

**Assessment of potential  
health risks from exposures  
to arsenic complex  
associated with CCA-treated utility poles**

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## Overview

Gradient Corporation (Gradient) prepared a human health risk assessment (HHRA) for Arch Wood Protection, Inc., (Arch) to quantify potential health risks from exposures to arsenic complex<sup>1</sup> associated with extant wooden utility poles treated with chromated copper arsenate (CCA).

Based on consideration of populations with potential exposures to CCA-treated utility poles, including child and adult residents, teenage trespassers, and adult utility pole workers, Gradient identified a child resident ages 2-6 years old and an adult utility pole worker ages 18-30 years old as the receptors with the greatest potential exposure and risk. The exposure routes considered for these receptors include incidental ingestion, dermal contact, and inhalation of arsenic complex. Gradient concluded that incidental ingestion and dermal contact are the relevant and potentially most significant exposure routes.

Reasonable maximum exposure (RME) assumptions and parameters, which likely overestimate rather than underestimate potential risks, were used to quantify exposures to in-service CCA-treated utility poles and CCA-impacted soil adjacent to utility poles. The results of this risk assessment indicate that potential health risks to a child resident or an adult utility pole worker would not exceed the U.S. Environmental Protection Agency's (USEPA's) target risk levels. A summary of the potential cancer and non-cancer health risks is provided below in Tables 1 and 2, respectively.

## Introduction

This HHRA was conducted in accordance with current USEPA risk assessment guidance (USEPA, 1989; 1993a; 2001a; 2002) and recent scientific literature. Also considered during preparation of this report were prior risk assessments conducted for CCA-treated wood, including a comprehensive risk assessment prepared by Gradient in October of 2001 (Gradient, 2001<sup>2</sup>). Cancer and non-cancer health risks were assessed separately for a residential and a worker exposure scenario. The residential scenario included a male/female child ages 2-6 years old who was assumed to play outdoors near a CCA-treated utility pole located on the residential property. The worker scenario includes a male/female utility pole worker ages 18-30 years old who was assumed to perform maintenance and repair services that requires climbing a utility pole. Receptors in both scenarios were evaluated for incidental ingestion and dermal contact with arsenic complex in dislodgeable residue (DR), which is residue on the surface of CCA-treated wood that can be removed by dermal contact with the hands. These receptors were also evaluated for incidental ingestion and dermal contact with soils adjacent to a utility pole that have been impacted with arsenic complex. It was assumed that most of the arsenic complex in these soils is the result of residue that has migrated (*via* rainwater run-off) from the pole to adjacent soils.

**Table 1**  
**Summary of Potential Cancer Risks**

Exposure Medium	Exposure Route	Receptor	
		Child Resident	Adult Worker
Arsenic Complex in Dislodgeable Residue Arsenic Complex in Soil	Incidental Ingestion and Dermal Contact	$9.2 \times 10^{-7}$	$1.2 \times 10^{-6}$
	Incidental Ingestion and Dermal Contact	$1.5 \times 10^{-5}$	$4.5 \times 10^{-6}$
	Estimated Cumulative Risk: USEPA Cancer Target Risk Range:	$1.6 \times 10^{-5}$ $1.0 \times 10^{-6}$ to $1.0 \times 10^{-4}$	$5.7 \times 10^{-6}$

**Table 2**  
**Summary of Potential Non-Cancer Risks**

Exposure Medium	Exposure Route	Receptor	
		Child Resident	Adult Worker
Arsenic Complex in Dislodgeable Residue Arsenic Complex in Soil	Incidental Ingestion and Dermal Contact	$5.7 \times 10^{-4}$	$1.6 \times 10^{-2}$
	Incidental Ingestion and Dermal Contact	$9.0 \times 10^{-3}$	$5.7 \times 10^{-2}$
	Estimated Cumulative Risk: USEPA Non-Cancer Target Risk Level:	$9.6 \times 10^{-3}$	$7.3 \times 10^{-2}$ 1.0

## Exposure Assessment

An exposure assessment is used to identify constituents of concern (COCs), and to quantify human exposure to COCs. Arsenic complex from CCA-treated utility poles is the only COC evaluated in this HHRA. Based on a review of the current peer-reviewed literature, the most likely chemical species of arsenic complex in DR is chromium (III) arsenate (Bull, 2000; 2001). The work by Bull (2000; 2001) is supported by recent high energy x-ray absorption spectroscopy (XAS) analysis of DR collected from the surface of CCA-treated deck boards. The XAS analysis demonstrates that arsenic complex in DR is in the pentavalent (As V) oxidation state and that chromium is in the trivalent state (Cr III) (Lytle, 2003). The XAS results are consistent with the chromium (III) arsenate species described by Bull (2000; 2001). The chemical compound, chromium (III) arsenate, is referred to as arsenic complex in this report. Because most of the arsenic in soils located below and adjacent to a CCA-treated utility pole is likely the result of residue that has migrated from the structure to the soil (arsenic is naturally occurring in soil), the form of arsenic in soil is also likely to be an arsenate.

Because most of the potential subchronic and chronic health risks associated with treated wood are from exposures to arsenic complex, neither of the two other metals in CCA-treated wood (*i.e.*, chromium (III)<sup>3</sup> and copper) were evaluated in this HHRA. Based on the toxicity of chromium (III) and copper, both of which are non-carcinogens according to the USEPA (2004a), their inclusion would not significantly affect estimated non-cancer risks. The comprehensive risk assessment prepared by Gradient (2001) provides more information regarding the chemical species of chromium in CCA-treated wood and why this metal is not considered further in the assessment of potential health risks associated with treated wood.

In addition to identifying COCs, an exposure assessment contains a description of the exposure scenarios, potential receptors, exposure media, and exposure pathways evaluated. Also described in this section is the calculation of exposure point concentrations (EPCs) and chemical intake for each complete exposure pathway, including the exposure factors used.

## Exposure Scenarios

As noted earlier, a residential and a worker exposure scenario were evaluated for exposure *via* incidental ingestion and dermal contact with DR and impacted soil.

It was assumed in the residential scenario that a CCA-treated utility pole is the source of DR, and nearly all of the arsenic complex in soil (arsenic is naturally occurring in soil). A male/female child, ages 2-6 years old, was used in the residential scenario to quantify subchronic exposures

for a period of 5 years. A child was used in the residential scenario because of the potential for greater exposure as a result of increased hand-to-mouth behavior in children under 6 years of age, and increased dose because of a child's low bodyweight. The combination of increased exposure and dose can result in potentially greater non-cancer risks for a child compared to an adult. Children under the age of 2 years were not considered to be independent or mobile enough to be playing around utility poles, which are typically located near streets or at the outer edge of a residential property.

A male/female adult utility pole worker, ages 18-30 years old, was used to quantify chronic exposures for a period of 12 years. An exposure duration of 12 years is based on feed-back from a number of utility companies in the U.S., which indicated that climbing utility poles is typically done by younger workers, often in apprentice programs, and that after 10 years or so, these workers often move into more supervisory roles where pole climbing is much less likely to occur. The utility pole worker was assumed to perform maintenance and repair services (*e.g.*, stringing wire, hanging hardware, *etc.*) that require climbing the pole. These types of activities were assumed to result in maximum exposure to utility poles.

This evaluation is considered a screening-level risk assessment, which means that conservative (*i.e.*, health protective) assumptions and parameters were used to estimate exposure and risk. For example, 20% of the time a child resident is outside playing in the yard on a daily basis, he/she was assumed to be exposed to soil and DR, simultaneously. According to USEPA guidance, if the results of a screening-level assessment indicate no significant risk, then no further assessment is warranted (USEPA, 1996). Consistent with this type of an assessment, reasonable maximum exposure (RME) assumptions and parameters were used, when appropriate, to quantify exposures. RME assumptions and parameters are used to estimate health risks associated with high-end exposures.

## Calculation of Exposure Point Concentrations

In a risk assessment, an exposure point concentration (EPC) is a conservative estimate of the average concentration of a constituent (*e.g.*, a chemical) to which an individual could be exposed. Calculation of the EPCs used for arsenic complex in DR and in soil are described below.

## Arsenic Complex in DR

As mentioned previously, DR refers to residue that can be removed from the surface of CCA-treated wood by dermal contact, usually with the hands. We reviewed twelve studies that evaluated the amount of arsenic complex removed using wipes and/or the hands of human volunteers to calculate an EPC for arsenic complex in DR. Removal

of arsenic complex from the surface of CCA-treated wood with hands is referred to as arsenic complex “hand loading”. Four of the twelve studies included arsenic complex hand loading data. The remaining studies provided only wipe sampling data. Relative to wipe data, hand loading data provide a more accurate estimate of the amount of dislodgeable arsenic complex that is typically removed via dermal contact during normal human exposure to a CCA-treated structure. The basis for this observation is discussed below.

Most wipe sampling methods remove dislodgeable material more efficiently than hand sampling methods, and therefore, may not provide an accurate measure of exposure, especially in the absence of any adjustment. Two studies conducted by SCS (1998, 2001) using CCA-treated wood showed that a wipe (*i.e.*, Kimwipes™) sampling method removed approximately 13- to 20-fold more dislodgeable arsenic complex than when the same sized surface area of treated wood was sampled using hands. The wipe sampling procedure in a large study conducted by Research Triangle Institute International (RTI, 2003) removed approximately 13-fold more arsenic complex than hands, when the data were normalized to the surface area of wood sampled. The results of these studies are similar to those observed in a study conducted by the U.S. Consumer Product Safety Commission (CPSC, 2003). The CPSC estimated that wipes remove approximately 5-fold more dislodgeable arsenic from CCA-treated wood than hands (CPSC, 2003).

Based on our review of the four available hand loading studies, the RTI (2003) study was selected to calculate an EPC for arsenic complex in DR because this study was the best documented (a sample tracking system and database were developed for the study), and it provided the most representative (*e.g.*, 25 in-service CCA-treated residential decks from different geographic locations in the U.S.) and comprehensive data set (*e.g.*, 20 or more hand and 20 or more wipe samples were collected per structure).

A brief description of the RTI (2003) study and some of the results are presented below. The study investigators plan to submit a manuscript for publication that will contain a full description of the study and its results.

RTI conducted a hand and wipe study of in-service Southern Yellow Pine (SYP) deck boards treated to a CCA retention of approximately 0.4 pounds per cubic foot (pcf) and ranging in age from 0.6 to 23 years (mean = 6.6 years). These boards were collected under RTI supervision from residences in the vicinity of Pittsburgh, Pennsylvania; Atlanta, Georgia; and Gainesville, Florida, and were shipped to RTI’s facility in Research Triangle Park, North Carolina for testing. A total of 702 coupons (*i.e.*, approximately 3-foot long sections of board) were

sampled using a weighted block wipe method based on a methodology developed in the CPSC (2003) study, and using the bare hands of human volunteers (each end of the same coupon was sampled using either a block wipe or a hand). Untreated and recently treated wood types<sup>4</sup> were also evaluated in the study; however, only aged (*i.e.*, weathered) deck wood was used to assess exposure in this risk assessment because this wood type most accurately represents what most people are exposed to.

Hand sizes (left and right) for 29 adult male and female volunteers within 18-65 years of age that provided informed consent were determined by tracing the outline of each hand. With a 1.1 kg weight<sup>5</sup> attached to the back of the hand, each volunteer rubbed one end of a coupon 10 times (for a total of 20 passes up and down) with his/her hand, which was moistened with a 0.9% saline solution. Each hand was then rinsed/washed 4 times with deionized water and then wiped with a saline-moistened polyester wipe to recover as much of the dislodged residue as possible. The hand rinsate and hand wipe were then analyzed for total chromium, copper, and arsenic.

The mass of arsenic complex in the hand rinsate and in the digested wipe for each hand was totaled and converted to hand loading (in mg/cm<sup>2</sup>) of arsenic complex by dividing the total amount of arsenic complex by the measured hand size for each volunteer. These hand load data were analyzed statistically in accordance with current USEPA risk assessment guidance (USEPA, 2002). This analysis indicated that these data were best fit by a lognormal distribution with a 95% upper confidence limit on the mean (UCLM) of 0.076 mg/cm<sup>2</sup>. A summary of the sta-

Hand Arsenic Complex Loading (µg/cm <sup>2</sup> ) <sup>1</sup>					
Sample Size <sup>2</sup>	Minimum	Maximum	Mean	Median	95% UCLM <sup>3</sup>
699	0.0039	2.19	0.069	0.037	0.076

Notes:  
<sup>1</sup>Based on non-transferable data.  
<sup>2</sup>Three coupons were not included in the analysis because some results were missing.  
<sup>3</sup>Based on 95% Chebyshev upper confidence limit (USEPA, 2002).

tistical analysis of the RTI (2003) data is provided above in Table 3.

