12-2001

A Meta-Analysis and Risk Assessment of Heavy Metal Uptake in Common Garden Vegetables.

Trent David LeCoultre
East Tennessee State University

A Meta-Analysis and Risk Assessment of Heavy Metal Uptake in Common Garden Vegetables

---

A thesis presented to
the faculty of the Department of Environmental Health
East Tennessee State University

In partial fulfillment
of the requirements for the degree
Master of Science in Environmental Health

---

by
Trent David LeCoultre
December 2001

Phillip Scheuerman, Chair
Creg Bishop
John Kalbfleisch

Keywords: Heavy Metal, Meta-Analysis, Risk Assessment, Vegetable, Uptake, Bioaccumulation, Monte Carlo

ABSTRACT

A Meta-Analysis and Risk Assessment of Heavy Metal Uptake in
Common Garden Vegetables
by
Trent David LeCoultre

Peer reviewed literature was searched to identify research pertaining to the uptake of heavy metals (As, Cd, Pb, and Zn) by vegetables (cabbage, carrot, lettuce, and radish). The objectives of this research were to 1) determine the relationship between heavy metal concentrations in the soil and heavy metal concentrations in vegetables and 2) determine the level of risk associated with exposure to heavy metals through ingestion of contaminated vegetables. Highly variable estimates and biologically implausible regression equations resulted from this meta-analysis. Exposure to arsenic through the ingestion of lettuce grown on contaminated soil significantly increases cancer risk, especially in children. Highly variable hazard quotients prevent strong statements concerning toxic effects from exposure to Pb, Cd, or Zn. A more in-depth meta-analysis (multiple-regression and nonlinear curve-fitting) and an upgrade in data reporting standards are recommended.

DEDICATION

This work is most sincerely dedicated to my wife, Amberly. Because of her unfaltering love, devotion, and motivation, I have been able to find within myself the fortitude and wherewithal to acheive my academic goals. Her ambition and drive inspire me. She is my companion, my best friend, and my soul mate. A greater gift has no man.

ACKNOWLEDGEMENTS

I would like to thank the members of my thesis committee Dr. Phil Scheuerman, Dr. Creg Bishop, and Dr. John Kalbfleisch for their participation in my research. I would also like to thank the Environmental Health Departmental secretaries, Christy Hoffman and Sandy Peacock, for going out of their way to help me during my time here.

A special thanks is extended to Gino Begliutti and Doug Dulaney for their comments and advice during the writing of this thesis. Their support and genuine friendship are greatly appreciated. Thanks also to Brian Evanshen and everyone else in the 'zoo' for the help and encouragement they have provided.

Finally, I would like to express my sincere gratitude to my family. Without their encouragement and loving support of my endeavors, academic or otherwise, I would not have accomplished my goals. I will never forget the personal sacrifices they have made to ensure that I succeed. I hope I never take them for granted and I pray that I've made them proud.

CONTENTS

LIST OF TABLES

Table Page

CHAPTER 1

INTRODUCTION

Background

Toxicity of ingested heavy metals has been an important human health issue for decades. The prevalence of contamination from both natural and anthropogenic sources has increased concern about the health effects of chronic low-level exposures. Many researchers have shown that some common garden vegetables are capable of accumulating high levels of metals from the soil (Garcia et al. 1981, Khan and Frankland 1983, Xiong 1998, Cobb et al. 2000). Certain *Brassica* species (cabbage) are hyperaccumulators of heavy metals into the edible tissues of the plant (Xiong 1998). This is an important exposure pathway for people who consume vegetables grown in heavy metal contaminated soil.

Natural and anthropogenic sources of soil contamination are widespread and variable. Heavy metals occur naturally in rocks. Arsenic is found in sulfide ores such as Arsenopyrite (FeAsS), cadmium is associated with sphalerite, and lead is found in many ores and is the natural byproduct of radioactive decay of uranium^{206} and other elements (ATSDR 1999b). Anthropogenic sources of heavy metal contaminants are more likely the cause of the higher more toxic concentrations in soil. Sources may include mining and smelting of ores, electroplating operations, fungicides and pesticides, sewage and sludge from treatment plants, and the burning of fossil fuels (John and VanLaerhoven 1972; Woolson 1973; Boon and Soltanpour 1992; Cobb et al. 2000).

Certain plants can accumulate heavy metals in their tissues. Uptake is generally increased in plants that are grown in areas with increased soil concentrations. Many people could be at risk of adverse health effects from consuming common garden vegetables cultivated in contaminated soil. Often the condition of garden soil is unknown or undocumented; therefore, exposure to toxic levels can occur. (Xu and Thornton 1985) suggest that there are health risks from consuming vegetables with elevated heavy metal concentrations. The populations most affected by heavy metal toxicity are pregnant women or very young children (Boon and Soltanpour 1992). Neurological disorders, CNS destruction, and cancers of various body organs are some of the reported effects of heavy metal poisoning (ATSDR 1994; ATSDR 1999a; ATSDR 1999b; ATSDR 2000). Low birth weight and severe mental retardation of newborn

children have been reported in some cases where the pregnant mother ingested toxic amounts of a heavy metal (Mahaffey et al. 1981).

Objectives

The objectives of this research were to 1) determine the relationship between heavy metal concentrations in the soil and heavy metal concentrations in vegetables and 2) determine the level of risk associated with exposure to heavy metals through ingestion of contaminated vegetables.

CHAPTER 2
LITERATURE REVIEW

Arsenic

Uses, Sources, Fate, and Transport

Approximately 90% of all arsenic produced in the United States is used to preserve lumber. Chromated copper arsenate (CCA) is the preservative used to retard the rotting and deterioration of wood exposed to weathering and insects (ATSDR 2000). Arsenic has also been used for decades as an ingredient in pesticides and fungicides. Arsenic acid ($As_2O_3 \cdot H_2O$) is used as a weed killer and in leaf desiccation of cotton plants (Woolson 1973). Arsenic is also used in the smelting of ores and in electroplating (Cobb et al. 2000). Atmospheric fallout from smelting and other manufacturing processes can be a significant source of As in the environment.

Arsenic is typically immobile in agricultural soil and, therefore, accumulates in the upper soil horizons (ATSDR 2000). Janssen et al. (1997) used a regression analysis of pH, organic matter content, clay content, iron oxide content, aluminum oxide content, and cation exchange capacity versus As mobility to determine how each parameter affected As mobility in soil (Janssen et al. 1997). They found that iron oxide content was the only soil characteristic significantly positively correlated with As mobility. Arsenic mobility is more dependant on ligand exchange mechanisms, particularly with iron oxides, than the pH-dependant dissolution-precipitation reactions that regulate the movement of most other metals in the soil (Darland and Inskeep 1997; Jones et al. 1997). Darland and Inskeep (1997) found that arsenate (AsO_4) transport through sand containing free iron oxides was very slow at pH 4.5 and 6.5, and significantly more rapid at pH 8.5. They suggested that liming soil to increase the pH and promote metal precipitation to decrease metal mobility, may actually facilitate the movement of As.

Arsenates (As(V)) are more toxic and more mobile in the soil than arsenites (As(III)) (McGeehan 1996). Some aquatic organisms and soil bacteria can reduce As(V) to As(III), increasing its toxicity and its mobility in the soil (Honschopp et al. 1996; Turpeinen et al. 1999; ATSDR 2000). Under reducing conditions, such as temporarily flooded or saturated soil, inorganic arsenicals may be methylated to produce the less toxic organic forms, monomethyl arsonic acid (MMA) or dimethyl arsinic acid (DMA) (Honschopp et al. 1996).

Toxicity

Inorganic arsenic is highly toxic, and acute exposures cause vomiting, diarrhea, and gastrointestinal hemorrhage. Death can occur at doses that range from 22 to 121 mg As/kg of body weight. For example, 2 people in a family of 8 died after 1 week of drinking water that contained 110 ppm As (2 mg As/kg/day) (Armstrong et al. 1984). Death usually results from fluid loss and circulatory collapse. Chronic, low-dose exposure causes several adverse health effects. Cough, sputum, rhinorrhea, and sore throat have been reported by people exposed to 0.03-0.05 mg/kg/day. Because people in areas of Taiwan receive doses of 0.014-0.065 mg/kg/day in drinking water, "Blackfoot disease" is endemic. Blackfoot disease is decreased circulation in the extremities, which leads to necrosis and gangrene. Although there are limited data to support developmental toxicity of arsenic, Golub et al. (1998) used animal models to show a dose-dependant increase in stillbirths and postnatal growth retardation in females chronically exposed before and during pregnancy (Golub et al. 1998). Anemia and luecopoenia have also been reported at acute, intermediate, and chronic exposure levels. Arsenic exposure causes several dermal effects. Generalized hyperkeratosis and the formation of hyperkeratotic warts and corns on the palms and the soles of the feet are caused by chronic arsenic ingestion. Discoloration of the skin of the face, neck, and back also can occur. Squamous cell carcinomas can form from hyperkeratotic warts and corns. Basal cell carcinomas are also caused by arsenic exposure but they do not form from the warts or corns. Chronic low-level exposure also increases the incidence of internal cancers. Cancers of the bladder, kidney, liver, lung, and prostate have been documented in animal studies. The Department of Health and Human Services (DHHS), the International Agency for Research on Cancer (IARC), the United States Environmental Protection Agency (EPA), and the National Toxicology Program (NTP) have all classified inorganic arsenic as a known human carcinogen (ATSDR 2000).

The EPA has determined that the reference dose (RfD) for inorganic arsenic is 0.0003 mg As/kg/day (Anonymous2001b). This RfD is derived from the NOAEL of 0.009 mg As/L using an uncertainty factor of 3. The NOAEL was established based on the occurrence of skin lesions in humans exposed to As. Skin lesions were the most sensitive endpoint. Tseng et al. (1968) conducted the principle study used to determine the NOAEL. The arithmetic mean of the concentration of arsenic in the well water of the control group was used as the NOAEL. To

derive the RfD, the NOAEL (0.009 mg/L) was first converted to mg As/kg/day. Assumptions used included consumption of 4.5L water/day, 55 kg body weight, and a food concentration of 0.002 mg As/day. The resulting NOAEL was 0.0008 mg As/kg/day. An uncertainty factor of 3 was used to account for limited data and the possible exclusion of sensitive individuals (ATSDR 2000).

Organic arsenicals are less toxic than the inorganic forms. The 2 primary forms of organic arsenic are monomethyl arsonic acid (MMA) and dimethyl arsinic acid (DMA). MMA and DMA are primarily used as agricultural pesticides. Data about the toxicity of organic forms of arsenic are limited; however, based on available data, organic arsenic is not toxic (ATSDR 2000).

Cadmium

Uses, Sources, Fate, and Transport

Although cadmium is a naturally occurring element, it is rarely found as a pure metal in nature. It is generally associated with oxygen, chlorides, sulfates, and sulfides. Cadmium is often a byproduct of the extraction of Pb, Zn, and Cu from their respective ores (ATSDR 1999a). Carbonaceous shale, coal, and other fossil fuels are also sources of Cd. Volcanism is the largest natural source of Cd (ATSDR 1999a). Anthropogenic sources of Cd in the soil and groundwater include the use of commercially available fertilizers and the disposal of sewage sludges as soil amendments (Baker et al. 1979; Garcia et al. 1979; Kosla 1986; Peles et al. 1998; Gallardo-Lara et al. 1999).

Cadmium can accumulate in high concentrations in soils. John et al. (1972) report a Cd concentration of 95 ppm in a sample collected near a battery smelter near Vancouver, BC, Canada. Cadmium is recalcitrant in the soil profile, particularly in the surface horizons (John et al. 1972; Khan and Frankland 1983). Most soil profiles have an A horizon, which is primarily topsoil composed of decaying organic matter such as leaves and grass, and a B horizon, which is composed of smaller clay-sized particles. In general, heavy metal concentrations are higher in the B horizons than in the A horizons (Lee et al. 1997). Heavy metals tend to accumulate in the clay fraction of most soil profiles (Boon et al. 1992; Lee et al. 1997). Boon et al. (1992) concluded that the concentration of heavy metals in soil is dependant on clay content because clay-sized particles have a large number of ionic binding sites due to the higher amount of

surface area. This results in the immobilization of heavy metals, and there is very little leaching through the soil profile (Khan and Frankland 1983). Immobilization can increase the Cd concentration of the soil and ultimately lead to the increased toxicity of the contaminated soil. Higher soil Cd concentrations can result in higher levels of uptake by plants (John et al. 1972). However, specific soil properties can have a significant effect on the amount of heavy metal assimilated by the plant (John and VanLaerhoven 1972; Peles et al. 1998).

Increased levels of Ca^{2+} can decrease the amount of Cd that is assimilated by plants (Larlson et al. 2000). Because of their similar size, Ca(II) is almost indistinguishable from Cd(II) (Ochiai 1995). A higher affinity for the essential trace metal Ca results in the decreased uptake of Cd into the plant. A similar relationship exists between P and Cd. John et al. (1972) showed that the addition of 1000 ppm of phosphorus to a Cd contaminated soil deceased the concentration of Cd 43% in the roots of oats. Trace metal deficiencies in plants have been associated with increases in heavy metal uptake (Khan and Frankland 1983).

Soil pH significantly influences heavy metal concentrations in both soil and plant tissues. The effect of soil pH on mobility of heavy metals is a well-researched topic (Cataldo et al. 1981; Chen et al. 1997; Peles et al. 1998; Li and Wu 1999). As the soil pH decreases, metals are desorbed from organic and clay particles, enter the soil solution and, become more mobile (Li and Wu 1999). When the pH is higher (i.e., >7), metals remain adsorbed and what metals in solution precipitate out in the form of salts (Chen et al. 1997). Variability in pH also affects the amount of Cd assimilated by the plant. John and VanLaerhoven (1972) showed that higher pH resulted in lower Cd uptake. Peles et al. (1998) concluded that the addition of lime to contaminated soils (essentially increasing the pH) decreased the uptake of heavy metals. In unlimed soils *Ambrosia trifida* accumulated 13.6 μg Cd g^{-1} of tissue and in limed soils *A.trifida* accumulated 2.5 μg Cd g^{-1} of tissue.

