.7*	*<4.5*	*<7.4*	33.7
9	*<17.6*	522	0.38	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	1.4	*<23.1*	1,206	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
10	*<17.6*	532	0.83	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	20.2	34.24	7,630	*<720*	60.6	*<4.4*	6.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
11	*<17.6*	3,543	2.4	*<19.2*	21.6	*<1.2*	*<5.9*	0.84	3.3	*<23.1*	3,292	*<720*	69.7	*<4.4*	3.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
12	*<17.6*	2,153	0.36	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	2.3	*<23.1*	606	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
13	*<17.6*	1,961	1.1	*<19.2*	16.4	*<1.2*	*<5.9*	*<0.84*	1.5	*<23.1*	1,980	*<720*	*<45.0*	*<4.4*	2.8	*<7.7*	*<4.5*	*<7.4*	*<33.4*
14	*<17.6*	2,184	1.2	*<19.2*	11.2	*<1.2*	*<5.9*	*<0.84*	2.3	*<23.1*	2,750	1,389	*<45.0*	*<4.4*	2.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
15	*<17.6*	2,304	2.0	*<19.2*	15.5	*<1.2*	*<5.9*	*<0.84*	2.8	*<23.1*	2,643	810	56.7	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
16	*<17.6*	1,013	0.85	*<19.2*	10.0	*<1.2*	*<5.9*	*<0.84*	2.8	*<23.1*	2,206	*<720*	*<45.0*	*<4.4*	1.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
17	*<17.6*	3,042	2.0	*<19.2*	19.8	*<1.2*	*<5.9*	*<0.84*	3.1	*<23.1*	3,237	915	98.3	*<4.4*	3.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
18	*<17.6*	1,100	1.3	*<19.2*	12.3	*<1.2*	*<5.9*	*<0.84*	2.4	*<23.1*	2,029	*<720*	*<45.0*	*<4.4*	1.8	*<7.7*	*<4.5*	*<7.4*	*<33.4*
19	*<17.6*	1,998	1.5	*<19.2*	19.2	1.6	*<5.9*	1.20	38.0	37.7	20,210	*<720*	206	*<4.4*	13.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
20	*<17.6*	10,048	0.84	*<19.2*	14.7	*<1.2*	*<5.9*	*<0.84*	4.2	*<23.1*	2,575	1054	76.6	*<4.4*	5.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
21	*<17.6*	*<311*	0.57	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	5.9	*<23.1*	6,364	*<720*	*<45.0*	*<4.4*	2.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
22	*<17.6*	3,944	4.8	*<19.2*	24.5	2.2	*<5.9*	1.7	40.4	75.6	29,680	1,080	595	9.6	20.6	25.7	*<4.5*	9.1	179
23	*<17.6*	1,980	1.2	*<19.2*	19.7	*<1.2*	*<5.9*	*<0.84*	2.0	*<23.1*	1,987	*<720*	*<45.0*	*<4.4*	1.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
24	*<17.6*	1,247	1.2	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	6.1	*<23.1*	4,678	*<720*	45.3	*<4.4*	3.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
25	*<17.6*	2,164	0.72	*<19.2*	18.8	*<1.2*	*<5.9*	*<0.84*	2.5	*<23.1*	2,260	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
26	*<17.6*	1,906	1.5	*<19.2*	19.6	*<1.2*	*<5.9*	*<0.84*	2.3	*<23.1*	2,276	*<720*	*<45.0*	*<4.4*	2.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
27	*<17.6*	4,799	3.0	*<19.2*	25.4	1.92	*<5.9*	1.8	50.0	61.8	29,550	1,656	499	*<4.4*	29.8	8.5	*<4.5*	*<7.4*	*<33.4*
28	*<17.6*	1,849	0.95	*<19.2*	22.9	*<1.2*	*<5.9*	*<0.84*	4.2	*<23.1*	2,613	*<720*	*<45.0*	*<4.4*	2.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
29	*<17.6*	*<311*	0.38	*<19.2*	75.7	*<1.2*	*<5.9*	1.8	22.6	*<23.1*	3,394	*<720*	*<45.0*	4.7	36.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
30	*<17.6*	1,788	0.79	*<19.2*	27.8	*<1.2*	*<5.9*	*<0.84*	1.7	*<23.1*	2,420	*<720*	*<45.0*	*<4.4*	1.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
31	*<17.6*	321	0.41	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	8.7	*<23.1*	2,593	*<720*	*<45.0*	*<4.4*	12.3	*<7.7*	*<4.5*	*<7.4*	*<33.4*
32	*<17.6*	1,148	0.82	*<19.2*	13.6	*<1.2*	*<5.9*	*<0.84*	2.0	*<23.1*	2,744	*<720*	*<45.0*	*<4.4*	1.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
33	*<17.6*	*<311*	0.13	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	*<1.0*	70.0	<352	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	44.2
34	*<17.6*	1,325	0.74	*<19.2*	12.1	*<1.2*	*<5.9*	*<0.84*	2.1	3318	1,556	*<720*	*<45.0*	*<4.4*	8.9	19.0	*<4.5*	*<7.4*	1640
35	*<17.6*	1,852	1.2	*<19.2*	20.8	*<1.2*	*<5.9*	*<0.84*	*<1.0*	*<23.1*	3,333	*<720*	*<45.0*	*<4.4*	1.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
36	*<17.6*	2,406	1.9	*<19.2*	17.2	*<1.2*	*<5.9*	*<0.84*	2.1	*<23.1*	3,625	*<720*	*<45.0*	*<4.4*	2.3	*<7.7*	*<4.5*	*<7.4*	*<33.4*
37	*<17.6*	1,681	1.3	*<19.2*	14.4	*<1.2*	*<5.9*	*<0.84*	2.4	*<23.1*	3,021	*<720*	63.5	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
38	*<17.6*	*<311*	0.13	*<19.2*	27.4	*<1.2*	*<5.9*	*<0.84*	1.6	*<23.1*	740	*<720*	*<45.0*	*<4.4*	14.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
39	*<17.6*	1,595	0.52	*<19.2*	151	*<1.2*	*<5.9*	*<0.84*	8.3	60.3	4,004	797	117	*<4.4*	107	*<7.7*	*<4.5*	*<7.4*	*<33.4*
40	*<17.6*	1,630	0.87	*<19.2*	14.9	*<1.2*	*<5.9*	*<0.84*	1.5	*<23.1*	1,781	*<720*	*<45.0*	*<4.4*	1.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
41	*<17.6*	*<311*	0.12	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	*<1.0*	*<23.1*	1,969	*<720*	*<45.0*	*<4.4*	2.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
42	*<17.6*	954	0.97	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	1.6	*<23.1*	2,169	*<720*	*<45.0*	*<4.4*	1.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
43	*<17.6*	1,813	0.58	*<19.2*	37.2	*<1.2*	*<5.9*	*<0.84*	6.9	*<23.1*	3,877	*<720*	74.9	*<4.4*	8.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*

[a] FIN = foundry identification number; see Table 2-2 for details.
Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-2. Element-Specific Concentrations in Spent Foundry Molding Sands Collected September 2005 (Concentrations in mg kg^{-1})

FIN[a]	**Ag**	**Al**	**As**	**B**	**Ba**	**Be**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Mg**	**Mn**	**Mo**	**Ni**	**Pb**	**Sb**	**V**	**Zn**
1	*<17.6*	3,496	2.4	*<19.2*	38.2	1.3	*<5.9*	1.1	21.4	115	22,989	1,295	199	*<4.4*	17.0	18.4	*<4.5*	*<7.4*	88.2
3	*<17.6*	1,740	2.0	*<19.2*	23.7	*<1.2*	*<5.9*	*<0.84*	5.1	*<23.1*	5,635	1,946	139	*<4.4*	3.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
4	*<17.6*	1,304	0.42	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	4.0	*<23.1*	2,516	*<720*	*<45.0*	*<4.4*	3.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
5	*<17.6*	1,482	1.8	*<19.2*	9.6	*<1.2*	*<5.9*	*<0.84*	5.6	*<23.1*	5,754	*<720*	121	*<4.4*	6.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
6	*<17.6*	353		*<19.2*	39.7	*<1.2*	*<5.9*	*9.10*	25.2	*<23.1*	4,558	26,994	184	*<4.4*	139	*<7.7*	*<4.5*	*<7.4*	*<33.4*
7	*<17.6*	*<311*	0.18	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	7.9	*<23.1*	727	*<720*	*<45.0*	9.2	3.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
8	*<17.6*	655	1.4	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	9.0	*<23.1*	4,938	*<720*	81.6	*<4.4*	4.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
9	*<17.6*	512	0.83	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	1.3	*<23.1*	1,071	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
10	*<17.6*	520	2.0	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*1.07*	51.6	*85.1*	21,860	*<720*	149	*<4.4*	18.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
11	*<17.6*	2,114	2.1	*<19.2*	13.1	*<1.2*	*<5.9*	*<0.84*	2.0	*<23.1*	2,073	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
12	*<17.6*	2,264	0.64	*<19.2*	<8.7	*<1.2*	*<5.9*	*<0.84*	2.5	*<23.1*	904	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
13	*<17.6*	1,865	1.9	*<19.2*	15.1	*<1.2*	*<5.9*	*<0.84*	2.1	*<23.1*	4,170	*<720*	*<45.0*	*<4.4*	3.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
14	*<17.6*	2,206	2.1	*<19.2*	<8.7	*<1.2*	*<5.9*	*<0.84*	2.5	*<23.1*	3,357	1,678	*<45.0*	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
16	*<17.6*	1,143	1.5	*<19.2*	28.8	*<1.2*	*<5.9*	*<0.84*	3.7	*<23.1*	2,520	*<720*	45.1	*<4.4*	2.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
17	*<17.6*	3,173	1.5	*<19.2*	19.2	*<1.2*	*<5.9*	*<0.84*	2.9	*<23.1*	3,942	971	94.6	*<4.4*	3.3	*<7.7*	*<4.5*	*<7.4*	*<33.4*
18	*<17.6*	1,044	2.2	*<19.2*	10.1	*<1.2*	*<5.9*	*<0.84*	2.7	*<23.1*	2,391	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
19	*<17.6*	3,574	3.4	*<19.2*	27.1	*3.5*	*<5.9*	*2.9*	196	114	60,020	1,310	920	19.8	36.7	11.0	*<4.5*	19.3	*<33.4*
20	*<17.6*	6,940	1.9	*<19.2*	19.0	*<1.2*	*<5.9*	*<0.84*	7.5	*<23.1*	3,132	1,267	135	*<4.4*	6.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
21	*<17.6*	<311	0.66	*<19.2*	<8.7	*<1.2*	*<5.9*	*<0.84*	3.5	*<23.1*	5,386	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
22	*<17.6*	3,267	5.1	*<19.2*	20.4	*1.6*	*<5.9*	*1.26*	32.5	53.5	20,340	954	458	6.1	15.4	14.0	*<4.5*	*<7.4*	87.8
23	*<17.6*	2,705	2.7	*<19.2*	29.3	*<1.2*	*<5.9*	*<0.84*	2.6	*<23.1*	3,343	1,031	*<45.0*	*<4.4*	2.8	*<7.7*	*<4.5*	*<7.4*	*<33.4*
24	*<17.6*	2,150	2.4	*<19.2*	12.0	*<1.2*	*<5.9*	*<0.84*	4.0	*<23.1*	4031	*<720*	62.2	*<4.4*	3.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
25	*<17.6*	2,103	1.2	*<19.2*	18.7	*<1.2*	*<5.9*	*<0.84*	2.6	*<23.1*	2,265	*<720*	*<45.0*	*<4.4*	1.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
26	*<17.6*	1,865	2.0	*<19.2*	19.6	*<1.2*	*<5.9*	*<0.84*	2.1	*<23.1*	2,739	*<720*	*<45.0*	*<4.4*	2.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
27	*<17.6*	2,431	2.2	*<19.2*	15.1	*<1.2*	*<5.9*	*<0.84*	13.0	*<23.1*	10,358	979	89.9	*<4.4*	20.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
28	*<17.6*	2,500	2.1	*<19.2*	31.2	*<1.2*	*<5.9*	*<0.84*	5.5	*<23.1*	3,810	813	57.0	*<4.4*	3.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
29	*<17.6*	<311	0.39	*<19.2*	68.4	*<1.2*	*<5.9*	*<0.84*	7.0	*<23.1*	1,087	*<720*	*<45.0*	*<4.4*	15.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
30	*<17.6*	2,120	1.7	*<19.2*	30.0	*<1.2*	*<5.9*	*<0.84*	2.0	*<23.1*	2,704	859	*<45.0*	*<4.4*	1.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
31	*<17.6*	1,213	1.5	*<19.2*	14.5	*<1.2*	*<5.9*	*<0.84*	11.0	*<23.1*	3,564	*<720*	*<45.0*	*<4.4*	14.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
33	*<17.6*	*<311*	0.82	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	2.5	14,360	789	*<720*	*<45.0*	*<4.4*	34.5	20.6	*<4.5*	*<7.4*	986
34	*<17.6*	513	0.78	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	3.5	14,220	1,704	*<720*	*<45.0*	*<4.4*	21.1	28.9	*<4.5*	*<7.4*	1732
35	*<17.6*	2,072	2.6	*<19.2*	25.9	*<1.2*	*<5.9*	*<0.84*	1.5	*<23.1*	4,196	807	*<45.0*	*<4.4*	2.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
36	*<17.6*	2,529	3.0	*<19.2*	20.9	*<1.2*	*<5.9*	*<0.84*	2.4	*<23.1*	3,962	*<720*	50.6	*<4.4*	2.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
37	*<17.6*	1,811	1.0	*<19.2*	13.1	*<1.2*	*<5.9*	*<0.84*	2.3	*<23.1*	3,048	*<720*	67.1	*<4.4*	2.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
40	*<17.6*	1,823	0.67	*<19.2*	15.8	*<1.2*	*<5.9*	*<0.84*	1.7	*<23.1*	1,760	*<720*	*<45.0*	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
41	*<17.6*	*<311*	0.13	*<19.2*	24.4	*<1.2*	*<5.9*	*<0.84*	<1.0	*<23.1*	4,926	*<720*	*<45.0*	*<4.4*	5.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
42	*<17.6*	1,162	1.1	*<19.2*	10.0	*<1.2*	*<5.9*	*<0.84*	1.7	*<23.1*	2,743	*<720*	<45	*<4.4*	<1.2	*<7.7*	*<4.5*	*<7.4*	*<33.4*
43	*<17.6*	1,856	1.4	*<19.2*	72.5	*<1.2*	*<5.9*	*<0.84*	5.7	*<23.1*	2,647	*<720*	<45	*<4.4*	7.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 15, 32, 38, and 39.

Table B-3. Element-Specific Concentrations in Spent Foundry Molding Sands Collected July 2006 (Concentrations in mg kg^{-1})

FIN[a]	Ag	Al	As	B	Ba	Be	Cd	Co	Cr	Cu	Fe	Mg	Mn	Mo	Ni	Pb	Sb	V	Zn
1	*<17.6*	3,431	2.0	*<19.2*	45.5	*<1.2*	*<5.9*	*<0.84*	5.0	31.3	5,265	*<720*	80.7	*<4.4*	3.2	10.6	*<4.5*	*<7.4*	*<33.4*
3	*<17.6*	1,780	1.0	*<19.2*	24.1	*<1.2*	*<5.9*	*<0.84*	*<1.0*	*<23.1*	1,575	2,218	46.5	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
4	*<17.6*	1,072	0.47	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	2.2	*<23.1*	1,542	*<720*	*<45.0*	*<4.4*	1.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
6	*<17.6*	*<311*	0.24	*<19.2*	*<8.7*	*<1.2*	*<5.9*	6.1	2.9	*<23.1*	4,287	15,990	59.7	*<4.4*	110.8	*<7.7*	*<4.5*	*<7.4*	*<33.4*
7	*<17.6*	*<311*	0.07	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	5.3	*<23.1*	566	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
8	*<17.6*	816	0.68	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	6.8	*<23.1*	4,549	*<720*	62.6	*<4.4*	4.6	9.6	*<4.5*	*<7.4*	*<33.4*
9	*<17.6*	370	0.31	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	*<1.0*	*<23.1*	931	*<720*	*<45.0*	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
10	*<17.6*	650	1.2	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	30.5	78.1	15,596	*<720*	128	*<4.4*	10.5	19.6	*<4.5*	*<7.4*	*<33.4*
11	*<17.6*	2,197	1.1	*<19.2*	28.8	*<1.2*	*<5.9*	*<0.84*	2.1	*<23.1*	2,530	*<720*	*<45.0*	*<4.4*	3.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
12	*<17.6*	1,416	0.34	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	1.6	*<23.1*	710	*<720*	*<45.0*	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
13	*<17.6*	1,870	1.0	*<19.2*	23.0	*<1.2*	*<5.9*	*<0.84*	*<1.0*	*<23.1*	1,841	*<720*	*<45.0*	*<4.4*	2.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*
14	*<17.6*	2,086	1.3	*<19.2*	9.5	*<1.2*	*<5.9*	*<0.84*	2.2	*<23.1*	2,970	*<720*	*<45.0*	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
16	*<17.6*	981	0.68	*<19.2*	19.4	*<1.2*	*<5.9*	*<0.84*	4.0	*<23.1*	2,770	*<720*	*<45.0*	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
17	*<17.6*	3,068	1.4	*<19.2*	53.1	*<1.2*	*<5.9*	*<0.84*	4.0	*<23.1*	3,751	*<720*	81.4	*<4.4*	4.7	*<7.7*	*<4.5*	*<7.4*	*<33.4*
18	*<17.6*	1,044	1.8	*<19.2*	12.3	*<1.2*	*<5.9*	*<0.84*	2.2	*<23.1*	2,263	*<720*	*<45.0*	*<4.4*	2.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
19	*<17.6*	2,092	0.72	*<19.2*	25.5	*<1.2*	*<5.9*	*<0.84*	12.7	*<23.1*	7,727	*<720*	99.0	*<4.4*	4.8	12.8	*<4.5*	*<7.4*	*<33.4*
20	*<17.6*	4,680	0.85	*<19.2*	12.9	*<1.2*	*<5.9*	*<0.84*	7.6	47.0	2,473	*<720*	83.8	*<4.4*	7.1	*<7.7*	*<4.5*	*<7.4*	23.4
21	*<17.6*	*<311*	0.47	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	2.9	*<23.1*	3,760	*<720*	*<45.0*	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
22	*<17.6*	6,189	4.9	*<19.2*	49.1	*<1.2*	*<5.9*	2.2	32.1	63.2	25,310	906	483	*<4.4*	17.7	62.6	*<4.5*	*<7.4*	352
23	*<17.6*	2,777	1.4	*<19.2*	27.1	*<1.2*	*<5.9*	*<0.84*	2.7	*<23.1*	2,999	*<720*	*<45.0*	*<4.4*	3.3	*<7.7*	*<4.5*	*<7.4*	*<33.4*
24	*<17.6*	1,936	1.1	*<19.2*	12.2	*<1.2*	*<5.9*	*<0.84*	3.0	*<23.1*	3,115	*<720*	60.8	*<4.4*	3.6	8.2	*<4.5*	*<7.4*	*<33.4*
25	*<17.6*	2,075	0.47	*<19.2*	20.6	*<1.2*	*<5.9*	*<0.84*	2.4	*<23.1*	2,209	*<720*	*<45.0*	*<4.4*	2.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
26	*<17.6*	1,592	2.0	*<19.2*	18.4	*<1.2*	*<5.9*	*<0.84*	1.9	*<23.1*	2,543	*<720*	*<45.0*	*<4.4*	2.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
27	*<17.6*	2,741	1.6	*<19.2*	26.1	*<1.2*	*<5.9*	1.2	8.5	*<23.1*	4,748	*<720*	66.7	*<4.4*	7.0	9.0	*<4.5*	*<7.4*	*<33.4*
28	*<17.6*	1,795	0.67	*<19.2*	21.1	*<1.2*	*<5.9*	*<0.84*	4.5	*<23.1*	4,703	*<720*	80.8	*<4.4*	3.1	8.1	*<4.5*	*<7.4*	*<33.4*
29	*<17.6*	387	0.17	*<19.2*	110	*<1.2*	*<5.9*	*<0.84*	8.0	*<23.1*	1,047	*<720*	*<45.0*	*<4.4*	8.6	*<7.7*	*<4.5*	*<7.4*	*<33.4*
30	*<17.6*	1,792	0.66	*<19.2*	27.0	*<1.2*	*<5.9*	*<0.84*	1.9	*<23.1*	2,578	*<720*	*<45.0*	*<4.4*	2.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
31	*<17.6*	410	0.27	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	2.5	*<23.1*	1,696	*<720*	*<45.0*	*<4.4*	3.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
33	*<17.6*	*<311*	0.08	*<19.2*	*<8.7*	*<1.2*	*<5.9*	*<0.84*	*<1.0*	38.5	*<352*	*<720*	*<45.0*	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
34	*<17.6*	1,681	1.2	*<19.2*	14.9	*<1.2*	*<5.9*	*<0.84*	5.7	4668	4,339	*<720*	*<45.0*	*<4.4*	16.3	212	*<4.5*	*<7.4*	2829
36	*<17.6*	2,739	1.5	*<19.2*	22.3	*<1.2*	*<5.9*	*<0.84*	2.4	*<23.1*	2,862	*<720*	50.6	*<4.4*	3.4	*<7.7*	*<4.5*	*<7.4*	*<33.4*
37	*<17.6*	1,269	1.0	*<19.2*	12.1	*<1.2*	*<5.9*	*<0.84*	1.7	*<23.1*	2,281	*<720*	*<45.0*	*<4.4*	2.5	*<7.7*	*<4.5*	*<7.4*	*<33.4*
38	*<17.6*	911	3.0	*<19.2*	60.3	2.5	*<5.9*	9.1	132	177	45,120	16,566	845	54.6	189	46.6	*<4.5*	9.7	*<33.4*
39	*<17.6*	1,298	0.34	*<19.2*	149	*<1.2*	*<5.9*	*<0.84*	5.6	25.7	3,162	*<720*	85.2	*<4.4*	15.1	*<7.7*	*<4.5*	*<7.4*	*<33.4*
40	*<17.6*	1,612	0.70	*<19.2*	15.1	*<1.2*	*<5.9*	*<0.84*	1.6	*<23.1*	1,628	*<720*	*<45.0*	*<4.4*	2.0	*<7.7*	*<4.5*	*<7.4*	*<33.4*
42	*<17.6*	799	0.77	*<19.2*	13.8	*<1.2*	*<5.9*	*<0.84*	1.3	*<23.1*	2,787	*<720*	*<45.0*	*<4.4*	*<1.2*	*<7.7*	*<4.5*	*<7.4*	*<33.4*
43	*<17.6*	1,092	0.59	*<19.2*	39.3	*<1.2*	*<5.9*	*<0.84*	14.3	*<23.1*	1,682	*<720*	*<45.0*	*<4.4*	3.9	*<7.7*	*<4.5*	*<7.4*	*<33.4*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 5, 15, 32, 35, and 41.

Table B-4. Polycyclic Aromatic Hydrocarbon Concentrations in Spent Foundry Molding Sands Collected June 2005 (Concentration in mg kg^{-1})

FIN[a]	**Acenaphthene**	**Acenaphthylene**	**Anthracene**	**Benz[a]anthracene**	**Benzo[b]fluoranthene**	**Benzo[k]fluoranthene**	**Benzo[ghi]perylene**	**Benzo[a]pyrene**	**Chrysene**	**Dibenz[a,h]anthracene**	**Fluoranthene**	**Fluorene**	**Indeno[1,2,3-cd]pyrene**	**Naphthalene**	**Phenanthrene**	**Pyrene**
1	*<0.04*	0.29	0.13	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.15	*<0.14*	4.2	0.46	0.31
2	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.22	*<0.14*	0.39	0.49	0.07
3	*<0.04*	0.13	0.11	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.18	*<0.14*	0.75	0.70	0.10
4	0.04	*<0.03*	0.16	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.07	*<0.14*	0.09	0.29	0.08
5	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.27	*<0.14*	0.41	0.52	0.31
6	0.07	0.09	0.37	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.15	*<0.14*	48.1	0.45	0.06
7	*<0.04*	0.09	0.11	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.16	*<0.14*	0.59	0.23	0.06
8	0.40	0.09	0.31	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*0.50*	0.33	*<0.14*	0.67	0.72	0.53
9	*<0.04*	0.05	0.38	*0.17*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*0.18*	*<0.16*	*<0.06*	0.49	*<0.14*	0.53	0.76	0.19
10	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.14	0.08	*<0.03*
11	*<0.04*	0.15	0.87	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.83	*<0.14*	2.1	1.6	*<0.03*
12	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	*<0.03*	*<0.03*
13	*<0.04*	*<0.03*	0.81	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.71	*<0.14*	3.3	1.5	0.35
14	*<0.04*	*<0.03*	0.53	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.41	*<0.14*	0.45	0.42	*<0.03*
15	0.04	0.04	0.57	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.29	*<0.14*	2.1	0.62	*<0.03*
16	0.09	*<0.03*	0.52	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.36	*<0.14*	0.86	0.71	0.29
17	*<0.04*	0.09	0.25	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.35	*<0.14*	0.94	0.73	0.26
18	*<0.04*	*<0.03*	0.08	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.07	*<0.14*	0.66	0.30	*<0.03*
19	*<0.04*	0.10	0.36	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.28	*<0.14*	0.66	0.62	0.21
20	11.7	*<0.03*	0.89	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.69	*<0.14*	1.1	1.2	0.46
21	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.18	*<0.14*	0.31	0.45	0.19
22	*<0.04*	*<0.03*	0.95	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.42	*<0.14*	0.89	1.2	0.23
23	*<0.04*	*<0.03*	0.37	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.10	*<0.14*	0.54	0.17	*<0.03*
24	*<0.04*	0.08	0.15	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.31	*<0.14*	1.1	0.70	0.05
25	*<0.04*	0.07	0.27	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.21	*<0.14*	0.74	0.32	*<0.03*
26	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.21	*<0.14*	0.73	0.57	*<0.03*
27	2.9	0.10	0.67	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.43	0.49	*<0.14*	3.0	1.1	0.35
28	0.26	0.03	0.70	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.12	*<0.14*	1.1	0.69	*<0.03*
29	0.19	*<0.03*	0.11	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.05	*<0.14*	0.22	0.10	*<0.03*
30	0.04	*<0.03*	0.14	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.08	*<0.14*	0.26	0.15	*<0.03*
31	*<0.04*	*<0.03*	0.06	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.06	*<0.14*	*<0.03*	0.06	*<0.03*
32	0.06	*<0.03*	0.61	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.38	*<0.14*	6.8	0.60	*<0.03*
33	*<0.04*	0.09	0.04	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.05	*<0.14*	27.6	0.06	*<0.03*
34	*<0.04*	0.17	0.91	*0.30*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*0.30*	*<0.16*	*<0.06*	2.58	*<0.14*	5.3	2.2	0.43
35	*<0.04*	0.06	0.09	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.37	*<0.14*	0.25	0.62	0.27
36	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	3.5	0.99	0.27
37	*<0.04*	0.19	0.55	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.67	*<0.14*	2.4	1.8	0.41
38	*<0.04*	*<0.03*	0.31	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.04	*<0.14*	*<0.03*	*<0.03*	*<0.03*
39	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.09	*<0.14*	0.16	0.13	*<0.03*
40	0.42	0.26	0.90	*0.19*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*0.22*	*<0.16*	*<0.06*	0.75	*<0.14*	0.65	1.2	0.30
41	*<0.04*	0.03	0.11	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.12	*<0.14*	32.9	0.45	*<0.03*
42	*<0.04*	*<0.03*	0.18	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.15	*<0.14*	0.95	0.36	*<0.03*
43	<0.04	<0.03	<0.03	<0.10	<0.12	<0.13	<0.14	<0.20	<0.08	<0.16	<0.06	<0.04	<0.14	0.27	0.09	<0.03

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-5. Polycyclic Aromatic Hydrocarbon Concentrations in Spent Foundry Molding Sands Collected September 2005 (Conc. in mg kg^{-1})

FIN[a]	Acenaph-thene	Acenaph-thylene	Anthracene	Benz[a]-anthracene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[ghi]-perylene	Benzo[a]-pyrene	Chrysene	Dibenz[a,h]-anthracene	Fluoranthene	Fluorene	Indeno[1,2,3-cd]pyrene	Naphthalene	Phenan-threne	Pyrene
1	*<0.04*	0.17	0.24	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.25	*<0.14*	1.6	1.4	0.06
3	*<0.04*	*<0.03*	0.18	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.26	0.18	*<0.03*
4	*<0.04*	*<0.03*	0.33	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.34	0.37	*<0.03*
5	0.06	0.06	0.62	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.10	0.50	*<0.14*	0.84	0.62	0.24
6	*<0.04*	0.28	0.38	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.06	0.38	*<0.14*	8.3	0.43	*<0.03*
7	0.09	*<0.03*	0.27	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.25	*<0.14*	0.43	0.29	0.06
8	0.05	*<0.03*	0.52	0.20	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*0.11*	*<0.16*	1.0	0.11	*<0.14*	0.16	1.29	0.86
9	*<0.04*	*<0.03*	0.69	0.13	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.11	0.47	*<0.14*	0.38	0.94	0.20
10	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.39	0.94	*<0.03*
11	0.11	*<0.03*	0.99	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.55	*<0.14*	1.5	1.4	0.18
12	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	*<0.03*	*<0.03*
13	*<0.04*	0.03	0.43	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.44	*<0.14*	1.9	0.91	0.12
14	*<0.04*	*<0.03*	0.84	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.53	*<0.14*	0.59	1.0	0.47
16	*<0.04*	*<0.03*	0.49	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.36	*<0.14*	0.55	0.77	*<0.03*
17	*<0.04*	*<0.03*	0.44	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.11	*<0.14*	0.34	0.39	*<0.03*
18	*<0.04*	0.08	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.36	*<0.14*	0.27	1.1	0.16
19	*<0.04*	*<0.03*	0.36	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.25	*<0.14*	0.51	0.40	0.07
20	0.09	*<0.03*	0.83	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.56	*<0.14*	0.66	0.97	0.47
21	*<0.04*	*<0.03*	0.23	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	0.20	*<0.03*
22	*<0.04*	0.04	0.62	0.10	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.45	*<0.14*	0.81	0.64	0.23
23	0.17	0.10	0.54	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.41	*<0.14*	0.09	0.57	0.22
24	*<0.04*	*<0.03*	0.97	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	1.2	*<0.14*	1.4	1.9	0.80
25	*<0.04*	*<0.03*	0.33	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.19	*<0.14*	0.43	0.45	0.09
26	*<0.04*	*<0.03*	0.50	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.49	*<0.14*	0.89	1.3	0.33
27	*<0.04*	*<0.03*	0.28	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.10	*<0.14*	0.19	0.54	*<0.03*
28	*<0.04*	*<0.03*	0.14	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.08	*<0.14*	0.95	0.14	0.07
29	0.06	*<0.03*	0.35	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.16	*<0.14*	*<0.03*	0.29	*<0.03*
30	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.71	*<0.14*	0.74	1.1	0.49
31	*<0.04*	*<0.03*	0.06	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.11	*<0.14*	0.36	0.26	*<0.03*
33	*<0.04*	0.32	0.46	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.54	*<0.14*	9.8	0.66	0.23
34	*<0.04*	0.25	0.45	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.84	*<0.14*	0.74	0.90	0.27
35	*<0.04*	0.11	0.36	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.26	*<0.14*	0.29	0.57	*<0.03*
36	0.12	*<0.03*	0.35	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	1.1	0.88	0.10
37	*<0.04*	0.12	0.66	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.23	0.88	*<0.14*	2.2	1.7	0.16
40	0.18	0.09	0.48	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.41	*<0.14*	1.1	0.77	0.04
41	*<0.04*	*<0.03*	0.53	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	0.17	*<0.06*	0.38	*<0.14*	14.6	0.65	0.13
42	0.04	*<0.03*	0.53	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.24	*<0.14*	0.68	0.55	*<0.03*
43	*<0.04*	0.20	0.05	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.16	*<0.14*	0.10	0.23	*<0.03*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 15, 32, 38, and 39.

Table B-6. Polycyclic Aromatic Hydrocarbon Concentrations in Spent Foundry Molding Sands Collected July 2006 (Concentrations in mg kg^{-1})

FIN[a]	Acenaph-thene	Acenaph-thylene	Anthracene	Benz[a]-anthracene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[ghi]-perylene	Benzo[a]-pyrene	Chrysene	Dibenz[a,h]-anthracene	Fluoranthene	Fluorene	Indeno[1,2,3-cd]pyrene	Naphthalene	Phenanthrene	Pyrene
1	*<0.04*	0.03	0.09	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.14	*<0.14*	0.65	0.30	0.07
3	*<0.04*	*<0.03*	0.10	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.14	*<0.14*	0.67	0.39	0.09
4	*<0.04*	*<0.03*	0.10	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.07	0.18	0.06
6	*<0.04*	*<0.03*	0.11	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.39	*<0.14*	42.2	0.18	0.06
7	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.06	*<0.14*	0.16	0.09	*<0.03*
8	*<0.04*	*<0.03*	0.25	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.18	0.28	*<0.14*	0.35	0.43	0.19
9	*<0.04*	*<0.03*	0.13	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	*<0.03*	*<0.03*
10	*<0.04*	*<0.03*	0.07	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.08	*<0.14*	0.03	0.19	0.06
11	*0.11*	0.14	0.69	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.33	0.64	*<0.14*	2.8	1.9	0.29
12	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	*<0.03*	*<0.03*
13	*0.10*	0.07	0.25	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.06	0.32	*<0.14*	0.63	0.80	0.11
14	*0.11*	*<0.03*	0.18	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.24	*<0.14*	0.14	0.64	0.11
16	*<0.04*	*<0.03*	0.16	*0.15*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.10	0.14	*<0.14*	0.50	0.43	0.10
17	*<0.04*	*<0.03*	0.10	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.07	*<0.14*	0.17	0.22	0.06
18	*<0.04*	0.25	0.60	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.56	*<0.14*	0.21	0.80	0.09
19	*<0.04*	*<0.03*	0.13	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.16	*<0.14*	0.43	0.43	0.10
20	*<0.04*	*<0.03*	0.37	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.30	*<0.14*	0.64	1.4	0.27
21	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	*<0.03*	0.09	0.04
22	*<0.04*	*<0.03*	0.17	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.17	*<0.14*	0.64	0.61	0.12
23	*<0.04*	0.07	0.19	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.12	*<0.14*	0.30	0.31	0.08
24	*0.40*	0.16	0.56	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	1.0	*<0.14*	1.9	1.1	0.73
25	*<0.04*	*<0.03*	0.04	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.06	*<0.14*	0.08	0.37	0.09
26	*<0.04*	0.05	0.23	*0.15*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.42	*<0.14*	0.89	0.44	0.13
27	*<0.04*	0.04	0.15	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.06	0.14	*<0.14*	1.0	0.44	0.13
28	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.05	*<0.14*	0.53	0.74	0.04
29	*<0.04*	*<0.03*	0.17	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.07	*<0.14*	0.08	0.18	0.06
30	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.07	*<0.14*	0.12	0.25	0.08
31	*<0.04*	0.05	0.05	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.04	*<0.14*	0.17	0.10	0.03
33	*<0.04*	0.08	0.20	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.46	*<0.14*	9.6	0.46	0.05
34	*0.06*	0.10	0.35	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.83	*<0.14*	2.0	1.6	0.24
36	*<0.04*	*<0.03*	*<0.03*	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.22	*<0.14*	5.8	0.48	0.09
37	*0.05*	0.07	0.27	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.41	*<0.14*	0.67	0.81	0.12
38	*<0.04*	*<0.03*	0.10	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.03	0.11	0.05
39	*<0.04*	*<0.03*	0.07	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.05	0.21	*<0.03*
40	0.25	0.33	0.60	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	0.09	0.41	*<0.14*	0.54	0.73	0.06
42	0.04	*<0.03*	0.46	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	0.18	*<0.14*	0.41	0.43	0.05
43	*<0.04*	*<0.03*	0.10	*<0.10*	*<0.12*	*<0.13*	*<0.14*	*<0.20*	*<0.08*	*<0.16*	*<0.06*	*<0.04*	*<0.14*	0.03	0.11	0.05

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 5, 15, 32, 35, and 41.

Table B-7. Phenolics Concentrations in the Spent Foundry Molding Sands Collected June 2005 (Concentrations in mg kg^{-1})

FIN[a]	2-Sec-Butyl-4,6-Dinitrophenol	4-Chloro-3-Methylphenol	2-Chlorophenol	2,4-Dichlorophenol	2,6-Dichlorophenol	2,4-Dimethylphenol	2,4-Dinitrophenol	2-Methylphenol	3- and 4-Methylphenol	2-Methyl-4,6-Dinitrophenol	2-Nitrophenol	4-Nitrophenol	Pentachlorophenol	Phenol	2,3,4,6-Tetrachlorophenol	2,4,6-Trichlorophenol	2,4,5-Trichlorophenol
1	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.70	*<0.24*	2.0	1.1	*<0.16*	*<0.09*	*<0.44*	*<0.24*	5.2	*<0.09*	*<0.12*	*<0.14*
2	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.17	*<0.09*	*<0.12*	*<0.14*
3	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.30	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.22	*<0.09*	*<0.12*	*<0.14*
4	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.51	*<0.24*	0.33	0.53	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.56	*<0.09*	*<0.12*	*<0.14*
5	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.39	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.89	*<0.09*	*<0.12*	*<0.14*
6	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	4.1	0.86	8.6	4.9	*<0.16*	*<0.09*	*<0.44*	*<0.24*	186.2	*<0.09*	*<0.12*	*<0.14*
7	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.19	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.3	*<0.09*	*<0.12*	*<0.14*
8	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.40	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.4	*<0.09*	*<0.12*	*<0.14*
9	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.36	*<0.24*	1.05	0.49	*<0.16*	*<0.09*	*<0.44*	*<0.24*	3.3	*<0.09*	*<0.12*	*<0.14*
10	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
11	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	3.85	*<0.24*	6.26	3.42	*<0.16*	*<0.09*	*<0.44*	*<0.24*	14.9	*<0.09*	*<0.12*	*<0.14*
12	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
13	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	7.5	*<0.24*	9.5	3.7	*<0.16*	*<0.09*	*<0.44*	*<0.24*	8.1	*<0.09*	*<0.12*	*<0.14*
14	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.4	*<0.24*	1.5	0.54	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.94	*<0.09*	*<0.12*	*<0.14*
15	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.68	*<0.24*	2.7	0.74	*<0.16*	*<0.09*	*<0.44*	*<0.24*	3.6	*<0.09*	*<0.12*	*<0.14*
16	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.1	*<0.24*	1.9	1.2	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.5	*<0.09*	*<0.12*	*<0.14*
17	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.79	*<0.24*	1.0	0.81	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.6	*<0.09*	*<0.12*	*<0.14*
18	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.38	*<0.09*	*<0.12*	*<0.14*
19	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.64	*<0.24*	1.5	0.64	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.9	*<0.09*	*<0.12*	*<0.14*
20	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	12.30	*<0.24*	11.7	6.1	*<0.16*	*<0.09*	*<0.44*	*<0.24*	24.7	*<0.09*	*<0.12*	*<0.14*
21	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
22	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.92	0.49	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.7	*<0.09*	*<0.12*	*<0.14*
23	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.09	*<0.24*	1.5	0.3	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.3	*<0.09*	*<0.12*	*<0.14*
24	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.46	*<0.24*	1.5	1.2	*<0.16*	*<0.09*	*<0.44*	*<0.24*	3.1	*<0.09*	*<0.12*	*<0.14*
25	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.90	0.13	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.7	*<0.09*	*<0.12*	*<0.14*
26	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.27	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.39	*<0.09*	*<0.12*	*<0.14*
27	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.53	*<0.24*	2.9	1.1	*<0.16*	*<0.09*	*<0.44*	*<0.24*	10.5	*<0.09*	*<0.12*	*<0.14*
28	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.16	*<0.24*	0.26	0.24	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.48	*<0.09*	*<0.12*	*<0.14*
29	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.65	*<0.24*	*<0.21*	0.20	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.36	*<0.09*	*<0.12*	*<0.14*
30	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.11	*<0.09*	*<0.12*	*<0.14*
31	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.61	*<0.09*	*<0.12*	*<0.14*
32	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.2	*<0.24*	6.8	2.5	*<0.16*	*<0.09*	*<0.44*	*<0.24*	28.6	*<0.09*	*<0.12*	*<0.14*
33	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.25	*<0.24*	1.2	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	18.9	*<0.09*	*<0.12*	*<0.14*
34	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	4.57	*<0.24*	14.9	4.8	*<0.16*	*<0.09*	*<0.44*	*<0.24*	70.6	*<0.09*	*<0.12*	*<0.14*
35	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.16	*<0.24*	0.29	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.50	*<0.09*	*<0.12*	*<0.14*
36	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.23	*<0.24*	1.06	0.30	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.5	*<0.09*	*<0.12*	*<0.14*
37	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.3	*<0.24*	3.8	2.8	*<0.16*	*<0.09*	*<0.44*	*<0.24*	6.3	*<0.09*	*<0.12*	*<0.14*
38	*<0.21*	0.31	*<0.11*	*<0.13*	*<0.06*	0.48	*<0.24*	0.29	0.23	*<0.16*	*<0.09*	*<0.44*	*<0.24*	10.8	*<0.09*	*<0.12*	*<0.14*
39	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.25	0.18	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.55	*<0.09*	*<0.12*	*<0.14*
40	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.1	*<0.24*	3.3	2.1	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.7	*<0.09*	*<0.12*	*<0.14*
41	*<0.21*	0.81	*<0.11*	*<0.13*	*<0.06*	0.30	*<0.24*	0.65	0.16	*<0.16*	*<0.09*	*<0.44*	*<0.24*	46.1	*<0.09*	*<0.12*	*<0.14*
42	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.50	*<0.24*	3.17	0.52	*<0.16*	*<0.09*	*<0.44*	*<0.24*	10.0	*<0.09*	*<0.12*	*<0.14*
43	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-8. Phenolics Concentrations in the Spent Foundry Molding Sands Collected September 2005 (concentrations in mg kg^{-1})

FIN[a]	2-Sec-Butyl-4,6-Dinitro-phenol	4-Chloro-3-Methylphenol	2-Chloro-phenol	2,4-Dichloro-phenol	2,6-Dichloro-phenol	2,4-Dimethyl-phenol	2,4-Dinitro-phenol	2-Methyl-phenol	3- and 4-Methylphenol	2-Methyl-4,6-Dinitrophenol	2-Nitrophenol	4-Nitrophenol	Pentachloro-phenol	Phenol	2,3,4,6-Tetrachloro-phenol	2,4,6-Trichloro-phenol	2,4,5-Trichlo-rophenol
1	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.67	0.20	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.3	*<0.09*	*<0.12*	*<0.14*
3	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.19	*<0.24*	0.26	0.21	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.41	*<0.09*	*<0.12*	*<0.14*
4	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.66	*<0.24*	1.3	0.43	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.85	*<0.09*	*<0.12*	*<0.14*
5	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.19	*<0.24*	1.23	0.46	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.7	*<0.09*	*<0.12*	*<0.14*
6	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.86	*<0.24*	2.9	0.73	*<0.16*	*<0.09*	*<0.44*	*<0.24*	50.0	*<0.09*	*<0.12*	*<0.14*
7	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.10	*<0.24*	*<0.21*	0.14	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.74	*<0.09*	*<0.12*	*<0.14*
8	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.09	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.63	*<0.09*	*<0.12*	*<0.14*
9	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.23	*<0.24*	0.67	0.34	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.7	*<0.09*	*<0.12*	*<0.14*
10	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
11	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	3.1	*<0.24*	5.3	2.3	*<0.16*	*<0.09*	*<0.44*	*<0.24*	8.1	*<0.09*	*<0.12*	*<0.14*
12	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
13	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.1	*<0.24*	3.0	1.3	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.8	*<0.09*	*<0.12*	*<0.14*
14	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.5	*<0.24*	1.3	0.36	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.47	*<0.09*	*<0.12*	*<0.14*
16	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.78	*<0.24*	1.5	0.65	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.6	*<0.09*	*<0.12*	*<0.14*
17	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.13	*<0.24*	0.34	0.30	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.44	*<0.09*	*<0.12*	*<0.14*
18	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.08	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.37	*<0.09*	*<0.12*	*<0.14*
19	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.46	*<0.24*	1.1	0.42	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.7	*<0.09*	*<0.12*	*<0.14*
20	*<0.21*	*0.45*	*<0.11*	*<0.13*	*<0.06*	7.5	*<0.24*	5.5	1.7	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.9	*<0.09*	*<0.12*	*<0.14*
21	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
22	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.09	*<0.24*	0.56	0.20	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.3	*<0.09*	*<0.12*	*<0.14*
23	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.16	*<0.09*	*<0.12*	*<0.14*
24	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.52	*<0.24*	0.68	0.65	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.93	*<0.09*	*<0.12*	*<0.14*
25	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.27	0.11	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.54	*<0.09*	*<0.12*	*<0.14*
26	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.23	*<0.24*	0.46	0.87	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.89	*<0.09*	*<0.12*	*<0.14*
27	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.21	0.17	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.53	*<0.09*	*<0.12*	*<0.14*
28	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.15	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.24	*<0.09*	*<0.12*	*<0.14*
29	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.71	*<0.24*	0.26	0.25	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.77	*<0.09*	*<0.12*	*<0.14*
30	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.13	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.10	*<0.09*	*<0.12*	*<0.14*
31	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.48	*<0.09*	*<0.12*	*<0.14*
33	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.21	*<0.24*	0.69	0.19	*<0.16*	*<0.09*	*<0.44*	*<0.24*	13.2	*<0.09*	*<0.12*	*<0.14*
34	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.13	*<0.24*	0.27	0.12	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.73	*<0.09*	*<0.12*	*<0.14*
35	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.23	0.96	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.6	*<0.09*	*<0.12*	*<0.14*
36	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.33	*<0.24*	0.74	0.32	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.3	*<0.09*	*<0.12*	*<0.14*
37	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.7	*<0.24*	4.4	3.2	*<0.16*	*<0.09*	*<0.44*	*<0.24*	11.0	*<0.09*	*<0.12*	*<0.14*
40	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	3.3	*<0.24*	9.9	4.0	*<0.16*	*<0.09*	*<0.44*	*<0.24*	14.2	*<0.09*	*<0.12*	*<0.14*
41	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.27	*<0.24*	0.74	0.09	*<0.16*	*<0.09*	*<0.44*	*<0.24*	20.2	*<0.09*	*<0.12*	*<0.14*
42	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.63	*<0.24*	3.36	0.78	*<0.16*	*<0.09*	*<0.44*	*<0.24*	11.2	*<0.09*	*<0.12*	*<0.14*
43	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.09	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.12	*<0.09*	*<0.12*	*<0.14*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Data were not available for FIN 2, 15, 32, 38, and 39.

Table B-9. Phenolic Concentrations in the Spent Foundry Molding Sands Collected July 2006 (Concentrations in mg kg^{-1})

FIN[a]	2-Sec-Butyl-4,6-Dinitro-phenol	4-Chloro-3-Methylphenol	2-Chloro-phenol	2,4-Dichloro-phenol	2,6-Dichloro-phenol	2,4-Dimethyl-phenol	2,4-Dinitro-phenol	2-Methyl-phenol	3- and 4-Methylphenol	2-Methyl-4,6-Dinitrophenol	2-Nitrophenol	4-Nitrophenol	Pentachloro-phenol	Phenol	2,3,4,6-Tetrachloro-phenol	2,4,6-Trichloro-phenol	2,4,5-Trichlo-rophenol
3	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.0	*<0.24*	2.6	1.6	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.5	*<0.09*	*<0.12*	*<0.14*
4	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.59	*<0.24*	0.69	0.23	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.38	*<0.09*	*<0.12*	*<0.14*
6	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.61	*<0.24*	1.7	0.33	*<0.16*	*<0.09*	*<0.44*	*<0.24*	28.5	*<0.09*	*<0.12*	*<0.14*
7	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
8	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.28	*<0.24*	0.53	0.20	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.91	*<0.09*	*<0.12*	*<0.14*
9	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
10	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	0.11	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.47	*<0.09*	*<0.12*	*<0.14*
11	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	4.4	*<0.24*	8.8	4.1	*<0.16*	*<0.09*	*<0.44*	*<0.24*	23.4	*<0.09*	*<0.12*	*<0.14*
12	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
13	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	4.2	*<0.24*	5.4	2.6	*<0.16*	*<0.09*	*<0.44*	*<0.24*	7.7	*<0.09*	*<0.12*	*<0.14*
14	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	10.9	*<0.24*	8.7	2.7	*<0.16*	*<0.09*	*<0.44*	*<0.24*	6.1	*<0.09*	*<0.12*	*<0.14*
16	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.80	*<0.24*	1.6	0.64	*<0.16*	*<0.09*	*<0.44*	*<0.24*	3.9	*<0.09*	*<0.12*	*<0.14*
17	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
18	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.32	*<0.24*	*<0.21*	0.15	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.10	*<0.09*	*<0.12*	*<0.14*
19	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.62	*<0.24*	2.3	0.76	*<0.16*	*<0.09*	*<0.44*	*<0.24*	4.5	*<0.09*	*<0.12*	*<0.14*
20	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	6.4	*<0.24*	10.5	4.7	*<0.16*	*<0.09*	*<0.44*	*<0.24*	23.7	*<0.09*	*<0.12*	*<0.14*
21	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
22	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.22	0.27	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.2	*<0.09*	*<0.12*	*<0.14*
23	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.26	*<0.24*	0.73	0.26	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.5	*<0.09*	*<0.12*	*<0.14*
24	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.78	*<0.24*	0.84	0.83	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.4	*<0.09*	*<0.12*	*<0.14*
25	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	0.29	0.08	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.6	*<0.09*	*<0.12*	*<0.14*
26	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.1	*<0.24*	1.7	3.2	*<0.16*	*<0.09*	*<0.44*	*<0.24*	3.1	*<0.09*	*<0.12*	*<0.14*
27	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.24	*<0.24*	1.5	0.49	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.4	*<0.09*	*<0.12*	*<0.14*
28	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.29	*<0.09*	*<0.12*	*<0.14*
29	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.44	*<0.24*	*<0.21*	0.13	*<0.16*	*<0.09*	*<0.44*	*<0.24*	1.0	*<0.09*	*<0.12*	*<0.14*
30	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.09	*<0.24*	0.21	0.20	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.65	*<0.09*	*<0.12*	*<0.14*
31	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.46	*<0.09*	*<0.12*	*<0.14*
33	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.56	*<0.24*	1.8	0.41	*<0.16*	*<0.09*	*<0.44*	*<0.24*	20.0	*<0.09*	*<0.12*	*<0.14*
34	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.82	*<0.24*	3.8	1.5	*<0.16*	*<0.09*	*<0.44*	*<0.24*	11.5	*<0.09*	*<0.12*	*<0.14*
36	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.01	*<0.24*	2.9	1.4	*<0.16*	*<0.09*	*<0.44*	*<0.24*	10.0	*<0.09*	*<0.12*	*<0.14*
37	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	2.2	*<0.24*	3.9	2.9	*<0.16*	*<0.09*	*<0.44*	*<0.24*	7.1	*<0.09*	*<0.12*	*<0.14*
38	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
39	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	*<0.08*	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	*<0.07*	*<0.09*	*<0.12*	*<0.14*
40	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	1.4	*<0.24*	4.00	2.4	*<0.16*	*<0.09*	*<0.44*	*<0.24*	6.1	*<0.09*	*<0.12*	*<0.14*
42	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.23	*<0.24*	1.1	0.30	*<0.16*	*<0.09*	*<0.44*	*<0.24*	2.7	*<0.09*	*<0.12*	*<0.14*
43	*<0.21*	*<0.18*	*<0.11*	*<0.13*	*<0.06*	0.11	*<0.24*	*<0.21*	*<0.08*	*<0.16*	*<0.09*	*<0.44*	*<0.24*	0.10	*<0.09*	*<0.12*	*<0.14*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Data were not available for FIN 2, 5, 8, 15, 32, 35, and 41.

Table B-10. Element Concentrations in the Toxicity Characteristic Leaching Procedure (TCLP) Extracts from the Spent Foundry Molding Sands Collected June 2005 (Concentrations in mg L^{-1})

FIN[a]	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.14	*<0.14*	*<0.05*	*<0.02*	*<0.41*
2	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.94	*<0.05*	*<0.02*	*<0.41*
3	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
4	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
5	*<0.04*	0.011	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
6	*<0.04*	*<0.001*	1.1	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
7	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
8	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
9	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
10	*<0.04*	0.001	*<0.86*	*0.04*	0.02	*<0.46*	0.22	0.18	*<0.05*	*<0.02*	*<0.41*
11	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
12	*<0.04*	2.4	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
13	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
14	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
15	*<0.04*	0.005	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
16	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
17	*<0.04*	0.006	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
18	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
19	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.19	*<0.05*	*<0.02*	*<0.41*
20	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
21	*<0.04*	*<0.001*	*<0.86*	*0.02*	*<0.01*	*<0.46*	0.10	*<0.14*	*<0.05*	*<0.02*	*<0.41*
22	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.25	*<0.05*	*<0.02*	2.5
23	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
24	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
25	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
26	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
27	*<0.04*	*<0.001*	*<0.86*	0.03	0.02	*<0.46*	0.20	0.44	*<0.05*	*<0.02*	*<0.41*
28	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
29	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.19	0.61	*<0.05*	*<0.02*	*<0.41*
30	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
31	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.20	*<0.05*	*<0.02*	*<0.41*
32	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
33	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	2.1	*<0.14*	*<0.05*	*<0.02*	1.7
34	*<0.04*	0.001	*<0.86*	*<0.01*	0.06	*<0.46*	3.5	*<0.14*	0.10	*<0.02*	37.6
35	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
36	*<0.04*	0.008	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
37	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
38	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.33	*<0.05*	*<0.02*	*<0.41*
39	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.11	1.5	*<0.05*	*<0.02*	*<0.41*
40	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
41	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
42	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
43	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-11. Element Concentrations in the TCLP Extracts from the Spent Foundry Molding Sands Collected September 2005 (Concentrations in mg L^{-1})

FIN[a]	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.14	0.15	*<0.05*	*<0.02*	*<0.41*
3	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
4	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
5	*<0.04*	0.018	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
6	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
7	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
8	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	0.58
9	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
10	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
11	*<0.04*	0.019	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
12	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
13	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
14	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
16	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
17	*<0.04*	0.013	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
18	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
19	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.14	0.25	*<0.05*	*<0.02*	*<0.41*
20	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
21	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
22	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	0.11	0.17	*<0.05*	*<0.02*	1.32
23	*<0.04*	0.007	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
24	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
25	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
26	*<0.04*	0.008	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
27	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
28	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
29	*<0.04*	0.005	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
30	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
31	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.30	*<0.05*	*<0.02*	*<0.41*
33	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	43.9	0.14	*<0.05*	*<0.02*	6.5
34	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*0.65*	0.26	*<0.05*	*<0.02*	40.3
35	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
36	*<0.04*	0.013	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
37	*<0.04*	0.005	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
40	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
41	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
42	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
43	*<0.04*	0.006	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Data were not available for FIN 2, 15, 32, 38, and 39.

Table B-12. Element Concentrations in the TCLP Extracts from the Spent Foundry Molding Sands Collected July 2006
(Concentrations in mg L^{-1})

FIN[a]	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
3	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
4	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
6	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
7	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
8	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	0.68
9	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
10	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
11	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
12	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
13	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
14	*<0.04*	0.006	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
16	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
17	*<0.04*	0.007	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
18	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
19	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
20	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
21	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
22	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.27	*<0.05*	*<0.02*	4.49
23	*<0.04*	0.005	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
24	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
25	*<0.04*	0.003	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
26	*<0.04*	0.012	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
27	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
28	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
29	*<0.04*	0.002	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	0.20	*<0.05*	*<0.02*	*<0.41*
30	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
31	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
33	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
34	*<0.04*	0.001	*<0.86*	*<0.02*	0.06	*<0.46*	5.4	0.23	1.1	*<0.02*	42.5
36	*<0.04*	0.017	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
37	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
38	*<0.04*	0.007	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	1.71	*<0.05*	*<0.02*	0.71
39	*<0.04*	0.004	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
40	*<0.04*	0.001	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
42	*<0.04*	*<0.001*	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*
43	*<0.04*	0.005	*<0.86*	*<0.01*	*<0.01*	*<0.46*	*<0.10*	*<0.14*	*<0.05*	*<0.02*	*<0.41*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Data were not available for FIN 2, 5, 15, 32, 35, and 41.

Table B-13. Element-Specific Concentrations in the Synthetic Precipitation Leaching Procedure (SPLP) Extracts from the Spent Foundry Molding Sands Collected June 2005 (Concentrations in mg L^{-1})

FIN[a]	**Ag**	**As**	**Ba**	**Be**	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Sb**	**Zn**
1	*<0.08*	3.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
2	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
3	*<0.08*	9.8E-02	2.9E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
4	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
5	*<0.08*	1.7E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
6	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
7	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
8	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
9	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
10	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
11	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
12	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
13	*<0.08*	4.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
14	*<0.08*	9.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
15	*<0.08*	1.0E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
16	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
17	*<0.08*	7.0E-03	2.5E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
18	*<0.08*	7.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
19	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
20	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
21	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
22	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
23	*<0.08*	8.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
24	*<0.08*	3.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
25	*<0.08*	4.0E-03	4.3E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
26	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
27	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
28	*<0.08*	5.0E-03	3.0E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
29	*<0.08*	3.0E-03	2.7E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
30	*<0.08*	5.0E-03	2.9E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
31	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
32	*<0.08*	4.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
33	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	3.4E-01
34	*<0.08*	3.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	5.5E-01	*<0.05*	*<0.08*	*<0.04*	3.1E+00
35	*<0.08*	7.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
36	*<0.08*	9.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
37	*<0.08*	9.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
38	*<0.08*	*<0.001*	2.7E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
39	*<0.08*	4.0E-03	6.1E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	2.4E-01	*<0.08*	*<0.04*	*<0.18*
40	*<0.08*	4.0E-03	2.9E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
41	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
42	*<0.08*	6.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
43	*<0.08*	7.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-14. Element-Specific Concentrations in the SPLP Extracts from the Spent Foundry Molding Sands Collected September 2005 (Concentrations in mg L^{-1})

FIN[a]	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
3	*<0.08*	1.7E-02	2.5E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
4	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
5	*<0.08*	2.4E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
6	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
7	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	8.9E-02	*<0.08*	*<0.04*	*<0.18*
8	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
9	*<0.08*	4.9E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
10	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
11	*<0.08*	1.9E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
12	*<0.08*	3.5E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
13	*<0.08*	3.9E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
14	*<0.08*	1.6E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
16	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
17	*<0.08*	1.5E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
18	*<0.08*	9.6E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
19	*<0.08*	1.8E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
20	*<0.08*	2.7E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
21	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
22	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
23	*<0.08*	2.3E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
24	*<0.08*	9.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
25	*<0.08*	1.0E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
26	*<0.08*	1.9E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
27	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	6.4E-02	*<0.08*	*<0.04*	*<0.18*
28	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
29	*<0.08*	5.2E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
30	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
31	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
33	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
34	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	7.5E-01	*<0.05*	*<0.08*	*<0.04*	1.6E+00
35	*<0.08*	1.5E-02	2.6E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
36	*<0.08*	1.4E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
37	*<0.08*	1.5E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
40	*<0.08*	6.8E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
41	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
42	*<0.08*	1.2E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
43	*<0.08*	1.3E-02	3.7E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	5.0E-02	*<0.08*	*<0.04*	*<0.18*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 15, 32, 38, and 39.

Table B-15. Element Concentrations in the SPLP Extracts from the Spent Foundry Molding Sands Collected July 2006 (Concentrations in mg L^{-1})

FIN[a]	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.08*	5.3E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
3	*<0.08*	9.5E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
4	*<0.08*	1.7E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
6	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
7	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
8	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
9	*<0.08*	2.2E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
10	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
11	*<0.08*	4.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
12	*<0.08*	1.8E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
13	*<0.08*	4.3E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
14	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
16	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
17	*<0.08*	5.6E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
18	*<0.08*	7.4E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
19	*<0.08*	3.4E-03	3.2E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
20	*<0.08*	2.0E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
21	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
22	*<0.08*	2.5E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
23	*<0.08*	1.2E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
24	*<0.08*	6.7E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
25	*<0.08*	6.2E-03	3.9E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
26	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
27	*<0.08*	2.2E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
28	*<0.08*	3.8E-03	3.1E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
29	*<0.08*	2.8E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
30	*<0.08*	4.7E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
31	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
33	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
34	*<0.08*	1.6E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	1.7E+00	*<0.05*	2.8E-01	*<0.04*	4.0E+00
36	*<0.08*	1.7E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
37	*<0.08*	1.1E-02	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
38	*<0.08*	*<0.001*	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
39	*<0.08*	3.4E-03	6.3E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	7.0E-02	*<0.08*	*<0.04*	*<0.18*
40	*<0.08*	4.5E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
42	*<0.08*	4.7E-03	*<0.23*	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*
43	*<0.08*	7.5E-03	3.9E-01	*<0.02*	*<0.01*	*<0.01*	*<0.21*	*<0.05*	*<0.08*	*<0.04*	*<0.18*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 5, 15, 32, 35, and 41.