The results from the RTI study are comparable to the results from several other treated wood exposure studies, including the two SCS (1998, 2001) studies discussed above, which calculated mean arsenic complex hand loading values of 0.039 and 0.047 mg/cm<sup>2</sup>, respectively, and the CPSC (2003) study with a mean value of 0.055 mg/cm<sup>2</sup> (based on a reported adult mean hand size of 141 cm<sup>2</sup>).

### **EPC Calculation for Arsenic Complex in DR**

The 95% UCLM from the RTI (2003) study was used to calculate an EPC for arsenic complex in DR from CCA-treated utility poles. Most of the treated deck boards in the RTI (2003) study were SYP, which is considered the most representative species of wood used for CCA-treated utility poles. Because the measured retention of most of the deck boards used in the RTI (2003) study was 0.4 pcf, the EPC from this study was adjusted upward to represent the higher retention recommended for utility poles by the American Wood-Preservers' Association (AWPA, 2002).

The minimum AWPA (2002) standard for CCA-treated poles is 0.6 pcf; however, in practice, the approximate outer half inch of a pole is typically treated to a retention above this minimum. Based on information from Arch, a retention of 0.8 pcf was used to represent the retention for poles in this outer "zone." Thus, the EPC from the RTI (2003) study was adjusted upward by a factor of 2.0 to represent the higher CCA retention used for poles, according to the calculation described below:

$$0.08 \text{ mg/cm}^2 (0.8 \text{ pcf}/0.4 \text{ pcf}) = 0.16 \text{ mg/cm}^2$$

Thus, the adjusted EPC used for utility poles is 0.16 mg/cm<sup>2</sup>

### **Arsenic Complex in Soil**

As noted earlier, this refers to arsenic complex in residue that has been mobilized from a CCA-treated utility pole to adjacent soils via rainwater. Gradient used a study conducted by Cooper and Ung (1997) to assess the concentrations of soil arsenic complex from CCA-treated utility poles.

During the summers of 1992-1994, Cooper and Ung identified 39 Red Pine and 14 Jack Pine utility poles of varying age, and located in a variety of soil conditions, in Canada (Red Pine and Jack Pine refers to the species of wood used for the utility pole). CCA retention in the outer 5 millimeters was measured for each pole and ranged from 0.6 to 2.1 pcf for the Red Pine poles, and 0.4 to 1.3 pcf for the Jack Pine poles. The authors noted that some of the measured retentions were higher than what was required to protect the poles from decay. As indicated earlier, the AWPA's recommended retention for utility poles is 0.6 pcf. Therefore, the higher retentions measured in some of the poles included in this study may result in higher concentrations of arsenic complex in adjacent soils due to leaching, than in soils adjacent to CCA-treated utility poles in the U.S.

Soil samples were collected at each pole using a soil auger at depths ranging from 0 to 2.0 meters (if possible), and at distances ranging from 0 to 25 meters. Soil samples were analyzed for total chromium, copper, and arsenic. The results were presented graphically as average

concentrations of CCA metals for poles in different soil conditions (e.g., clay, sandy, high organic content, wet), with two types of pressure-applied treatments (i.e., CCA or CCA with polyethylene glycol or PEG), and different ages (range 1-13 years). Results were also summarized based on soil depth and distance from the pole, and for each species of wood.

The authors noted that the highest concentrations of metals were reported for surface soil samples collected immediately adjacent to the pole, confirming that rainwater run-off is a significant source of the elevated concentrations of metals in these soils. The authors also noted that the concentration of the CCA metals in soil increased with the age of the pole and that in most cases background concentrations of these metals were observed within approximately one foot of the pole.

### **EPC Calculation for Arsenic Complex in Soil**

Gradient conservatively used the surface soil samples collected immediately adjacent to the poles in the Cooper and Ung (1997) study because these samples had the highest reported concentrations of CCA constituents, and therefore, are unlikely to underestimate exposure (and risk). The concentrations of arsenic complex in these soil samples (estimated from the graphs) were pooled for the different soil conditions, types of pressure-applied treatments, and age of the poles. The concentration of soil arsenic complex in this dataset ranged from 25 to 550 mg/kg with a mean of 138 mg/kg and a 95% UCLM (based on a lognormal distribution) of 214 mg/kg. The 95% UCLM was used as the EPC for soil arsenic complex. Attachment 1 contains a summary of the statistical analysis of the data used for soil arsenic complex.

## **Quantification of Exposure**

This section describes the exposure assumptions and factors used to calculate potential human intake levels of arsenic complex from incidental ingestion and dermal contact with CCA-treated utility poles and CCA-impacted soil (refer to Attachment 2 for a summary of the key exposure assumptions and factors). Intake estimates represent the daily dose of a chemical taken into the body, averaged over an appropriate exposure period, and expressed in the units of milligram (mg) of chemical per kilogram (kg) of human body weight a day (mg/kg-day). The exposure equations used in the HHRA are based on the USEPA's "Risk Assessment Guidance for Superfund Volume 1 Human Health Evaluation Manual (Part A)" (USEPA, 1989). The ingestion and dermal exposure equations for dislodgeable arsenic complex are modified slightly to calculate the daily dose of arsenic complex for these unique exposure pathways. Refer to Attachments 3-6 for a description of the exposure equations used.

## Incidental Ingestion of Arsenic Complex in DR

*EPC for Arsenic Complex in DR.* As discussed previously, the EPC for this pathway is the 95% UCLM of hand arsenic complex loadings ( $\text{mg}/\text{cm}^2$ ) for CCA-treated residential decks from the RTI (2003) study, adjusted upward by a factor of 2.0 to reflect the higher retention of utility poles.

*Hand Transfer Efficiency (HTE).* The results of the RTI (2003) study were used to calculate the loading or mass (in mg) of arsenic complex in DR per unit of skin surface area on the hands (*i.e.*,  $\text{mg}/\text{cm}^2$ ). To estimate the amount of DR on the hands that might be ingested *via* hand-to-mouth contact, a hand transfer efficiency (HTE) factor was calculated to estimate the proportion of DR on the hands that might be subsequently ingested. The basis for the HTE factor is briefly described below, the data and calculations used to derive the HTE factor are discussed in more detail in Gradient (2001).

To develop the HTE factor, data regarding children's incidental ingestion of soil, adherence of soil to the hands, and the skin surface area of the hands were reviewed. The HTE factor was calculated based on data regarding lead loading onto hands, which were collected as part of a community study of children's exposures in the vicinity of a smelter (Roels *et al.*, 1980). In this study, researchers measured the mass of lead adhering to children's hands by rinsing the front surface of the children's hands and analyzing the rinsate for lead. Average lead concentrations in soil samples were then used to estimate the average amount of soil adhering to the hands.

The average amount of soil adhering to the hands was then divided by the "available" (for contact with a treated wood surface) skin surface area of the hands for the average age of the children in the Roels *et al.* study (*i.e.*, 11 year olds) to generate a soil adherence factor (AF) of  $1.1 \text{ mg}/\text{cm}^2$  for both boys and girls. The "available" skin surface area of the hands was assumed to be approximately one-third of the total surface area of both hands (the basis for this assumption is discussed below). Using median skin surface area data specific to the child resident in this HHRA, the soil AF of  $1.1 \text{ mg}/\text{cm}^2$  was used to estimate the average mass of soil on the hands of this receptor.

This soil loading estimate was then combined with an estimated soil ingestion rate to derive the HTE factor, which is an estimate of the fraction of the mass of soil adhering to the hands that would need to be ingested to yield the estimated soil ingestion rate. A median soil ingestion rate of 36 mg/day for children, age 2-6 years, was calculated based on a soil ingestion study conducted in Amherst, Massachusetts (Calabrese *et al.*, 1989; Stanek and Calabrese, 1995). This soil ingestion rate is the mean estimate for the 50<sup>th</sup> percentile child. When divided by the hand soil loading estimate for the child resident (146 mg on both hands), a daily HTE factor of approximately 0.25

hand loads/day is calculated. This value indicates that, on average, incidental ingestion of approximately one-fourth of the soil adhering to the front surface of the child resident's hands yields the typical estimated daily soil ingestion rate. In this HHRA, the HTE factor is used to estimate the percentage of the total amount of DR on the surface of both hands that is incidentally transferred to the mouth during incidental hand-to-mouth contact.

An HTE factor of 0.13 was used for the adult utility pole worker (ages 18-30 years old). This value is approximately one-half of the HTE derived for the child resident (ages 2-6 years old). The HTE factor for the adult worker is smaller to reflect the reduced hand-to-mouth behavior in people greater than 6 years of age. This assumption is supported by the fact that the USEPA-recommended mean soil ingestion rate for adults is exactly one-half of the recommended value for children less than 6 years of age (USEPA, 1997a).

The Roels *et al.* (1980) study was selected as the basis for the HTE factor because it provides direct empirical measurements of soil adherence to the palmar surface of children's hands following a variety of activities. By contrast, many of the more recent studies of soil adherence have focused on adherence associated with specific activities or have evaluated soil adherence over more extensive skin surface areas. Palmar surfaces are the primary skin area for contact with treated wood surfaces. And soil adherence to palmar surfaces is generally greater than adherence to the skin of other body parts. Thus, Roels *et al.* was selected as the best study for estimating the HTE factor because the data provided in this study are most relevant to the assessment of exposure to DR on children's hands.

*Skin Surface Area (SA).* It was assumed that DR on the surface of both hands could be ingested *via* hand-to-mouth contact. The skin surface area of the hands assumed to be available for contact with a treated wood surface is 1/3 of the total surface area of both hands. The total surface area of both hands is based on male and female hand size data in the USEPA's *Exposure Factors Handbook* (1997a). The 2/3 reduction in total hand surface area is based on four separate sources, described below.

1. The USEPA's Science Advisory Council for Exposure recommended a number of revisions to the Office of Pesticide's standard operating procedures (SOPs) for residential exposure assessment (USEPA, 2001b). One of the recommended revisions, regarding the assessment of children's exposure to pesticide residue, indicates that only the palmar surface area of both hands should be considered in quantifying exposure *via* hand-to-mouth contact. This recommendation also indicates the approximate palmar surface areas for children age

- 3-4, the values of which are approximately 1/3 of the total surface area of both hands for children in this age range according to hand size data in USEPA (1997a).
2. In a study by Rodes *et al.* (2001), hand-press trials were used to quantify the transfer of particles from indoor surfaces to human skin. One of the findings in this study was that only approximately 1/3 of the hand surface typically came in contact with a smooth test surface (Rodes *et al.*, 2001).
  3. The results of the Rodes *et al.* (2001) study are supported by a study by Brouwer *et al.* (1999) where it was reported that only 4-16% of the total surface area of the palm was exposed to a contaminated glass plate surface after a single hand press, and that this value increased to about 40% after 12 contacts.
  4. And finally, measured hand sizes based on tracing each hand of the adult male and female volunteers in the SCS (1998; 2001) and RTI (2003) hand loading studies were compared to the total surface area of one hand for adults in USEPA (1997a). Based on this comparison, the measured hand surface areas from these studies were on average, approximately 1/3 of the total hand surface area calculated for an adult based on data in USEPA (1997a).

The hand surface areas used to quantify incidental ingestion exposure to DR are 132 cm<sup>2</sup> for the child resident and 300 cm<sup>2</sup> for the adult worker.

*Relative Bioavailability Absorption (RBA).* The basis for the bioavailability of arsenic complex in DR is briefly discussed here. Refer to Gradient (2001) for a more detailed discussion of the studies and issues related to the bioavailability of this complex.

A critical factor determining the magnitude of potential exposures and risks associated with a chemical is its bioavailability or the amount of the chemical that is actually absorbed into the body. A chemical's bioavailability is influenced by such factors as the species of the chemical, the matrix in which it is present, the amount of time that a chemical is in a matrix, and the route by which exposure occurs. When chemicals are ingested, bioavailability is determined by the amount of a chemical that is dissolved in gastrointestinal fluids and absorbed across the gastrointestinal tract into the bloodstream. An ingested chemical that is adsorbed to soil or some other solid medium like wood dust may be absorbed less completely than the same ingested dose of the chemical when dissolved in water (NEPI, 2000).

Another important factor to consider is the relative bioavailability of the chemical under the exposure conditions of interest when compared to the bioavailability of the chemical under the exposure conditions present in the study that forms the basis for the quantitative toxicity criteria for

the chemical (USEPA, 1989). Frequently, quantitative toxicity criteria are calculated based on studies where the chemical was administered in food or water. By contrast, risk assessments for chemicals in the environment often require assessments of exposures and risks associated with chemicals in soil or other solid media. Where the bioavailability of the chemical observed in the toxicity study is likely to differ from that under the exposure conditions of interest, an RBA factor is derived. The RBA factor for a specific chemical reflects the absorption fraction from the medium of interest relative to the absorption from the exposure medium used in the relevant toxicity study (e.g., food or water).