Khan and Frankland (1983) reported that extremely high concentrations (180 μg g^{-1}) of Cd in soil adversely affected plant development. In their research, radish plants were grown on soils contaminated with Cd and Pb. Within 3 weeks of planting, all plants that were grown in soil contaminated with 1000 μg Cd g^{-1} were dead. The concentrations of Cd in the soil that produced a 50% inhibition in growth were higher at the seedling stage than at the edible stage. John et al. (1972) also showed that plant size and yield were reduced when 50 mg Cd (dosed as $CdCl_2$) was added to 500g of soil. In both studies, chlorosis of the leaves was reported. Khan

and Frankland (1983) suggest additive effects from the application of Cd and Pb at the same time. They document a considerable reduction in growth when Cd was added at 50 μg g^{-1} and Pb was added at 1000 μg g^{-1} (Khan and Frankland 1983).

Toxicity

The Agency for Toxic Substances and Disease Registry (ATSDR) reports that the average American ingests about 30 μg Cd/day (ATSDR 1999a). However, only about one tenth of this amount is actually absorbed into the tissues. Intake of Cd can double if one smokes cigarettes because each cigarette contains about 2 μg Cd. Acute doses (10-30 mg/kg-day) of cadmium can cause severe gastrointestinal irritation, vomiting, diarrhea, and excessive salivation, and doses of 25 mg CdI_2/kg body weight can cause death.

Low-level chronic exposure to Cd can cause adverse health effects including gastrointestinal, hematological, musculoskeletal, renal, neurological, and reproductive effects. The main target organ for Cd following chronic oral exposure is the kidney (ATSDR 1999a). Because cadmium tends to accumulate in the kidneys, the EPA has based the RfD for cadmium on the concentration of the metal in the human renal cortex (EPA 1994a). The highest Cd level in the renal cortex that does not cause significant proteinuria is 200μg Cd/g (EPA 1994a; ATSDR 1999a). A toxicokinetic model was used to determine the no-observable-adverse-effect-level (NOAEL) dose that would result in a renal cortex concentration of 200 μg Cd/g. To use the model, it was assumed that 0.01% of the daily Cd body burden is excreted in the urine or feces and that 2.5% of the Cd in food and 5% of the Cd in water are actually absorbed into the body tissues. Based on these assumptions, the model estimate of the NOAEL is 0.01 mg Cd/g for food and 0.005 mg Cd/g for water. The RfD is determined using the NOAEL and an uncertainty factor of 10. The uncertainty factor is used to take into account biological variability. EPA has established RfDs for Cd of 0.001 mg Cd/kg/day for food and 0.0005 mg Cd/kg/day for water. These amounts represent an estimated daily oral exposure that is likely not to cause adverse health effects (EPA 1994a).

The ATSDR concludes that there is insufficient evidence to determine whether oral exposure to Cd increases the risk for cancer. However, the United States Department of Health and Human Services (DHHS) has stated that cadmium compounds may be carcinogenic (ATSDR 1999a). The International Agency for the Research on Cancer (IARC) has classified

Cd and Cd salts as possible human carcinogens. This classification is based on human lung cancer data from occupational inhalation (ATSDR 1999a).

Lead

Uses, Sources, Fate, and Transport

Lead is a naturally occurring heavy metal. It is seldom found in its elemental form; however, it is part of several ores including its own (galena, PbS). Pb is also a product of the radioactive decay of uranium^{206}, thorium^{208}, and actinium^{207} (Sax and Lewis Sr. 1987). Pb has many industrial and commercial uses. It is used in the production of ammunition, as solder, in ceramic glass, and the production of batteries (ATSDR 1999b). Other sources of Pb in the environment include automobile exhaust, industrial wastewater, wastewater sludge, and pesticides (Balba et al. 1991). Because of its high toxicity, the use of lead in some products has been discontinued. Lead is no longer used in house paint because of the concern about the toxic effects of the accidental ingestion of paint chips or the inhalation of aerosolized lead from decaying paint. In 1991, the amount of Pb was greatly reduced in gasoline (Anonymous2001a). Most of the environmental lead contamination comes either from landfill leachate or from airborne lead particles deposited onto the soil (ATSDR 1999b).

Pb behavior in soil is similar to Cd behavior in soil. However, Khan and Frankland (1983) showed that Pb was less mobile in soil than Cd. Very little of either Pb or Cd was leached through the soil profile. In fact, more Pb and Cd were removed from the soil by plants than was leached through the profile (Khan and Frankland 1983). Several factors may influence the content and distribution of heavy metals in soil. Some of these factors are parent material, organic matter, particle size distribution, drainage, pH, type of vegetation amount of vegetation, and aerosol deposition (Lee et al. 1997).

Heavy metals, including Pb, tend to accumulate in the clay fraction of the soil profile (Boon and Soltanpour 1992; Lee et al. 1997; Li and Wu 1999). Strong ionic bonds are formed between the cation and the clay particle. Acidic conditions will cause desorption of these cations into solution making them available for uptake by plants. Desorption to the soil solution also increase cation mobility through the profile (John and VanLaerhoven 1972; Cataldo et al. 1981; Chen et al. 1997; Peles et al. 1998; Li and Wu 1999).

Decreased growth and yield have been observed in plants grown in Pb contaminated soils. Balba et al. (1991) showed a significant decrease in plant biomass yield with increasing Pb treatments that varied with soil type. The highest adverse effects were on those plants grown in soils with high clay content. Khan and Frankland (1983) also showed decreased plant growth and yield in soils with Pb contamination.

Toxicity

Ninety-nine percent (99%) of the lead that enters the adult human body and 33% that enters a child's body is excreted in about 2 weeks (ATSDR 1999b). Because of this, lead poisoning is a greater concern in children. Most of the accumulated lead is sequestered in the bones and teeth. This causes brittle bones and weakness in the wrists and fingers. Lead that is stored in bones can reenter the blood stream during periods of increased bone mineral recycling (i.e., pregnancy, lactation, menopause, advancing age, etc.). Mobilized lead can be redeposited in the soft tissues of the body and can cause musculoskeletal, renal, ocular, immunological, neurological, reproductive, and developmental effects (Todd et al. 1996; ATSDR 1999b).

Replacement of calcium in the bone and muscle tissue by lead can impair normal bone growth, and bone density and calcium content can decrease. High exposures (i.e., > 30 mg Pb/kg/day) to lead cause muscle weakness, cramps, and joint pain. Impaired kidney function and a weakened immune system can also result from over-exposure to Pb. Various reproductive effects including decreased pregnancy rate, ovarian damage, testicular damage, testicular atrophy, cellular degeneration, and irregular estrous cycles have been shown in animal studies (ATSDR 1999b). Renal toxicity is now used as a biochemical and physiologic marker of chronic subclinical lead toxicity (Todd et al. 1996).

Although over-exposure to lead causes serious health effects in adults, especially pregnant women, the toxicity of lead is greatly increased in children. The Centers for Disease Control (CDC) report that nearly 1 million children in the United States have blood-lead levels that exceed the 10 μg Pb/dL level of concern (ATSDR 1999b). Dirt, dust, and lead-based paint chips from old houses can be sources of increased exposure to children. Because lead can cross the placenta, prenatal exposure can be significant. A pregnant woman and her fetus will have virtually the same blood-lead level (Todd et al. 1996). *In utero* exposure can lead to low birth weight, premature birth, or miscarriage. Lead can also be transmitted through breast milk.

Anemia, colic, impaired vitamin D metabolism, and growth retardation result from lead exposure during infancy or early childhood. Lead exposure is also associated with several neurological effects, such as delayed neurological development, cognitive impairment, IQ deficits, and effects on general brain function. Some of these effects are irreversible and continue into adulthood (ATSDR 1999b). The United States Environmental Protection Agency has classified inorganic lead as a possible human carcinogen. Although human data are insufficient, there are significant increases in renal tumors with high (i.e., >500 ppm) exposure of lead based on animal studies (EPA 1991).

Zinc

Uses, Sources, Fate, and Transport

Zinc can be found in nearly all soils. It is present in most rocks and is weathered out and deposited into the soil. Zinc is also released by thermal outgassing and other volcanic events. Fallout from such events can be a significant source of zinc in soils and plants. Anthropogenic release is the primary source of zinc in the environment. Zinc is released from industrial and manufacturing facilities in wastewater effluent or from incinerators. Zinc is used as a constituent in several alloys, including brass, bronze, die-cast metals, and is combined with copper for the production of US pennies. Zinc is also used in electroplating, smelting, and ore processing (ATSDR 1994). Mine tailings and drainage from mines can contain high concentrations of zinc (Cobb et al. 2000).

The fate and transport of zinc (Zn^{+2}) in the environment is dependant on cation exchange capacity, pH, organic matter content, nature of complexing ligands, and the concentration of the metal in the soil. As pH increases, there is an increase in negatively charged binding sites on soil particles, which facilitates the adsorption of zinc ions and removal from solution (ATSDR 1994). The Zn concentration in the soil and clay content are positively correlated (Lee et al. 1997). The most common form of zinc in anaerobic soils is the insoluble zinc sulfide. Therefore, mobility is limited in anaerobic conditions. Zinc mobility increases with low pH (e.g. < 7) under oxidizing conditions and low cation exchange capacity (ATSDR 1994). The presence of competing metal ions and organic ions such as humic material may cause the adsorption of Zn^{+2} ions to the soil, particularly in soils with an elevated pH, via ligand exchange reactions (ATSDR 1994). These

reactions reduce the solubility of zinc in the soil solution and, therefore, reducing its mobility and limit its bioavailability.

Toxicity

Of the metals considered in this research, zinc is the least toxic. Zinc is an essential element in the human diet because it is required to maintain the proper functions of the immune system. It is also important for normal brain activity and is fundamental in the growth and development of the fetus. Zinc deficiency in the diet may be more detrimental to human health than too much zinc in the diet (ATSDR 1994). Although the average daily intake of zinc in the United States is 7-16.3 mg Zn/day, the Recommended Daily Allowance (RDA) for zinc is 15 mg Zn/day for men and 12 mg Zn/day for women (ATSDR 1994). To compensate for Zinc deficiency some people use Zinc supplements. Ingestion of large doses (390 mg Zn/kg/day for 3-13 days, or about 27g Zn/day) of Zn can cause death (ATSDR 1994). If doses 10-15 times higher than the RDA are taken over a long period, anemia and damage to the pancreas and kidney can develop. Vomiting, diarrhea, abdominal cramping, and, in some cases, intestinal hemorrhage can occur from long-term exposure to high (i.e., >85 mg/kg/day) doses of zinc. Murphy (1970) documented a 16-year-old boy who had ingested 12g of elemental Zn over a 2-day period (86 mg Zn/kg/day). He presented with lightheadedness, lethargy, staggering gait, and decreased motor skills. These high oral doses of Zn can also impair the immune system (Murphy 1970).

All of these adverse health effects are from oral doses greater than 85 mg Zn/kg/day and are usually related to either accidental ingestion (i.e., drinking water from galvanized buckets) or through improper use of nutritional supplements. Food may contain from 2 ppm Zn in leafy vegetables up to 29 ppm Zn in poultry, fish, and other meats (ATSDR 1994). The most commonly reported health effects from high oral exposure (i.e., >85 mg/kg/day) to Zn are anemia (and copper anemia), caused by Zn displacing iron and copper in the blood, and decreased HDL cholesterol, which can lead to cardiac disease. Zinc is not a human carcinogen.

Risk Assessment

Section 121(d)(1) of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (42 U.S.C. 9601 et seq.) states that remediation of hazardous waste sites must be to the degree that ensures the protection of human health and the environment. The Agency for Toxic Substances and Disease Registry was statutorily formed (CERCLA §104(i)), in part, to carry out human health assessments for hazardous waste sites. CERCLA requires that health assessments include a preliminary assessment of risk to human health, identification of potential exposure pathways, the characteristics of the affected community, the short and long-term health effects for each chemical, and analysis of morbidity and mortality data on diseases caused by exposure to the contaminant (CERCLA §104(i)(6)(F)).

The United States Environmental Protection Agency published the *Risk Assessment Guidance for Superfund (RAGS) Volume I: Human Health Evaluation Manual (Part A)* in an effort to comply with CERCLA requirements (EPA 1989). This manual is designed for use by EPA contractors, state agencies, federal agencies, and individuals conducting human health risk assessments. It contains information on the human health risk assessment process used in CERCLA mandated remedial investigations and feasibility studies (RI/FS). The purpose of the RI/FS is to obtain information, including health risk data, needed to determine the appropriate remedial action for a particular site (EPA 1989). RAGS has become the predominant regulatory guidance document used for conducting risk assessments. The United States Department of Energy-Oak Ridge Operations (DOE-ORO) has developed a document consistent with and, in part, based on RAGS entitled *Guidance for Conducting Risk Assessments and Related Risk Activities for the DOE-ORO Environmental Management Program* (DOE 1999). Risk assessment is defined by DOE as a tool used by decision-makers to assess the potential adverse human health effects that may result from exposure to contaminants at a particular site (DOE 1999).

Risk assessment is done in 4 steps or stages. EPA (1989) and DOE-ORO (1999) designate the stages as: 1) data compilation and evaluation, 2) exposure assessment, 3) toxicity assessment, and 4) risk characterization (EPA 1989; DOE 1999). The type, quality, and availability of data from a particular site will determine the extent of the investigation. The data evaluation steps outlined in RAGS are given in Table 1.