Table B-16. Element Concentrations in Water Extracts from the Spent Foundry Molding Sands Collected June 2005 (Concentrations in mg L^{-1})

FIN[a]	**Ag**	**As**	**Ba**	**Be**	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Sb**	**Zn**
1	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
2	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
3	*<0.05*	8.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
4	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
5	*<0.05*	1.8E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
6	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
7	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
8	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
9	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
10	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
11	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
12	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
13	*<0.05*	5.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
14	*<0.05*	1.0E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
15	*<0.05*	1.0E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
16	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
17	*<0.05*	7.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
18	*<0.05*	5.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
19	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
20	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
21	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
22	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
23	*<0.05*	9.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
24	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
25	*<0.05*	6.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
26	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
27	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
28	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
29	*<0.05*	3.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
30	*<0.05*	5.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
31	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*0.05*	*<0.11*	*<0.04*	*<0.22*
32	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
33	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	4.6E-02	*<0.11*	*<0.04*	*<0.22*
34	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
35	*<0.05*	8.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	1.1E+00	*<0.05*	*<0.11*	*<0.04*	1.0E+00
36	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	3.0E-01	*<0.05*	*<0.11*	*<0.04*	1.3E+00
37	*<0.05*	9.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
38	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
39	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
40	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
41	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
42	*<0.05*	6.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
43	*<0.05*	6.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Table B-17. Element Concentrations in the Water Extracts from the Spent Foundry Molding Sands Collected September 2005 (Concentrations in mg L^{-1})

FIN[a]	*Ag*	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
3	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
4	*<0.05*	1.8E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
5	*<0.05*	2.3E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
6	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
7	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
8	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
9	*<0.05*	5.1E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
10	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
11	*<0.05*	1.9E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
12	*<0.05*	4.1E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
13	*<0.05*	6.5E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
14	*<0.05*	1.7E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
16	*<0.05*	1.5E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
17	*<0.05*	1.6E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
18	*<0.05*	7.4E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
19	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
20	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
21	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
22	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
23	*<0.05*	2.4E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
24	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
25	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
26	*<0.05*	1.9E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
27	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
28	*<0.05*	1.3E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
29	*<0.05*	4.7E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
30	*<0.05*	1.2E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
31	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
33	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	7.0E-02	*<0.05*	*<0.11*	*<0.04*	*<0.22*
34	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	2.2E-01	*<0.05*	*<0.11*	*<0.04*	*<0.22*
35	*<0.05*	1.8E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
36	*<0.05*	1.7E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
37	*<0.05*	1.8E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
40	*<0.05*	9.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
41	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
42	*<0.05*	1.3E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
43	*<0.05*	1.3E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 15, 32, 38, and 39.

Table B-18. Element Concentrations in the Water Extracts from the Spent Foundry Molding Sands Collected July 2006 (Concentrations in mg L^{-1})

FINa	Ag	As	Ba	Be	Cd	Cr	Cu	Ni	Pb	Sb	Zn
1	*<0.05*	6.6E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
3	*<0.05*	7.8E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
4	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
6	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
7	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
8	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
9	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
10	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
11	*<0.05*	4.7E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
12	*<0.05*	2.6E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
13	*<0.05*	4.8E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
14	*<0.05*	1.3E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
16	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
17	*<0.05*	7.3E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
18	*<0.05*	8.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
19	*<0.05*	3.1E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
20	*<0.05*	2.5E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
21	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
22	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
23	*<0.05*	1.4E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
24	*<0.05*	6.7E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
25	*<0.05*	5.8E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
26	*<0.05*	1.2E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
27	*<0.05*	4.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
28	*<0.05*	3.0E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
29	*<0.05*	2.8E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
30	*<0.05*	4.6E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
31	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
33	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
34	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	1.6E+00
36	*<0.05*	1.7E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
37	*<0.05*	1.1E-02	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
38	*<0.05*	*<0.003*	*<0.24*	*<0.01*	*<0.01*	*<0.02*	8.0E-02	*<0.05*	*<0.11*	*<0.04*	*<0.22*
39	*<0.05*	2.9E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
40	*<0.05*	5.1E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
42	*<0.05*	5.3E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*
43	*<0.05*	6.7E-03	*<0.24*	*<0.01*	*<0.01*	*<0.02*	*<0.07*	*<0.05*	*<0.11*	*<0.04*	*<0.22*

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

No data were available for FIN 2, 5, 15, 32, 35, and 41.

Table B-19. Summary of Total Elemental Content of 43 Spent Foundry Molding Sands Using a Microwave-Assisted Aqua Regia Digest (U.S. EPA, 3051a)

FIN[a]	Al	As	B	Ba	Be	Ca	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Mo	Na	Ni	P	Pb	S	Sb	Se	Tl	V	Zn
Units	g kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	g kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	g kg^{-1}	mg kg^{-1}	g kg^{-1}	mg kg^{-1}	mg kg^{-1}	g kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	g kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
LOQ	0.001	0.1	20.0	10.0	0.1	0.005	0.04	0.5	0.5	0.5	0.005	50.0	0.002	0.5	1.0	0.02	0.5	5.0	1.0	0.05	0.04	0.4	0.04	1.0	10.0
1	11.7	3.72	*10*	*5*	0.383	3.05	0.16	1.74	17.4	82.6	12.0	610	3.20	237	1.85	1.39	6.60	78.6	15.3	1.64	0.353	*0.2*	0.083	6.60	63.7
2	3.44	0.395	*10*	*5*	*0.05*	1.09	0.05	92.5	49.3	7.04	54.7	321	287	639	*0.5*	0.305	2560	17.6	3.03	0.152	0.094	*0.2*	*0.02*	2.10	46.3
3	8.09	1.13	*10*	*5*	0.148	1.22	0.05	*0.25*	*0.25*	*0.25*	2.90	215	3.02	52.4	*0.5*	1.38	3.47	54.6	4.46	0.588	0.277	*0.2*	0.045	1.21	5
4	4.95	0.633	*10*	*5*	*0.05*	1.20	0.051	0.806	5.52	8.42	3.08	618	1.03	34.6	*0.5*	0.280	3.46	81.2	2.22	0.302	0.12	*0.2*	*0.02*	1.20	22.8
5	7.57	2.14	20.2	*5*	0.157	0.975	0.08	1.11	15.7	13.0	12.8	248	1.16	288	1.94	1.82	5.64	34.3	4.87	0.352	0.08	*0.2*	0.051	2.45	24.5
6	1.70	0.498	*10*	120	*0.05*	1.32	0.103	42.1	134	23.2	27.0	203	124	570	2.49	0.320	1160	36.4	2.55	0.248	0.11	*0.2*	*0.02*	3.11	26.0
7	0.219	0.126	*10*	*5*	*0.05*	0.285	*0.02*	*0.25*	19.1	3.26	2.88	*25*	0.103	14.1	*0.5*	*0.01*	11.7	11.1	*0.50*	1.31	0.15	*0.2*	*0.02*	*0.5*	5
8	3.00	1.22	*10*	*5*	0.101	3.23	0.199	1.25	16.4	32.9	9.68	268	0.963	107	1.98	0.225	6.39	49.4	5.06	0.234	0.21	*0.2*	*0.02*	4.02	40.2
9	1.59	0.363	*10*	*5*	*0.05*	1.61	*0.02*	*0.25*	1.98	2.94	1.58	204	0.713	20.5	*0.5*	0.165	1.11	36.6	1.41	0.139	*0.02*	*0.2*	*0.02*	2.14	12.1
10	1.51	0.972	*10*	*5*	*0.05*	1.17	*0.02*	1.82	25.1	46.0	10.4	253	0.521	79.4	2.90	0.140	8.73	44.9	1.63	0.137	0.31	*0.2*	*0.02*	3.13	5
11	5.78	2.55	*10*	*5*	0.272	1.72	0.085	1.01	3.27	4.69	2.91	544	1.14	65.4	*0.5*	1.46	2.90	55.9	4.38	0.813	0.06	*0.2*	0.063	2.13	5
12	4.06	0.241	*10*	*5*	*0.05*	0.503	*0.02*	*0.25*	4.70	*0.25*	1.28	153	0.192	24.1	*0.5*	*0.01*	1.32	36.0	1.28	*0.025*	0.17	*0.2*	*0.02*	6.25	5
13	6.15	2.11	*10*	*5*	0.242	2.09	0.063	0.952	2.68	3.57	3.49	647	1.37	37.4	0.604	0.820	3.85	56.0	3.04	0.616	0.16	*0.2*	0.040	3.09	5
14	5.11	1.05	59.4	*5*	0.096	3.10	0.051	*0.25*	2.98	2.60	3.62	376	1.95	38.5	*0.5*	0.560	2.62	46.0	3.26	0.145	0.17	*0.2*	*0.02*	4.98	5
15	7.87	3.19	*10*	*5*	0.252	2.59	0.05	1.29	4.78	3.13	4.37	547	1.79	67.1	*0.5*	1.13	3.30	62.0	5.02	0.826	0.09	*0.2*	0.082	4.95	5
16	4.23	0.921	*10*	*5*	0.122	1.57	*0.02*	0.856	8.64	8.22	4.51	343	1.27	54.5	*0.5*	0.305	2.82	57.5	2.22	0.528	0.10	*0.2*	*0.02*	3.24	14.4
17	7.33	3.01	*10*	*5*	0.264	1.83	0.062	1.14	4.21	4.03	4.87	374	1.72	114	1.02	1.37	4.09	58.5	5.92	0.957	0.14	*0.2*	0.070	2.88	5
18	4.82	1.62	*10*	*5*	0.114	1.59	*0.02*	*0.25*	5.26	5.99	4.26	419	1.16	30.0	*0.5*	1.16	3.43	46.9	2.35	0.674	0.93	*0.2*	0.061	3.39	72.1
19	6.02	3.85	*10*	*5*	0.207	2.78	0.112	5.88	115	88.2	55.7	292	1.50	482	22.9	0.975	42.9	71.4	7.04	0.564	1.23	*0.2*	0.043	11.3	5
20	11.2	0.993	*10*	*5*	0.110	2.36	0.099	0.958	4.93	12.6	3.49	370	1.83	109	*0.5*	1.22	5.92	64.9	2.20	0.557	*0.02*	*0.2*	*0.02*	4.42	34.6
21	0.705	1.26	*10*	*5*	0.05	0.393	*0.02*	*0.25*	14.8	15.1	13.4	*25*	0.129	90.6	3.33	0.160	11.5	18.3	1.90	0.190	0.24	*0.2*	*0.02*	2.47	5
22	6.24	7.79	*10*	*5*	0.227	3.10	0.36	5.99	95.1	137	57.1	531	1.60	707	21.8	0.983	49.4	82.5	22.9	1.18	1.04	*0.2*	0.089	9.03	245
23	6.66	1.24	*10*	*5*	0.147	2.70	0.051	0.65	3.59	2.11	3.64	403	1.65	40.2	*0.5*	1.39	2.29	47.4	3.28	0.713	0.150	*0.2*	0.051	2.69	5
24	5.73	2.14	*10*	*5*	0.151	1.27	0.043	1.15	7.85	12.5	6.70	396	1.06	93.7	1.41	0.930	4.24	73.4	7.07	0.430	1.71	*0.2*	*0.02*	3.92	14.3
25	3.57	0.585	*10*	*5*	0.369	2.65	0.055	*0.25*	2.93	3.16	3.16	224	1.11	33.7	*0.5*	1.71	2.13	49.5	4.14	0.591	0.72	*0.2*	*0.02*	1.84	5
26	7.19	2.54	*10*	*5*	0.186	2.79	0.046	0.794	3.79	4.76	3.96	416	1.62	34.6	2.51	1.16	2.68	47.1	3.74	0.898	0.14	*0.2*	0.049	2.98	5
27	10.5	6.44	*10*	*5*	0.328	2.57	0.19	6.62	87.1	107	64.4	1300	2.29	670	19.7	1.11	117	85.9	8.63	1.13	0.65	0.438	0.090	9.90	30.2
28	6.33	0.899	*10*	*5*	0.171	2.51	*0.02*	0.633	5.39	2.11	4.25	251	1.55	51.1	*0.5*	1.26	2.84	50.9	4.00	0.850	0.25	*0.2*	0.056	2.48	5
29	0.572	0.335	*10*	*5*	0.05	0.405	0.061	2.74	109	46.3	5.49	1780	0.236	61.2	10.6	0.408	81.5	32.7	1.10	0.119	0.11	*0.2*	*0.02*	2.73	44.5
30	7.31	0.770	*10*	*5*	0.205	2.92	0.067	0.880	2.94	14.5	4.22	453	1.91	40.4	*0.5*	1.93	2.51	59.3	3.91	0.802	0.17	*0.2*	0.041	3.80	5
31	1.57	0.767	*10*	*5*	*0.05*	0.993	0.067	1.31	16.9	23.9	7.66	114	0.545	84.2	3.10	0.347	23.1	17.0	2.41	2.04	0.16	*0.2*	*0.02*	1.32	5
32	2.44	0.767	*10*	*5*	0.141	2.12	*0.02*	*0.25*	2.96	2.92	3.61	189	1.31	21.8	*0.5*	1.46	2.15	41.0	2.84	0.660	0.17	*0.2*	0.047	1.47	5
33	0.650	0.223	*10*	*5*	*0.05*	0.075	0.103	*0.25*	1.36	85.2	0.536	*25*	0.060	14.0	*0.5*	*0.01*	1.02	20.2	1.79	*0.025*	0.10	1.15	*0.02*	1.27	34.7
34	5.14	2.09	*10*	*5*	0.127	1.69	3.79	*0.25*	4.17	3805	2.94	445	1.39	18.7	*0.5*	0.800	15.3	176	20.8	0.545	0.74	59.5	0.065	2.88	2474
35	7.89	2.25	*10*	*5*	0.226	4.09	0.066	1.02	2.60	4.57	5.98	437	1.96	46.5	*0.5*	1.27	2.53	66.6	4.64	0.641	0.29	*0.2*	0.062	2.96	5
36	7.98	3.25	*10*	*5*	0.282	2.05	0.119	0.981	3.50	6.46	5.47	337	1.51	58.1	*0.5*	1.85	2.78	58.8	5.83	0.940	0.12	*0.2*	0.096	2.59	5
37	6.28	2.56	*10*	*5*	0.599	1.88	0.078	1.44	3.83	6.22	4.87	242	1.32	119	0.68	1.02	3.04	53.8	4.30	0.684	0.24	*0.2*	0.049	3.91	5
38	1.11	0.164	*10*	*5*	*0.05*	0.370	*0.02*	*0.25*	8.87	21.0	1.29	328	0.305	28.4	*0.5*	0.630	36.0	17.8	1.40	*0.025*	0.07	*0.2*	*0.02*	*0.5*	5
39	5.65	0.578	*10*	141	0.110	1.89	0.074	1.07	34.0	72.4	5.73	289	1.47	161	6.64	1.31	102	96.6	4.88	0.327	0.26	*0.2*	*0.02*	2.48	17.5
40	4.57	0.256	*10*	*5*	0.177	2.07	*0.02*	0.524	2.59	2.15	2.33	266	1.06	20.9	*0.5*	0.837	1.61	42.5	3.63	0.646	0.25	0.06	0.056	2.44	11.4
41	0.193	0.256	*10*	*5*	*0.05*	0.094	*0.02*	*0.25*	0.97	3.46	3.13	*25*	0.050	5.56	*0.5*	*0.01*	2.18	5.41	*0.5*	*0.025*	0.08	*0.2*	*0.02*	1.18	5
42	2.72	0.664	*10*	*5*	0.153	0.835	*0.02*	0.781	2.50	2.84	3.00	299	0.629	27.0	*0.5*	0.605	2.06	44.4	3.13	0.320	0.26	*0.2*	*0.02*	2.90	5
43	5.46	0.771	*10*	17.7	0.171	3.13	0.055	0.808	18.2	12.8	5.70	316	1.28	110	2.11	1.37	19.6	58.9	4.30	0.397	0.16	*0.2*	*0.02*	2.62	13.2

[a] FIN = foundry identification number; see Table 2-2 for details.

Entries in italics were below the limit of quantification (LOQ) and were calculated as 0.5 times the LOQ.

Table B-20. Total Metal Concentrations in Spinach Leaves Grown on Spent Foundry Sand Blends
(Concentrations in mg kg^{-1})

Element	Control			IGS-1			IGS-2			AGS-1			AGS-2			NBS-1			NBS-2			Sufficiency Range (Jones et al., 1991)
Al	10.1	±	2.3*	50.3	±	12.3a	65.4	±	28.7a	34.4	±	18.7	25.5	±	7.6	9.94	±	2.57	14.9	±	1.2	
B	79.9	±	33.7	51.7	±	6.2	47.4	±	6.7	58.1	±	39.4	41.3	±	3.1	74.2	±	13.3	57.6	±	8.8	25–60
Ba	*<0.12*			1.35	±	0.90	1.80	±	1.23	2.20	±	1.62a	3.55	±	0.26a	0.79	±	0.53	1.52	±	0.17	
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	0.24	±	0.16	0.20	±	0.15	0.47	±	0.27	1.00	±	0.81	0.39	±	0.16	0.70	±	0.26	0.30	±	0.22	
Co	0.72	±	0.10	0.40	±	0.17	0.72	±	0.24	*<0.01*			0.67	±	0.16	1.38	±	0.32	0.96	±	0.16	
Cr	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cu	4.74	±	0.50	3.07	±	1.05a	3.62	±	0.70	6.15	±	0.57	10.9	±	1.3a	6.87	±	1.02	3.85	±	0.34a	5–25
Fe	68.1	±	8.8	41.9	±	12.7	58.2	±	10.0	73.9	±	17.0	59.5	±	4.7	110	±	20a	110	±	18a	60–200
Mg	8,511	±	3,242	3,475	±	1,466a	5,515	±	1,198	3,339	±	1,315a	6,009	±	548	10,182	±	835a	12,758	±	1,988	6,000–10,000
Mn	25.0	±	4.1	28.9	±	7.2	69.6	±	16.5a	262	±	43a	119	±	12a	54.8	±	13.0	32.0	±	3.5	30–250
Mo	0.97	±	0.14	0.65	±	0.45	0.55	±	0.65	0.70	±	0.53	0.98	±	0.11	0.81	±	0.55	0.89	±	0.07	
Ni	2.06	±	0.13	1.23	±	0.50	1.19	±	0.36	*<0.05*			1.59	±	0.25	9.91	±	6.47	3.78	±	0.89a	
Pb	*<0.28*			3.96	±	1.22a	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	27.0	±	1.4	29.9	±	2.9	38.1	±	3.7	84.0	±	12.3a	50.2	±	4.5a	76.3	±	15.8a	22.4	±	1.8	25–100

* Average value of four replicates ± standard deviation.

"a" values indicate a significant difference than the control ($P<0.05$).

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Note: AGS = aluminum green sand, IGS= iron green sand, NBS = steel no-bake sand

Table B-21. Total Element-Specific Concentrations in Radish Globes and Leaves Grown on Spent Foundry Sand Blends (Concentrations in mg kg^{-1})

Element	Control			IGS-1			IGS-2			AGS-1			AGS-2			NBS-1			NBS-2			Sufficiency Range (Jones et al., 1991)
Radish Globes																						
Al	52.7	±	12.5*	319	±	275	258	±	89	2,393	±	1,157a	777	±	499	55.6	±	111	270	±	55	
B	24.9	±	2.9	27.0	±	6.2	26.0	±	1.6	30.2	±	2.5	45.3	±	27.6	32.7	±	9.9	34.1	±	11.3	
Ba	*<0.12*			3.78	±	2.84	4.19	±	1.46	5.46	±	6.30	*<0.12*			*<0.12*			*<0.12*			
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Co	0.58	±	0.08	0.08	±	0.16	0.28	±	0.12	*<0.01*			0.11	±	0.21	0.78	±	0.38	1.13	±	0.69	
Cr	0.38	±	0.12	0.31	±	0.48	*<0.01*			0.70	±	0.81	0.22	±	0.44	*<0.01*			*<0.01*			
Cu	5.48	±	1.50	2.43	±	1.75	2.26	±	0.60	7.05	±	8.88	13.49	±	8.28	4.08	±	1.22	1.39	±	1.72	
Fe	139	±	23	256	±	296	132	±	80	652	±	232a	438	±	197	165	±	93	178	±	61	
Mg	2,112	±	471	1,162	±	449	1,313	±	192	2,497	±	437	1847	±	486	3704	±	2,332	3,614	±	1,619	
Mn	18.4	±	1.8	11.5	±	4.3	23.0	±	11.2	104	±	37a	22.7	±	5.2	14.0	±	4.2	17.1	±	3.9	
Mo	*<0.07*			1.27	±	0.52a	1.33	±	0.38a	*<0.07*			*<0.07*			*<0.07*			*<0.07*			
Ni	1.46	±	0.16	*<0.05*			0.17	±	0.34	*<0.05*			*<0.05*			5.13	±	2.37a	4.81	±	3.91	
Pb	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	17.8	±	3.2	16.3	±	3.2	17.4	±	6.9	18.7	±	12.6	19.9	±	3.4	15.9	±	2.6	18.8	±	2.7	
Radish Leaves																						
Al	9.9	±	19.8	58.7	±	44.0	75.9	±	88.1	288	±	214a	118	±	119	*<2.00*			47.2	±	94.5	
B	98.8	±	12.4	49.2	±	5.5	66.0	±	7.5	91.8	±	8.5	81.2	±	64.8	100.8	±	10.5	88.5	±	9.2	25–125
Ba	*<0.12*			8.12	±	0.61a	14.0	±	1.9a	*<0.12*			6.20	±	0.56a	3.10	±	2.38a	8.14	±	1.55a	
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	0.11	±	0.22	0.21	±	0.42	*<0.01*			*<0.01*			*<0.01*			*<0.01*			0.19	±	0.38	
Co	1.17	±	0.42	0.03	±	0.07	0.26	±	0.19	*<0.01*			*<0.01*			2.29	±	1.59	2.39	±	1.20	
Cr	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cu	7.10	±	3.30	2.75	±	0.94a	3.22	±	0.53a	0.75	±	1.49a	4.86	±	0.82	4.75	±	1.22	3.75	±	1.16a	5–25
Fe	131	±	58	106	±	27	109	±	36	214	±	65	134	±	65	137	±	41	245	±	90	50–200
Mg	8,907	±	1,060	3,315	±	299a	4,842	±	1120a	4,353	±	407a	3,774	±	1,254a	12,357	±	3,134a	10,212	±	2,341	5,000–45,000
Mn	88.2	±	15.2	60.6	±	11.3	143	±	7a	192	±	33a	128	±	18a	125	±	13a	86.8	±	3.8	50–250
Mo	5.00	±	1.05	6.28	±	1.77	7.84	±	1.48a	10.7	±	2.0a	6.67	±	0.82	2.87	±	0.12	4.26	±	0.41	
Ni	3.93	±	1.27	*<0.05*			0.28	±	0.55	*<0.05*			*<0.05*			19.7	±	11.7a	7.07	±	2.98	
Pb	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	29.2	±	4.3	24.1	±	2.7	28.8	±	5.2	31.6	±	4.5	25.3	±	3.1	37.5	±	8.3	30.0	±	3.2	25–100

* Average value of four replicates ± standard deviation.

“a” values indicate a significant difference than the control (*P*<0.05).

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Note: AGS = aluminum green sand, IGS= iron green sand, NBS = steel no-bake sand

Table B-22. Total Metal Concentrations in Perennial Ryegrass Grown on Spent Foundry Sand Blends (Concentrations in mg kg^{-1})

Element	Control			IGS-1			IGS-2			AGS-1			AGS-2			NBS-1			NBS-2			Sufficiency Range (Jones et al., 1991)
Harvest 1																						
Al	*<2.00*			17.8	±	35.5	*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			52–922 **
B	113	±	34.3 *	132	±	136	57.6	±	7.7	296	±	60.2a	42.7	±	6.2	120	±	28	79.8	±	7.1	5–17
Ba	*<0.12*			12.7	±	2.7a	7.85	±	1.81a	1.49	±	2.97	6.53	±	1.33a	1.78	±	2.07	8.59	±	0.43a	
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	0.09	±	0.11	0.19	±	0.30	0.14	±	0.19	0.09	±	0.19	0.25	±	0.16	0.19	±	0.22	0.44	±	0.28	
Co	0.92	±	0.05	0.41	±	0.09a	0.35	±	0.07a	*<0.01*			0.33	±	0.05a	0.58	±	0.06a	0.34	±	0.07a	
Cr	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cu	5.61	±	1.29	7.97	±	1.55a	9.53	±	2.08a	13.8	±	1.0a	11.1	±	0.9a	7.96	±	1.10a	5.01	±	0.48	6–38
Fe	49.4	±	1.0	64.3	±	10.8	63.1	±	12.2	87.2	±	11.7	61.9	±	10.2	59.0	±	7.0	48.3	±	2.6	97–934
Mg	3,806	±	155	2,576	±	939	3,231	±	884	1,872	±	2,167	2,736	±	753	4,978	±	1569	4,380	±	95	1,600–3,200
Mn	100	±	4	121	±	12	200	±	23a	192	±	25a	191	±	20a	123	±	11	135	±	4a	30–73
Mo	1.51	±	0.24	3.54	±	0.54a	5.01	±	0.49a	9.69	±	1.27a	4.27	±	0.56a	1.25	±	0.84a	2.38	±	0.15	0.5–1.0
Ni	1.66	±	0.30	2.49	±	0.31a	2.30	±	0.58	*<0.05*			2.19	±	0.16	3.35	±	0.77a	2.35	±	0.30	
Pb	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	34.5	±	1.7	30.8	±	3.6	25.8	±	3.6a	46.1	±	4.0a	26.9	±	2.7a	45.6	±	3.9a	23.5	±	0.5a	14–64
Harvest 2																						
Al	*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			52–922 **
B	68.9	±	11.6	46.3	±	5.8a	37.2	±	9.5a	184	±	13.0a	28.2	±	12.6a	63.3	±	5.3	39.8	±	3.4a	5–17
Ba	*<0.12*			10.9	±	2.9a	7.71	±	2.55a	4.14	±	0.58a	13.2	±	3.7a	*<0.12*			2.98	±	0.24	
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	0.11	±	0.04	0.16	±	0.12	0.05	±	0.06	0.04	±	0.09	0.06	±	0.08	0.13	±	0.10	0.12	±	0.11	
Co	0.61	±	0.08	0.16	±	0.05a	0.14	±	0.11a	*<0.01*			0.14	±	0.10a	0.45	±	0.02a	0.07	±	0.08a	
Cr	*<0.01*			*<0.01*			*<0.01*			0.1	±	0.1	*<0.01*			*<0.01*			*<0.01*			
Cu	11.2	±	1.0	7.70	±	1.79	9.22	±	2.33	11.4	±	3.4	10.0	±	4.5	9.06	±	0.31	6.75	±	0.29	6–38
Fe	71.4	±	8.9	59.7	±	6.8	56.5	±	12.7	75.4	±	18.2	56.6	±	17.0	62.4	±	1.4	54.8	±	2.6	97–934
Mg	6,755	±	732	3,709	±	161a	4,188	±	585a	4,552	±	600a	4,321	±	682a	5,678	±	287a	5,542	±	231a	1.600–3.200
Mn	68.8	±	9.2	119	±	7	271	±	62a	326	±	9a	320	±	73a	75	±	5	119	±	9	30–73
Mo	1.76	±	0.26	4.84	±	1.25a	5.33	±	2.04a	6.79	±	1.70a	3.20	±	1.56	1.64	±	0.12	1.82	±	0.34	0.5–1.0
Ni	1.60	±	0.31	1.27	±	0.36	1.11	±	0.53	0.48	±	0.34a	1.03	±	0.40	4.82	±	0.94a	1.73	±	0.24	
Pb	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	23.2	±	3.4	33.7	±	6.2	22.9	±	5.3	48.3	±	12.1a	24.0	±	11.1	42.2	±	3.1a	22.1	±	1.9	14–64

Element	Control			IGS-1			IGS-2			AGS-1			AGS-2			NBS-1			NBS-2			Sufficiency Range (Jones et al., 1991)
													Harvest 3									
Al	*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			*<2.00*			52–922 **
B	43.1	±	1.9	43.3	±	6.9	44.2	±	5.3	96.4	±	11.3a	35.3	±	5.4	44.3	±	2.3	34.5	±	2.5	5–17
Ba	*<0.12*			17.2	±	2.8a	16.7	±	4.1a	3.34	±	0.50	22.5	±	4.7a	*<0.12*			*<0.12*			
Be	*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			*<0.01*			
Cd	0.06	±	0.07	0.09	±	0.07	0.09	±	0.09	0.25	±	0.16	0.10	±	0.09	0.04	±	0.05	0.10	±	0.07	
Co	0.23	±	0.03	0.04	±	0.07a	0.11	±	0.08a	*<0.01*			0.16	±	0.02a	0.24	±	0.04	*<0.01*			
Cr	*<0.01*			*<0.01*			*<0.01*			0.1	±	0.1	*<0.01*			*<0.01*			*<0.01*			
Cu	4.29	±	0.63	6.37	±	0.90a	7.75	±	1.06a	8.80	±	1.20a	9.15	±	0.58a	4.86	±	0.33	2.56	±	0.79a	6–38
Fe	57.7	±	4.4	51.8	±	4.0	59.3	±	7.8	61.7	±	6.3	56.5	±	5.5	45.3	±	1.6a	38.0	±	1.8a	97–934
Mg	5,092	±	511	4,373	±	387	4,698	±	522	3,719	±	359a	5,062	±	864	5,313	±	236	4,405	±	696	1,600–3,200
Mn	57.3	±	2.6	99.4	±	22.0a	221	±	23a	374	±	17a	280	±	14a	40.5	±	4.7	61.5	±	6.7	30–73
Mo	1.03	±	0.22	5.50	±	1.66	6.39	±	0.94a	5.97	±	1.0a	3.96	±	0.18a	0.75	±	0.52	0.25	±	0.51	0.5–1.0
Ni	0.75	±	0.06	1.04	±	0.15a	1.14	±	0.29a	*<0.05*			1.01	±	0.08a	3.71	±	0.23a	0.94	±	0.12	
Pb	*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			*<0.28*			
V	*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			*<0.09*			
Zn	13.8	±	0.6	34.5	±	5.5a	29.6	±	4.5a	48.9	±	4.2a	33.4	±	6.4a	31.8	±	2.4a	15.8	±	1.0	14–64

* Average value of four replicates ± standard deviation.

** Non-essential element

“a” values indicate a significant difference than the control ($P<0.05$).

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

Note: AGS = aluminum green sand, IGS= iron green sand, NBS = steel no-bake sand

Table B-23. Lettuce Tissue Elemental Content for 10 Spent Foundry Sands and a Silica Sand Control Plant Nutrient Tissue Adequacy Levels, Elemental Normal and Toxic Levels

			Spent Foundry Sands										Nutrient Adequate	Normal	Toxic
Element	**units**	**Control**	**1**	**3**	**4**	**10**	**16**	**20**	**24**	**25**	**28**	**40**		**Range**	
As	mg kg^{-1}	1.43	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1		1.0–1.7[a]	5–20[a]
B	mg kg^{-1}	142	75.9	57.1	101	57.1	82.9	94.6	78.7	56.3	65.4	75.3	7–75[c]		>100[b]
Ba	mg kg^{-1}	0.676	1.10	1.68	1.18	1.16	1.31	1.03	1.68	1.09	0.786	1.13			>500[a]
Ca	g kg^{-1}	2.09	4.53	2.65	4.66	3.64	4.73	3.69	3.26	3.21	3.83	3.85	5–30[d]		
Cd	mg kg^{-1}	<0.2	0.72	0.413	0.387	1.14	0.29	0.485	1.88	0.368	0.393	0.412		0.1–2.4[e]	5–30[a]
Co	mg kg^{-1}	2.82	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2		0.02–1.0[a]	15–50[a]
Cu	mg kg^{-1}	43.6	34.2	15.9	21.3	38.9	16.2	10.4	21.7	19.4	21.8	18.3	5–30[a]		20–100[a]
Fe	mg kg^{-1}	201	148	89.6	140	215	176	220	215	98.1	271	135	50–150[d]		
K	g kg^{-1}	24.5	40.9	34.2	24.2	30.6	41.0	38.6	36.1	37.7	38.7	41.9	14–35[d]		
Mg	g kg^{-1}	4.52	2.49	1.99	2.97	2.63	2.99	2.53	2.30	1.55	2.33	2.06	3–10[d]		
Mn	mg kg^{-1}	141	131	65.2	92.3	93.8	102	118	72.2	65.6	99.2	45.9	30–300[a]		>800[b]
Mo	mg kg^{-1}	<2	<2	3.94	<2	3.04	<2	7.18	<2	3.52	<2	<2	0.25–5.0[a]		10–50[a]
N	g kg^{-1}	49.8	31.7	27.5	21.3	22.5	31.9	50.8	33.0	26.4	33.5	37.3	17–50[d]		
Na	g kg^{-1}	0.661	17.4	28.9	6.33	7.80	9.85	29.7	21.2	28.0	20.8	19.2			
Ni	mg kg^{-1}	5.68	<2	<2	<2	10.3	2.92	<2	<2	<2	<2	<2		0.1–5.0[a]	10–100[a]
P	g kg^{-1}	9.70	4.23	4.83	3.84	3.37	5.20	3.84	5.82	4.10	4.05	4.75	2.0–5.0[d]		
S	g kg^{-1}	3.05	3.95	2.48	1.71	1.92	2.75	4.68	3.18	2.68	3.38	3.78	1.5–5.0[d]		
Zn	mg kg^{-1}	91.0	55.1	94.8	28.2	29.1	36.8	64.1	63.5	68.2	63.8	99.0	27–150[a]		>500[c]

Note: Other elements evaluated were below the detection limit, including Be, Tl, and V (<0.2 mg kg^{-1}) and Cr, Pb, Sb, and Se (<2 0.2 mg kg^{-1}).

[a] Kabata-Pendias, 2001

[b] Adriano, 2001

[c] Pais and Jones, 1997

[d] Johnson et al., 2000

[e] Bowen, 1979

Table B-24. Total Element-Specific Concentrations in *Eisenia fetida* After 28 Days in the Spent Foundry Sand Blends (Average Concentrations of Four Replicates in mg kg^{-1} Dry Biomass)

Sample	Control	AGS-1			AGS-2			IGS-1			IGS-2			NBS			BGS
Ratio (%)	**100**	**10**	**30**	**50**	**10**	**30**	**50**	**10**	**30**	**50**	**10**	**30**	**50**	**10**	**30**	**50**	**10**
Al	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*369.0*	*662.7*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*	*<256.3*
B	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*	*<34.6*
Ba	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*	*<11.0*
Be	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*	*<0.5*
Ca	3,310.6	2,206.0	2,938.0	2,631.1	2,839.1	2,847.8	2,621.5	2,292.0	2,100.9	1,445.6	2,779.1	2,632.0	2,312.2	2,170.6	1,585.2	2,561.3	2,529.4
Cd	1.3	0.7	0.7	1.1	1.6	1.5	1.3	1.2	1.0	0.7	0.9	1.2	0.8	1.5	0.8	1.0	1.4
Co	1.8	1.2	0.8	1.3	1.9	1.7	1.6	1.6	1.3	0.9	1.1	1.5	1.2	1.7	1.1	1.8	1.3
Cr	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*	*<2.1*
Cu	10.2	*<4.4*	*<4.4*	7.6	10.5	7.2	7.1	6.9	6.1	*<4.4*	*<4.4*	7.4	*<4.4*	8.6	6.5	7.0	104.1
Fe	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*	*<307.2*
Mg	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*	*<51.6*
Mn	20.9	19.8	22.2	20.8	24.3	18.6	23.8	15.6	13.3	17.4	16.3	15.8	16.3	17.3	13.8	17.5	6.9
Mo	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*	*<1.9*
Na	3,181.8	3,083.5	2,748.0	2,719.3	2,844.3	2,506.4	2,787.0	2,712.2	2,463.3	2,265.8	2,635.0	2,770.9	3,024.1	2,428.7	2,677.6	2,779.2	2,380.1
Ni	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*	*<1.8*
Pb	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*	*<49.7*
V	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*	*<4.1*
Zn	43.2	*<38.6*	*<38.6*	45.1	*<38.6*	41.1	43.8	*<38.6*	38.8	*<38.6*	40.5	*<38.6*	*<38.6*	53.4	42.0	45.1	75.9

Entries in italics with the less than symbol (<) identify samples that were below the quantitative detection limit shown for that element.

For brass green sand (BGS), the earthworm biomass was insufficient to run metal analyses on 30% and 50% BGS blends.

Note: ASG = aluminum green sand, IGS= iron green sand, NBS = steel no-bake sand

Table B-25. Particle-Size Distribution, U.S. Department of Agriculture Textural Class and Bulk Density for Spent Foundry Sand

FIN[a]	**% Sand (0.05–2 mm)**	**% Silt (2–50 µm)**	**% Clay (<2 µm)**	**Textural Class**	**Bulk Density (g cm^{-3})**
1	82.7	7.7	9.6	Loamy sand	1.60
2	79.3	9.4	11.3	Sandy loam	1.57
4	94.6	2.1	3.3	Sand	1.66
5	87.2	3.6	9.2	Loamy sand	1.61
6	98.3	1.7	0	Sand	1.66
7	99.9	0.1	0	Sand	1.66
8	94.8	2.3	2.9	Sand	1.66
9	98.6	0.8	0.6	Sand	1.66
10	99.8	0.2	0	Sand	1.66
11	82.9	7.0	10.1	Loamy sand	1.59
12	98.9	1.1	0	Sand	1.66
13	89.1	2	8.9	Loamy sand	1.62
14	91.7	1.1	7.2	Sand	1.66
15	85	5	10	Loamy sand	1.6
16	95.1	1	3.9	Sand	1.66
17	84.6	4.3	11.1	Loamy sand	1.58
18	94	0.5	5.5	Sand	1.66
19	90.5	4.1	5.4	Sand	1.66
20	91.6	0.9	7.5	Sand	1.66
21	99	0	1	Sand	1.66
22	89.9	5.7	4.4	Sand	1.66
23	89.6	1.8	8.6	Sand	1.63
24	92.1	3.7	4.2	Sand	1.66
25	88.1	4.6	7.3	Loamy sand	1.65
26	89.3	3.5	7.2	Sand	1.66
27	76.6	16.9	6.5	Loamy sand	1.64
28	87.7	3.9	8.4	Loamy sand	1.63
29	98.1	1.9	0	Sand	1.66
30	89.5	3.9	6.6	Sand	1.66
31	97	2.1	0.9	Sand	1.66
32	90.1	3.8	6.1	Sand	1.66
33	99.1	0.9	0	Sand	1.68
34	89	4.5	6.5	Sand	1.65
35	86.7	5.3	8	Loamy sand	1.63
36	82.9	7.5	9.6	Loamy sand	1.6
37	86.6	4.9	8.5	Loamy sand	1.62
38	97.7	1.6	0.7	Sand	1.66
39	91	2.2	6.8	Sand	1.66
40	89.6	2.8	7.6	Sand	1.66
41	100	0	0	Sand	1.66
42	93.3	2	4.7	Sand	1.66
43	87.9	4.1	8	Loamy sand	1.63

[a] FIN = foundry identification number; see Table 2-2 for details.
No data were available for FIN 3.

Table B-26. Pore Water Content (mg kg^{-1}) of Spent Foundry Sand Measured in a 1:1 Deionized Water:Spent Foundry Sand Extraction

	Al	As	B	Ba	Be	Ca	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Mo	Na	Ni	P	Pb	S	Se	Tl	V	Zn
PQL	**0.2**	**0.02**	**0.1**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.05**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**
1	*0.1*	*0.01*	0.519	0.06	*0.01*	81.8	*0.01*	*0.01*	*0.01*	0.027	1.14	22.1	5.65	*0.01*	0.108	456	*0.01*	0.053	*0.025*	4.31	*0.01*	*0.01*	*0.01*	*0.01*
2	*0.1*	*0.01*	0.135	0.082	*0.01*	25.2	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	37.7	29.4	*0.01*	0.041	165	*0.01*	0.026	*0.025*	84	*0.01*	*0.01*	*0.01*	*0.01*
3	39.8	0.054	1.1	0.778	*0.01*	4.37	*0.01*	*0.01*	*0.01*	*0.01*	7.75	13.9	11.8	*0.01*	0.252	321	*0.01*	0.841	*0.025*	156	*0.01*	*0.01*	*0.01*	0.123
4	*0.1*	*0.01*	1.37	0.083	*0.01*	211	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	54.6	14.9	0.114	0.024	167	*0.01*	0.02	*0.025*	107	*0.01*	*0.01*	*0.01*	*0.01*
5	1,720	0.162	4	2.85	*0.01*	94	*0.01*	0.47	0.29	0.176	402	24.8	235	1.58	0.246	577	0.219	0.543	0.148	198	0.022	*0.01*	0.054	1.54
6	*0.1*	*0.01*	3.96	0.376	*0.01*	60	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	25.3	64.2	*0.01*	2.85	56	*0.01*	0.278	*0.025*	147	*0.01*	*0.01*	*0.01*	*0.01*
7	*0.1*	*0.01*	0.295	0.159	*0.01*	261	*0.01*	0.029	*0.01*	*0.01*	*0.01*	27.3	25.9	1.78	*0.01*	0.635	0.373	0.017	*0.025*	376	*0.01*	*0.01*	*0.01*	0.045
8	0.248	*0.01*	0.172	*0.01*	*0.01*	5	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	19.6	2.69	*0.01*	*0.01*	116	*0.01*	0.061	*0.025*	43.1	*0.01*	*0.01*	*0.01*	*0.01*
9	*0.1*	0.024	0.208	*0.01*	*0.01*	14.8	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	17.6	1.1	*0.01*	0.047	66.4	*0.01*	0.514	*0.025*	33.4	*0.01*	*0.01*	0.035	*0.01*
10	0.225	*0.01*	0.273	*0.01*	*0.01*	65	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	16.9	0.993	*0.01*	0.044	10.8	*0.01*	0.114	*0.025*	44.2	*0.01*	*0.01*	*0.01*	*0.01*
11	75.9	0.033	4.42	0.049	*0.01*	27.2	*0.01*	*0.01*	*0.01*	*0.01*	15.6	45.9	13.2	0.059	0.229	480	*0.01*	0.08	*0.025*	221	*0.01*	*0.01*	*0.01*	0.096
12	*0.1*	*0.01*	0.393	*0.01*	*0.01*	31.1	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	15.2	4.65	0.047	*0.01*	2.17	*0.01*	*0.01*	*0.025*	28.1	*0.01*	*0.01*	0.028	*0.01*
13	0.782	0.022	0.534	0.026	*0.01*	25.6	*0.01*	*0.01*	*0.01*	*0.01*	0.242	50.6	2.11	*0.01*	*0.01*	242	*0.01*	*0.01*	*0.025*	167	*0.01*	*0.01*	*0.01*	*0.01*
14	437	0.109	42.2	0.486	*0.01*	44.8	*0.01*	*0.01*	0.054	0.107	141	25.3	103	1.02	0.116	337	0.059	1.62	0.07	125	*0.01*	*0.01*	0.1	0.426
15	3.89	0.066	0.36	*0.01*	*0.01*	16.4	*0.01*	*0.01*	*0.01*	*0.01*	0.935	25.9	2.36	0.022	0.156	357	*0.01*	0.224	*0.025*	256	*0.01*	*0.01*	*0.01*	*0.01*
16	*0.1*	*0.01*	0.267	0.022	*0.01*	38.2	*0.01*	*0.01*	*0.01*	*0.01*	0.04	29	13.5	*0.01*	*0.01*	120.5	*0.01*	0.391	*0.025*	115	*0.01*	*0.01*	*0.01*	*0.01*
17	1,107	0.059	0.449	0.119	*0.01*	107	*0.01*	*0.01*	*0.01*	*0.01*	251	22	187	*0.01*	0.119	461	*0.01*	0.345	*0.025*	261	*0.01*	*0.01*	*0.01*	0.271
18	55.5	0.048	0.598	0.05	*0.01*	17.5	*0.01*	*0.01*	*0.01*	*0.01*	11.7	30.8	12.3	0.034	0.079	279	*0.01*	4.39	*0.025*	171	*0.01*	*0.01*	*0.01*	0.031
19	111	*0.01*	0.357	0.578	*0.01*	12.2	*0.01*	*0.01*	*0.01*	*0.01*	46.5	16.4	24.2	0.298	0.126	271	0.042	0.169	*0.025*	117	*0.01*	*0.01*	*0.01*	0.3
20	0.982	*0.01*	1.67	*0.01*	*0.01*	45.7	*0.01*	*0.01*	*0.01*	*0.01*	0.206	32	4.24	0.02	0.111	429	*0.01*	0.106	*0.025*	248	*0.01*	*0.01*	*0.01*	*0.01*
21	*0.1*	*0.01*	0.118	*0.01*	*0.01*	4.5	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	15	0.464	0.022	0.048	68.7	*0.01*	*0.01*	*0.025*	34.1	*0.01*	*0.01*	*0.01*	*0.01*
22	*0.1*	*0.01*	0.206	*0.01*	*0.01*	49.6	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	34.1	9.7	0.404	0.092	281	*0.01*	0.037	*0.025*	291	*0.01*	*0.01*	*0.01*	*0.01*
23	1.79	0.071	0.428	*0.01*	*0.01*	25.3	*0.01*	*0.01*	*0.01*	*0.01*	0.376	27.8	1.84	*0.01*	0.13	498	*0.01*	0.249	*0.025*	343	*0.01*	*0.01*	*0.01*	*0.01*
24	145	*0.01*	0.531	0.068	*0.01*	20.8	*0.01*	*0.01*	*0.01*	*0.01*	45.7	23.6	20	*0.01*	0.105	215	0.025	0.585	*0.025*	97.4	*0.01*	*0.01*	*0.01*	0.309
25	1,847	0.116	1.49	4.5	*0.01*	124	*0.01*	0.054	0.083	0.083	323	39.3	313	0.777	0.136	659	0.135	1.02	0.205	223	0.039	*0.01*	0.099	1.88
26	*0.1*	0.082	0.56	*0.01*	*0.01*	36.9	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	27.8	1.17	*0.01*	1.12	407	*0.01*	0.218	*0.025*	300	*0.01*	*0.01*	*0.01*	*0.01*
27	*0.1*	*0.01*	0.828	0.036	*0.01*	47	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	53	18.6	*0.01*	0.067	374	0.021	0.071	*0.025*	275	*0.01*	*0.01*	*0.01*	*0.01*
28	65.1	0.019	0.842	0.068	*0.01*	32.5	*0.01*	*0.01*	*0.01*	*0.01*	28.5	19.3	29.6	0.117	0.117	155	0.03	0.46	*0.025*	104	*0.01*	*0.01*	0.016	0.176
29	9.9	0.049	0.208	0.336	*0.01*	8.87	*0.01*	*0.01*	0.245	0.832	8.22	854	1.652	0.15	0.518	123	0.402	4.89	*0.025*	14.5	*0.01*	*0.01*	0.186	0.071
30	74.3	0.047	0.7	0.032	*0.01*	26.3	*0.01*	*0.01*	*0.01*	0.07	12.7	25.6	14	0.054	0.104	468	*0.01*	1.17	*0.025*	368	*0.01*	*0.01*	*0.01*	0.023
31	*0.1*	*0.01*	3.54	0.05	*0.01*	245	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	19.1	27.4	8.64	*0.01*	147	*0.01*	*0.01*	*0.025*	518	*0.01*	*0.01*	*0.01*	0.065
32	266	0.055	3.31	0.16	*0.01*	35.6	*0.01*	*0.01*	0.039	0.012	51.7	23.1	54.1	0.126	0.089	345	0.035	0.406	*0.025*	204	*0.01*	*0.01*	0.039	*0.01*
33	0.322	*0.01*	0.345	0.03	*0.01*	4.73	*0.01*	*0.01*	*0.01*	0.469	0.57	12.5	1.62	0.154	0.054	1.71	*0.01*	0.036	*0.025*	1.2	*0.01*	*0.01*	*0.01*	*0.01*
34	*0.1*	*0.01*	0.223	0.03	*0.01*	41.9	0.023	*0.01*	*0.01*	1.14	0.57	38.2	15.3	0.127	0.063	180	*0.01*	0.661	*0.025*	66.6	0.647	*0.01*	*0.01*	2.25
35	*0.1*	0.086	0.888	0.205	*0.01*	44.6	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	35.3	74.4	0.162	0.145	526	0.032	0.453	*0.025*	264	*0.01*	*0.01*	*0.01*	0.617
36	1,116	0.124	0.786	1.21	*0.01*	84.9	*0.01*	0.033	0.059	0.087	252	53.3	156	0.817	0.183	531	0.066	3.89	*0.025*	238	*0.01*	*0.01*	0.084	1.44
37	366	0.09	0.852	0.136	*0.01*	35.3	*0.01*	*0.01*	*0.01*	*0.01*	98.6	19.1	63.6	0.222	0.132	237	0.023	0.712	*0.025*	101	*0.01*	*0.01*	0.053	0.216
38	37.6	0.021	0.134	0.104	*0.01*	7.8	*0.01*	*0.01*	*0.01*	0.217	9.02	130	8.42	0.146	0.143	594	0.202	1.54	*0.025*	27	*0.01*	*0.01*	*0.01*	0.055
39	1,418	0.089	1.17	0.979	*0.01*	80.7	*0.01*	0.048	0.09	1.7	272	90.6	261	3.02	0.55	472	2.9	8.84	0.169	138	0.024	*0.01*	0.063	0.819
40	841	0.074	0.546	0.465	*0.01*	75.1	*0.01*	0.026	0.056	0.081	158	30	165	0.203	0.099	287	0.046	0.598	0.087	122	*0.01*	*0.01*	0.087	0.273
41	*0.1*	*0.01*	0.138	*0.01*	*0.01*	16.8	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	10.6	2.83	0.104	*0.01*	10.9	*0.01*	*0.01*	*0.025*	12.2	*0.01*	*0.01*	*0.01*	*0.01*
42	193	0.076	0.416	0.067	*0.01*	16	*0.01*	*0.01*	*0.01*	*0.01*	37.5	17.2	32.3	0.093	0.116	204	*0.01*	0.522	*0.025*	68.7	*0.01*	*0.01*	*0.01*	0.172
43	264	0.059	1.13	0.12	*0.01*	27.4	*0.01*	*0.01*	*0.01*	0.09	46.9	38.6	43.2	0.239	0.207	383	0.085	0.782	*0.025*	241	*0.01*	*0.01*	0.03	0.22

Entries in italics were below the practical quantitation limit (PQL) and were calculated as 0.5 times the PQL.

Table B-27. Tentative Gas Chromatography-Mass Spectrometry Characterization of Products in the Pyrolyzates

Peak	Retention Time	Assignment	Molecular Weight
1	6:10	Ethylbenzene	106
2	12:47	1-Ethyl-2-methylbenzene	118
3	13:07	Phenol	94
4	15:11	1-Propynylbenzene	116
5	16:22	2-Methylphenol	108
6	17:18	1-Ethoxy-4-methylbenzene	136
7	17:20	3- and 4-Methylphenol	108
8	19:53	2,5-Dimethylphenol	122
9	20:47	2,3-Dimethylphenol	122
10	20:51	Naphthalene	128
11	20:55	2-Ethenyl-1,3,5-trimethylbenzene	146
12	21:52	2-Methyl-8-propyldodecane	226
13	23:25	2-Ethyl-4-methylphenol	136
14	24:25	2-Methylnaphthalene	142
15	24:27	(E)-5-Tetradecene	196
16	24:51	1-Methylnaphthalene	142
17	26:42	2,6-Dimethylheptadecane	268
18	26:48	2-Ethenylnaphthalene	154
19	27:13	2-Ethylnaphthalene	156
20	27:32	1,7-Dimethylnaphthalene	156
21	27:52	1,2-Dimethylnaphthalene	156
22	28:03	1,8-Dimethylnaphthalene	156
23	28:45	Acenaphthylene	152
24	28:52	Tridecanol	200
25	29:01	4-Methyloctadecane	268
26	30:28	Dibenzofuran	168
27	30:37	1,6,7-Trimethylnaphthalene	170
28	31:30	3-Ethyl-1-methylnaphthalene	170
29	31:33	1,4,5-Trimethylnaphthalene	170
30	32:08	Fluorene	166
31	32:09	1,4,6-Trimethylnaphthalene	170
32	32:50	1,2-Dimethyl-4-(phenylmethyl)-benzene	196
33	33:25	1-Methyl-7-(1-methylethyl)-naphthalene	184
34	33:35	2,6,10-Trimethylpentadecane	254
35	34:10	1,6-Dimethyl-4-(1-methylethyl)-naphthalene	198
36	35:25	1-Nonadecane	266
37	36:40	Anthracene	178
38	37:05	2,6,10,14-Tetramethylhexadecane	282
39	38:54	(Z)-9-Octadecenal	266

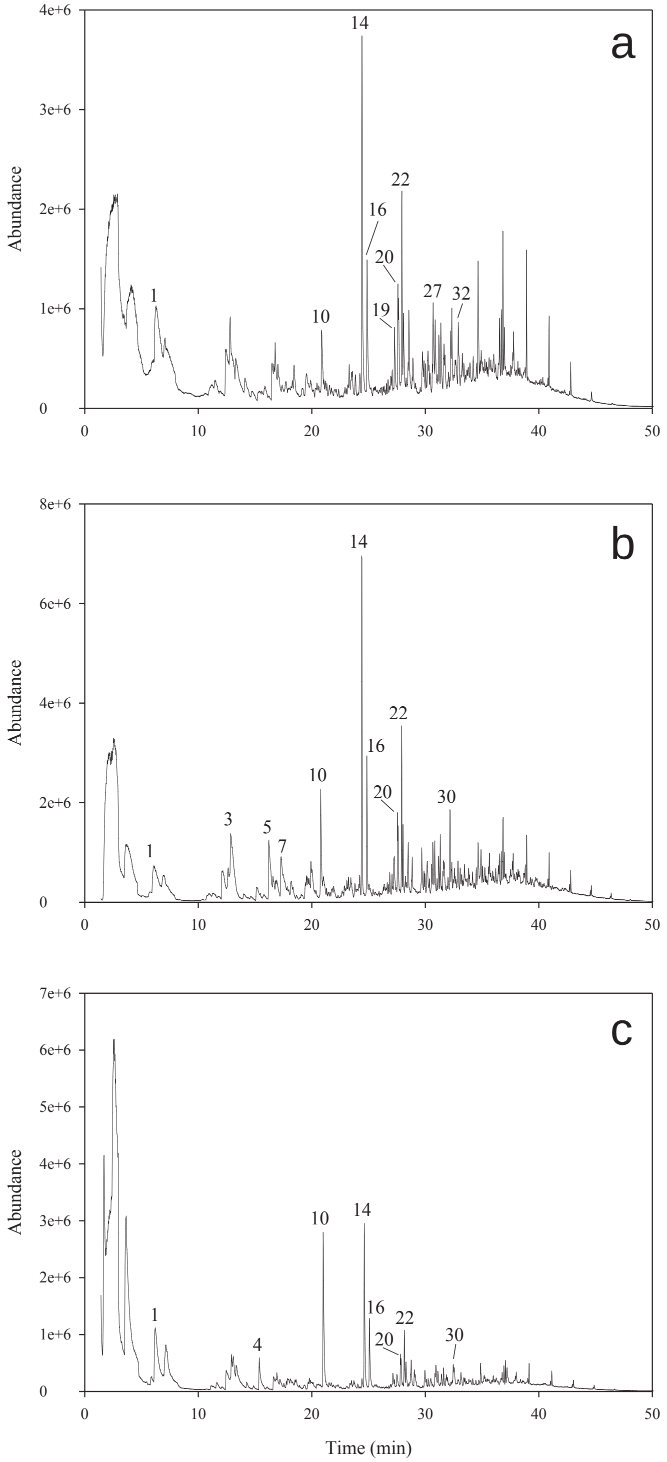

Figure B-1. Gas chromatogram of pyrolysis products from fresh green sand at a) 500°C, b) 750°C, and c) 1000°C.

The fresh green sand contained 92% silica sand, 4% sodium bentonite, 2% calcium bentonite, and 2% seacoal (w/w). Assignments of the labeled peaks are shown in Table B-27.

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Appendix C

Explanation of USDA Eco Screening Values for Cu, Ni, and Zn

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Appendix C

Explanation of USDA Eco Screening Values for Cu, Ni, and Zn

Inspection of Cu, Ni, and Zn concentrations in silica-based iron, steel and aluminum SFSs reveals a few samples with concentrations higher than the 95th percentile of U.S. and Canadian background soils. The 95th percentile was used to represent reasonable maximum background levels in soils, which have caused no known adverse effects in the environment (Scheckel et al., 2009; Chaney 2010). This does not mean that the 95th percentile of background is the beginning of potential toxicity; rather, that without more evaluation, we are not comfortable suggesting that the higher levels are free from concern about possible adverse effects.

The issue of some Ecological-Soil Screening Levels (Eco-SSLs) that were lower than some SFS samples and considerably lower than 95th percentile soil background levels has been noted. We have discussed the source of the Eco-SSL values, which are based on the worst case for each element. For Cu, Ni, and Zn, the source is acidic soils, low in clay, Fe, Mn and Al oxides, and organic matter, as well as the fresh addition of soluble metal salts (which have immediate, near 100% bioavailability, but react with soils over time to less bioavailable forms) in toxicological tests. USDA argues that these conditions are more severe than the environment where SFS and manufactured soils containing SFS would be used. Thus, alternative, more realistic limits were developed for Cu, Ni, and Zn, and an explanation of the derivation of these less conservative levels than the 95th percentile and Eco-SSL values was required.

It is important to note that the matrix of manufactured soils containing SFS is near neutral pH, with organic matter (typically 5-10% or higher) and balanced fertility ready to be sold as a topsoil. Furthermore, if soil pH is allowed to fall to below 5.5 (for Mn^{2+}) or 5.2 (for Al^{3+}) over time due to acidic rainfall and/or use of ammoniacal fertilizers, the soil will eventually become Mn, or Al and Mn phytotoxic and prevent growth of garden crops and even lawn grasses. Many garden crops fail at pH 5.5, which is still well above the worst case of the EcoSSL baseline for metals (pH 4.0). Most garden crops perform much better at a pH ranging from 6.5 to 7 than at lower pH; gardeners are advised to maintain soil pH in this range.

Copper: Cu is strongly bound by soil organic matter even at relatively low pH. Copper phytotoxicity has occurred in locations where mine wastes were dispersed, or where excessive fungicidal sprays were applied to trees growing in very strongly acidic, sandy, low organic matter soils. As shown in **Table C-1**, Cu levels in some agricultural soils have risen above the geochemical background levels from long term applications of Cu fertilizers and Cu-pesticides. Some peat soils require the addition of as much as 100 kg Cu ha^{-1} to achieve adequate Cu fertility for vegetable crops susceptible to Cu deficiency. Because field phytotoxicity of Cu to sensitive crops has not been observed until acidic sandy soils approach well over 200 mg Cu kg^{-1}, we conclude that 200 mg Cu kg^{-1} in a land-applied byproduct such as SFS is not a source of concern for ecological receptors.

Nickel: Soil Ni is transformed to insoluble solid phases at soil pH levels appropriate for crop production. Even added soluble Ni salts rapidly convert to insoluble solids, and those become decreasingly bioavailable over time as additional reactions occur with silicates (Scheckel and Sparks, 2001). Nickel has been shown to be phytotoxic in highly acidic soils surrounding Ni smelters in Canada, but not in soils with reasonable pH management. Natural serpentine soils contain 1000-2000 mg Ni kg^{-1} and are seldom phytotoxic until pH drops to pH 5.5 or below; phytotoxicity is readily reversed by simple addition of limestone (Kukier and Chaney, 2004; Siebielec et al., 2007). Because field phytotoxicity from Ni has been observed only when acidic soils exceed about 1000 mg kg^{-1}, we conclude that 200 mg Ni kg^{-1} in a land-applied byproduct such as SFS is not a source of concern for soil fertility or ecological receptors.