An appropriate RBA factor for assessing exposure to arsenic complex in DR is based on a study conducted by Casteel *et al.* (2003a). A brief description of the Casteel *et al.* (2003a) study and some of the results are presented below. The study investigators plan to submit a manuscript for publication that will contain a full description of the study and its results.

For this study, DR was collected from the top (*i.e.*, weathered) surface of aged CCA-treated wood coupons cut from in-service deck boards by gently rubbing the coupons with a soft bristle test tube brush under deionized water. The dislodgeable material was filtered to remove any splinters and concentrated to a fine powder, which constituted the test material for the study. The study design involved three different doses of arsenic complex in DR, two soluble sodium arsenate doses, and a control group. Twenty-four hour urinary arsenic measurements were obtained on three separate days after dosing in each group. A total of 30 juvenile swine were used (*i.e.*, 6 dose groups, 5 animals per group). The mean RBA for arsenic complex in the DR was 29% (90% confidence interval, 26-32%) and is the value used in this risk assessment for incidental ingestion exposure to DR in the residential and worker scenarios.

The results of the Casteel *et al.* (2003a) study were compared to two RBA studies using dogs. Peoples (1976) and Peoples and Parker (1979) conducted two studies involving dogs (n=3) fed sawdust from CCA-treated wood. Absorption of arsenic was assessed by comparing the amount of arsenic excreted in urine to the total ingested dose of arsenic complex. The mean RBA estimate derived from these two studies was 47%.

The results of Casteel *et al.* (2003a) were used in the assessment of oral exposure to arsenic complex in DR because this study used an established animal model and was the only available study that evaluated the relative bioavailability of arsenic complex in the same matrix that an individual is exposed to when making contact with the surface of CCA-treated wood.

In addition to the animal studies used to derive an estimate of oral bioavailability, other factors support an

assumption of reduced bioavailability for arsenic complex in DR. First, the chemical process that occurs during wood treatment is designed to bind the CCA in the wood so that the fixative will persist and prevent deterioration of the wood over a long period of time (Bull, 2001). Second, data from leaching studies indicate limited release of arsenic from treated wood under normal outdoor conditions (Osmose, 1996; Osmose, 2000). And finally, analysis of dislodgeable surface residue collected from CCA-treated wood coupons found that over 90% of the surface arsenic was insoluble in water (Osmose 2001).

*Exposure Frequency (EF)*. Different EF parameters were used for the child resident and adult worker exposure scenarios.

The EF used for the child resident is 350 days/year and assumes that 2 weeks per year is spent away from the home on vacation (USEPA, 1993a).

The EF for the adult worker is based on feedback from a number of utility companies in the U.S. Based on this information, it appears that for younger workers (e.g., 18-30 years old) a high-end estimate of the number of times a utility pole is actually climbed is once per workday. Thus, we used USEPA's RME estimate of 250 days/year for this receptor, which assumes a 5 day work week, 50 weeks per year (2 weeks/year are assumed to be spent on vacation) (USEPA, 1993a).

*Fraction of intake (FI)*. This parameter describes the fraction of each day that an individual may be exposed to DR and/or impacted soil. This parameter is based on the following assumptions:

- The FI parameter is only applicable to the incidental ingestion exposure route
- Exposure occurs nearly entirely outdoors during daily waking hours (assumed to be 12 hours/day)
- For both exposure scenarios, all time outdoors is spent on or near a CCA-treated wood utility pole
- Once a receptor is no longer in contact with either DR or impacted soil, there is no further intake (i.e., intake is proportional to time in contact with the exposure medium)

Because incidental ingestion typically occurs via hand-to-mouth contact, once DR and/or soil has been removed via hand-to-mouth contact, oral intake can only continue if the hands are reloaded with DR and/or soil. For example, in the residential scenario, once a child is no longer exposed to DR from a treated wood utility pole and the residue has been removed either orally or by hand washing, intake via oral ingestion will cease unless the child's hands are reloaded. Thus, a child's daily exposure is limited to the fraction of the day she/he is outdoors and in contact with the pole or impacted soil near the pole.

The fraction of each day that the child resident was assumed to be exposed to DR and/or impacted soil is 5.1 hours/day and is based on data in USEPA (1997b) regarding the 90<sup>th</sup> percentile amount of time spent outdoors at a residence for the two most relevant age ranges available (i.e., 1-4 and 5-11 years old). The value of 5.1 hours/day was reduced by 80% under the assumption that 20% of the time a child is outdoors and in the yard, he/she may be playing around a treated wood utility pole. Thus, the FI value used for this receptor is  $(5.1 \text{ hrs/day} \times 0.20)/12 \text{ hrs/day}$  or 0.09 (unitless). As discussed above, the denominator used to calculate the value of the FI parameter assumes 12 waking hours per day. This is a conservative assumption, especially for older receptors who are likely awake for more than 12 hours/day, because as the denominator is increased, the value of the FI parameter is decreased.

The FI for the adult worker is based on an 8 hour work day and feedback from a number of utility companies in the U.S. who reported that pole workers wear shoes, fire-retardant long pants, long sleeve shirts, and gloves. Thus, actual skin contact with either a utility pole or nearby soils is unlikely. However, Gradient conservatively assumed that part of the time (i.e., 20%) a worker may not be wearing gloves and may roll-up the shirtsleeves, thereby exposing the forearms. Thus, the value of the FI parameter for this receptor is  $(8 \text{ hrs/day} \times 0.20)/12 \text{ hrs/day}$  or 0.13 (unitless).

*Exposure Duration (ED)*. The values of the ED parameter used for the two receptors are based on their assumed age ranges. The ED for the child resident, ages 2-6, is 5 years. The ED for the adult utility pole worker, ages 18-30, is 12 years. The age range for the adult worker is based on feed-back from a number of utility companies in the U.S., which indicates that climbing utility poles is typically done by younger workers and that after 10 years or so, these workers often move into more supervisory roles where pole climbing is much less likely to occur.

*Body Weight (BW)*. In accordance with current USEPA risk assessment guidance, the use of a mean bodyweight is appropriate where RME parameters are used to quantify exposures (USEPA, 1989). A bodyweight of 17.8 kg was used for the child resident, ages 2-6, and is based on the mean bodyweight of boys and girls ages 2-6 (USEPA, 1997a). A bodyweight of 70 kg was used for the adult worker, ages 18-30, and is based on the mean bodyweight of male and female adults (USEPA, 1997a).

*Averaging Time (AT)*. The values of the AT parameter used to estimate non-cancer health risks for the two receptors are based on their assumed age ranges. The AT used to estimate non-cancer risks for the child receptor, ages 2-6 years, is 5 years. The AT used to estimate non-cancer risks for the adult worker is 12 years, based on an age range of 18-30 years.

For cancer risk, exposures are averaged over a lifetime. The current USEPA-recommended average life expectancy for women and men in the U.S. is 75 years (USEPA, 1997a). However, the USEPA used a lifetime expectancy of 70 years to calculate cancer potency values (i.e., cancer slope factors and unit risks) in its on-line chemical toxicity database, i.e., the Integrated Risk Information System or IRIS (USEPA, 1997a). Therefore, to be consistent with the derivation of cancer potency values recommended by the Agency and used for arsenic complex in this assessment, a 70-year life expectancy (which equates to an AT of 25,550 days) was used to estimate potential cancer risks for both receptors.

### **Dermal Contact with Arsenic Complex in DR**

There are two exposure factors in the assessment of dermal contact with arsenic complex in DR that differ from those discussed above regarding exposure via the incidental ingestion route, and include the available skin surface area (SA), and the dermal absorption fraction (DA). Only these exposure factors, which are unique to the dermal contact route, are discussed here.

*Available Skin Surface Area (SA).* This factor represents the amount of skin that is assumed to be available for contact with DR from a treated wood surface. The child resident was assumed to wear shoes, shorts and a T-shirt; however, only the hands were considered to be in contact with a utility pole during a playing event. Therefore, as described above, 1/3 of the total surface area of both hands (i.e., 132 cm<sup>2</sup>) was used as the skin surface available for contact with a treated wood surface.

As described above, the adult worker was assumed to wear shoes, fire-retardant long pants, long sleeve shirts, and gloves. Thus, actual skin contact with a utility pole is unlikely. However, Gradient conservatively assumed that part of the time a worker may not be wearing gloves and may roll-up the shirtsleeves, thereby exposing the forearms. Therefore, the SA for this receptor is based on 1/3 of the total surface area of both hands, and the entire surface area of both forearms for a total of 1,610 cm<sup>2</sup>. This calculated value is based on skin surface area data for adult men and women in USEPA (1997a).

*Dermal Absorption Fraction (DA).* This parameter represents the amount of a chemical that is in contact with the skin and that is absorbed through the skin and into the bloodstream.

A recent study by Wester *et al.* (2004) was used to assess dermal absorption of arsenic complex in DR. A brief description of this study and some of the results are presented below. Refer to the published article for a full description of the study and its results.

Like the Casteel *et al.* (2003a) study discussed above, Wester *et al.* (2004) used a sample of the same dislodgeable material collected from the weathered

surface of aged CCA-treated boards from in-service decks. The study design involved application of DR to a 100 cm<sup>2</sup> area of the abdomen of three female Rhesus monkeys that were maintained on a low arsenic diet. Each primate received a dose of soluble sodium arsenate, and of arsenic complex in DR, with a minimum of 14 days between dosing trials. Thus, each animal functioned as its own internal control. Urinary arsenic was collected during each of the 8 hour dosing periods, and at 24-hour intervals thereafter for the following 7 days.

The results indicate that dermal absorption of the soluble arsenic applied in solution was consistent with earlier research conducted using radiolabeled tracers (Wester *et al.*, 1993). Following application of the DR, absorption of arsenic complex in two of the animals was not detectable above background, and the third animal absorbed approximately 0.1% of the applied dose. Therefore, a value of 0.1% was used in this risk assessment to assess dermal absorption of arsenic complex in DR. This value represents the upper bound of absorption from the three animals in the study.

### **Incidental Ingestion of Arsenic Complex in Soil**

There are four exposure factors in the assessment of incidental ingestion of soil arsenic complex that differ from those discussed above regarding ingestion of arsenic complex in DR, and include the exposure point concentration (EPC), fraction of source (FS), soil ingestion rate (IR), and oral relative bioavailability of arsenic in soil (RBA). Only these exposure factors, which are unique to the soil ingestion route, are discussed here.

*EPC for Soil Arsenic Complex.* As discussed previously, the 95% UCLM of 214 mg/kg was used as the EPC for exposures to soil arsenic complex near a utility pole. The EPC is based on the estimated concentrations of arsenic in surface soil samples collected immediately adjacent to CCA-treated utility poles in Canada (Copper and Ung, 1997).

*Relative Oral Bioavailability Absorption (RBA).* The basis for the bioavailability of soil arsenic complex is discussed below.

It is widely recognized that the bioavailability of many metals and organic chemicals in soil tends to be considerably lower than bioavailability from food or water (see, for example, Ruby *et al.*, 1999 and Alexander, 2000). Bioavailability from soil can be affected by a number of factors, including the form of the chemical, its solubility, the size distribution of the ingested soil particles, the type of soil, the degree of encapsulation of the chemical within an insoluble matrix, and the nutritional status of the exposed individual.

USEPA guidance recognizes the need to make adjustments for the reduced bioavailability of constituents in soil. Specifically, in Appendix A of the USEPA's "Risk Assess-

ment Guidance for Superfund" (USEPA, 1989, pg. A-3), the Agency notes:

If the medium of exposure in the site exposure assessment differs from the medium of exposure assumed by the toxicity value (e.g., RfD values usually are based on or have been adjusted to reflect exposure via drinking water, while the site medium of concern may be soil), an absorption adjustment may, on occasion, be appropriate. For example, a substance might be more completely absorbed following exposure to contaminated drinking water than following exposure to contaminated food or soil (e.g., if the substance does not desorb from soil in the gastrointestinal tract).

USEPA guidance also recommends the use of RBA factors "to adjust a food or soil ingestion exposure estimate to match an RfD or slope factor based on the assumption of drinking water ingestion" (USEPA, 1989, pg. A-3).

Only a few studies have specifically examined the bioavailability of arsenic complex from CCA-treated wood, in soil. However, the results from these studies are consistent with the general results observed in other studies of soil containing arsenic from various sources - that the bioavailability of CCA-related arsenic complex in soil is less than the bioavailability of soluble arsenic.