Table 1: Data Evaluation Steps Outlined in the USEPAs Risk Assessment Guidance for Superfund (RAGS) (EPA 1989).

1	Gather all data available from the site investigation and sort by medium
2	Evaluate the analytical methods used
3	Evaluate the quality of data with respect to sample quantitation limits
4	Evaluate the quality of data with respect to qualifiers and codes
5	Evaluate the quality of data with respect to blanks
6	Evaluate tentatively identified compounds
7	Compare potential site-related contamination with background
8	Develop a set of data for use in the risk assessment
9	If appropriate, further limit the number of chemicals to be carried through the risk assessment

Site-specific criteria may require alteration to any or all of these steps. An exposure assessment is defined by EPA and DOE-ORO as the determination or estimation (quantitative or qualitative) of the magnitude, frequency, duration, and route of exposure for each potential or actual population to be evaluated in the risk assessment (EPA 1989; DOE 1999). This step should include a site characterization, identification of the potential exposure pathways and quantification of actual or potential exposure (DOE 1999). Toxicity assessments have 2 parts, hazard identification and dose-response evaluation. The hazard identification step is done to establish causation of adverse health effects and exposure to a particular agent. Dose-response evaluations are done to determine the relationship between the dose of contaminant and the incidence of adverse health effects. Toxicity values such as RfD and reference concentration (RfC) are used in this evaluation. These values are derived from epidemiological studies and animal data (EPA 1989).

The 4 step in risk assessment is the risk characterization. This step involves the compilation of data from the previous steps and its incorporation into a mathematical model to derive a value for risk. Models can change significantly depending on several factors. These factors may include daily intake value, exposure level, RfD, RfC, specific data concerning the people exposed (e.g. age, body weight, inhalation rate, etc.), chemical specific constants such as uptake factors, absorption factors, and residency time. Assumptions and generalizations are used because it is impractical to determine the exact values for each site, each chemical, and each

potentially exposed individual. Uncertainty factors are incorporated into the model to account for these issues and the variability of the toxic effects of chemicals.

A distinction is made in the methodology for assessing cancer and noncancer risk. For determining the probability of developing cancer from exposure to a carcinogen, a slope factor is used in the model. The slope factor describes the dose-response relationship. The slope factor is directly related to intake and risk. Risk is expressed as a unitless probability of developing cancer (EPA 1989). The potential for developing noncarcinogenic effects is expressed as a ratio of time weighted exposure level and a reference dose or concentration. This ratio is called a hazard quotient (EPA 1989).

The technique of exposure assessment and risk characterization can be applied to any exposure scenario. Pitten et al. (1999) performed a risk assessment of uptake of arsenic from contaminated soil at a former military base. They found that there was low arsenic accumulation in plant material compared to arsenic levels in the soil; therefore, there was low risk (Pitten et al. 1999). Edberg (1996) evaluated the health risk associated with biologically contaminated drinking water (Edberg 1996). Using an equation to evaluate the health effect of the microorganisms in water, Edberg determined risk based on the number of microbes, their virulence, and the immune status of the host. Boffetta et al. (2000) compared childhood cancer risk and adult lung cancer risk after childhood exposure to side-stream tobacco smoke. In this case, meta-analysis was used to combine odds ratios and relative risks to extrapolate the effect of interest.

According to the EPA (EPA 1989), no risk characterization (or risk assessment) should be considered complete until text containing a full description and interpretation of the derived risk is included. This should be written in terms that can be easily understood by administrators or other officials who may not be trained in areas such as epidemiology, toxicology, or risk assessment.

Meta-Analysis

The technique of quantitatively combining, synthesizing, and summarizing data and results from different studies is known as meta-analysis (Putzrath and Ginevan 1991; Hasselblad 1995). This type of analysis was first used for the social sciences but has been used in a variety

of fields including environmental health, epidemiology, and risk assessment (Putzrath and Ginevan 1991; Blair et al. 1995; Hasselblad 1995).

Table 2: Situations Where Meta-analysis May be Useful as Outlined by Blair et al. (1995).

1	When sources of heterogeneity are to be examined.
2	When the relationship between environmental exposures and health effects is not clear.
3	When refinement of the estimate of an effect is important.
4	When there are questions about the generalizability of results.
5	When it is clear that there is a hazard, but no indication of its magnitude.
6	When information beyond that provided by individual studies or narrative review is needed.

The selection of the studies to be included in the meta-analysis should be done within the confines of specific criteria (Blair et al. 1995). The areas that should be included in the determination of the eligibility of a study are study design, multiple studies of the same overlapping populations, study quality, statistical properties, and publication bias (Blair et al. 1995). Because of the scarcity of data that meet all of the predefined criteria, studies should only be excluded if there are major problems in methodology, design, or analysis (Blair et al. 1995). Homogeneity of effects between studies is necessary for effective analysis. For example, it is not logical to compare studies reporting only plant growth inhibition from metal exposure to studies reporting plant metal concentrations with no measure of plant weight or dimension. Studies included in the meta-analysis must be representative of data from the same universe (Putzrath and Ginevan 1991). Once homogeneity of the selected studies has been established or heterogeneity has been addressed, data combination and analysis can begin.

Hasselblad (1995) discussed ways to quantitatively combine environmental health data. The first method described was the combination of *P*-values. This method could be used to determine if there is any significant difference in the effects of exposure. Combining *P*-values could be problematic if one or more studies in the meta-analysis do not report an exact *P*-value (i.e., $P<0.05$). In this case, a *P*-value of 0.05 could be used for an individual study and would be considered conservative (Hasselblad 1995). Hasselblad (1995) identifies 5 methods for combining *P*-values as a hypothesis test. An inverse variance weighted technique can be used to pool estimates of some effect of exposure from different studies. This method involves the summation of the weighted inverses of the variations of the effect estimates. Putzrath and

Ginevan (1991) described a similar method for combining pooled variations of estimates where each variation is assumed equally representative of the actual effect and is, therefore, given equal weight. There are often different ways that data and results are reported because of the inherent variability between studies. To account for this variability, effect sizes can be compared. Hasselblad (1995) describes a method to create an outcome measure independent of the scale of measurement in each study. The effect size is determined by dividing the difference of the sample means of the treated and control groups by the estimated standard deviation of a single observation.

Risk assessments should be made based on all available studies concerning the particular focus of the risk assessment. Meta-analysis provides a comprehensive, quantitative summation of similar data to more aptly identify risk. In the past, risk assessments often been based on one representative study, usually one that shows high risk. Meta-analysis can provide a more accurate approximation of the degree of risk (Putzrath and Ginevan 1991).

CHAPTER 3
RESEARCH DESIGN

Inclusion Criteria

The peer-reviewed literature was searched, using PubMed and Infotrac databases, for articles pertaining to heavy metal contamination of soils and uptake by plants with no limitation on publication date. Certain criteria have been set for inclusion into this study. Each study must have evaluated metal uptake of one or more of the following vegetables: lettuce, cabbage, radish, or carrot. Each study must also have investigated vegetable uptake of at least one of the following metals: cadmium, arsenic, lead, or zinc. Concentrations of metals in plant tissue must be reported in the article or obtainable from the author. All articles must have reported soil metal concentrations or the dosed metal concentrations for each experimental condition. Detailed soil analyses (i.e., pH, organic matter content, cation exchange capacity, soil type, etc.) were preferred but not required.

Table 3: Studies That Have Been Included Into the Meta-analysis.

Author	Plant-Metal
(Cobb et al. 2000)	Arsenic-Lettuce
	Arsenic-Radish
	Cadmium-Lettuce
	Cadmium-Radish
	Lead-Lettuce
	Lead-Radish
	Zinc-Lettuce
	Zinc-Radish
(Khan and Frankland 1983)	Cadmium-Radish
	Lead-Radish
(Xu and Thornton 1985)	Arsenic-Lettuce
	Arsenic-Carrot
(Boon and Soltanpour 1992)	Cadmium-Lettuce
	Lead-Lettuce
(Nwosu et al. 1995a)	Cadmium-Lettuce
	Cadmium-Radish
	Lead-Lettuce
	Lead-Radish
(Xiong 1998)	Lead-Cabbage
(Helgesen and Larsen 1998)	Arsenic-Carrot
(Carbonell-Barrachina et al. 1999)	Arsenic-Radish
	Arsenic-Radish
(Jinadasa et al. 1997)	Cadmium-Lettuce
	Cadmium-Cabbage
(Garcia et al. 1981)	Lead-Lettuce
	Cadmium-Lettuce
	Zinc-Lettuce
	Lead-Radish
	Cadmium-Radish
	Zinc-Radish
(De Pieri et al. 1997)	Cadmium-Cabbage
	Lead-Cabbage
	Cadmium-Lettuce
	Lead-Lettuce
	Zinc-Lettuce
(Sloan et al. 1997)	Cadmium-Lettuce
	Lead-Lettuce
	Zinc-Lettuce
(Davies 1978)	Lead-Radish
(Haghiri 1973)	Cadmium-Lettuce
	Cadmium-Radish
	Cadmium-Radish
(Alloway et al. 1988)	Cadmium-Cabbage

[a] Study numbers were assigned based on order of inclusion. Missing numbers indicate the exclusion of those studies.
[b] Block numbers were assigned beginning with the first study and were based on individual plant-metal combinations.

Database Compilation

Data from all qualifying studies were compiled into a Microsoft Excel Spreadsheet (database). Data extracted included the name of metal, form of the metal, source of the contamination, dosed concentration, plant type, plant tissue, metal concentration in the plant, method of detection, and soil parameters. Data in the spreadsheet were organized by study. Data were also grouped by plant type and metal (e.g., cadmium radish, lead lettuce, zinc carrot).

Meta-Analysis

Soil or dosed concentrations and plant concentrations were analyzed using regression analyses. The resulting R^2-values represented the fraction of the variation of the plant-metal concentrations that can be explained by the variation in soil-metal concentrations. The slopes of the lines, θ_j, were combined using the inverse variance weighted method (Hasselblad 1995) (Equation 1). Where m = number of studies in a group, j = study number, θ_j = slope of the regression line from study j, and w_j = 1 / Variance [θ_j], in this case, the standard deviation of the slopes.

$$\theta = \frac{\left[\sum_{j=1}^{m} w_j \theta_j\right]}{\left[\sum_{j=1}^{m} w_j\right]} \qquad \text{Equation 1}$$

The resulting θ can be used in a pooled regression equation to extrapolate a plant concentration from a given soil concentration. The y-intercepts were pooled and weighted using their respective standard deviations. Equation 1 was also used to pool R^2-values as reported in Table 5. R^2-values (θ) were weighted with the inverse of the square of the S-values from the regression output. These S-values represent the standard error of the points about the regression line (Moore 1995). The variance of the pooled R^2-values can be determined using the Equation 2. Background concentrations for metals in plants can be determined using Equation 3. Where θ = Combined slope from a particular group and β = Pooled y-intercepts from the studies in the groups.

$$Var(\theta) = \frac{1}{\left[\sum_{j=1}^{m} w_j\right]} \qquad \text{Equation 2}$$

$$\textit{Background metal concentration} = \textit{Analytical detection limit}\ (\theta) + \beta \qquad \text{Equation 3}$$

Risk Assessment

Risk was estimated using equations (Equation 4) from the U.S. Environmental Protection Agency's (USEPA) Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (1989). Values used in Equation 4 were taken from the Environmental Protection Agency's Exposure Factors Handbook (1997).

$$Intake(mg/kg-day) = \frac{CF \times IR \times FI \times EF \times ED}{BW \times AT} \qquad \text{Equation 4}$$

Where *CF*= Contaminant concentration in food (mg/kg), *IR*= Ingestion rate (kg/meal), *FI*= Fraction ingested from contaminated source (unitless), *EF*= Exposure frequency (meals/year), *ED*= Exposure duration (years), *BW*= Body weight (kg), and *AT*= Averaging time (period over which exposure is averaged – days).

Table 4: Mean Per Capita Intake Rates (As Consumed) For Vegetables (EPA 1997)

Vegetable	**Average Daily Consumption (grams/kg body weight-day)[a]**	**Average Quantity Consumed Per Meal (g/meal)[a]**
Cabbage	0.0936 ± (0.0039)[b]	68 ± 45
Carrot	0.1735 ± (0.0042)	43 ± 40
Lettuce	0.2123 ± (0.0059)[c]	65 ± 59
Radish	0.0016 ± (0.0002)[d]	0.78[e]

[a] Values are reported as mean ± standard error
[b] For red and green varieties of cabbage
[c] For head varieties of lettuce
[d] For radish roots
[e] Data were unavailable for radishes. The value listed was extrapolated from the Average Daily Consumption by assuming a BW of 70 kg and 1 radish eating occasion every 7 days.