Zinc: Zn is a common soil contaminant because of its widespread commercial use in products and farm and garden implements. Urban emissions have raised levels of soil Zn in city centers as well. Zinc toxicity is the most common phytotoxic effect observed in the environment because of these uses (Chaney, 1993). Most cases of Zn phytotoxicity involved mine wastes, Zn smelter emissions, burned rubber tires, or pesticide sprays where high levels of Zn accumulated over time, and the soils were strongly acidic or very highly contaminated. Alkaline soils can contain over 1000 mg Zn kg^{-1} with no adverse effects, and even as high as 10,000 mg Zn kg^{-1} without harming plants or wildlife (USEPA, 2007). An example of home garden metals levels from the general Baltimore area was published by Mielke et al. (1983) (Table C-2). When some of the highly Pb- and Zn-contaminated soils were used in pot experiments to test uptake of metals by lettuce, even soil with 3,490 mg Zn, 5,210 mg Pb and 269 mg Cu kg^{-1} did not cause any adverse effects on the lettuce (Sterrett et al., 1993). A plant response test with Montreal, Quebec, Canada soils similarly found no adverse effects of substantial soil Cu and Zn levels on plant growth (Tambasco et al., 2000; Ge et al., 2002). Comparing SFS to urban soils shows that use of manufactured soils in urban gardens will usually provide lower soil Zn levels than background urban soils. The recognized adverse effect of excessive soil Zn is phytotoxicity if soil pH falls below 5.5 and especially below 5.0; simply incorporating agricultural limestone corrects and prevents future Zn phytotoxicity. Added soluble Zn fertilizers react over time to form solids or adsorbed species with lower phytoavailability such that additional Zn fertilizers may be required after 5-10 years. Higher soil Zn levels provide a reservoir of plant-available Zn that roots can access to obtain adequate Zn for plant growth and improve plant quality by increasing plant Zn concentrations. However, plant accumulation of Zn to levels above about 400-500 mg kg^{-1} dry leaves causes visibly evident phytotoxicity, but ruminant livestock tolerate diets with at least 500 mg salt Zn kg^{-1}, and monogastric animals tolerate higher dietary soluble Zn. Plant storage tissues (grain, fruits, edible roots) contain considerably lower Zn levels than do leaves. Thus the suggested investigatory limit of 300 mg Zn kg^{-1} in land-applied SFS is protective of soil fertility and ecological receptors.

Table C-1. Comparison of USDA Recommended maximum concentration of Cu, Ni, and Zn in SFS before additional investigation is required with Eco-SSL and 95th percentile of background U.S. and Canadian soils
(mg kg^{-1} DW)

Element	SFS 95th Percentile	SFS Maximum	USDA Recommendations	EPA Eco-SSL (Plants)	95th Percentile (Smith)	95th Percentile (Holmgren)
Cu	107	137	200	70	30.1	216
Ni	102	117	200	38	37.5	154
Zn	72.1	245	300	160	103	170

SOURCES: Holmgren et al. (1993); Smith et al. (2005); U.S. EPA (2007a, 2007b, and 2007c)

Table C-2. Concentrations of Zn, Cu and Ni extractable with 1.0 M HNO_3 in 422 Baltimore, MD, area gardens (mg kg^{-1} DW)

Element	Minimum	Median	Mean	90th Percentile	Maximum
Cu	0.7	17.2	25	64.4	96.7
Ni	0.5	2.8	4.9	8.4	53.4
Zn	0.3	92	211	521	4,880

SOURCE: Mielke et al. (1983)

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Appendix D

Meteorological Data

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Appendix D

Meteorological Data

This analysis reflects 5 years of representative meteorological data, including surface data and upper-air data. These data were obtained from 41 meteorological stations selected to represent the nine general climate regions of the continental United States. These observational data were processed and used as input to the home garden model and to the Industrial Source Complex, Short-Term Model, Version 3 (ISCST3).[1] Using the locations of the economic feasibility areas and their associated meteorological stations, a subset of the national data were extracted for use in the SFS analysis. This appendix describes the approach that was applied in selecting the representative meteorological stations and describes how the meteorological data were processed for use in the modeling.

D.1 Meteorological Station Selection

Forty-one meteorological stations were chosen to represent the nine general climate regions of the continental United States. The approach used the following three main steps:

1. Identify contiguous areas that are sufficiently similar, as defined by Bailey regions. Bailey's ecoregions and subregions of the United States (Bailey et al., 1994) are used to associate coverage areas with meteorological stations. This hierarchical classification scheme is based primarily on rainfall regimes; subregions are delineated by elevation and other factors affecting ecology.
2. Select one meteorological station to represent each contiguous area. The station selection step considered the following parameters:
   - **Major National Weather Service (NWS) station preferred**. NWS stations are expected to have high-quality equipment that is kept in good repair and is suitably sited.
   - **Number of years of surface-level meteorological data available**. More years of data provide a more realistic long-term estimate of air concentration and deposition.
   - **Central location within the area**. All other factors being equal, central locations are more likely to be representative of the entire contiguous area because they have the smallest average distance from all points in the region.
3. Identify the boundaries of the area to be represented by each meteorological station. Thiessen polygons, which are created by a geographic information systems (GIS) procedure that assigns every point on a map to the closest station, were used as the first step in drawing the boundaries. The borders of adjacent areas that were in different Bailey ecoregions were adjusted along the Bailey boundaries.

Table D-1 lists the selected stations for the continental United States and Figure D-1 shows these stations and their boundaries.

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[1] ISCST3 modeling was not performed specifically for this analysis. National ISCST3 modeling was performed to support EPA's 503 biosolids program. The SFS analysis applies a subset of the national outputs to estimate deposition impacts in SFS economic feasibility areas.

Table D-1. Surface-Level Meteorology Stations Used, by State

Station Number	Station Name	State
13963	Little Rock/Adams Field	AR
23183	Phoenix/Sky Harbor International Airport	AZ
93193	Fresno /Air Terminal	CA
23174	Los Angeles /International Airport	CA
23234	San Francisco /International Airport	CA
94018	Boulder Airport	CO
14740	Hartford/Bradley International Airport	CT
12839	Miami/International Airport	FL
12842	Tampa/International Airport	FL
13874	Atlanta/Atlanta-Hartsfield International	GA
24131	Boise/Air Terminal	ID
94846	Chicago/O'Hare International Airport	IL
12916	New Orleans/International Airport	LA
13957	Shreveport/Regional Airport	LA
14764	Portland/International Jetport	ME
14840	Muskegon/County Airport	MI
14922	Minneapolis-St Paul/International Airport	MN
13865	Meridian/Key Field	MS
24033	Billings/Logan International Airport	MT
03812	Asheville/Regional Airport	NC
13722	Raleigh/Raleigh-Durham Airport	NC
24011	Bismarck/Municipal Airport	ND
14935	Grand Island/Airport	NE
23050	Albuquerque/International Airport	NM
23169	Las Vegas/McCarran International Airport	NV
24128	Winnemucca/WSO Airport	NV
14820	Cleveland/Hopkins International Airport	OH
13968	Tulsa/International Airport	OK
24232	Salem/McNary Field	OR
14751	Harrisburg/Capital City Airport	PA
13739	Philadelphia/International Airport	PA
14778	Williamsport-Lycoming/County Airport	PA
13880	Charleston/International Airport	SC
13897	Nashville/Metro Airport	TN
12960	Houston/Intercontinental Airport	TX
24127	Salt Lake City/International Airport	UT
13737	Norfolk/International Airport	VA
14742	Burlington/International Airport	VT
24233	Seattle/Seattle-Tacoma International Airport	WA
03860	Huntington/Tri-State Airport	WV
24089	Casper/Natrona County International Airport	WY

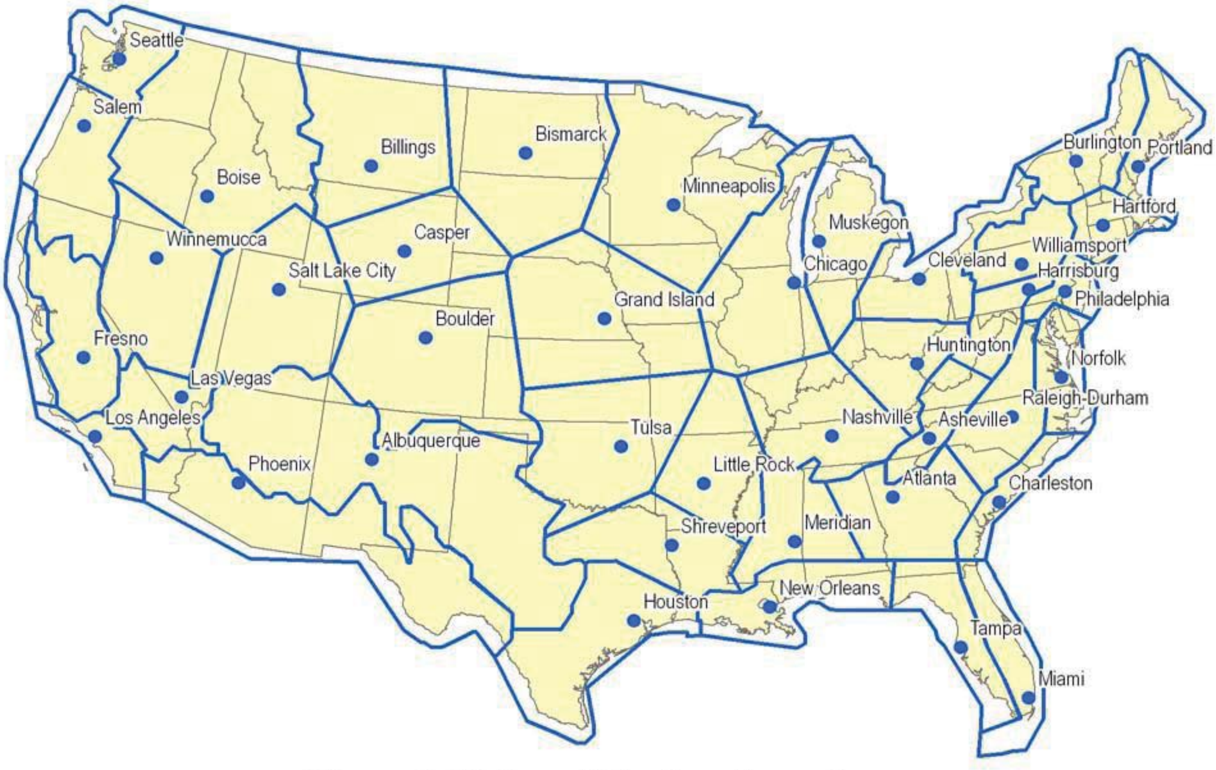

Figure D-1. Meteorological stations and regions

For purposes of this discussion, the contiguous United States was divided into the following sections: West Coast, Western Mountains, Desert Southwest, Gulf Coast, Southeast, Middle Atlantic, Northeast, Great Lakes, and Central States. The process of selecting stations and delineating the region assigned to each station is discussed in these sections.

D.1.1 West Coast

The California coast is divided just north of Los Angeles. This northern section is represented by the **San Francisco** International Airport (23234).

The southern California coast contains the Los Angeles basin south to the California/Mexico border. This region is represented by the **Los Angeles** International Airport (23174).

The California Central Valley Region, which encompasses the Sacramento Valley to the north and the San Joaquin Valley to the south, is defined by the Coast Range and Diablo Range to the west and the Sierra Nevada Mountains to the east. The valley extends south to the northern rim of the Los Angeles basin. The region is represented by **Fresno** Air Terminal (93193).

The coastal half of Oregon includes the Pacific Coast, the Central Valley Region, and the Great Sandy Desert, east to the Columbia Plateau. This region is represented by the station at McNary Field in **Salem**, OR (24232).

The coastal half of Washington is bounded by the edge of the Humid Temperate Domain to the east, the Washington/Canada border to the north, and the Columbia River to the south. This region is represented by the **Seattle-Tacoma** International Airport (24233).

D.1.2 Western Mountains

The **Boise** Air Terminal (24131) in Idaho represents the northern Rocky Mountains.

Almost all of Nevada and southeastern Oregon are represented by the station at **Winnemucca** WSO Airport (24128) in Nevada.

The Salt Lake Basin and the Great Divide Desert in Utah and Colorado are represented by the station at **Salt Lake City** International Airport (24127) in Utah.

D.1.3 Desert Southwest

The Desert Southwest is defined by various deserts and mountain ranges. One distinguishing feature is the transition between low desert in southern Arizona and high desert in northern Arizona. The southern boundary of this section is the U.S./Mexico border.

Southern Arizona, New Mexico, and western Texas comprise a region of low desert that is represented by the station at **Phoenix**/Sky Harbor International Airport (23183). The region is bounded to the north between Phoenix and Prescott, AZ, along the southern edge of the Columbia Plateau, which represents the transition from low to high desert.

Southeastern California, southern Nevada, and a small portion of northeastern Arizona are represented by the station at **Las Vegas**/McCarran International Airport (23169). This region is characterized by high desert.

The station at **Albuquerque** International Airport (23050) represents the mountainous region of northern Arizona, most of New Mexico, and central Texas.

D.1.4 Gulf Coast

The Texas Gulf Coast is represented by the station at **Houston** Intercontinental Airport (12960).

The Central Gulf Coast extends from western Louisiana through the Florida panhandle. This entire region is part of the Outer Coastal Plain Mixed Forest Province. The station at **New Orleans** International Airport (12916) in Louisiana was chosen to represent this region.

The West Coast of the Florida Peninsula is heavily influenced by the Gulf of Mexico, which has warmer water than the Atlantic Ocean off the East Coast of the Florida Peninsula. This region of the West Coast of Florida extends from the Florida Panhandle to the southern tip of Florida. The station at **Tampa** International Airport (12842) was chosen to represent this region.

D.1.5 Southeast

The Southeast section extends from the Atlantic coastal region of Florida and the Florida Keys northward through Georgia and North and South Carolinas. This region has an extremely broad coastal plain, requiring it to be divided between the coastal region and more inland regions for Georgia and South Carolina. This region also includes the inland areas of Louisiana, Mississippi, and Alabama.

The southern tip of Florida includes the Everglades, which have been drained along the Atlantic Coast to provide land for Miami, Ft. Lauderdale, West Palm Beach, and other coastal cities. This region north to the Georgia border is represented by the station at **Miami** International Airport (12839).

A long stretch of the Southeastern Atlantic Coast extends from the Georgia-Florida border through Georgia, South Carolina, and the southern portion of North Carolina. The boundary between the more forested coast and more agricultural inland area forms the western boundary. The station at **Charleston** International Airport (13880) represents this region.

The Blue Ridge region is further inland in Georgia and South Carolina. The station at **Atlanta** Hartsfield International Airport (13874) represents this region.

The inland areas of Alabama and Mississippi are represented by the station at **Meridian** Key Field (13865), which is located in Mississippi near the Alabama border. This area extends from the Central Gulf Coast region northward to southern Tennessee and westward to the Mississippi River Valley in western Mississippi.

The inland portion of Louisiana and eastern Texas is part of the Coastal Plain. This region extends northward to the Ouachita Mountains, which are just south of the Ozark Plateau in Arkansas. The hill country in eastern Texas is included. This region is represented by the station at **Shreveport** Regional Airport (13957) in Louisiana.

D.1.6 Middle Atlantic

The Middle Atlantic section includes coastal areas with bays, sounds, inlets, and barrier islands; a broad coastal plain; and the southern Appalachian Mountains.

The northern portion of the coastal region of North Carolina, coastal Virginia, and the Delmarva Peninsula is represented by the station at **Norfolk** International Airport (13737) in Virginia.

The Piedmont region of North Carolina, South Carolina, and Virginia is just inland from the coastal region. The station at **Raleigh-Durham** Airport (13722) in North Carolina represents this region.

The southern Appalachian Mountains lie to the west of the Piedmont region of North Carolina and Virginia. This region extends to the southwest to include a portion of western South Carolina and northeastern Georgia and to the northeast to include the southeastern portion of West Virginia. The station at **Asheville** Regional Airport (03812) in North Carolina was chosen to represent this region.

The Appalachian Mountains of West Virginia and eastern Kentucky are represented by the station at **Huntington** Tri-State Airport (03860) in West Virginia.

The inland region encompassing northern Virginia, part of Maryland, and eastern Pennsylvania is composed of another section of the Appalachian Mountains. Boundaries are approximated by the Bailey's Central Appalachian Forest province. The station at **Harrisburg**/Capital City Airport (14751) in Pennsylvania represents this region.

The area just to the north of the Chesapeake Bay northward through New Jersey, eastern Pennsylvania, and New York City is characterized by the Eastern Broadleaf Forest (Oceanic) Province in the Coastal Plain. The station at **Philadelphia** International Airport (13739) in Pennsylvania represents this region.

D.1.7 Northeast

The Northeast section includes Maine and New England. This region is characterized by forests to the north, large urban areas along the southern Coastal Plain, and the mountain ridges and valleys of the northern Appalachian Mountains. This section is bounded by the Atlantic Ocean on the east, the U.S.-Canada border on the north, and the coastal plain of the eastern Great Lakes to the west.

The station at Bradley International Airport (14740) in **Hartford**, CT, represents the New England region, which encompasses Connecticut, Massachusetts, Rhode Island, and a small portion of Vermont, New Hampshire, and eastern New York.

Northern New England and Maine are represented by the station located at the International Jetport (14764) in **Portland**, ME. This region includes Maine and most of New Hampshire and Vermont.

The station at the International Airport (14742) in **Burlington**, VT, represents northeastern New York, Vermont, New Hampshire, and western Maine.

The remainder of the northern Appalachian Mountains in New York and Pennsylvania is represented by the station at **Williamsport-Lycoming** (14778) in Pennsylvania. This region is bounded on the west by the Adirondack Mountains, just to the east of the coastal plain of Lake Ontario.

D.1.8 Great Lakes

The Eastern Great Lakes divide the United States and Canada. On the U.S. side, the western portion of New York, a small portion of Pennsylvania, and northeastern Ohio border the eastern shores of Lake Ontario and Lake Erie. Mountains form the eastern boundary. The western border is just inland from the western shore of Lake Erie. The station at Hopkins International Airport (14820) in **Cleveland**, OH, represents this region.

The Lower Peninsula of Michigan is bordered by the Great Lakes on three sides. As previously noted, the eastern portion along Lake Erie is represented by the station in Cleveland, OH. The remainder of the Lower Peninsula of Michigan and the eastern portion of the Upper Peninsula of Michigan are represented by the station at **Muskegon** County Airport (14840).

The western shore of Lake Michigan, which includes Green Bay, is formed by the northeastern portion of Illinois, eastern Wisconsin, and part of the Upper Peninsula of Michigan.

Lake Superior forms the northern boundary of this region, and the western boundary is formed by the hills to the east of the Wisconsin River and the Upper Mississippi River. Most of Illinois, western Indiana, eastern Iowa, and northeastern Missouri are included in this region, which is represented by the station at O'Hare International Airport (94846) in **Chicago**, IL.

D.1.9 Central States

This section includes the Central Lowlands (south of the Great Lakes), the Midwest, and the Great Plains. The elevation for this section is generally lowest in the Mississippi Valley, which extends through the Midwest and drains a large portion of the center of the continental United States. This section also includes other major river valleys, including the Ohio, Tennessee, and Missouri. This section is bordered on the east by the Appalachian Mountains, on the west by the Rocky Mountains, on the north by the border with Canada, and on the south by the Southeast section, Texas, and the Desert Southwest section.

One region includes western Kentucky, central and western Tennessee north of Memphis, and southeastern Missouri east of the Ozark Plateau, southern Illinois, and southern Indiana. This region is represented by the station at **Nashville** Metropolitan Airport (13897) in Tennessee.

A large region is assigned to the station at Adams Field (13963) in **Little Rock**, AR.

The northern portion of the Midwest includes the portion of Wisconsin west of the Lake Michigan coastal plain, the western portion of the Upper Peninsula of Michigan, Minnesota, and the eastern portion of North and South Dakota. This region is represented by the station at **Minneapolis-St. Paul** International Airport (14922) in Minnesota.

The Great Plains lie between the Central Lowlands to the east and the Rocky Mountains to the west. Lands at higher elevations are more grassland and shrubland used for cattle ranges, whereas the lower elevations are more frequently used for crops. The region that includes most of North and South Dakotas is represented by the station at **Bismarck** Municipal Airport (24011) in North Dakota.

The central portion of Montana is more rugged, but still part of the Great Plains. The Rocky Mountains form the western and southwestern boundaries of this region, which is represented by the station at **Billings** Logan International Airport (24033) in Montana.

The station at **Casper**/Natrona County International Airport (24089) in Wyoming represents most of Wyoming, southwestern South Dakota, and northwestern Nebraska.

Most of Colorado, southwestern Nebraska, western Kansas, and the panhandle of Oklahoma are represented by the station at the **Boulder** Airport (23062) in Colorado.

The north central portion of the Great Plains includes most of Nebraska, northern Kansas, western Iowa, southeastern South Dakota, and northwestern Missouri. This region is represented by the station at **Grand Island** Airport (14935) in Nebraska.

The southern portion of the Great Plains includes most of Kansas, part of Missouri, and eastern Oklahoma. This region is represented by the station at **Tulsa** International Airport (13968).

D.2 Processing Meteorological Data

Surface Data. Hourly surface meteorological data used in air dispersion and deposition modeling were processed from the Solar and Meteorological Surface Observation Network (SAMSON) CD-ROM (U.S. DOC and U.S. DOE, 1993). Variables included the following:

- Temperature
- Pressure

- Wind direction
- Wind speed
- Opaque cloud cover
- Ceiling height
- Current weather
- Hourly precipitation.

Upper-Air Data. Twice-daily mixing-height data were calculated from upper-air data contained in the radiosonde data of the North America CD-ROM set (NCDC, 1997). This set contains upper-air data from 1946 through 1996 for most upper-air stations in the United States. The upper-air data were combined with the SAMSON data to create the mixing-height files. EPA’s Support Center for Regulatory Air Models (SCRAM) bulletin board was also used to obtain mixing-height data (if available) when mixing-height data could not be successfully calculated from the radiosonde data. This risk assessment used variable mixing heights that were based on hourly ceiling height observations used in the ISCST3 air model.

Filling in Missing Data. The program SQAQC identified missing surface data by searching for incidents of missing data on the observation indicator, opaque cloud cover, temperature, station pressure, wind direction and wind speed, and ceiling height. Years that were missing 10% or more of the data were discarded (Atkinson and Lee, 1992). Verification (quality control [QC]) checks were performed on the SQAQC program by applying it to station data where the missing data were known.

For years missing less than 10% of the data, missing surface data were filled in by a program called METFIX. This program fills in up to 5 consecutive hours of data for cloud cover, ceiling height, temperature, pressure, wind direction, and wind speed. For single missing values, the METFIX program follows the objective procedures developed by Atkinson and Lee (1992). For two to five consecutive missing values, other rules were developed because the subjective methods provided by Atkinson and Lee (1992) rely on professional judgment and could not be programmed. The METFIX program flagged files where missing data exceeded five consecutive values. In the few cases where this occurred and the missing data did not constitute 10% of the file, they were filled in manually using procedures from Atkinson and Lee (1992).

All upper-air files were checked for missing data using a program called QAQC. QAQC produces a log file containing occurrences of missing mixing height. Verification (QC) checks were performed on the QAQC program by applying it to station data where the missing data were known.

Missing mixing heights were filled in by interpolating one to five consecutive missing values. According to Atkinson and Lee (1992), if there are one to five consecutive missing values, then the values should be filled in subjectively using professional judgment. Again, programming these subjective procedures was not feasible, and the program used simple linear interpolation to automatically fill in these values. Information from Atkinson and Lee (1992) was used to determine which files should be discarded (i.e., files missing more than five consecutive missing values or missing 10% or more of the data). After the missing mixing heights were filled in for all upper-air files, they were checked again for missing data using the QAQC program.

Other Input Data. Processing of meteorological data also required the following site/NWS specific parameters:

- Anemometer height (m)
- Bowen ratio
- Minimum Monin-Obukhov length (m)
- Noontime albedo
- Roughness length (m), surface meteorological station
- Fraction net radiation absorbed by the ground.
- Roughness length (m), area around facility
- Anthropogenic heat flux (W m^{-2})

Anemometer height was collected from local climatic data summaries (NOAA, 1983). When anemometer height was not available, the station was assigned the most common anemometer height from the other stations (6.1 m).

Land-use information is required for determining a number of inputs. To obtain this information, a GIS determined the land uses within a 3-km radius around each meteorological station using Geographic Retrieval and Analysis System (GIRAS) spatial data with Anderson land-use codes (Anderson et al., 1976). A weighted average of these land uses was used to estimate the Bowen ratio, minimum Monin-Obukhov length, the noontime albedo, the roughness length at the meteorological station, and the fraction of net radiation absorbed by the ground. The Bowen ratio is a measure of the amount of moisture at the surface around a meteorological station. The wetness of a location was determined based on the average annual precipitation amount. For this analysis, the annual average values were applied. The minimum Monin-Obukhov length, which is a measure of the atmospheric stability at a meteorological station, was correlated with the land-use classification. Noontime albedo values were also correlated with land use around a meteorological station. Table D-2 presents the crosswalk between the Anderson land-use codes from the GIRAS and the PCRAMMET land-use designations used in air modeling. Other data used in the ISCST3 modeling are presented in Tables D-3 through D-6. These are the Bowen ratio (Table D-3), the minimum Monin-Obukhov length (Table D-4), Albedo values (Table D-5), and surface roughness length (Table D-6).

The surface roughness length is a measure of the height of an obstacle to the wind flow. It is not equal to, but generally proportional to the physical dimensions of the obstacle. The roughness length was assumed to be the same at the meteorological station and at the garden site.

Table D-2. Relation between Anderson Land-Use Codes and PCRAMMET Land-Use Codes

Anderson Code and Description[a]	PCRAMMET Type and Description[b]
51 Streams and canals	1 Water surface
52 Lakes	1 Water surface
53 Reservoirs	1 Water surface
54 Bays and estuaries	1 Water surface
41 Deciduous forest land	2 Deciduous forest
61 Forested wetland	2 Deciduous forest
42 Evergreen forest land	3 Coniferous forest
43 Mixed forest land	4 Mixed forest
62 Nonforested wetland	5 Swamp (nonforested)
84 Wet tundra	5 Swamp (nonforested)
21 Cropland and pasture	6 Agricultural
22 Orchards-groves-vineyards-nurseries-ornamental	6 Agricultural
23 Confined feeding operations	6 Agricultural
24 Other agricultural land	6 Agricultural
31 Herbaceous rangeland	7 Rangeland (grassland)
32 Shrub and brush rangeland	7 Rangeland (grassland)
33 Mixed rangeland	7 Rangeland (grassland)
11 Residential	9 Urban
12 Commercial and services	9 Urban
13 Industrial	9 Urban
14 Transportation-communication-utilities	9 Urban
15 Industrial and commercial complexes	9 Urban
16 Mixed urban or built-up land	9 Urban
17 Other urban or built-up land	9 Urban
71 Dry salt flats	10 Desert shrubland
72 Beaches	10 Desert shrubland
73 Sandy areas not beaches	10 Desert shrubland
74 Bare exposed rock	10 Desert shrubland
75 Strip mines-quarries-gravel pits	10 Desert shrubland
76 Transitional areas	10 Desert shrubland
81 Shrub and brush tundra	10 Desert shrubland
82 Herbaceous tundra	10 Desert shrubland
83 Bare ground	10 Desert shrubland
85 Mixed tundra	10 Desert shrubland
91 Perennial snowfields	10 Desert shrubland
92 Glaciers	10 Desert shrubland

[a] Anderson codes from Anderson and colleagues (1976)
[b] PCRAMMET codes from U.S. EPA (1995)

Table D-3. Daytime Bowen Ratio by Land Use and Season

Land-Use Type	Spring			Summer			Autumn			Winter			Annual Average		
	Dry	**Wet**	**Avg.**	**Dry**	**Wet**	**Avg.**	**Dry**	**Wet**	**Avg.**	**Dry**	**Wet**	**Avg.**	**Dry**	**Wet**	**Avg.**
Water surface	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	2.0	0.3	1.5	0.575	0.15	0.45
Deciduous forest	1.5	0.3	0.7	0.6	0.2	0.3	2.0	0.4	1.0	2.0	0.5	1.5	1.53	0.35	0.875
Coniferous forest	1.5	0.3	0.7	0.6	0.2	0.3	1.5	0.3	0.8	2.0	0.3	1.5	1.4	0.275	0.825
Swamp	0.2	0.1	0.1	0.2	0.1	0.1	0.2	0.1	0.1	2.0	0.5	1.5	0.65	0.2	0.45
Cultivated land (agricultural)	1.0	0.2	0.3	1.5	0.3	0.5	2.0	0.4	0.7	2.0	0.5	1.5	1.63	0.35	0.75
Grassland	1.0	0.3	0.4	2.0	0.4	0.8	2.0	0.5	1.0	2.0	0.5	1.5	1.75	0.425	0.825
Urban	2.0	0.5	1.0	4.0	1.0	2.0	4.0	1.0	2.0	2.0	0.5	1.5	3.0	0.75	1.6
Desert shrubland	5.0	1.0	3.0	6.0	5.0	4.0	10.0	2.0	6.0	10.0	2.0	6.0	7.75	2.5	4.75

Source: U.S. EPA (1995)
Averages were computed for this effort.

Table D-4. Minimum Monin-Obukhov Length (Stable Conditions)

Urban Land-Use Classification	Length (m)
Agriculture (open)	2
Residential	25
Compact residential/industrial	50
Commercial (19–40 story buildings) (>40 story buildings)	100 150

Source: U.S. EPA (1995)

Table D-5. Albedo Values of Natural Ground Covers for Land-Use Types and Seasons

Land-Use Type	Spring	Summer	Autumn	Winter	Annual Average
Water surface	0.12	0.1	0.14	0.2	0.14
Deciduous forest	0.12	0.12	0.12	0.5	0.22
Coniferous forest	0.12	0.12	0.12	0.35	0.18
Swamp	0.12	0.14	0.16	0.3	0.18
Cultivated land (agricultural)	0.14	0.2	0.18	0.6	0.28
Grassland	0.18	0.18	0.20	0.6	0.29
Urban	0.14	0.16	0.18	0.35	0.21
Desert shrubland	0.3	0.28	0.28	0.45	0.33

Source: U.S. EPA (1995) Average values were computed for this analysis.

Table D-6. Surface Roughness Length for Land-Use Types and Seasons (m)

Land-Use Type	Spring	Summer	Autumn	Winter	Annual Average
Water surface	0.0001	0.0001	0.0001	0.0001	0.0001
Deciduous forest	1.0	1.3	0.8	0.5	0.9
Coniferous forest	1.3	1.3	1.3	1.3	1.3
Swamp	0.2	0.2	0.2	0.05	0.16
Cultivated land (agricultural)	0.03	0.2	0.05	0.01	0.07
Grassland	0.05	0.2	0.01	0.001	0.04
Urban	1.0	1.0	1.0	1.0	1.0
Desert shrubland	0.3	0.3	0.3	0.15	0.26

Source: U.S. EPA (1995) Average values were computed for this analysis.

During daytime hours, the heat flux into the ground is parameterized as a fraction of the net radiation incident on the ground. This fraction varies based on land use. A value of 0.15 was used for rural locations. Suburban and urban locations were given values of 0.22 and 0.27, respectively (U.S. EPA, 1995).

Anthropogenic heat flux is negligible for meteorological stations outside of highly urbanized locations; however, in areas with high population densities or energy use, such as industrial facilities, this flux may not always be negligible (U.S. EPA, 1995). For this analysis, anthropogenic heat flux was assumed to be zero for all meteorological stations.

D.3 References

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Appendix E

Soil Data

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Appendix E

Soil Data

A soils dataset was developed to represent the variability in soil conditions in areas that produce SFS. This dataset was defined from a national dataset currently used by EPA to support the 503 biosolids program. Using the locations of the SFS economic feasibility areas, a subset of the national data was extracted for use in the SFS analysis. This appendix describes the approach that was applied in developing the national dataset.

A representative sample of soils was generated by overlaying 7,000 points on a soils layer using a geographic information system (GIS). The 7,000 points were distributed proportionally to the number of farms located in each meteorological region. The points were located randomly within each meteorological region, and information on the soil map unit corresponding to each point was extracted. The predominant texture by depth for the top 20 cm of soil was determined using soil texture data by layer. Additional details on the data sources and methods used to collect soil textures and relate them to the hydrologic soil properties needed for modeling are provided below.

E.1 Data Sources

The primary source for soil properties data was the State Soil Geographic (STATSGO) database. STATSGO is a repository of nationwide soil properties primarily compiled by USDA from county soil survey data (USDA, 1994). STATSGO includes a 1:250,000-scale GIS coverage that delineates soil map units and an associated database containing soil data for each STATSGO map unit. (Map units are areas used to spatially represent soils in the database.)

In addition, two compilations of STATSGO data, each keyed to the STATSGO map unit GIS coverage, and land-use data from the Geographic Information Retrieval and Analysis System (GIRAS) land-use database were used as convenient sources of average soil properties:

- **USSOILS**. USSOILS (Schwarz and Alexander, 1995) averages STATSGO data over the entire soil column for each map unit.
- **CONUS**. CONUS (Miller and White, 1998) provides average STATSGO data by map unit and a set of 11 standardized soil layers.
- **GIRAS**. The GIRAS land-use database (U.S. EPA, 1994) provides comprehensive landuse data in a digital GIS format for the contiguous 48 states.

Soil properties derived directly from STATSGO, CONUS, or USSOILS data include organic matter content, Universal Soil Loss Equation (USLE) K (erodibility) and S (slope) factors, and pH. A complete set of hydrologic soil properties was not available from STATSGO.[1] To ensure consistent and realistic values, it was necessary to rely on established, nationwide relationships between hydrologic properties and soil texture or hydrologic soil group, both of

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[1] Hydrologic soil properties required for modeling include bulk density, saturated water content, residual water content, field moisture content, wilting point, saturated hydraulic conductivity, soil moisture coefficient b, and soil moisture retention parameters alpha and beta.

which are available from STATSGO. Sources for these relationships include Carsel and Parrish (1988), Carsel et al. (1988), and Clapp and Hornberger (1978). These peer-reviewed references provide a consistent set of correlated hydrologic properties for each soil texture or hydrologic group. Table E-1 lists soil properties collected for this analysis and their data sources.

Table E-1. Summary of Soil of Properties Collected for Sewage Sludge Risk Analysis

Soil Variable	Units	Data Source
Properties Derived from Soil Texture		
USDA soil texture	Unitless	CONUS/STATSGO
Saturated hydraulic conductivity	cm h^{-1}	Relationship from Carsel and Parrish (1988)
Saturated water content	L L^{-1}	Relationship from Carsel and Parrish (1988)
Soil moisture coefficient b	Unitless	Relationship from Clapp and Hornberger (1978)
Soil bulk density	mg L^{-1}	Calculated from saturated water content
Root zone depth	cm	Relationship (with land use) from Dunne and Leopold (1978)
Properties Derived from Soil Hydrologic Class		
Soil Conservation Service (SCS) hydrologic class	Unitless	CONUS/STATSGO
Field capacity	% (vol.)	Relationship from Carsel et al. (1988)
Wilting point	% (vol.)	Relationship from Carsel et al. (1988)
SCS curve number	Unitless	Relationship (with land use) from USDA (1986)
Properties Obtained Directly from STATSGO		
Fraction organic carbon	g g^{-1}	STATSGO
Silt content	% (wt.)	STATSGO
USLE erodibility factor (K)	kg m^{-2}	STATSGO
USLE slope (S)	Degrees	STATSGO
Properties Derived from Slope		
USLE slope length (L)	m	Relationship from Lightle and Weesies (1998)
USLE length/slope factor (LS)	Unitless	Calculated from L and S per Williams and Berndt (1977)

Finally, two parameters—root zone depth and Soil Conservation Service (SCS) curve number (used for recharge calculations)—required site-based land-use data, as well as soil texture or hydrologic soil group. The land-use data were obtained for each of the 41 meteorological regions from the GIRAS land-use database (U.S. EPA, 1994). Land-use and land-cover information in GIRAS was mapped and coded using the Anderson classification system (Anderson et al., 1976), which is a hierarchical system of land-use characterizations. This nationwide coverage is based on late-1970s to early-1980s satellite images and aerial photography. The relationships used to convert the land-use and soil data were obtained from Dunne and Leopold (1978) for root-zone depth and USDA (1986) for the SCS curve number.

E.2 Data Collection

Soil data collection began by overlaying the boundaries of the 41 meteorological regions onto the STATSGO map units to determine the STATSGO map units and their areas within each region. These data were then used to derive predominant soil properties within each meteorological region, either through direct calculations or by applying established relationships in lookup tables. Soil model inputs were based on the soil properties of the predominant soil type (texture and hydrologic group) for each STATSGO map unit having agricultural land use within the meteorological region.

Twelve common soil textures were collected to develop soil and hydrologic properties. Using CONUS data for the top three surface soil layers (20 cm) in each STATSGO map unit, the soil texture of the thickest CONUS layer was considered the predominant texture for the map unit. The textures were ranked according to predominance across all map units and, when there were two soil textures with equal depths, the texture with the higher ranking was chosen for that map unit. For the 303 out of 7,000 map units without one of the 12 common soil textures (e.g., those with water or organic matter), the predominant soil texture (i.e., loam) was selected. Soil column texture was obtained in a similar manner, except that all CONUS layers were used. Attachment A to this appendix presents the percentage of soil textures within each meteorological region.

To limit data collection to agricultural soils in each meteorological region, GIS programs overlayed the STATSGO map units with the GIRAS land-use coverage to determine which map units (and their respective areas) occur in cropland use and pastureland use (i.e., Anderson land-use code 21). These data were then processed to create a set of the 12 soil textures, ranked by percentage of land in agricultural use with each texture, for each region. These textures were used to derive soil properties for this analysis for each region/texture combination as described in the next section.

Because certain soil properties were derived from SCS hydrologic soil groups, it was necessary to develop a hydrologic soil group that would be consistent with the soils of each texture within a region. To do so, a table of hydrologic soil groups by STATSGO map unit was created using STATSGO data for hydrologic soil groups by the component soils within the map unit. Based on the predominant texture for each map unit, hydrologic soil groups for the component soils with the same texture were averaged across each map unit (weighted by component percent) using the numeric conversion: Group A=1, Group B=2, Group C=3, and Group D=4. These values were then averaged again (weighted by map unit area) for each soil texture occurring in a region. These regional average textures were converted back to letters using the same conversion, resulting in a hydrologic soil group for each texture occurring within a meteorological region. A hydrologic soil group applies to the entire soil column and is not layer-specific.

E.3 Development of Soil Properties

After the distribution of soil textures and their related hydrologic class were determined for each meteorological region, average soil properties were determined for each soil texture present in a region by relationships with soil texture or hydrologic class or by extracting the data for soils of each texture directly from STATSGO.

Soil Properties Based on Relationship with Soil Texture—Several soil hydrologic properties were derived directly from the soil texture using database lookup tables relating mean properties to texture class (see Table E-2):

- **Saturated hydraulic conductivity (cm h^{-1})** was determined for both surface soil and the entire soil column using a national relationship from Carsel and Parrish (1988).
- **Saturated water content (unitless)** was determined for both surface soil and the entire soil column using a relationship from Carsel and Parrish (1988).
- **Bulk density (g cm^{-3})** was calculated for surface soil from saturated water content using the following equation:

$$\rho_b = 2.65(1 - \varphi) \quad \text{(E-1)}$$

Where

ρ_b = Bulk density of the soil (U.S. EPA, 1997)
2.65 = Particle density in g cm^{-3} (assumed to be quartz)
φ = Saturated water content

- **Soil moisture coefficient (unitless)** was determined for both the surface soil and the entire soil column using a relationship from Clapp and Hornberger (1978).
- **Depth to root zone (cm)** was determined using a Dunne and Leopold (1978) table of rooting depth by vegetation type and soil texture. For each soil texture, a minimum and a maximum root zone depth (for shallow and deep-rooted crops) were used to represent the range across cropland and pastureland use. Because Dunne and Leopold (1978) included only five soil textures, these five textures were mapped across the 12 basic textures used in this analysis (see Table E-3).

Table E-2. Hydrological Soil Parameters Correlated to Soil Texture

Soil Texture	Saturated Hydraulic Conductivity[a] (cm h^{-1})	Saturated Water Content[a] (L L^{-1})	Bulk Density [b] (g cm^{-3})	Soil Moisture Coefficient [c]
Clay	0.20	0.38	1.643	11.4
Sandy clay	0.12	0.38	1.643	10.4
Silty clay	0.02	0.36	1.696	10.4
Clay loam	0.26	0.41	1.5635	8.52
Sandy clay loam	1.31	0.39	1.6165	7.12
Silty clay loam	0.07	0.43	1.5105	7.75
Sand	29.70	0.43	1.5105	4.05
Loamy sand	14.59	0.41	1.5635	4.38
Sandy loam	4.42	0.41	1.5635	4.90
Loam	1.04	0.43	1.5105	5.39
Silt	0.25	0.46	1.431	—
Silt loam	0.45	0.45	1.4575	5.30

[a] Carsel and Parrish (1988)
[b] Calculated from WCS using equation from U.S. EPA (1997)
[c] Clapp and Hornberger (1978)

Table E-3. Depth to Root Zone Values

USDA Soil Texture	Dunne and Leopold Texture	Shallow-Rooted Crops (cm)	Deep-Rooted Crops (cm)
Clay	Clay	25	67
Sandy clay			
Silty clay			
Clay loam	Clay loam	40	100
Sandy clay loam			
Silty clay loam			
Sand	Fine sand	50	100
Loamy sand	Fine sandy loam	50	100
Sandy loam			
Loam	Silt loam	62	125
Silt			
Silt loam			

Source: Derived from Dunne and Leopold (1978)

Soil Parameters Based on Relationship with Hydrologic Group—The following soil parameters are all based on the average hydrologic soil group for each texture within a meteorological region. Mean values by hydrologic group were obtained using the following relationships (see Tables E-4 and E-5):

- **Soil moisture field capacity (volume %)**. A single field capacity value was obtained for each soil group by averaging the layered property values from Carsel et al. (1988).
- **Soil moisture wilting point (volume %)**. A single wilting point value was obtained for each soil group by averaging the layered property values from Carsel et al. (1988).
- **SCS curve number (unitless)**. Minimum and maximum SCS curve number values were determined for each regional soil texture based on a USDA (1986) table of curve numbers by cover type and hydrologic soil group, assuming a good condition pasture-land use for the minimum and poor-condition cropland use for the maximum. A lookup table with minimum and maximum SCS curve numbers by hydrologic soil group was used to assign the appropriate value for each regional soil texture according to its hydrologic soil group.

Table E-4. Field Capacity and Wilting Point Values

Hydrologic Group	Layer	Field Capacity	Wilting Point
A	1	9.4	3.1
	2	8.1	2.3
	3	5.9	2.1
	4	5.8	1.9
	Average	*7.3*	*2.4*
B	1	19.1	8.7
	2	18.8	9.3
	3	18.7	8.9
	4	17.5	8.4
	Average	*18.5*	*8.8*
C	1	22.5	10.4
	2	23.2	12.1
	3	22.9	11.9
	4	21.3	11.5
	Average	*22.5*	*11.5*
D	1	24.2	13.8
	2	26.3	17.0
	3	25.6	16.3
	4	24.4	15.1
	Average	*25.1*	*15.6*

Source: Carsel et al. (1988)

Table E-5. SCS Curve Number Values by SCS Hydrologic Soil Group

SCS Hydrologic Soil Group	SCS Curve Number	
	Minimum	Maximum
A	39	72
B	61	81
C	74	88
D	80	91

Source: Derived from USDA (1986)

E.4 Parameters Collected Directly from STATSGO-Based Data Sources

Several variables were obtained directly from STATSGO (Schwarz and Alexander, 1995). Although these variables were not derived from soil texture, they were extracted and averaged based only on soil map units with the predominant texture to ensure consistent soil properties.

These variables include the following:

- **USLE erodibility factor—top 20 cm (tons acre^{-1})**. An area-weighted average erodibility factor for the top 20 cm of soil was calculated from STATSGO data by layer and component. STATSGO layer data were translated into *K* values using standardized CONUS layers and calculating a depth-weighted average value. Furthermore, a component percent-weighted average *K* was calculated for each CONUS layer across all components contained in each map unit. The resulting table contains *K* values by map unit and standardized CONUS layer. To derive one value for *K* by map unit for the top 20 cm of soil, a depth-weighted average for the top three CONUS layers was calculated. The final *K* value by meteorological region and soil texture was obtained by averaging the map units for each surface soil texture present within the meteorological region.
- **Fraction organic carbon (foc)—top 20 cm (mass fraction)**. An area-weighted average *foc* for surface soils was calculated for each region and soil texture using only the map units with the predominant surface soil texture of interest within the region. Percent organic matter for the top 20 cm of soil was obtained from STATSGO organic matter data by layer and component (Schwarz and Alexander, 1995) and converted to *foc* by dividing by 174 (100 × 1.74 g organic matter g^{-1} of organic carbon) (U.S. EPA, 1997). Percent organic matter values were translated from STATSGO layer and component into standardized CONUS layers using the same methodology described for the USLE erodibility factor *K*. Then, a depth-weighted average percent organic matter was calculated for the top three CONUS layers (top 20 cm of soil).
- **Silt content—top 20 cm (weight percent)**. An area-weighted average silt content for surface soils was derived from STATSGO data for each region and soil texture in the same manner described for USLE erodibility factor.

The USLE's length slope factor (*LS*) was derived from STATSGO slope data. Percent slope was obtained by region and soil texture using only the map units with the predominant texture of interest. An area-weighted average slope was calculated for each texture occurring in a region. Length (ft) was then obtained from a Lightle and Weesies (1998) lookup table of default flow lengths by slope, using slope values rounded to the nearest integer (Table E-6). All slopes less than 0.5 were given the length corresponding to 0.5, and all slopes greater than 24 were given the length corresponding to 24. The USLE length/slope factor *LS* (unitless) was then calculated using the following equation from Williams and Berndt (1977):

$$LS = (L/72.6)^{m} (0.065 + 0.0454S + 0.0065S2) \qquad \text{(E-2)}$$

Where

L = Flow length
S = Slope in percent
and
m = 0.2 for slope <1%
m = 0.3 for slope ≥1% and <3%
m = 0.4 for slope ≥3% and <5%
m = 0.5 for slope ≥5%

Table E-6. Default Flow Lengths by Slope

Slope	Length (ft)	Slope	Length (ft)
≤0.5	100	13	90
1	200	14	80
2	300	15	70
3	200	16	60
4	180	17	60
5	160	18	50
6	150	19	50
7	140	20	50
8	130	21	50
9	125	22	50
10	120	23	50
11	110	≥24	50
12	100		

Source: Lightle and Weesies (1998)

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Appendix E

Attachment E-A: Soil Data

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Table E-A-1. Soil Textures for Meteorological Regions

Meteorological Station (Station number)	
Soil Texture	**Percent of Total Soil**
Albuquerque (23050)	
Clay	12.2
Clay Loam	29.4
Loam	14.1
Sand	12.7
Sandy Clay Loam	0.2
Silty Clay	1.2
Silty Clay Loam	9.7
Silt Loam	0.8
Sandy Loam	19.6
Ashville (03812)	
Clay Loam	5.0
Loam	30.4
Sandy Clay	2.2
Silty Clay Loam	2.6
Silt Loam	44.6
Sandy Loam	15.2
Atlanta (13874)	
Clay	0.9
Loam	3.2
Loamy Sand	46.8
Sand	3
Sandy Clay	0.3
Sandy Clay Loam	0.5
Silt Loam	8
Sandy Loam	36.5
Billings (24033)	
Clay	6.8
Clay Loam	32
Loam	34.9
Sandy Clay Loam	0.4
Silty Clay	3.3
Silty Clay Loam	9.1
Silt Loam	9.6
Sandy Loam	3.8
Bismarck (24011)	
Clay	3.5
Clay Loam	4
Loam	56.1
Sand	3.5
Silty Clay	5.3
Silty Clay Loam	7.8
Silt Loam	9.6
Sandy Loam	10.2
Boise (24131)	
Clay	0.3
Clay Loam	1.9
Loam	12.5
Loamy Sand	0.5
Sand	3.7
Silty Clay Loam	1.9
Silt Loam	67.6
Sandy Loam	11.6
Boulder (94018)	
Clay	2.3
Clay Loam	6.3
Loam	20.8
Loamy Sand	2.2
Sand	5.6
Silty Clay	0.3
Silty Clay Loam	7.3
Silt Loam	37.2
Sandy Loam	17.5
Burlington (14742)	
Clay	7.8
Loam	11.4
Loamy Sand	5.3
Sand	1.8
Silty Clay	2.9
Silty Clay Loam	11.3
Silt Loam	20.7
Sandy Loam	38.7
Caspar (24089)	
Clay	5.2
Clay Loam	10.5
Loam	31
Loamy Sand	3.8
Sand	7.4
Sandy Clay Loam	2.6
Silty Clay	1.6
Silty Clay Loam	3.1
Silt Loam	13
Sandy Loam	21.8

Table E-A-1. Soil Textures for Meteorological Regions (cont'd)

Meteorological Station (Station number)		Meteorological Station (Station number)	
Soil Texture	**Percent of Total Soil**	**Soil Texture**	**Percent of Total Soil**
Charleston (13880)		**Harrisburg (14751)**	
Clay	0.3	Clay Loam	3.4
Loam	2.2	Loam	14.1
Loamy Sand	24	Silty Clay	0.1
Sand	45.6	Silty Clay Loam	4.4
Silty Clay Loam	0.4	Silt Loam	71.5
Silt Loam	1	Sandy Loam	6.2
Sandy Loam	26.5		
		Hartford (14740)	
Chicago (94846)		Loam	10
Clay Loam	0.5	Loamy Sand	3.4
Loam	5.5	Sand	1.6
Loamy Sand	0.8	Silt Loam	44.5
Sand	3.6	Sandy Loam	40.3
Silty Clay Loam	10.3		
Silt Loam	75.9	**Houston (12960)**	
Sandy Loam	3.3	Clay	33.8
		Clay Loam	8.3
Cleveland (14820)		Loam	3
Clay	10	Loamy Sand	0.3
Loam	7.8	Sand	10.1
Loamy Sand	1.2	Sandy Clay Loam	4.6
Sand	1.1	Silty Clay	2.6
Silty Clay	1.6	Silty Clay Loam	1.3
Silty Clay Loam	3.6	Silt Loam	6.8
Silt Loam	70.8	Sandy Loam	29.2
Sandy Loam	3.5		
		Huntington (03860)	
Fresno (93193)		Loam	1.3
Clay	22.3	Silty Clay	7.5
Clay Loam	14.8	Silty Clay Loam	5.9
Loam	11.7	Silt Loam	84.8
Sand	1.2	Sandy Loam	0.3
Silty Clay Loam	8.1		
Silt Loam	5.5	**Las Vegas (23169)**	
Sandy Loam	35.7	Clay	16.4
		Loam	11.4
Grand Island (14935)		Loamy Sand	40.5
Clay Loam	6.3	Sand	10.1
Loam	13.4	Silty Clay	1
Loamy Sand	0.3	Silt Loam	0.8
Sand	4.4	Sandy Loam	19.7
Silty Clay	2.9		
Silty Clay Loam	26.7	**Little Rock (13963)**	
Silt Loam	43.8	Clay	13.6
Sandy Loam	2.1	Loam	2.7
		Silty Clay	9.5
		Silty Clay Loam	7.8
		Silt Loam	56.5

Table E-A-1. Soil Textures for Meteorological Regions (cont'd)

Meteorological Station (Station number)		Meteorological Station (Station number)	
Soil Texture	Percent of Total Soil	Soil Texture	Percent of Total Soil
		Sandy Loam	9.9
Los Angeles (23174)		**Nashville (13897)**	
Clay	2.6	Clay	0.5
Clay Loam	2.4	Loam	3.7
Loam	10.7	Sand	0.4
Loamy Sand	14	Silty Clay	2.7
Sand	4.5	Silty Clay Loam	4.2
Silty Clay Loam	0.9	Silt Loam	85.4
Silt Loam	2.9	Sandy Loam	3
Sandy Loam	61.5		
		New Orleans (12916)	
Meridian (13865)		Clay	8.3
Clay	8.6	Loam	1
Loam	3.9	Loamy Sand	10.7
Loamy Sand	1.5	Sand	2.4
Silt	0.3	Silty Clay Loam	4.7
Silty Clay	7.4	Silt Loam	29.5
Silty Clay Loam	4.3	Sandy Loam	43.4
Silt Loam	40.7		
Sandy Loam	33.3	**Norfolk (13737)**	
		Loam	7.6
Miami (12839)		Loamy Sand	10.2
Clay Loam	0.1	Sand	4.8
Loam	4.3	Silty Clay Loam	0.6
Loamy Sand	1.5	Silt Loam	14.1
Sand	93.7	Sandy Loam	62.6
Silt Loam	0.4		
		Philadelphia (13739)	
Minneapolis (14922)		Loam	22.8
Clay	0.7	Sand	3
Clay Loam	11.2	Silt Loam	63.5
Loam	32.3	Sandy Loam	10.5
Loamy Sand	2.9		
Sand	3.2	**Phoenix (23183)**	
Sandy Clay Loam	0.6	Clay	5.5
Silty Clay	1	Clay Loam	10.2
Silty Clay Loam	13.9	Loam	26.2
Silt Loam	20.8	Sand	0.6
Sandy Loam	13.5	Sandy Clay Loam	1.3
		Silty Clay Loam	25.6
Muskegon (14840)		Silt Loam	1.4
Clay	0.3	Sandy Loam	29
Clay Loam	0.5		
Loam	34.3	**Portland (14764)**	
Loamy Sand	11.7	Loam	19.2
Sand	7.3	Loamy Sand	5.7
Silty Clay Loam	2.3	Silt Loam	44.2
Silt Loam	26.1	Sandy Loam	30.9
Sandy Loam	17.5		

Table E-A-1. Soil Textures for Meteorological Regions (cont'd)

Meteorological Station (Station number) Soil Texture	Percent of Total Soil
Raleigh-Durham (13722	
Loam	19.4
Loamy Sand	18.5
Sand	11.5
Sandy Clay	2.2
Silty Clay Loam	1.5
Silt Loam	13.9
Sandy Loam	32.7
Salem (24232)	
Clay Loam	3.5
Loam	18.3
Loamy Sand	0.3
Sand	3
Silt	1
Silty Clay	0.6
Silty Clay Loam	31.4
Silt Loam	34.1
Sandy Loam	7.8
Salt Lake City (24127)	
Clay Loam	2.8
Loam	30
Sand	0.9
Silty Clay Loam	8.2
Silt Loam	47.9
Sandy Loam	9.3
San Francisco (23234)	
Clay	20.1
Clay Loam	17.1
Loam	33.7
Loamy Sand	1.7
Sand	3.3
Silty Clay Loam	6
Silt Loam	8.6
Sandy Loam	9.5
Seattle (24233)	
Loam	11.9
Loamy Sand	1.5
Sand	1.1
Silty Clay Loam	5.6
Silt Loam	52.4
Sandy Loam	27.4
Shreveport (13957)	
Clay	29
Clay Loam	5
Loam	4.4
Sand	6.5
Silty Clay	3
Silty Clay Loam	3.9
Silt Loam	14.2
Sandy Loam	34
Tampa (12842)	
Loamy Sand	25.9
Sand	73
Sandy Loam	1.1
Tulsa (13968)	
Clay	1.9
Clay Loam	2.8
Loam	10.3
Sand	5.7
Silty Clay	0.9
Silty Clay Loam	8.8
Silt Loam	51.5
Sandy Loam	18.2
Williamsport (14778)	
Loam	11.8
Silt Loam	86.1
Sandy Loam	1.4
Winnemucca (24128)	
Clay	4.2
Clay Loam	6.5
Loam	24.2
Loamy Sand	1.8
Sand	1.7
Silty Clay	5.1
Silty Clay Loam	8.2
Silt Loam	20.6
Sandy Loam	27.6

Appendix F

Chemical Data

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Table F-1. Chemical Parameters for Antimony (7440-36-0)
Ecological Assessment

Parameter	Description	Value	Reference
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	6.6 E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	2.66E-5	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 1.26., max 501, mean 200, stdev 12.6)	U.S. EPA, 2005
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	1.2E+02	U.S. EPA, 2008
Ecological Benchmark			
Eco-SSL	EPA Soil Screening Level (mg kg^{-1}soil)	Soil Biota: 78 Mammals: 0.27	U.S. EPA, 2014a

Table F-2. Chemical Parameters for Arsenic (7440-38-2)
Human Health Soil/Produce Assessment

Parameter	Description	Value	Reference
Biotransfer Factors			
BrExfruit	Soil-to-plant bioconcentration factor, exposed fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	2.00E-03	Calculated based on U.S. EPA, 1999
BrExveg	Soil-to-plant bioconcentration factor, exposed vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	1.00E-02	Calculated based on U.S. EPA, 1999
BrProfruit	Soil-to-plant bioconcentration factor, protected fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	2.00E-03	Calculated based on U.S. EPA, 1999
BrProveg	Soil-to-plant bioconcentration factor, protected vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	2.00E-03	Calculated based on U.S. EPA, 1999
BrRoot	Soil-to-plant bioconcentration factor, root vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	4.60E-03	Calculated based on U.S. EPA, 1999
KpPar	Plant surface loss coefficient, particulate (1 yr^{-1})	1.81E+01	U.S. EPA, 1997
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	5.73E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	3.25E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 2, max 19953, mean 1585, stdev 5)	U.S. EPA, 2005
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	7.49E+01	U.S. EPA, 2008
Human Health Benchmark			
CSF	Cancer Slope Factor (mg kg^{-1}-d^{-1})$^{-1}$	1.50E+00	U.S. EPA, 2012
RfD	Reference Dose (mg kg^{-1}-d^{-1})	3.00E-04	U.S. EPA, 2012

Table F-3. Chemical Parameters for Chromium III (16065-83-1)
Ecological Assessment

Parameter	Description	Value	Reference
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	7.1 E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	4.6E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 10, max 50,119, mean 6310, stdev 2.5)	U.S. EPA, 2005
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	5.1E+01	U.S. EPA, 2008
Ecological Benchmark			
Eco-SSL	EPA Soil Screening Level (mg kg^{-1}soil)	Mammals: 34	U.S. EPA, 2014a

Table F-4. Chemical Parameters for Cobalt (7440-48-4)
Human Health Soil/Produce Assessment

Parameter	Description	Value	Reference
Biotransfer Factors			
BrExfruit	Soil-to-plant bioconcentration factor, exposed fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	7.0E-03	Baes et al., 1984
BrExveg	Soil-to-plant bioconcentration factor, exposed vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	2.0E-02	Baes et al., 1984
BrProfruit	Soil-to-plant bioconcentration factor, protected fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	7.0E-03	Baes et al., 1984
BrProveg	Soil-to-plant bioconcentration factor, protected vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	7.0E-03	Baes et al., 1984
BrRoot	Soil-to-plant bioconcentration factor, root vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	2.0E-02	Baes et al., 1984
KpPar	Plant surface loss coefficient, particulate (1 yr^{-1})	1.81E+01	U.S. EPA, 1997
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	8.8E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	4.89E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 0.06, max 12,589, mean 126, stdev 15.8)	U.S. EPA, 2005
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	5.8E+01	U.S. EPA, 2008
Human Health Benchmark			
RfD	Reference Dose (mg kg^{-1}-d^{-1})	3.00E-04	U.S. EPA, 2014b

Table F-65. Chemical Parameters for Copper (7440-50-8)
Ecological Assessment

Parameter	Description	Value	Reference
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	8.9E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^{2} s^{-1})	4.68E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 1.26, max 3981, mean 316, stdev 4.0)	U.S. EPA, 2005 (from literature data)
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	6.3E+01	U.S. EPA, 2008
Ecological Benchmark			
Eco-SSL	EPA Soil Screening Level (mg kg^{-1}soil)	Terr. Plants: 70 Soil Biota: 80 Mammals: 49	U.S. EPA, 2014a

Table F-3. Chemical Parameters for Iron (7439-89-6)
Human Health Soil/Produce Assessment

Parameter	Description	Value	Reference
Biotransfer Factors			
BrExfruit	Soil-to-plant bioconcentration factor, exposed fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	1.00E-03	Baes et al., 1984
BrExveg	Soil-to-plant bioconcentration factor, exposed vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	4.00E-03	Baes et al., 1984
BrProfruit	Soil-to-plant bioconcentration factor, protected fruit (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	1.00E-03	Baes et al., 1984
BrProveg	Soil-to-plant bioconcentration factor, protected vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	1.00E-03	Baes et al., 1984
BrRoot	Soil-to-plant bioconcentration factor, root vegetables (mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$	4.00E-03	Baes et al., 1984
KpPar	Plant surface loss coefficient, particulate (1 yr^{-1})	1.81E+01	U.S. EPA, 1997
Chemical Properties			
Density	Density of the chemical (g mL^{-1})	7.8E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^{2} s^{-1})	4.68E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	25	Baes et al., 1984
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	5.5E+01	U.S. EPA, 2008
Human Health Benchmark			
RfD	Reference Dose (mg kg^{-1}-d^{-1})	7.0E-01	U.S. EPA, 2012

Table F-3. Chemical Parameters for Manganese (7439-96-5)
Ecological Assessment

Parameter	Description	Value	Reference
Density	Density of the chemical (g mL^{-1})	7.3E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	4.48E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution Kd values (min 251, max 50,119, mean 1585, stdev 5.0)	Allison, 2003
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	5.4E+01	U.S. EPA, 2008
Ecological Benchmark			
Eco-SSL	EPA Soil Screening Level (mg kg^{-1}soil)	Terr. Plants: 220 Soil Biota: 450 Mammals: 4000	U.S. EPA, 2014a

Table F-4. Chemical Parameters for Nickel (7440-02-0)
Ecological Assessment

Parameter	Description	Value	Reference
Density	Density of the chemical (g mL^{-1})	8.9E+00	U.S. EPA, 2008
Dw	Diffusion coefficient in water (cm^2 s^{-1})	4.90E-05	Calculated based on U.S. EPA, 2001
Kd	Soil-water partition coefficient (L Kg^{-1})	Lognormal distribution – Kd values (min 10, max 794, mean 6310, stdev 3.2)	U.S. EPA, 2005
ksoil	Degradation rate for soil (1 day^{-1})	0.00E+00	Set to zero for metals
MW	Molecular weight (g mol^{-1})	5.8E+01	U.S. EPA, 2008
Ecological Benchmark			
Eco-SSL	EPA Soil Screening Level (mg kg^{-1}soil)	Terr. Plants: 38 Soil Biota: 280 Mammals: 130	U.S. EPA, 2014a

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Appendix G

Home Garden Source Model

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Appendix G
Home Garden Source Model

G.1 Introduction

For the home gardening scenario, the Land Application Unit (LAU) Module from the Multimedia, Multipathway, Multi-receptor Risk Assessment (3MRA) modeling system was used to predict the release of metals from SFS-manufactured soil used in a home garden. In support of EPA's 503 program, the LAU Module has been modified to simulate chemical losses from farm fields (rather than land application units) that apply biosolids as a soil amendment. This appendix describes the modified LAU Module[1] (henceforth referred to as the home garden source model [HGSM]) that was used, primarily, to estimate annual average concentrations of metals in garden soil based on the predicted losses associated with various environmental processes (e.g., overland runoff, particulate emissions, leaching).