This risk assessment relied on another study conducted by Casteel *et al.* (2003b) that used juvenile swine to measure the RBA of arsenic in soil collected adjacent to CCA-treated utility poles. A brief description of the Casteel *et al.* (2003b) study and some of the results are presented below. The study investigators plan to submit a manuscript for publication that will contain a full description of the study and its results.

The Casteel *et al.* (2003b) study involved 25 juvenile swine. The juvenile swine model is appropriate for assessing the bioavailability of metals to humans, particularly young children (Casteel *et al.*, 2001). There were 5 animals per dose group, two different soluble sodium arsenate doses, two different test soil doses, and a control group. The RBA was determined by comparing total urinary arsenic excretion (mg/day) in the soil test group to the sodium arsenate group. Casteel *et al.* (2003b) reported that the mean RBA for soil arsenic was 49% (90% confidence interval, 41-58%). This value was used in the assessment of oral exposure to arsenic complex in soil in the residential and worker exposure scenarios.

The RBA from the Casteel *et al.* (2003b) study is comparable to the results of one other animal study that used soil obtained from CCA treatment sites and higher

than results from a second study. The first study involved soil samples collected from a CCA wood treatment site fed to rats (Ng and Moore, 1996). This study yielded a RBA for the test soil of 38% relative to the absorption of sodium arsenate. In the second study, five *Cebus apella* monkeys were fed soil samples collected from five different sites, including a CCA wood treatment site (Roberts *et al.*, 2002). Relative bioavailability of the test soil was based on urinary excretion of arsenic compared to that observed following an oral dose of sodium arsenate. In this animal model, the relative bioavailability of arsenic complex in the CCA treatment site soils was estimated to be approximately 16.0%.

The results from the Casteel *et al.* (2003b) study are the most relevant for use in this risk assessment for several reasons, including:

1. The Casteel *et al.* (2003b) data are based on soil collected from adjacent to an in-place CCA-treated wood structure, whereas the Ng and Moore (1996) and Roberts *et al.* (2002) studies used soil from CCA treatment sites that may have included sources of arsenic (e.g., spilled CCA formulation, drippings from recently treated wood) other than fixed CCA in treated wood, which is the primary source of arsenic in soil near treated wood utility poles.
2. The Ng and Moore (1996) study also used rats, which are known to metabolize arsenic quite differently than humans (ATSDR, 2000).
3. Although the Roberts *et al.* (2002) study used a more representative animal model (*i.e.*, the monkey), this study had a smaller sample size than the Casteel *et al.* (2003b) study, suggesting a less reliable RBA estimate.

The chemistry of arsenic in soil also suggests a reduced bioavailability for this metalloid. As noted earlier, chromium (III) arsenate is the primary form of arsenic complex found in treated wood, and most likely the predominant form that migrates via rainwater run-off to nearby soils. Arsenates are insoluble and immobile in most soils, often due to interactions with iron and aluminum oxides, which are abundant in soils, especially those with a high clay content (Sadiq, 1997; Lin and Puls, 2000). Unless there are unusual reducing conditions, arsenic will remain in the arsenate form in soil. Reducing conditions can alter the chemical form of arsenic in the environment; however, such conditions are rare in surface soils. The chemical species of arsenic in soil near treated wood structures, its strong binding capacity to certain inorganic constituents in soil, and its reduced solubility and mobility in soil, all support the notion of a reduced bioavailability of CCA-derived arsenic complex in soil.

*Fraction of Source (FS)*. This parameter refers to the fraction of impacted soil that a receptor could potentially be exposed to.

Based on the results of the Cooper and Ung (1997) study, it was assumed that elevated concentrations of soil arsenic complex are limited to an approximate one-foot radius around a utility pole. This observation is supported by data showing that the area of soil affected by treated wood structures is relatively limited (Malcolm Pirnie, 2002; Stillwell and Graetz, 2001), and that arsenic is not very mobile in soil (Cooper, 1990; USDA, 1980).

Based on best professional judgment, an FS of 50% was used to assess exposures to arsenic complex in soil for the child resident via incidental ingestion and dermal contact. This factor is based on the limited area of impacted soil around a CCA-treated utility pole, and the EPC used for soil, which is based on the concentrations of arsenic complex in surface soil samples collected immediately adjacent to the base of the pole.

An FS of 50% was also used for the adult worker based on the reasons provided above, and because a significant portion of each workday (*i.e.*, 8 hours/day) involves activities where exposures to soil are possible. Therefore, on the job activities likely constitute a significant portion of this receptor's daily exposure to soil.

*Soil Ingestion Rate ( $IR_{soil}$ )*. Separate incidental ingestion rates for soil were used for the child resident and adult worker.

A soil ingestion rate of 100 mg/day was used as the RME estimate for the child resident. Based on recent soil ingestion studies in the published literature, a soil ingestion rate of 100 mg/day is consistent with an RME estimate (*i.e.*, 95<sup>th</sup> percentile). For example, the 95<sup>th</sup> percentile soil ingestion rate from Stanek and Calabrese (2000) ranges from 106 to 133 mg/day, depending on the time period used; and the 95<sup>th</sup> percentile soil ingestion rate from Stanek *et al.* (2001) is 91 mg/day.

A soil ingestion rate of 50 mg/day was used as the RME estimate for the adult worker. This value is based on data presented in USEPA (1997a) and empirical data indicating a central tendency estimate of 10 mg/day for this age group (Stanek *et al.* 1997). Relative to the RME estimate used for the child resident (100 mg/day), this value also conservatively assumes that incidental soil ingestion rates in older children and adults are one-half of those identified for younger individuals.

## **Dermal Contact with Arsenic Complex in Soil**

There are three exposure factors in the assessment of dermal contact with soil that differ from those discussed above regarding ingestion of soil arsenic complex, and include the skin surface area (SA), the soil adherence factor (AF), and the Dermal Absorption Fraction (DA). Only

these exposure factors, which are unique to the soil dermal contact route, are discussed here.

*Skin Surface Area Exposed (SA)*. This factor represents the amount of skin that is assumed to be available for exposure to soil. Consistent with USEPA risk assessment guidance, median skin surface areas were used as RME estimates to assess dermal exposure to CCA-impacted soil (USEPA, 1989).

The child resident was assumed to be wearing shoes, shorts and a T-shirt. Therefore, the SA used (3,165 cm<sup>2</sup>) is based on the surface area of both hands (total), the forearms, and lower legs for boys and girls ages 2-6 (USEPA, 1997a).

The adult worker was assumed to wear shoes, fire-retardant long pants, long sleeve shirts, and gloves. Thus, actual skin contact with soil is unlikely. However, Gradient conservatively assumed that part of the time a worker may not be wearing gloves and may roll-up the shirtsleeves-exposing the forearms. Therefore, the SA used (2,210 cm<sup>2</sup>) is based on the surface area of both hands (total) and the forearms for adult men and women (USEPA, 1997a).

*Soil to Skin Adherence Factor (AF)*. This parameter represents the amount of soil that adheres to the skin per unit surface area (USEPA, 2001a). Adherence factors vary depending on the properties of the soil, the part of the body exposed, and the type of activity. The AF used to quantify dermal exposure to soil for both the child resident and the adult worker is 0.2 mg/cm<sup>2</sup> and is recommended as an RME estimate for a child resident (ages 1-6) and an adult commercial/industrial worker (USEPA, 2001a).

*Dermal Absorption Fraction (DA)*. The dermal absorption value used to quantify exposure to soil impacted with arsenic complex is the USEPA's recommended value of 3% (USEPA, 2001b). This value is based on an older study by Wester *et al.* (1993) involving dermal absorption of soluble arsenic mixed with soil.

## **Toxicity Assessment**

This section describes the basis and application of the arsenic toxicity criteria used to quantify risks.

### **Overview of Dose-Response Data**

Gradient assessed potential cancer and non-cancer risks from exposures to arsenic complex associated with CCA-treated utility poles using dose-response relationships for carcinogenicity (*i.e.*, the oral cancer slope factor or CSF<sub>oral</sub>) and systemic toxicity (*i.e.*, the oral Reference Dose or RfD<sub>oral</sub>).

The primary source for arsenic toxicity criteria is the USEPA's Integrated Risk Information System (IRIS) (USEPA, 2004b). Toxicity criteria in IRIS undergo a peer review process and represent the generally accepted approach in

the Agency. A subchronic oral Reference Dose (subchronic RfD<sub>oral</sub>) for arsenic was used to quantify non-cancer health risks for the resident child. The subchronic RfD was developed by USEPA, Region 8 and is considered appropriate to assess acute and subchronic exposures to inorganic arsenic in drinking water, food, and soil for periods up to 7 years (USEPA, Region 8, 2001).

### Arsenic Cancer Toxicity Criterion

The CSF<sub>oral</sub> was used to assess the potential risk of cancer from oral exposure to arsenic. In general, the CSF<sub>oral</sub> is an upper-bound estimate of carcinogenic potency that relates estimates of lifetime average chemical intake to the incremental risk of an individual developing cancer over their lifetime (USEPA, 1992). The CSFs recommended by the USEPA are conservative upper-bound estimates, which means that the USEPA is reasonably confident that the “true” cancer risk does not exceed the estimated risk based on the CSF, and may be as low as zero.

The USEPA concluded that arsenic is a “human carcinogen”, a weight-of-evidence classification for carcinogenicity of “A” (USEPA, 2004b). This classification is based on sufficient evidence of carcinogenicity in human populations. Lung cancer has been associated with inhalation of arsenic, and skin, bladder, and possibly other internal cancers have been associated with ingestion of arsenic in drinking water.

The USEPA-recommended CSF<sub>oral</sub> for arsenic in IRIS is 1.5 (mg/kg-day)<sup>-1</sup>. This toxicity criterion is based on the incidence of skin cancer from a study of a large population (over 40,000 people) in Taiwan with chronic exposure to arsenic in drinking water and food (Tseng, 1977; Tseng *et al.*, 1968). The CSF<sub>oral</sub> was calculated using a multi-stage model, assuming a drinking water ingestion rate of 3.5 L/day for Taiwanese males and 2 L/day for Taiwanese females, an average Taiwanese body weight of 55 kg, and an average U.S. body weight of 70 kg.

There is currently considerable debate among the scientific community regarding the arsenic CSF<sub>oral</sub>. It is possible that the current value of 1.5 (mg/kg-day)<sup>-1</sup> may overestimate cancer risks for U.S. populations (Chappell *et al.*, 1997; Slayton and Beck, 2001; Schoen *et al.*, 2004). The key uncertainties regarding arsenic cancer toxicity are discussed later in the Uncertainty Assessment section.

There are no USEPA-derived toxicity criteria for studies involving dermal exposures. In the absence of dermal-specific CSFs, the CSF<sub>oral</sub> is used, assuming that once a chemical is absorbed into the blood stream, the carcinogenic effect is similar regardless of whether the route of exposure was oral or dermal. However, since a CSF<sub>oral</sub> is based on the amount of a chemical administered per unit time and body weight (chemical intake), it needs to be adjusted to be applicable to absorbed doses (dermal

exposures are expressed as absorbed intake levels) (USEPA, 1989; 2001a). If oral absorption is very high (near 100%), then the absorbed dose is virtually the same as the administered dose, and no adjustment of the CSF<sub>oral</sub> is necessary. If oral absorption is very low (e.g., 5%), the absorbed dose is much smaller than the administered dose, and an adjustment of the toxicity criteria is necessary. For any given chemical, the USEPA recommends adjusting the applicable toxicity criterion to evaluate dermal risks only when the oral absorption for a chemical is less than 50%, to “obviate the need to make comparatively small adjustments in the toxicity value that would otherwise impart on the process a level of accuracy that is not supported by the scientific literature” (USEPA, 2001a).

Because oral absorption for inorganic arsenic is about 95% in water (USEPA, 2001a), and the USEPA recommends adjusting toxicity for the dermal route only when oral absorption is less than 50%, no adjustment was made to the CSF<sub>oral</sub>, RfD<sub>oral</sub> or subchronic RfD<sub>oral</sub> in the assessment of health risks from dermal exposure to arsenic complex in DR or in soil.

### Arsenic Non-Cancer Toxicity Criterion

The RfD<sub>oral</sub> is used to assess potential non-cancer risk from oral exposure to a chemical and is an estimate of daily chemical intake that a sensitive population can experience over a lifetime with a negligible risk of adverse systemic health effects. The USEPA (1993b) derives RfDs by first identifying the highest dose level that does not cause observable adverse health effects (*i.e.*, the No Observed-Adverse Effect Level or NOAEL). If a NOAEL is not identified, a Lowest Observed Adverse Effect-Level, or LOAEL, may be used. This dose level is then divided by uncertainty factors to calculate an RfD. An uncertainty factor of 100 is often used, to account for interspecies differences (if animal studies were used) and sensitive human subpopulations (e.g., children and the elderly) (USEPA, 1993b). Additional uncertainty factors may be used, depending on the quality of the toxicity study and/or confidence in the data.