When the average quantity consumed per meal is provided, as it is for cabbage, carrot and lettuce, ingestion rates (kg/meal) can be calculated by multiplying the quantities consumed by the number of meals per day. After this is done, the products can be inserted into equation 4. For radishes, quantity consumed per meal was not provided in the Exposure Factors Handbook. In this case, quantity consumed per meal was extrapolated from the average daily consumption by assuming a body weight of 70 kg and a radish consumption rate of one day per week (See Equation 5). Where *QC*=Quantity consumed per eating occasion (g/meal), *ADC*=Average daily consumption (g/kg body weight-day), *BW*=Body weight (kg), and *M*=Eating occasions per day (meal/day).

$$QC(g/meal) = \frac{ADC \times BW}{M} \qquad \text{Equation 5}$$

EPA recommends the fraction ingested from the contaminated source (FI) in Risk Assessment Guidance for Superfund (RAGS) (EPA 1989). This variable represents the fraction of consumed vegetables that come from home gardens. The average value is 0.25, and the 'worst-case' value is 0.4. EPA states that the 'worst-case' value can be used to represent the upper 95th percentile. This was the value used in my calculations. Exposure frequency (EF) describes the number of meals per year in which the vegetable is consumed. This number can be extrapolated from the Exposure Factors Handbook values reported in Table 1 using Equation 6. Where *M*=Meals per day (meals/day), *ADC*=Average Daily Consumption (g/kg body weight-day), *BW*=Body weight (kg), and *QC*=Quantity consumed per eating occasion (g/meal).

$$M = \frac{ADC \times BW}{QC} \qquad \text{Equation 6}$$

The number of years someone is potentially exposed is used in this calculation as the exposure duration (ED). Individual body weights (BW) are also considered. Usually in exposure assessments, 70 kg is used as an average adult body weight. The Exposure Factors Handbook lists average body weights based on age and gender. Exposure is averaged over a specified period. In general, this period, or averaging time (AT), is the product of ED and 365 days/year for noncarcingenic effects and the 25500 days (70 years x 365 days/year) for

carcinogenic effects. When assessing carcinogenic risk, lifetime exposure is assumed. This assumption is justified by the belief that an acute exposure to high concentrations of toxicants is equivalent to a chronic low-dose of toxicant (EPA 1989).

The concentrations of the contaminants in the food (CI) were obtained from data provided in the literature. To allow for a range of possible metal concentrations, a Monte Carlo simulation was performed. A spreadsheet was made incorporating Equation 4 and all of its components. The distribution of CI was assumed lognormal. The mean and standard deviation of CI were derived from the literature that met the inclusion criteria. Using @Risk (Palisade Corporation, 2000), 10,000 iterations were performed and each sample from the defined distribution was included into the spreadsheet to determine risk. The final output was descriptive statistics based on the results of the Monte Carlo simulation. Means from these outcomes were used to describe the risk (see Tables 6,7, and 8).

The Dietary Exposure Potential Model is software developed by the National Exposure Research Laboratory of the Office of Research and Development of the USEPA (EPA 2001). This model was used to determine background intakes of each plant-metal combination. The model includes a consumption database (Continuing Survey of Food Intake by Individuals, 1994-1996) and a residue database (Food and Drug Administration—Total Diet Study Residue Database, 1982-1994). Exposures for each plant-metal combination are reported based on population age. These exposures were used as CIs in Equation 4 to determine background intake and risk. Background risk and risk calculated from elevated exposure were compared. The DEPM and its associated databases do not report residual zinc. Gerrior and Bente (2001) report the mineral concentration in the U.S. food supply in milligrams per capita per day (15 mg Zn per capita per day) (Gerrior and Bente 2001). Of the total amount of zinc in the food supply, vegetables make up 6 percent. In the report, vegetables are divided into ‘white potatoes’, ‘dark green/deep yellow’, ‘tomatoes’, and ‘other’ and percentages are given for each category. The vegetables used in this study were included in the other category so the percentage for ‘other’ was used (2.4%). Daily intake was calculated using Equation 7. Where *DI*=Daily intake (mg/kg-day), *FS*=Amount of mineral in U.S. Food Supply (mg/day), *VC*=Percentage of FS contributed by vegetables (unitless), and *BW*=Body weight (kg). It is not possible to separate zinc concentrations into the specific vegetables or age groups, as is done in DEPM, so one value was used for all ages and vegetable types.

$$DI = \frac{FS \times VC}{BW}$$ Equation 7

Non-carcinogenic effects were determined by dividing the Intake from Equation 4 by the RfD of each metal as reported in the IRIS database (Anonymous2001b). This calculation is described in EPA's RAGS (1989) and is shown in Equation 8. Where *HQ*=Noncancer Hazard Quotient, *E*=exposure level (or intake) (mg/kg-day), and *RfD*=reference dose (mg/kg-day). Carcinogenic effects were calculated for known carcinogens by multiplying the Intake from Equation 4 by the slope factor reported in the IRIS database. This calculation is also described in RAGS and is shown in Equation 9. Where *CDI*=Chronic daily intake averaged over 70 years (mg/kg-day), and *SF*=Slope Factor $(mg/kg\text{-}day)^{-1}$.

$$HQ = E / RfD$$ Equation 8

$$Risk = CDI \times SF$$ Equation 9

The U.S. EPA considers it inappropriate to establish an RfD for inorganic lead (EPA 1991). Some of the toxic effects of lead poisoning such as changes in blood enzyme levels and neurobehavioral impairment in children, can occur at blood lead levels that appear to be without a threshold. To determine a hazard quotient, one must have the reference dose (RfD). Because the EPA has not established an RfD, it was necessary to determine a value that can be used with a certain level of confidence. Because no threshold exists for lead toxicity, the lowest quantifiable value could be used and would be considered conservative. EPA's SW-846 lists the detection limit for lead using atomic absorption spectrophotometry—graphite furnace method as 1 µg/L. This value was used in Equation 10 to derive a value to be treated as the lowest observable adverse effect level (LOAEL). Where *LOAEL*=Lowest observable adverse effect level (µg/kg-day), *DL*=Detection limit (µg/kg), and *VC*=Contaminant concentration in vegetables (mg/kg). Equation 11 incorporates the LOAEL along with uncertainty and modifying factors to produce a value equivalent in nature to an RfD. Where *RfD*=Reference dose (µg/kg-day), *LOAEL*= Lowest observable adverse effect level (µg/kg-day), *UF*=Uncertainty Factor (unitless), *MF*=Modifying Factor (unitless). This value is then used in Equation 6 to determine

the noncancer hazard quotient. Because the detection limit of lead, which results in an RfD of $2.1x10^{-6}$ (mg/kg-day), may severely overestimate risk of toxic effects, another value was used. A value (0.05 mg/kg-day) that represents the dose at which severe toxic effects (reproductive, neurological and behavioral) begin to appear was also used to calculate noncancer hazard quotients.

$$LOAEL = (DL \times VC) / BW \qquad \text{Equation 10}$$

$$RfD = \frac{LOAEL}{UF \times MF} \qquad \text{Equation 11}$$

CHAPTER 4
RESULTS AND DISCUSSION

Fifty-two studies concerning heavy metal uptake by plants were identified from the literature. Of the plants studied, vegetables have been studied the most [31 (60%) studies]. Lettuce, cabbage, radish, and carrot were studied more often than any other vegetables. The metals studied most often were arsenic, cadmium, lead, and zinc. These metals all occur ubiquitously in the environment at trace levels. They can be associated with the underlying geology. The natural concentrations of these metals are typically at or near detection limits and pose no recognized risk to human health. However, metals used in industry or other anthropogenic activities can accumulate to potentially harmful concentrations. Of the studies reviewed in this investigation, 70% used field-contaminated soil (i.e., they either used contaminated soil collected in the field to culture plants in the laboratory or conducted field studies using plants growing in field-contaminated soil). Sources of contamination for soils used in these studies included mine tailings (20%), sludge amendments (20%), industrial wastes (20%), automobile exhaust (12%), fertilizers, and pesticides or herbicides (12%) and 16% were background concentrations. By pooling these data and using risk assessment methodology, general statements can be made concerning metal uptake in vegetables and their potential impact on human health.

The results of this meta-analysis include regressions of metal concentrations in the plants, the dependant variable, with metal concentrations in the soil, the independent variable (Table 4 and Figure 1). The estimated risks were calculated using exposure concentrations from the Monte Carlo simulation (Tables 5 through 8, Figures 2, 3, and 4). Because the arsenic-lettuce and lead-lettuce groups resulted in significantly higher risks, compared to the other groups, the inset graphs were created in Figures 2 and 3. Figure 4 was created using hazard quotients derived from an alternative RfD for lead.

Table 5: Pooled Equations from the Regression of the Dependant Plant-Metal Concentration and the Independent Soil-Metal Concentration and Associated R^2-Values for Each Plant-Metal Group.

Equation	R^2 (%) ± SD
As-Lettuce conc. = (Soil As conc.) 0.000034 + 29.1	60.2 ± (67.7)
As-Radish conc. = (Soil As conc.) 0.03 + 457.8	9.6 ± (84.1)
As-Carrot conc. = (Soil As conc.) 0.000002 + 0.1	72.9 ± (1.0)
Cd-Lettuce conc. = (Soil Cd conc.) 0.011 + 256.5	86.3 ± (0.3)
Cd-Cabbage conc. = (Soil Cd conc.) 0.026 + 0.11	4.1 ± (1.6)
Cd-Radish conc. = (Soil Cd conc.) 0.001 + 197.2	1.5 ± (2.2)
Pb-Lettuce conc. = (Soil Pb conc.) 0.000024 – 625.4	26.3 ± (14.0)
Pb-Cabbage conc. = (Soil Pb conc.) 0.035 – 59523.2	17.2 ± (0.4)
Pb-Radish conc. = (Soil Pb conc.) 0.000065 + 753.9	0.9 ± (6.0)
Zn-Lettuce conc. = (Soil Zn conc.) 0.016 – 41958.4	25.8 ± (79.3)
Zn-Radish conc. = (Soil Zn conc.) 0.0002 + 45167.8	0.3 ± (70.1)

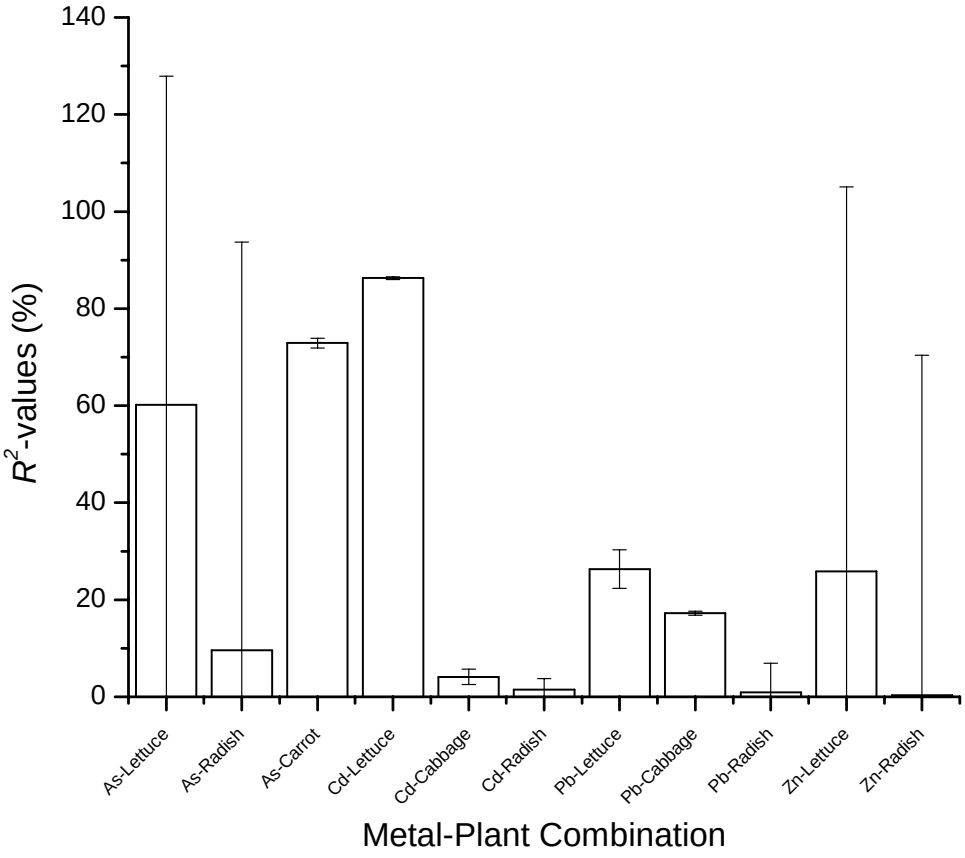

Figure 1: R^2-Values for Each Plant-Metal Combination. Values are from Table 4.

Table 6: Cancer Risk for Populations Exposed to Arsenic Contaminated Vegetables.

Population	CDI[a] (mg/kg-day)	SF (mg/kg-day)$^{-1}$	Weight of Evidence	Type of Cancer	SF source	SF Basis (Vehicle)	Exposure Risk[b]
Exposure Pathway: Ingestion of contaminated vegetables.							
Arsenic							
Carrot		1.5	A[a]	Skin Cancer	IRIS	Food	
Children 1-6 years old	3.53e-5 ± (4.88e-5)						5e-5 ± (8e-5)
Average Adult	8.15e-6 ± (1.17e-5)						1e-5 ± (2e-5)
Adults 55+ years old	8.16e-6 ± (1.2e-5)						1e-5 ± (2e-5)
Lettuce		1.5	A	Skin Cancer	IRIS	Food	
Children 1-6 years old	4.86e-3 ± (3.34e-3)						7e-3 ± (5e-3)
Average Adult	1.13e-3 ± (7.79e-4)						2e-3 ± (1e-3)
Adults 55+ years old	1.13e-3 ± (7.77e-4)						2e-3 ± (1e-3)
Radish		1.5	A	Skin Cancer	IRIS	Food	
Children 1-6 years old	1.61e-5 ± (1.98e-5)						2e-5 ± (3e-5)
Average Adult	3.65e-6 ± (4.12e-6)						6e-6 ± (6e-6)
Adults 55+ years old	3.68e-6 ± (4.43e-6)						6e-6 ± (6e-6)

[a] CDI (Chronic Daily Intake) is not reported in this table. CDI was determined through a Monte Carlo simulation.

SF = Slope Factor

[b] 'A' classification denotes a known human carcinogen.

[c] Exposure Risk is reported as a mean ± Standard deviation.