The HGSM is based on the Generic Soil Column Model (GSCM), a generalized solution that was developed to simulate the dynamic changes in chemical constituent mass fate and transport within the field and near-surface soils in watershed subareas. Governing equations for the GSCM are similar to those used by Jury and colleagues (1983 and 1990) and Shan and Stephens (1995). However, the analytical solution techniques used by these researchers were not applicable to the source emission module developed here because they did not consider constituent mass loss rates in the surface soil from runoff, wind and water erosion, leaching, and mechanical processes.

Section G.2 describes the assumptions, governing equations, boundary conditions, and solution technique that were originally developed as the GSCM. Section G.3 describes the implementation of the HGSM to the garden scenario; specifically, how the GSCM and various components (e.g., hydrology, soil erosion, and runoff water quality) are integrated with the local and regional watersheds. Additional details are included in three attachments: **Attachment A** lists and defines all symbols used in Sections G.2 and G.3; **Attachment B** provides supplementary information on particulate emission equations; and **Attachment C** presents the HGSM input parameters used in the SFS analysis. **Attachment D** describes the modeling that was performed to estimate the location-specific dispersion and deposition factors originally generated to support EPA's biosolids evaluation. The subset of these factors relevant to SFS economic feasibility areas were mapped to and applied in modeling of the SFS gardening scenario. **Attachment E** describes the soil Kd evaluation that was performed to examine the impact of Kd distributions on SFS screening levels.

G.2 Generic Soil Column Model

G.2.1 Assumptions

The GSCM includes the following assumptions:

- The contaminant partitions to three phases: adsorbed (solid), dissolved (liquid), and

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[1] The information presented in this appendix on the LAU model is based on U.S. EPA, 1999.

gaseous (as in Jury et al., 1983 and 1990).

$$C_T = \rho_b C_s + \theta_w C_L + \theta_a C_G \qquad \text{(G-1)}$$

where

C_T = Total contaminant concentration in soil (g m^{-3} of soil)
ρ_b = Soil dry bulk density (kg m^{-3})
C_s = Adsorbed-phase contaminant concentration in soil (g kg^{-1} of dry soil)
θ_w = Soil volumetric water content (m^3 soil water m^{-3} soil)
C_L = Aqueous-phase contaminant concentration in soil (g m^{-3} of soil water)
θ_a = Soil volumetric air content (m^3 soil air m^{-3} soil)
C_G = Gas-phase contaminant concentration in soil (g m^{-3} of soil air)

- The contaminant undergoes reversible, linear equilibrium partitioning between the adsorbed and dissolved phases (as in Jury et al., 1983 and 1990),

$$C_S = K_d C_L \qquad \text{(G-2)}$$

where K_d is the linear equilibrium partitioning coefficient (m^3 kg^{-1}). For inorganic contaminants K_d is a specified input parameter.[2] For organic contaminants,

$$K_d = foc \cdot K_{oc} \qquad \text{(G-3)}$$

where *foc* is the fraction organic carbon in soil and K_{oc} is the equilibrium partition coefficient (m^3 kg^{-1}), normalized to organic carbon.

- The contaminant is in equilibrium between the dissolved and gaseous phases, and follows Henry's law (as in Jury et al., 1983 and 1990),

$$C_G = H'C_L \qquad \text{(G-4)}$$

where $\acute{H}$ is the dimensionless Henry's law constant.

- The total contaminant concentration in soil can also be expressed in units of µg of contaminant mass per g of dry soil (µg g^{-1}):

$$C'_T = \frac{C_T}{\rho_b} \qquad \text{(G-5)}$$

- Using the linear equilibrium approximations in Equations G-2 through G-5, C_T can be expressed in terms of C_L, C_S , or C_G:

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[2] Linear equilibrium partitioning assumes that the sorptive capacity of the soil column solids does not become exhausted.

$$C_T = K_{TL} C_L = \frac{K_{TL}}{K_d} C_s = \frac{K_{TL}}{H'} C_G \tag{G-6}$$

where

$$K_{TL} = \rho_b K_d + \theta_w + \theta_a H' \tag{G-7}$$

K_{TL} is the dimensionless equilibrium distribution coefficient between the total and aqueous-phase constituent concentrations in soil.

- The total water flux or infiltration rate (I, m d^{-1}) is constant in space and time (as in Jury et al., 1983 and 1990) and ≥ 0. It is specified as an annual average.
- The soil column is an unconsolidated, homogeneous, and porous medium whose basic properties (ρ_b, *foc*, θ_w, θ_a, and η—the total soil porosity) are average annual values, constant in space.
- Contaminant mass may be lost from the soil column as a result of one or more first-order loss processes.
- The total chemical flux is the sum of the vapor flux and the flux of the dissolved solute (as in Jury et al., 1983 and 1990).
- The chemical is transported in one dimension through the soil column (as in Jury et al., 1983 and 1990).
- The vapor-phase and liquid-phase porosity and tortuosity factors obey the model of Millington and Quirk (1961) (as in Jury et al., 1983 and 1990) (see Equation G-9 below).
- The modeled spatial domain of the soil column remains constant in volume and fixed in space with respect to a vertical reference (e.g., the water table).

G.2.2 Governing Mass Balance Equation

Under the previously mentioned assumptions, the governing mass fate and transport equation can be written as follows:

$$\frac{\partial C_T}{\partial t} = D_E \frac{\partial^2 C_T}{\partial z^2} - V_E \frac{\partial C_T}{\partial z} - kC_T \tag{G-8}$$

Where *k* (1 d^{-1}) is the total first-order loss rate and D_E (m^2 d^{-1}) is the effective diffusivity in soil calculated as follows:

$$D_E = \frac{\left(\theta_a^{10/3} D_a H' + \theta_w^{10/3} D_w\right) 8.64}{\eta^2 K_{TL}} \tag{G-9}$$

Where D_a and D_w (cm^2 s^{-1}) are air and water diffusivities, respectively, and 8.64 is a conversion factor ((m^2-s) (cm^2-d)$^{-1}$). D_E is the sum of the effective gaseous and water diffusion coefficients in soil, $D_{E,a}$, and $D_{E,w}$, respectively, where

$$D_{E,a} = \frac{\theta_a^{10/3} D_a H' 8.64}{\eta^2 K_{TL}} \tag{G-10}$$

and

$$D_{E,w} = \frac{\theta_w^{10/3} D_w 8.64}{\eta^2 K_{TL}} \tag{G-11}$$

The effective solute convection velocity (V_E, m d^{-1}) is equal to the water flux corrected for the contaminant partitioning to the water phase as follows:

$$V_E = \frac{I}{K_{TL}} \tag{G-12}$$

G.2.3 Parameter Estimation Methodologies

Water content (θw) is estimated as a function of the annual average infiltration rate (I, m d^{-1}) using Equation G-13, from Clapp and Hornberger (1978):

$$\theta_w = \eta \cdot \left(\frac{I}{0.24 K_{sat}} \right)^{1/(2SM_b+3)} \tag{G-13}$$

Where K_{sat} (cm h^{-1}) is saturated hydraulic conductivity, SM_b is a unitless exponent specified by soil-type, and 0.24 ((m-h) $(\text{cm-d})^{-1}$) is a unit conversion factor.

Volumetric air content is estimated using Equation G-14:

$$\theta = \eta - \theta_w \tag{G-14}$$

Ή, Da, and Dw are either estimated as a function of temperature in the soil column (Tsc, °C) or specified directly as input parameters if pre-adjusted values are available.

G.2.4 Solution Technique

G.2.4.1 Background

The governing equation (Equation G-8) was solved to evaluate the following in a soil column of depth z_{sc},

- Total contaminant concentration as a function of time, t, and depth below the surface, z; and
- Contaminant mass fluxes across the upper (z=0) and lower boundaries ($z=z_{sc}$) of the soil column.

A numerical solution of Equation G-8, with zero concentration at the soil surface and zero release at the bottom of the soil column, was first examined using a straightforward explicit finite difference approach. This approach resulted in such a high numerical diffusion that it was impossible to distinguish diffusion effects. Subdividing each section into relatively thinner sections reduced the numerical diffusion to more manageable levels, but also required smaller time steps, and the computation time became quite long. In addition, the numerical solution was not stable in extreme situations (e.g., high/low V_E or D_E).

An alternative solution was developed using a quasi-analytical approach. The quasi-analytical solution allows for relative computational speed and significantly reduces concern about numerical diffusion and lack of stability. The tradeoff is a loss of ability to evaluate short-term trends in concentration and diffusive flux profiles. The alternative solution estimates long-term (i.e., annual average) contaminant concentration profiles and mass fluxes.

The alternative solution consists of a superposition of analytic solutions of the three components of the governing equation (Equation G-8) on the same grid. The solution for a simplified case where the soil column consists of one homogeneous zone whose properties are uniform in space and time is described below. Adaptations of the solution technique to account for variations from this simplified case are described in the module-specific sections.

G.2.4.2 Description of Quasi-Analytical Approach

The quasi-analytical approach is a step-wise solution of the three components of the governing equation (Equation G-8) on the same grid. That is, the following equations are solved individually:

$$\frac{\partial C_T}{\partial t} = D_E \frac{\partial^2 C_T}{\partial z^2} \tag{G-15}$$

$$\frac{\partial C_T}{\partial t} = -V_E \frac{\partial C_T}{\partial z} \tag{G-16}$$

$$\frac{\partial C_T}{\partial t} = -kC_T \tag{G-17}$$

Boundary conditions of $C_T=0$ at both the upper and lower boundaries of the soil column are assumed, although some flexibility exists for specifying the lower boundary condition, as discussed below.

Equations G-15 through G-17 each have an analytical solution that can be combined to obtain a pure diffusion solution that moves with velocity V_E through the porous medium (Jost, 1960). The solution of the general differential equation is then the solution of the diffusive portion with its time dependence, translating in space with velocity V_E, and decaying exponentially with time.

The first two solutions for a point source are graphically illustrated in Figures G-1 and G-2. If it were possible to compute such point source solutions for each position in the soil column and each time of interest, then the governing differential equations would be linear and the contributions at each point could be added to obtain a global solution. That is, each point in the soil column could be treated as if it were the only point for which there is a nonzero concentration.

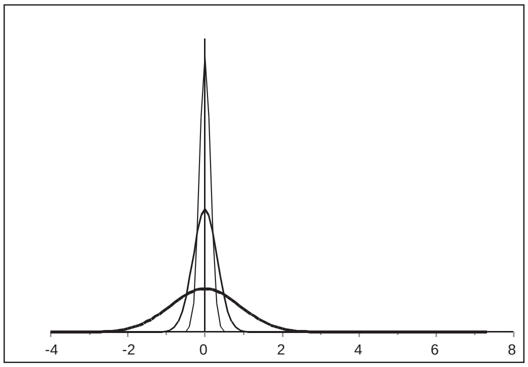

Figure G-1. Diffusive spreading from a point source with time, at times 0.01, 0.05, and 0.4.

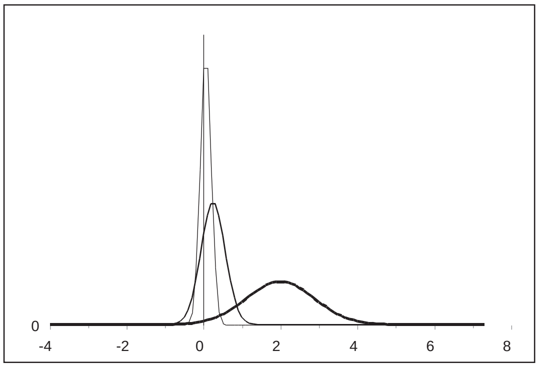

Figure G-2. Diffusive spreading from a point source with a constant velocity to the right, at times 0.01, 0.05, and 0.4.

To make the analysis tractable, instead of a point source, the soil column is divided into layer sources each of depth dz (i.e., a grid). A layer source can be thought of as multiple point sources packed closely together. In such a case, Equation G-15 has a solution for one-dimensional diffusion, with the concentration at any point and any time given by Equation G-18 for a layer of width dz centered at z'=0 (Jost, 1960):

$$C_T(z',t) = \frac{C_{T0}}{2}\left[erf\left(\frac{z'+dz/2}{\sqrt{4D_E t}}\right) + erf\left(\frac{dz/2-z'}{\sqrt{4D_E t}}\right)\right] \tag{G-18}$$

The concentration profile is assumed to be initially uniform from z'=−dz/2 to z'=+dz/2 and zero everywhere else. With time, the profile spreads outward and the concentration at the origin decreases, as shown in Figure G-3 for dz=2. The concentration profile also moves down through the soil column at velocity V_E, as illustrated in Figure G-4. Layer solutions assume uniform average concentrations within each layer. Thus, the thickness of the layers determines the spatial resolution available.

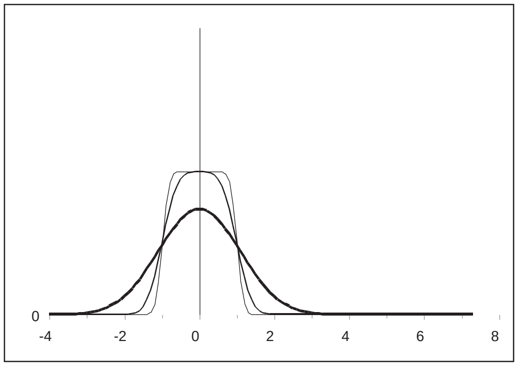

Figure G-3. Diffusive spreading from a layer source with time, at times 0.01, 0.05, and 0.4.

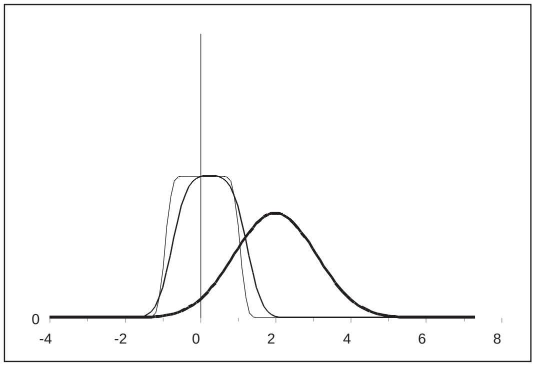

Figure G-4. Diffusive spreading from a layer source with a constant velocity to the right, at times 0.01, 0.05, and 0.4.

The total amount of material, m (g m^{-2}), that has passed any ordinate z' after time, t, is given by the integral of the concentration from z' to ∞ with one-half leaving to the left (negative z' values) and one-half to the right (positive z' values):

$$m(z',t) = 2\int_{z'}^{\infty} C_T(z,t)dz \tag{G-19}$$

Deriving the integral in Equation G-19 results in

$$m(z',t) = C_{T0}\sqrt{4D_E t}\left[\int_{(z'-dz/2)/\sqrt{4D_E t}}^{\infty} erfc(y)dy - \int_{(z'+dz/2)/\sqrt{4D_E t}}^{\infty} erfc(y)dy\right] \tag{G-20}$$

which is evaluated using the relationship from Abramowitz and Stegun (1970):

$$\int \mathrm{erfc}(x)dx = x\,\mathrm{erfc}(x) - \frac{1}{\sqrt{\pi}}\exp\left(-x^2\right) + \text{constant} \tag{G-21}$$

The fraction of the original mass that diffuses past a boundary at z' in any time period 0 to t, Df(z',t) (g m^{-2}), is one-half m(z',t) divided by the amount of mass initially present in the source layer (C_{T0}·dz):

$$Df(z',t) = 0.5\cdot\frac{\sqrt{4D_E t}}{dz}\left[\int_{(z'-dz/2)/\sqrt{4D_E t}}^{\infty} erfc(y)dy - \int_{(z'+dz/2)/\sqrt{4D_E t}}^{\infty} erfc(y)dy\right] \tag{G-22}$$

The fraction of mass that remains in the original layer of width dz after diffusion in the time period 0 to t, $Df_0(t)$, is

$$Df_0(t) = 1 - 2\cdot Df(z' = 0.5dz, t) \tag{G-23}$$

By evaluating all the layer boundaries (z'=0.5 dz, 1.5 dz, 2.5 dz, ...), the amount of contaminant mass transported to any layer via diffusion after time, t, can be calculated as the difference between the amount outside the upstream boundary and the amount outside the downstream boundary. For example, the fraction of mass originally present in the source layer that ends up in the layer adjacent to the source layer in time, t, is Df(z'=0.5 dz, t) -Df(z'=1.5 dz, t). The integrated amount of material that has crossed into the adjacent layer and the amount that remains in the source layer after time, t, are given directly by Equations G-22 and G-23, respectively, and only have to be computed once for fixed time steps and layer thicknesses.

The amount of mass that diffuses from a given layer out the lower boundary of the soil column in time, t, can be tracked by multiplying Df(z',t)—evaluated where z' is at the bottom of the soil column ($z=z_{sc}$)—by (C_{T0} ·dz) for that layer. Diffusive losses across the bottom boundary from all the soil column layers are summed to calculate the total diffusive (aqueous- and gaseous-phase) loss across the bottom boundary, $M_{lchd}(t)$ (g m^{-2}), in time, t.

Likewise, the total diffusive loss out the top of the soil column, $M_0(t)$ (g m^{-2}), is the sum of the total diffusive losses across the upper boundary from each layer. The volatilization loss from the surface of the soil column, $M_{vol}(t)$ (g m^{-2}), is assumed to be from gaseous-phase diffusion only and is determined by

$$M_{vol}(t) = M_0(t) \cdot \frac{D_{E,a}}{D_E} \tag{G-24}$$

where $(D_{E,a}/D_E)$ is the fraction of the total diffusive loss from any layer that is due to diffusion in the gaseous phase in the soil.

It is assumed that mass is not lost across the top soil boundary due to diffusion from the aqueous phase. To maintain mass balance, mass calculated to be lost this way is added back into the top soil layer, augmenting the total contaminant concentration there by ($M_0(t)\ D_{E,w}/D_E$). This method of approximating $M_{vol}(t)$ is justified on the basis of computational efficiency. A more rigorous treatment would include a mathematical transition layer across which diffusion from the soil to the air occurs. However, use of such a transition layer would require a more computationally intensive solution technique, as well as specification of the thickness of the transition layer.

Without this approximation (i.e., if $M_{vol}(t)=M_0(t)$), $M_{vol}(t)$ could be >0 for nonvolatile contaminants ($D_a=H'=0$) because of the possible contribution to M_0 from the aqueous-phase diffusive flux. Estimating $M_{vol}(t)$ and augmenting the total contaminant concentration in the surface layer is considered a reasonable approximation of what actually occurs. That is, contaminant mass diffuses to the surface in both the aqueous and gaseous phases. While the contaminant mass in the gas phase volatilizes out the surface of the soil column, the contaminant mass in the aqueous phase is left behind, concentrating the contaminant mass in surface soil (approximated here as the surface soil column layer).

To account for decay, Equation G-17 is solved using the technique of separation of variables (Jost, 1960). The solution takes the form

$$C_T = C_{T0} \exp(-kt) \tag{G-25}$$

As Equation G-25 is applied to each layer, the amount of mass lost as a result of first-order decay in time, t, M_{loss} (g m^{-2}), can be tracked using

$$M_{loss}(t) = (1 - \exp(-kt))C_{T0} \cdot dz \tag{G-26}$$

If multiple first-order loss processes occur (i.e., $k=\sum k_j$), the fraction of initial mass lost as a result of each process j is determined using the following equation:

$$M_{loss,j}(t) = \frac{k_j}{k} M_{loss}(t) \tag{G-27}$$

A potential difficulty with the layer solution is that the convection of material leads to an artificial numerical diffusion because the concentration within each layer can only be expressed as an average value. This component of numerical diffusion can be avoided completely if the contents of each layer are transferred completely to the next layer at the end of each time step by making the time step equal to the layer thickness divided by the effective velocity, V_E:

$$dt = \frac{dz}{V_E} \tag{G-28}$$

The contaminant mass in the bottom layer is convected out of the lower boundary. Total mass lost due to advection in dt, M_{lcha} (g m^{-2}), is simply C_{T0} in the lowest soil column layer multiplied by dz.

To summarize the overall solution technique, the three processes (diffusion, first-order losses, and advective transport) are considered separately, in series, and then combined (under the justification of the superposition principle for linear differential equations) to result in the chemical concentration vertical profile at the end of a computational time step. Specifically, the chemical concentration profile after diffusion only is simulated first. Next, the chemical mass in each computational cell (the mass after diffusion) is decreased to account for first-order loss. Finally, after sufficient time has elapsed (which may take multiple time steps) for the chemical mass in a cell to advect (at the sorption-corrected velocity) the thickness of the cell, all remaining chemical mass translates to the next lower cell. This completes the series solution of the overall fate and transport governing equation.

G.2.4.2.1 Boundary Conditions

Zero concentration is assumed at the upper boundary of the soil column. This is consistent with the assumption that the air is a sink for volatilized contaminant mass, but requires the approximate method for estimating $M_{vol}(t)$ described above.

At the lower boundary of the soil column, the flexibility exists with this solution technique to specify a value between zero and 1 for the ratio (bcm) of the total contaminant concentration in the soil directly below the modeled soil column and in the soil column. A ratio of one (bcm=1) corresponds to a zero gradient boundary condition ($dC_T/dz=0$). A ratio of zero (bcm=0) corresponds to a zero concentration boundary condition ($C_T=0$).

When bcm is equal to zero, diffusive fluxes at the upper and lower boundaries of the soil column are calculated directly as previously described. When bcm is >0, a reflection of the soil column is created. The contaminant concentrations in the reflected soil column cells are set equal to bcm multiplied by the contaminant concentration in the soil column cell being reflected (i.e., the concentration in the first cell of the reflected soil column is set to bcm multiplied by the contaminant concentration in the lowest cell of the actual soil column). The upward diffusive flux from the reflected soil column cells (1) offsets the diffusive flux out the lower boundary of the soil column, (2) increments the contaminant concentrations in the soil column, and (3) augments the diffusive flux out the upper boundary of the soil column. Hence, when bcm is equal to 1 (the no diffusion boundary condition), the downward diffusive flux out the bottom boundary of the soil column is completely offset by the upward diffusive flux across the same boundary from the reflected soil column cells.

G.2.4.2.2 Algorithm

The general algorithm for applying the individual solutions to Equations G-15 through G-17 is as follows for a homogeneous soil column and an averaging time period of 1 year:

1. Specify
   - Lower boundary condition multiplier (bcm)
   - Initial conditions in soil column (C_{T0})
   - Soil column size (z_{sc}) and properties (ρ_b, foc, η, K_{sat}, SM_b)
   - First-order loss rates (k_j)
   - Chemical properties (K_{oc}, H', D_a, D_w)
   - Upper and lower averaging depths (z_{ava}, z_{avb}).
2. Calculate/read K_d, which is internally calculated for organics ($K_d = K_{oe} \times F_{oc}$), and read as a user input for metals.
3. Subdivide the soil column into multiple layers of depth, dz, that are an integral fraction of z_{sc}. Calculate the total number of layers, $N_{dz} = z_{sc}/dz$.
4. Derive an annual average infiltration rate (I) for the year.
5. Calculate θ_w, θ_a, K_{TL}, D_E, and V_E.
6. Calculate the time to cross a single layer at velocity V_E (Equation G-28). This is the convection-based computing time step, dt (see also the note below).
7. Evaluate the fraction of mass that remains in a layer (Equation G-25) and that diffuses across layer boundaries z'=0.5 dz, 1.5 dz, 2.5 dz,... (Equation G-24) at t = dt. (These fractions are constant for a fixed dt.)
8. Calculate the amount of mass present in the soil column at the beginning of the year (M_{col1}, g m^{-2}).
9. Initialize cumulative mass loss variables (M_{vol}, M_{lchd}, M_{lcha}, and $M_{loss,j}$).
10. Diffusion. Adjust the concentration profile to reflect diffusive fluxes for one time step. This redistributes material throughout the whole soil column. Increment M_{vol} and M_{lchd}.
11. First-order losses. Allow the concentration profile to decay in each layer (Equation G-27) for one time step. Increment mass lost due to all applicable first-order loss processes, j, $M_{loss,j}$ (Equation G-25).
12. Convection. Propagate the concentration profile one layer downstream. Increment M_{lcha}.
13. Repeat Steps 10 through 12 until it is time to add and/or remove contaminant mass (proceed to Step 14) or until the end of the year (proceed to Step 15).
14. To account for the addition of contaminant mass, update the contaminant concentrations in the affected layers. Track total mass added (M_{add}, g m^{-2}) and/or removed (M_{rem}, g m^{-2}). Begin the algorithm again at Step 10.
15. At end of the year, calculate/report

   - Total mass in the soil column (M_{col2}, g m^{-2})
   - Mass balance error for the year (M_{err}, g m^{-2}):

$$M_{err} = M_{col2} - M_{col1} - M_{add} + M_{rem} + M_{vol} + M_{lcha} + M_{lchd} + \sum_j M_{loss,j} \qquad \text{(G-29)}$$

   - Annual average total concentration in surface layer
   - Annual, deptG-weighted average total concentration ($z_{ava} \leq z \leq z_{avb}$)

− Annual average volatilization flux (J_{vol}, g m^{-2} d^{-1}):

$$J_{vol} = \frac{M_{vol}}{365} \tag{G-30}$$

− Annual average leaching flux (J_{lch}, g m^{-2} d^{-1}):

$$J_{lch} = \frac{M_{lchd} + M_{lcha}}{365} \tag{G-31}$$

16. Begin the algorithm again at Step 4 until mass is no longer added to the soil column and mass has been depleted from the soil (i.e., M_{col2}=0).

Note that the convection time step cannot be any greater than the length of time between mass additions or removals (e.g., soil applications). For example, if contaminant mass is added every 30 days, then this is the maximum time step, regardless of how small the velocity is. This limited dt is used to calculate the number of time steps required before convective transfer takes place, and the convective transfer step is performed on an as-needed basis. If the calculated convective time step in the above example is 60 days, then the convective transfer occurs every other time step. Over several steps this results in a temporal distortion of the concentrations within the layers, but the effects average out by the end of the year.

To check the performance of the solution algorithm, Equation G-29 tests if the change in mass in the system over the year is equal to the difference between mass additions and losses. If the mass balance error (M_{err}) is >10^{-8} g m^{-2}, then a message is written to the warning file.

G.3 HGSM Implementation

G.3.1 Introduction

The HGSM provides annual average contaminant mass flux rates from the surface of the field and contaminant mass emission rates due to particulate emissions. To ensure transparency, this report documents all of the major theory, algorithms and functionality implemented in the HGSM, and identifies those used in this evaluation.[3]

The HGSM assumes that the home garden is one component of a broader watershed, and so is affected by runoff and erosion from upslope land areas. The watershed, including the home garden, is referred to as the “local” watershed and is illustrated in Figure G-6. A local watershed is defined as that drainage area that just contains the home garden or a portion thereof (there can be multiple local watersheds) in the lateral (perpendicular to runoff flow) direction, and in which runoff occurs as overland flow (sheet flow) only. This distinguishes it from the “regional” watershed, which is modeled when estimates are needed for downslope soil concentrations and waterbody loadings of chemical constituents that are released from the field. Although the local watershed extends downslope to the point that runoff flows and eroded soil loads would enter a

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[3] For other analyses, the HGSM is also used to predict the delivery of chemical constituents to downslope land areas and waterbodies due to runoff and erosion.

well-defined drainage channel (e.g., a ditch, stream, lake, or some other waterbody), the SFS risk modeling screening did not "track" the constituent loads once released from the field. Rather, the HGSM was used only to predict the metal concentrations in soil after losses (e.g., runoff, erosion) have occurred.

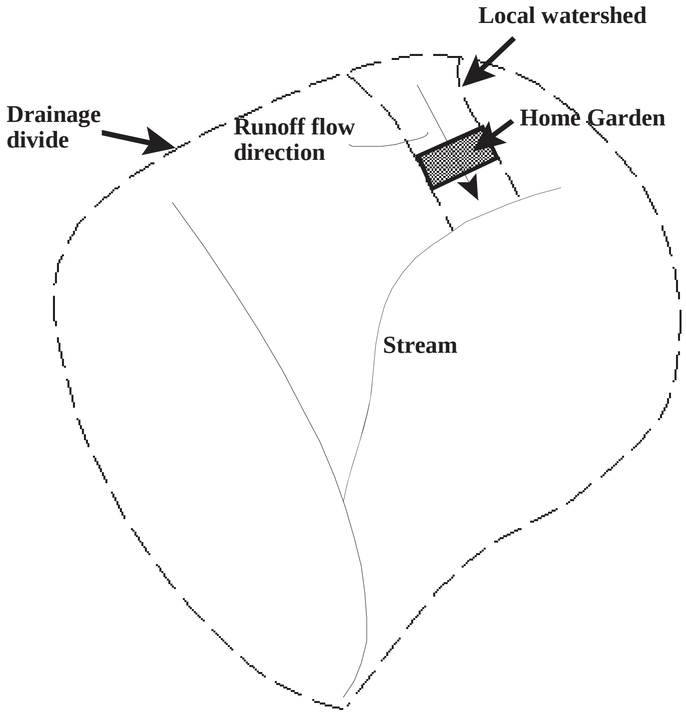

Figure G-5. Regional watershed containing the home garden.

Figures G-6 and G-7 illustrate how the local watershed is conceptualized for the combined Local Watershed/Soil Column Module (i.e., as a two-dimensional, two-medium system. The dimensions are longitudinal (i.e., downslope or in the direction of runoff flow) and vertical (i.e., through the soil column). The media are the soil column and, during runoff events, the overlying runoff water column. In the longitudinal direction, the local watershed is made up of a number of land subareas that may have differing surface or subsurface characteristics (e.g., land uses, soil properties, and chemical concentrations). For example, subarea 2 might be a home garden, subarea 1 an upslope area, and subareas 3 through N downslope buffer areas extending to the waterbody.

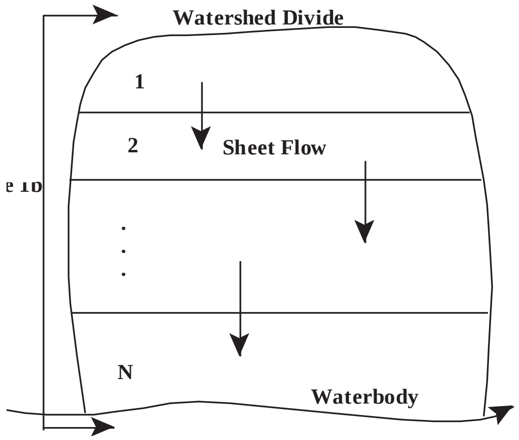

Figure G-6. Local watershed.

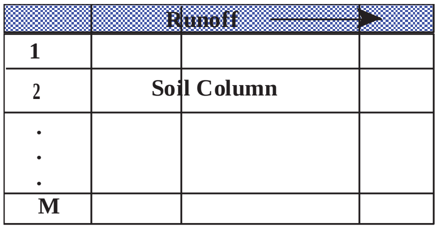

Figure G-7. Cross-sectional view of a watershed for the soil column model.

G.3.2 Hydrology

G.3.2.1 Overview

Hydrologic modeling simulates watershed runoff and groundwater recharge (termed here as “infiltration”). The hydrology module is based on a daily soil moisture water balance performed for the root zone of the soil column. At the end of a given day, t, the soil moisture in the root zone of an arbitrary watershed subarea, i, is estimated as

$$SM_{i,t} = SM_{i,t-1} + P_t + RO_{i-1,t} - RO_{i,t} - ET_{i,t} - IN_{i,t} \tag{G-33}$$

where

$SM_{i,t}$ = Soil moisture (cm) in root zone at end of day t for subarea i
$SM_{i,t-1}$ = Soil moisture (cm) in root zone at end of previous day for subarea i
P_t = Total precipitation (cm) on day t
$RO_{i-1,t}$ = Storm runoff (cm) on day t coming onto subarea i from i-1
$RO_{i,t}$ = Storm runoff (cm) on day t leaving subarea i
$ET_{i,t}$ = Evapotranspiration (cm) from root zone on day t for subarea i
$IN_{i,t}$ = Infiltration (groundwater recharge) on day t (cm) for subarea i

Frozen precipitation is treated as rainfall. Runoff, evapotranspiration, and infiltration losses from the root zone are discussed in subsequent sections. The equations presented in these sections refer to “day t and subarea i” in accordance with the water balance equation (see Equation G-33).

G.3.2.2 Runoff

G.3.2.2.1 Governing Equations

Daily runoff is based on the Soil Conservation Service’s (SCS’s) widely used “curve number” procedure (USDA, 1986) and is a function of current and antecedent precipitation and land use. Land use is considered empirically by the curve numbers, which are catalogued by land use or cover type (e.g., woods, meadow, impervious surfaces), treatment or practice (e.g., contoured, terraced), hydrologic condition, and hydrologic soil group.

Runoff depth is calculated by the SCS procedure as

$$RO = \frac{(P - Ia)^2}{P - Ia + S} \; for\ P \geq Ia \tag{G-34}$$

where

RO = Runoff depth (cm)
P = Precipitation depth (cm)
Ia = Initial abstraction (threshold precipitation depth for runoff to occur) (cm)
S = Watershed storage (cm)

By experimentation with more than 3,000 soil types and cover crops, the SCS developed the following relationships for watershed storage as a function of curve number (CN) and initial abstraction as a function of storage:

$$S = \frac{2540}{CN} - 25.4 \tag{G-35}$$

$$Ia = 0.2S \tag{G-36}$$

Combining Equations G-34 and G-35 results in

$$RO = \frac{(P - 0.2S)^2}{P + 0.8S} \; for\ P \geq 0.2S \tag{G-37}$$

$$RO = 0 \; for\ P \prec 0.2S \tag{G-38}$$

where S is given by Equation G-35. For impervious surfaces (CN=100), it can be observed that RO=P.

Three antecedent moisture classes (AMCs) were used to adjust the SCS curve numbers as shown in Table G-1. The growing season is assumed to be June through August (Julian Day 152 to 243) throughout the country.

Curve numbers are typically presented in the literature, assuming average antecedent moisture conditions (AMC II), and can be adjusted for drier (AMC I) or wetter (AMC III) conditions as (Chow et al., 1988).

Table G-1. Antecedent Moisture Classes for SCS Curve Number Methodology

Antecedent Moisture Class	Total 5-Day Antecedent Rainfall (cm)	
	Dormant Season	Growing Season
I	<1.3	<3.6
II	1.3 to 2.8	3.6 to 5.3
III	>2.8	>5.3

Source: U.S. EPA (1985b)

These adjustments have the effect of increasing runoff under wet antecedent conditions and decreasing runoff under dry antecedent conditions, relative to average conditions.

G.3.2.2.2 Implementation

In the conceptual model for the local watershed (Figure G-6), the subareas may have different land uses and different curve numbers for each subarea. Equation G-37 is nonlinear in the curve number; therefore, the method by which the SCS procedure is applied to multiple subareas can make a significant difference in the resulting cumulative runoff values for downslope subareas. There are essentially two options for implementing the procedure. The first is based on runoff **routing** from each subarea to the next downslope subarea. That is, the runoff depth from subarea 1 would first be calculated from Equation G-37. The cumulative runoff depth from subareas 1 and 2 would then be calculated by applying Equation G-37 to subarea 2 and adding (routing) the runoff depth from subarea 1. This would be repeated for all subareas. This method is **not** appropriate for the sheet flow assumption of the local watershed and can give much higher cumulative runoff depths (volumes) than would actually occur under the sheet flow assumption. (The implicit assumption of the routing method is that the subareas are not hydrologically connected [e.g., runoff from subarea 1 is captured in a drainage system [non-sheet-flow] and diverted directly to the watershed outlet without passing through/over downslope subareas.)

A different, nonrouting method is appropriate for implementing the SCS procedure for the local (sheet flow) watershed. The method is based on determining composite curve numbers and is analogous to the nonsoil routing implementation of the Universal Soil Loss Equation (USLE) soil erosion module presented in Section G.3.3.

$$CN(I) = \frac{4.2CN(II)}{10 - 0.058CN(II)} \tag{G-39}$$

$$CN(III) = \frac{23CN(II)}{10 + 0.13CN(II)} \tag{G-40}$$

The methodology used for implementing this method is illustrated by the following pseudo-code:

```
FOR i=1,...,N (subareas)
    $CNeff_i$=Area-weighted composite $CN_i$ for all subareas j, j=1,...,i
    Calculate $S_i$ from equation (3.2.2-2) using $CNeff_i$
```

```
        Calculate $RO_i$ from equation (3.2.2-1) using $S_i$. ($RO_i$ is the ***average*** runoff depth over all upslope subareas j, j=1,...,i).
        Calculate $Q_i=RO_i \times WSA_i$ where $Q_i$ is cumulative runoff volume and $WSA_i$ is cumulative area.
        IF i=1 THEN
                $H1_i=RO_i$ where $H1_i$ is subarea-specific runoff depth for subarea I (i.e., $RO_i-RO_{i-1}$)
        ELSE
                $H1_i=(Q_i-Q_{i-1})/A_i$ where $A_i$ is subarea-specific surface area
        IF $H1_i<0$ THEN $H1_i=0$
        END IF
NEXT i
```

G.3.2.3 Evapotranspiration

Potential evapotranspiration (PET) is the demand for soil moisture from evaporation and plant transpiration. When soil moisture is abundant, actual ET equals PET. When soil moisture is limiting, ET will be less than PET. The extent to which it is less under limiting conditions has been expressed as a function of PET, available soil water (AW), and available soil water capacity (AWC), as shown in Equation G-41 (Dunne and Leopold, 1978):

$$ET = PET * f\left(\frac{AW}{AWC}\right) \tag{G-41}$$

where

f = A functional relationship of the arguments

and

$$AW = (SM - WP)\frac{DRZ}{100} \tag{G-42}$$

$$AWC = (FC - WP)\frac{DRZ}{100} \tag{G-43}$$

where

WP = Soil wilting point (% volume), which is the minimum soil moisture content that is available to plants. (Plants can exert a maximum suction of approximately 15 atmospheres. The wilting point is that moisture that would not be available at 15 atmospheres.)

DRZ = Depth of the root zone (cm).

FC = Soil field capacity (% volume), which is the maximum soil moisture content that can be held in the soil by capillary or osmotic forces. Soil moisture above the field capacity is readily drained by gravity.

The functional relationship in Equation G-41 is assumed here to be linear, so that ET (cm) is calculated as

$$ET = \min\left[PET\,,PET\left(\frac{SM - WP}{FC - WP}\right)\right] \tag{G-44}$$

PET is estimated as described below.

The more theoretically based modules for daily ET (e.g., the Penman-Monteith equation [Monteith, 1965]) rely on the availability of significant daily meteorological data, including temperature gradient between surface and air, solar radiation, wind speed, and relative humidity. All of these variables may not be readily available for all application sites and, therefore, the less data-demanding Hargreaves equation was used (Shuttleworth, 1993). The Hargreaves method, which is primarily temperature-based, has been shown to provide reasonable estimates of evaporation (Jensen et al., 1990)—presumably because it also includes an implicit link to solar radiation through its latitude parameter (Shuttleworth, 1993).

The Hargreaves equation is

$$PET = 0.0023 S_0 \Delta_T^{0.5} \left(T + 17.8\right) * 0.1 \tag{G-45}$$

where

PET = Potential evapotranspiration (cm d^{-1})
S_0 = water equivalent of extraterrestrial radiation (mm d^{-1}) and is given as (Duffie and Beckman, 1980)
Δ_T = Difference in mean monthly maximum and mean monthly minimum air temperature
T = Mean daily air temperature (°C)

$$S_0 = 15.392 d_r \left(\varpi_s Sin\,\phi\, Sin\,\theta + Cos\,\phi\, Cos\,\theta\, Sin\,\varpi_s\right) \tag{G-46}$$

where

$$d_r = 1 + 0.033 Cos\left(\frac{2\pi}{365} J\right) \tag{G-47}$$

J = Julian day
ϖ_s = Sunset hour angle (radians) given by

$$\varpi_s + Arccos\left(- Tan\,\phi\, Tan\,\theta\right) \tag{G-48}$$

ϕ = Site latitude (positive for northern hemisphere, negative for southern)
θ = Solar declination (radians) given by

$$\theta = 0.4093 Sin\left(\frac{2\pi}{365}J - 1.405\right) \tag{G-49}$$

G.3.2.4 Infiltration (Recharge)

Any soil moisture in excess of the soil's field capacity (FC) that is not used to satisfy ET, is available for gravity drainage from the root zone as infiltration to subroot zones (Dunne and Leopold, 1978). This infiltration rate will, however, be limited by the root zone soil's saturated hydraulic conductivity. Accordingly, infiltration is calculated as

$$IN = \min\left[K_{sat}, (SM - FC)\frac{DRZ}{100}\right] \tag{G-50}$$

where

IN = Infiltration rate (cm d^{-1})
K_{sat}= Saturated hydraulic conductivity (cm d^{-1}).

If infiltration is limited by K_{sat}, the hydrology algorithm includes a feedback loop that increases the previously calculated runoff volume by the amount of excess soil moisture (i.e., the water above the field capacity that exceeds K_{sat}). This adjustment preserves water balance and is based on the assumption that the runoff curve number method, which is only loosely sensitive to soil moisture (through the antecedent precipitation adjustment), has admitted more water into the soil column than can be accommodated by ET, infiltration, and/or increased soil moisture. After the runoff is increased for this excess, the ET, infiltration, and soil moisture are updated to reflect this modification and preserve the water balance.

G.3.3 Soil Erosion

G.3.3.1 General

The soil erosion module is based on the USLE, which is an empirical methodology (see, e.g., Wischmeier and Smith, 1978) based on measured soil losses from experimental field-scale plots in the United States for approximately 40,000 storms. The USLE predicts sheet and rill erosion from hillsides upslope of defined drainage channels, such as streams; however, it does not predict streambank erosion.

Let SL (kg m^{-2} $time^{-1}$) denote the eroded soil flux (unit load) from a hillside area over some time period. SL is predicted by the USLE as the product of the following six variables:

$$SL = R \times K \times C \times P \times LS \times Sd \tag{G-51}$$

Where

R = Rainfall factor ($time^{-1}$). Accounts for the erosive (kinetic) energy of falling raindrops, which is essentially controlled by rainfall intensity. The kinetic energy of an individual storm multiplied by its maximum 30-minute intensity is sometimes called the erosivity index (EI) factor. R factors are developed by cumulating these individual storm EI factors. R factors have been compiled throughout the United States on a long-term annual average basis.

K = Soil erodibility factor (kg m^{-2}). An experimentally determined property and is a function of soil type, including particle size distribution, organic content, structure, and profile. K values are available from soil surveys and databases (e.g., State Soil Geographic [STATSGO]).

C = Dimensionless "cropping management" factor. Varies between 0 and 1. C accounts for the type of cover (e.g., sod, grass type, fallow) on the soil, and is used to correct the USLE prediction relative to the cover type for which the experimentally determined K values were measured (fallow or freshly plowed fields).

P = Dimensionless practice factor. Accounts for the effect of erosion control practices such as contouring or terracing. P is never negative, but could be >1.0 if land practices actually encourage erosion relative to the original experimental plots on which K was measured.

LS = Length-slope factor**,** accounts for the effects of the length and angle of the slope of a field on erosion losses. LS is calculated by the following equation from U.S. EPA (1985b):

$$LS_i = (.045X_i)^b(65.41Sin^2\theta + 4.56Sin\theta + .065) \tag{G-52}$$

where

X_i = Flow length (m) from the point at which sheet flow originates (the upslope drainage divide) to the point of interest on the hillside.
Θ = Slope angle (degrees), where θ may be calculated from percent slope, S, as

$$\theta = arctan(S/100) \tag{G-53}$$

and b, the exponent, is determined as a function of S as

$b = 0.5$, if $S>.05$
$b = 0.4$, if $.035 \leq S \leq .045$
$b = 0.3$, if $.01 \leq S < .035$
$b = 0.2$, if $S<.01$.

LS increases with increasing flow length because runoff quantity generally increases with flow length. It increases with slope because runoff velocity generally increases with slope.

Sd = **Sediment delivery ratio.** Estimates the fraction of onsite eroded soil that reaches a particular downslope or downstream location in a watershed sub-basin (Shen and Julien, 1993). The sediment delivery ratio is used to account for deposition of eroded soil from the local watershed in ditches, gullies, or other depressions.

Vanoni (1975) developed the sediment delivery ratio as a function of watershed drainage area:

$$Sd = a \times A^{-.125} \tag{G-54}$$

where

Sd = Sediment delivery ratio (dimensionless)

a = Normalized to give $Sd = 1.0$ for an area of 0.001 mi^2 as per Vanoni (1975) (for area in m^2, $a=2.67$).

A = Sub-basin area (m^2)

G.3.3.2 Daily USLE Implementation

The HGSM implements the USLE on a storm event basis using a modified USLE procedure. This implementation requires determining a daily R value (R_t, d^{-1}) that specifies the erosivity of each daily storm.

For this evaluation, R_t is supplied from published long-term annual total R values. These long-term annual total R values (published in the form of isopleths across the country) are disaggregated down to daily values using the following method:

Given: Long-term annual total R for a site, R_{ann}, (obtained from the isopleths)

Given: Number of years in the simulation, NYR

Given: Hourly time series of precipitation amounts for the complete record of NYR years

1. Compute cumulative R over record, $R_{total}=R_{ann} \times NYR$.
2. Compute cumulative precipitation over NYR years, PPT_{total}.
3. For each hourly precipitation value in the record, allocate R_{total} to that hour based on the fraction of PPT_{total} represented by the hourly precipitation. Denote an hourly allocation as R_{hour}.
4. For each day of the record, cumulate all R_{hour} values to the daily total. The result is R_t for each day of the NYR record.

G.3.3.3 Spatial Implementation

For the local watershed application, the daily USLE is applied spatially to a hillside comprised of N subareas (see Figure G-6). Pseudo-code for this application is

```
LET $CSL_i$=Cumulative soil load (kg $d^{-1}$) for subarea i (i.e., eroded load from subarea i) and all upslope subareas j, j=1,...,i
LET $WSA_i$=Cumulative land area ($m^2$) upslope of and including subarea i
FOR I=1,...,N
    $Keff_i$=Area-weighted $K_i$ for all subareas j, j=1,...,i
    $Ceff_i$=Area-weighted $C_i$ for all subareas j, j=1,...,i
    $Peff_i$=Area-weighted $P_i$ for all subareas j, j=1,...,i
```

```
        $CSL_i = R \times WSA_i \times Keff_i \times Ceff_i \times Peff_i \times LS_i \times Sd_i$
    NEXT i
```

G.3.4 Chemical Fate and Transport

G.3.4.1 Runoff Compartment

G.3.4.1.1 Introduction

The module used to estimate chemical and suspended solids concentrations in storm event runoff is based on mass balances of solids and chemical in the runoff and the top soil column layer of thickness dz. The soil compartment is external to this module (see Section G.3.4.2), and results from that compartment are called as needed by the software. Solids and chemical concentrations in the runoff are assumed to be at steady-state during each individual runoff event, but can vary among runoff events (i.e., a quasi-dynamic approach). The assumption of steady-state within each storm event is appropriate for the following reasons:

- Run-time considerations (i.e., maximize the numerical time step).
- Data are not available at the temporal scale to accurately track within-storm event conditions (e.g., rainfall hyetographs).
- Because of the anticipated relatively small surface areas of the watershed subareas and the associated relatively small runoff volumes, the actual time to steady-state may not differ significantly from the 1 day or less implicitly assumed here. (A sensitivity analysis was performed using a dynamic form of the runoff compartment module that suggested relatively little difference in soil concentrations as a function of the steady-state versus dynamic assumption.)
- To the extent that the actual time to steady-state would be >1 day, the module is biased toward overestimating downslope concentrations and waterbody loads (i.e., it is a protective assumption from the risk standpoint).

Figure G-8 presents the conceptual runoff quality module, showing the two compartments and the fate and transport processes considered. Development of mass balance equations for solids and chemical follow.[4]

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[4] Hydrolysis, volatilization, and biodegradation processes are not simulated in the runoff compartment. The percentage of time that runoff is actually occurring will be sufficiently short that any additional losses from these processes should be minimal. In addition, these processes are continuously simulated in the surface layer of the soil column: To also include them in the runoff compartment would be double-counting.

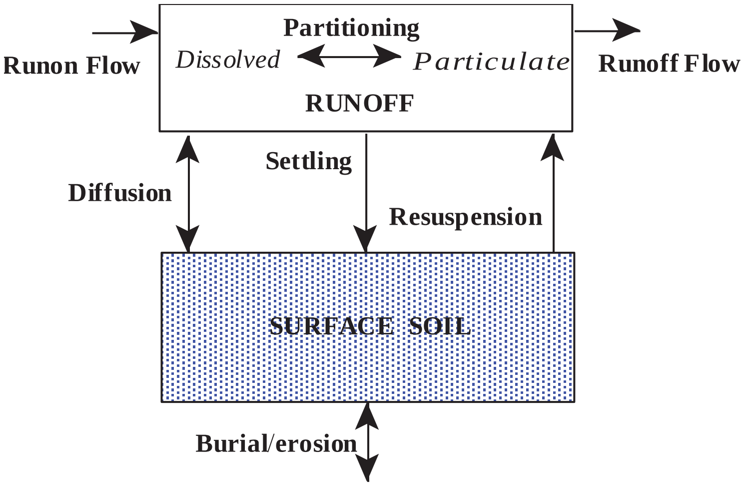

Figure G-8. Runoff quality conceptual model.

G.3.4.1.2 Solids in Runoff Compartment

A steady-state mass balance of solids in the runoff (i.e., suspended solids from erosion), written for local watershed subarea i is given by the following equation (in the subsequent module development, units are presented in general dimensional format (i.e., M[ass]-L[ength]-T[ime], for simplicity of presentation):

$$0 = Q'_{i-1} m_{1,i-1} - Q'_i m_{1,i} - vs_i A_i m_{1,i} + vr_i A_i M_2 \qquad \text{(G-55)}$$

$$Q'_{i-1} = Q_{i-1} + \frac{CSL_{i-1}}{\rho} \qquad \text{(G-56)}$$

$$Q'_i = Q_i + \frac{CSL_i}{\rho} \qquad \text{(G-57)}$$

where

$m_{1,i}$= Solids concentration (M L^{-3}) in the subarea i runoff (suspended solids)
M_2 = Solids concentration (M L^{-3}) in the top soil column layer of subarea i
Q_i = Runoff flow (L^3 T^{-1}) leaving subarea i
Q_{i-1}= Runon flow (L^3 T^{-1}) from subarea i−1
A_i = Surface area (L^2) of subarea i
vs_i = Settling velocity (L T^{-1})
vr_i = Resuspension velocity (L T^{-1})
$\acute{Q}_i$ = Total runoff flow volume (L^3 T^{-1}) (water plus solids) leaving subarea i
CSL_i =Cumulative soil load leaving subarea i (M T^{-1})
ρ = Particle density (M L^{-3}) (i.e., 2.65 g m^{-3}).

Note: Subscript “1” denotes the runoff compartment, whereas “2” denotes the top soil column layer compartment.

The first term in Equation G-55 is the flux of soil across the upslope interface of subarea i. The second term is the flux of soil across the downslope interface, the third term is an internal sink of soil due to settling, and the fourth term is an internal source due to resuspension.

G.3.4.1.3 Solids in Soil Compartment

The HGSM does not consider chemical mass transport among watershed subareas due to soil erosion because it is based on a single subarea only; therefore, that transport is considered here. The HGSM assumes that solids mass transport to or from the soil compartment of any given watershed subarea occurs only in a vertical direction (i.e., there is no downgradient advection of the top soil column layer). (This is analogous to the assumption of a stationary sediment bed in stream/sediment quality modules.) The downslope mass transport of soil occurs due to vertical erosion or resuspension of soil followed by advective transport of the soil in the runoff water as suspended solids. The transport is described in terms of the following three parameters: settling, resuspension, and burial/erosion velocities. Under the assumption of no advective transport of the soil column layer, the steady-state mass balance equation for the surficial soil layer is

$$0 = vs_i m_{1,i} A_i - vr_i m_{2,i} A_i - vb_i m_{2,i} A_i \qquad \text{(G-58)}$$

where

vb_i = Burial/erosion velocity ($L\ T^{-1}$).

The first term of Equation G-58 is a source of soil mass to the surficial soil column layer due to settling from the overlying runoff water. The second term is a sink from resuspension. The third term is either a source or a sink depending on the sign of the burial/erosion velocity as described below.

Consider the solids balances in the runoff and soil compartments, Equations G-55 through G-58. These equations involve three parameters (i.e., vs, vr, and vb) and two solids concentrations (i.e., m_1 and m_2). Which of these five variables is known for arbitrary subarea i? It can be assumed that the solids concentration in the soil (m_2) is a known value—it is simply the bulk soil density. Consider now the suspended solids concentration in subarea i, $m_{1,i}$. From the soil erosion module, the total solids mass fluxes moving across both the upslope and downslope interfaces of subarea i are known, and these two fluxes are, respectively, the first two terms on the right side of Equation G-55 $m_{1,i}$ and can then be determined as

$$m_{1,i} = CSL_i / Q'_i \qquad \text{(G-59)}$$

where

CSL_i	=	the cumulative soil load leaving subarea i, as determined by the soil erosion module
Q'_I	=	the cumulative runoff flow volume (including solids' volume) leaving subarea i, as determined by the runoff quantity model.

Therefore, because the soil concentration (m_2) is assumed to be known, and the soil erosion and runoff quantity modules can be used to determine the suspended solids concentrations (the $m_{1,i}$), Equations G-55 through G-58 can now be considered as two equations in three unknowns (i.e., vs, vr, and vb).

The settling (vs) and resuspension (vr) parameters reflect processes internal to subarea i, whereas the burial/erosion parameter (vb) reflects net changes across subarea i and is completely determined by the difference in the soil fluxes entering and leaving subarea i. This can be observed by adding the right sides of Equations G-55 and G-58 and setting the result to zero. All terms involving vs and vr cancel, and the burial/erosion velocity is then derived by

$$vb_i = \frac{CSL_{i-1} - CSL_i}{A_i m_2} \tag{G-60}$$

where CSL_{i-1} and CSL_i denote the soil fluxes into and out of subarea i, respectively, as previously discussed. From Equation G-60 it can be observed that, if the soil load entering subarea i (CSL_{i-1}) is greater than the soil load leaving (CSL_i), then the burial/erosion velocity is positive and soil is being deposited (buried). Conversely, as will typically be the case, if the load leaving is greater than the load entering, then the burial/erosion velocity will be negative and erosion is occurring.

With the net soil flux across the subarea having been determined, Equations G-55 and G-58 are in fact the same equation—the burial velocity term is explicitly shown in Equation G-58 and implicitly shown in Equation G-55. Thus, either Equation G-55 or G-58 represents one equation in two unknowns (i.e., vs and vr). If one of these is known, then the other can be solved. Of the two, it would be very difficult to obtain estimates for the resuspension velocity, and the settling velocity could be assumed similar to, for example, hindered or compaction settling in sludge thickeners. Accordingly, the following equation determines vr as a function of vs (and vb, which is determined using Equation G-60) for subarea i:

$$vr_i = vs_i \frac{m_{1,i}}{m_2} - vb_i \tag{G-61}$$

The settling velocity, vs, is assigned values from a uniform random distribution between the range 0.05 and 1.0 m d^{-1}, based on observed settling velocities for “mineral” sludges in sludge thickening experiments.

In summary, with m_2 known and m_1 calculated from results of the soil erosion and runoff modules, the solids mass balance equations are used to determine the burial/erosion and resuspension parameters for subsequent use in the chemical (contaminant) model.

G.3.4.1.4 Contaminant in Runoff Compartment

As shown in Figure G-8, a steady-state mass balance of contaminant in the runoff results in the following equation:

$$0 = Q'_{i-1} c_{1,i-1} - Q'_i c_{1,i} - vs_i A_i Fp_{1,i} c_{1,i} + vr_i A_i Fp_{2,i} Er_i c_{2,i} + vd_i A_i \Phi_2 \left(\frac{Fd_{2,i}}{\Phi_2} c_{2,i} - \frac{Fd_{1,i}}{\Phi_{1,i}} c_{1,i} \right) \tag{G-62}$$

where

$c_{1,i}$ = Total contaminant concentration (particulate + dissolved) in runoff in subarea i ($M\ L^{-3}$)

$c_{2,i}$ = Total contaminant concentration in soil ($M\ L^{-3}$)

$Fp_{1,i}$ = Fraction particulate in runoff

Er_i = Enrichment ratio

vd_i = Diffusive exchange velocity ($L\ T^{-1}$)

$\Phi_{1,i}$ = Porosity of the runoff, calculated as

$$\Phi_{1,i} = 1 - \frac{m_{1,i}}{\rho} \tag{G-63}$$

where ρ is the density ($M\ L^{-3}$) of suspended solids (e.g., 2.65 g cm^{-3}).

$Fd_{1,i}$ = Fraction dissolved in runoff ($1 - Fp_{1,i}$)

$V_{1,i}$ = Subarea-specific (not cumulative) runoff volume for subarea i (L^3)

Φ_2 = Soil porosity, calculated as

$$\Phi_2 = 1 - \frac{m_2}{\rho} \tag{G-64}$$

Note that Φ_2 is equivalent to porosity (η) in the HGSM.

The diffusive flux term in Equation G-62 (last term) deserves some explanation regarding the porosities. Recall that the concentration is a total concentration (sorbed plus dissolved) expressed as mass of chemical per *total* volume (solids plus water) in either the soil or the runoff water. Multiplication of the total concentration by Fd converts total concentration to dissolved concentration, but still based on total volume. Thus, the runoff water and soil porosities are included in the denominators to express the dissolved concentration per volume of *water* (i.e., the actual pore water (or runoff water) concentration). Regarding the soil porosity in the $vd_i\ A_i\ \Phi_2$ term, Φ_2 is used to account for the fact that diffusion of dissolved chemical will only occur across the interstitial area, not the entire interface area.

Equation G-62 can be used to express $c_{1,i}$ as a function of $c_{1,i-1}$ and $c_{2,i}$ as

$$c_{1,i} = \frac{Q'_{i-1} c_{1,i-1} + \lfloor vr_i A_i Fp_{2,i} Er_i + vd_i A_i Fd_{2,i} \rfloor c_{2,i}}{Q'_i + vs_i A_i Fp_{1,i} + vd_i A_i \Phi_2 \left(Fd_{1,i} / \Phi_{1,i} \right)} \tag{G-65}$$

where $c_{2,i}$ is determined by the HGSM as described in Section G.2. Determination of the individual terms constituting this equation is described below.

$Fp_{1,i}$ is calculated using the following equation from Thomann and Mueller (1987):

$$Fp_{1,i} = \frac{\left(k_d / \Phi_{1,i} \right) m_{1,i}}{1 + \left(k_d / \Phi_{1,i} \right) m_{1,i}} \tag{G-66}$$

where

k_d = Chemical-specific partition coefficient (L^3 M^{-1}) (Note: k_d is divided by porosity to attain the porosity-corrected k_d with units of mass per total [liquid plus solids] volume.)

$Fp_{2,i}$ is similarly calculated as

$$Fp_{2,i} = \frac{(k_d / \Phi_2)m_2}{1 + (k_d / \Phi_2)m_2} \tag{G-67}$$

where Fp_2 (and Fd_2) will be constant among all subareas i.

$Fd_{1,i}$ and $Fd_{2,i}$ are then determined as

$$Fd_{1,i} = 1 - Fp_{1,i} \tag{G-68}$$

$$Fd_{2,i} = 1 - Fp_{2,i} \tag{G-69}$$

Assuming that resistance to vertical diffusion is much greater in the soil than in the runoff (Thomann and Mueller, 1987, p. 548), the diffusive exchange velocity, vd_i, can be expressed as

$$vd_i = \frac{Dw}{Lc} \tag{G-70}$$

where

Dw = Water diffusivity (L^2 T^{-1}).