The USEPA recommends an RfD<sub>oral</sub> for arsenic of 3 x 10<sup>-4</sup> mg/kg-day to quantify non-cancer risk from chronic oral exposure to arsenic (USEPA, 2004b). The arsenic RfD<sub>oral</sub> is based on increased incidence of hyperpigmentation, keratosis, and possible vascular complications in the same Taiwanese population used as the basis for the CSF<sub>oral</sub> (Tseng, 1977; Tseng *et al.*, 1968). The USEPA calculated a NOAEL of 0.0008 mg/kg-day for skin lesions in the Tseng study, based on the reported drinking water concentration in the NOAEL group (0.009 mg/L), an assumed drinking water ingestion rate of 4.5 L/day, a daily arsenic intake from sweet potatoes and rice of 0.002 mg/day, and an average Taiwanese body weight of 55 kg, *i.e.*, [(0.009 mg/L × 4.5 L/day) + 0.002 mg/day]/55 kg

(Abernathy *et al.*, 1989). An uncertainty factor of 3 (based on a lack of reproductive toxicity data and uncertainty regarding toxicity in sensitive individuals) was applied to the NOAEL to derive an RfD of  $3 \times 10^{-4}$  mg/kg-day (0.0008/3). Overall, the USEPA has “medium” confidence in the study, “medium” confidence in the database (due to poor characterization of the exposure levels in the Tseng and other supporting studies), and “medium” confidence in the RfD<sub>oral</sub> for arsenic. It is noted in the IRIS substance file for arsenic that a clear consensus does not exist among Agency scientists regarding systemic arsenic toxicity (USEPA, 2004b). The IRIS file also notes that solid scientific arguments can be made for values within a factor of 2 or 3 of the current recommended RfD<sub>oral</sub> value (*i.e.*, 0.1 to 0.2 µg/kg-day).

### Arsenic Subchronic RfD<sub>oral</sub>

The IRIS database does not provide a toxicity criterion for assessing subchronic oral exposures to arsenic. However, USEPA’s Region 8 Office (Region 8) has derived an RfD<sub>oral</sub> for arsenic of 0.015 mg/kg-day ( $1.5 \times 10^{-2}$  mg/kg-day) that addresses both acute and subchronic exposures (USEPA, Region 8, 2001). According to Region 8, the subchronic RfD<sub>oral</sub> is appropriate to quantify non-cancer health risks from acute exposures to inorganic arsenic lasting one to fourteen days, and subchronic exposures lasting 15 days to 7 years (USEPA, Region 8, 2001). Based on these criteria, the subchronic RfD<sub>oral</sub> was used to quantify non-cancer health risks for the child resident for both oral and dermal exposures to arsenic complex.

Region 8 reviewed 18 different studies where arsenic exposure was primarily *via* drinking water. Based on this review as a whole, and a study by Mazumder *et al.* (1998) in particular, Region 8 determined that the NOAEL for arsenic was 0.015 mg/kg-day. At this exposure level, signs of arsenic-related skin effects (hyperkeratosis, hyperpigmentation) were absent in children exposed to arsenic in drinking water (USEPA, Region 8, 2001). Because the NOAEL is based on a review of a large number of studies in human populations, including some involving sensitive subgroups, Region 8 determined that additional safety factors were not required in order to derive an RfD<sub>oral</sub> from the NOAEL. An alternative derivation,

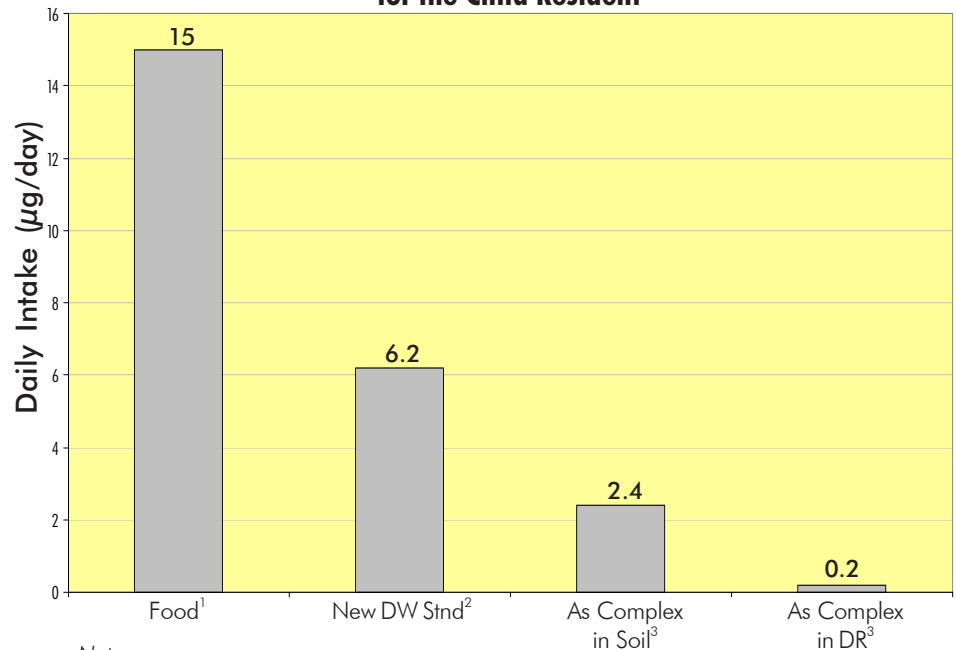
based on applying safety factors to the LOAEL reported in the studies, yielded a similar value of 0.02 mg/kg-day. Because many of the epidemiological studies included a large number of children, the acute/subchronic RfD<sub>oral</sub> is protective of children. Region 8 also noted that the NOAEL (and hence the RfD<sub>oral</sub>) could be as high as 0.03 to 0.04 mg/kg-day based on an evaluation of the studies by Tseng (1977) and Tseng *et al.* (1968). Thus, use of an RfD<sub>oral</sub> of 0.015 mg/kg-day may actually result in an overestimate of non-cancer risk. An analysis by Tsuji *et al.* (2004) concluded that available data indicate an acute/subchronic oral reference level for children in the range of 0.005 to 0.015 mg/kg-day. The reference levels developed in this study support Region 8’s acute/subchronic RfD<sub>oral</sub> value.

## Exposure and Risk Characterization

### Exposure Comparison

In order to put potential exposures to arsenic complex associated with CCA-treated utility poles into perspective, Figure 1 (below) provides a comparison for the child resident between the daily intake of arsenic complex in DR and in soil, to the daily intake of inorganic arsenic in drinking water (based on the new federal drinking water standard for arsenic) and in a typical U.S. diet. Figure 2 contains the same type of comparison for the adult worker.

**Figure 1**  
**Comparison of Inorganic Arsenic Intakes for the Child Resident**



Notes:

<sup>1</sup>Drinking water (DW) intakes are based on the new federal drinking water standard for arsenic (10 mg/L) and are calculated using USEPA (1997) recommended 90<sup>th</sup> or 95<sup>th</sup> percentile drinking water ingestion rates for children ages 1-10 years old (1.5 L/day) and adults (2.3 L/day).

<sup>2</sup>Daily intake of inorganic arsenic in food for children (ages 1-6 years old) is from Yost *et al.* (2004) and is the 95<sup>th</sup> percentile intake rate. It should be noted that the average

These graphs indicate that even when RME assumptions and parameters are used to quantify potential daily intakes of arsenic complex associated with CCA-treated utility poles, these intakes are significantly less than the intake of inorganic arsenic from the typical U.S. diet or from drinking tap water at the new federal drinking water standard for arsenic.

### Calculation of Cancer Risks

Cancer risks are characterized as the incremental probability that an individual will develop cancer during his or her lifetime due to chemical exposure under a specific exposure scenario (e.g., the residential scenario evaluated in this risk assessment). The term “incremental” implies the risk above the background cancer risk experienced by all individuals in the course of daily life. At least 1 in 3 Americans will develop cancer during their lifetime (ACS, 2004), so the background cancer risk is 0.33, or 330,000 in one million. The incremental risk is a measure of the additional estimated cancer risk due to a specific exposure. Cancer risks are expressed as a unitless probability (e.g., one in a million, or  $1 \times 10^{-6}$ ) of an individual developing cancer over a lifetime, above background risk, as a result of exposure to a carcinogen.

Excess (incremental) cancer risks for the exposure routes (i.e., oral ingestion and dermal contact) evaluated in this risk assessment are calculated using intake estimates (averaged over a lifetime or lifetime daily dose) and cancer slope factors (CSFs) (e.g.,  $CSF_{oral}$ ). Estimated intakes and CSFs are combined to calculate an excess lifetime cancer risk (ELCR) according to the following equation (USEPA, 1989):

$$Cancer\ Risk = Intake \left( \frac{mg}{kg \cdot day} \right) \times CSF \left( \frac{mg}{kg \cdot day} \right)^{-1}$$

Potential cancer risk is calculated by multiplying intake (i.e., the dose via either the oral or dermal exposure route) by the  $CSF_{oral}$ .

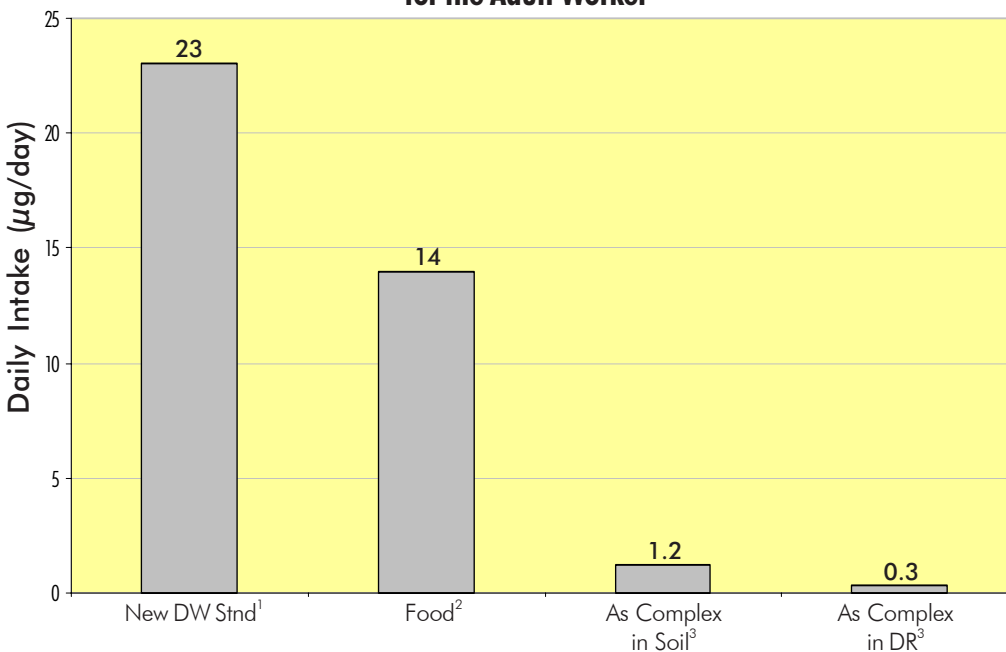
### Summary of Cancer Risks

Because this risk assessment was prepared in accordance with current USEPA risk assessment guidance, the estimated cancer risks were compared to the Agency’s target cancer risk range of  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  (USEPA, 1991). A cancer risk of  $1 \times 10^{-6}$  represents one case of cancer in every 1,000,000 population and a cancer risk of  $1 \times 10^{-4}$  represents one case of cancer in every 10,000 population.

The cumulative cancer risk (i.e., risks associated with the same toxicologic mechanism) estimated for the child resident is  $1.6 \times 10^{-5}$ . This represents an upper bound estimate of approximately two additional cases of cancer in every 100,000 population and is within the USEPA’s cancer target risk range. (It should be noted that the actual number of cancer cases could be zero). Of this cumulative risk, approximately 94% is attributable to soil arsenic complex and the remaining 6% attributable to arsenic complex in DR. By exposure pathway, approximately 77% of the cumulative risk is from dermal contact, the remaining 23% is from incidental ingestion.

The cumulative estimated cancer risk for the adult utility pole worker is  $5.7 \times 10^{-6}$ . This represents an upper bound estimate of approximately six additional cases of cancer in every

**Figure 2  
Comparison of Inorganic Arsenic Intakes  
for the Adult Worker**



<sup>1</sup>dietary intake of inorganic arsenic in Yost et al. (2004) is nearly 3-fold less than that estimated in Yost et al. (1998), which provided an average estimate for a child ages 6 months-2 years old. The reason for this difference between the studies is not clear. Daily intake of inorganic arsenic in food for adults is from Yost et al. (1998). Yost et al. (2004) does not provide dietary inorganic arsenic intakes for adults.