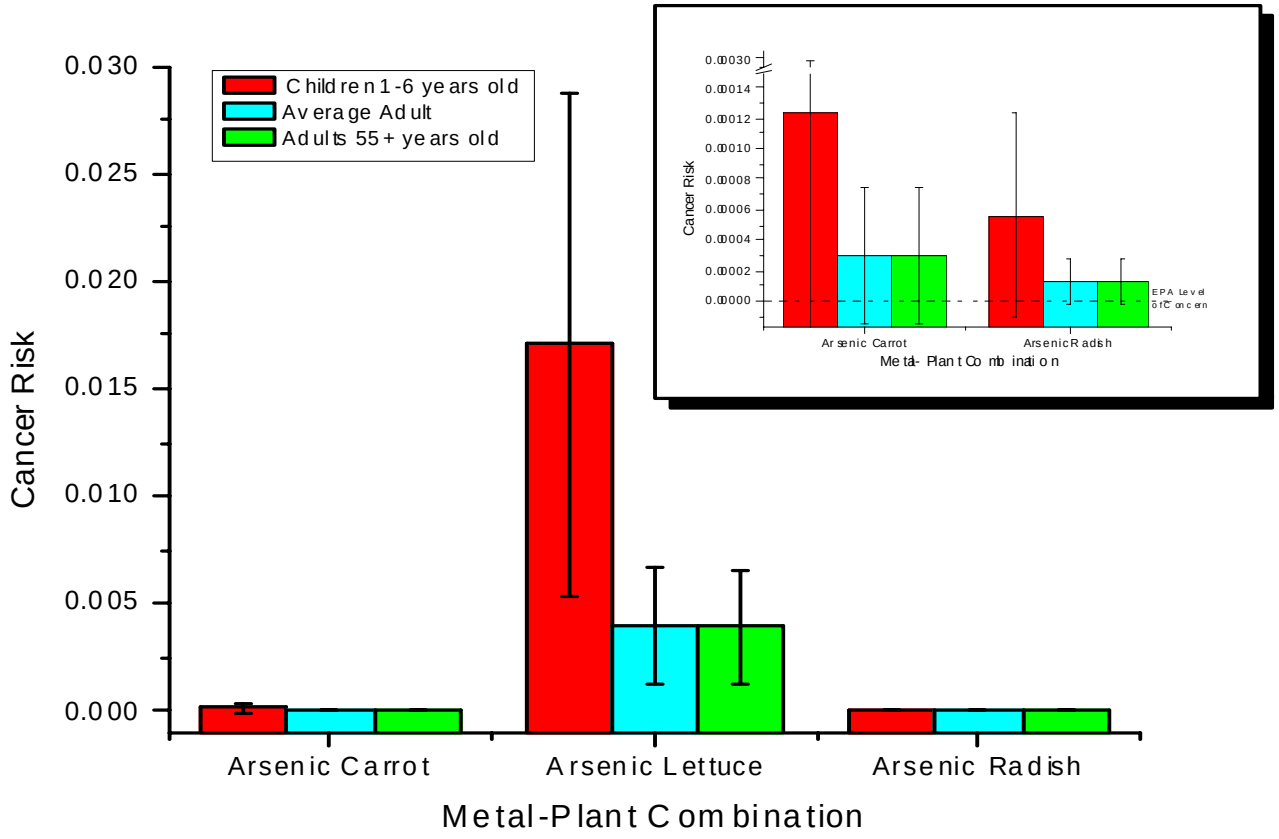

Figure 2: Cancer Risk for Populations Potentially Exposed to Heavy Metal Contaminated Vegetables. The inset graph shows cancer risk with the Arsenic-Lettuce combination removed to show the smaller scale for the other combinations and to show the EPA level of concern.

Table 7: Noncaner Hazard Quotients for Children 1-6 Years Old Exposed to Heavy Metal Contaminated Vegetables.

Exposure Scenario	CDI[a] (mg/kg-day)	Oral RfD (mg/kg-day)	Confidence Level[c]	Critical Effect	RfD Basis/ RfD Source	Uncertainty and Modifying Factors	Hazard Quotient	Chemical Specific Risk	Total Exposure Hazard Index
Exposure Pathway: Ingestion of contaminated vegetables									
Children 1-6 years old									
Cadmium		0.001	High	Protienuria	IRIS	UF = 10 for H MF = none			
Cabbage	2.26e-4 ± (5.54e-4)						0.2 ± (0.4)		
Lettuce	3.46e-2 ± (6.5e-2)						34 ± (62)		
Radish	2.4e-4 ± (4.97e-4)						0.2 ± (0.4)		
								34 ± (63)	
Lead		2.1e-6[b]	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS	UF = 10 for L MF = none			
Cabbage	3.35e-4 ± (6.45e-4)						2e2 ± (3e2)		
Lettuce	3.78e-2 ± (8.57e-2)						2e4 ± (4e4)		
Radish	1.6e-4 ± (2.73e-4)						8e1 ± (1e2)		
								2e4 ± (4e4)	
Zinc		0.3	Medium	Decrease in ESOD	IRIS	UF = 3 for L, S MF = none			
Lettuce	1.32e-2 ± (9.33e-3)						4e-2 ± (3e-2)		
Radish	3.51e-4 ± (5.66e-4)						1e-3 ± (2e-3)		
								5e-2 ± (3e-2)	
									2e4 ± (4e4)

[a] CDI is reported as mean ± SD. Values are results from Monte Carlo simulation.
[b] Not an EPA RfD. Value determined from analytical detection limits for AAS and adjusted with the specified uncertainty factor.
[c] Confidence level from IRIS: either High, Medium, or Low

Uncertainty Adjustments:

H = Intrahuman variability (UF = 10)
L = Evaluation from LOAEL instead of NOAEL
S = Evaluation from Subchronic to chronic exposure

CDI = Chronic Daily Intake
RfD = Reference Dose
AAS = Atomic Absorption Spectroscopy
ESOD= Erytrocyte Superoxide Dismutase

Table 8: Noncaner Hazard Quotients for Average Adults Exposed to Heavy Metal Contaminated Vegetables.

Exposure Scenario	CDI[a] (mg/kg-day)	Oral RfD (mg/kg-day)	Confidence Level[c]	Critical Effect	RfD Basis/ RfD Source	Uncertainty and Modifying Factors	Hazard Quotient	Chemical Specific Risk	Total Exposure Hazard Index
Exposure Pathway: Ingestion of contaminated vegetables									
Average Adult									
Cadmium		0.001	High	Protienuria	IRIS	UF = 10 for H MF = none			
Cabbage	5.07e-5 ± (1.1e-4)						5e-2 ± (1e-1)		
Lettuce	7.98e-3 ± (1.53e-2)						8 ± (15)		
Radish	5.46e-5 ± (9.29e-5)						6e-2 ± (1e-1)		
								8 ± (15)	
Lead		2.1e-6[b]	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS	UF = 10 for L MF = none			
Cabbage	7.96e-5 ± (1.59e-4)						37 ± (69)		
Lettuce	8.83e-3 ± (2.2e-2)						4e3 ± (1e4)		
Radish	3.81e-5 ± (7.47e-5)						18 ± (32)		
								4e3 ± (1e4)	
Zinc		0.3	Medium	Decrease in ESOD	IRIS	UF = 3 for L, S MF = none			
Lettuce	3.05e-3 ± (2.16e-3)						1e-2 ± (7e-3)		
Radish	8.1e-5 ± (1.24e-4)						3e-4 ± (4e-4)		
								1e-2 ± (8e-3)	
									4e3 ± (1e4)

[a] CDI is reported as mean ± SD. Values are results from Monte Carlo simulation.
[b] Not an EPA RfD. Value determined from analytical detection limits for AAS and adjusted with the specified uncertainty factor.
[c] Confidence level from IRIS: either High, Medium, or Low

Uncertainty Adjustments:

H = Intrahuman variability (UF = 10)
L = Evaluation from LOAEL instead of NOAEL
S = Evaluation from Subchronic to chronic exposure

CDI = Chronic Daily Intake
RfD = Reference Dose
AAS = Atomic Absorption Spectroscopy
ESOD= Erythrocyte Superoxide Dismutase

Table 9: Noncaner Hazard Quotients for Adults 55+ Years Old Exposed to Heavy Metal Contaminated Vegetables.

Exposure Scenario	CDI[a] (mg/kg-day)	Oral RfD (mg/kg-day)	Confidence Level[c]	Critical Effect	RfD Basis/ RfD Source	Uncertainty and Modifying Factors	Hazard Quotient	Chemical Specific Risk	Total Exposure Hazard Index
Exposure Pathway: Ingestion of contaminated vegetables									
Adults 55+ years old									
Cadmium		0.001	High	Protienuria	IRIS	UF = 10 for H MF = none			
Cabbage	5.06e-5 ± (1.05e-4)						5e-2 ± (1e-1)		
Lettuce	7.78e-3 ± (1.42e-2)						8 ± (15)		
Radish	5.42e-5 ± (9.67e-5)						5e-2 ± (9e-2)		
								8 ± (15)	
Lead		2.1e-6[b]	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS	UF = 10 for L MF = none			
Cabbage	7.97e-5 ± (1.56e-4)						38 ± (71)		
Lettuce	8.87e-3 ± (2.16e-2)						4e3 ± (1e4)		
Radish	3.73e-5 ± (6.70e-5)						18 ± (33)		
								4e3 ± (1e4)	
Zinc		0.3	Medium	Decrease in ESOD	IRIS	UF = 3 for L, S MF = none			
Lettuce	3.03e-3 ± (2.12e-3)						1e-2 ± (7e-3)		
Radish	7.97e-5 ± (1.21e-4)						3e-4 ± (4e-4)		
								1e-2 ± (8e-3)	
									4e3 ± (1e4)

[a] CDI is reported as mean ± SD. Values are results from Monte Carlo simulation.
[b] Not an EPA RfD. Value determined from analytical detection limits for AAS and adjusted with the specified uncertainty factor.
[c] Confidence level from IRIS: either High, Medium, or Low

Uncertainty Adjustments:

H = Intrahuman variability (UF = 10)
L = Evaluation from LOAEL instead of NOAEL
S = Evaluation from Subchronic to chronic exposure

CDI = Chronic Daily Intake
RfD = Reference Dose
AAS = Atomic Absorption Spectroscopy
ESOD= Erythrocyte Superoxide Dismutase

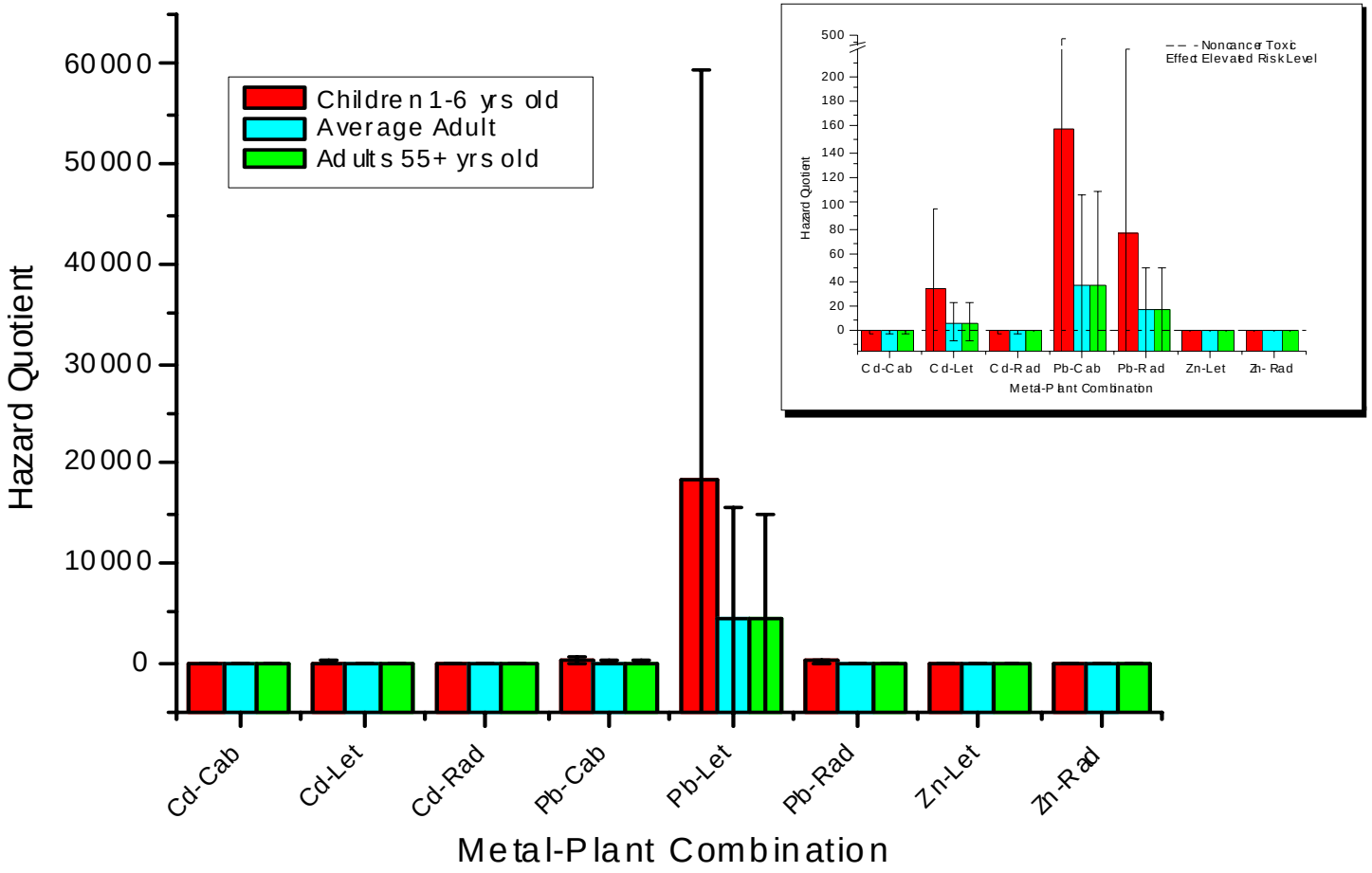

Figure 3: Noncancer Hazard Quotients for Populations Potentially Exposed to Heavy Metal Contaminated Vegetables. The inset graph shows hazard quotients with the Lead-Lettuce combination removed to show the smaller scale for the other combinations and to show the Elevated Risk Level.