Lc = Characteristic mixing length (L) over which a concentration gradient exists; assumed to be the depth of the runoff volume, including the solids (H1'):

$$Lc = Hl'_i = \frac{Q'_i}{A_i} \tag{G-71}$$

The enrichment ratio, Er_i, is used to account for preferential erosion of finer soil particles, with higher specific surface areas and more sorbed chemical per unit area, as rainfall intensity decreases. That is, large (i.e. highly erosive) runoff events may result in average eroded soil particle sizes and associated sorbed chemical loads that do not differ much from the average sizes/loads in the surficial soil column layer. However, less intense runoff events will erode the finer materials, and resulting runoff chemical loads could be significantly higher than represented by the average soil concentration. U.S. EPA (1985b) gives the storm event-specific enrichment ratio as a power function of sediment discharge flux (M L^{-2}). This formulation results in

$$Er_i = \frac{a}{(CSL_i / WSA_i)^{0.2}} \tag{G-72}$$

where

a = 7.39 for CSL_i/WSA_i in kg ha^{-1} (U.S. EPA, 1985b).

(CSL_i is the event soil load leaving subarea I, and WSA_i is the local watershed surface area from the drainage divide down to and including subarea i.) The enrichment ratio is ≥ 1.0. Should specific values of the sediment discharge (the denominator) result in an enrichment ratio <1.0, it is reset to 1.0 in the code.

G.3.4.2 Soil Compartment

The GSCM (see Section G.2.2) is coupled to the runoff compartment module (see Section G.3.4.1) in this section and applied to the several subareas that constitute the sheet flow for the local watershed of which the home garden is an integral part. Continuing the chemical concentration indexing scheme (i.e., subscript "1" denoting runoff compartment, and subscript "2" denoting surficial soil compartment), let the total (dissolved, particulate, and gaseous phase) chemical concentration in the surficial soil column layer of any local watershed subarea i be denoted as $C_{2,i}$. $C_{2,i}$ is equivalent to C_T. From Section G.2.2 (GSCM), the governing differential equation for the surface soil layer of subarea i is

$$\frac{\partial C_{2,i}}{\partial t} = D_E \frac{\partial^2 C_{2,i}}{\partial z^2} - V_E \frac{\partial C_{2,i}}{\partial z} - \sum k_j C_{2,i} + ss_i \qquad \text{(G-73)}$$

where k_j represents the first-order rate constant due to process j, and does not include runoff/erosion processes (i.e., biological decay and hydrolysis and wind/mechanical action). The last term, ss_i, is a source/sink term representing the net effect of runoff and erosion processes on $C_{2,i}$ as shown in Figure G-8. This term is derived by the following equation:

$$ss_i = \frac{vs_i Fp_{1,i} C_{1,i} - vr_i Fp_{2,i} Er_i C_{2,i} - vd_i \Phi_2 \left(\frac{Fd_{2,i}}{\Phi_2} C_{2,i} - \frac{Fd_{1,i}}{\Phi_{1,i}} C_{1,i} \right) - vb_i Fp_2 C_{2,i}}{dz} \qquad \text{(G-74)}$$

where vs_i, vr_i, vd_i, and vb_i denote, respectively, the settling, resuspension, diffusive exchange, and burial/erosion velocities for subarea i as described in the runoff compartment model. Thus, the terms comprising ss_i are, respectively, a source of chemical due to settling from the overlying runoff water, a sink of chemical due to resuspension, and a source or sink (depending on the relative values of $C_{1,i}$ and $C_{2,i}$) due to chemical diffusion to and from the runoff.

The burial/erosion mechanism introduces a minor mass balance error into the model. The module for surface soil/runoff water fate and transport (Section G.3.4.1) is based on a conceptual model originally developed for use in a stream/sediment application (e.g., Thomann and Mueller, 1987) where the sediment compartment location relative to a reference point below the surface can move vertically ("float") as burial and erosion occur. In that moving frame of reference, burial/erosion of contaminant does not introduce a mass balance error because, with respect to the modeled sediment, this sink/source of contaminant is **exogenous** to the modeled system (i.e., it is coming from/going to outside of the modeled system). There is internal (endogenous) mass balance consistency within the modeled system. However, the frame of reference is not allowed to float, but is fixed by the elevation of the lower boundary (e.g., top of the vadose zone). Thus, if a sorbed chemical is eroded from the surface cell, then that surface cell, which is vertically fixed, must have a "source" that is internal to the modeled soil column to compensate for this sink or its internal mass balance is not maintained. The magnitude of this mass balance error is equal to the mass of eroded soil from the surface over the duration of the simulation multiplied by its average sorbed chemical concentration. In most cases, this error as a percentage of the total chemical mass in the modeled home garden will be quite small, and that has been confirmed in multiple executions of the module. Conceptually at least, the GSCM could be designed so that, after each runoff event, the surficial soil compartment could decrease or increase in size to accommodate the event's erosion/burial magnitude, while maintaining a fixed vertical reference.

Grouping coefficients of $C_{1,i}$ and $C_{2,i}$, Equation G-74 can be rewritten as

$$ss_i = a_i C_{1,i} - b_i C_{2,i} - k_{bu,i} C_{2,i} \tag{G-75}$$

where

$$a_i = \frac{vs_i Fp_{1,i} + vd_i \Phi_2 \dfrac{Fd_{1,i}}{\Phi_{1,i}}}{dz} \tag{G-76}$$

$$b_i = \frac{vs_i Fp_{2,i} Er_i + vd_i Fd_{2,i}}{dz} \tag{G-77}$$

$$k_{bu,i} = \frac{vb_i Fp_{2,i}}{dz} \tag{G-78}$$

and $k_{bu,i}$ is the first-order rate constant ($1\ T^{-1}$) associated with the burial/erosion process.

Using Equation G-75, Equation G-73 can be rewritten as

$$\frac{\partial C_{2,i}}{\partial t} = D_E \frac{\partial^2 C_{2,i}}{\partial z^2} - V_E \frac{\partial C_{2,i}}{\partial z} - \sum k_j C_{2,i} + a_i C_{1,i} - b_i C_{2,i} - k_{bu,i} C_{2,i} \tag{G-79}$$

Equation G-79 demonstrates that $C_{2,i}$ is a function of $C_{1,i}$. Similarly, Equation G-65 of the runoff compartment module demonstrates that $C_{1,i}$ is a function of $C_{2,i}$. Thus, $C_{2,i}$ and $C_{1,i}$ are jointly determined at any time, t, by simultaneous solution of their two respective equations.

$C_{2,i}$ at time, t, can be determined by substitution for $C_{1,i}$. Using Equation G-65, $C_{1,i}$ can be expressed as

$$C_{1,i} = \frac{Q'_{i-1} C_{1,i-1}}{d_{2,i}} + \frac{d_{1,i}}{d_{2,i}} C_{2,i} \tag{G-80}$$

where

$$d_{1,i} = vr_i A_i Fp_{2,i} Er_i + vd_i A_i Fd_{2,i} \tag{G-81}$$

$$d_{2,i} = Q'_i + vs_i A_i Fp_{1,i} + vd_i A_i \Phi_2 \frac{Fd_{1,i}}{\Phi_{1,i}} \tag{G-82}$$

Substituting for $C_{1,i}$ from Equation G-80 into Equation G-79, the differential equation for $C_{2,i}$ is now expressed implicitly as a function of $C_{1,i}$ as

$$\frac{\partial C_{2,i}}{\partial t} = D_E \frac{\partial^2 C_{2,i}}{\partial z^2} - V_E \frac{\partial C_{2,i}}{\partial z} - \left(\sum k_j + b_i + k_{bu,i} - \frac{a_i d_{1,i}}{d_{2,i}} \right) C_{2,i} + \frac{a_i Q'_{i-1} C_{1,i-1}}{d_{2,i}} \tag{G-83}$$

After $C_{2,i}$ at time, t, is determined by solution of Equation G-83, the associated value for $C_{1,i}$ can be found from Equation G-80, thus completing the simultaneous solution. (The value for $C_{1,i-1}$ [i.e., the runoff concentration in the immediately upslope subarea] will have been determined previously during the simultaneous solution for the i−1 subarea at time, t.)

To implement the simultaneous solution, Equation G-83 can be simplified to

$$\frac{\partial C_{2,i}}{\partial t} = D_E \frac{\partial^2 C_{2,i}}{\partial z^2} - V_E \frac{\partial C_{2,i}}{\partial z} - k_i' C_{2,i} + ld_{i-1} \tag{G-84}$$

where

$$k'_i = \sum k_j + k_{ev,i} + k_{bu,i} \tag{G-85}$$

$$k_{ev,i} = b_i - a_i \frac{d_{1,i}}{d_{2,i}} \tag{G-86}$$

$$ld_{i-1} = \frac{a_i}{d_{2,i}} Q'_{i-1} C_{1,i-1} \tag{G-87}$$

k'_i is the lumped first-order loss rate, which includes the effects of abiotic hydrolysis (j=hy), aerobic biodegradation (j=ae), and wind/mechanical activity (j=wd), in addition to runoff and erosion. $k_{ev,i}$ is the storm event (or runoff and erosion) first-order loss rate. k_{hy} and k_{ae} are inputs to the module. The last term, ld_{i-1}, is the run-on load from upslope subareas in g m^{-3} d^{-1}.

Recall that the GSCM, the governing equation is broken up into three component equations: diffusion, convection, and first-order losses (Equations G-15 through G-17), and each equation is solved individually on a grid. In the subsurface layers, the solution technique described in Section G.2 is applied directly. However, for the surface soil column layer, the first two-component equations remain the same, but the third equation has been revised to

$$\frac{\partial C_{2,i}}{\partial t} = -k\,' C_{2,i} + ld_{i-1} \tag{G-88}$$

which has the following analytical solution for $C_{2,i}=C^0_{2,i}$ at t=0:

$$C_{2,i} = \begin{cases} C_{2,i}^{0} \exp\left(-k't\right) + ld_{i-1}\left[\dfrac{1-\exp\left(-k't\right)}{k'_i}\right] & k'_i > 0 \\ C_{2,i}^{0} + ld_{i-1}t & k'_i = 0 \end{cases} \tag{G-89}$$

To track mass losses, the total mass added to the soil column in subarea i in any time period zero to t due to settling from runoff water, $M_{add,i}$ (M L^{-2}), is evaluated using

$$M_{add,i} = ld_{i-1}t \;\; dz \tag{G-90}$$

A mass balance on the soil column in time, t, gives

$$\Delta M_i = M_{add,i} - M_{loss,i} \tag{G-91}$$

where ΔM_i (M $L^{-2)}$ is the change in mass in the soil column in subarea i as given by $(C_{2,i} - C_{2,i}) \times dz$, and $M_{loss,i}$ (M L^{-2}) is the total mass lost from the subarea i soil column in any time period zero to t. By substituting Equation G-89 for $C_{2,i}$ and Equation G-90 for $M_{add,i}$ and rearranging, $M_{loss,i}$ when $k'_i=0$ is found to equal 0, and the following equation for $M_{loss,i}$ was derived for $k'_i>0$:

$$M_{loss,i} = \left[C_{2,i}^{0}\left(1-\exp\left(-k'_i\,t\right)\right) + ld_{i-1}\left(\frac{k'_i\,t + \exp\left(-k'_i\,t\right) - 1}{k'_i}\right)\right]dz \tag{G-92}$$

The total mass lost in any time period zero to t from subarea i soil column can be attributed to specific first-order loss processes, j, $M_i(t)$ (M L^{-2}) using

$$M_{j,i} = M_{loss,i}\frac{k_j}{k'_i} \tag{G-93}$$

where j is hy for hydrolysis, ae for aerobic degradation, wd for losses due to wind/mechanical activity, ev for runoff/erosion events, and bu for burial/erosion.

Equation G-80 provides the contaminant concentration in the runoff water at time, t. The average contaminant concentration in the runoff water ($C_{1,i}$) over time zero to t is determined using

$$\overline{C}_{1,i} = \frac{Q'_{i-1}\,\overline{C}_{1,i-1}}{d_{2,i}} + \frac{d_{1,i}}{d_{2,i}}\overline{C}_{2,i} \tag{G-94}$$

where $C_{2,i}$ is the time-weighted average contaminant concentration in the soil compartment over the same time period. Given the short time step (i.e., 1 day) used in the integration of the local watershed/soil column module, $C_{2,i}$ is approximated using

$$\overline{C}_{2,i} = \frac{C_{2,i}^{0} + C_{2,i}}{2} \tag{G-95}$$

where the 0 superscript denotes concentration at the beginning of the day.

G.3.5 Implementation

G.3.5.1 Overview

The HGSM implementation essentially links the regional watershed, GSCM, and local watershed at scales that are appropriate to the modeling questions. For example, at the regional level, the infiltration is assumed constant and convection events occur at regular intervals throughout the simulation. At the local watershed level, the infiltration rate (I) is allowed to vary from year to year; as a result, convection events are not "required" to occur at regular intervals (see Figures G-9 and G-10 for the local watershed logic). To determine the appropriate time to initiate a convection event, a variable (fadv) tracking the fraction of mass in the bottom soil column layer that would have convected is incremented by ($dt \cdot V_E/dz$) at the end of every time step. If fadv is sufficiently close to 1, then a convection event is initiated, and fadv is reset to zero.

M_{lcha} is incremented by fadv multiplied by dz multiplied by C_T in the lowest layer, and C_T in the lowest layer is adjusted accordingly. Leachate flux for the final year is then calculated using Equation G-31.

G.3.5.2 Simulation-Stopping Criterion

For a given local watershed, i, the simulation is stopped in each successive subarea when the amount of contaminant mass in local watershed i and all upslope subareas j (j<i) is determined to be insignificant. "Insignificance" is defined by the input parameter TermFrac, and this simulation criterion is implemented as follows:

1. During the years before the end of the operating life of the farming field, the year-end cumulative subarea contaminant mass in each subarea is determined. Here, cumulative subarea mass ($samass_i$) refers to the sum of the contaminant mass in subarea i and all upslope subareas j (j<i). The maximum cumulative subarea contaminant mass ($maxsamass_i$) is stored for each subarea.
2. After farming operation ceases, the year-end cumulative subarea contaminant mass in each subarea is compared to the stored maximum for that subarea. The simulation in subarea i is stopped when

$$Samass_i \leq TermFrac \bullet maxsamass_i$$

where *TermFrac* is the user-specified fraction ranging from 0 to 1.0 (unless the NyrMax parameter is reached first, at which point the simulation is automatically stopped). The year the simulation ceases in each local watershed and subarea is stored in an internal two-dimensional array dimensioned on local watershed and subarea.[5]

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[5] When the source code was run, computer memory requirements resulted in an inability to make full use of the previously described *TermFrac* stopping criterion for highly persistent chemicals. Time series outputs are kept in random access memory (RAM) for post-processing. When the length of the time series became too large for available RAM and memory-caching occurred, run time increased drastically. To mitigate this problem, the length of the time series was determined by the *TermFrac* criterion, as previously described, or 200 years, whichever came first.

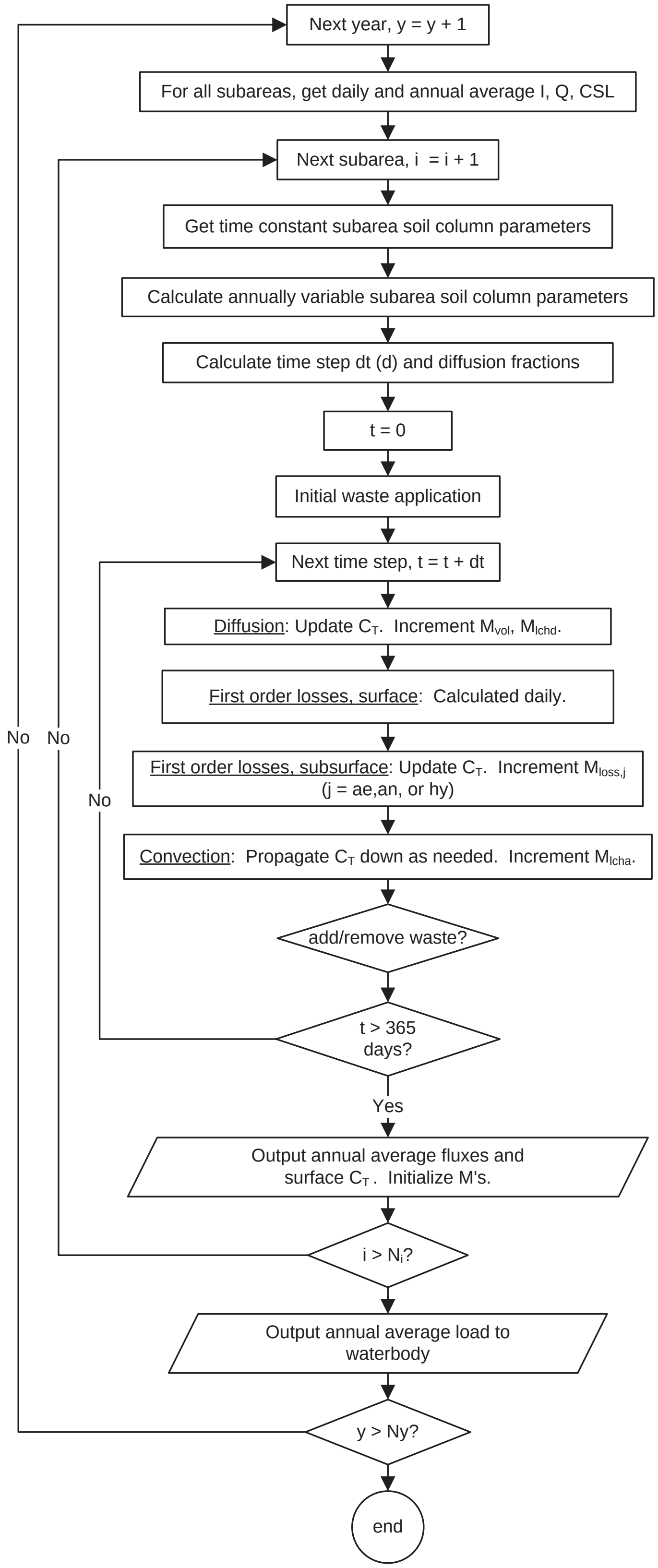

Figure G-9. Overview of algorithm for local watershed/GSCM.

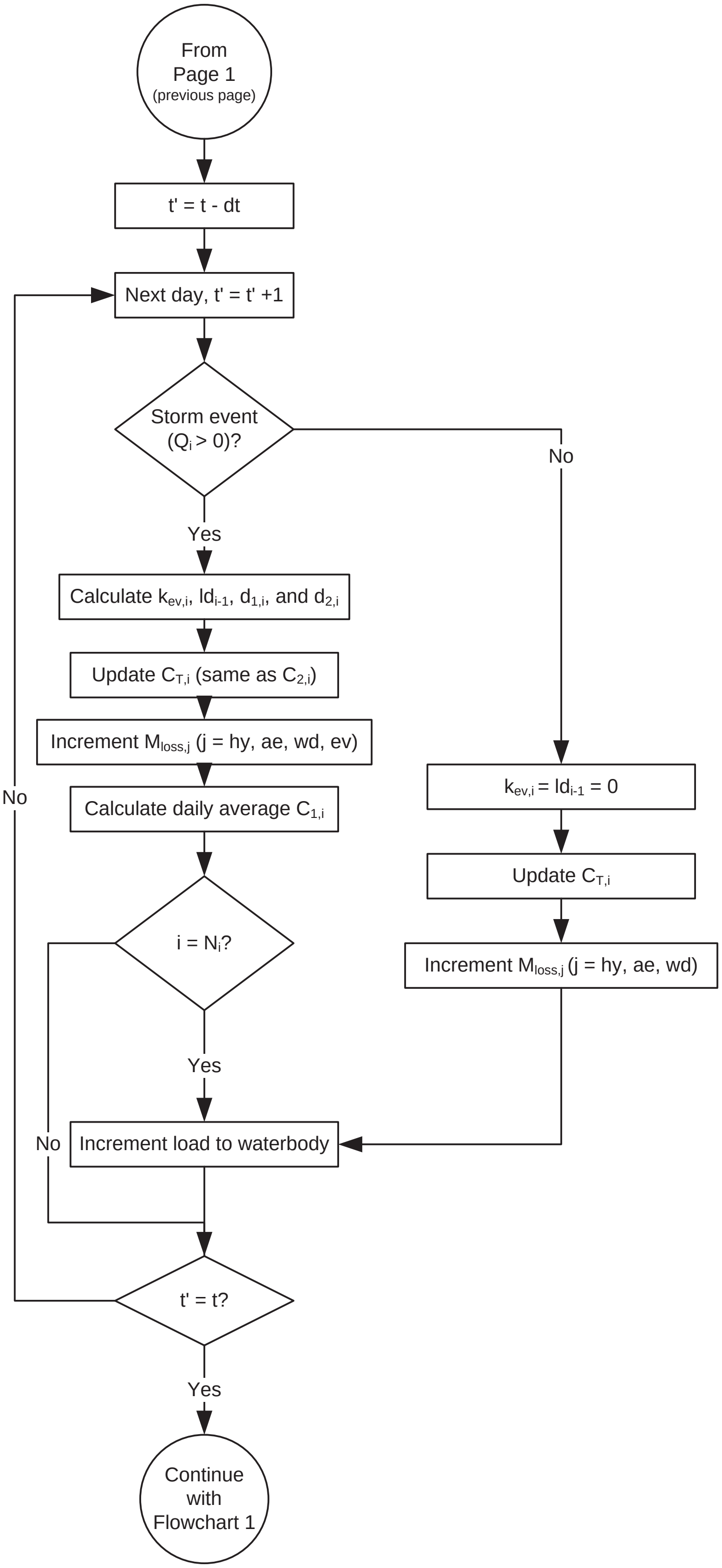

Figure G-10. Detail on calculation of first-order losses in surface layer.

G.3.5.3 Leachate Flux Processing

Preliminary module runs during model development demonstrated many cases in which the convective transfer step occurred less than once per year, sometimes even less than once in the entire simulation period. In these cases, the leachate flux was nonzero in the years when a convection event occurred, and zero in years when it did not. This is a limitation of the solution technique. In reality, leaching occurs more or less continuously over the time between convection events. To mitigate this limitation, a leachate flux post-processing algorithm was developed. The entire simulation (0<j#NyrMax) is split into the following three time periods, where j is used here as the year index:

1. Home garden operating years ($0 \le j \le y_{op}$)
2. Non-operating years ($y_{op} < j \le LeachFluxNY$)
3. No leachate flux years ($LeachFluxNY < j \le NyrMax$)

where LeachFluxNY is the last year when there is a positive leachate flux. The processed leachate fluxes (J_{lchp}, g m^{-2} d^{-1}) in time periods 1 and 2 are calculated from J_{lch} in each year, j, using

$$J_{lchp,j} = \frac{I_j}{\bar{I}(b-a+1)} \sum_{j=a}^{j=b} J_{lch,j} \qquad \text{(G-96)}$$

where, in time period 1, a=0 and b=y_{op}. In time period 2, a=y_{op} and b=LeachFluxNY. The first term in Equation G-96 is an infiltration-based weight where I_j is the annual average infiltration rate in year j and I is the average infiltration rate between years a and b. In time period 3, J_{lchp} is zero.

Using Equation G-96 to estimate the leachate flux conserves mass. That is, the total mass lost due to leaching over the course of the simulation is the same using the processed or unprocessed leachate fluxes. However, using the processed leachate fluxes provides a smoother function of leachate flux over time.

G.3.5.4 End-of-Simulation Mass Balance Check

At the end of the simulation, the code performs a mass balance check, including the home garden subarea and all other subarea “soil columns.” The mass balance error (fMerr) is computed as a fraction of the total contaminant mass added to the system from the mass balance equation

$$fMerr = 1 - \left(fMrem + fMlost \right) \qquad \text{(G-97)}$$

where fMrem is the fraction of added contaminant mass that remains in the system at the end of the simulation. fMlost is the fraction of the added contaminant mass that was lost from the system by the end of the simulation. fMlost is the sum of the variables listed and defined in Table G-2.

Table G-2. Variables Summarizing Contaminant Mass Losses

Variable	Definition: Fraction of the Total Mass Added or Lost Due to
fMvol_wmu	Volatilization from the home garden
fMlch_wmu	Leaching from the home garden
fMwnd_wmu	Wind/mechanical action on the home garden surface
fMdeg_wmu	Abiotic and biodegradation within the home garden
fMrmv_wmu	Removal from the home garden
fMvol_sa	Volatilization from the non-home garden subarea soil columns
fMlch_sa	Leaching from the non-home garden subarea soil columns
fMdeg_sa	Abiotic and biodegradation in the non-home garden subarea soil columns
fMswl	Runoff/erosion from the most downslope subarea
fMbur[a]	Burial/erosion in all subareas (see k_{bu} in Equation G-87)

[a] fMbur is the only listed variable that can be negative (indicating a mass gain). This results from including a burial/erosion term when linking the runoff and soil compartments (see Figure G-8 and the discussion in Section G.3.4.2)

G.3.6 Output Summary

Table G-3 summarizes the HGSM outputs used in the SFS analysis.

- *Emissions to Estimate Air Impacts*. All annual time series outputs to ISCST3 are reported up to and including the last year that there are nonzero volatile or particulate emission rates (V_E or C_E).
- *Soil Concentrations to Estimate Soil and Food Chain Exposures*. The annual time series of depth-weighted average soil concentration (CTda), used in plant root zone calculations, is reported until soil concentrations reach zero. The same is true for the surface soil concentration (CTss) used in SFS evaluation to estimate exposures due to incidental soil ingestion as well as impact due to particulate emissions.
- *Leachate to Estimate Groundwater Impacts with EPACMTP*. The annual time series of LeachFlux is reported until LeachFlux is zero. AnnInfil is reported from year 1 to the last year that meteorological data are available.

Table G-3. Output Summary for the Home garden Source Model

Variable Name[a]		Definition	Units
Documentation	Code		
I	AnnInfil	Leachate infiltration rate (annual average; home garden subarea[s] only)	m d^{-1}
J_{vol}	VE	Volatile emission rate	g m^{-2} d^{-1}
	VERY	Year associated with output	Year
	VENY	Number of years in outputs	Unitless
CE30	CE	Constituent mass emission rate-PM_{30}	g m^{-2} d^{-1}
	CEYR	Year associated with output	Year
	CENY	Number of years in outputs	Unitless
E30	PE30	Eroded solids mass emission rate-PM_{30}	g m^{-2} d^{-1}
	PE30YR	Year associated with output	Year
	PE30NY	Number of years in outputs	Unitless
pmf	PMF	Particulate emission particle size distribution	Mass fraction
	PMFYR	Year associated with output	Year
	PMFNY	Number of years in outputs	Unitless
J_{lch}	LeachFlux	Leachate contaminant flux	g m^{-2} d^{-1}
	LeachFluxYR	Year associated with output	Year
LeachFluxNY	LeachFluxNY	Number of years in outputs	Unitless
	SWLoadChem	Chemical load to waterbody	g d^{-1}
	SWLoadChemYr	Year associated with output	Year
	SWLoadChemNY	Number of years in outputs	Unitless
C_T	CTss	Soil concentration in surface soil layer	µg g^{-1}
	CTssYR	Year associated with output	Year
	CTssNY	Number of years in outputs	Unitless
C_T	CTda	Depth-weighted average soil concentration (from zava to zavb)	µg g^{-1}
	CTdaYR	Year associated with output	Year
	CTdaNY	Number of years in outputs	Unitless
	SrcSoil	Flag for soil presence (true)	Logical
	SrcOvl	Flag for overland flow presence (true)	Logical
	SrcLeachMet	Flag for leachate presence when leachate is met-driven (true)	Logical
	SrcLeachSrc	Flag for leachate presence when leachate is not met-driven (false)	Logical
	SrcVE	Flag for volatile emissions presence (true)	Logical
	SrcCE	Flag for chemical sorbed to particulates emissions presence (true)	Logical
	NyrMet	Number of years in the available met record	Unitless

[a] When the variable name is used in the code but not in the documentation, the first column is left blank.

G.2.7 Limitations Related to the Use of GSCM

The following limitations are noted for the GSCM:

- The GSCM was developed originally for organic contaminants and assumes that the partition coefficient, K_d, is linear and is estimated as the product of K_{oc} and foc. Partitioning for metals involves complex chemistry, including the dynamic effects of aqueous-phase contaminant concentration, precipitation, dissolution, adsorption/desorption, and media geochemistry (e.g., oxidation-reduction conditions) on the value of K_d and the fate and transport behavior of metals in general. This complexity is not modeled by the GSCM for metals partitioning. Rather, K_d is externally provided as a randomly sampled value by the chemical properties processor (CPP).

- The algorithm estimates annual average source releases. Some of the inputs (e.g., infiltration) are long-term annual averages, whereas others are annual averages. Therefore, the outputs are not strictly applicable to individual years.

- The model in its current form considers only one contaminant at a time and does not simulate fate and transport of reaction products. With further model development, it would be possible to track the production of reaction products in each soil column layer and use basically the same algorithm that is used for the parent compound to model the fate of reaction products.

- The chosen solution technique (i.e. sequential solutions to the three-component differential equations of the governing differential equation) allows computational efficiency. However, the choice of the order in which these solutions are applied could result in systematic errors. The size of the errors depends on the relative loss rates associated with the three processes. For example, if the first-order loss rate due to degradation was high and those losses were calculated first, then less contaminant mass would be available for diffusive and advective losses. The current algorithm solves for diffusive losses first. This is followed by first-order losses and advection, respectively.

G.4 References

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Appendix G

Attachment G-A:
Symbols, Units, and Definitions

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Appendix G
Attachment A:
Symbols, Units, and Definitions

Table G-A-1. Symbols, Units, and Definitions[1]

Symbol	Units	Definition
η_j	–	Total porosity where j is a subscript indicating waste, w; waste/soil mixture in the till zone, till; and soil, s
η	–	Total porosity
θ_a	–	Soil volumetric air content
$\theta_{a,j}$	–	Soil volumetric air content where j is a subscript indicating waste, w; waste/soil mixture in the till zone, till; and soil, s
θ_w	–	Soil volumetric water content
$\theta_{w,j}$	–	Soil volumetric water content where j is a subscript indicating waste, w; waste/soil mixture in the till zone, till; and soil, s
ρ_b	g cm^{-3}	Soil dry bulk density. Same as m^2. (Note: g cm^{-3}=mg m^{-3})
$\rho_{b,j}$	g cm^{-3}	Dry bulk density where j is a subscript indicating waste, w; waste/soil mixture in the till zone, till; and soil, s
$\rho_{b,w}^{wet}$	g cm^{-3}	Wet bulk density of home garden soil amendment
A	m^2	Area of home garden
a_i	1 d^{-1}	Calculated parameter (Equation G-76) for subarea i
bcm	–	Lower coil column boundary condition multiplier
b_i	1 d^{-1}	Calculated parameter (Equation G-77) for subarea i
C'_T	mg g^{-1}	Total mass-based contaminant concentration in dry soil
$C'_{T,W}$	mg g^{-1}	Total mass-based contaminant concentration in incoming dry waste
$C_{2,i}$	g m^{-3}	Contaminant concentration in surface soil grid space in subarea i (equivalent to C_T)
C_G	g m^{-3}	Contaminant concentration in gaseous phase in soil
C_L	g m^{-3}	Contaminant concentration in aqueous phase in soil
C_L^{sol}	g m^{-3}	Contaminant aqueous solubility
CN	Unitless	SCS runoff module curve number parameter
C_S	mg g^{-1}	Contaminant concentration in adsorbed phase in soil
$CSL_{i,t}$	Kg	Cumulative soil load leaving subarea i, day t
C_T	g m^{-3}	Total volume-based contaminant concentration in soil
C_{T0}	g m^{-3}	Initial total volume-based contaminant concentration in soil
$d_{1,i}$	m^3 d^{-1}	Calculated parameter (Equation G-81) for subarea i
$d_{2,i}$	m^3/d^{-1}	Calculated parameter (Equation G-82) for subarea i
D_a	cm^2 s^{-1}	Diffusivity in air

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[1] Based on Table A-1, U.S. EPA, 1999.

Symbol	Units	Definition
D_E	$m^2\ d^{-1}$	Effective diffusivity in soil
$D_{E,a}$	$m^2\ d^{-1}$	Effective diffusivity in soil air
$D_{E,w}$	$m^2\ d^{-1}$	Effective diffusivity in soil water
D_f	–	Fraction of original mass in soil column grid space that diffuses past a boundary in time, t
D_{f0}	—	Fraction of original mass in soil column grid space that remains after time, t
DRZ	cm	Depth of the root zone
d_s	m	Thickness of soil in unmixed home garden till zone
dt	d	Length of time step in GSCM solution algorithm
d_w	m	Thickness of waste in unmixed home garden till zone
D_w	$cm^2\ s^{-1}$	Diffusivity in water
dz	m	Soil column grid size in GSCM solution algorithm
ER_i	Unitless	Erosion chemical enrichment ratio for subarea i
$ET_{i,t}$	$cm\ d^{-1}$	Evapotranspiration from root zone on day t for subarea i
FC_i	cm	Soil moisture field capacity for subarea i
foc	–	Fraction organic carbon in soil
foc_j	–	Fraction organic carbon where j is a subscript indicating waste, w; waste/soil mixture in the till zone, till; and soil, s
$\acute{H}$	–	Dimensionless Henry's law constant
I	$m\ d^{-1}$	Average annual water infiltration rate
$IN_{i,t}$	$cm\ d^{-1}$	Daily infiltration for subarea i, day t
J_{lch}	$g\ m^{-2}\ d^{-1}$	Annual average leachate flux at lower soil column boundary
J_{vol}	$g\ m^{-2}\ d^{-1}$	Annual average volatilization flux at upper soil column boundary
k	$1\ d^{-1}$	Total first-order loss rate
$k_{bu,i}$	$m\ d^{-1}$	First-order rate constant due to burial/erosion for subarea i
K_d	$cm^3\ g^{-1}$	Soil-water partition coefficient
k_j	$1\ d^{-1}$	Annual average first-order loss rate due to process j, where j indicates hydrolysis, h; aerobic biodegradation, ae; anaerobic biodegradation, an; storm events in subarea i, ev,i; and wind/mechanical activity, wd
K_{oc}	$cm^3\ g^{-1}$	Equilibrium partition coefficient normalized to organic carbon
K_{sat}	$cm\ hr^{-1}$	Saturated hydraulic conductivity
K_{TL}	–	Equilibrium distribution coefficient between the total (g/m3) and aqueous phase (g/m3) contaminant concentrations in soil
L	$mg\ yr^{-1}$	Bulk waste mass loading rate into WMU
ld_{i-1}	$g\ m^{-3}\ d^{-1}$	Run-on load to subarea i from subarea i-1
$\acute{L}$	$mg\ yr^{-1}$	Bulk waste loading rate adjusted for mass losses due to unloading
$m1_i$	$g\ m^{-3}$	Suspended solids concentration in runoff water, subarea i
m	$g\ m^{-2}$	Total amount of material from soil column grid space that has passed a boundary at time, t
M_{col1}	$g\ m^{-2}$	Total mass in soil column at start of year
M_{col2}	$g\ m^{-2}$	Total mass in soil column at end of year

Symbol	Units	Definition
M_i	g m^{-2}	Annual contaminant mass loss due to process i, where i is a subscript indicating: Total diffusive loss at the surface, 0 Gas phase diffusive losses (volatilization) at the surface, vol Aqueous phase leaching due to diffusion, lchd Aqueous phase leaching due to advection, lcha First-order loss process j where j is as defined in kj
$\mathrm{M}_{\mathrm{add}}$	g m^{-2}	Annual mass added to soil column
$\mathrm{M}_{\mathrm{rem}}$	g m^{-2}	Annual mass removed from soil column
N_{appl}	1 yr^{-1}	Number of home garden applications per year
N_{dz}	–	Total number of grid spaces of depth dz in soil column
PET_i	cm d^{-1}	Potential evapotranspiration for day t
P_t	cm	Total precipitation on day t
$Q_{i,t}$	m^3 d^{-1}	Runoff flow volume (water only) leaving subarea I, day t
$Q'_{i,t}$	m^3 d^{-1}	Total runoff flow volume (including solids) leaving subarea i, day t
R_{appl}	mg m^{-2} yr^{-1}	Home garden waste application rate
Sd	Unitless	Sediment delivery ratio for subarea/watershed i
$RO_{i,t}$	cm	Stormwater runoff depth leaving subarea i, day t
sd	w/w, %	Weight percent of solids in raw waste applied to home garden
SM_b	–	Unitless soil-specific exponent in Equation G-13
$SM_{i,t}$	cm	Soil moisture in root zone at end of day t for subarea i
t	d	Time since start of simulation
t_{bet}	d	Time between LAU waste applications
vb_i	m d^{-1}	Burial/erosion velocity for subarea i
vd_i	m d^{-1}	Diffusive exchange velocity between runoff and surficial soil
vr_i	m d^{-1}	Stormwater runoff resuspension velocity for subarea i
C_T^z	g m^{-3}	DeptG-weighted average $_{\mathrm{CT}}$ at time, t
V_E	m d^{-1}	Effective solute velocity in soil
W	mg m^{-2}	Average mass of waste added per LAU application
WP_i	cm	Soil moisture wilting point for subarea i
y_{op}	yr	Last year of operation of home garden
z	m	Distance down from soil surface
z_{sc}	m	Total depth of soil column
z_{till}	m	Distance from soil surface to bottom of home garden till (mixing) zone

G-A. References

U.S. EPA (Environmental Protection Agency). 1999. *Source Modules For Nonwastewater Waste Management Units (Land Application Units, Wastepiles, And Landfills) Background And Implementation For The Multimedia, Multipathway, And Multireceptor Risk Assessment (3mra) For Hwir99*. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Office of Solid Waste, Washington, DC. October. Available at http://www.epa.gov/osw/hazard/wastetypes/wasteid/hwirwste/pdf/risk/modules/s0056.pdf (accessed 19 February 2014).

Appendix G

Attachment G-B:
Particulate Emission Equations

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Attachment B: Particulate Emission Equations

G-B.1 Introduction

The HGSM estimates the annual average, area-normalized emission rate of contaminant mass adsorbed to particulate matter <30 µm in diameter, CE30 (g of contaminant m^{-2} d^{-1}), as well as annual average particle size distribution information in the form of the mass fractions of the total particulate emissions in four aerodynamic particle size categories: 30–15 µm, 15–10 µm, 10–2.5 µm, and <2.5 µm.

Table G-B-1 identifies the various release mechanisms and references for the algorithms implemented within the model. The SFS analysis only considered emissions due to wind erosion and tilling. This attachment describes the algorithms and assumptions used to estimate annual releases for each mechanism:

- **$E30_i$ (g of particulates ≤30 µm in diameter m^{-2} d^{-1})**. The annual average PM_{30} emissions rate due to release mechanism i, where mechanisms of release considered are summarized in Table G-B-1
- **Particle size range mass fractions**. The mass fractions of $E30_i$ in the aerodynamic particle size categories previously identified.

Table G-B-1. Summary of Mechanisms of Release of Particulate Emissions

Mechanism	Symbol	Home garden		Algorithm Reference
		Active	Fallow	
Wind erosion from open area	wd	X	X	Cowherd et al. (1985)
Vehicular activity	ve	X		U.S. EPA (1995)
Spreading/compacting or tilling	sc	X		U.S. EPA (1985)

G-B.2 Particulate Emission Rate ($E30_i$) Algorithms and Particle Size Range Mass Fractions

G-B.2.1 Wind Erosion from Open Fields ($E30_{wd}$)

The algorithm for the estimation of PM_{30} emissions due to wind erosion from an open field is based on the procedure developed by Cowherd et al. (1985). This algorithm was adapted for implementation in a computer code and is presented in detail here. $E30_{wd}$ is estimated in the source emission module. The user-specified input parameters are summarized in Table G-B-2.

To account for the fact that home gardens can differ in the degree of vegetation (veg'), surface roughness height (z'_0), and frequency of disturbances per month (fd'), different values are assigned to these parameters in the following equations according to whether the field is active or inactive (i.e., fallow). The value assignments are summarized in Table G-B-3 where veg, z_0, and fd are user input values.

Table G-B-2. Input Parameter Units and Definitions for E30$_{wd}$

Symbol	Units	Definition
asdm	mm	Mode of the aggregate size distribution
Lc	—	Ratio of the silhouette area of roughness elements too large to be included in sieving to total base area
veg	—	Fraction of surface covered with vegetation (fallow field)
Z_0	cm	Surface roughness height (fallow field)
S	w/w, %	Silt content of surface material
U+	m s-1	Observed or probable fastest mile of wind between disturbances
PE	—	Thornthwaite Precipitation Evaporation Index
U	m s-1	Mean annual wind speed
P	d yr-1	Mean number of days per year with ≥0.01 in precipitation
Fd	1 mo-1	Frequency of disturbance per month where a disturbance is defined as an action that exposes fresh surface material (fallow field)

Table G-B-3. Active/Inactive Fields Assignments for veg', z'$_0$, fd'

Symbol	Units	Active Field	Fallow Field
veg'	---	0.0	veg
z'_0	cm	1.0	z_0
fd'	1 mo^{-1}	fd	0.0

Step 1: Calculate U_{*t}

Calculate the threshold friction velocity, U_{*t} (m s^{-1}), the threshold wind speed for the onset of wind erosion:

$$U_{*_t} = 0.650 \cdot cf \cdot (asdm)^{0.425} \qquad \text{(G-B-1)}$$

Where

$$cf = \begin{cases} 1.0 & Lc < 2x10^{-4} \\ 1.05 + 50.18Lc - 647.89Lc^2 + 6863.50Lc^3 & 2x10^{-4} \leq Lc \leq 1x10^{-1} \end{cases} \qquad \text{(G-B-2)}$$

Table G-B-2 provides definitions of asdm and Lc. Lc is measured by inspection of a representative 1–m^2 transect of the site surface. Lc can range from 0 to 0.1. High Lc ($\geq 2 \times 10^{-4}$) increases the threshold friction velocity, which results in a relatively low or zero particulate emission rate due to wind erosion. Low Lc ($< 2 \times 10^{-4}$) is indicative of a bare surface with homogeneous finely divided material (e.g., an agricultural field). Such surfaces have a relatively low threshold friction velocity and increased particulate emissions. Equations (G-B-1 and G-B-2) were derived from work from Cowherd et al. (1985, Figures 3-4 and 3-5).

Step 2: Calculate U_t

U_t (m s^{-1}) is the threshold wind velocity at a height of 7.0 m (7.0 m is the typical weather station anemometer height). It is calculated using an equation from Cowherd et al. (1985, Equation, 4-3, with z=700 cm):

$$U_t = \frac{U_{*t}}{0.4} \ln\left(\frac{700}{z'_0}\right) \qquad z'_0 < 700 \qquad \text{(G-B-3)}$$

where z'_0 is the roughness height in cm. Values for z'_0 for various surface conditions are provided from Cowherd et al. (1985, Figure 3-6).

Step 3: Calculate E30wd

$E30_{wd}$ is the annual average emission rate of particulate matter <30 µm in diameter per unit area of the contaminated surface. Note that the methodology developed by Cowherd et al. (1985) estimates emission rates of particulate matter <10 µm (or $E10_{wd}$). $E30_{wd}$ can be approximated from $E10_{wd}$ with knowledge of the ratio between PM_{30} and PM_{10} for wind erosion. Cowherd (1998) advises that a good first approximation of this ratio is provided by the particle size multiplier information presented from U.S. EPA (1995) for wind erosion from open fields where PM_{30}/PM_{10} is equal to 2. Therefore, the HGSM incorporates a factor of 2 into the Cowherd et al. (1985) equations for $E10_{wd}$ to allow estimation of $E30_{wd}$.

For Sites with Limited Erosion Potential (U*t>0.75 m s-1)

The following equation was derived by using equations from Cowherd et al. (1985, Equations 4-1 to 4-3), applying a factor of 2 as previously discussed, and converting the units to g m^{-2} d^{-1}:

$$E30_{wd} = \begin{cases} \dfrac{11.12\left(U^+ - U_t\right)\left(1-veg'\right)fd'}{\left(PE/50\right)^2} \cdot \dfrac{24}{10^3} & U^+ \geq U_t \\ 0 & U^+ < U_t \end{cases} \qquad \text{(G-B-4)}$$

Data for mean annual U^+ and PE (Thornthwaite Precipitation Evaporation Index) for locations throughout the United States can be found in climatic atlases (e.g., U.S. DOC, 1968) and Cowherd et al. (1985, Figure 4-2), respectively. Cowherd et al. (1985) advise that, in the worst case, fd should be assumed to be 30 per month.

For Sites with Unlimited Erosion Potential (U_{*t}≤0.75 m s^{-1})

When U_{*t} is <0.75 m s^{-1}, the site is considered to have unlimited erosion potential, and $E30_{wd}$ is calculated using an equation from Cowherd et al. (1985, Equation 4-4) with a factor of 2 applied as previously discussed:

$$E30_{wd} = 0.072\left(1-veg'\right)\left(\frac{u}{U_t}\right)^3 g(x) \cdot 24\frac{h}{d} \qquad \text{(G-B-5)}$$

where

$$x = 0.886 \frac{U_t}{u} \tag{G-B-6}$$

$$g(x) = \begin{cases} 1.91 & 0 \le x < 0.5 \\ 2.2 - 0.6x & 0.5 \le x \le 1.0 \\ 2.9 - 1.3x & 1.0 < x \le 2.0 \\ 0.18\left(8x^3 + 12x\right)\exp\left(-x^2\right) & x > 2.0 \end{cases} \tag{G-B-7}$$

Where g(x) was derived from Cowherd et al. (1985, Figure 4-3). Data for U for locations throughout the United States can be found in climatic atlases (e.g., U.S. DOC, 1968).

Step 4: Apply Particle Size Range Mass Fractions

Particle size range mass fractions allow estimation of the fraction of the PM_{30} emitted that is in specific size fractions. As previously mentioned, Cowherd (1998) suggests using U.S. EPA's (1995) particle size multipliers provided for wind erosion from industrial fields. The U.S. EPA (1995) distribution was adapted to get the fraction of the emissions in the designated size categories as presented in Table G-B-4.

Table G-B-4. Aerodynamic Particle Size Range Mass Fractions for $E30_{wd}$ and $E30_{wp}$

30–15 µm	15–10 µm	10–2.5 µm	≤2.5 µm
0.4	0.10	0.3	0.2

G-B.2.2 Spreading/Compacting or Tilling Operations ($E30_{sc}$)

To estimate the rate of PM_{30} emissions due to spreading and compacting or tilling operations ($E30_{sc}$, g m^{-2} d^{-1}), the HGSM adapted an equation from U.S. EPA (1985, Equation 1, p. 11.2.2-1) that was developed to estimate emissions due to agricultural tilling in units of kilogram of particulate emissions per hectare per tilling (or spreading/compacting) event. The first two terms in Equation G-B-8 represent EPA's equation with the particle size multiplier for <30 µm applied.

$$E30_{sc} = (1.77)\, S^{0.6} \cdot N_{op} \cdot \frac{10^3 g}{kg} \cdot \frac{ha}{10^4 m^2} \tag{G-B-8}$$

Parameter definitions are provided in Table G-B-5. The particle size range mass fractions were developed from information provided in U.S. EPA (1985) and are presented in Table G-B-6.

Table G-B-5. Parameter Units and Definitions for $E30_{sc}$

Symbol	Units	Definition
S	w/w, %	Silt content of surface material[a] (1.7–88) [b]
$N_{op}{}^{c}$	1 d^{-1}	Number of tilling (or spreading and compacting) operations per day
$fcult$	—	Number of cultivations per application

[a] Silt is defined as particles <75 μm in diameter. Silt content is determined by the percent of loose dry surface material that passes through a 200-mesh screen using the ASTM-C-136 method (U.S. EPA, 1985).

[b] Values in parentheses are the ranges of source conditions that were tested in developing the U.S. EPA (1985) equation.

[c] N_{op} = (Nappl/365 × fcult)

Table G-B-6. Aerodynamic Particle Size Range Mass Fractions for $E30_{sc}$

30–15 μm	15–10 μm	10–2.5 μm	≤2.5 μm
0.24	0.12	0.34	0.30

G-B.3 Particle Size Range Mass Fractions for Total PM_{30} Emission Rate

Particle size range mass fractions characterizing the total annual average PM_{30} emission rate ($E30_i$ summed over all applicable mechanisms) are determined annually by applying the mechanism-specific mass fractions to the $E30_i$ estimates to obtain size-specific emission rate estimates $E_{i,j}$ (g m^{-2} d^{-1}) where subscript j identifies the particle size range (j=1 indicates 30–15 μm; 2 indicates 15–10 μm; 3 indicates 10–2.5 μm; and 4 indicates ≤2.5 μm). The total particle size range mass fraction, pmf_j, is calculated as

$$pmf_j = \frac{\sum_i E_{i,j}}{\sum_i E30_i} \tag{G-B-9}$$

G-B.4 Annual Average Constituent Emission Rate (CE30) Equations

The amount of mass lost due to wind and mechanical disturbances, $M_{loss,wd}$ (g m^{-2}), estimated using Equation G-92 and accumulated throughout the simulated year is used to estimate CE30 (g m^{-2} $d^{-}1$), the annual average, area-normalized emission rate of contaminant mass adsorbed to particulate matter <30 μm in diameter.

$$CE30 = \frac{M_{loss,wd}}{365} \tag{G-B-10}$$

Equation G-B-10 is directly applicable to the home garden during both the active and fallow periods.

G-B.5 Estimation of First-Order Loss Rate (k_{wd})

An equation for k_{wd} was derived by performing a mass balance on the surface layer of the "soil" column to a depth of dz (the depth of the surface soil column cell) and considering losses due to wind and mechanical activity only:

$$\frac{\partial C_T}{\partial t} = -k_{wd} C_T \qquad \text{(G-B-11)}$$

where

$$k_{wd} = \frac{1}{dz} \cdot \frac{K_d}{K_{TL}} \cdot \frac{g}{10^{6}\, \mu g} \cdot \sum_i E30_i \qquad i \neq un \qquad \text{(G-B-12)}$$

G-B.6 References

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Appendix G

Attachment G-C:
Home Garden Input Parameters Used for SFS Analysis

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Attachment C: Home Garden Source Model Input Parameters Used for SFS Analysis

Table G-C-1 lists the HGSM input parameter values used to model particulate emissions, erosion, and leaching from manufactured soil containing SFS applied to residential gardens. Each variable entry also includes a parameter description, units, and a data source for each variable. Variables are grouped by national constants, variables that are derived from other variables, site-specific soil and land-use variables, and location and sites-specific variables.

Table G-C-1. Source Parameters

Parameter	Description	Value	Reference
Constants			
AppDepth	Depth of waste incorporation (m)	0.2	Per EPA directive
asdm	Mode of the aggregate size distribution (mm)	0.5	Based on U.S. EPA, 1989
CutOffYr	Operating life (years)	40	U.S. EPA (typical value for manufactured soils)
fwmu	Fraction of waste in WMU (Waste Management Unit) (unitless)	Set to 1, assuming that waste is not mixed	
mt	Distance vehicle travels on (m)	0	Set to 0, assuming that no regular vehicular activity occurs on the agricultural field
Nappl	Waste applications per year (1 year^{-1})	1	Per EPA directive
nv	Vehicles per day on HGSM (1 day^{-1})	0	Set to 0, assuming that no regular vehicular activity occurs on the agricultural field
NyrMax	Maximum model simulation time (years)	200	Chosen to ensure that the entire period in which receptors may be exposed was modeled. Value is based on the fact that exposure must begin sometime during the operation of the unit; the maximum operation of the unit is 40 years, and the exposure maximum exposure duration is 100 years.

Table G-C-1. Source Parameters

Parameter	Description	Value	Reference
td	Time period of deposition (years)	200	Assumed time period for modeling
Vv	Volatilization velocity (m day^{-1})	0	Assumption that degradation rates account for volatilization
Derived			
foc_soil	Fraction organic carbon for soil (unitless)	Calculated using % organic matter	Calculated based on U.S. EPA EPACMTP, 1997b
focW	Fraction organic carbon of waste solids (unitless)	set to native soil (foc_soil)	set to native soil (foc_soil)
LS	USLE length-slope factor (unitless)	Calculated from X and Theta	Calculated from length and slope based on Williams and Berndt, 1977
Rappl	Wet waste application rate (Mg m^{-2} year^{-1})	Rappl = Nappl x application rate x ha m^{-2}	Process Design Manual, U.S. EPA 1995
X	Flowlength for local watershed (m)	Default flow lengths by slope (Theta)	Lightle and Weesies, 1998
Distributions			
Area_LAU	Area of the home garden (m^2)	Set to 404.7 square meters (i.e., 0.1 acre)	Home Garden Scenario
DRZ	Root zone depth (cm)	Uniform distribution min=value for shallow-rooted crops max=value for deep-rooted crops see Table 3	Dunne and Leopold, 1978
effdust	Dust suppression control efficiency for controlled areas (unitless)	normal distribution min=0 max=1 mean=0.5 stdev=0.3	Based on U.S. EPA, 1989
Lc	Roughness ratio (unitless)	Lognormal distribution min 1E-04 max 1E-03 mean 3E-04 stdev 0.304	U.S. EPA, 1989
veg	Fraction vegetative cover (fraction)	Normal distribution min=0.8 max=1.0 mean=0.9 stdev=0.1	Protective distribution for screening purposes

Table G-C-1. Source Parameters

Parameter	Description	Value	Reference
Hydrologic Soil Group-Specific			
CN	SCS curve number (unitless)	HGSM: based on cover type and hydrologic soil group all other areas: uniform distribution group A: min=39, max=72 group B: min=61, max=81 group C: min=74, max=88 group D: min=80, max=91	Based on Wanielista and Yousef, 1993
SMFC	Soil moisture field capacity (Volume %)	Based on average hydrologic soil group for each soil texture	Carsel and Parrish, 1988
SMWP	Soil moisture wilting point (Volume %)	Based on average hydrologic soil group for each soil texture	Carsel and Parrish, 1988
SoilHydGrp	Hydrologic soil group	Based on hydrologic soil properties	USDA, 1994 (STATSGO)
Landuse-Specific			
C	USLE cover management factor (unitless)	Set to 0.1	Based on Parameter Guidance Document, U.S. EPA, 1997a
fcult	Number of cultivations per application (unitless)	Set to 5	Based on U.S. EPA, 1989
fd	Frequency of surface disturbance per month on active HGSM (1 mo^{-1})	Calculated from cultivations per application	Based on U.S. EPA, 1989
P	USLE supporting practice factor (unitless)	Set to 1	Wanielista and Yousef, 1993
PI	Percent impervious (percent)	HGSM: 0%	Center for Watershed Protection, 1998
zruf	Roughness height (cm)	Set to 1	Based on information in U.S. EPA, 1989
Regional			
AirTemp	Long-term average air temperature (°C)	Calculated from hourly ambient air temperature data	U.S. DOC and U.S. DOE, 1993
R	Meteorologic parameter - USLE rainfall/erosivity factor (1 year^{-1})	Based on 22-year station rainfall records	Wischmeier and Smith, 1978
Twater	Waterbody temperature (° K)	Based on HUC region	van der Leeden et al., 1990
uw	Meteorologic parameter - mean annual wind speed (m sec^{-1})	Calculated from hourly windspeed data	U.S. DOC and U.S. DOE, 1999
Scenario-Specific			
Area_buffer	Area of the buffer (m^2)	10m x length of source - where length of source is the total buffer length	Buffer width based on 40 of 1993
Site-Specific			
SiteLatitude	Site latitude (degrees)	Angular distance in degrees north or south of the equator	U.S. DOC and U.S. DOE, 1993

Table G-C-1. Source Parameters

Parameter	Description	Value	Reference
Soil Texture-Specific			
%Organic-Matter	Percent organic matter for surface soil (percent)	By predominant soil texture; calculated based on area-weighted average across all map units for region (Appendix D)	USDA, 1994 (STATSGO)
BD	Dry bulk density (g cm^{-3})	Surface soil: calculated from saturated water content (WCS); see Appendix D	Surface soil: Calculated based on U.S. EPA EPACMTP, 1997b from saturated water content (WCS)
BDw	Dry bulk density for waste solids (g cm^{-3})	Set to bulk density for the native soil	Gunn et al., 2004
K	USLE soil erodibility factor (Kg m^{-2})	Area-weighted average for each soil texture within meteorological region (Appendix D)	USDA, 1994 (STATSGO)
Ksat	Saturated hydraulic conductivity (cm h^{-1})	Based on surface soil texture	Carsel and Parrish, 1988
SMb	Soil moisture coefficient (unitless)	Based on surface soil texture; see Appendix D	Clapp and Hornberger, 1978
SoilTexture	Texture of surface soil	Distribution of agricultural soil textures within meteorological region (Appendix D)	USDA, 1994 (STATSGO)
Ss	Silt content of soil (mass percent)	Area-weighted average silt content for each soil texture within meteorological region (Appendix D)	USDA, 1994 (STATSGO)
Theta	Slope of the local watershed (degrees)	Area-weighted average slope for each soil texture within meteorological region (Appendix D)	USDA, 1994 (STATSGO)
WCS	Saturated volumetric water content, porosity for soil (ml cm^{-3})	Based on surface soil texture; see Appendix D	Carsel and Parrish, 1988
WSpH	Watershed soil pH (pH units)	Area-weighted average value for each soil texture within meteorological region (Appendix D)	USDA, 1994 (STATSGO)

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Appendix G

Attachment G-D: Source Air Dispersion and Deposition Modeling

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Attachment G-D:
Source Air Dispersion and Deposition Modeling

The constituent-specific emission rates (predicted by the source model) are combined with air concentrations and deposition rates (supplied by the air dispersion model) to calculate constituent-specific vapor- and particle-phase air concentrations and deposition rates. These constituent-specific air concentrations and deposition rates are used in estimating aboveground produce concentrations as shown in the equations presented in **Appendix H**. Rather than performing new air dispersion modeling, the SFS evaluation used pre-existing dispersion and deposition rates generated as part of EPA's evaluation of dioxins in biosolids applied to agricultural land (U.S. EPA, 2003b), as well as ongoing biosolids-related risk assessment work. The biosolids dataset reflects national-scale dispersion modeling for farms with areas that span a range from 111 to 180 acres. The SFS evaluation used the portion of these data specific to regions where SFS might be used (i.e. the "economic feasibility areas" described in Section 3.2.2). Although dispersion data from farm-size applications will likely overestimate impacts due to residential gardening in a 404.7 sq. meter (i.e., 0.1 acre) area, the simplifying approach is appropriate for this conservative SFS screening.

The remainder of Attachment G-D describes the biosolids modeling that was performed to estimate the location-specific dispersion and deposition factors that were mapped to and applied in modeling the SFS gardening scenario.

G -D.1 Conceptual Air Model

Air dispersion and deposition modeling uses a computer-based set of calculations to estimate ambient ground-level constituent concentrations and deposition rates associated with constituent releases from land-use practices and wind erosion. The dispersion model uses information on meteorology (e.g., wind speed, wind direction, temperature) to estimate the movement of constituents through the atmosphere. Movement downwind is largely determined by wind speed and wind direction. Dispersion around the centerline of the plume is estimated using empirically derived dispersion coefficients that account for the movement of pollutants in the horizontal and vertical directions. Pollutant movement from the atmosphere to the ground is also modeled, to account for deposition processes driven by gravitational settling and removal by precipitation.

The air model used in biosolids assessment (and, by extension, this SFS evaluation) is the Industrial Source Complex–Short Term Model, version 3 (ISCST3),[1] a steady-state Gaussian plume model used for modeling concentration, dry deposition, and wet deposition from point, area, volume, and open-pit sources. ISCST3 was designed primarily to support EPA's regulatory modeling programs. The ISCST3 estimates annual average air concentration of dispersed constituents and annual deposition rate estimates for vapors and particles at various locations in and surrounding a source. The air concentrations and deposition rates developed by ISCST3

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[1] Modeling the deposition of particle-bound metals released from soil (e.g., via windblown emissions) onto plant surfaces requires a model capable of estimating air concentrations and deposition rates. Although SCREEN3 (i.e., the model used to conduct SFS Phase I screening of inhalation exposures) is an appropriate model for assessing maximum inhalation exposures, it does not calculate wet or dry deposition. The ISCST3 model was therefore chosen to support the refined probabilistic modeling of the SFS home garden scenario.

were based on a unit emission rate (i.e., 1 µg s^{-1} m^{-2}). The resulting air concentrations are called unitized air concentrations (UACs) (i.e., µg m^{-3} per unit emission rate of 1 µg s^{-1} m^{-2}), and these are multiplied by the constituent-specific emission rates (predicted by the source models) and appropriate conversion factors to calculate chemical-specific vapor- and particle-phase air concentrations and deposition rates. **Appendix H** presents the equations used to develop the final constituent-specific ambient air estimates and deposition rates.