<sup>3</sup>Intake of arsenic complex (As Complex) based on incidental ingestion and skin contact.

1,000,000 population and is within the USEPA's cancer target risk range. (It should be noted that the actual number of cancer cases could be zero). Of this cumulative cancer risk, approximately 79% is attributable to soil arsenic complex and the remaining 21% attributable to arsenic complex in DR. By exposure pathway, approximately 74% of the cumulative cancer risk is from dermal contact, the remaining 26% is from incidental ingestion.

Attachments 3 through 6 contain the equations used to calculate the cancer and non-cancer risk estimates for both the residential and worker scenarios.

### Calculation of Non-Cancer Risks

Non-cancer health risks are expressed as hazard quotients rather than probabilities. A hazard quotient compares the estimated daily intake or average daily dose of a chemical to an applicable Reference Dose (RfD) derived by the USEPA. The hazard quotient is calculated using an RfD according to the following equation (USEPA, 1989):

$$\text{Hazard Quotient} = \frac{\text{Intake} \left( \frac{\text{mg}}{\text{kg} \cdot \text{day}} \right)}{\text{RfD} \left( \frac{\text{mg}}{\text{kg} \cdot \text{day}} \right)}$$

Potential non-cancer risk is calculated by dividing intake (i.e., the dose via either the oral or dermal exposure route) by an RfD<sub>oral</sub>.

In accordance with USEPA risk assessment guidance, hazard quotients are calculated for each receptor and exposure route, and then summed across the different exposure routes to calculate a hazard index (USEPA, 1989). Because a hazard quotient is simply a ratio of estimated intakes of a chemical to its RfD, a hazard index does not represent the probability that an adverse health effect may occur. Instead, a hazard index indicates whether estimated chemical intakes for an individual present a potentially significant non-cancer health risk based on a comparison to a USEPA-recommended RfD.

### Summary of Non-Cancer Risks

According to USEPA risk assessment guidance, if a hazard index is less than 1.0, no further evaluation of non-cancer risks is necessary (USEPA, 1989). The estimated hazard index for the child resident is  $9.6 \times 10^{-3}$  (0.0096), and includes all exposures to arsenic complex in DR and in soil. The estimated hazard index for the adult worker is  $7.3 \times 10^{-2}$  (0.073), and also includes all exposures to arsenic complex in DR and in soil. These estimated hazard indices are well below the USEPA's target hazard index of 1.0.

Approximately 94% of the hazard index for the child resident is attributable to soil arsenic complex; the remaining 6% is from arsenic complex in DR. For the adult worker, approximately 78% of the hazard index is attributable to arsenic complex in soil, the remaining 22% is from arsenic complex in DR.

### The Use of Cancer Risk Targets in Risk Management Decisions

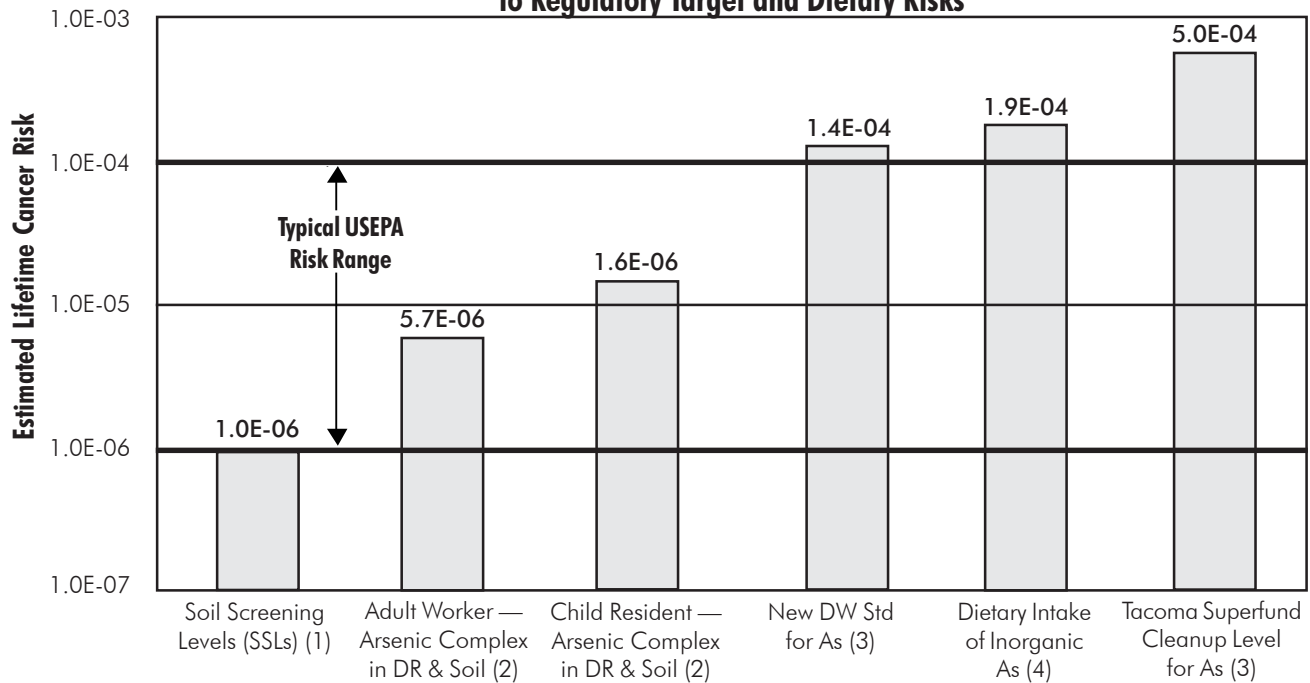
This section discusses the issue of significant risk in the regulation of chemical exposures. It is included to provide some perspective on the estimated risks from exposures to arsenic complex associated with CCA-treated utility poles.

An acceptable cancer risk in the federal government is not defined as a single precise value, but rather a range of values that allows the selection of an acceptable risk within this range based on a number of considerations. As discussed below, USEPA uses  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  as a "target range" for managing risks under its Superfund program (USEPA, 1991). In general, site-specific exposures to chemicals are regulated so that estimated risks are within this target range. However, based on a review of the published literature and several federal regulatory decisions, we observed that cancer risks associated with USEPA-approved site remediations, and air and drinking water standards, can, under certain circumstances, exceed this range. Several factors, including feasibility, availability of alternatives, and the size of the affected population, can influence the precise risk limit selected (see for example Travis *et al.*, 1987).

The 1990 National Contingency Plan (NCP), which is an environmental guidance document prepared by the USEPA under the Comprehensive Environmental Response, Compensation, and Liability Act or CERCLA (more commonly known as Superfund), contains language indicating that remediation of hazardous waste sites should be managed so that concentrations of chemicals remaining in soil are associated with cancer risks within a range of  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  (USEPA, 1990). The NCP states that risks within this range are "generally acceptable" and that risks greater than  $1 \times 10^{-4}$  may be permitted depending on site-specific considerations. Furthermore, an April 1991 memo from Assistant Administrator Donald Clay in the Office of Solid Waste and Emergency response (OSWER), states that cumulative cancer risks up to  $1 \times 10^{-4}$  can be used to develop remedial alternatives for Superfund sites and that remediation would not typically be required at a site if risks associated with RME parameters were  $1 \times 10^{-4}$  or less (USEPA, 1991). The memo goes on to state that in certain cases the Agency "may consider risk estimates slightly greater than  $1 \times 10^{-4}$  to be protective."

Thus, cancer risks within the  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  target range are generally considered acceptable for Agency decisions regarding hazardous waste site cleanups, with the

**Figure 3**  
**Comparison of Estimated Risks for the Child Resident and Adult Worker**  
**to Regulatory Target and Dietary Risks**



Notes:

- 1 SSLs are derived by the USEPA (2001d) and are based on a target risk of  $1 \times 10^{-6}$  for carcinogens. SSLs are considered protective for residential and industrial exposures to soil.
- 2 Cumulative potential risk based on incidental ingestion and dermal contact with arsenic complex from CCA-treated utility poles, in dislodgeable residue (DR) and soil.
- 3 Upper-bound estimated risk based on drinking water at the new federal standard for arsenic of  $10 \mu\text{g/L}$  (USEPA, 2001c). Does not include exposure to arsenic in food or in water used for cooking.
- 4 Estimated risk based on exposure in inorganic arsenic in typical U.S. diet (Yost et al., 1998) for a child and adult receptor, ages 2-31.
- 5 Estimated risk for exposures to arsenic in soil at a Superfund site in Tacoma, Washington (USEPA, Region 10, 1993).

possibility that risks above  $1 \times 10^{-4}$  may be considered acceptable depending on site-specific considerations. Several examples exist of higher risk levels being allowed at hazardous waste sites. One instance is the 1993 Record of Decision (ROD) for Commencement Bay in Tacoma, Washington (Operable Unit 04, Ruston/North Tacoma Study Area), where a remediation action level for soil arsenic of 230 mg/kg was established based on a lifetime cancer risk of  $5 \times 10^{-4}$  (USEPA, Region 10, 1993). Another example is reflected in a USEPA memorandum from OSWER regarding cleanup goals (CUGs) for dioxin and related compounds (*i.e.*, dioxin toxicity equivalents or TEQs) in soil at Superfund and RCRA sites, which states that the recommended residential CUG for dioxin/TEQs corresponds to a  $2.5 \times 10^{-4}$  lifetime cancer risk (USEPA, 1998).

The USEPA has also applied a target risk range to set emission standards for the Clean Air Act (CAA), and to establish Maximum Contaminant Levels (MCLs) for the Safe Drinking Water Act (SDWA). Air emission standards for stationary sources are based on an upper-bound cancer risk of  $1 \times 10^{-4}$  for the maximally exposed individual (Sadowitz and Graham, 1995). In setting limits for car-

cinogens in drinking water (*i.e.*, MCLs) the USEPA Office of Drinking Water tries to ensure that the standards for carcinogens do not exceed the  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  range (Rosenthal et al., 1992). However, MCLs for vinyl chloride and ethylene dibromide are associated with excess lifetime cancer risks between  $1 \times 10^{-4}$  and  $1 \times 10^{-3}$  (Rosenthal et al., 1992). And the recent MCL for arsenic of 10 mg/L carries an upper bound risk of  $1.4 \times 10^{-4}$  (USEPA, 2001c).

Thus, there is no bright-line that the USEPA uses to establish acceptable cancer risks. Acceptable risks typically range between  $10^{-6}$  and  $10^{-4}$ , with the precise value determined on a case-by-case basis and consideration of a number of different factors.

The cumulative potential cancer risks for the child resident and adult utility worker evaluated in this risk assessment are  $1.6 \times 10^{-5}$  and  $5.7 \times 10^{-6}$ , respectively. These risk estimates are based on RME assumptions and parameters and are within the USEPA's cancer target risk range. Figure 3 (above) provides a comparison of the potential cancer risks estimated in this risk assessment, to several regulatory target risks used by the USEPA, and to the estimated risk from ingestion of naturally occurring inorganic arsenic in food. This comparison shows that

potential, high-end risk estimates for exposures to arsenic complex associated with CCA-treated utility poles are significantly less than potential risks associated with the new federal drinking water standard for arsenic, soil arsenic remediation goals at the Tacoma Superfund site, and from exposure to naturally occurring inorganic arsenic in food. While the potential risks for a child resident and an adult utility pole worker exceed the target risk used for carcinogens in the USEPA's soil screening levels (SSLs), it should be noted that the SSLs do not represent cleanup levels. They are used to assess whether any further site investigation is needed (USEPA, 2001d). In the case of arsenic in particular, many locations in the U.S. have naturally occurring levels of arsenic in soil that exceed the concentration for arsenic that the SSL is based on (Shacklette and Boerngen, 1984; Dragun and Chiasson, 1991). Thus, Gradient concludes that the potential risks from exposures to CCA-treated utility poles for a child resident and an adult utility pole worker do not exceed acceptable limits and do not present a health concern.

## Uncertainty Assessment

The process of evaluating human health risks from exposures to impacted environmental media involves multiple steps. Inherent in each step of the process are uncertainties that ultimately affect the final risk estimates. Uncertainties may exist throughout the assessment process, including the collection of samples used to identify constituents, laboratory analysis of samples, estimation of potential exposures, and derivation of toxicity criteria. These uncertainties may result in either an over- or under-estimation of risks. However, because this is a screening-level risk assessment, conservative assumptions and RME estimates were used so as to overestimate rather than underestimate potential exposures and risks.