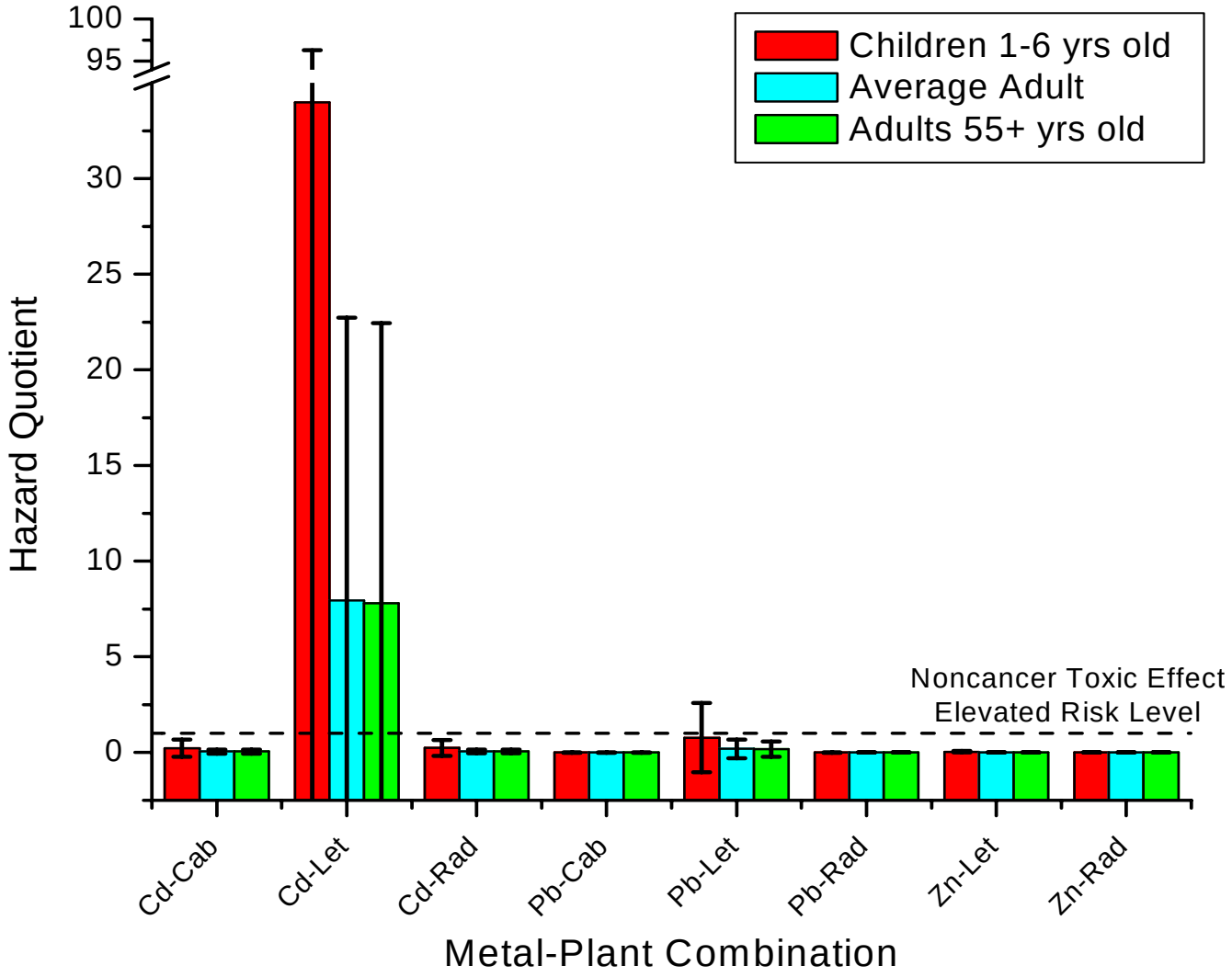

Figure 4: Noncancer Hazard Quotients for Populations Potentially Exposed to Heavy Metal Contaminated Vegetables.

Table 10: Noncancer Hazard Quotients for Exposure to Lead Using a RfD of 0.05 mg/kg-day.

Exposure Scenario	CDI[a] (mg/kg-day)	Oral RfD (mg/kg-day)	Confidence Level[c]	Critical Effect	RfD Basis/ RfD Source	Hazard Quotient	Specific Risk
Exposure Pathway: Ingestion of contaminated vegetables							
Lead							
Children 1-6 years old		0.05	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS		
Cabbage	3.35e-4 ± (6.45e-4)					7e-3±(1e-3)	
Lettuce	3.78e-2 ± (8.57e-2)					8e-1±(2)	
Radish	1.6e-4 ± (2.73e-4)					3e-3±(6e-3)	
							8e-1±(2)
Average Adult		0.05	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS		
Cabbage	3.35e-4 ± (6.45e-4)					2e-3±(3e-3)	
Lettuce	3.78e-2 ± (8.57e-2)					2e-1±(5e-1)	
Radish	1.6e-4 ± (2.73e-4)					7e-4±(1e-3)	
							2e-1±(5e-1)
Adults 55+ years old		0.05	Low	Hematologic, Musculoskeletal, and Neurologic	IRIS		
Cabbage	3.35e-4 ± (6.45e-4)					2e-3±(3e-3)	
Lettuce	3.78e-2 ± (8.57e-2)					2e-1±(4e-1)	
Radish	1.6e-4 ± (2.73e-4)					8e-4±(1e-3)	
							2e-1±(4e-1)

[a] CDI is reported as mean ± SD. Values are results from Monte Carlo simulation.
[b] Not an EPA RfD. Value determined from analytical detection limits for AAS and adjusted with the specified uncertainty factor.
[c] Confidence level from IRIS: either High, Medium, or Low

CDI = Chronic Daily Intake
RfD = Reference Dose
AAS = Atomic Absorption Spectroscopy
ESOD= Erytrocyte Superoxide Dismutase

Table 11: Noncancer Hazard Quotients for Populations Exposed to Lead in Lettuce for Two Different Reference Doses.

Population	Hazard Quotient	Standard Deviation
Lead RfD = 0.0000021 mg/kg-day		
Children 1-6 years old	18190	41385
Average Adult	4336	11210
Adults 55+ years old	4311	10665
Lead RfD = 0.05 mg/kg-day		
Children 1-6 years old	0.78	1.82
Average Adult	0.19	0.49
Adults 55+ years old	0.18	0.41

Meta-Analysis

The effect combined in the meta-analysis of results from these studies was the relationship between the metal concentrations in the soil to the metal concentrations found in the plants grown in these soils. This relationship was determined using regression analysis of soil metal concentrations with plant metal concentrations for each defined plant-metal combination (Table 3). Only 50% of the articles reviewed met the meta-analysis inclusion criteria. The criteria (other than the requisite use of the plants and metals previously mentioned) that must have been met for inclusion, were the availability of metal concentrations in soil or dosed metal concentrations and the quantitative concentrations of metals in plants.

Articles were excluded from the meta-analysis for violating any of the inclusion criteria. Some articles were excluded because metal concentrations in plants were not analyzed. The inhibition of plant growth caused by metal contamination was the only endpoint measured in 2 studies (Woolson 1973; Zaman and Zereen 1998). Plant-metal concentrations were not determined in these studies; therefore, they did not meet the inclusion criteria. Several researchers reported metal concentrations in the vegetables but did not report the corresponding soil or dosed concentrations (Hutchinson et al. 1974; Zurera et al. 1987; Adaudi et al. 1990; Alloway et al. 1990; Ward and Savage 1994; Muller and Anke 1994; De Pieri et al. 1996; Miner et al. 1997; Lee et al. 1997; Carbonell-Barrachina et al. 1999; Rahlenbeck et al. 1999; Albering et al. 1999). Unless there were other reasons for exclusion, an attempt was made to contact the authors of studies that were missing data to obtain these data. These attempts were successful in 2 cases and led to the inclusion of the studies in the meta-analysis (Boon and Soltanpour 1992; Sloan et al. 1997).

After studies that met the inclusion criteria were identified and divided based on plant-metal combination, 5 of these combinations (As-cabbage, Cd-carrot, Pb-carrot, Zn-cabbage, and Zn-carrot) were dropped from the analysis because only one study that dealt with the particular plant-metal combination met the inclusion criteria. Fifteen studies (of the original 31) met the inclusion criteria and fell into a plant-metal group that contained at least 2 studies. The exclusion of 4 of the possible 16 plant-metal combinations from this investigation was not expected.

When combining data, 2 sources of variation inherent in environmental and ecological data were considered. Type I error results from sampling error within a particular study.

Gurevitch and Hedges (1999) describe this type of variation as the deviation of the actual effect-size estimate from the value that would have been obtained if the sample size were large enough to reduce the sampling error to virtually zero. This type of error is common, especially with environmental data. Because of the wide variety of ways data are collected and analyzed, Type II is introduced. Type II error describes the between-study differences (Gurevitch and Hedges 1999).

The inverse variance weighted method has been widely used to combine effects (Hasselblad 1995). The weighting process of meta-analysis that was used accounted for variability between studies. After regression analyses were performed on the individual studies, the resulting R^2-values represented the fraction of the total variation in plant-metal concentrations that can be explained by the variation in soil-plant concentrations. R^2-values were weighted using the inverse of the *S*-values (the standard error of the values about the regression line). The slopes of the regression lines were weighted using the R^2-values. Type I error, introduced through within-study sampling error was not addressed by any of the weighting in this meta-analysis.

Weighted R^2-values and regression equations are reported in Table 6. The R^2-values span from 0.3% for the Zn-Radish group, which indicates no significant soil concentration effect (i.e., no linear relationship between concentration of Zn in the soil to that in the plant), to 86.3% for the Cd-lettuce group representing an apparent correlation between soil Cd concentration and plant Cd concentration. The Zn-Radish and Cd-Lettuce groups also illustrate the differences in standard deviations (SD) of the R^2-values. The relatively high (compared to other plant-metal groups) R^2-value and low SD for the Cd-lettuce group, shows that as Cd concentrations in soil increase the amount of Cd assimilated by the plant increases linearly. Conversely, with the low (0.3%) R^2-value of the Zn-radish group and its high (70.1%) SD, very little can be said, with any level of confidence, about the true slope of the regression line.

The lead-lettuce, lead-cabbage, and zinc-lettuce groups have a negative pooled y-intercept. This implies that when the metal concentration in the soil is zero, there is a negative metal concentration in the plant. This scenario is biologically implausible. Because the concentration of a metal cannot drop below zero, attempting to explain these relationships linearly is inappropriate for these groups. Therefore, these relationships must be nonlinear.

There are many possible explanations for the variability in these regression equations. Soil parameters, such as pH, organic matter content, and soil particle size affect the amount of the metal accumulated in the plant tissue (John and VanLaerhoven 1972; Boon and Soltanpour 1992; Nwosu et al. 1995b; De Pieri et al. 1996; Jinadasa et al. 1997; Miner et al. 1997; Albering et al. 1999). Soil parameters were not considered in this meta-analysis because they were reported in only 4 of the included studies (Garcia et al. 1981; Nwosu et al. 1995b; Sloan et al. 1997; Jinadasa et al. 1997). Soil parameters should be measured and reported by default whenever there is interest in plant contamination or the potential for plant contamination due to the presence of metals in the environment. Because of the deficiency in the reporting of soil parameters, the confidence in the results of this meta-analysis is low. Several researchers have shown significant relationships between the concentrations of metals in the soil and the concentrations of the metals in the plants (Davies 1978; Garcia et al. 1981; Xu and Thornton 1985; De Pieri et al. 1996; De Pieri et al. 1997; Sloan et al. 1997; Xiong 1998; Zaman and Zereen 1998; Gallardo-Lara et al. 1999). If multiple-regression products were pooled, the resulting regression equations would be looked upon with considerably more confidence. As coefficients are assigned for the various factors affecting metal uptake, the dependent variable (the concentration of the metal in the plant) will be better explained by the change in the independent variable (the metal concentration in the soil). As a result, the R^2-value will increase thereby lending more confidence to the pooled estimate. The importance of meta-analysis in this situation is that it can identify the site parameters that confound predictor variates such as the metal concentration in the soil.

Few studies have attempted to apply meta-analytic techniques to environmental data (i.e., environmental contaminants). Several studies, however, have successfully applied these techniques to epidemiological and environmental health data (Morris et al. 1992; Morris 1994; Blair et al. 1995; Hasselblad 1995; Boffetta et al. 2000). These studies typically combined odds ratios and relative risks, both of which are universally reported in epidemiological studies. Poor data reporting severely limited not only the number of studies that could be included but also the confidence of the variability of the studies that were included. Gurevitch and Hedges (1999) identify the most serious limitation to ecological meta-analysis as poor data reporting. Common deficiencies in the studies reviewed for this meta-analysis were failure to report sample sizes and/or some measure of variance (standard deviation, standard error, range, etc.) about the mean.

Studies that could have significantly impacted the results of the meta-analysis had to be excluded because of inadequacies data reporting.