G-D.2 Air Model Inputs

The key inputs to the air dispersion and deposition model include the following:

- **Emission rates.** The air concentrations and deposition rates developed by ISCST3 were based on a unit emission rate (i.e., 1 µg s^{-1} m^{-2}). The resulting air concentrations are called UACs (i.e., µg m-3 per unit emission rate of 1 µg s^{-1} m^{-2}).
- **Surface area of the farm.** As discussed previously, the size of the farm was varied stochastically by sampling from a distribution using data from Hoppe et al. (2001) reflecting lifestyle farms.
- **Meteorological data for the site.** Meteorological conditions at the site were modeled using surface and upper air data obtained for the 41 climatic regions (See Appendix D).
- **Locations of potential receptors.** Receptors were placed uniformly over the modeling domain. Outputs for these receptor points were averaged and used to estimate the mean air concentrations and deposition rates.
- **Particle diameter and mass fraction.** Particle diameter and mass fraction are also required inputs when modeling deposition. As input, a fixed distribution, consistent with the Multimedia, Multipathway, Multireceptor Risk Assessment Modeling System (3MRA) air modeling (U.S. EPA, 1999b), was used. The four size categories modeled were 30–15 µm, 15–10 µm, 10–2.5 µm and <2.5 µm, with mass fractions of 0.4, 0.1, 0.3, and 0.2, respectively.

G-D.3 Air Model Outputs

The air dispersion and deposition data were used to calculate environmental media concentrations and food chain concentrations. The dispersion model outputs included annual average air concentrations of the vapors and particles, wet deposition of the vapors and particles, and dry deposition of the particles. Dry deposition of the vapors was also calculated, but outside the dispersion model, based on an assumed dry deposition velocity of vapors of 1 cm s^{-1}. These outputs were produced for the grid of receptor points. These outputs were processed and averaged in a GIS to produce areal averages for the following, based on the unit emission rate approach:

- Air concentration of vapors and particles
- Wet deposition of vapors and particles
- Dry deposition of particles.

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Appendix G

Attachment G-E :
Arsenic Soil Partitioning (Kd) Evaluation

In environmental modeling, how a constituent partitions between soil components (i.e., how much adsorbs to soil solids, compared to how much transfers into soil water) is addressed through the use of a soil/water partition coefficient, or Kd. The higher the Kd, the more constituent adsorbs to soil solids, rather than transferring into soil water). Under the home garden scenario, the properties and characteristics of the manufactured soil are assumed to mimic those of natural soil in the area. Accordingly, the SFS-specific screening levels (generated as part of Phase II national-scale modeling) were developed based on soil Kd values from U.S. EPA (2005). Given the complexities of arsenic behavior in soil, an analysis was performed examining the impact of Kd distributions on SFS arsenic screening levels**.** To better understand the uncertainties and the sensitivity of these screening values to Kd, source modeling was also performed for arsenic using a distribution of SFS waste-specific Kds. This distribution was developed using the full set of whole waste/leachate pairs presented in Appendix B (i.e., the SFS total waste concentration for each sample was divided by the corresponding leachate concentration for that sample). It is important to note that the SFS waste-specific Kd distribution reflects partitioning in pure SFS, and therefore would not accurately estimate partitioning in soils. Modeling results using the SFS waste-specific Kd distribution can be seen as bounding estimates.

Table G-E-1 compares the arsenic SFS-derived waste Kd distribution to the soil Kd distribution from U.S. EPA (2005). The SFS-derived waste Kd distribution is relatively narrow and the Kds are generally well below the soil-Kds. The minimum Kd values for the two distributions are very similar, however, the mean waste Kd is approximately 6 times lower, and the maximum waste Kd is about 10 times lower than corresponding soil-Kd values. Given that Kd is a measure of sorption to solids, the SFS-waste Kd distribution would therefore tend to estimate lower retention of arsenic in the soil and higher releases to groundwater than would the soil Kd distribution. This is not surprising, as soils tend to have much higher levels of adsorbent sources (e.g., Fe, Al, and Mn hydroxides and organic matter) compared to SFS, and would therefore retain more arsenic in the solid phase. **Figure G-E-1** provides a graphical comparison of the cumulative distributions for the soil Kds and SFS waste-specific Kds.

Table G-E-1. Arsenic: Comparison of Soil Kds with SFS Waste Kds (L kg^{-1})

	Soil Kd Distribution (U.S. EPA, 2005)		SFS Waste-Specific Kd Distribution (Derived based on Appendix B data)	
Statistic	**Kd**	**log Kd**	**Kd**	**log Kd**
Minimum	2	0.3	5	0.7
Mean	1,585	3.2	241	2.4
Maximum	19,952	4.3	1,960	3.3

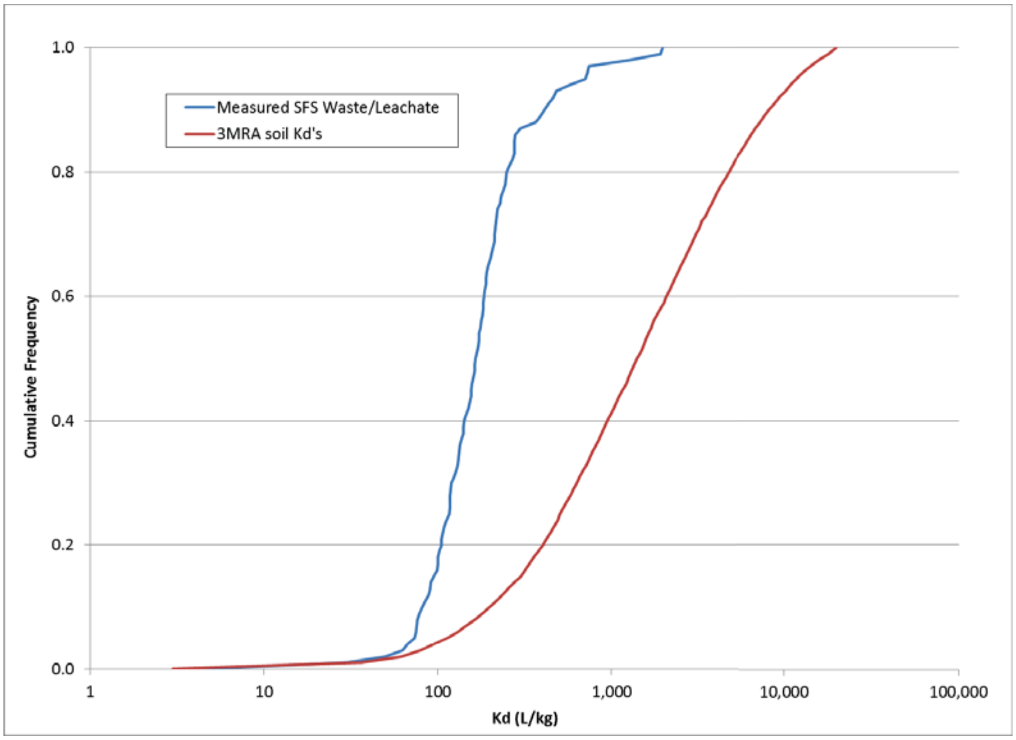

Figure G-E-1. Comparison of cumulative distributions for Arsenic Kd: SFS Waste Partition Coefficients versus Soil Partition Coefficients (L kg^{-1})

Output from the analysis was a distribution of soil/produce and groundwater risks and corresponding SFS-specific screening levels. **Table G-E-2** presents the screening levels that were developed using the two Kd distributions. As seen from this table, it is clear that application of the SFS waste-specific Kd distribution results in a significantly lower screening level for the groundwater pathway. However, it is important to note that this lower screening level is nearly identical to the soil/produce screening level of 8.0 mg kg^{-1} obtained using the soil Kd distribution. The similarity between the established SFS-specific screening level and the bounding waste-specific estimate fosters a high level of confidence that the SFS screening level will be protective of human health under a range of pathways and environmental conditions.

Table G-E-2. Home Gardening 90th Percentile Arsenic Screening Levels for SFS in Manufactured Soil

Pathway	Arsenic SFS Screening Levels (mg kg^{-1})	
	Based on Soil Kd Distribution	Based on SFS Waste-Specific Kd Distribution
Soil/Produce	8.0	9.5
Groundwater	59	7.7

References

U.S. EPA (Environmental Protection Agency). 2005. *Partition Coefficients for Metals in Surface Water, Soil, and Waste*. EPA/600R-05/074. U.S. Environmental Protection Agency, Office of Research and Development. July. Available at http://www.epa.gov/athens/publications/reports/Ambrose600R05074PartitionCoefficients.pdf (accessed 9 December 2013).

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Appendix H:
Fate, Transport, Exposure, and Hazard Calculations for Human Health and Ecological Effects

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Table H1-1. Total Concentration in Air (mg m^{-3})

C_{air}

$$C_{air} = Q \times \left\lfloor F_v \times C_{yv} + \left(1 - F_v\right) \times C_{yp} \right\rfloor \times 0.001$$

Name	Description	Value
F_v	Fraction of air concentration in vapor phase (unitless)	F_v=0; modeled constituents present only in particle phase
Q	Emission rate from source (g s^{-1}-m^{-2})	Calculated from source model output
C_{yp}	Normalized particulate air concentration (μg-s-m^2 g^{-1}-m^{-3})	Calculated from dispersion modeling
C_{yv}	Normalized vapor-phase air concentration (ug-s-m^2 g^{-1}-m^{-3})	Calculated from dispersion modeling
0.001	Conversion factor (mg μg^{-1})	

Source: Based on U.S.EPA IEM, 1998 and U.S.EPA HHRAP, 2005

Table H2-1. Particulate Deposition Onto Plants
(mg m^{-2}-yr^{-1})

D_p

$$D_p = \left(1000 \times Q\right) \times \left(1 - F_v\right) \times \left(D_{ydp} + \left(F_w \times D_{ywp}\right)\right)$$

Name	Description	Value
Q	Emission rate from source (g s^{-1}-m^{-2})	Calculated from source model output
F_v	Fraction of air concentration in vapor phase (unitless)	F_v=0; modeled constituents present only in particle phase
D_{ydp}	Normalized annual average dry deposition from particle phase (ug-s-m^2 g^{-1}-m^{-3})	Calculated from dispersion modeling
F_w	Fraction of wet deposition adhering to plant surface (unitless)	Set to 0.6 (U.S. EPA HHRAP, 2005)
D_{ywp}	Normalized annual average wet deposition from particle phase (ug-s-m^2 g^{-1}-m^{-3})	Calculated from dispersion modeling
1000	Conversion factor (mg g^{-1})	

Source: Based on U.S.EPA IEM, 1998 and U.S.EPA HHRAP, 2005

Table H3-1. Concentration in Aboveground Vegetation Due to Deposition, Transfer, and Uptake ($mg\ kg^{-1}$ WW)

P_{ag}

$$P_{ag} = \left(P_d + P_v + P_r\right) \times \frac{100 - MAF}{100}$$

Name	Description	Value
P_d	Vegetative concentration due to particle deposition ($mg\ kg^{-1}$ DW)	Calculated; see Table H3-3
P_v	Vegetative concentration due to air-to-plant transfer ($mg\ kg^{-1}$ DW)	P_v=0; modeled constituents present only in particle phase
P_r	Aboveground vegetation concentration due to root uptake ($mg\ kg^{-1}$ DW)	Calculated; see Table H3-2
MAF	Plant tissue-specific moisture adjustment factor to convert DW concentration into WW (percent)	Exposed fruit: 85 Exposed vegetables: 91.77 Protected fruit: 89.59 Protected vegetables: 80.23
100	Conversion factor to percent (unitless)	

Source: Based on U.S.EPA IEM, 1998 and U.S.EPA HHRAP, 2005
Considered exposed and protected fruits and vegetables. Pv and Pd are always assumed to be zero for protected fruit and vegetables.

Table H3-2. Aboveground Vegetation Concentration Due to Root Uptake ($mg\ kg^{-1}$ DW)

P_r

$$P_r = C_{soil} \times B_r$$

Name	Description	Value
C_{soil}	Concentration of contaminant in soil ($mg\ kg^{-1}$)	Output from Source Model
B_r	Soil-to-plant bioconcentration factor (($mg\ kg^{-1}$ DW plant) ($mg\ kg^{-1}$ soil)$^{-1}$)	Chemical data; see Appendix F

Source: Based on U.S. EPA IEM, 1998 and U.S.EPA HHRAP, 2005

Table H3-3. Vegetative Concentration Due to Particle Deposition (mg kg^{-1} DW)

P_d

$$P_d = \frac{D_p \times R_p}{Y_p \times K_{pPar}}$$

Name	Description	Value
D_p	Particle deposition term for plants (mg m^{-2}-yr^{-1})	Calculated; see Table H2-1
R_p	Interception fraction (unitless)	Exposed fruit: 0.48 Exposed vegetables: 0.48
Y_p	Crop yield (kg DW m^{-2})	Exposed fruit: 1.17 Exposed vegetables: 1.17
K_{pPar}	Plant surface loss coefficient, particulate (1 yr^{-1})	Chemical data; see Appendix F

Source: Based on U.S. EPA IEM, 1998 and U.S. EPA HHRAP, 2005(Steady-state solution)

Table H3-4. Concentration in Belowground Vegetation Due to Root Uptake (mg kg^{-1} DW)

P_{bg}

For metals:

$$P_{bg} = C_{soil} \times Br_{root} \times DW_r$$

$$DW_r = \frac{100 - MAF_{bg}}{100}$$

Name	Description	Value
C_{soil}	Concentration of contaminant in soil (mg kg^{-1})	Output from Source Model
Br_{root}	Soil-to-plant bioconcentration factor for roots ((mg kg^{-1} DW plant) (mg kg^{-1} soil)$^{-1}$)	Chemical data; see Appendix F
DW_r	Dry weight fraction for root vegetables (unitless)	Calculated above
MAF_{bg}	Plant tissue-specific moisture adjustment factor for root vegetables to convert DW concentration into WW (percent)	Below ground vegetables: 87.32

Source: Based on U.S.EPA IEM, 1998 and U.S. EPA HHRAP, 2005

Table H4-1. Average Daily Dose from Total Ingestion
(mg kg^{-1} BW d^{-1})

$ADD_{Total\ Ingestion}$

$$ADD_{TotalIngestion} = ADD_{soil} + ADD_{produce} \text{ (soil pathways)}$$
$$ADD_{TotalIngestion} = ADD_{dw} \quad \text{(groundwater pathway)}$$

Name	Description	Value
ADD_{soil}	Average daily dose from ingestion of soil (mg kg^{-1} BW d^{-1})	Calculated; see Tables H4-2
$ADD_{produce}$	Average daily dose from consumption of produce (mg kg^{-1} BW d^{-1})	Calculated; see Tables H4-3
ADD_{dw}	Average daily dose from ingestion of drinking water (mg kg^{-1} BW d^{-1})	Calculated; see Tables H4-3

Table H4-2. Average Daily Dose from Ingestion of Soil
(mg kg^{-1} BW d^{-1})

ADD_{soil}

$$ADD_{soil} = \frac{C_{soil} \times CR_{s} \times F_{soil}}{BW} \times 0.000001$$

Name	Description	Value
C_{soil}	Concentration of contaminant in soil (mg kg^{-1})	Output from Source Model
CR_{s}	Soil ingestion rate (mg day^{-1})	Human exposure data; see Appendix I
F_{soil}	Fraction of ingested soil that is contaminated (unitless)	1 (i.e. 100%)
BW	Body weight (kg)	Human exposure data; see Appendix I
0.000001	Conversion factor (kg mg^{-1})	

Source: Based on U.S. EPA IEM, 1998 and U.S. EPA HHRAP, 2005

Table H4-3. Average Daily Dose from Consumption of Produce
(mg kg^{-1} BW d^{-1})

$ADD_{produce}$

$$ADD_{produce} = \frac{1}{1000}\sum_{i=1}^{i=n} P_i \times CR_{Pi} \times F_{Pi} \times \left(1 - L_{Pi}\right)$$

Name	Description	Value
P_i	Concentration in vegetation as wet weight (g kg^{-1}WW)	Calculated; see Tables H3-1 and H3-4
CR_{Pi}	Daily human consumption rate of produce (g WW kg^{-1} BW day^{-1})	Human exposure data; see Appendix I
F_{Pi}	Fraction of vegetables grown in contaminated soil (unitless)	Human exposure data; see Appendix I
L_{Pi}	Food preparation loss (unitless)	Human exposure data; see Appendix I
0.001	Conversion factor (g kg^{-1})	

Source: Based on U.S.EPA IEM, 1998 and U.S. EPA HHRAP, 2005

Table H4-4. Average Daily Dose from Ingestion of Drinking Water
(mg kg^{-1} BW d^{-1})

ADD_{dw}

$$ADD_{dw} = C_{dw} \times CR_{dw} \times F_{dw} \times 0.001$$

Name	Description	Value
C_{dw}	Concentration of contaminant in drinking water (mg L^{-1})	Output from EPACMTP Model
CR_{dw}	Drinking water ingestion rate (mL kg^{-1} d^{-1})	Human exposure data; see Appendix I
F_{dw}	Fraction of ingested drinking water that is contaminated (unitless)	1 (i.e. 100%)
0.001	Conversion factor (L ml^{-1})	

Source: Based on U.S. EPA IEM, 1998 and U.S. EPA HHRAP, 2005

Table H4-5. Lifetime Average Daily Dose
(mg kg^{-1} d^{-1})

LADD

$$LADD = \frac{ADD \times ED \times EF}{AT \times 365}$$

Name	Description	Value
ADD	Average daily dose (mg kg^{-1}-day^{-1})	Calculated; see Tables H4-1 to H4-4
ED	Exposure duration (yr)	Human exposure data; see Appendix I
EF	Exposure frequency (d yr^{-1})	Human exposure data; see Appendix I
AT	Averaging time (yr)	Human exposure data; see Appendix I
365	Conversion factor (days yr^{-1})	

Source: Based on U.S.EPA IEM, 1998

Table H5-1. Unitized Human Dose Ratio Due to Ingestion
(unitless)

UDR

$$UDR = \frac{ADD \text{ or } LADD}{HealthBenchmark}$$

Name	Description	Value
ADD	Average daily dose for Noncarcinogens (mg kg^{-1} day^{-1})	Calculated; see Tables H4-1 to H4-4
LADD	Lifetime average daily dose for Carcinogens (mg kg^{-1} day^{-1})	Calculated; see Table H4-5
Health Benchmark	RfD for noncancer or cancer risk level of 1E-05/CSF for cancer (mg kg^{-1} day^{-1})	Chemical data; see Appendix F

Table H6-1. Unitized Ecological Dose Ratio (unitless)

UDR

$$UDR = \frac{SoilConc}{Eco - SSL}$$

Name	Description	Value
SoilConc	Annual average SFS constituent-specific soil concentration (mg kg^{-1})	Calculated by home garden source model; see Appendix G
Eco-SSL	EPA's Ecological Soil Screening Levels (mg kg^{-1})	Chemical data; see Appendix F

H.1 References

U.S. EPA (Environmental Protection Agency). 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*. Update to EPA/600/6-90/003 Methodology for Assessing Health Risks Associated With Indirect Exposure to Combustor Emissions. EPA 600/R-98/137. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Cincinnati, OH. December.

U.S. EPA (Environmental Protection Agency). 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*. EPA 530-R-05-006. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response. September. Available (with supporting documentation in a self-extracting file) at http://www.epa.gov/osw/hazard/tsd/td/combust/risk.htm (accessed 19 March 2012).

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Appendix I

Human Exposure Factors

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Appendix I:
Human Exposure Factors

This appendix describes the collection or derivation of the human exposure factors that were used in the SFS beneficial use in soils risk analysis. Exposure factors define the magnitude, frequency, duration, and routes of exposure to SFS constituents that an individual may experience.

The term "exposure," as defined by EPA's exposure guidelines (1992), is the condition that occurs when a contaminant comes into contact with the outer boundary of the body. The exposure of an individual to a contaminant completes an exposure pathway. After the body is exposed, the constituent can cross the outer boundary and enter the body. The amount of contaminant that crosses and is available for adsorption at internal exchange boundaries is referred to as the "dose" (U.S. EPA, 1992).

Exposure factors are data that quantify human behavior patterns (e.g., ingestion rates of soil and fruit) and characteristics (e.g., body weight) that affect human exposure to environmental contaminants. These data can be used to construct realistic assumptions concerning an individual's exposure to and subsequent intake of a constituent in the environment. The exposure factors data also enable EPA to differentiate the exposures of individuals of different ages (e.g., a child versus an adult). **Section I.1** presents an overview of the receptors and selected exposure pathways considered for this analysis. The derivation and values used for the human exposure factors in this risk assessment are described in **Section I.2**.

I.1 Receptors and Exposure Pathways

In the home gardening scenario, both adult and child members of a residential family are exposed to chemicals through the use of SFS manufactured soil on their property. The adults are 20 years old or older when exposure begins, and the children begin exposure at 1 year of age.

As described in Section 5.3, Phase II refined probabilistic modeling was performed for four constituents to evaluate potential exposures under the home garden soil/produce ingestion pathway: arsenic, lead, manganese, and nickel. In addition, arsenic was also retained for more refined evaluation under the groundwater pathway. The Phase II methodology as implemented generates data to support the development of SFS concentrations based on cumulative exposure across pathways or for individual pathways. **Table I-1** lists each receptor along with the specific ingestion exposures that apply to that receptor for a given pathway. For the home gardening groundwater pathway, receptors are exposed through the ingestion of groundwater used as a drinking water source. For the soil/produce pathway, adult and child gardeners are exposed via ingestion of soil, and homegrown above- and belowground produce.[1]

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[1] Although receptor exposures via the groundwater and soil/produce pathways were evaluated concurrently, separate target SFS concentrations were developed for each pathway based on analyses discussed in Section 5.3.5 and Appendix J that indicate that these exposures will not occur within the same timeframe.

Table I-1. Receptors and Ingestion Exposure Pathways

Receptor	Groundwater Pathway	Soil/Produce Pathway					
	Drinking Water	**Soil**	**Protected Vegetables**	**Exposed Vegetables**	**Root Vegetables**	**Protected Fruits**	**Exposed Fruits**
Adult Resident	√	√	√	√	√	√	√
Child Resident	√	√	√	√	√	√	√

I.1.1 Childhood Exposure

Children are an important subpopulation to consider in a risk assessment because they are likely to be more susceptible to exposures than adults. For example, children may eat more fruit per unit of body weight than adults. This higher intake-rate-to-body-weight ratio can result in a higher average daily dose (ADD) for children than for adults.

As children mature, their physical characteristics and behavior patterns change. To capture these changes in the analysis, the life of a child was considered in stages represented by the following cohorts: Cohort 1 (aged 1–5), Cohort 2 (aged 6–11), Cohort 3 (aged 12–19), and Cohort 4 (aged 20–70). Each cohort is associated with distributions of exposure parameter values that are required to calculate exposure to an individual. The exposure parameter distributions for each cohort reflect the physical characteristics and behavior patterns of that age range. Data from the 2011 *Exposure Factors Handbook* (EFH) and *Child-Specific Exposure Factors Handbook* (CSEFH; U.S. EPA, 2008a) were used to derive distributions appropriate for each cohort. The distributions for Cohort 4, the 20- to 70-year-olds, were the same as those used for adult receptors.

The development of the child exposure parameters consisted of the following two steps:

1. Define the start age of the child
2. Select the exposure duration of the child.

To capture the higher intake-rate-to-body-weight ratio of children, a start age of 1 was selected. The distribution of exposure durations for Cohort 1 (aged 1–5) was used to define exposure duration for each of the Monte Carlo iterations in the probabilistic analysis.

I.1.2 Exposure Pathways

Human receptors may come into contact with chemicals present in environmental media through a variety of pathways. In general, exposure pathways are either direct (e.g., ingestion of groundwater) or indirect (e.g., food chain pathways). The exposure pathways considered in this assessment were ingestion of soil, drinking water, and produce.

I.1.2.1 Ingestion of Soil

In the home gardening scenario, both adult and child receptors were exposed to soil based on incidental ingestion, mostly as a result of hand-to-mouth behavior.

I.1.2.2 Ingestion of Drinking Water

In the home gardening scenario, both the adult and child receptors were assumed to ingest groundwater contaminated by SFS constituents leaching from the manufactured soil used in the garden.

I.1.2.3 Ingestion of Above- and Belowground Produce

The home gardening scenario included ingestion of the following categories of produce: exposed fruit, protected fruit, exposed vegetables, protected vegetables, and root vegetables. For aboveground produce, the term "exposed" indicates that the edible portion of the plant is exposed to the atmosphere, and the term "protected" indicates that the edible portion of the plant is protected from the atmosphere by an inedible skin. Home gardeners were assumed to grow their fruits and vegetables in manufactured soil. The "aboveground" fruits and vegetables were assumed to become contaminated via soil and air deposition. "Belowground produce" refers to root crops grown by the gardener and were assumed to become contaminated via root uptake. The evaluation used data developed by EPA on home gardeners, as well as data on the general population, to define the amount of home grown produce consumed by adult and child receptors.

I.2 Exposure Parameters Used in Probabilistic Analysis

I.2.1 Introduction

The general methodology for collecting human exposure data for the probabilistic assessment used the EFH (U.S. EPA, 2011) and *Child-Specific Exposure Factors Handbook* (CSEFH; U.S. EPA, 2008) in one of the following three ways:

1. When data were adequate (most input variables), selected parametric distributions were fit to the EFH or CSEFH data. The best distribution was then chosen using the chi-square measure of goodness of fit. Parameter uncertainty information (e.g., averages, standard deviations) was also derived.
2. If percentile data were not adequate for statistical model fitting, in most cases distributions were selected based on the results for other age cohorts or, if no comparable information was available, by assuming lognormal as a default distribution and reasonable coefficients of variation (CVs).
3. Other variables for which data were not adequate for either approaches 1 or 2 above were fixed at EFH-recommended mean values or according to established EPA policy.

Table I-2 summarizes all of the parameters that were varied in the probabilistic assessment. Fixed variables are presented later in Section I.2.4.

Probabilistic risk assessments involve "sampling" values from probability distribution functions (PDFs) and using the values to estimate risk. In some cases, distributions are infinite, and there is a probability, although very small, that very large or very small values might be selected from the distributions. Because selecting extremely large or extremely small values is unrealistic (e.g., the range of adult body weights is not infinite), maximum and minimum values were imposed on the distributions consistent with a methodology developed for the 3MRA modeling system. For the probabilistic analyses, the maximum intake rates for most food items were defined as 2 × (mean + 3 standard deviations). For exposed fruit (adult gardener) and exposed vegetable (children aged 12–19), twice the 99th percentile value was used as the maximum intake rate. Minimum intake values for all food items were zero. The minimum and maximum values are also included in Table I-2.[2]

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[2] The 3MRA methodology for defining minimum and maximum values has been extensively peer reviewed and reviewed by the Science Advisory Board. The defined minimum and maximum values preserve the shape and scale of the distribution. For this reason, these values will typically not match the lower- and upper- most percentiles (e.g., 1st and 99th) presented in the EFH (U.S. EPA, 2011).

Table I-2. Summary of Exposure Parameters Used in Home Gardening Analysis

Parameter	Units	Distribution Type	Mean (or Shape)[a]	Std Dev (or Scale)[a]	Minimum	Maximum	Reference
Body weight (Child 1)	kg	Lognormal	1.55E+01	2.05E+00	4.00E+00	5.00E+01	U.S. EPA (2011); Table 8-3
Body weight (Child 2)	kg	Lognormal	3.07E+01	5.96E+00	6.00E+00	2.00E+02	
Body weight (Child 3)	kg	Lognormal	5.82E+01	1.02E+01	1.30E+01	3.00E+02	
Body weight (Adult)	kg	Lognormal	7.12E+01	1.33E+01	1.50E+01	3.00E+02	
Exposed fruit (Child 1)	g WW kg^{-1} d^{-1}	Gamma	1.43E+00	1.58E+00	0.00E+00	1.60E+01	U.S. EPA (2011); Table 13-58
Exposed fruit (Child 2)	g WW kg^{-1} d^{-1}	Lognormal	2.78E+00	5.12E+00	0.00E+00	3.60E+01	
Exposed fruit (Child 3)	g WW kg^{-1} d^{-1}	Lognormal	1.54E+00	2.44E+00	0.00E+00	1.18E+01	
Exposed fruit (Adult)	g WW kg^{-1} d^{-1}	Lognormal	1.57E+00	2.3E+00	0.00E+00	1.29E+01	
Exposed vegetables (Child 1)	g WW kg^{-1} d^{-1}	Gamma	9.70E-01	2.62E+00	0.00E+00	2.10E+01	U.S. EPA (2011); Table 13-60
Exposed vegetables (Child 2)	g WW kg^{-1} d^{-1}	Lognormal	1.64E+00	3.95E+00	0.00E+00	2.70E+01	
Exposed vegetables (Child 3)	g WW kg^{-1} d^{-1}	Gamma	9.10E-01	1.19E+00	0.00E+00	1.10E+01	
Exposed vegetables (Adult)	g WW kg^{-1} d^{-1}	Weibull	1.57E+00	1.76E+00	0.00E+00	1.03E+01	
Protected fruit (Child 1)	g WW kg^{-1} d^{-1}	Gamma	7.37E-01	1.59E+01	0.00E+00	4.50E+01	U.S. EPA (2011); Table 13-59
Protected fruit (Child 2)	g WW kg^{-1} d^{-1}	Gamma	7.37E-01	8.15E+00	0.00E+00	2.60E+01	
Protected fruit (Child 3)	g WW kg^{-1} d^{-1}	Gamma	7.36E-01	3.56E+00	0.00E+00	3.80E+01	
Protected fruit (Adult)	g WW kg^{-1} d^{-1}	Lognormal	6.63E+00	1.57E+01	0.00E+00	4.73E+01	
Protected vegetables (Child 1)	g WW kg^{-1} d^{-1}	Lognormal	1.88E+00	1.98E+00	0.00E+00	1.60E+01	U.S. EPA (2011); Table 13-61
Protected vegetables (Child 2)	g WW kg^{-1} d^{-1}	Lognormal	1.07E+00	1.04E+00	0.00E+00	8.00E+00	
Protected vegetables (Child 3)	g WW kg^{-1} d^{-1}	Lognormal	7.70E-01	6.90E-01	0.00E+00	6.00E+00	
Protected vegetables (Adult)	g WW kg^{-1} d^{-1}	Lognormal	1.01E+00	1.19E+00	0.00E+00	6.49E+00	
Root vegetables (Child 1)	g WW kg^{-1} d^{-1}	Lognormal	2.31E+00	6.05E+00	0.00E+00	4.10E+01	U.S. EPA (2011); Table 13-62
Root vegetables (Child 2)	g WW kg^{-1} d^{-1}	Weibull	6.80E-01	1.06E+00	0.00E+00	1.50E+01	
Root vegetables (Child 3)	g WW kg^{-1} d^{-1}	Weibull	8.40E-01	9.10E-01	0.00E+00	9.00E+00	
Root vegetables (Adult)	g WW kg^{-1} d^{-1}	Weibull	1.15E+00	1.32E+00	0.00E+00	7.47E+00	
Drinking water (Child 1)	mL kg^{-1} d^{-1}	Weibull	1.15E+00	2.56E+01	2.23E-03	1.86E+02	U.S. EPA (2008); Table 3-19
Drinking water (Child 2)	mL kg^{-1} d^{-1}	Weibull	1.14E+00	1.75E+01	2.23E-03	1.86E+02	
Drinking water (Child 3)	mL kg^{-1} d^{-1}	Weibull	1.08E+00	1.14E+01	2.23E-03	1.86E+02	
Drinking water (Adult)	mL kg^{-1} d^{-1}	Weibull	1.16E+00	1.66E+01	1.00E-02	1.26E+02	U.S. EPA (2011) Table 3-38
Exposure duration (Child)	yr	Weibull	1.32E+00	7.06E+00	1.00E+00	3.80E+01	U.S. EPA (2011); Table 16-109
Exposure duration (Adult)	yr	Weibull	1.34E+00	1.74E+01	1.00E+00	5.00E+01[b]	

a. Shape and scale are presented for Gamma and Weibull distributions.

b. Exposure duration was capped at 50 years so it would never exceed the 70-year lifetime assumption implicit in the averaging time used, given the starting age of 20 years.

I.2.2 Exposure Parameter Distribution Methodology

This section describes how stochastic or distributed input data for each exposure factor were collected and processed. Most exposure factor distributions were developed by analyzing data from the EFH or CSEFH to fit selected parametric distributions (i.e., gamma, lognormal, Weibull). Development steps included preparing data, fitting distributions, assessing fit, and preparing parameters to characterize uncertainty in the distribution inputs.

For many exposure factors, EFH and CSEFH data include sample sizes and estimates of the following parameters for specific receptor types and age groups: mean, standard deviation, standard error, and percentiles corresponding to a subset of the following probabilities: 0.01, 0.02, 0.05, 0.10, 0.15, 0.25, 0.50, 0.75, 0.85, 0.90, 0.95, 0.98, and 0.99. Where available, these percentile data were used as the basis for fitting distributions. Although in no case were all of these percentiles actually provided for a single factor, the EFH typically included seven or more. Therefore, using the percentiles is a fuller use of the available information than simply fitting data based on the method of moments (e.g., selecting models that agree with the data mean and standard deviation). For some factors, sample sizes were too small to justify the use of certain percentiles in the fitting process. Percentiles were used only if at least one data point was in the tail of the distribution. If the exposure factor data repeated a value across several adjacent percentiles, only one value (the most central or closest to the median) was used in most cases (e.g., if both the 98th and 99th percentiles had the same value, only the 98th value was used).

The EFH and CSEFH do not use standardized age cohorts across exposure factors. Different exposure factors have data reported for different age categories. Therefore, to obtain the percentiles for fitting the four standardized age cohorts (i.e., aged 1–5, 6–11, 12–19, and 20–70), each EFH or CSEFH cohort-specific value for a given exposure factor was assigned to one of these four cohorts. When multiple cohorts fit into a single cohort, the percentiles were averaged within each cohort (e.g., data on children aged 1–2 and 3–5 were averaged for Cohort 1 [aged 1–5]). If sample sizes were available, then weighted averages were used, with weights proportional to sample sizes. If sample sizes were not available, then equal weights were assumed (i.e., the percentiles were simply averaged).

Because the EFH and CSEFH data are always positive and almost always skewed to the right (i.e., have a long right tail), three two-parameter probability distributions commonly used to characterize such data (gamma, lognormal, and Weibull) were selected. The data were also fit to a three-parameter distribution (generalized gamma) that unifies the two-parameter distributions and allows for a likelihood ratio test of the fit of the two-parameter distributions. [1] This was a considerable improvement over the common practice of using a lognormal model in which adequate EFH data were available to support maximum likelihood estimation. However, in a few cases (e.g., inhalation rate) the data were not adequate to fit a distribution, and the lognormal distribution was assumed as the default.

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[1] The SFS evaluation ultimately only used the two-parameter distributions because the three-parameter distribution did not significantly improve the goodness of fit over the two-parameter distributions.

I.2.3 Variable Parameters

I.2.3.1 Exposed Fruit Consumption

Table I-3 presents exposed fruit consumption data. Data for consumption of homegrown exposed fruit come from Table 13-58 of the EFH (U.S. EPA, 2011). Data (in g WW kg^{-1} d^{-1}) are presented by child age groups and for adults. For the age group of 1- to 5-year olds, data were only available for those aged 3–5 years (not available for children aged 1–2); therefore, these data were used for the entire 1- to 5-year-old age group.

Table I-3. Exposed Fruit Consumption Data and Distributions

Age Cohort	N	EFH Data (g WW kg^{-1} d^{-1}) Data Mean	P01	P05	P10	P25	P50	P75	P90	P95	P99
1–5	49	2.6			0.373	1	1.82	2.64	5.41	6.07	32.5
6–11	68	2.52		0.171	0.373	0.619	1.11	2.91	6.98	11.7	15.7
12–19	50	1.33		0.123	0.258	0.404	0.609	2.27	3.41	4.78	5.9
Adult	596	1.55	0.042	0.158	0.258	0.449	0.878	1.73	3.41	5	12.9

N = number of samples, P01–P99 = percentiles

I.2.3.2 Protected Fruit Consumption

Data for consumption of homegrown protected fruit come from Table 13-59 of the EFH (U.S. EPA, 2011) and are presented in **Table I-4**. Data (in g WW kg d^{-1}) were presented for the following age cohorts: those aged 12–19, 20–39, 40–69, and all ages combined. No data for adults or children aged 1–5 and 6–11 were available for homegrown protected fruit consumption. However, per capita intake data for protected fruit (including store-bought products) were available from the EFH for those aged 1–2, 3–5, and 6–11. Therefore, data for the general population were used to calculate adjustment factors to develop distributions for the non-adult age groups for consumption of homegrown protected fruit. The population estimated mean and standard deviation for adults aged 20 and older (derived from the weighted average of means and standard deviations of those aged 20–39 and those aged 40–69) were used to represent adults for the analysis.

Table I-4. Protected Fruit Consumption Data and Distributions

Source	Age Cohort	EFH Data (g WW kg^{-1} d^{-1}) Data Mean	P01	P05	P10	P25	P50	P75	P90	P95	P99
EFH (gen)	All ages	1.9					0.38	2.6	5.4	8.1	16.3
EFH (gen)	1–5	5.45					2.7	7.7	14.38	20	32.3
EFH (gen)	6–11	2.7					0.17	3.8	8.1	11.4	19.8
EFH (gen)	12–19	1.8					1.8*	2.6	5.4	8.4	15.4
EFH (gen)	20–69	1.4					0.93*	2.1	4.2	5.8	10.5

HP	1–5										
HP	6–11										
EFH (HP)	12–19	2.960	0.12	0.16	0.283	0.393	1.23	2.84	7.44	11.4	19.1
EFH (HP)	20–69	5.1	0.13	0.3	0.39	0.94	2	6.9	15	19	36.59
EFH (HP)	All ages	5.740	0.15	0.266	0.335	0.933	2.34	7.45	16	19.7	47.3
EFH (HP)	Adult	5.9	0.12	0.265	0.335	1.116	2.42	7.46	16	19.1	47.3

gen = general population data, EFH = U.S. EPA (2011), HP = home-produced data, P05–P95 = percentiles,
* based on mean

The relative standard deviations (RSD) for consumption rates were assumed to be the same for all age groups; the similarity of coefficients of variation (CVs) suggests that this is a reasonable approximation for the general population. To develop consumption of homegrown protected fruit distributions for the child age groups, it was also assumed that the mean intake rates have the same fixed ratio for all the age groups of a given food type. That is, the ratio of the mean amount consumed of homegrown protected fruit divided by the mean amount consumed of protected fruit in the general population is the same for any two age groups. These two assumptions (i.e., constant RSD and constant mean ratio) were used to infer the parameters of the gamma distributions for the home-produced foods from those of the general population. Each age-specific ratio (or adjustment factor) was multiplied by the “all ages” group data (e.g., mean, standard deviation) to estimate each age-specific consumption rate.

I.2.3.3 Exposed Vegetable Consumption

Table I-5 presents exposed vegetable consumption data and distributions. Data for consumption of homegrown exposed vegetables come from Table 13-60 of the EFH (U.S. EPA, 2011). Data (in g WW kg^{-1} d^{-1}) were presented for those aged 1–2, 3–5, 6–11, 12–19, 20–39, and 40–69, as well as for all adults. Weighted averages of percentiles, means, and standard deviations were calculated for the age group of 1- to 5-year-olds (combining groups of children aged 1–2 years and 3–5).

Table I-5. Exposed Vegetable Consumption Data and Distributions

Age Cohort	N	EFH Data (g WW kg^{-1} d^{-1})								
		Data Mean	P05	P10	P25	P50	P75	P90	P95	P99
1–5	105	2.453	0.102	0.37	0.833	1.459	3.226	6.431	8.587	9.3
6–11	134	1.39	0.044	0.094	0.312	0.643	1.6	3.22	5.47	13.3
12–19	143	1.07	0.029	0.142	0.304	0.656	1.46	2.35	3.78	5.67
Adult	1361	1.57	0.089	0.168	0.413	0.889	1.97	3.63	5.45	10.3

N = number of samples, P01–P99 = percentiles

I.2.3.4 Root Vegetable Consumption

Table I-6 presents root vegetable consumption rates and distributions. Homegrown root vegetable consumption data come from Table 13-62 of the EFH (U.S. EPA, 2011). Data (in g WW kg^{-1} d^{-1}) were presented for those aged 1–2, 3–5, 6–11, 12–19, 20–39, and 40–69, and for all adults. Weighted averages of percentiles, means, and standard deviations were calculated for the Cohort 1 age group (combining groups of children aged 1–2 and 3–5).

Table I-6. Root Vegetable Consumption Data and Distributions

Age Cohort	N	EFH Data (g WW kg^{-1} d^{-1})									
		Data Mean	P01	P05	P10	P25	P50	P75	P90	P95	P99
1-5	45	1.886	0.08	0.081	0.167	0.291	0.686	2.653	5.722	7.502	7.50
6-11	67	1.32		0.014	0.036	0.232	0.523	1.63	3.83	5.59	7.47
12-19	76	0.937	0.01	0.008	0.068	0.269	0.565	1.37	2.26	3.32	5.13
Adult	682	1.15		0.036	0.117	0.258	0.674	1.5	2.81	3.64	7.47

N = number of samples, P01–P99 = percentiles

I.2.3.5 Protected Vegetable Consumption

Homegrown protected vegetable consumption data come from Table 13-61 of the EFH (U.S. EPA, 2011) and are presented in **Table I-7** below. Data (in g WW kg^{-1} d^{-1}) were presented for those aged 1–2, 3–5, 6–11, 12–19, 20–39, and 40–69 years, as well as for adults. Weighted averages of percentiles, means, and standard deviations were calculated for Cohort 1 (children aged 1–5), combining groups of children aged 1–2 and 3–5.

Table I-7. Protected Vegetable Consumption Data and Distributions

Age Cohort	N	EFH Data (g WW kg^{-1} d^{-1})									
		Data Mean	P01	P05	P10	P25	P50	P75	P90	P95	P99
1–-5	53	1.76	0.27	0.265	0.408	0.829	1.397	2.066	3.053	6.812	6.94
6–11	63	1.1	0.19	0.208	0.318	0.387	0.791	1.31	2.14	3.12	5.4
12–19	51	0.776	0.06	0.161	0.239	0.354	0.583	0.824	1.85	2.2	2.69
Adults	602	1.01	0.103	0.153	0.192	0.336	0.642	1.21	2.32	3.05	6.49

N = number of samples, P01–P99 = percentiles

I.2.3.6 Body Weight

Table I-8 presents body weight data and distributions. Body weight data come from Table 8-3 of the CSEFH (U.S.EPA, 2008) and Table 8-3 of the EFH (U.S. EPA, 2011). Data (in kg) were presented by age and gender. Weighted averages of percentiles and means were calculated for those aged 1–5, 6–11, 12–19, and adult age groups; male and female data were weighted and combined for each age group. These percentile data were used as the basis for fitting distributions.

Table I-8. Body Weight Data and Distributions

Age Cohort	N	Body Weight Data (kg)									
		Data Mean	P05	P10	P15	P25	P50	P75	P85	P90	P95
1–5	4,638	15.6	11.7	12.4	12.8	13.6	15.1	17.0	18.2	19.2	20.9
6–11	3,593	31.8	19.7	21.3	22.3	24.4	29.3	36.8	42.1	45.6	52.5
12–19	10,148	63.9	40.8	44.3	47.2	51.4	60.6	72.5	81.5	88.1	98.0
20+	14,698	81.63	53.6	57.7	61.1	67.0	79.0	92.9	102.0	108.5	118.8

I.2.3.7 Drinking Water

Table I-9 presents drinking water data and distributions. Drinking water data come from Table 3-19 of the CSEFH (U.S.EPA, 2008) and Table 3-38 of the EFH (U.S. EPA, 2011). Data (in mL kg^{-1} d^{-1}) were presented by age and gender. Weighted averages of percentiles, and means were calculated for those aged 1–5, 6–11, 12–19, and adult age groups; male and female data were weighted and combined for each age group. These percentile data were used as the basis for fitting distributions.

Table I-9. Drinking Water and Distributions

Age Cohort	N	Drinking Water Data (mL kg^{-1} d^{-1})								
		Data Mean	P10	P15	P25	P50	P75	P90	P95	P99
1–5	5462	24.8	3.3	8.3	19.5	34	34	50.6	66.1	103
6–11	1410	17	2	6	13	23	23	35	47	78
12–19	4143	11	1	4	8	15	15	25	34	58
20+	7616	16	2	-	6	12	22	34	42	64

I.2.3.8 Exposure Duration

Table I-10 presents exposure duration data and distributions. Exposure duration was assumed to be equivalent to the average residence time for each receptor. Exposure durations for adult and child residents were determined using data on residential occupancy from the EFH, Table 16-109 (U.S. EPA, 2011). The data represent the total time a person is expected to live at a single location, based on age. The table presented male and female data combined. Adult residents aged 21–90 were pooled. Children aged 3 were used to represent those aged 1–5.

Table I-10. Exposure Duration Data and Distributions

EFH Data		Distributions		
Age Cohort	Data Mean (yr)	Distribution	Pop-Estd Shape (yr)	Pop-Estd Scale (yr)
Child (1–5)	6.5	Weibull	1.32	7.059
Adult resident	16.0	Weibull	1.34	17.38

Pop-Estd = population-estimated

I.2.4 Fixed Parameters

Certain exposure factors were fixed based on central tendency values from the best available source (usually EFH recommendations), either because no variability was expected or because the available data were not adequate to generate distributions. Fixed (constant) exposure factors are shown in **Table I-11** along with the selected value and data source.

Table I-11. Summary of Human Exposure Factor Data Used in Modeling: Constants

Description	Average	Units	Source
Averaging time for carcinogens	7.00E+01	yr	U.S. EPA (1989)
Exposure frequency	3.50E+02	d y^{-1}	U.S. EPA (1991)
Fraction food preparation loss			
Exposed fruit	2.10E-01	Fraction	U.S. EPA (2011); Table 13-69
Exposed vegetables	1.61E-01	Fraction	U.S. EPA (2011); Table 13-69
Protected fruit	2.90E-01	Fraction	U.S. EPA (2011); Table 13-69
Protected vegetables	1.30E-01	Fraction	U.S. EPA (2011); Table 13-69
Root vegetables	5.30E-02	Fraction	U.S. EPA (2011); Table 13-69
Ingestion rate: soil			
Children aged 1–5, 6-11, and 12-19	1.00E+02	mg d^{-1}	U.S. EPA (2011); Table 5-1
Adult	5.00E+01	mg d^{-1}	U.S. EPA (2011); Table 5-1

- When evaluating carcinogens, total dose was averaged over the lifetime of the individual, assumed to be 70 years.
- Exposure frequency was set to 350 days per year in accordance with EPA policy, assuming that residents take an average of 2 weeks' vacation time away from their homes each year.
- Mean soil ingestion rates were cited as 100 mg d^{-1} for children and 50 mg d^{-1} for adults (U.S. EPA, 2011, Table 5-1). The EFH did not recommend any percentile data. The soil ingestion rates were not varied for the probabilistic analysis.

Exposure Parameters Used for General Population

Consumption rate data for the general population were obtained directly from the EFH based on per capita intake rates. Data for most parameters included 50th and 90th percentiles. However, for exposed fruit (adults and 11–19 year olds) and protected vegetables (all cohorts), mean data were used in the absence of 50th percentile data. Data for children aged 1–5 reflect a weighted average for consumption rates reported for children aged 1–2 and 3–5. Data for adults reflect a weighted average for consumption rates reported for adults aged 20–39 and 40–69. **Table I-12** summarizes the parameters that were used in the analysis of the general population. The fraction contaminated was assumed to be 0.5.

Table I-12. Summary of Produce Consumption Rates (CR) for General Population

Parameters	General Population Estimates grams (WW) kg^{-1} body weight day^{-1} 50th Percentile	General Population Estimates grams (WW) kg^{-1} body weight day^{-1} 90th Percentile	Source
Exposed Fruit			
Children aged 1–5	3.9	21.24	U.S. EPA (2011); Table 9-18
Children aged 6–11	2.2 [a]	6.3	U.S. EPA (2011); Table 9-18
Children aged 12–19	0.87 [a]	2.9	U.S. EPA (2011); Table 9-18
Adults	0.646 [a]	2.12	U.S. EPA (2011); Table 9-18
Exposed Vegetables			
Children aged 1–5	0.638	4.96	U.S. EPA (2011); Table 9-20
Children aged 6–11	0.6	3.4	U.S. EPA (2011); Table 9-20
Children aged 12–19	0.53	2.5	U.S. EPA (2011); Table 9-20
Adults	0.906	3.26	U.S. EPA (2011); Table 9-20
Protected Fruit			
Children aged 1–5	2.7	14.38	U.S. EPA (2011); Table 9-19
Children aged 6–11	0.17	8.1	U.S. EPA (2011); Table 9-19
Children aged 12–19	1.8 [a]	5.4	U.S. EPA (2011); Table 9-19
Adults	0.926 [a]	4.18	U.S. EPA (2011); Table 9-19
Protected Vegetables			
Children aged 1–5	1.26 [a]	3.86	U.S. EPA (2011); Table 9-21
Children aged 6–11	0.78 [a]	2.6	U.S. EPA (2011); Table 9-21
Children aged 12–19	0.46 [a]	1.5	U.S. EPA (2011); Table 9-21
Adults	0.548 [a]	1.7	U.S. EPA (2011); Table 9-21
Root Vegetables			
Children aged 1–5	1.44	6.02	U.S. EPA (2011); Table 9-22
Children aged 6–11	1.0	4.2	U.S. EPA (2011); Table 9-22
Children aged 12–19	0.82	3.0	U.S. EPA (2011); Table 9-22
Adults	0.7	2.58	U.S. EPA (2011); Table 9-22

[a] Based on mean values.

I.3 References

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Appendix J

EPACMTP Groundwater Modeling

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Appendix J:
EPACMTP Groundwater Modeling

National-scale probabilistic groundwater modeling was performed for arsenic using EPACMTP (U.S. EPA, 2003a,b,c; 1997). The EPACMTP model addresses chemical reactions by adsorption and decay processes. For the simulation of metals, EPACMTP utilizes nonlinear sorption isotherms which generally have a linear range at lower leachate concentrations and behave nonlinearly at higher leachate concentrations. The use of nonlinear metal sorption isotherms enables EPACMTP to model nonlinear behavior in the unsaturated zone module for a wide array of subsurface conditions. In the case of arsenic, the model supports species-specific modeling of either arsenic III or V with arsenic III being the more mobile of the two species. In this analysis, arsenic was modeled as arsenic III supporting the development of conservative SFS Screening Levels for the groundwater pathway.

Nonlinear Sorption Isotherms

A nonlinear sorption isotherm is an expression of the equilibrium relationship between the sorbed concentration of a metal (or other constituent) and the aqueous concentration for a representative set of subsurface system conditions. Nonlinear sorption isotherms are important when modeling metals because metal sorption coefficients (Kds), which influence metal fate and transport, are significantly affected by metal concentration in the aqueous phase. In general, metal mobility tends to be higher (and thus, Kds lower) as leachate concentrations increase. Therefore, as leachate concentrations decrease during unsaturated zone (soil) transport, metal mobility also tends to decrease (and Kds tend to increase).

The leachate fluxes and annual average leachate infiltration rates estimated by the home garden source model were used as input to EPACMTP, to estimate arsenic concentrations at the receptor well. For both the child and adult receptors, the model generated distributions of maximum time-average concentrations. These concentrations were calculated using receptor-specific exposure durations and EPACMTP estimated peak well concentrations. The averaging period for each iteration in the simulation was centered on the peak well concentration and spanned the exposure duration for the receptor of interest (i.e., child or adult).

Under the SFS home garden scenario, the well was assumed to be placed 1 meter from the edge of the garden in the centerline of the plume. The depth of the well was varied uniformly throughout the aquifer thickness or throughout the upper 10 m of the aquifer thickness, whichever was less. That is, the well depth was never allowed to exceed 10 m below the water table. This limitation for well depth has been used in previous analyses primarily for two reasons: (1) to be consistent with a residential well scenario (these wells are generally shallow because of the higher cost of drilling a deeper well) and (2) to produce a conservative estimate of exposure (because the infiltration rate is generally lower than the groundwater seepage velocity, groundwater plumes tend to be relatively shallow).

The distributions of receptor drinking water concentrations were developed concurrently with the soil pathway modeling using an initial soil concentration of 1 ppm for arsenic under the “unitized” approach. As described in **Chapter 5**, the “unitized” approach scales the 90th percentile unitized hazard quotient (HQ) to estimate a protective SFS-specific concentration based on EPA’s risk management criteria (e.g., HQ of 1). These SFS-specific concentrations represent conservative estimates of the constituent concentration in SFS which, if the SFS were used as a component of manufactured soil in a home garden, would be protective of human health and the environment. To ensure the appropriateness of applying the unitized approach to the groundwater pathway, it was necessary to demonstrate that arsenic would behave linearly in the subsurface under anticipated environmental conditions and at concentration levels found in SFS. **Section J.1** describes the analysis that was performed to make the determination that the linear approach was valid and defensible.

The remainder of this appendix discusses model inputs and outputs. **Section J.2** discusses and presents the EPACMTP input parameters used in the national-scale assessment. **Section J.3** discusses key outputs including predicted arrival times for peak receptor well concentrations.

J.1 Linear Behavior

EPACMTP simulates the migration of constituents from the source model through the unsaturated and saturated zones to receptor drinking water wells. In the unsaturated zone, EPACMTP simulates the effects of both linear and nonlinear sorption reactions. For metal constituents such as arsenic with nonlinear sorption isotherms, the unsaturated zone module simulates partitioning by using concentration-dependent partitioning coefficients. These coefficients generally have a linear range at lower leachate concentrations and behave nonlinearly at higher leachate concentrations, with Kd generally decreasing with increasing leachate concentration. The saturated zone module uses a linearized isotherm, based upon the maximum constituent concentration at the water table. The linear assumption applied in the saturated zone reflects dilution of the leachate in the ambient groundwater (as the leachate enters the saturated zone) to a range in which constituent isotherms generally are linear. In order to apply the “unitized” approach to develop SFS-specific Screening Levels, each modeling component along the exposure pathway, including the unsaturated zone, must be linear.

To ensure that a linear partitioning assumption is valid in the unsaturated zone, consideration was given to the following. The assumption of linearity from emplacement to exposure is dependent on the selection of Kd values from the linear range of the isotherms in the unsaturated zone. Therefore, it was necessary to review the leachate concentrations generated by the garden source model to ensure that the arsenic concentrations leaching from the garden would not exceed the upper bound of an isotherm’s linear range. In addition, it was necessary to ensure that the predicted leachate concentrations associated with the estimated SFS-specific Screening Levels would also fall within the linear range of the arsenic isotherms. For this reason, the below analysis used the 95th percentile leachate concentrations derived from both SPLP and ASTM testing methods as applied to pure SFS samples.

Analysis Overview

The analysis consisted of the following steps:

1. Establish a statistically representative leachate concentration from the population of SFS leachate data compiled by EPA
2. Visually investigate the tabulated isotherms for arsenic III to identify if linear regions exist and
3. Visually compare a conservatively representative leachate value for arsenic III to the linear ranges of the isotherms to see if the assumption of linearity will hold for leachate

values expected in the garden. When the representative leachate values are well within the linear range, the defensible use of a unitized, scaling approach can be established.

4. Lastly, establish a bounding SFS concentration to benchmark that the calculated SFS-specific Screening Levels will be associated with leachate concentrations that fall within the linear range.

Establish a statistically representative leachate concentration

The USDA collected samples of SFS from U.S. foundries and conducted leaching analyses of the materials in their raw form. **Table J-1** presents 95th percentile leachate concentrations reflecting the USDA's leaching analyses, the synthetic precipitation leaching procedure (SPLP), and the American Society for Testing and Materials (ASTM) International method D 3897. The higher of the two values (i.e., ASTM value of 0.018 mg L^{-1}) was used in the linearity analysis for comparison to MINTEQA2-derived sorption isotherms.

Table J-1. USDA SPLP and ASTM Results for Arsenic

	SFS 95th %ile	
Metal	**SPLP (mg L^{-1})**	**ASTM (mg L^{-1})**
Arsenic	0.017	0.018

Reference: Chapter 4, Table 4-2

Given that the home garden scenario assumes that SFS will be mixed in a 50:50 ratio with native soils, the USDA 95th percentile leachate values serve as a conservatively high estimate for the maximum likely SFS leachate concentration to be observed under the scenario.

Visual Inspection of Isotherms for Linearity

A visualization tool developed with the MATLAB (MathWorks, 2013) scientific programming platform was used to plot individual MINTEQA2-derived tabulated sorption isotherms of arsenic III. **Figure J-1** presents plots of two isotherms for a unique set of subsurface conditions. The x- and y-axes represent aqueous dissolved concentration (mg L^{-1}) and Kd (L kg^{-1}) on a base 10 logarithmic scale. The two isotherms are plotted in the main figure window, one for the unsaturated (in blue) and one for the saturated (in red) regions of the subsurface. The plotted curves representing the isotherms correspond to the same set of specific subsurface conditions as specified by the selections shown on the left side of the figure:

- Groundwater compositional type (carbonate or non-carbonate)
- Dissolved concentration of representative anthropogenic (leachate) organic acids (LOM)
- pH of the receiving domain
- Concentration of adsorbents – ferric oxide [goethite] (FeOX) and particulate natural organic matter (NOM).

Both curves display the same characteristic behavior: Kd is constant (i.e., linear) for aqueous concentrations less than or equal to 0.3 mg L^{-1}, above which Kd behaves nonlinearly, decreasing with increasing aqueous concentration.

Figure J-1. Visualization of 2 (unsaturated and saturated) arsenic III nonlinear sorption isotherms generated by MINTEQA2 in non-carbonate groundwater compositional environment.

Visual Inspection and Comparison to SFS Concentrations

Figure J-2 is a duplicate of Figure J-1 with the addition of a vertical line representing the 95th percentile ASTM leachate concentration of 0.018 mg L^{-1} for arsenic in SFS. This value is over an order of magnitude less than the concentration at which the Kd begins to be dependent on the dissolved concentration. If this behavior is consistent for all isotherms, then a linear sorption assumption is reasonable.

Figure J-2. Visualization of 2 (unsaturated and saturated) Arsenic III nonlinear sorption isotherms in non-carbonate groundwater compositional environment with SFS 95th percentile ASTM leachate concentration of 0.018 mg L^{-1} superimposed.

Figure J-3 and **Figure J-4** show all nonlinear sorption isotherms for arsenic III and the 95th percentile ASTM leachate concentration of 0.018 mg L^{-1} for arsenic in SFS for carbonate (karst) and non-carbonate aquifer environments. In all cases, the benchmark leachate concentration is comfortably less than the upper bound on the linear range of Kd.

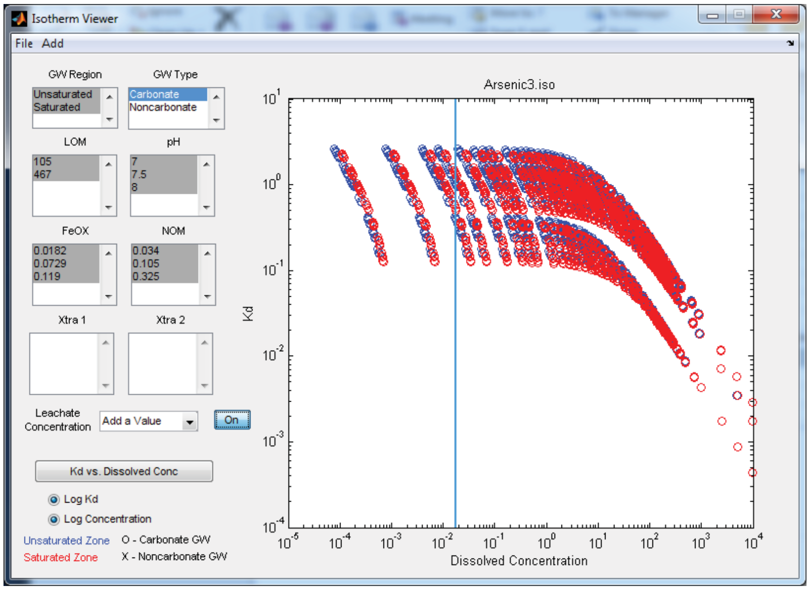

Figure J-3. Visualization of all (unsaturated and saturated conditions) Arsenic III nonlinear sorption isotherms in carbonate groundwater compositional environment with SFS 95th percentile ASTM leachate concentration of 0.018 mg L^{-1} superimposed.

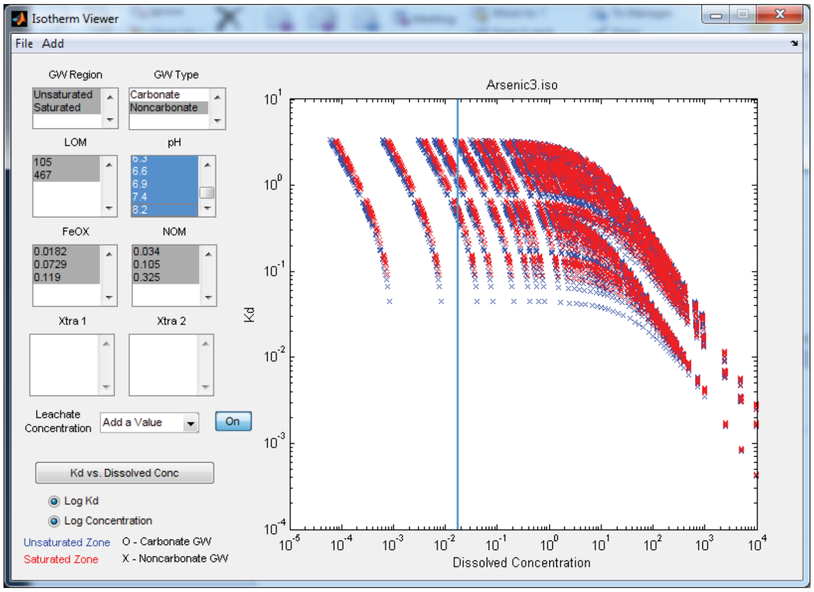

Figure J-4.Visualization of all (unsaturated and saturated conditions) Arsenic III nonlinear sorption isotherms in non-carbonate groundwater compositional environment with SFS 95th percentile ASTM leachate concentration of 0.018 mg L^{-1} superimposed.