Below is a discussion of the most significant sources of uncertainty in the Exposure Assessment, and in the Toxicity Assessment sections of the HHRA. Other sources of uncertainty exist than those evaluated here; however, their impact on the overall estimated risks are comparatively insignificant.

## Exposure Assessment

*Exposure Point Concentration for Soil Arsenic Complex.* The approach used to calculate the EPC for soil arsenic complex is likely to result in an overestimate of risk because only surface soil samples collected immediately adjacent to the base of utility poles were used. It is unlikely that a receptor would be exposed to only these soils while working/playing near utility poles. The concentration of soil arsenic complex in soil samples collected just one foot

from the utility pole approached background levels (Cooper and Ung, 1997).

*Relative Oral Bioavailability Absorption.* The RBA factors used for arsenic complex in DR and in soil are based on well conducted studies (*i.e.*, Casteel *et al.*, 2003a,b) that used the same test material (*i.e.*, DR or soil) to which people may be exposed when contacting CCA-treated wood and CCA-impacted soils. Additional data supporting a reduced RBA for arsenic complex in DR and in soil includes animal bioavailability studies of inorganic arsenic in sawdust (Peoples, 1976; Peoples and Parker, 1979) and in soil (Roberts *et al.*, 2002). Furthermore, data from leaching studies, the chemistry of CCA fixation, and the chemical species of arsenic complex in DR all indicate a reduced RBA for arsenic complex in DR. Thus, the oral bioavailability estimates used in this risk assessment are considered reliable.

Uncertainty exists in results obtained from *in vivo* bioavailability studies because the anatomy and physiology of the animals used in these studies may differ from those of humans (Ruby *et al.*, 1999; Valberg *et al.*, 1997). However, juvenile swine are considered a good physiological model for gastrointestinal absorption in children (Weis and LaVelle, 1991).

*Dermal Absorption Fraction.* The DA used to represent dermal absorption of arsenic complex in DR is based on a recent study by Wester *et al.* (2004) that measured the absorption of arsenic complex from DR applied to the skin of Rhesus monkeys. The Rhesus monkey is considered a relevant animal model to assess dermal absorption in humans (Wester and Maibach 1975; 1989). And like the oral bioavailability studies (*i.e.*, Casteel *et al.*, 2003a,b), Wester *et al.* (2004) used the same test material (*i.e.*, DR) to which people may be exposed when contacting CCA-treated wood.

Dermal absorption of arsenic complex in soil is based on an older study by Wester *et al.* (1993) that involved adding soluble arsenic to soil (with no aging) and applying this mixture to the skin of a Rhesus monkey to measure absorption of radio-labeled arsenic. We used this study because it provides the only available estimate of dermal absorption of arsenic from soil. However, limitations in this study add uncertainty to the estimate and likely bias the value high. The two most significant limitations in this study are discussed here. First, the form of arsenic from CCA-treated wood is primarily chromium (III) arsenate (Bull, 2000; 2001; Lytle, 2003), and arsenates are known to be insoluble and immobile in most soils, especially those with a high clay content (Sadiq 1997; Lin and Puls 2000). A soluble form of arsenic (*i.e.*, sodium arsenate) was used in Wester *et al.* (1993). Second, arsenic in aged soils is less bioavailable via the oral route than in fresh soils (Loehr, 1996; Alexander, 2000). Therefore, using the DA from Wester *et al.* (1993) without any adjustment to account

for the reduced solubility of arsenic complex in aged soils could substantially overestimate dermal absorption (and risk) of arsenic complex in soil.

**Exposure Frequency.** The number of days/year (*i.e.*, 350) that the child resident was assumed to be exposed to arsenic complex from CCA-treated utility poles may overestimate exposure for this receptor because this value assumes that a child resident is in contact with a treated wood utility pole and nearby soils nearly everyday of the year that he/she is outdoors, which is probably not realistic.

Implicit in the exposure frequency assumption is that once a child resident or an adult worker is exposed dermally to either DR or soil, exposure is assumed to occur all day with no adjustment to account for washing hands or other body parts. This is a conservative assumption that likely overestimates dermal exposure for both receptors.

**Skin Surface Area.** Based on feed-back from a number of utility companies across the U.S., utility pole workers typically wear shoes, and fire-retardant long pants, long sleeve shirts and gloves. Thus, actual skin contact with a utility pole or nearby soils is unlikely while working. However, Gradient conservatively assumed that for part of the time (*i.e.*, 20%), a utility pole worker may not be wearing gloves and may roll-up the shirtsleeves, thereby exposing the forearms. This is a conservative assumption, which based on the best available information, is likely to result in an overestimate of the skin surface area available for exposure to DR and impacted soils for this receptor.

## Toxicity Assessment

**Toxicity Via Dermal Exposure.** The approach used to evaluate risks from dermal exposure addresses systemic cancer and non-cancer effects by assuming that once a chemical is absorbed into the blood stream the health effects are similar regardless of whether the route of exposure was oral or dermal. However, there are uncertainties associated with this approach because dermally absorbed chemicals may have different patterns of distribution, metabolism, and excretion than orally absorbed chemicals (USEPA, 2001a). Use of oral toxicity criteria to evaluate dermal exposures may over- or under-estimate risks, depending on the chemical. Furthermore, this approach does not address potential dermal toxicity associated with direct contact (*i.e.*, “port of entry” effects), such as allergic contact dermatitis, chemical irritation, and skin cancer. Although the USEPA is currently in the process of developing chemical-specific dermal toxicity criteria for these types of health effects, such values are not currently available (USEPA, 2001a).

**Oral Cancer Slope Factor.** There is considerable debate among the scientific community regarding the CSF<sub>oral</sub> for arsenic. It is possible that the current value of 1.5 (mg/kg-day)<sup>-1</sup> may overestimate cancer risk for U.S. populations (Chappell *et al.*, 1997; Slayton *et al.*, 1996;

Slayton and Beck, 2001; Schoen *et al.*, 2004). The key uncertainties regarding the CSF<sub>oral</sub> for arsenic are summarized below:

- **Exposure Assessment.** There are considerable scientific concerns about the exposure estimates in the Taiwanese study (Slayton *et al.*, 1996; Chappell *et al.*, 1997; Brown *et al.*, 2000; Brown and Ross, 2002; Lamm *et al.*, 2003). Individual exposures were not characterized and exposures were estimated based on average well arsenic concentrations in each village. The original data are not available. The analytical method used to measure arsenic concentrations may not have been accurate at low levels. Other possible sources of exposure (arsenic in rice and yams) were not controlled. Therefore, the Taiwanese data are inadequate for quantitative dose-response assessment for arsenic and skin cancers.
- **Dose-Response Modeling.** The USEPA calculated the current CSF<sub>oral</sub> using a nearly linear dose-response relationship (USEPA, 2004b). However, a review of the literature by Schoen *et al.* (2004) concluded that mechanistic considerations support a non-linear dose-response relationship for arsenic. A similar conclusion was reached by a scientific peer review panel (ERG, 1997). Epidemiological studies in the U.S. (*e.g.*, see Lewis *et al.*, 1999; Steinmaus *et al.*, 2003) provide no convincing evidence of arsenic carcinogenicity for U.S. populations. A fuller discussion of these studies is provided in Schoen *et al.* (2004).
- **Human-to-Human Variations.** In general, dose levels, genetic factors, dietary patterns, or other lifestyle factors may alter arsenic metabolism and detoxification in different populations (Beck and Slayton, 1998; Del Razo *et al.*, 1997). Protein deficiencies in the Taiwanese diets could have affected their ability to methylate, and therefore, detoxify arsenic (NRC, 1999). A recent case-control study in West Bengal, India by Mitra *et al.* (2004) demonstrated that several nutritional deficiencies, such as protein and calcium deficiency may increase susceptibility to arsenic-caused skin lesions. Thus, extrapolations from the Taiwan population, which was nutritionally disadvantaged, to the U.S. population is likely to overestimate risk from exposure to arsenic.

In a draft probabilistic risk assessment for children who contact CCA-treated wood structures, USEPA used an arsenic CSF<sub>oral</sub> of 3.67 (mg/kg-day)<sup>-1</sup> (USEPA, 2003). This value is more than 2-fold greater than the current arsenic CSF<sub>oral</sub> in IRIS (USEPA, 2004b), and is based on the same Taiwanese data. Even using this higher value, the cumulative cancer risk estimates for the child resident and the adult utility pole worker in this risk assessment are within the USEPA cancer target risk range.

**Subchronic Oral Reference Dose.** The level of confidence or certainty in a toxicity criterion such as an RfD is generally governed by four factors: (1) a preference for human rather than animal toxicity data, (2) the existence of multiple studies showing similar results in different populations, (3) the quality of the studies being considered, and (4) the availability of a NOAEL rather than a LOAEL (USEPA, 1993b). Considering these factors, a high degree of confidence can be ascribed to the subchronic RfD<sub>oral</sub> derived by USEPA, Region 8 and used in this risk assessment. This toxicity criterion is based on a large number of epidemiological studies that were conducted in different populations and under different exposure conditions. Furthermore, although the data define a LOAEL, Region 8 scientists were able to use the extensive data of Tseng (1977) and Tseng *et al.* (1968) to define a likely boundary range for a NOAEL. The lower boundary on this range (0.02 mg/kg-day) is essentially the same as the subchronic RfD<sub>oral</sub> (0.015 mg/kg-day). This suggests that the effect of any error in the subchronic RfD<sub>oral</sub> would be quite small. And the fact that an independent group of ATSDR scientists derived a similar value of 0.005

mg/kg-day lends support to the Region 8 criterion (ATSDR, 2000). This is further supported by Tsuji *et al.* (2004), which concluded that available data indicate an acute/subchronic oral reference level for children in the range of 0.005 to 0.015 mg/kg-day.

## Results and Conclusions

### Results

The estimated cancer risks for the child resident and the adult utility worker are summarized below in Table 1. The estimated non-cancer risks for these receptors are summarized below in Table 2.

### Conclusions

The cancer risk estimates for the child resident and the adult worker, which are based on RME assumptions and parameters, are within the USEPA's cancer target risk range of  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$ . The non-cancer risk estimates for both of these receptors are below the USEPA's target non-cancer hazard index of 1.0.

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## Footnotes

<sup>1</sup> The predominant form of arsenic in CCA-treated wood is chromium (III) arsenate (Bull 2000; 2001).

<sup>2</sup> Gradient (2001) is available on the World Wide Web at: [www.woodpreservativescience.org/docs/011020cca-body.pdf](http://www.woodpreservativescience.org/docs/011020cca-body.pdf) It should be noted that this risk report contains several changes from Gradient (2001). These changes were made to clarify certain exposure assumptions and to incorporate the results of several new studies regarding exposure to arsenic complex in dislodgeable residue and in soil.

<sup>3</sup> Chromium (VI) used in the CCA formulation is reduced to chromium (III) when the metals are "fixed" to the wood (Bull, 2000, 2001). Indeed, the reduction of chromium (VI) to chromium (III) is essential for CCA to properly bind to the wood.

<sup>4</sup> Type, as it is used here, is the same as sample, however, in this description, sample refers to what is collected from the different wood types. Thus, type is used to avoid confusion.

<sup>5</sup> The 1.1 kg weight on the hand corresponds to the weight of the block sampling device. This was done to assess the difference in removal efficiency of DR between each sampling method with approximately the same downward pressure.

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# Attachment I

## Summary of Statistical Analysis

<b>Utility Pole Soil Data</b>
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Input Data

25	Number of samples		Uncensored values	
35	Uncensored	20	Mean	137.50
50	Censored		Lognormal mean	137.69
50	Detection limit or PQL		Std. devn.	128.13
50	Method detection limit		Median	100
50	TOTAL	20	Min.	25
50			Max.	550

65	Lognormal distribution?		Normal distribution?	
75	r-squared is:	0.967	r-squared is:	0.757
100	Recommendations:			
100	Assume lognormal distribution.			
100	W value is 0.964. This exceeds the tabled value of 0.905			

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95% UCLM (Land's method) is 213.5

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Note: Statistical analysis was performed using MTCA Stat 3.0, a statistical software program available from the State of Washington Department of Ecology, Olympia, WA. January 1995.

300  
300  
550



Gradient is an environmental consulting firm, headquartered in Cambridge, Mass., with nationally recognized specialties in risk assessment, contaminant fate and transport, and environmental chemistry.

Arch Wood Protection manufactures wood preservatives and other chemicals to enhance the properties of wood.