Risk Assessment

A comprehensive risk assessment was performed in an attempt to quantify the carcinogenic and noncarcinogenic risks, associated with ingestion of common garden vegetables contaminated with heavy metals. The U.S. Environmental Protection Agency has established a protocol for conducting such risk assessments in its *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part A)* (RAGS) (EPA 1989). Exhibit 6-18 of RAGS, Equation 3 of this document, is used in the determination of residential exposure via the food pathway, specifically, the ingestion of contaminated fruits and vegetables (EPA 1989). This model incorporates variables that are specific to vegetable, exposure, and population characteristics.

The populations for which risk would be calculated were determined based on sensitivity to toxic effects. Children below the age of 6 and adults 55 and older were identified as sensitive sub-populations. Risk was calculated for average adults (18-54 years old) to highlight the risk to the average person. Consideration was initially made for gender; however, there was no significant difference ($P>0.05$) between sexes. All gender specific values for each age category were averaged.

Some assumptions had to be made where data were not provided in the Exposure Factors Handbook. Average Daily Consumption was provided for all 4 plants (cabbage, carrot, lettuce and radish). However, the Average Quantity Consumed per Meal was not provided for radish. The value needed for the risk equation (Equation 3) had to be a quantity (kg) per meal. Because this value was not provided for radishes, a value was extrapolated from the Average Daily Consumption. This process, as described in the Research Plan, involved an assumption of the number of meals each day that include radishes. Reverse calculations were done on the other 3 vegetables to determine this value for each of them. It was determined that cabbage was assumed to be eaten once every 10 days, lettuce-once every 4 days, and carrot-twice every 7 days. Based on the frequency of ingestion of lettuce and the relative infrequent consumption of radish, an ingestion frequency of once per week was assumed and considered conservative. For the purposes of assessing risk, conservative assumptions are not only appropriate, but necessary.

Because toxic effects have been shown at very low levels that suggest there is no threshold, EPA has resisted establishing a reference dose (RfD). An RfD is an exposure level at which no toxic effects will occur. To determine the risk associated with lead exposure, an RfD value must be assumed. Therefore, I initially assumed that the RfD was equal to the lead the detection limit of 1 μg/L. This number was converted into an RfD as described in the Research Plan. Using this estimate introduces a significant amount of uncertainty into the calculated risk value. It is possible that the hazard quotients derived from this number could be overestimated. We have to assume a certain level of risk for many decisions we make each day. By stepping outside, we subject ourselves to harmful radiation from the sun. The EPA does not attempt to regulate this exposure, but it can regulate acute exposure to radiation. In the same way, if lead is ubiquitous (even at trace concentrations) in the environment, we can never have zero exposure. There must be some acceptable level of risk. For this reason, an alternate value was selected as the RfD for lead. At an intake of 0.05 mg Pb/kg body weight per day reproductive effects, such as decreased motility of spermatozoa in male rats (Krasovskii et al. 1979), neurological effects, including impaired operant learning (Rice 1985) and behavioral dysfunction in monkeys (Rice and Karpinski 1988) begin to appear. This level represents the dose at which severe toxic effects begin to occur, and was, therefore, used as an RfD in the calculation of hazard quotients.

EPA's noncancer toxic effect elevated risk level, the level at which there is no increased risk of toxic effects, is a hazard quotient of one. This line is shown in the inset graph in Figure 3. All exposure scenarios associated with lead and the Cd-lettuce combination for all age groups exceed this level. The large graph in Figure 3 shows the extremely high ($2x10^4$) hazard quotient for children exposed to lead through the ingestion of lettuce. The hazard quotients for average adults and adults 55 and older are shown in the large graph of Figure 3. These values ($4x10^3$) also far exceed the elevated risk level. For all age categories the Pb-cabbage, Pb-radish, and Cd-lettuce combinations exceed the elevated risk level (Figure 3). Exposures to zinc through ingestion of lettuce in radish and exposures to cadmium through ingestion of cabbage and radish by all age groups result in hazard quotients below the elevated risk level, indicating no risk.

The hazard quotients derived from using 0.05 mg/kg-day as the RfD for lead are represented in Figure 4. This graph also includes the hazard quotients from all other metals. Therefore, the only difference between Figure 3 and Figure 4 are the hazard quotients for the lead-plant combinations. By using a higher RfD for lead, the risk of noncancer toxic effects is 5

orders of magnitude lower. For example, using the detection limit of lead in the computation of the RfD the hazard quotient of children exposed to lead in lettuce is $2x10^4$. This same exposure in Figure 4 results in a hazard quotient of 0.8 (see Table 11). The mean hazard quotients for each lead exposure scenario in Figure 4 are below EPA's noncancer toxic effect elevated risk level. The variation of the hazard quotient associated with lead-lettuce exposure in children extends beyond the elevated risk level. This means that there is a possibility that the true hazard quotient falls above this level, indicating an increased risk of toxic effects in children exposed to lead in lettuce.

Estimates of the contaminant concentration in these vegetables were determined from Monte Carlo simulation. The Monte Carlo technique produces an estimate of the standard deviation for each of the hazard quotients and cancer risk estimates. Traditional risk assessments use point estimates of exposure yielding point estimates of risk with no measure of the variation. Monte Carlo analysis provides an entirely different perspective on actual risk. For example, the hazard quotient for the children exposed to lead through the ingestion of lettuce is $2x10^4$, but the standard deviation is two times the magnitude of that estimate ($4x10^4$). This means that the real hazard quotient could be as high as $6x10^4$ or as low as zero, implying that there is no risk. Similarly, for every combination that exceeds the elevated effect level (Figure 3), in each case the respective error bars drop well below the elevated effect level. Risk assessors and regulatory entities must evaluate the uncertainty of the estimate of risk when making decisions based on these estimates. Single-value estimates of risk conceal this uncertainty. Because single-value risks are derived from one set of data, depending on how those data are chosen, the calculated risk or hazard quotient could be either over- or underestimated. By pooling all available data and using Monte Carlo simulations to enhance these data, the central tendency (mean) could serve as the point estimate for regulatory purposes. The added benefit of using this value would be that the variation of the estimate is known. If questions arise about the confidence of a traditionally derived single-point estimate, regulatory decisions using these data should be made with caution. The current practices of deriving point estimates of risk include built-in safety factors, such as the uncertainty factors and modifying factors used in the derivation of the RfD. This provides some level of conservatism, however, the validity of the data set used to derive that point estimate remains in question.

The cancer risks associated with ingestion of vegetables contaminated with arsenic, a known human carcinogen, are shown in Figure 2. EPA's level of concern for cancer risk is $1x10^{-6}$. All arsenic exposure scenarios evaluated in this study result in cancer risks above the level of concern. The cancer risk of children exposed to arsenic from ingestion of lettuce is four orders of magnitude greater than this level ($2x10^{-2}$). The cancer risk for the average adult and adults 55 and older are also well above the level of concern ($4x10^{-3}$). The standard deviations of As-carrot and As-radish combinations for all age groups extend beyond EPA's level of concern. This situation invokes the same level of uncertainty as the hazard quotient calculations previously discussed. This is not the case, however, for the As-lettuce combination for all age categories. None of the error bars in this group extends beyond EPA's level of concern. Because even at the lowest limit of the standard deviation of each age category, cancer risks remain several orders of magnitude above EPA's level of concern, it can be said with certainty that there is a high risk of cancer for those individuals consuming lettuce grown in soils contaminated with arsenic. Point estimate risk assessments would have lead to the same conclusion concerning the As-lettuce combination. However, by using Monte Carlo risk assessment we are able to quantify the variation in these estimates and conclude with a higher level of confidence that there is a significant risk. The summation of all hazard quotients from all exposure pathways for children, average adults, and adults 55 and older are given in Tables 8, 9, and 10. This additive hazard quotient is known as the Total Exposure Hazard Index.

Background levels of metal ingestion were determined using the Dietary Exposure Potential Model (DEPM). These background levels, based on age, were used as the CFs in Equation 3 to determine intake and risk associated consumption of vegetables with background levels of metals. These values were all well below not only the respective contaminated concentrations but also EPA's level of concern for carcinogenic effects and the noncancer toxic effect elevated risk level. Therefore, there appears to be no risk, either carcinogenic or noncarcinogenic, associated with background concentrations of metals in plants.

Of the 30 papers used in this study, only 13 made some determination as to the human health risk associated with exposure to heavy metal contaminated vegetables. Increased risk was found in 7 studies (John et al. 1972; Boon and Soltanpour 1992; Muller and Anke 1994; Nwosu et al. 1995b; Carbonell-Barrachina et al. 1999; Cobb et al. 2000), and no increased risk was found in 6 studies (Woolson 1973; Zurera et al. 1987; Helgesen and Larsen 1998; Gallardo-Lara

et al. 1999; Rahlenbeck et al. 1999). These risk estimates were all qualitative and involved merely comparing estimated potential exposure to background exposure. No actual risk or hazard quotient was determined.

CHAPTER 6
CONCLUSIONS AND RECOMMENDATIONS

Conclusions

This is the first attempt at developing a method to combine this type of data. This process has revealed that the use of linear regressions to characterize the relationship between metal concentrations in plants and metal concentrations in soils are probably inappropriate. Nonlinear regressions would more accurately represent the true relationship. Also, the variability of the pooled R^2-values (Table 4) may be a result of variables, such as soil parameters, climate, or method of detection, not accounted for in this meta-analysis. Inconsistent data reporting between studies prevented these factors from being considered.

The use of Monte Carlo simulations in risk assessment provides an understanding of the degree of uncertainty and variability around a risk estimate that single-point estimates of risk cannot provide (EPA 1994b). Single-point estimates, although often accompanied by a qualitative discussion of uncertainty, do not have associated confidence intervals. Monte Carlo risk assessment gives an idea of the level of variability in the risk estimate. Understanding the variation in the risk estimate, decision makers and regulatory agencies will be better informed about the significance of the risk estimate. In this risk assessment, extremely high (5 orders of magnitude over EPA levels of concern) hazard quotients for exposure of children to lead in lettuce (Figure 3) are reported, yet the associated error bars indicate that the true risk estimate could range between zero, indicating no risk, to 3 times the point estimate. This extreme variability limits the level of confidence with which any regulatory decision can be made concerning these data. On the other hand, it can be concluded, with a high level of confidence, that exposure to arsenic in lettuce at concentrations found in the 2 studies included in this group greatly increases the risk of developing cancer (see Figure 2). The standard deviations of the estimates for the arsenic-lettuce combination for each age category do not include EPA's level of concern. This means that even at the lowest value within the confidence interval, the cancer risk exceeds the level of concern indicating that there is an elevated risk of carcinogenic effects.

A limitation to this risk assessment is that studies related only by the plant and metal of concern were combined to establish risk. This means that other possible confounders, including soil parameters, were not considered. Also, by having a low N (number of studies included in

the risk assessment), the effect of outliers is more evident. Type I and type II error could both affect the risk estimates. Monte Carlo simulation does reduce these sources of error; however, it is not possible to distinguish between the two (Gurevitch and Hedges 1999).

Recommendations

Because soil characteristics such as pH, organic matter content, clay content, and cation exchange capacity affect the uptake of metals by plants, it would lend power to future meta-analyses to combine the products of a multiple regression where these factors are included, which can only be done if the data are available. In addition, it is suggested that nonlinear curve fitting be explored in the future to explain the relationship between the concentration of the metal in the soil and the concentration of the metal in the plant.

I have shown that both meta-analysis and risk assessment using environmental and ecological data are severely limited by the data that are available. It has been suggested by Gurevitch and Hedges (1999) that publication standards should be upgraded, and authors, reviewers, and editors should be made aware that basic information should be required from all research before publication. By unifying data reporting standards, possibilities for the application of meta-analysis on environmental and ecological data will greatly increase.

REFERENCES

[Anonymous]. Cadmium. CyberIndiana . 2001a.

[Anonymous]. Integrated Risk Information System. US Environmental Protection Agency . 2001b.

Adaudi AO, Gbodi TA, Aliu YO. 1990. The lead content of plants and animals as indicators of environmental contamination. Vet Hum Toxicol 32(5):454-6.

Albering HJ, van Leusen SM, Moonen EJ, Hoogewerff JA, Kleinjans JC. 1999. Human health risk assessment: A case study involving heavy metal soil contamination after the flooding of the river Meuse during the winter of 1993-1994. Environ Health Perspect 107(1):37-43.

Alloway BJ, Jackson AP, Morgan H. 1990. The accumulation of cadmium by vegetables grown on soils contaminated from a variety of sources. Sci Total Environ 91:223-36.

Alloway BJ, Thornton I, Smart GA, Sherlock JC, Quinn MJ. 1988. The Shipham report. An investigation into cadmium contamination and its implications for human health. Metal availability. Sci Total Environ 75(1):41-69.

Armstrong C, Stroube R, Rubio T. Outbreak of fatal arsenic poisoning caused by contaminated drinking water. Arch Environ Health 44[5], 283-290. 1984.

ATSDR. Toxicological Profile for Zinc. 1994. US Department of Health and Human Services, Public Health Service. 205-88-0608.

ATSDR. Toxicological Profile for Cadmium. 1999a. US Department of Health and Human Services, Public Health Service. 205-93-0606.

ATSDR. Toxicological Profile for Lead. 1999b. US Department of Health and Human Services, Public Health Service. 205-93-0606.

ATSDR. Toxicological Profile for Arsenic. 2000. US Department of Health and Human Services, Public Health Service. 205-1999-00024.

Baker DE, Amacher MC, Leach RM. 1979. Sewage sludge as a source of cadmium in soil-plant-animal systems. Environ Health Perspect 28:45-9.

Balba A, Shibiny G, El-Khatib E. Effect of Lead Increments on the Yield and Lead Content of Tomato Plants. Water, Air, and Soil Pollution 57-58, 93-99. 1991.