Screening Level Leachate within Linear Range

To determine whether the resulting SFS Screening Levels would result in predicted leachate concentrations that are within the linear range, a two-step approach was implemented: Under the first step, a point of reference of 150 ppm was calculated and used for comparison to the arsenic groundwater SFS screening level as an approximate breaking point indicator for linear/non-linear behavior. This breaking point was estimated based on an initial unitized source model runs where the 90th percentile maximum arsenic leachate concentration was identified to be 0.004 mg L^{-1}. Based on the above demonstration of linearity, the linearity/non-linearity leachate concentration of 0.3 mg L^{-1} was used to back-calculate to a corresponding SFS concentration of 150 ppm (corresponding to a manufactured soil concentration of 75 ppm).[14] Under the second step, the manufactured soil concentration corresponding to the final 90th percentile groundwater SFS Screening Level was used as input to the source model. The resulting leachate distribution was reviewed and the 90th percentile maximum leachate concentration was found to be 0.03 mg L^{-1} which is well below the established leachate concentration of 0.3 mg L^{-1} discussed above.

Results from the analysis described above demonstrated that there is high confidence that a linear assumption for the groundwater pathway modeling of arsenic III is reasonable and defensible for calculating SFS Screening Levels. As shown above, the SFS 95th percentile leachate concentration is, in all cases, comfortably within the linear range of all isotherms for arsenic III.

J.2 Model Input Parameters

Attachment J-A identifies the key EPACMTP input parameters, values and distributions used in evaluating the groundwater pathway.

Table J-A-1 presents all the EPACMTP input parameters organized by the primary components of the groundwater modeling scenario:

- **Aquifer** (or saturated zone) parameters
- **Chemical** parameters associated with the leachate
- **Exposure** parameters associated with the receptor well
- **Vadose** (or unsaturated zone) parameters
- **Waste Management Unit** (garden) parameters.

For each input parameter, **Table J-A-1** provides the EPACMTP variable ID, the parameter description, units, distribution type, default values (if applicable), and data sources (where appropriate).

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[14] It should be noted that this concentration was estimated only as a point of reference for this analysis and should not be viewed as a definitive cut point between linear and non-linear behavior for any arsenic leaching scenario. Rather, it is an approximation based on MINTEQ modeling that captures key controlling factors within the subsurface environment. Therefore, the purpose of showing the approximate cut point should be recognized, and the value should be used with caution.

Table J-A-2 presents detailed information on the four empirical correlated groundwater pathway parameters: GRADNT, XKX, ZB, and DSOIL. Each record in the table represents a correlated sampling of each of the four parameters from a single site. Table J-A-2 represents data collected from 400 hazardous waste sites in the United States (Newell et al., 1990) grouped into 12 subsurface environments, identified by Aquifer Code. The value "-999" denotes that site data were unavailable. This value (-999) triggers the model to estimate the value using methods described in Section 5.5 of the *EPACMTP Technical Background Document* (U.S. EPA, 2003a). Details regarding data origins, organization, and use in EPACMTP are provided in Section 5.3.4 of U.S. EPA (2003b).

Table J-A-3 presents parameters represented by empirical distributions. For each parameter, a set of paired values consisting of a parameter value and the associated cumulative distribution function percentile (e.g., 0.25 denotes the 25th percentile) are shown. The derivation of each distribution is discussed in detail in various sections of U.S. EPA (2003b).

The percolation of water through garden (infiltration) and soils surrounding the garden (recharge) was estimated using modeling results from the Hydrogeologic Evaluation of Landfill Performance (HELP) model (Schroeder et al., 1994). As shown in **Table J-A-4**, the rates are correlated with the cover soil and the climate center nearest the garden. The compilation and creation of these data are described in detail in Appendix A of U.S. EPA (2003b).

J.3 Model Outputs

This section discusses key outputs including arrival time predictions used to support the development of separate SFS screening levels for the soil/produce and groundwater pathways.

The EPACMTP model outputs peak and average receptor well concentrations and the estimated year when these concentrations are predicted to occur. The reported year is measured from the time of initial contaminant release, and corresponds roughly to the middle of the averaging period. To determine if surface and groundwater pathway exposures would occur during the same or overlapping timeframes, the EPACMTP outputs were examined to characterize arrival times. The timeframe estimates for arrival of plume at the receptor well are presented in **Table J-1**. These estimates represent the year (after the SFS manufactured soil is placed in the home garden) when the contaminant plume front would arrive at a well [Beginning] and the year when the contaminant plume would pass the well [End]. Arrival of peak concentration would only occur somewhere within this timeframe. The estimates shown are based on EPACMTP outputs from the unsaturated zone transport simulation, including the first arrival time of leachate at the water table and cessation time of leachate arrival at the water table. Retardation effects in the aquifer due to sorption of arsenic onto soils were also accounted for in these estimates using the following equation:

$$t_{Aq} = x \frac{\phi R}{v_x}$$

where

t_{Aq} = estimated travel time in the aquifer [yr]

x = distance from source to well in X direction (along ground water flow direction [m]
ϕ = porosity of aquifer [-]
R = Retardation factor in aquifer [-]
v_x = average groundwater velocity in X direction [m/yr]

Table J-2. EPACMTP Arrival Times of Arsenic Plume at the Receptor Well

Percentile	Arrival Time Zone (year)	
	Beginning	End
90 %	29	200
80 %	61	200
70 %	100	202
60 %	150	220
50 %	201	272
40 %	203	345
30 %	207	457
20 %	229	663
10 %	398	1112

The travel time in the aquifer was added to the water table information from the unsaturated zone to estimate the windows shown in Table J-2. Based on these data, the front edge of 90% of the simulated plumes would arrive at the receptor well no sooner than 29 years after placed in the garden. Based on the end time of 200 years, the peak or maximum average concentration would not occur until well beyond the initial introduction into the well. The maximum exposures via the soil/produce pathway will occur during the first few years immediately following the application of the manufactured soil. Given the predicted lag time between the surface and groundwater pathway exposures, it is very unlikely that these exposures would occur within the same timeframe. As a result, separate SFS Screening Levels were developed for the soil/produce and the groundwater pathways.

J.4 References

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Attachment J-A
EPACMTP Input Parameters

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Table J-A-1. Groundwater Model Variables

Variable ID	Parameter	Units	Type	Value	Comments	Reference
Aquifer						
DIAM	Avg. particle diameter	cm	Empirical	Randomly selected from national distribution		See attached Table J-A-3
POR	Effective porosity	$cm^3\ cm^{-3}$	Derived	Calculated by EPACMTP	Derived from particle diameter	U.S. EPA, 2003b
BULKD	Bulk density (dry)	$g\ cm^{-3}$	Derived	Calculated by EPACMTP	Derived from porosity	U.S. EPA, 2003b
ZB	Aquifer thickness	m	Empirical correlated	Aquifer-specific	Randomly selected based on aquifer code (IGWR) using EPACMTP	See attached Table J-A-2
XKX	Longitudinal hydraulic conductivity (K)	$m\ yr^{-1}$	Empirical correlated	Aquifer-specific	Randomly selected based on aquifer code (IGWR) using EPACMTP	See attached Table J-A-2
ANIST	Anisotropy ratio	Unitless	Constant	1	No anisotropy	U.S. EPA, 2003b
GRADNT	Hydraulic gradient	$m\ m^{-1}$	Empirical correlated	Aquifer-specific	Randomly selected based on aquifer code (IGWR) using EPACMTP	See attached Table J-A-2
VXCS	Groundwater seepage velocity	$m\ yr^{-1}$	Derived	Calculated by EPACMTP	Derived from conductivity and gradient	U.S. EPA, 2003b
RETARD	Retardation factor	Unitless	Derived	Calculated by EPACMTP	Derived from bulk density, Kd, and porosity	U.S. EPA, 2003b
AL	Longitudinal dispersivity (aquifer)	m	Empirical	Calculated from a national distribution and radial distance to well (RADIS)		See attached Table J-A-3
AT	Transverse dispersivity	m	Derived	Calculated by EPACMTP	Derived from longitudinal dispersivity	U.S. EPA, 2003b
AV	Vertical dispersivity	m	Derived	Calculated by EPACMTP	Derived from longitudinal dispersivity	U.S. EPA, 2003b
TEMP	Groundwater temperature	Degrees C	Constant	Site-specific	Groundwater temperature map; average for 20- mi radius around location	Todd, 1980
PH	Groundwater pH	std. Units	Empirical	Randomly selected from national distribution	Assumed equal to average soil pH for 20-mi radius around location	See attached Table J-A-3
FOC	Fraction organic carbon (FOC)	$g\ g^{-1}$	N/A		Not used in this analysis	
RADIS	Radial distance to well (R)	m	N/A		Not used in this analysis	
ANGLE	Angle of well off plume centerline	Degrees	N/A		Not used in this analysis	
XWELL	Longitudinal distance to well	m	Constant	1m down-gradient of SFS home garden		SFS Home Garden Scenario
YWELL	Distance from plume centerline to well	m	Constant	0 - Well is in middle of plume		SFS Home Garden Scenario
ZWELL	Depth of well below water table	Fraction	Uniform	Randomly selected from uniform distr. within ZB or within uppers of 3, 5 & 10 m of aquifer if ZB> 3, 5 or 10 m, respectively	Uniform Distribution Parameters: Min = 0.0 m; Max = minimum(3.0m,5.0m,10.0m, ZB)	U.S. EPA, 2003b

Table J-A-1. Groundwater Model Variables (continued)

Variable ID	Parameter	Units	Type	Value	Comments	Reference
Chemical						
SFCOF	Freundlich isotherm coefficient (Kd) in aquifer	$cm^3\ g^{-1}$	Derived or empirical isotherm	Chemical-specific	Function of KOC and POM for organics; chosen from tabulated empirical nonlinear isotherms generated by MINTEQ2A for metals	U.S. EPA, 2003b
SFEXP	Freundlich isotherm exponent in aquifer	Unitless	Constant	1	Not used in this analysis	
CSLAM	Chemical decay rate in aquifer	$1\ yr^{-1}$	Derived	Calculated by EPACMTP	Not used in this analysis	U.S. EPA, 2003b
BIOS	Biodegradation rate in aquifer	$1\ yr^{-1}$	Constant	0	Not used in this analysis	
RTEMP	Reference temperature	Degrees C	Constant	Chemical-specific	Not used in this analysis	
KOC	Organic carbon distribution coefficient (KOC)	$mL\ g^{-1}$	Constant	Chemical-specific	Not used in this analysis	
DSTAR	Molecular diffusion coefficient	$m^2\ yr^{-1}$	Constant	Chemical-specific		U.S. EPA, 1987, U.S.EPA, 2001
CZERO	Leachate concentration	$mg\ L^{-1}$	Empirical	Randomly selected from national distribution	3MRA LAU Module Output	
CWASTE	Leachable concentration in waste	$mg\ kg^{-1}$	Empirical	Randomly selected from national distribution	Not used in this analysis	
UFCOF	Freundlich isotherm coefficient (Kd) in vadose zone	$cm^3\ g^{-1}$	Derived or empirical isotherm	Chemical-specific	Not used in this analysis	U.S. EPA, 2003b
UFEXP	Freundlich isotherm exponent vadose zone	Unitless	Constant	1	Not used in this analysis	
Exposure						
CARC	Groundwater averaging time	yr	Empirical	Distributions of Child and Adult exposure times	Model Output used to represent long-term concentration for chronic exposure	
Metal						
METALID	Metal ID	Unitless	Constant	Constant for Arsenic III	SFS Home Garden scenario assumption	U.S. EPA, 2003b
USPH	Unsaturated Zone pH	std. Units	Empirical	Randomly selected from national distribution	Randomly selected value from national distribution from STORET (U.S. EPA, 1996)	See attached Table J-A-3
FEOX	Iron Hydroxide Content	wt. %Fe	Empirical	Randomly selected from uniform distribution	Randomly selected value from uniform distribution: Min=0.0126;Max=1.115 (Loux et al., 1990)	U.S. EPA, 2003b
USLOM	Leachate Organic Matter	$mg\ L^{-1}$	Empirical	Randomly selected from uniform distribution	Randomly selected value from uniform distribution: Min=0.001173;Max=0.00878	U.S. EPA, 2003b
USNOM	Percent Organic Matter	wt. %OM	Empirical	Randomly selected from Johnson SB distribution	Randomly selected value from Johnson SB distribution: Min=0.0;Max=8.51 (Carsel et al., 1988)	U.S. EPA, 2003b

Table J-A-1. Groundwater Model Variables (continued)

Variable ID	Parameter	Units	Type	Value	Comments	Reference
ASNOM	Fraction Organic Content	g g^{-1}	Empirical	Randomly selected from Johnson SB distribution	Randomly selected value from Johnson SB distribution: Min=0.0;Max=11.0 (Carsel and Parrish, 1988)	U.S. EPA, 2003b
IGWT	Groundwater Type	Unitless	Empirical correlated	Calculated by EPACMTP	IGWT=1 if IGWR=1-11; IGWT=2 if IGWR=12	U.S. EPA, 2003b
Vadose						
SATK	Saturated hydraulic conductivity	cm hr^{-1}	Empirical	Site-specific	Correlated to site location	U.S. EPA, 2003b
ALPHA	Moisture retention parameter (a)	1 cm^{-1}	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
BETA	Moisture retention parameter (b)	Unitless	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
WCR	Residual water content	L L^{-1}	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
WCS	Saturated water content	L L^{-1}	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
DSOIL	Unsaturated zone thickness	m	Empirical correlated	Aquifer-specific	Randomly selected based on aquifer code (IGWR)	See attached Table J-A-2
DISPR	Longitudinal dispersivity (vadose)	m	Derived	Calculated by EPACMTP	Derived from unsaturated zone thickness	U.S. EPA, 2003b
POM	Percent organic matter (POM)	%	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
RHOB	Soil bulk density	G cm^{-3}	Empirical	Site-specific	Randomly selected value from distribution for a given soil texture (ISTYPE1)	U.S. EPA, 2003b
WMU						
AREA	WMU area	m^2	Empirical	Constant 0.1 acre	SFS Home Garden scenario assumption	
XY	WMU length	m	Derived	Calculated by EPACMTP	Square root of 0.1 acre area	U.S. EPA, 2003b
YD	WMU width	m	Derived	Calculated by EPACMTP	Square root of 0.1 acre area	U.S. EPA, 2003b
RECH	Recharge rate	m yr^{-1}	Derived	Site-specific	Selected based on climate center (ICLR) and regional soil type (ISTYPE1) for all scenarios	See attached Table J-A-4
SINFIL	Infiltration rate	m yr^{-1}	Derived	Site-specific	3MRA LAU Module Output	
TSOURCE	Duration of leaching period	yr	Derived	Calculated by EPACMTP	Calculated based on transient mass fluxes read by EPACMTP	U.S. EPA, 2003b
DEPTH	WMU depth	m	N/A	TBD	TBD	
FRAC	Fractional volume of waste in garden	unitless	Constant or Uniform	1.0 or Randomly selected from uniform distribution	Uniform Distribution Parameters: Min = 0.036; Max = 1.0;	U.S. EPA, 2003b

Table J-A-1. Groundwater Model Variables (continued)

Variable ID	Parameter	Units	Type	Value	Comments	Reference
DBGS	Depth of WMU base below ground surface	m	Constant	0	Source is on or very near ground surface	
ISTYPE1	Regional soil type	Integer	Empirical	Site-specific		
IGWR	Hydrogeologic Environment	Integer	Empirical	Site-specific	Derived using state-specific aquifer maps	See attached Table J-A-2; U.S. EPA, 1990
ICLR	Climate Center	Integer	Empirical	Site-specific	Nearest appropriate climate center to WMU locations	U.S. EPA, 2003b, See attached Table J-A-4

Table J-A-2. Correlated Empirical Distributions

Hydrogeologic Environment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
1	-999	25.	-	0.016	
1	3.15	16.8	152	-	
1	-999	15.2	15.2	-	
1	-999	610	-999	0.000	
1	-999	5.79	9.14	0.05	
1	946	4.57	-999	0.014	
1	1580	3.05	-999	0.014	
1	63.1	4.88	12.2	0.07	
1	3470	6.1	152	0.03	
1	28.4	2.04	9.14	0.01	
1	126	6.1	7.32	0.03	
1	15.8	3.81	32.9	0.09	
1	315	21.3	3.05	-	
1	-999	6.1	6.1	0.000007	
1	11000	3.05	18.3	0.02	
1	94.6	1.83	4.27	0.04	
1	-999	1.22	9.14	0.01	
1	7570	1.52	3.05	0.000007	
1	6.31	0.914	6.1	0.038	
1	6.31	1.83	7.62	0.1	
1	31.5	6.1	-999	0.06	
1	31.5	0.305	6.1	0.005	
1	-999	9.14	152	0.008	
1	-8.52129	2.8144	3.76962	-3.97399	Mean
1	6.82319	1.0747	1.80348	-0.39418	Covariance
1	1.07478	0.800	0.55257	0.436	Covariance
1	1.80348	0.5525	1.1956	0.17788	Covariance
1	-0.39418	0.436	0.17788	0.81424	Covariance
1	3.15	0.305	3.05	0.000007	Minimum
1	11000	610	152	0.81424	Maximum
2	63.1	6.1	22.9	0.08	
2	28.4	6.1	79.3	-	
2	1890	76.5	-999	0.008	
2	5990	30.5	183	0.001	
2	315	65.5	45.7	0.005	
2	31.5	15.2	21.3	0.1	
2	1580	174	30.5	-	
2	315	5.97	3.6	-	
2	22.1	12.2	10.7	0.028	
2	284	16.8	3.05	0.003	
2	9.46	6.1	152	0.031	
2	221	9.14	-999	0.008	
2	3.15	3.96	4.57	0.01	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	**Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)**	**Unsaturated Zone Thickness [DSOIL] (m)**	**Aquifer Thickness [ZB] (m)**	**Hydraulic Gradient [GRADNT] (m/m)**	**Notes**
2	3.1	4.57	91.4	0.001	
2	2210	15.2	30.5	0.033	
2	11000	18.3	91.4	-	
2	126	13.4	7.62	0.004	
2	1330	6.1	21.3	0.005	
2	31500	1.83	3.05	-	
2	-999	4.27	89	-	
2	1890	53.6	6.1	0.043	
2	9780	18.3	30.5	0.012	
2	6.31	12.2	24.4	0.015	
2	3.15	12.2	12.2	0.025	
2	12.6	3.7	30	0.01	
2	22100000	9.14	1.52	1	
2	34700	12.2	4.57	0.008	
2	31500	15.2	6.1	0.05	
2	3.15	3.66	9.14	0.04	
2	315	9.14	21.3	0.005	
2	315	8.53	19	0.025	
2	-999	4.88	-999	-	
2	-999	3.05	-999	0.024	
2	63.1	4.57	19.8	0.04	
2	189	6.1	61	0.023	
2	22100000	4.57	1.83	1	
2	-999	183	12.2	0.000	
2	22.1	2.74	3.05	-	
2	189	15.2	61	0.012	
2	11000	15.2	22.9	0.000	
2	-999	3.66	18.3	-	
2	63.1	8.23	518	0.007	
2	126	4.57	107	0.03	
2	-999	1.52	91.4	-	
2	-7.68877	3.469	4.2618	-4.42479	Mean
2	12.3279	1.3250	0.47331	-1.46902	Covariance
2	1.32509	0.5420	-0.01357	-0.1757	Covariance
2	0.47331	-0.01357	1.61831	-0.39626	Covariance
2	-1.46902	-	-0.39626	1.75145	Covariance
2	3.15	1.52	1.52	0.000	Minimum
2	22100000	183	518	1	Maximum
3	25500	3.66	3.66	0.000	
3	946	9.14	5.33	0.005	
3	1260	1.77	6.1	0.000000004	
3	28.4	6.1	-999	0.034	
3	3780	16.8	1.52	0.04	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
3	2680	6.7	2.4	0.009	
3	31.5	9.45	-999	0.05	
3	-999	7.62	-999	0.01	
3	63.1	2.3	4.12	0.007	
3	6620	30.5	21.3	0.02	
3	126	3.06	15.2	0.01	
3	31.5	-999	-999	0.01	
3	8830	5.33	45.7	0.000	
3	158	0.91	4.57	0.003	
3	6.31	1.37	3.66	0.027	
3	9.46	2.56	2.74	0.042	
3	-7.81342	2.7277	2.93298	-4.6888	Mean
3	21.2765	2.7807	0.6463	-1.30916	Covariance
3	2.78074	1.0703	0.17468	0.29718	Covariance
3	0.646	0.1746	0.96341	-0.64536	Covariance
3	-1.30916	0.2971	-0.64536	1.970	Covariance
3	6.31	0.914	1.52	0.000000004	Minimum
3	25500	30.5	45.7	0.05	Maximum
4	50800	4.57	9.14	0.005	
4	13900	-999	33.5	0.028	
4	-999	6.1	-999	-	
4	-999	12.2	4.57	0.01	
4	1580	2.13	12.2	0.001	
4	3.15	19.8	2.44	0.007	
4	12.6	4.57	10.7	0.07	
4	-999	0.91	6.1	0.043	
4	2520	1.52	3.05	0.02	
4	3150	2.44	-999	0.000002	
4	9.46	1.83	6.04	0.055	
4	94.6	0.61	3.96	0.006	
4	-999	6.98	53.3	-	
4	11600	15.2	76.2	0.004	
4	12600	7.62	6.4	0.049	
4	4100	2.13	32	0.003	
4	-999	10.7	8.53	0.000	
4	-999	0.61	7.62	0.001	
4	3150	0.30	9.14	0.003	
4	221	1.52	7.62	0.004	
4	-999	4.57	27.4	0.015	
4	3.15	3.05	3.05	0.02	
4	631	2.44	7.62	0.005	
4	-999	50.8	145	0.092	
4	-999	15.2	6.1	0.0000001	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
4	31.	33.5	-	0.023	
4	315	9.14	3.05	0.002	
4	4420	1.52	19.8	0.002	
4	631	2.21	0.33	0.001	
4	-999	1.22	-999	-	
4	-999	9.14	3.05	0.005	
4	7880	22.9	3.05	0.02	
4	5360	3.05	6.1	0.001	
4	-6.82634	2.6587	3.3063	-4.9212	Mean
4	9.60704	0.5103	1.46619	-1.4956	Covariance
4	0.51036	1.522	-0.01024	0.093	Covariance
4	1.46619	-0.01024	1.28413	-0.02391	Covariance
4	-	0.093	-0.02391	1.83998	Covariance
4	3.15	0.305	0.33	0.0000001	Minimum
4	11600	50.8	145	0.092	Maximum
5	5680	3.05	21.3	0.002	
5	-999	0.91	3.96	-	
5	946	-999	15.2	0.093	
5	-999	3.05	6.1	0.01	
5	15800	6.1	3.05	0.000	
5	63100	5.18	1.52	0.005	
5	-999	6.1	3.05	0.005	
5	15.6	38.1	1.52	0.025	
5	12600	4.57	4.57	0.001	
5	-999	4.57	22.9	0.03	
5	7570	30.5	-999	-	
5	-999	101	15.2	0.05	
5	1580	33.5	914	0.001	
5	31500	30.5	24.4	0.001	
5	-999	9.75	15.2	-	
5	6.31	3.38	7.62	0.003	
5	-999	32.9	4.57	-	
5	23700	42.7	6.1	0.003	
5	-999	10.7	1.07	-	
5	1580	19.8	24.4	0.005	
5	1260	2.44	-999	-	
5	3150	12.2	3.81	-	
5	126	15.2	4.57	0.002	
5	946	3.05	3.05	0.002	
5	-999	4.57	-999	-	
5	-999	2.44	-999	-	
5	1390	34.1	91.4	0.003	
5	-999	12.2	85.3	-	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	**Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)**	**Unsaturated Zone Thickness [DSOIL] (m)**	**Aquifer Thickness [ZB] (m)**	**Hydraulic Gradient [GRADNT] (m/m)**	**Notes**
5	-999	3.6	-	-	
5	-999	27.4	-999	0.006	
5	-999	15.9	16.2	0.000	
5	94.6	7.01	9.14	0.000	
5	2840	42.7	30.5	0.002	
5	158	1	130	0.001	
5	-999	18.3	3.66	0.01	
5	1260	7.32	18.3	0.000	
5	63.1	82.3	-999	-	
5	15800	36.6	-999	0.001	
5	3470	7.62	15.2	0.02	
5	-999	12.2	15.2	0.001	
5	126	1.83	11	0.002	
5	2210	15.2	9.14	-	
5	3.15	3.66	2.44	0.005	
5	-999	12.2	48.8	0.01	
5	-999	36.6	-999	0.068	
5	63700	6	-999	-	
5	3.15	6	15.2	0.015	
5	-999	7.01	18.3	-	
5	631	14.6	24.4	0.003	
5	3190000	9.14	0.30	0.000002	
5	3150	10.7	3.05	0.006	
5	3.15	4.72	18.3	0.07	
5	946	13.7	6.1	0.008	
5	3150	7.62	7.62	-	
5	315	4.88	9.14	0.017	
5	11000	2.44	6.1	-	
5	-999	2.44	5.18	0.04	
5	-999	3.96	18.3	-	
5	12.6	2.13	0.61	-	
5	2210	9.14	1.52	0.025	
5	-999	3.05	6.1	0.013	
5	22100	6.1	91.4	0.001	
5	-5.61434	3.4383	3.53678	-5.61773	Mean
5	9.98295	0.2801	0.08839	-2.96927	Covariance
5	0.28014	0.839	0.54136	0.044	Covariance
5	0.08839	0.5413	2.05569	-0.71488	Covariance
5	2.96927	0.044	-0.71488	4.17328	Covariance
5	3.15	0.914	0.30	0.000002	Minimum
5	3190000	101	914	0.093	Maximum
6	-999	15.2	18.3	0.005	
6	-999	1.83	9.14	0.002	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
6	315	4.8	15.	0.001	
6	631	8.53	9.14	0.01	
6	10700	3.51	7.32	0.005	
6	1890	24.4	36.6	0.001	
6	3.15	2.74	3.66	0.003	
6	-999	21.3	7.62	0.001	
6	4100	27.4	3.05	0.001	
6	16700	2.44	6.4	0.004	
6	11000	5.49	13.1	0.002	
6	315	1.52	3.05	0.002	
6	-999	1.22	1.83	0.008	
6	11000	5.79	-999	0.000	
6	-999	3.96	4.27	0.017	
6	-999	12.2	16.8	0.002	
6	1580	4.57	7.62	0.04	
6	33100	30.5	22.9	0.01	
6	-999	4.57	7.62	0.1	
6	252	11.5	-999	0.005	
6	14200	4.57	18.3	0.000	
6	3150	1.52	1.52	0.0000004	
6	5680	3.05	6.1	0.001	
6	1890	3.66	6.1	0.002	
6	315	3.66	0.61	0.000001	
6	31.5	1.52	-999	0.00000002	
6	3150	1.19	3.66	-	
6	15500	5.18	7.93	0.006	
6	5520	3.66	5.49	0.01	
6	3150	3.05	16.8	0.013	
6	158	1.52	3.05	0.012	
6	22.1	1.22	13.7	0.004	
6	-999	1.83	9.14	0.011	
6	9.46	0.914	6.1	0.008	
6	-999	10.7	15.2	0.00008	
6	-999	12.2	12.2	0.000001	
6	-	2.6584	3.15814	-5.6184	Mean
6	13.8058	1.6770	2.14642	-0.09303	Covariance
6	1.67704	0.898	0.34951	-0.23716	Covariance
6	2.14642	0.3495	0.86919	0.00252	Covariance
6	-0.09303	-0.23716	0.00252	1.23921	Covariance
6	3.15	0.914	0.61	0.00000002	Minimum
6	10700	30.5	36.6	0.1	Maximum
7	946	2.44	8.23	0.002	
7	1260	2.13	305	0.003	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydrogeologic Environment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
7	-999	35.	-	-	
7	6940	-999	22.9	0.003	
7	23300	15.2	36.6	0.004	
7	4420	1.83	38.1	0.000	
7	56100	3.05	10.1	0.002	
7	55200	3.05	61	-	
7	9460	57.9	9.14	0.000001	
7	-999	9.14	9.14	0.000	
7	-999	12.2	9.14	0.002	
7	946	3.05	3.05	0.008	
7	9780	3.05	3.05	0.013	
7	-999	5.18	12.2	0.002	
7	4420	3.66	15.2	0.005	
7	4420	24.4	21.3	0.01	
7	1580	1.52	24.4	0.01	
7	82000	14.9	8.53	0.003	
7	946	12.2	18.3	0.000002	
7	11000	3.05	4.57	-	
7	-999	4.57	13.7	0.01	
7	6940	2.13	7.99	0.004	
7	6310	7.01	5.18	0.049	
7	23700	4.88	18.3	0.033	
7	17700	5.79	42.7	0.002	
7	1890	4.57	10.7	0.000004	
7	14500	1.52	18.3	0.012	
7	12000	2	-999	0.01	
7	2520	1.52	6.1	0.011	
7	12.6	5.79	4.27	0.021	
7	315	0.61	4.57	0.006	
7	31.5	0.457	-999	0.001	
7	-999	45.7	3.05	-	
7	-5.22204	2.8144	3.78819	-5.30668	Mean
7	13.0649	-1.10808	0.50353	-0.73884	Covariance
7	-1.10808	1.1384	0.0496	0.26902	Covariance
7	0.50353	0.049	1.11517	-0.46202	Covariance
7	-0.73884	0.2690	-0.46202	1.11713	Covariance
7	12.6	0.457	3.05	0.000001	Minimum
7	12000	57.9	305	0.049	Maximum
8	6310	7.62	61	0.001	
8	24000	4.88	22.9	0.002	
8	30000	2.99	18.9	0.004	
8	-999	12.2	6.71	0.001	
8	2520	3.05	21.3	0.0000008	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydrogeologic Environment [IGWR]	**Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)**	**Unsaturated Zone Thickness [DSOIL] (m)**	**Aquifer Thickness [ZB] (m)**	**Hydraulic Gradient [GRADNT] (m/m)**	**Notes**
8	11000	9.1	21.	0.004	
8	13300	5.49	12.2	0.006	
8	37800	4.57	9.14	0.003	
8	1260	10.7	-999	0.008	
8	2210	3.05	22.9	0.000	
8	9780	3.35	15.2	0.000	
8	1890	48.8	32	0.03	
8	34400	7.62	26.2	0.006	
8	44200	4.88	18.6	0.002	
8	15800	2	24.4	0.001	
8	7250	9.14	39.6	0.000	
8	13900	12.2	122	0.002	
8	29000	2.74	10.1	-	
8	99700	2.13	7.01	0.000	
8	-999	4.57	6.1	0.003	
8	14800	1.83	61	0.001	
8	7880	2.44	3.05	0.03	
8	-999	15.2	76.2	0.000	
8	5680	2.44	6.1	0.001	
8	18900	4.57	7.62	0.005	
8	3880	3.66	7.62	0.004	
8	-999	2	18.3	0.000	
8	473	6.1	4.57	0.017	
8	10400	7.62	30.5	0.001	
8	22100	9.14	7.62	0.005	
8	27800	7.62	24.4	0.002	
8	27800	7.62	24.4	0.002	
8	-999	6.1	4.57	0.00004	
8	11000	12.2	3.05	0.075	
8	19200	5.33	12.2	0.008	
8	631	0.91	10.7	0.01	
8	19200	18.3	10.7	0.013	
8	5050	0.61	12.2	0.003	
8	-999	7.62	30.5	0.002	
8	33100	15.2	30.5	0.000	
8	-999	4.57	22.9	0.01	
8	2210	2.13	3.66	0.02	
8	60900	2	30.5	0.003	
8	-3.59646	2.9737	3.92385	-5.86511	Mean
8	5.02	0.4862	0.15471	-0.8019	Covariance
8	0.486	0.8555	0.26963	0.07004	Covariance
8	0.154	0.2696	0.75329	-0.62236	Covariance
8	-	0.0700	-0.62236	1.62199	Covariance

Table J-A-2. Correlated Empirical Distributions (continued)

Hydrogeologic Environment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
8	473	0.6	3.0	0.0000008	Minimum
8	11000	48.8	122	0.075	Maximum
9	946	2.1	13.7	0.05	
9	315	13.7	12.2	0.001	
9	18.9	3.66	5.49	0.008	
9	21800	6.1	15.2	0.004	
9	3470	39.6	54.9	0.017	
9	3150	21.3	4.57	0.01	
9	126	1	30	-	
9	31.5	7.62	3.05	0.009	
9	-999	3.05	30.5	0.0000005	
9	31.5	5.18	10.7	0.03	
9	315	3.96	22.9	0.007	
9	63.1	4.57	2.96	0.022	
9	915	2.44	12.2	0.000	
9	-999	7.32	12.2	-	
9	1890	1.83	0.91	0.005	
9	3150	7.62	7.62	-	
9	631	3.66	2.13	-	
9	6310	2.44	9.14	0.00000004	
9	-999	2.13	7.62	0.009	
9	4100	1.52	6.1	0.01	
9	126	3.05	4.57	0.05	
9	126	3.05	7.62	0.02	
9	-999	0.61	1.83	-	
9	12.6	1.83	-999	0.04	
9	8830	1.52	18.3	0.004	
9	315	1.52	6.1	-	
9	284	1.74	9.14	0.01	
9	9.46	18.3	2.44	0.003	
9	1580	3.35	6.1	0.000004	
9	-7.67984	2.4855	3.22796	-4.68545	Mean
9	11.25	0.1708	0.72472	-0.72109	Covariance
9	0.17085	0.8731	0.13478	-0.12094	Covariance
9	0.72472	0.1347	0.81983	-0.0043	Covariance
9	-0.72109	-0.12094	-0.0043	1.28625	Covariance
9	9.46	0.61	0.91	0.00000004	Minimum
9	21800	39.6	54.9	0.05	Maximum
10	-999	3.35	14.6	0.03	
10	4420	11.6	54.9	0.005	
10	284	4.57	7.62	0.01	
10	19600	39.6	21.4	0.000	
10	158	4.57	3.05	0.000	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
10	315	1.5	6.1	0.004	
10	-999	6.1	3.66	0.000001	
10	126	7.62	2.29	0.005	
10	315	15.2	10.7	0.01	
10	31.5	2.74	6.86	0.017	
10	126	3.05	4.12	0.003	
10	-999	3.81	6.1	0.00001	
10	-999	3.66	15.2	0.1	
10	631	4.57	0.91	0.005	
10	3470	3.05	3.05	0.002	
10	2210	25.9	7.62	0.00001	
10	-999	1.52	15.2	0.002	
10	2840	2.74	4.57	-	
10	-999	1.83	2.44	0.008	
10	2210	13.7	7.62	0.01	
10	126	12.2	12.2	0.025	
10	-999	3.81	16.8	0.002	
10	-999	3.32	1.83	0.06	
10	3.15	3.66	11.6	0.01	
10	25.2	1.83	4.57	0.009	
10	4420	10.7	9.14	0.014	
10	-999	6.1	42.7	0.00175	
10	-6.97635	2.8094	3.15655	-5.57335	Mean
10	4.99889	1.2799	0.51266	-1.74813	Covariance
10	1.27993	0.8603	0.40799	-0.71454	Covariance
10	0.51266	0.4079	0.8467	0.03369	Covariance
10	-1.74813	-0.71454	0.03369	3.61694	Covariance
10	3.15	1.52	0.91	0.000001	Minimum
10	19600	39.6	54.9	0.1	Maximum
11	946	2.13	305	0.01	
11	63.1	2.74	30.5	0.03	
11	7250	9.14	36.6	0.000	
11	24300	4.57	10.7	0.006	
11	-999	1.52	305	0.001	
11	7570	3.05	45.7	0.006	
11	12600	0.91	4.57	0.005	
11	631	0.91	6.1	0.01	
11	3150	1.52	6.1	-	
11	1260	1.22	10.7	0.002	
11	31.5	0.914	15.2	0.005	
11	13900	1.52	61	0.002	
11	-999	1.68	15.2	0.002	
11	2520	2	2	0.002	

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
11	1260	1.2	3.0	0.017	
11	-999	0.91	7.62	-	
11	315	1.52	1.52	0.05	
11	1580	2.74	4.57	0.023	
11	-999	3.35	4.27	0.019	
11	315	3.05	24.4	0.001	
11	284	1.07	30.5	0.003	
11	946	2.13	1.68	0.000	
11	-999	2.74	21.3	0.00003	
11	8170	7.01	6.1	0.003	
11	-999	-999	6.71	-	
11	-999	3.05	42.7	0.000	
11	-5.38023	1.899	3.7492	-5.61773	Mean
11	3.48349	0.5251	-0.00422	-0.63963	Covariance
11	0.52513	0.4690	0.18069	-0.2284	Covariance
11	-0.00429	0.1806	2.02612	-0.08327	Covariance
11	-0.63963	-	-0.08327	1.97797	Covariance
11	31.5	0.914	1.52	0.00003	Minimum
11	24300	9.14	305	0.05	Maximum
12	15800	3	3	0.006	
12	-999	5	10	0.005	
12	1580	50.8	144	0.023	
12	-999	15.2	91.4	-	
12	-999	3.05	-999	0.012	
12	1580	45.7	-999	-	
12	126	3.05	15.2	0.00005	
12	315	12.2	61	0.033	
12	-999	30.5	-999	0.02	
12	-999	320	-999	0.009	
12	-999	5.33	15.2	0.001	
12	15800	29.3	19.5	-	
12	-999	18.3	-999	-	
12	221	-999	39.6	0.002	
12	315	3.96	3.05	0.018	
12	24900	1.52	-999	0.002	
12	12300	3.96	18.3	0.009	
12	-999	3.05	305	0.001	
12	94.6	7.62	19.8	0.01	
12	1260	400	1	0.000002	
12	2180	1.68	7.32	0.00042	
12	6310	1.22	3.05	-	
12	-	3.4776	4.32063	-5.49537	Mean
12	12.0503	1.4325	0.53279	0.79733	Covariance

Table J-A-2. Correlated Empirical Distributions (continued)

Hydro-geologic Environ-ment [IGWR]	Long. Hydraulic Conductivity Aquifer [XKX] (m/yr)	Unsaturated Zone Thickness [DSOIL] (m)	Aquifer Thickness [ZB] (m)	Hydraulic Gradient [GRADNT] (m/m)	Notes
12	1.43257	1.25667	0.99541	1.35511	Covariance
12	0.53279	0.9954	1.2437	0.81321	Covariance
12	0.79733	1.3551	0.81132	4.45451	Covariance
12	94.6	1.22	3.05	0.000002	Minimum
12	15800	400	305	0.033	Maximum
13	1890	5.18	10.1	0.005	
13	-	3.4776	4.32063	-5.49537	Mean
13	12.0503	1.4325	0.53279	0.79733	Covariance
13	1.43257	1.2566	0.99541	1.35511	Covariance
13	0.53279	0.9954	1.2437	0.81321	Covariance
13	0.79733	1.3551	0.81132	4.45451	Covariance
13	3.15	0.305	0.30	0.000000004	Minimum
13	22100000	610	914	1	Maximum

References:

U.S.EPA, 2003b

Table J-A-3. Empirical Distributions of Selected Parameters for Groundwater Modeling

VariableID	Parameter [Reference]	Units	Value	CDF
AL	Longitudinal dispersivity (aquifer)	m	0.1	0
			1	0.1
			10	0.7
			100	1
DIAM	Avg. particle diameter	cm	0.00039	0
			0.00078	0.038
			0.0016	0.104
			0.0031	0.171
			0.0063	0.262
			0.0125	0.371
			0.025	0.56
			0.05	0.792
			0.1	0.904
			0.2	0.944
			0.4	0.976
			0.8	1
PH	Groundwater pH	std. Units	3.2	0
			3.6	0.01
			4.5	0.05
			5.2	0.1
			6.07	0.25
			6.8	0.5
			7.4	0.75
			7.9	0.9
			8.2	0.95
			8.95	0.99
			9.7	1
USPH	Unsaturated Zone pH	std. Units	3.2	0
			3.6	0.01
			4.5	0.05
			5.2	0.1
			6.07	0.25
			6.8	0.5
			7.4	0.75
			7.9	0.9
			8.2	0.95
			8.95	0.99
			9.7	1

CDF = Cumulative distribution function
References: U.S. EPA, 2003b

Table J-A-4. HELP Infiltration Rates for Regional Recharge

ICLR	City	State	Silt Loam (ISTYPE= 1)	Sandy Loam (ISTYPE1 = 2)	Silty Clay Loam (ISTYPE = 3)
1	Fresno	CA	0.0307	0.0368	0.0381
2	Boise	ID	0.0008	0.0094	0.0038
3	Denver	CO	0.0008	0.0008	0.0036
4	Grand Junction	CO	0.0000	0.0000	0.0003
5	Pocatello	ID	0.0000	0.0000	0.0000
6	Glasgow	KY	0.0099	0.0074	0.0099
7	Bismarck	ND	0.0239	0.0300	0.0196
8	Pullman	WA	0.0069	0.0132	0.0084
9	Yakima	WA	0.0000	0.0023	0.0003
10	Cheyenne	WY	0.0005	0.0013	0.0086
11	Lander	WY	0.0033	0.0053	0.0094
12	Los Angeles	CA	0.0787	0.0950	0.0699
13	Sacramento	CA	0.1024	0.0876	0.0945
14	San Diego	CA	0.0221	0.0340	0.0241
15	Santa maria	CA	0.0947	0.1151	0.0841
16	Ely	NV	0.0000	0.0000	0.0003
17	Rapid City	SD	0.0005	0.0071	0.0033
18	Cedar City	UT	0.0000	0.0008	0.0000
19	Albuquerque	NM	0.0000	0.0000	0.0003
20	Las Vegas	NV	0.0000	0.0000	0.0018
21	Phoenix	AZ	0.0000	0.0003	0.0003
22	Tucson	AZ	0.0000	0.0003	0.0005
23	El Paso	TX	0.0076	0.0130	0.0081
24	Medford	OR	0.2073	0.2309	0.2096
25	Great Falls	MT	0.0036	0.0069	0.0074
26	Salt Lake City	UT	0.0130	0.0269	0.0185
27	Grand Island	NE	0.0442	0.0627	0.0323
28	Flagstaff	AZ	0.0239	0.0630	0.0226
29	Dodge City	KS	0.0135	0.0345	0.0226
30	Midland	TX	0.0180	0.0254	0.0135
31	St. Cloud	MN	0.0602	0.0831	0.0554
32	E. Lansing	MI	0.1090	0.1452	0.1102
33	North Omaha	NE	0.0671	0.0795	0.0536

Table J-A-4. HELP Infiltration Rates for Regional Recharge (continued)

ICLR	City	State	Silt Loam (ISTYPE1 = 1)	Sandy Loam (ISTYPE1 = 2)	Silty Clay Loam (ISTYPE1 = 3)
34	Tulsa	OK	0.0686	0.1006	0.0465
35	Brownsville	TX	0.0549	0.1049	0.0384
36	Dallas	TX	0.0599	0.1067	0.0531
37	Oklahoma City	OK	0.0612	0.0942	0.0389
38	Concord	NH	0.1585	0.2057	0.1372
39	Pittsburg	PA	0.0894	0.1313	0.0792
40	Portland	OR	0.4171	0.4387	0.3927
41	Caribou	ME	0.1082	0.1491	0.0886
42	Chicago	IL	0.0798	0.1138	0.0620
43	Burlington	VT	0.1359	0.1781	0.1166
44	Bangor	ME	0.1471	0.2045	0.1227
45	Rutland	VT	0.1212	0.1598	0.1008
46	Seattle	WA	0.4384	0.4582	0.4077
47	Montpelier	VT	0.1062	0.1483	0.0879
48	Sault St. Marie	MI	0.1651	0.2101	0.1435
49	Put-in-Bay	OH	0.0508	0.1003	0.0495
50	Madison	WI	0.0912	0.1400	0.0686
51	Columbus	OH	0.0765	0.1158	0.0663
52	Cleveland	OH	0.0780	0.1212	0.0823
53	Des Moines	IA	0.1143	0.1641	0.1156
54	E. St. Louis	IL	0.1435	0.1676	0.0704
55	Columbia	MO	0.1529	0.1989	0.1224
56	Topeka	KS	0.1049	0.1483	0.0762
57	Tampa	FL	0.0658	0.1031	0.0475
58	San Antonio	TX	0.1095	0.1646	0.0820
59	Hartford	CT	0.1709	0.2228	0.1405
60	Syracuse	NY	0.2545	0.3251	0.2118
61	Worchester	MA	0.2022	0.2591	0.1697
62	Augusta	ME	0.2116	0.2700	0.1674
63	Providence	RI	0.2131	0.2863	0.1753
64	Portland	ME	0.2294	0.2840	0.1872
65	Nashua	NH	0.2268	0.2812	0.1943
66	Ithaca	NY	0.1684	0.2136	0.1392
67	Boston	MA	0.2332	0.2383	0.1542

Table J-A-4. HELP Infiltration Rates for Regional Recharge (continued)

ICLR	City	State	Silt Loam (ISTYPE1 = 1)	Sandy Loam (ISTYPE1 = 2)	Silty Clay Loam (ISTYPE1 = 3)
68	Schenectady	NY	0.1473	0.1928	0.1224
69	Lynchburg	VA	0.3081	0.3612	0.2570
70	New York City	NY	0.2436	0.2944	0.1969
71	Philadelphia	PA	0.2007	0.2609	0.1641
72	Seabrook	NJ	0.1814	0.2428	0.1427
73	Indianapolis	IN	0.1300	0.1862	0.1064
74	Cincinnati	OH	0.1554	0.2210	0.1539
75	Bridgeport	CT	0.1953	0.2464	0.1615
76	Orlando	FL	0.1016	0.1697	0.0805
77	Greensboro	NC	0.3256	0.3896	0.2705
78	Jacksonville	FL	0.1511	0.2106	0.1102
79	Watkinsville	GA	0.2891	0.3556	0.2332
80	Norfolk	VA	0.3122	0.0000	0.2685
81	Shreveport	LA	0.2296	0.2939	0.1842
82	Astoria	OR	1.0762	1.1494	0.9647
83	New Haven	CT	0.3520	0.4628	0.2855
84	Plainfield	MA	0.1900	0.2540	0.1521
85	Knoxville	TN	0.4107	0.4460	0.3543
86	Central Park	NY	0.3363	0.4171	0.2738
87	Lexington	KY	0.3294	0.3970	0.2700
88	Edison	NJ	0.3122	0.3914	0.2492
89	Nashville	TN	0.4674	0.5395	0.3769
90	Little Rock	AK	0.3531	0.4336	0.2824
91	Tallahassee	FL	0.5913	0.7308	0.4564
92	New Orleans	LA	0.5893	0.7445	0.4503
93	Charleston	SC	0.2609	0.3287	0.2123
94	W. Palm Beach	FL	0.2611	0.3490	0.1783
95	Atlanta	GA	0.3416	0.3993	0.2822
96	Lake Charles	LA	0.3647	0.4641	0.2817
97	Miami	FL	0.1450	0.2201	0.1019
98	Annette	AK	1.6833	1.8354	1.4610
99	Bethel	AK	0.0564	0.0721	0.0554
100	Fairbanks	AK	0.0104	0.0234	0.0117
101	Honolulu	HI	0.0523	0.0945	0.0366
102	San Juan	PR	0.1267	0.1923	0.0945

Reference: U.S.EPA, 2003b

Appendix K:
Detailed Human Health Results

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Table K-1. Detailed Human Health Results

(Based on Home Gardener Consumption Rate Distributions for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
Arsenic - Cancer							
228	Child-HG	50	Soil Ingestion	4.2E-01	1.8E-07	2.7E-02	80
8883	Child-HG	50	Protected Veg	3.9E-04	3.1E-08	4.7E-03	471
3686	Child-HG	50	Exposed Veg	8.2E-04	7.5E-08	1.1E-02	197
4474	Child-HG	50	Protected Fruit	2.0E-04	6.5E-08	9.7E-03	227
8971	Child-HG	50	Exposed Fruit	2.1E-04	3.1E-08	4.7E-03	472
6301	Child-HG	50	Root Veg	5.7E-04	3.8E-08	5.7E-03	386
5114	Child-HG	50	Total Ingestion	NA	5.4E-07	8.1E-02	27
430	Child-HG	90	Soil Ingestion	4.5E-01	3.2E-07	4.8E-02	46
8373	Child-HG	90	Protected Veg	3.9E-04	1.0E-07	1.5E-02	148
6864	Child-HG	90	Exposed Veg	8.4E-04	3.1E-07	4.6E-02	48
590	Child-HG	90	Protected Fruit	2.0E-04	2.6E-07	3.9E-02	56
2015	Child-HG	90	Exposed Fruit	3.0E-04	1.1E-07	1.6E-02	136
6176	Child-HG	90	Root Veg	5.8E-04	2.2E-07	3.3E-02	67
4734	Child-HG	90	Total Ingestion	NA	1.1E-06	1.7E-01	13
2638	Adult-HG	50	Soil Ingestion	4.5E-01	2.7E-08	4.0E-03	549
2958	Adult-HG	50	Protected Veg	3.7E-04	2.6E-08	3.9E-03	565
2631	Adult-HG	50	Exposed Veg	7.9E-04	6.6E-08	9.9E-03	222
5299	Adult-HG	50	Protected Fruit	2.1E-04	4.4E-08	6.6E-03	333
130	Adult-HG	50	Exposed Fruit	3.0E-04	2.4E-08	3.6E-03	608
1697	Adult-HG	50	Root Veg	5.5E-04	3.8E-08	5.6E-03	392
4772	Adult-HG	50	Total Ingestion	NA	3.7E-07	5.6E-02	39
6628	Adult-HG	90	Soil Ingestion	5.0E-01	8.2E-07	1.2E-02	180
11	Adult-HG	90	Protected Veg	3.1E-04	1.8E-07	2.7E-02	82
6058	Adult-HG	90	Exposed Veg	8.0E-04	5.9E-07	8.9E-02	25
9680	Adult-HG	90	Protected Fruit	2.0E-04	4.5E-07	6.7E-02	33
6685	Adult-HG	90	Exposed Fruit	2.9E-04	1.8E-07	2.7E-02	81
6301	Adult-HG	90	Root Veg	5.6E-04	3.7E-07	5.4E-02	40
7831	Adult-HG	90	Total Ingestion	NA	1.8E-06	2.8E-01	8
Arsenic- Noncancer							
5759	Child-HG	50	Soil Ingestion	7.4E-01	2.6E-06	1.5E-02	150
1301	Child-HG	50	Protected Veg	3.8E-04	4.1E-07	1.4E-03	1,600
4323	Child-HG	50	Exposed Veg	8.1E-04	1.1E-06	3.6E-03	616
529	Child-HG	50	Protected Fruit	2.0E-04	9.2E-07	3.1E-03	718
7266	Child-HG	50	Exposed Fruit	3.1E-04	4.4E-07	1.5E-03	1,508
9022	Child-HG	50	Root Veg	5.8E-04	5.2E-07	1.7E-03	1,259
6244	Child-HG	50	Total Ingestion	NA	7.1E-06	2.4E-02	93
464	Child-HG	90	Soil Ingestion	7.3E-01	3.9E-06	2.2E-02	102
5066	Child-HG	90	Protected Veg	3.9E-04	1.1E-06	3.8E-03	580

Table K-1. Detailed Human Health Results

(Based on Home Gardener Consumption Rate Distributions for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
2672	Child-HG	90	Exposed Veg	8.0E-04	3.5E-06	1.2E-02	186
1342	Child-HG	90	Protected Fruit	2.1E-04	3.1E-06	1.0E-02	213
4818	Child-HG	90	Exposed Fruit	3.0E-04	1.1E-06	3.8E-03	576
1383	Child-HG	90	Root Veg	5.7E-04	2.4E-06	8.1E-03	272
1587	Child-HG	90	Total Ingestion	NA	1.2E-05	3.9E-02	56
254	Adult-HG	50	Soil Ingestion	5.9E-01	2.3E-07	1.3E-03	1,694
197	Adult-HG	50	Protected Veg	3.9E-04	2.2E-07	7.3E-04	3,022
4519	Adult-HG	50	Exposed Veg	7.7E-04	6.6E-07	2.2E-03	1,002
2590	Adult-HG	50	Protected Fruit	1.9E-04	3.8E-07	1.3E-03	1,758
5585	Adult-HG	50	Exposed Fruit	2.9E-04	2.1E-07	6.9E-04	3,174
4446	Adult-HG	50	Root Veg	5.4E-04	3.8E-07	1.3E-03	1,718
6559	Adult-HG	50	Total Ingestion	NA	3.0E-06	1.0E-02	219
7850	Adult-HG	90	Soil Ingestion	9.7E-01	4.0E-07	2.2E-03	986
6578	Adult-HG	90	Protected Veg	3.9E-04	7.4E-07	2.5E-03	897
6391	Adult-HG	90	Exposed Veg	7.8E-04	2.5E-06	8.4E-03	263
2392	Adult-HG	90	Protected Fruit	2.0E-04	2.2E-06	7.3E-03	302
2095	Adult-HG	90	Exposed Fruit	2.9E-04	8.2E-07	2.7E-03	802
4086	Adult-HG	90	Root Veg	5.8E-04	1.5E-06	5.0E-03	440
9631	Adult-HG	90	Total Ingestion	NA	6.2E-06	2.1E-02	106
Cobalt							
1898	Child-HG	50	Soil Ingestion	5.2E-01	3.6E-06	1.2E-02	1,038
5384	Child-HG	50	Protected Veg	1.3E-03	1.3E-06	4.3E-03	283
7962	Child-HG	50	Exposed Veg	1.1E-03	1.9E-06	6.3E-03	143
2528	Child-HG	50	Protected Fruit	7.0E-04	2.8E-06	9.4E-03	99
7592	Child-HG	50	Exposed Fruit	1.0E-03	1.3E-06	4.4E-03	267
2253	Child-HG	50	Root Veg	2.3E-03	2.0E-06	6.6E-03	111
495	Child-HG	50	Total Ingestion	NA	1.7E-05	5.5E-02	38
2268	Child-HG	90	Soil Ingestion	6.5E-01	6.2E-06	2.1E-02	106
3048	Child-HG	90	Protected Veg	1.4E-03	3.7E-06	1.2E-02	178
1940	Child-HG	90	Exposed Veg	1.6E-03	6.7E-06	2.2E-02	98
289	Child-HG	90	Protected Fruit	7.2E-04	1.0E-05	3.4E-02	65
5040	Child-HG	90	Exposed Fruit	1.0E-03	3.6E-06	1.2E-02	184
622	Child-HG	90	Root Veg	2.1E-03	9.8E-06	3.3E-02	67
5049	Child-HG	90	Total Ingestion	NA	3.1E-05	1.0E-01	22
5203	Adult-HG	50	Soil Ingestion	6.0E-01	3.1E-07	1.0E-03	2,099
6853	Adult-HG	50	Protected Veg	1.4E-03	6.3E-07	2.1E-03	1,045
1048	Adult-HG	50	Exposed Veg	1.6E-03	1.1E-06	3.6E-03	608
390	Adult-HG	50	Protected Fruit	6.8E-04	1.1E-06	3.6E-03	612
8993	Adult-HG	50	Exposed Fruit	2.3E-04	5.8E-07	1.9E-03	1133
1736	Adult-HG	50	Root Veg	2.3E-03	1.3E-06	4.5E-03	492

Table K-1. Detailed Human Health Results

(Based on Home Gardener Consumption Rate Distributions for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
569	Adult-HG	50	Total Ingestion	0.0E+00	7.9E-06	2.6E-02	83
5751	Adult-HG	90	Soil Ingestion	8.1E-01	6.4E-07	2.1E-03	1,038
9465	Adult-HG	90	Protected Veg	1.2E-03	2.3E-06	7.8E-03	283
4894	Adult-HG	90	Exposed Veg	1.5E-03	4.6E-06	1.5E-02	143
1399	Adult-HG	90	Protected Fruit	7.1E-04	6.7E-06	2.2E-02	99
5914	Adult-HG	90	Exposed Fruit	1.0E-03	2.5E-06	8.2E-03	267
6178	Adult-HG	90	Root Veg	2.5E-03	6.0E-06	2.0E-02	111
5661	Adult-HG	90	Total Ingestion	NA	1.8E-05	5.8E-02	38
Iron							
1613	Child-HG	50	Soil Ingestion	5.5E-01	3.0E-06	4.3E-06	507,821
6612	Child-HG	50	Protected Veg	1.9E-04	1.9E-07	2.7E-07	Capped
506	Child-HG	50	Exposed Veg	2.8E-04	4.0E-07	5.8E-07	Capped
5073	Child-HG	50	Protected Fruit	9.4E-05	4.3E-07	6.1E-07	Capped
2396	Child-HG	50	Exposed Fruit	1.2E-04	2.1E-07	3.0E-07	Capped
5087	Child-HG	50	Root Veg	4.8E-04	4.2E-07	5.9E-07	Capped
7672	Child-HG	50	Total Ingestion	NA	5.5E-06	7.9E-06	277,777
1658	Child-HG	90	Soil Ingestion	9.0E-01	5.9E-06	8.5E-06	260,230
362	Child-HG	90	Protected Veg	1.8E-04	5.4E-07	7.7E-07	Capped
7137	Child-HG	90	Exposed Veg	4.0E-04	1.4E-06	2.0E-06	Capped
4045	Child-HG	90	Protected Fruit	1.0E-04	1.5E-06	2.1E-06	Capped
5016	Child-HG	90	Exposed Fruit	1.6E-04	5.7E-07	8.1E-07	Capped
806	Child-HG	90	Root Veg	5.0E-04	2.0E-06	2.8E-06	260,230
3020	Child-HG	90	Total Ingestion	NA	9.6E-06	1.4E-05	160,912
7194	Adult-HG	50	Soil Ingestion	3.8E-01	2.3E-07	3.3E-07	Capped
5075	Adult-HG	50	Protected Veg	1.6E-04	9.2E-08	1.3E-07	Capped
8713	Adult-HG	50	Exposed Veg	3.1E-04	2.3E-07	3.3E-07	Capped
3929	Adult-HG	50	Protected Fruit	9.4E-05	1.6E-07	2.3E-07	Capped
9918	Adult-HG	50	Exposed Fruit	1.2E-04	9.0E-08	1.3E-07	Capped
3393	Adult-HG	50	Root Veg	2.5E-04	2.8E-07	4.0E-07	Capped
959	Adult-HG	50	Total Ingestion	NA	1.6E-06	2.2E-06	980,056
3159	Adult-HG	90	Soil Ingestion	8.1E-01	5.9E-07	8.4E-07	Capped
6636	Adult-HG	90	Protected Veg	1.8E-04	3.2E-07	4.6E-07	Capped
6766	Adult-HG	90	Exposed Veg	3.1E-04	8.8E-07	1.3E-06	Capped
6819	Adult-HG	90	Protected Fruit	1.0E-04	9.3E-07	1.3E-06	Capped
8211	Adult-HG	90	Exposed Fruit	1.8E-04	3.8E-07	5.4E-07	Capped
3179	Adult-HG	90	Root Veg	4.9E-04	1.1E-06	1.6E-06	Capped
9766	Adult-HG	90	Total Ingestion	NA	3.2E-06	4.6E-06	489,027

Capped = Modeling estimates indicated risks below levels of concern at concentrations above 1,000,000 mg kg^{-1} (i.e., SFS could be comprised entirely of this constituent and still not cause risk).