## Attachment 2

### Summary of Key Reasonable Maximum Exposure Assumptions and Factors CCA-Treated Utility Poles

Potential Receptor	Exposure Medium (1)	Fraction of Source	Exposure Route	Exposure Frequency	Fraction of Intake (2)	Exposure Intake Factors	Oral Bioavailability and Dermal Absorption
Child Resident (ages 2-6 years)	Dislodgeable Residue	100%	Incidental Ingestion	350 days/year (USEPA, 1993a)	$(5.1 \text{ hrs/day} \times 0.20)/(12 \text{ hrs/day}) = 0.09$ (unitless) (USEPA, 1997b)	0.25 hand loads/day (Gradient, 2001)	29% (Casteel et al., 2003a)
	Dislodgeable Residue	100%	Dermal Contact	Same as above	--	Available hand surface area: 132 cm <sup>2</sup> (USEPA, 1997a)	0.1% (Wester et al., 2004)
	Soil	50%	Incidental Ingestion	Same as above	$(5.1 \text{ hrs/day} \times 0.20)/(12 \text{ hrs/day}) = 0.09$ (unitless) (USEPA, 1997b)	100 mg/day (Stanek and Calabrese, 2000; Stanek et al., 2001)	49.0% (Casteel et al., 2003b)
	Soil	50%	Dermal Contact	Same as above	--	Skin surface area: 3,165 cm <sup>2</sup> (USEPA, 1997a) Soil adherence factor: 0.2 mg/cm <sup>2</sup> (USEPA, 2001a)	3% (USEPA, 2001a)
Adult Utility Pole Worker (ages 18-30 years)	Dislodgeable Residue	100%	Incidental Ingestion	250 days/year (USEPA, 1993a)	$(8 \text{ hrs/day} \times 0.20)/(12 \text{ hrs/day}) = 0.13$ (unitless) (USEPA, 1997b)	0.13 hand loads/day (Gradient, 2001)	29% (Casteel et al., 2003a)
	Dislodgeable Residue	100%	Dermal Contact	Same as above	--	Available hand surface area and forearms: 1,610 cm <sup>2</sup> (USEPA, 1997a)	0.1% (Wester et al., 2004)
	Soil	50%	Incidental Ingestion	Same as above	$(8 \text{ hrs/day} \times 0.20)/(12 \text{ hrs/day}) = 0.13$ (unitless) (USEPA, 1997b)	50 mg/day (USEPA, 1997a; Stanek et al., 1997)	49.0% (Casteel et al., 2003b)
	Soil	50%	Dermal Contact	Same as above	--	Skin surface area: 2,210 cm <sup>2</sup> (USEPA, 1997a) Soil adherence factor: 0.2 mg/cm <sup>2</sup> (USEPA, 2001a)	3% (USEPA, 2001a)

**Note:**

(1) Arsenic complex is the constituent of concern in both dislodgeable residue and impacted soil.

(2) As described in the report, the fraction of intake parameter (F1) quantifies the fraction of daily intake of dislodgeable residue and/or impacted soil, and is based on the proportion of daily waking hours (assumed to be 12 hours/day) that a receptor may be exposed to these media via incidental ingestion.

Dermal exposure to dislodgeable residue and/or impacted soil was quantified as a daily event because it's assumed that dermal uptake can continue post exposure.

USEPA = U.S. Environmental Protection Agency

As = inorganic arsenic complex

-- = not applicable

# Attachment 3

## Calculation of Potential Non-Cancer and Cancer Risks

### Incidental Ingestion and Dermal Contact with Arsenic Complex in Dislodgeable Residue

#### Child Resident, Ages 2-6 Years

#### Incidental Ingestion

##### Average Daily Dose and Non-Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{hand load}$ )	Hand Transfer Efficiency (hand loads/day)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Relative Oral Bioavailability (unitless)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Average Daily Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Subchronic Oral Reference Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Subchronic Hazard Quotient
CCA-Treated Utility Poles	1.60E-01	1.32E+02	2.50E-01	350	0.09	5	0.29	1.00E-03	17.8	1.83E+03	7.42E-06	1.50E-02	4.9E-04

##### Lifetime Average Daily Dose and Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{hand load}$ )	Hand Transfer Efficiency (hand loads/day)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Relative Oral Bioavailability (unitless)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Oral Cancer Slope Factor ( $\text{mg}/\text{kg}\cdot\text{day}$ ) <sup>-1</sup>	Excess Lifetime Cancer Risk
CCA-Treated Utility Poles	1.60E-01	1.32E+02	2.50E-01	350	0.09	5	0.29	1.00E-03	17.8	2.56E+04	5.30E-07	1.50E+00	8.0E-07

#### Dermal Contact

##### Average Daily Dose and Non-Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{day}$ )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Average Daily Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Subchronic Oral Reference Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Subchronic Hazard Quotient
CCA-Treated Utility Poles	1.60E-01	1.32E+02	1.00E-03	350	5	1.00E-03	17.8	1.83E+03	1.14E-06	1.50E-02	7.6E-05

##### Lifetime Average Daily Dose and Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{day}$ )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose ( $\text{mg}/\text{kg}\cdot\text{day}$ )	Oral Cancer Slope Factor ( $\text{mg}/\text{kg}\cdot\text{day}$ ) <sup>-1</sup>	Excess Lifetime Cancer Risk
CCA-Treated Utility Poles	1.60E-01	1.32E+02	1.00E-03	350	5	1.00E-03	17.8	2.56E+04	8.13E-08	1.50E+00	1.2E-07

**Attachment 4**  
**Calculation of Potential Non-Cancer and Cancer Risks**  
**Incidental Ingestion and Dermal Contact with Arsenic Complex in Dislodgeable Residue**  
**Adult Utility Pole Worker, Ages 18-30 Years**

**Incidental Ingestion**

Average Daily Dose and Non-Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{hand load}$ )	Hand Transfer Efficiency (hand loads/day)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Relative Oral Bioavailability (unitless)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Average Daily Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Chronic Oral Reference Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Chronic Hazard Quotient
CCA-Treated Utility Poles	1.60E-01	3.00E+02	1.30E-01	250	0.13	12	0.29	1.00E-03	70.0	4.38E+03	2.30E-06	3.00E-04	7.7E-03

Lifetime Average Daily Dose and Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Hand Surface Area ( $\text{cm}^2/\text{hand load}$ )	Hand Transfer Efficiency (hand loads/day)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Relative Oral Bioavailability (unitless)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Oral Cancer Slope Factor ( $\text{mg}/\text{kg}\text{-day}$ ) <sup>-1</sup>	Excess Lifetime Cancer Risk
CCA-Treated Utility Poles	1.60E-01	3.00E+02	1.30E-01	250	0.13	12	0.29	1.00E-03	70.0	2.56E+04	3.95E-07	1.50E+00	5.9E-07

**Dermal Contact**

Average Daily Dose and Non-Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Skin Surface Area ( $\text{cm}^2/\text{day}$ )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Average Daily Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Chronic Oral Reference Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Chronic Hazard Quotient
CCA-Treated Utility Poles	1.60E-01	1.61E+03	1.00E-03	250	12	1.00E-03	70.0	4.38E+03	2.52E-06	3.00E-04	8.4E-03

Lifetime Average Daily Dose and Cancer Risk

Arsenic Complex in Dislodgeable Residue	Exposure Point Concentration ( $\mu\text{g}/\text{cm}^2$ )	Available Skin Surface Area ( $\text{cm}^2/\text{day}$ )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor ( $\text{mg}/\mu\text{g}$ )	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose ( $\text{mg}/\text{kg}\text{-day}$ )	Oral Cancer Slope Factor ( $\text{mg}/\text{kg}\text{-day}$ ) <sup>-1</sup>	Excess Lifetime Cancer Risk
CCA-Treated Utility Poles	1.60E-01	1.61E+03	1.00E-03	250	12	1.00E-03	70.0	2.56E+04	4.32E-07	1.50E+00	6.5E-07

## Attachment 5

### Calculation of Potential Non-Cancer and Cancer Risks Incidental Ingestion and Dermal Contact with Impacted Soil Child Resident, Ages 2-6 Years

#### Incidental Ingestion

##### Average Daily Dose and Non-Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Soil Ingestion Rate (mg/day)	Relative Oral Bioavailability (unitless)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Average Daily Dose (mg/kg-day)	Subchronic Oral Reference Dose (mg/kg-day)	Subchronic Hazard Quotient
Utility Pole Soil	2.14E+02	5.00E-01	1.00E+02	4.90E-01	350	0.09	5	1.00E-06	17.8	1.83E+03	2.54E-05	1.50E-02	1.7E-03

##### Lifetime Average Daily Dose and Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Soil Ingestion Rate (mg/day)	Relative Oral Bioavailability (unitless)	Exposure Frequency (days/year)	Fraction of Intake (unitless)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose (mg/kg-day)	Oral Cancer Slope Factor (mg/kg-day) <sup>-1</sup>	Excess Lifetime Cancer Risk
Utility Pole Soil	2.14E+02	5.00E-01	1.00E+02	4.90E-01	350	0.09	5	1.00E-06	17.8	2.56E+04	1.82E-06	1.50E+00	2.7E-06

#### Dermal Contact

##### Average Daily Dose and Non-Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Skin Surface Area (cm <sup>2</sup> /day)	Soil Adherence Factor (mg/cm <sup>2</sup> )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Average Daily Dose (mg/kg-day)	Subchronic Oral Reference Dose (mg/kg-day)	Subchronic Hazard Quotient
Utility Pole Soil	2.14E+02	5.00E-01	3.17E+03	2.00E-01	3.00E-02	350	5	1.00E-06	17.8	1.83E+03	1.09E-04	1.50E-02	7.3E-03

##### Lifetime Average Daily Dose and Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Skin Surface Area (cm <sup>2</sup> /day)	Soil Adherence Factor (mg/cm <sup>2</sup> )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose (mg/kg-day)	Oral Cancer Slope Factor (mg/kg-day) <sup>-1</sup>	Excess Lifetime Cancer Risk
Utility Pole Soil	2.14E+02	5.00E-01	3.17E+03	2.00E-01	3.00E-02	350	5	1.00E-06	17.8	2.56E+04	7.82E-06	1.50E+00	1.2E-05

# Attachment 6

## Calculation of Potential Non-Cancer and Cancer Risks Incidental Ingestion and Dermal Contact with Impacted Soil Adult Utility Worker, Ages 18-30 Years

### Incidental Ingestion

Average Daily Dose and Non-Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Soil Ingestion Rate (mg/day)	Relative Oral Bioavailability (unitless)	Exposure Frequency (days/year)	Fraction of Source (unitless)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Average Daily Dose (mg/kg-day)	Chronic Oral Reference Dose (mg/kg-day)	Chronic Hazard Quotient
Utility Pole Soil	2.14E+02	5.00E-01	5.00E+01	4.90E-01	250	0.13	12	1.00E-06	70.0	4.38E+03	3.33E-06	3.00E-04	1.1E-02

Lifetime Average Daily Dose and Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Soil Ingestion Rate (mg/day)	Relative Oral Bioavailability (unitless)	Exposure Frequency (days/year)	Fraction of Source (unitless)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose (mg/kg-day)	Oral Cancer Slope Factor (mg/kg-day) <sup>-1</sup>	Excess Lifetime Cancer Risk
Utility Pole Soil	2.14E+02	5.00E-01	5.00E+01	4.90E-01	250	0.13	12	1.00E-06	70.0	2.56E+04	5.72E-07	1.50E+00	8.6E-07

### Dermal Contact

Average Daily Dose and Non-Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Skin Surface Area (cm <sup>2</sup> /day)	Soil Adherence Factor (mg/cm <sup>2</sup> )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Average Daily Dose (mg/kg-day)	Chronic Oral Reference Dose (mg/kg-day)	Chronic Hazard Quotient
Utility Pole Soil	2.14E+02	5.00E-01	2.21E+03	2.00E-01	3.00E-02	250	12	1.00E-06	70.0	4.38E+03	1.39E-05	3.00E-04	4.6E-02

Lifetime Average Daily Dose and Cancer Risk

Impacted Soil	Exposure Point Concentration (mg/kg)	Fraction of Source (unitless)	Skin Surface Area (cm <sup>2</sup> /day)	Soil Adherence Factor (mg/cm <sup>2</sup> )	Dermal Abs. Fraction (unitless)	Exposure Frequency (days/year)	Exposure Duration (years)	Conversion Factor (kg/mg)	Body Weight (kg)	Averaging Time (days)	Lifetime Average Daily Dose (mg/kg-day)	Oral Cancer Slope Factor (mg/kg-day) <sup>-1</sup>	Excess Lifetime Cancer Risk
Utility Pole Soil	2.14E+02	5.00E-01	2.21E+03	2.00E-01	3.00E-02	250	12	1.00E-06	70.0	2.56E+04	2.38E-06	1.50E+00	3.6E-06