Table K-2. Detailed Human Health Results

(Based on General Population Median Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
Arsenic - Cancer							
228	Child-GP Med	50	Soil Ingestion	4.2E-01	1.8E-07	2.7E-02	80
9569	Child-GP Med	50	Protected Veg	3.9E-04	1.5E-08	2.2E-03	1001
6290	Child-GP Med	50	Exposed Veg	8.3E-04	1.5E-08	2.3E-03	971
3055	Child-GP Med	50	Protected Fruit	2.1E-04	1.4E-08	2.0E-03	1092
7051	Child-GP Med	50	Exposed Fruit	3.2E-04	3.4E-08	5.1E-03	435
9569	Child-GP Med	50	Root Veg	5.8E-04	2.7E-08	4.0E-03	546
5208	Child-GP Med	50	Total Ingestion	NA	2.9E-07	4.4E-02	50
430	Child-GP Med	90	Soil Ingestion	4.5E-01	3.2E-07	4.8E-02	46
631	Child-GP Med	90	Protected Veg	3.8E-04	2.7E-08	4.1E-03	541
5371	Child-GP Med	90	Exposed Veg	8.0E-04	3.6E-08	5.4E-03	409
3314	Child-GP Med	90	Protected Fruit	2.0E-04	1.8E-08	2.7E-03	823
8637	Child-GP Med	90	Exposed Fruit	3.0E-04	5.6E-08	8.4E-03	263
5148	Child-GP Med	90	Root Veg	5.6E-04	5.4E-08	8.1E-03	271
2116	Child-GP Med	90	Total Ingestion	NA	5.0E-07	7.5E-02	30
2638	Adult-GP Med	50	Soil Ingestion	4.5E-01	2.7E-08	4.0E-03	549
1198	Adult-GP Med	50	Protected Veg	3.8E-04	1.2E-08	1.9E-03	1187
1198	Adult-GP Med	50	Exposed Veg	7.9E-04	4.1E-08	6.2E-03	357
1198	Adult-GP Med	50	Protected Fruit	2.0E-04	9.0E-09	1.3E-03	1635
455	Adult-GP Med	50	Exposed Fruit	2.9E-04	1.0E-08	1.5E-03	1451
1198	Adult-GP Med	50	Root Veg	5.6E-04	2.5E-08	3.8E-03	579
8883	Adult-GP Med	50	Total Ingestion	NA	1.2E-07	1.9E-02	118
6628	Adult-GP Med	90	Soil Ingestion	5.0E-01	8.2E-08	1.2E-02	180
3340	Adult-GP Med	90	Protected Veg	3.6E-04	5.3E-08	8.0E-03	276
3410	Adult-GP Med	90	Exposed Veg	6.8E-04	1.8E-07	2.7E-02	83
3340	Adult-GP Med	90	Protected Fruit	1.9E-04	3.9E-08	5.8E-03	380
2136	Adult-GP Med	90	Exposed Fruit	2.5E-04	4.4E-08	6.5E-03	337
3340	Adult-GP Med	90	Root Veg	5.4E-04	1.1E-07	1.6E-02	135
1770	Adult-GP Med	90	Total Ingestion	NA	5.0E-07	7.4E-02	30
Arsenic- Noncancer							
5759	Child-GP Med	50	Soil Ingestion	7.4E-01	2.6E-06	8.8E-03	250
34	Child-GP Med	50	Protected Veg	3.8E-04	2.1E-07	7.0E-04	3142
9883	Child-GP Med	50	Exposed Veg	8.1E-04	2.2E-07	7.2E-04	3048
34	Child-GP Med	50	Protected Fruit	2.0E-04	1.9E-07	6.4E-04	3413
8356	Child-GP Med	50	Exposed Fruit	3.2E-04	4.5E-07	1.5E-03	1452
34	Child-GP Med	50	Root Veg	5.7E-04	3.9E-07	1.3E-03	1712
2391	Child-GP Med	50	Total Ingestion	NA	4.0E-06	1.3E-02	166
464	Child-GP Med	90	Soil Ingestion	7.3E-01	3.9E-06	1.3E-02	169
9993	Child-GP Med	90	Protected Veg	3.9E-04	2.2E-07	7.2E-04	3057

Table K-2. Detailed Human Health Results
(Based on General Population Median Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
9686	Child-GP Med	90	Exposed Veg	8.4E-04	2.2E-07	7.5E-04	2936
9993	Child-GP Med	90	Protected Fruit	2.1E-04	2.0E-07	6.6E-04	3321
8455	Child-GP Med	90	Exposed Fruit	3.4E-04	5.2E-07	1.7E-03	1259
9993	Child-GP Med	90	Root Veg	5.8E-04	4.0E-07	1.3E-03	1666
2525	Child-GP Med	90	Total Ingestion	NA	5.4E-06	1.8E-02	121
254	Adult-GP Med	50	Soil Ingestion	5.9E-01	2.3E-07	7.8E-04	2823
6349	Adult-GP Med	50	Protected Veg	3.9E-04	9.2E-08	3.1E-04	7174
4527	Adult-GP Med	50	Exposed Veg	8.1E-04	3.1E-07	1.0E-03	2152
6349	Adult-GP Med	50	Protected Fruit	2.0E-04	6.7E-08	2.2E-04	9882
5117	Adult-GP Med	50	Exposed Fruit	3.0E-04	7.6E-08	2.5E-04	8681
6349	Adult-GP Med	50	Root Veg	5.7E-04	1.9E-07	6.3E-04	3498
70	Adult-GP Med	50	Total Ingestion	NA	9.7E-07	3.2E-03	682
7850	Adult-GP Med	90	Soil Ingestion	9.7E-01	4.0E-07	1.3E-03	1644
5770	Adult-GP Med	90	Protected Veg	3.9E-04	9.4E-08	3.1E-04	7024
8505	Adult-GP Med	90	Exposed Veg	8.4E-04	3.2E-07	1.1E-03	2056
5770	Adult-GP Med	90	Protected Fruit	2.1E-04	6.8E-08	2.3E-04	9676
4018	Adult-GP Med	90	Exposed Fruit	3.5E-04	8.8E-08	2.9E-04	7484
5770	Adult-GP Med	90	Root Veg	5.8E-04	1.9E-07	6.4E-04	3425
4444	Adult-GP Med	90	Total Ingestion	NA	1.2E-06	3.9E-03	570
Cobalt							
1898	Child-GP Med	50	Soil Ingestion	5.2E-01	3.6E-06	1.2E-02	181
4798	Child-GP Med	50	Protected Veg	1.2E-03	6.7E-07	2.2E-03	986
6625	Child-GP Med	50	Exposed Veg	1.6E-03	4.2E-07	1.4E-03	1,570
2100	Child-GP Med	50	Protected Fruit	7.1E-04	5.7E-07	1.9E-03	1,152
6312	Child-GP Med	50	Exposed Fruit	1.1E-03	1.4E-06	4.8E-03	460
3001	Child-GP Med	50	Root Veg	2.3E-03	1.5E-06	5.2E-03	427
3059	Child-GP Med	50	Total Ingestion	NA	8.2E-06	2.7E-02	80
2268	Child-GP Med	90	Soil Ingestion	6.5E-01	6.2E-06	2.1E-02	106
9540	Child-GP Med	90	Protected Veg	1.4E-03	7.5E-07	2.5E-03	876
8465	Child-GP Med	90	Exposed Veg	1.6E-03	4.4E-07	1.5E-03	1,495
9540	Child-GP Med	90	Protected Fruit	7.2E-04	6.9E-07	2.3E-03	951
7152	Child-GP Med	90	Exposed Fruit	1.1E-03	1.7E-06	5.5E-03	399
9540	Child-GP Med	90	Root Veg	2.5E-03	1.7E-06	5.7E-03	384
8674	Child-GP Med	90	Total Ingestion	NA	1.1E-05	3.8E-02	58
5203	Adult-GP Med	50	Soil Ingestion	6.0E-01	3.1E-07	1.0E-03	2,099
6413	Adult-GP Med	50	Protected Veg	1.3E-03	3.1E-07	1.0E-03	2,115
2740	Adult-GP Med	50	Exposed Veg	1.6E-03	5.9E-07	2.0E-03	1,112
6413	Adult-GP Med	50	Protected Fruit	6.9E-04	2.3E-07	7.6E-04	2,913
2503	Adult-GP Med	50	Exposed Fruit	1.0E-03	2.6E-07	8.5E-04	2,586
6413	Adult-GP Med	50	Root Veg	2.4E-03	7.9E-07	2.6E-03	830

Table K-2. Detailed Human Health Results

(Based on General Population Median Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
5410	Adult-GP Med	50	Total Ingestion	NA	2.5E-06	8.3E-03	265
5751	Adult-GP Med	90	Soil Ingestion	8.1E-01	6.4E-07	2.1E-03	1,038
6328	Adult-GP Med	90	Protected Veg	1.4E-03	3.3E-07	1.1E-03	2,012
7792	Adult-GP Med	90	Exposed Veg	1.6E-03	6.3E-07	2.1E-03	1,053
6328	Adult-GP Med	90	Protected Fruit	7.2E-04	2.4E-07	7.9E-04	2,772
9954	Adult-GP Med	90	Exposed Fruit	1.1E-03	2.7E-07	9.1E-04	2,405
6328	Adult-GP Med	90	Root Veg	2.5E-03	8.4E-07	2.8E-03	790
5260	Adult-GP Med	90	Total Ingestion	NA	2.9E-06	9.7E-03	226
Iron							
1613	Child-GP Med	50	Soil Ingestion	5.5E-01	3.0E-06	4.3E-06	507,821
1020	Child-GP Med	50	Protected Veg	1.8E-04	9.7E-08	1.4E-07	Capped
3002	Child-GP Med	50	Exposed Veg	3.1E-04	8.2E-08	1.2E-07	Capped
20	Child-GP Med	50	Protected Fruit	9.3E-05	8.9E-08	1.3E-07	Capped
1262	Child-GP Med	50	Exposed Fruit	1.4E-04	2.2E-07	3.1E-07	Capped
7968	Child-GP Med	50	Root Veg	5.0E-04	3.1E-07	4.5E-07	Capped
6883	Child-GP Med	50	Total Ingestion	NA	3.8E-06	5.4E-06	404,714
1658	Child-GP Med	90	Soil Ingestion	9.0E-01	5.9E-06	8.5E-06	260,230
5217	Child-GP Med	90	Protected Veg	1.9E-04	1.1E-07	1.5E-07	Capped
9656	Child-GP Med	90	Exposed Veg	3.4E-04	9.1E-08	1.3E-07	Capped
5217	Child-GP Med	90	Protected Fruit	1.0E-04	9.8E-08	1.4E-07	Capped
6010	Child-GP Med	90	Exposed Fruit	1.8E-04	2.8E-07	4.1E-07	Capped
5217	Child-GP Med	90	Root Veg	5.0E-04	3.4E-07	4.8E-07	Capped
4792	Child-GP Med	90	Total Ingestion	NA	6.8E-06	9.7E-06	226,140
7194	Adult-GP Med	50	Soil Ingestion	3.8E-01	2.3E-07	3.3E-07	Capped
1139	Adult-GP Med	50	Protected Veg	1.7E-04	4.1E-08	5.9E-08	Capped
3977	Adult-GP Med	50	Exposed Veg	2.9E-04	1.1E-07	1.6E-07	Capped
1139	Adult-GP Med	50	Protected Fruit	9.1E-05	3.0E-08	4.3E-08	Capped
2167	Adult-GP Med	50	Exposed Fruit	1.4E-04	3.5E-08	5.0E-08	Capped
1139	Adult-GP Med	50	Root Veg	4.4E-04	1.5E-07	2.1E-07	Capped
1301	Adult-GP Med	50	Total Ingestion	NA	5.9E-07	8.5E-07	Capped
3159	Adult-GP Med	90	Soil Ingestion	8.1E-01	5.9E-07	8.4E-07	Capped
6286	Adult-GP Med	90	Protected Veg	1.9E-04	4.6E-08	6.6E-08	Capped
9385	Adult-GP Med	90	Exposed Veg	3.4E-04	1.3E-07	1.9E-07	Capped
6286	Adult-GP Med	90	Protected Fruit	1.0E-04	3.4E-08	4.8E-08	Capped
9461	Adult-GP Med	90	Exposed Fruit	1.9E-04	4.8E-08	6.9E-08	Capped
6286	Adult-GP Med	90	Root Veg	5.0E-04	1.7E-07	2.4E-07	Capped
5677	Adult-GP Med	90	Total Ingestion	NA	1.0E-06	1.4E-06	Capped

Capped = Modeling estimates indicated risks below levels of concern at concentrations above 1,000,000 mg kg^{-1} (i.e., SFS could be comprised entirely of this constituent and still not cause risk).

Table K-3. Detailed Human Health Results

(Based on General Population High Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
Arsenic - Cancer							
228	Child–GP High	50	Soil Ingestion	4.2E-01	1.8E-07	2.7E-02	80
9569	Child–GP High	50	Protected Veg	3.9E-04	4.5E-08	6.7E-03	327
7457	Child–GP High	50	Exposed Veg	8.3E-04	1.2E-07	1.8E-02	125
5947	Child–GP High	50	Protected Fruit	2.1E-04	7.2E-08	1.1E-02	204
3555	Child–GP High	50	Exposed Fruit	2.9E-04	1.8E-07	2.7E-02	82
9569	Child–GP High	50	Root Veg	5.8E-04	1.1E-07	1.7E-02	131
2701	Child–GP High	50	Total Ingestion	NA	7.4E-07	1.1E-01	20
430	Child–GP High	90	Soil Ingestion	4.5E-01	3.2E-07	4.8E-02	46
1485	Child–GP High	90	Protected Veg	3.8E-04	8.7E-08	1.3E-02	170
3921	Child–GP High	90	Exposed Veg	8.0E-04	2.3E-07	3.4E-02	64
631	Child–GP High	90	Protected Fruit	2.0E-04	1.3E-07	1.9E-02	114
5141	Child–GP High	90	Exposed Fruit	2.9E-04	2.5E-07	3.7E-02	60
631	Child–GP High	90	Root Veg	5.6E-04	2.2E-07	3.4E-02	66
1692	Child–GP High	90	Total Ingestion	NA	1.3E-06	1.9E-01	12
2638	Adult–GP High	50	Soil Ingestion	4.5E-01	2.7E-08	4.0E-03	549
1198	Adult–GP High	50	Protected Veg	3.8E-04	3.9E-08	5.8E-03	383
1198	Adult–GP High	50	Exposed Veg	7.9E-04	1.5E-07	2.2E-02	99
1198	Adult–GP High	50	Protected Fruit	2.0E-04	4.1E-08	6.1E-03	362
455	Adult–GP High	50	Exposed Fruit	2.9E-04	3.3E-08	5.0E-03	442
1198	Adult–GP High	50	Root Veg	5.6E-04	9.4E-08	1.4E-02	157
7041	Adult–GP High	50	Total Ingestion	NA	3.8E-07	5.7E-02	39
6628	Adult–GP High	90	Soil Ingestion	5.0E-01	8.2E-08	1.2E-02	180
3340	Adult–GP High	90	Protected Veg	3.6E-04	1.7E-07	2.5E-02	89
3410	Adult–GP High	90	Exposed Veg	6.8E-04	6.4E-07	9.5E-02	23
3340	Adult–GP High	90	Protected Fruit	1.9E-04	1.8E-07	2.6E-02	84
2136	Adult–GP High	90	Exposed Fruit	2.5E-04	1.4E-07	2.1E-02	103
3340	Adult–GP High	90	Root Veg	5.4E-04	4.0E-07	6.0E-02	37
3447	Adult–GP High	90	Total Ingestion	NA	1.6E-06	2.4E-01	9
Arsenic- Noncancer							
5759	Child–GP High	50	Soil Ingestion	7.4E-01	2.6E-06	8.8E-03	250
34	Child–GP High	50	Protected Veg	3.8E-04	6.4E-07	2.1E-03	1025
217	Child–GP High	50	Exposed Veg	8.0E-04	1.7E-06	5.5E-03	397
34	Child–GP High	50	Protected Fruit	2.0E-04	1.0E-06	3.4E-03	641
9691	Child–GP High	50	Exposed Fruit	3.3E-04	2.5E-06	8.2E-03	268
34	Child–GP High	50	Root Veg	5.7E-04	1.6E-06	5.4E-03	410
2614	Child–GP High	50	Total Ingestion	NA	9.9E-06	3.3E-02	67
464	Child–GP High	90	Soil Ingestion	7.3E-01	3.9E-06	1.3E-02	169
9993	Child–GP High	90	Protected Veg	3.9E-04	6.6E-07	2.2E-03	998

Table K-3. Detailed Human Health Results
(Based on General Population High Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
6242	Child–GP High	90	Exposed Veg	8.4E-04	1.7E-06	5.8E-03	378
9993	Child–GP High	90	Protected Fruit	2.1E-04	1.1E-06	3.5E-03	623
6864	Child–GP High	90	Exposed Fruit	3.4E-04	2.9E-06	9.5E-03	231
9993	Child–GP High	90	Root Veg	5.8E-04	1.7E-06	5.5E-03	399
7163	Child–GP High	90	Total Ingestion	NA	1.2E-05	3.9E-02	56
254	Adult–GP High	50	Soil Ingestion	5.9E-01	2.3E-07	7.8E-04	2823
6349	Adult–GP High	50	Protected Veg	3.9E-04	2.9E-07	9.5E-04	2313
4527	Adult–GP High	50	Exposed Veg	8.1E-04	1.1E-06	3.7E-03	598
6349	Adult–GP High	50	Protected Fruit	2.0E-04	3.0E-07	1.0E-03	2189
5117	Adult–GP High	50	Exposed Fruit	3.0E-04	2.5E-07	8.3E-04	2645
6349	Adult–GP High	50	Root Veg	5.7E-04	7.0E-07	2.3E-03	949
6954	Adult–GP High	50	Total Ingestion	NA	2.9E-06	9.6E-03	229
7850	Adult–GP High	90	Soil Ingestion	9.7E-01	4.0E-07	1.3E-03	1644
5770	Adult–GP High	90	Protected Veg	3.9E-04	2.9E-07	9.7E-04	2264
8505	Adult–GP High	90	Exposed Veg	8.4E-04	1.2E-06	3.9E-03	571
5770	Adult–GP High	90	Protected Fruit	2.1E-04	3.1E-07	1.0E-03	2144
4018	Adult–GP High	90	Exposed Fruit	3.5E-04	2.9E-07	9.6E-04	2281
5770	Adult–GP High	90	Root Veg	5.8E-04	7.1E-07	2.4E-03	929
5719	Adult–GP High	90	Total Ingestion	NA	3.1E-06	1.0E-02	210
Cobalt							
1898	Child–GP High	50	Soil Ingestion	5.2E-01	1.2E-02	1.2E-02	181
2501	Child–GP High	50	Protected Veg	1.2E-03	6.9E-03	6.9E-03	318
5502	Child–GP High	50	Exposed Veg	1.6E-03	1.0E-02	1.0E-02	215
8654	Child–GP High	50	Protected Fruit	7.3E-04	1.1E-02	1.1E-02	203
5236	Child–GP High	50	Exposed Fruit	1.0E-03	2.5E-02	2.5E-02	87
994	Child–GP High	50	Root Veg	2.5E-03	2.2E-02	2.2E-02	102
9733	Child–GP High	50	Total Ingestion	NA	8.6E-02	8.6E-02	25
2268	Child–GP High	90	Soil Ingestion	6.5E-01	2.1E-02	2.1E-02	106
9540	Child–GP High	90	Protected Veg	1.4E-03	7.7E-03	7.7E-03	286
7830	Child–GP High	90	Exposed Veg	1.6E-03	1.1E-02	1.1E-02	192
9540	Child–GP High	90	Protected Fruit	7.2E-04	1.2E-02	1.2E-02	179
7152	Child–GP High	90	Exposed Fruit	1.1E-03	3.0E-02	3.0E-02	73
9540	Child–GP High	90	Root Veg	2.5E-03	2.4E-02	2.4E-02	92
4005	Child–GP High	90	Total Ingestion	NA	1.1E-01	1.1E-01	21
5203	Adult–GP High	50	Soil Ingestion	6.0E-01	1.0E-03	1.0E-03	2099
6413	Adult–GP High	50	Protected Veg	1.3E-03	3.2E-03	3.2E-03	682
2740	Adult–GP High	50	Exposed Veg	1.6E-03	7.1E-03	7.1E-03	309
6413	Adult–GP High	50	Protected Fruit	6.9E-04	3.4E-03	3.4E-03	645
2503	Adult–GP High	50	Exposed Fruit	1.0E-03	2.8E-03	2.8E-03	788
6413	Adult–GP High	50	Root Veg	2.4E-03	9.8E-03	9.8E-03	225

Table K-3. Detailed Human Health Results

(Based on General Population High Consumption Rates for Produce)

RunID	Receptor Type	%-tile	Exposure Pathway	Soil/Produce Concentration (mg kg^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (mg kg^{-1} SFS)
509	Adult–GP High	50	Total Ingestion	NA	2.7E-02	2.7E-02	80
5751	Adult–GP High	90	Soil Ingestion	8.1E-01	2.1E-03	2.1E-03	1038
6328	Adult–GP High	90	Protected Veg	1.4E-03	3.4E-03	3.4E-03	649
7792	Adult–GP High	90	Exposed Veg	1.6E-03	7.5E-03	7.5E-03	293
6328	Adult–GP High	90	Protected Fruit	7.2E-04	3.6E-03	3.6E-03	614
9954	Adult–GP High	90	Exposed Fruit	1.1E-03	3.0E-03	3.0E-03	733
6328	Adult–GP High	90	Root Veg	2.5E-03	1.0E-02	1.0E-02	214
9534	Adult–GP High	90	Total Ingestion	NA	3.0E-02	3.0E-02	74
Iron							
1613	Child–GP High	50	Soil Ingestion	5.5E-01	4.3E-06	4.3E-06	507,821
3135	Child–GP High	50	Protected Veg	1.8E-04	4.3E-07	4.3E-07	Capped
6911	Child–GP High	50	Exposed Veg	3.2E-04	9.0E-07	9.0E-07	Capped
2289	Child–GP High	50	Protected Fruit	9.4E-05	6.8E-07	6.8E-07	Capped
5124	Child–GP High	50	Exposed Fruit	1.4E-04	1.7E-06	1.7E-06	Capped
2931	Child–GP High	50	Root Veg	4.6E-04	1.9E-06	1.9E-06	Capped
2508	Child–GP High	50	Total Ingestion	NA	9.7E-06	9.7E-06	225,994
1658	Child–GP High	90	Soil Ingestion	9.0E-01	8.5E-06	8.5E-06	260,230
5217	Child–GP High	90	Protected Veg	1.9E-04	4.6E-07	4.6E-07	Capped
8669	Child–GP High	90	Exposed Veg	3.4E-04	1.0E-06	1.0E-06	Capped
5217	Child–GP High	90	Protected Fruit	1.0E-04	7.4E-07	7.4E-07	Capped
6010	Child–GP High	90	Exposed Fruit	1.8E-04	2.2E-06	2.2E-06	Capped
5217	Child–GP High	90	Root Veg	5.0E-04	2.0E-06	2.0E-06	Capped
7537	Child–GP High	90	Total Ingestion	NA	1.5E-05	1.5E-05	148,480
7194	Adult–GP High	50	Soil Ingestion	3.8E-01	3.3E-07	3.3E-07	Capped
1139	Adult–GP High	50	Protected Veg	1.7E-04	1.8E-07	1.8E-07	Capped
3977	Adult–GP High	50	Exposed Veg	2.9E-04	5.7E-07	5.7E-07	Capped
1139	Adult–GP High	50	Protected Fruit	9.1E-05	1.9E-07	1.9E-07	Capped
2167	Adult–GP High	50	Exposed Fruit	1.4E-04	1.6E-07	1.6E-07	Capped
1139	Adult–GP High	50	Root Veg	4.4E-04	7.8E-07	7.8E-07	Capped
7952	Adult–GP High	50	Total Ingestion	NA	2.2E-06	2.2E-06	991,820
3159	Adult–GP High	90	Soil Ingestion	8.1E-01	8.4E-07	8.4E-07	Capped
6286	Adult–GP High	90	Protected Veg	1.9E-04	2.1E-07	2.1E-07	Capped
9385	Adult–GP High	90	Exposed Veg	3.4E-04	6.7E-07	6.7E-07	Capped
6286	Adult–GP High	90	Protected Fruit	1.0E-04	2.2E-07	2.2E-07	Capped
9461	Adult–GP High	90	Exposed Fruit	1.9E-04	2.3E-07	2.3E-07	Capped
6286	Adult–GP High	90	Root Veg	5.0E-04	8.7E-07	8.7E-07	Capped
4181	Adult–GP High	90	Total Ingestion	NA	3.0E-06	3.0E-06	726,078

Capped = Modeling estimates indicated risks below levels of concern at concentrations above 1,000,000 mg kg^{-1} (i.e., SFS could be comprised entirely of this constituent and still not cause risk).

Table K-4. Detailed Human Health Results: Groundwater Ingestion

RunID	Receptor Type	%-tile	Exposure Pathway	Groundwater Concentration (mg L^{-1})	ADD or LADD (mg kg^{-1} d^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentration (Groundwater Pathway) (mg kg^{-1})
Arsenic - Cancer							
NA	Child	50	Groundwater	Pathway Incomplete	NA	NA	NA
4302	Child	90	Groundwater	3.2E-04		3.7E-02	59
NA	Adult	50	Groundwater	Pathway Incomplete	NA	NA	NA
9716	Adult	90	Groundwater	2.2E-04		3.1E-02	71
Arsenic – Noncancer							
NA	Child	50	Groundwater	Pathway Incomplete	NA	NA	NA
5146	Child	90	Groundwater	1.2E-04		1.3E-02	171
NA	Adult	50	Groundwater	Pathway Incomplete	NA	NA	NA
1578	Adult	90	Groundwater	1.0E-03		6.8E-03	321

Table K-5. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using Home Gardener Consumption Rate Distributions for Produce

RunID	%-tile	Consumption Rate				Units	Soil/Food Conc				Body Weight (kg)				ED (yr) *	EF (d yr^{-1})	AT (yr)	Days $Year^{-1}$	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc*** (mg kg^{-1} dry weight)
		Child 1-5	Child 6-11	Child 2-19	Adult		Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult										
Home Gardener Child – Cancer																								
5114		100	100	NA	NA	(mg d^{-1})	7.0E-01	6.5E-01	NA	NA	18	40	NA	NA	6	350	70	365	2.1E-06	1.7E-07	6.7E-06	2.5E-02	Soil	
		2.87	0.54	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	3.8E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	8.4E-07	6.9E-08	6.7E-06	1.0E-02	Protected Veg	
		1.03	0.12	NA	NA	(g WW kg^{-1} BW d^{-1})	8.1E-04	7.9E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	6.0E-07	4.9E-08	6.7E-06	7.4E-03	Exposed Veg	
		19.48	0.00	NA	NA	(g WW kg^{-1} BW d^{-1})	2.0E-04	2.0E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	2.4E-06	1.9E-07	6.7E-06	2.9E-02	Protected Fruit	
		1.84	0.83	NA	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	2.9E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	3.9E-07	3.2E-08	6.7E-06	4.8E-03	Exposed Fruit	
		0.21	2.86	NA	NA	(g WW kg^{-1} BW d^{-1})	5.7E-04	5.6E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	3.5E-07	2.9E-08	6.7E-06	4.3E-03	Root Veg	
	50	**25.44**																		5.4E-07	6.7E-06	8.1E-02	Total Ingestion	**27**
4734		100	100	100	NA	(mg d^{-1})	8.3E-01	5.5E-01	3.6E-01	NA	14	26	65	NA	13	350	70	365	1.8E-06	3.2E-07	6.7E-06	4.8E-02	Soil	
		0.99	2.60	1.30	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	3.8E-04	3.7E-04	NA	NA	NA	NA	NA	13	350	70	365	5.7E-07	1.0E-07	6.7E-06	1.6E-02	Protected Veg	
		1.44	0.10	2.85	NA	(g WW kg^{-1} BW d^{-1})	8.2E-04	8.0E-04	7.7E-04	NA	NA	NA	NA	NA	13	350	70	365	8.5E-07	1.6E-07	6.7E-06	2.3E-02	Exposed Veg	
		7.83	18.76	0.80	NA	(g WW kg^{-1} BW d^{-1})	2.1E-04	2.0E-04	2.0E-04	NA	NA	NA	NA	NA	13	350	70	365	1.7E-06	3.1E-07	6.7E-06	4.6E-02	Protected Fruit	
		4.64	1.24	8.80	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	2.9E-04	2.8E-04	NA	NA	NA	NA	NA	13	350	70	365	1.0E-06	1.8E-07	6.7E-06	2.7E-02	Exposed Fruit	
		1.01	0.81	0.01	NA	(g WW kg^{-1} BW d^{-1})	5.8E-04	5.6E-04	5.5E-04	NA	NA	NA	NA	NA	13	350	70	365	3.6E-07	6.5E-08	6.7E-06	9.8E-03	Root Veg	
	90	**15.91**	**23.51**	**13.77**																1.1E-06	6.7E-06	1.7E-01	Total Ingestion	**13**
Home Gardener Adult - Cancer																								
4772		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	4.6E-01	NA	NA	NA	96	10	350	70	365	1.5E-07	2.0E-08	6.7E-06	3.0E-03	Soil	
		NA	NA	NA	2.33	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.8E-04	NA	NA	NA	NA	10	350	70	365	7.8E-07	1.1E-07	6.7E-06	1.6E-02	Protected Veg	
		NA	NA	NA	1.00	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.0E-04	NA	NA	NA	NA	10	350	70	365	6.7E-07	9.2E-08	6.7E-06	1.4E-02	Exposed Veg	
		NA	NA	NA	2.94	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.0E-04	NA	NA	NA	NA	10	350	70	365	4.2E-07	5.8E-08	6.7E-06	8.7E-03	Protected Fruit	
		NA	NA	NA	2.37	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.9E-04	NA	NA	NA	NA	10	350	70	365	5.4E-07	7.5E-08	6.7E-06	1.1E-02	Exposed Fruit	
		NA	NA	NA	0.34	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.7E-04	NA	NA	NA	NA	10	350	70	365	1.8E-07	2.5E-08	6.7E-06	3.7E-03	Root Veg	
	50				**8.97**															3.7E-07	6.7E-06	5.6E-02	Total Ingestion	**39**
7831		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	9.0E-01	NA	NA	NA	97	42	350	70	365	2.8E-07	1.6E-07	6.7E-06	2.4E-02	Soil	
		NA	NA	NA	0.39	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	42	350	70	365	1.3E-07	7.7E-08	6.7E-06	1.2E-02	Protected Veg	
		NA	NA	NA	1.83	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.2E-04	NA	NA	NA	NA	42	350	70	365	1.3E-06	7.2E-07	6.7E-06	1.1E-01	Exposed Veg	
		NA	NA	NA	1.52	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	42	350	70	365	2.2E-07	1.3E-07	6.7E-06	1.9E-02	Protected Fruit	
		NA	NA	NA	0.64	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.0E-04	NA	NA	NA	NA	42	350	70	365	1.5E-07	8.7E-08	6.7E-06	1.3E-02	Exposed Fruit	
		NA	NA	NA	2.11	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	42	350	70	365	1.2E-06	6.7E-07	6.7E-06	1.0E-01	Root Veg	
	90				**6.50**															1.8E-06	6.7E-06	2.8E-01	Total Ingestion	**8**

Table K-5. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using Home Gardener Consumption Rate Distributions for Produce

RunID	%-tile	Consumption Rate				Units	Soil/Food Conc				Body Weight (kg)				ED (yr) *	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc*** (mg kg^{-1} dry weight)
		Child 1-5	Child 6-11	Child 2-19	Adult		Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult										
Home Gardener Child – Noncancer																								
6244		100	NA	NA	NA	(mg d^{-1})	8.7E-01	NA	NA	NA	21	NA	NA	NA	4	NA	NA	NA	2.5E-06	NA	3.0E-04	8.2E-03	Soil	
		1.85	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	4	NA	NA	NA	6.3E-07	NA	3.0E-04	2.1E-03	Protected Veg	
		0.77	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.4E-04	NA	NA	NA	NA	NA	NA	NA	4	NA	NA	NA	5.4E-07	NA	3.0E-04	1.8E-03	Exposed Veg	
		17.28	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.1E-04	NA	NA	NA	NA	NA	NA	NA	4	NA	NA	NA	2.5E-06	NA	3.0E-04	8.5E-03	Protected Fruit	
		2.13	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.5E-04	NA	NA	NA	NA	NA	NA	NA	4	NA	NA	NA	5.8E-07	NA	3.0E-04	1.9E-03	Exposed Fruit	
		0.67	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.8E-04	NA	NA	NA	NA	NA	NA	NA	4	NA	NA	NA	3.7E-07	NA	3.0E-04	1.2E-03	Root Veg	
	50	**22.70**																	7.1E-06		3.0E-04	2.4E-02	Total Ingestion	**93**
1587		100	NA	NA	NA	(mg d^{-1})	8.8E-01	NA	NA	NA	16	NA	NA	NA	3	NA	NA	NA	3.4E-06	NA	3.0E-04	1.1E-02	Soil	
		2.26	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	7.7E-07	NA	3.0E-04	2.6E-03	Protected Veg	
		0.04	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.4E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	2.8E-08	NA	3.0E-04	9.2E-05	Exposed Veg	
		10.67	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.1E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	1.6E-06	NA	3.0E-04	5.2E-03	Protected Fruit	
		5.54	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.4E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	1.5E-06	NA	3.0E-04	4.9E-03	Exposed Fruit	
		8.16	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.8E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	4.5E-06	NA	3.0E-04	1.5E-02	Root Veg	
	90	**26.66**																	1.2E-05		3.0E-04	3.9E-02	Total Ingestion	**56**
Home Gardener Adult – Noncancer																								
6559		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	9.0E-01	NA	NA	NA	61	4	NA	NA	NA	4.5E-07	NA	3.0E-04	1.5E-03	Soil	
		NA	NA	NA	2.35	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	4	NA	NA	NA	8.0E-07	NA	3.0E-04	2.7E-03	Protected Veg	
		NA	NA	NA	0.76	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.4E-04	NA	NA	NA	NA	4	NA	NA	NA	5.3E-07	NA	3.0E-04	1.8E-03	Exposed Veg	
		NA	NA	NA	4.15	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	4	NA	NA	NA	6.1E-07	NA	3.0E-04	2.0E-03	Protected Fruit	
		NA	NA	NA	0.53	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.3E-04	NA	NA	NA	NA	4	NA	NA	NA	1.4E-07	NA	3.0E-04	4.7E-04	Exposed Fruit	
		NA	NA	NA	0.86	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	4	NA	NA	NA	4.7E-07	NA	3.0E-04	1.6E-03	Root Veg	
	50				**8.65**														3.0E-06		3.0E-04	1.0E-02	Total Ingestion	**219**
9631		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	6.2E-01	NA	NA	NA	117	11	NA	NA	NA	1.6E-07	NA	3.0E-04	5.3E-04	Soil	
		NA	NA	NA	0.64	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.8E-04	NA	NA	NA	NA	11	NA	NA	NA	2.1E-07	NA	3.0E-04	7.1E-04	Protected Veg	
		NA	NA	NA	7.68	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.1E-04	NA	NA	NA	NA	11	NA	NA	NA	5.2E-06	NA	3.0E-04	1.7E-02	Exposed Veg	
		NA	NA	NA	1.15	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.0E-04	NA	NA	NA	NA	11	NA	NA	NA	1.6E-07	NA	3.0E-04	5.5E-04	Protected Fruit	
		NA	NA	NA	0.41	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.1E-04	NA	NA	NA	NA	11	NA	NA	NA	1.0E-07	NA	3.0E-04	3.4E-04	Exposed Fruit	
		NA	NA	NA	0.70	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.6E-04	NA	NA	NA	NA	11	NA	NA	NA	3.7E-07	NA	3.0E-04	1.2E-03	Root Veg	
	90				**10.58**														6.2E-06		3.0E-04	2.1E-02	Total Ingestion	**106**

* Model rounds exposure duration to whole number.

** Soil ingestion dose includes an adjustment to the arsenic soil concentration to reflect EPA's default relative bioavailability (RBA) value of 60%.

*** Includes conversion from wet to dry weight reflecting average modeled solids content of 90 percent (10 percent moisture).

Table K-6. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using General Population Median Consumption Rates and Assuming 50% of Consumed Produce is Homegrown

RunID	%-tile	Consumption Rate*				Units	Soil/Food Conc				Body Weight (kg)				ED (yr)	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc *** (mg kg^{-1} dry weight)
		Child 1-5	Child 6-11	Child 2-19	Adult		Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult										
Gen. Pop. Child – Cancer																								
5208		100	100	NA	NA	(mg d^{-1})	6.3E-01	NA	NA	NA	16	49	NA	NA	6	350	70	365	2.4E-06	1.8E-07	6.7E-06	2.7E-02	Soil	
		0.63	0.39	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	6	350	70	365	2.1E-07	1.6E-08	6.7E-06	2.4E-03	Protected Veg	
		0.32	0.30	NA	NA	(g WW kg^{-1} BW d^{-1})	8.1E-04	NA	NA	NA	NA	NA	NA	NA	6	350	70	365	2.2E-07	1.6E-08	6.7E-06	2.5E-03	Exposed Veg	
		1.35	0.09	NA	NA	(g WW kg^{-1} BW d^{-1})	2.0E-04	NA	NA	NA	NA	NA	NA	NA	6	350	70	365	2.0E-07	1.5E-08	6.7E-06	2.2E-03	Protected Fruit	
		1.95	1.10	NA	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	NA	NA	NA	NA	NA	NA	NA	6	350	70	365	4.6E-07	3.5E-08	6.7E-06	5.2E-03	Exposed Fruit	
		0.72	0.50	NA	NA	(g WW kg^{-1} BW d^{-1})	5.7E-04	NA	NA	NA	NA	NA	NA	NA	6	350	70	365	3.9E-07	3.0E-08	6.7E-06	4.5E-03	Root Veg	
	50	**4.97**	**2.38**																	2.9E-07	6.7E-06	4.4E-02	Total Ingestion	**50**
2116		100	100	100	50	(mg d^{-1})	6.0E-01	3.1E-01	2.1E-01	1.8E-01	17	26	76	52	22	350	70	365	7.6E-07	2.2E-07	6.7E-06	3.4E-02	Soil	
		0.63	0.39	0.23	0.27	(g WW kg^{-1} BW d^{-1})	4.0E-04	3.7E-04	3.7E-04	3.6E-04	NA	NA	NA	NA	22	350	70	365	1.2E-07	3.7E-08	6.7E-06	5.5E-03	Protected Veg	
		0.32	0.30	0.27	0.45	(g WW kg^{-1} BW d^{-1})	8.1E-04	7.8E-04	7.6E-04	7.5E-04	NA	NA	NA	NA	22	350	70	365	2.1E-07	6.1E-08	6.7E-06	9.2E-03	Exposed Veg	
		1.35	0.09	0.90	0.46	(g WW kg^{-1} BW d^{-1})	2.0E-04	2.0E-04	1.9E-04	1.9E-04	NA	NA	NA	NA	22	350	70	365	1.0E-07	3.0E-08	6.7E-06	4.5E-03	Protected Fruit	
		1.95	1.10	0.44	0.32	(g WW kg^{-1} BW d^{-1})	3.1E-04	2.9E-04	2.8E-04	2.7E-04	NA	NA	NA	NA	22	350	70	365	2.2E-07	6.6E-08	6.7E-06	9.9E-03	Exposed Fruit	
		0.72	0.50	0.41	0.35	(g WW kg^{-1} BW d^{-1})	5.7E-04	5.5E-04	5.4E-04	5.3E-04	NA	NA	NA	NA	22	350	70	365	2.6E-07	7.8E-08	6.7E-06	1.2E-02	Root Veg	
	90	**4.97**	**2.38**	**2.24**	**1.86**															5.0E-07	6.7E-06	7.5E-02	Total Ingestion	**30**
Gen. Pop. Adult – Cancer																								
8883		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	7.2E-01	NA	NA	NA	72	9	350	70	365	3.0E-07	3.6E-08	6.7E-06	5.4E-03	Soil	
		NA	NA	NA	0.27	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	9	350	70	365	9.3E-08	1.1E-08	6.7E-06	1.7E-03	Protected Veg	
		NA	NA	NA	0.45	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.1E-04	NA	NA	NA	NA	9	350	70	365	3.1E-07	3.7E-08	6.7E-06	5.6E-03	Exposed Veg	
		NA	NA	NA	0.46	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.0E-04	NA	NA	NA	NA	9	350	70	365	6.7E-08	8.1E-09	6.7E-06	1.2E-03	Protected Fruit	
		NA	NA	NA	0.32	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.0E-04	NA	NA	NA	NA	9	350	70	365	7.5E-08	9.1E-09	6.7E-06	1.4E-03	Exposed Fruit	
		NA	NA	NA	0.35	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.7E-04	NA	NA	NA	NA	9	350	70	365	1.9E-07	2.3E-08	6.7E-06	3.4E-03	Root Veg	
	50				**1.86**															1.2E-07	6.7E-06	1.9E-02	Total Ingestion	**118**
1770		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	3.3E-01	NA	NA	NA	71	44	350	70	365	1.4E-07	8.5E-08	6.7E-06	1.3E-02	Soil	
		NA	NA	NA	0.27	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.6E-04	NA	NA	NA	NA	44	350	70	365	8.7E-08	5.2E-08	6.7E-06	7.8E-03	Protected Veg	
		NA	NA	NA	0.45	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	7.6E-04	NA	NA	NA	NA	44	350	70	365	2.9E-07	1.7E-07	6.7E-06	2.6E-02	Exposed Veg	
		NA	NA	NA	0.46	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	1.9E-04	NA	NA	NA	NA	44	350	70	365	6.3E-08	3.8E-08	6.7E-06	5.7E-03	Protected Fruit	
		NA	NA	NA	0.32	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.8E-04	NA	NA	NA	NA	44	350	70	365	7.1E-08	4.2E-08	6.7E-06	6.4E-03	Exposed Fruit	
		NA	NA	NA	0.35	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.4E-04	NA	NA	NA	NA	44	350	70	365	1.8E-07	1.1E-07	6.7E-06	1.6E-02	Root Veg	
	90				**1.86**															5.0E-07	6.7E-06	7.4E-02	Total Ingestion	**30**

Table K-6. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using General Population Median Consumption Rates and Assuming 50% of Consumed Produce is Homegrown

		Consumption Rate*					Soil/Food Conc				Body Weight (kg)													
RunID	%-tile	Child 1-5	Child 6-11	Child 2-19	Adult	Units	Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult	ED (yr)	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc *** (mg kg^{-1} dry weight)
Gen. Pop. Child - Noncancer																								
2391		100	NA	NA	NA	(mg d^{-1})	7.0E-01	NA	NA	NA	16	NA	NA	NA	3	NA	NA	NA	2.6E-06	NA	3.0E-04	8.5E-03	Soil	
		0.63	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	2.1E-07	NA	3.0E-04	7.1E-04	Protected Veg	
		0.32	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.1E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	2.2E-07	NA	3.0E-04	7.2E-04	Exposed Veg	
		1.35	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.0E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	1.4E-07	NA	3.0E-04	4.7E-04	Protected Fruit	
		1.95	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	4.6E-07	NA	3.0E-04	1.5E-03	Exposed Fruit	
		0.72	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.7E-04	NA	NA	NA	NA	NA	NA	NA	3	NA	NA	NA	3.9E-07	NA	3.0E-04	1.3E-03	Root Veg	
	50	**4.97**																	4.0E-06		3.0E-04	1.3E-02	Total Ingestion	**166**
2525		100	NA	NA	NA	(mg d^{-1})	9.4E-01	NA	NA	NA	14	NA	NA	NA	1	NA	NA	NA	3.9E-06	NA	3.0E-04	1.3E-02	Soil	
		0.63	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	1	NA	NA	NA	2.2E-07	NA	3.0E-04	7.2E-04	Protected Veg	
		0.32	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.3E-04	NA	NA	NA	NA	NA	NA	NA	1	NA	NA	NA	2.2E-07	NA	3.0E-04	7.4E-04	Exposed Veg	
		1.35	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.1E-04	NA	NA	NA	NA	NA	NA	NA	1	NA	NA	NA	2.0E-07	NA	3.0E-04	6.6E-04	Protected Fruit	
		1.95	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.1E-04	NA	NA	NA	NA	NA	NA	NA	1	NA	NA	NA	4.8E-07	NA	3.0E-04	1.6E-03	Exposed Fruit	
		0.72	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.8E-04	NA	NA	NA	NA	NA	NA	NA	1	NA	NA	NA	4.0E-07	NA	3.0E-04	1.3E-03	Root Veg	
	90	**4.97**																	5.4E-06		3.0E-04	1.8E-02	Total Ingestion	**121**
Gen. Pop. Adult - Noncancer																								
70		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	8.2E-01	NA	NA	NA	109	1	NA	NA	NA	2.2E-07	NA	3.0E-04	7.5E-04	Soil	
		NA	NA	NA	0.27	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	1	NA	NA	NA	9.3E-08	NA	3.0E-04	3.1E-04	Protected Veg	
		NA	NA	NA	0.45	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.2E-04	NA	NA	NA	NA	1	NA	NA	NA	3.1E-07	NA	3.0E-04	1.0E-03	Exposed Veg	
		NA	NA	NA	0.46	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	1	NA	NA	NA	6.8E-08	NA	3.0E-04	2.3E-04	Protected Fruit	
		NA	NA	NA	0.32	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.1E-04	NA	NA	NA	NA	1	NA	NA	NA	7.9E-08	NA	3.0E-04	2.6E-04	Exposed Fruit	
		NA	NA	NA	0.35	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	1	NA	NA	NA	1.9E-07	NA	3.0E-04	6.4E-04	Root Veg	
	50				**1.86**														9.7E-07		3.0E-04	3.2E-03	Total Ingestion	**682**
4444		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	8.5E-01	NA	NA	NA	61	2	NA	NA	NA	4.2E-07	NA	3.0E-04	1.4E-03	Soil	
		NA	NA	NA	0.27	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	2	NA	NA	NA	9.4E-08	NA	3.0E-04	3.1E-04	Protected Veg	
		NA	NA	NA	0.45	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.2E-04	NA	NA	NA	NA	2	NA	NA	NA	3.1E-07	NA	3.0E-04	1.0E-03	Exposed Veg	
		NA	NA	NA	0.46	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	2	NA	NA	NA	6.8E-08	NA	3.0E-04	2.3E-04	Protected Fruit	
		NA	NA	NA	0.32	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.0E-04	NA	NA	NA	NA	2	NA	NA	NA	7.7E-08	NA	3.0E-04	2.6E-04	Exposed Fruit	
		NA	NA	NA	0.35	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	2	NA	NA	NA	1.9E-07	NA	3.0E-04	6.4E-04	Root Veg	
	90				**1.86**														1.2E-06		3.0E-04	3.9E-03	Total Ingestion	**570**

* Produce consumption rates scaled to reflect 50% homegrown produce.

** Soil ingestion dose includes an adjustment to the arsenic soil concentration to reflect EPA's default relative bioavailability (RBA) value of 60%.

*** Includes conversion from wet to dry weight reflecting average modeled solids content of 90 percent (10 percent moisture).

Table K-7. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using General Population High Consumption Rates and Assuming 50% of Consumed Produce is Homegrown

RunID	%-tile	Consumption Rate*				Units	Soil/Food Conc				Body Weight (kg)				ED (yr)	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc*** (mg kg^{-1} dry weight)
		Child 1-5	Child 6-11	Child 2-19	Adult		Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult										
Gen. Pop. Child - Cancer																								
2701		100	100	NA	NA	(mg d^{-1})	6.4E-01	4.1E-01	NA	NA	16	31	NA	NA	6	350	70	365	2.0E-06	1.7E-07	6.7E-06	2.6E-02	Soil	
		1.93	1.30	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	3.8E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	5.8E-07	5.0E-08	6.7E-06	7.5E-03	Protected Veg	
		2.48	1.70	NA	NA	(g WW kg^{-1} BW d^{-1})	8.0E-04	7.8E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	1.5E-06	1.3E-07	6.7E-06	1.9E-02	Exposed Veg	
		7.19	4.05	NA	NA	(g WW kg^{-1} BW d^{-1})	2.0E-04	2.0E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	9.2E-07	7.9E-08	6.7E-06	1.2E-02	Protected Fruit	
		10.62	3.15	NA	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	2.9E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	2.1E-06	1.8E-07	6.7E-06	2.7E-02	Exposed Fruit	
		3.01	2.10	NA	NA	(g WW kg^{-1} BW d^{-1})	5.7E-04	5.4E-04	NA	NA	NA	NA	NA	NA	6	350	70	365	1.5E-06	1.3E-07	6.7E-06	1.9E-02	Root Veg	
	50	**25.23**																		7.4E-07	6.7E-06	1.1E-01	Total Ingestion	**20**
1692		100	100	100	NA	(mg d^{-1})	3.8E-01	1.6E-01	1.0E-01	NA	15	29	60	NA	19	350	70	365	5.6E-07	1.4E-07	6.7E-06	2.1E-02	Soil	
		1.93	1.30	0.75	NA	(g WW kg^{-1} BW d^{-1})	3.8E-04	3.6E-04	3.5E-04	NA	NA	NA	NA	NA	19	350	70	365	4.0E-07	1.0E-07	6.7E-06	1.5E-02	Protected Veg	
		2.48	1.70	1.25	NA	(g WW kg^{-1} BW d^{-1})	7.9E-04	7.5E-04	7.2E-04	NA	NA	NA	NA	NA	19	350	70	365	1.1E-06	2.8E-07	6.7E-06	4.3E-02	Exposed Veg	
		7.19	4.05	2.70	NA	(g WW kg^{-1} BW d^{-1})	3.8E-04	1.9E-04	1.8E-04	NA	NA	NA	NA	NA	19	350	70	365	8.4E-07	2.2E-07	6.7E-06	3.2E-02	Protected Fruit	
		10.62	3.15	1.45	NA	(g WW kg^{-1} BW d^{-1})	2.9E-04	2.7E-04	2.6E-04	NA	NA	NA	NA	NA	19	350	70	365	1.0E-06	2.5E-07	6.7E-06	3.8E-02	Exposed Fruit	
		3.01	2.10	1.50	NA	(g WW kg^{-1} BW d^{-1})	5.6E-04	5.3E-04	5.1E-04	NA	NA	NA	NA	NA	19	350	70	365	1.1E-06	2.8E-07	6.7E-06	4.1E-02	Root Veg	
	90	**25.23**	**12.30**	**7.65**																1.3E-06	6.7E-06	1.9E-01	Total Ingestion	**12**
Gen. Pop. Adult - Cancer																								
7041		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	7.8E-01	NA	NA	NA	60	9	350	70	365	3.9E-07	4.8E-08	6.7E-06	7.3E-03	Soil	
		NA	NA	NA	0.85	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	9	350	70	365	2.9E-07	3.6E-08	6.7E-06	5.3E-03	Protected Veg	
		NA	NA	NA	1.63	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.3E-04	NA	NA	NA	NA	9	350	70	365	1.1E-06	1.4E-07	6.7E-06	2.1E-02	Exposed Veg	
		NA	NA	NA	2.09	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	9	350	70	365	3.1E-07	3.8E-08	6.7E-06	5.6E-03	Protected Fruit	
		NA	NA	NA	1.06	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.2E-04	NA	NA	NA	NA	9	350	70	365	2.7E-07	3.3E-08	6.7E-06	4.9E-03	Exposed Fruit	
		NA	NA	NA	1.29	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	9	350	70	365	7.0E-07	8.7E-08	6.7E-06	1.3E-02	Root Veg	
	50				**6.92**															3.8E-07	6.7E-06	5.7E-02	Total Ingestion	**39**
3447		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	2.2E-01	NA	NA	NA	64	46	350	70	365	1.0E-07	6.6E-08	6.7E-06	9.9E-03	Soil	
		NA	NA	NA	0.85	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.6E-04	NA	NA	NA	NA	46	350	70	365	2.7E-07	1.7E-07	6.7E-06	2.5E-02	Protected Veg	
		NA	NA	NA	1.63	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	7.5E-04	NA	NA	NA	NA	46	350	70	365	1.0E-06	6.5E-07	6.7E-06	9.8E-02	Exposed Veg	
		NA	NA	NA	2.09	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	1.9E-04	NA	NA	NA	NA	46	350	70	365	2.8E-07	1.8E-07	6.7E-06	2.7E-02	Protected Fruit	
		NA	NA	NA	1.06	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.7E-04	NA	NA	NA	NA	46	350	70	365	2.3E-07	1.5E-07	6.7E-06	2.2E-02	Exposed Fruit	
		NA	NA	NA	1.29	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.3E-04	NA	NA	NA	NA	46	350	70	365	6.5E-07	4.1E-07	6.7E-06	6.2E-02	Root Veg	
	90				**6.92**															1.6E-06	6.7E-06	2.4E-01	Total Ingestion	**9**

Table K-7. Example Calculations for Home Gardener Soil Pathway: Total Arsenic Ingestion Using General Population High Consumption Rates and Assuming 50% of Consumed Produce is Homegrown

		Consumption Rate*					Soil/Food Conc				Body Weight (kg)													
RunID	%-tile	Child 1-5	Child 6-11	Child 2-19	Adult	Units	Child 1-5	Child 6-11	Child 12-19	Adult	Child 1-5	Child 6-11	Child 2-19	Adult	ED (yr)	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD** (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc*** (mg kg^{-1} dry weight)
Gen. Pop. Child - Noncancer																								
2614		100	NA	NA	NA	(mg d^{-1})	6.4E-01	NA	NA	NA	16	NA	NA	NA	5	NA	NA	NA	2.4E-06	NA	3.0E-04	7.9E-03	Soil	
		1.93	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	5	NA	NA	NA	6.5E-07	NA	3.0E-04	2.2E-03	Protected Veg	
		2.48	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.1E-04	NA	NA	NA	NA	NA	NA	NA	5	NA	NA	NA	1.7E-06	NA	3.0E-04	5.6E-03	Exposed Veg	
		7.19	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.0E-04	NA	NA	NA	NA	NA	NA	NA	5	NA	NA	NA	1.0E-06	NA	3.0E-04	3.5E-03	Protected Fruit	
		10.62	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.0E-04	NA	NA	NA	NA	NA	NA	NA	5	NA	NA	NA	2.5E-06	NA	3.0E-04	8.3E-03	Exposed Fruit	
		3.01	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.7E-04	NA	NA	NA	NA	NA	NA	NA	5	NA	NA	NA	1.6E-06	NA	3.0E-04	5.4E-03	Root Veg	
	50	**25.23**																	9.9E-06		3.0E-04	3.3E-02	Total Ingestion	**67**
7163		100	NA	NA	NA	(mg d^{-1})	9.1E-01	NA	NA	NA	14	NA	NA	NA	2	NA	NA	NA	3.8E-06	NA	3.0E-04	1.3E-02	Soil	
		1.93	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.9E-04	NA	NA	NA	NA	NA	NA	NA	2	NA	NA	NA	6.6E-07	NA	3.0E-04	2.2E-03	Protected Veg	
		2.48	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	8.4E-04	NA	NA	NA	NA	NA	NA	NA	2	NA	NA	NA	1.8E-06	NA	3.0E-04	5.8E-03	Exposed Veg	
		7.19	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	2.1E-04	NA	NA	NA	NA	NA	NA	NA	2	NA	NA	NA	1.1E-06	NA	3.0E-04	3.5E-03	Protected Fruit	
		10.62	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	3.4E-04	NA	NA	NA	NA	NA	NA	NA	2	NA	NA	NA	2.9E-06	NA	3.0E-04	9.5E-03	Exposed Fruit	
		3.01	NA	NA	NA	(g WW kg^{-1} BW d^{-1})	5.8E-04	NA	NA	NA	NA	NA	NA	NA	2	NA	NA	NA	1.7E-06	NA	3.0E-04	5.5E-03	Root Veg	
	90	**25.23**																	1.2E-05		3.0E-04	3.9E-02	Total Ingestion	**56**
Gen. Pop. Adult - Noncancer																								
6954		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	5.2E-01	NA	NA	NA	55	25	NA	NA	NA	2.8E-07	NA	3.0E-04	9.4E-04	Soil	
		NA	NA	NA	0.85	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.8E-04	NA	NA	NA	NA	25	NA	NA	NA	2.8E-07	NA	3.0E-04	9.4E-04	Protected Veg	
		NA	NA	NA	1.63	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	7.9E-04	NA	NA	NA	NA	25	NA	NA	NA	1.1E-06	NA	3.0E-04	3.6E-03	Exposed Veg	
		NA	NA	NA	2.09	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.0E-04	NA	NA	NA	NA	25	NA	NA	NA	3.0E-07	NA	3.0E-04	9.9E-04	Protected Fruit	
		NA	NA	NA	1.06	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.9E-04	NA	NA	NA	NA	25	NA	NA	NA	2.5E-07	NA	3.0E-04	8.2E-04	Exposed Fruit	
		NA	NA	NA	1.29	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.6E-04	NA	NA	NA	NA	25	NA	NA	NA	6.9E-07	NA	3.0E-04	2.3E-03	Root Veg	
	50				**6.92**														2.9E-06		3.0E-04	9.6E-03	Total Ingestion	**229**
5719		NA	NA	NA	50	(mg d^{-1})	NA	NA	NA	9.6E-01	NA	NA	NA	76	1	NA	NA	NA	3.8E-07	NA	3.0E-04	1.3E-03	Soil	
		NA	NA	NA	0.85	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.9E-04	NA	NA	NA	NA	1	NA	NA	NA	2.9E-07	NA	3.0E-04	9.7E-04	Protected Veg	
		NA	NA	NA	1.63	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	8.5E-04	NA	NA	NA	NA	1	NA	NA	NA	1.2E-06	NA	3.0E-04	3.9E-03	Exposed Veg	
		NA	NA	NA	2.09	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	2.1E-04	NA	NA	NA	NA	1	NA	NA	NA	3.1E-07	NA	3.0E-04	1.0E-03	Protected Fruit	
		NA	NA	NA	1.06	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	3.5E-04	NA	NA	NA	NA	1	NA	NA	NA	2.9E-07	NA	3.0E-04	9.7E-04	Exposed Fruit	
		NA	NA	NA	1.29	(g WW kg^{-1} BW d^{-1})	NA	NA	NA	5.8E-04	NA	NA	NA	NA	1	NA	NA	NA	7.1E-07	NA	3.0E-04	2.4E-03	Root Veg	
	90				**6.92**														3.1E-06		3.0E-04	1.0E-02	Total Ingestion	**210**

* Produce consumption rates scaled to reflect 50% homegrown produce.

** Soil ingestion dose includes an adjustment to the arsenic soil concentration to reflect EPA's default relative bioavailability (RBA) value of 60%.

*** Includes conversion from wet to dry weight reflecting average modeled solids content of 90 percent (10 percent moisture).

Table K-8. Example Calculations for Home Gardener Groundwater Pathway for Arsenic

RunID	%-tile	Consumption Rate (mL kg^{-1} BW d^{-1})				Groundwater Conc (mg L^{-1})				ED (yr)	EF (d yr^{-1})	AT (yr)	Days Year^{-1}	ADD (mg kg^{-1} BW d^{-1})	LADD (mg kg^{-1} BW d^{-1})	Benchmark (RfD or 1E-5 CSF^{-1})	Unitized Dose Ratio (unitless)	Pathway	Allowable SFS Conc* (mg kg^{-1} dry weight)
		Child 1-5	Child 6-11	Child 2-19	Adult	Child 1-5	Child 6-11	Child 12-19	Adult										
Home Gardener Child – Cancer																			
4302	90	8.7128	12.233	NA	NA	3.2E-04	3.2E-04	NA	NA	6	350	70	365	2.8E-06	2.5E-07	6.7E-06	3.7E-02	Groundwater	59
Home Gardener Adult - Cancer																			
9716	90	NA	NA	NA	6.4184	NA	NA	NA	2.2E-04	11	350	70	365	1.4E-06	2.1E-07	6.7E-06	3.1E-02	Groundwater	71
Home Gardener Child - Noncancer																			
5146	90	33.096	9.6343	NA	NA	1.2E-04	NA	NA	NA	5	NA	NA	NA	3.9E-06	NA	3.0E-04	1.3E-02	Groundwater	171
Home Gardener Adult - Noncancer																			
1578	90	NA	NA	NA	1.9839	NA	NA	NA	0.001	13	NA	NA	NA	2.1E-06	NA	3.0E-04	6.8E-03	Groundwater	321

* Includes conversion from wet to dry weight reflecting average modeled solids content of 90 percent (10 percent moisture).

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Appendix L:
Detailed Ecological Results

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Table L-1. Detailed Ecological Results

Receptor Name	Percentile	Soil (mg kg^{-1})	Unitized Dose Ratio (unitless)	Foundry Sand-Specific Screening Concentrations (mg kg^{-1}SFS)
Antimony				
Soil Biota	90	9.6E-01	1.2E-02	179
Soil Biota	50	8.2E-01	1.0E-02	210
Mammals	90	1.4E-01	5.3E-01	4.1
Mammals	50	1.2E-01	4.5E-01	4.8
Chromium (III)				
Mammals	90	1.5E-01	4.3E-03	511
Mammals	50	1.4E-01	4.1E-03	532
Copper				
Plants	90	9.7E-01	1.4E-02	159
Plants	50	9.0E-01	1.3E-02	172
Soil Biota	90	9.7E-01	1.2E-02	181
Soil Biota	50	9.0E-01	1.1E-02	196
Mammals	90	1.5E-01	3.0E-03	741
Mammals	50	1.3E-01	2.7E-03	801
Manganese				
Plant	90	9.7E-02	4.4E-04	4970
Plants	50	9.3E-02	4.2E-04	5212
Soil Biota	90	9.7E-01	2.2E-03	1017
Soil Biota	50	9.3E-01	2.1E-03	1066
Mammals	90	9.7E-01	2.4E-04	9036
Mammals	50	9.3E-01	2.3E-04	9477
Nickel				
Plants	90	6.8E-02	1.8E-03	1230
Plants	50	6.4E-02	1.7E-03	1300
Soil Biota	90	9.7E-01	3.5E-02	634
Soil Biota	50	9.2E-01	3.3E-02	671
Mammals	90	9.7E-01	7.5E-03	294
Mammals	50	9.2E-01	7.1E-03	311

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