Blair A, Burg J, Foran J, Gibb H, Greenland S, Morris R, Raabe G, Savitz D, Teta J, Wartenberg D, . 1995. Guidelines for application of meta-analysis in environmental epidemiology. ISLI Risk Science Institute. Regul Toxicol Pharmacol 22(2):189-97.

Boffetta P, Tredaniel J, Greco A. Risk of Childhood Cancer and Adult Lung Cancer after Childhood Exposure to Passive Smoke: A Meta-Analysis. Env.Health.Persp. 108[1], 73-82. 2000.

Boon DY, Soltanpour PN. Lead, Cadmium, and Zinc Contamination of Aspen Garden Soils and Vegetation. J.Environmental Quality 21, 82-86. 1992.

Carbonell-Barrachina AA, Burlo F, Lopez E, Martinez-Sanchez F. 1999. Arsenic toxicity and accumulation in radish as affected by arsenic chemical speciation. J Environ Sci Health B 34(4):661-79.

Cataldo D, Garland T, Wildung R. Cadmium Distribution and Chemical Fate in Soybean Plants. Plant Physiology 68, 835-839. 1981.

Chen X, Wright J, Conca J, Peurrung L. Effects of pH on Heavy Metal Sorption on Mineral Apatite. Environ Sci and Technology 31[3], 624-631. 1997.

Cobb GP, Sands K, Waters M, Wixson BG, Dorward-King E. Accumulation of Heavy Metals by Vegetables Grown in Mine Wastes. Environmental Toxicology and Chemistry 19[3], 600-607. 2000.

Darland JE, Inskeep WP. Effects of pH and Phosphate Competition on the Transport of Arsenate. J.Environmental Quality 26, 1133-1139. 1997.

Davies BE. Plant-Available Lead and Other Metals in British Garden Soils. Sci.Total Environ. 9, 243-262. 1978.

De Pieri LA, Buckley WT, Kowalenko CG. Micronutrient concentrations of commercially grown vegetables and of soils in the Lower Fraser Valley of British Columbia. Canadian Journal of Soil Science 76, 173-182. 1996.

De Pieri LA, Buckley WT, Kowalenko CG. Cadmium and lead concentrations of commercially grown vegetables and of soils in the Lower Fraser Valley of British Columbia. Canadian Journal of Soil Science 77, 51-57. 1997.

DOE. 1999. Guidance for Conducting Risk Assessments and Related Risk Activities for the DOE-ORO Environmental Management Program. US Department of Energy, Office of Environmental Management; Report nr BJC/OR-271.

Edberg SC. Assessing Health Risk In Drinking Water From Naturally Occurring Microbes. J.Env.Health 58[6], 18-24. 1996.

EPA. 1989. Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part A). Washington, DC: US Environmental Protection Agency, Office of Emergency and Remedial Response; Report nr EPA/540/1-89/002.

EPA. 1991 Feb. Integrated Risk Information System-Lead and compounds (inorganic). US Environmental Protection Agency; Report nr 7439-92-1.

EPA. 1994a Feb. Integrated Risk Information System-Cadmium. US Environmental Protection Agency; Report nr 7440-43-9.

EPA. 1994b Feb. Use of Monte Carlo Simulation in Risk Assessments. Philadelphia, PA: US Environmental Protection Agency, Region III;Technical Guidance Manual Report nr EPA903-F-94-001.

EPA. 1997 Aug. Exposure Factors Handbook. National Center for Environmental Assessment: U.S. Environmental Protection Agency; Report nr EPA/600/8-89/043.

EPA. 2001 Dietary Exposure Potential Model. Office of Research and Development: National Exposure Research Laboratory.

Gallardo-Lara F, Azcon M, Quesada JL, Polo A. 1999. Phytoavailability and extractability of copper and zinc in calcareous soil amended with composted urban wastes. J Environ Sci Health B 34(6):1049-64.

Garcia WJ, Blessin CW, Inglett GE, Kwolek WF. Metal Accumulation and Crop Yield for a Variety of Edible Crops Grown in Diverse Soil Media Amended with Sewage Sludge. Environmental Science and Technology 15[7], 793-804. 1981.

Garcia WJ, Blessin CW, Sandford HW, Inglett GE. 1979. Translocation and accumulation of seven heavy metals in tissues of corn plants grown on sludge-treated strip-mined soil. J Agric Food Chem 27(5):1088-94.

Gerrior S, Bente L. 2001. Nutrient Content of the U.S. Food Supply, 1909-97. U.S. Department of Agriculture, Center for Nutrition Policy and Promotion, Home Economics Research Report; Report nr 54.

Golub MS, Macintosh MS, Baumrind N. 1998. Developmental and reproductive toxicity of inorganic arsenic: animal studies and human concerns. J Toxicol Environ Health B Crit Rev 1(3):199-241.

Gurevitch J, Hedges LV. Statistical issues in ecological meta-analyses. Ecology 80[4], 1142-1149. 1999.

Haghiri F. Cadmium Uptake by Plants. J.Environmental Quality 2[1], 93-96. 1973.

Hasselblad V. 1995. Meta-analysis of environmental health data. Sci Total Environ 160-161:545-58.

Helgesen H, Larsen EH. 1998. Bioavailability and speciation of arsenic in carrots grown in contaminated soil. Analyst 123(5):791-6.

Honschopp S, Brunken N, Nehrhorn A, Breunig HJ. 1996. Isolation and characterization of a new arsenic methylating bacterium from soil. Microbiol Res 151(1):37-41.

Hutchinson TC, Czuba M, Cunningham L. Lead, Cadmium, Zinc, Copper and Nickel Distributions in Vegetables and Soils of an Intensely Cultivated Area and Levels of Copper, Lead and Zinc in the Growers. Trace Substances in Environmental Health 8, 81-93. 1974.

Janssen R, Peijnenburg W, Posthuma L, VanDenHoop M. 1997. Equilibrium partitioning of heavy metals in Dutch field soils. I.Relationship between metal partition coefficients and soil characteristics. Environmental Toxicology and Chemistry 16[12], 2470-2478.

Jinadasa KBPN, Milham PJ, Hawkins CA, Cornish PS, Williams PA, Kaldor CJ, Conroy JP. Survey of Cadmium Levels in Vegetables and Soils of Greater Sydney, Australia. J.Environmental Quality 26, 924-933. 1997.

John MK, VanLaerhoven CJ. 1972. Lead distribution in plants grown on a contaminated soil. Environ Lett 3(2):111-6.

John MK, VanLaerhoven CJ, Chukwuma C, Sr. Factors Affecting Plant Uptake and Phytotoxicity of Cadmium Added to Soils. Environmental Science and Technology 6[12], 1005-1009. 1972.

Jones CA, Inskeep WP, Nguyen L. Arsenic transport in contaminated mine tailings following liming. J.Environmental Quality 26[2], 433-439. 1997.

Khan DH, Frankland B. Effects of cadmium and lead on radish plants with particular reference to movement of metals through soil profile and plant. Plant and Soil 70, 335-345. 1983.

Kosla T. 1986. [Studies of the levels of copper and zinc in the soil, grass and blood serum and hair of bulls from the areas irrigated with waste water from the Ner River]. Pol Arch Weter 25(1):145-54.

Krasovskii GN, Vasukovich LY, Chariev OG. Experimental studyof biological effects of lead and aluminum following oral administration. Env.Health.Persp. 30, 47-51. 1979.

Larlson J, Likens G, Fitzpatrick J, Crock J. Cadmium Toxicity Among Wildlife in the Colorado Rocky Mountains. Nature 406, 181-183. 7-13-2000.

Lee BD, Carter BJ, Basta NT, Weaver B. Factors influencing heavy metal distribution in six Oklahoma benchmark soils. Soil Sci.Soc.Am.J. 61, 218-223. 1997.

Li L, Wu G. Numerical Simulation of Transport of Four Heavy Metals in Kaolinite Clay. J.Environ Engineering 125[4], 314-324. 1999.

Mahaffey KR, Capar SG, Gladen BC, Fowler BA. 1981. Concurrent exposure to lead, cadmium, and arsenic. Effects on toxicity and tissue metal concentrations in the rat. J Lab Clin Med 98(4):463-81.

McGeehan SL. Arsenic Sorption and Redox Reactions: Relevance to Transport and Remediation. J.Environ.Sci.Health A31[9], 2319-2336. 1996.

Miner GS, Guitierrez R, King LD. Soil Factors Affecting Plant Concetrations of Cadmium, Copper, and Zinc on Sludge-Amended Soilsf. J.Environmental Quality 26, 989-994. 1997.

Moore DS. 1995. The Basic Practice of Statistics. New York: W.H. Freeman and Company.

Morris R. Meta-Analysis in Cancer Epidemiology. Env.Health.Persp. 102, Supplement 8, 61-66. 1994.

Morris R, Audet A, Angelillo IF, Chalmers TC, Mosteller F. Chlorination, Chlorination By-products, and Cancer: A Meta-Analysis. Am.J.Pub.Health 82[7], 955-963. 1992.

Muller M, Anke M. 1994. Distribution of cadmium in the food chain (soil-plant-human) of a cadmium exposed area and the health risks of the general population. Sci Total Environ 156(2):151-8.

Murphy JV. Intoxication following ingestion of elemental zinc. JAMA 212, 2119-2120. 1970.

Nwosu JU, Harding AK, Linder G. 1995. Cadmium and lead uptake by edible crops grown in a silt loam soil. Bull Environ Contam Toxicol 54(4):570-8.

Ochiai E. Toxicity of Heavy Metals and Biological Defense. Journal of Chemical Education 72[6], 479-484. 1995.

Peles J, Brewer S, Barrett G. Heavy Metal Accumulation by Old-field Plant Species During Recovery of Sludge-treated Ecosystems. The American Midland Naturalist 140[2], 245-251. 1998.

Pitten FA, Muller G, Konig P, Schmidt D, Thurow K, Kramer A. 1999. Risk assessment of a former military base contaminated with organoarsenic-based warfare agents: uptake of arsenic by terrestrial plants. Sci Total Environ 226(2-3):237-45.

Putzrath RM, Ginevan ME. 1991. Meta-analysis: methods for combining data to improve quantitative risk assessment. Regul Toxicol Pharmacol 14(2):178-88.

Rahlenbeck SI, Burberg A, Zimmerman RD. Lead and Cadmium in Ethiopian Vegetables. Bull.Environ.Contam Toxicol. 62, 30-33. 1999.

Rice DC. Chronic low-lead exposure from birth produces deficits in discrimination reversal in monkeys. Toxicol.Appl.Pharmacol. 77, 201-210. 1985.

Rice DC, Karpinski K. Lifetime low-level lead exposure produces deficits in delayed alteration in adult monkeys. Neurotoxicol.Teratol. 10, 207-214. 1988.

Sax N, Lewis Sr. R. 1987. Hawley's Condensed Chemical Dictionary. 11 ed. New York: Van Nostrand Reinhold Co.

Sloan JJ, Dowdy RH, Dolan MS, Linden DR. Long-Term Effects of Biosolids Applications on Heavy Metal Bioavailability in Agricultural Soils. J.Environmental Quality 26, 966-974. 1997.

Todd AC, Wetmur JG, Moline JM, Godbold JH, Levin SM, Landrigan PJ. 1996. Unraveling the chronic toxicity of lead: an essential priority for environmental health. Environ Health Perspect 104 Suppl 1:141-6.

Turpeinen R, Pantsar-Kallio M, Haggblom M, Kairesalo T. 1999. Influence of microbes on the mobilization, toxicity and biomethylation of arsenic in soil. Sci Total Environ 236(1-3):173-80.

Ward NI, Savage JM. Metal dispersion and transportational activities using food crops as bioindicators. Sci.Total Environ. 146/147, 309-319. 1994.

Woolson E. Arsenic Phytotoxicity and Uptake in Six Vegetable Crops. Weed Science 21[6], 524-527. 1973.

Xiong ZT. Lead Uptake and Effects on Seed Germination and Plant Growth in a Pb Hyperaccumulator *Brassica pekinensis* Rupr. Bull.Environ.Contam Toxicol. 60, 285-291. 1998.

Xu J, Thornton I. Arsenic in garden soils and vegetable crops in Cornwall, England: implications for human health. Environmental Geochemistry and Health 7[4], 131-133. 1985.

Zaman MS, Zereen F. 1998. Growth responses of radish plants to soil cadmium and lead contamination. Bull Environ Contam Toxicol 61(1):44-50.

Zurera G, Estrada F, Rincon F, Pozo R. Lead and Cadmium Contamination Levels in Edible Vegetables. Bull.Environ.Contam Toxicol. 38, 805-812. 1987.

APPENDICES

APPENDIX A

Meta-Analysis Data

Table 12: Raw Data for Weighting and Combining Using Meta-analysis.

Study	R^2 (%)	S[a]	Slope	Y-Int
Cobb et al. 2000	71.3	9.58	0.105	5.41
	37.7	18.52	0.1	-8.3
	84.1	4.578	1.58	-3.93
	26	4.812	0.461	1.97
	77.7	189.5	0.088	-39
	30.1	146.9	0.0263	41
	63.4	10.82	0.00658	8.98
	25.9	205.6	1.31	146
Khan and Frankland 1983	6.3	29.33	0.0169	28.1
	49.6	93.95	0.186	2.6
Xu and Thornton 1985	49	9.58	0.0038	-0.2176
	23	0.22311	0.00603	-0.62
Boon and Soltanpour 1992	74	2.059	0.95	2.07
	90.3	21.91	0.05	2.36
Nwosu et al. 1995	15.4	158.1	0.473	174
	61.7	108.5	0.967	41.6
	49	21.91	0.0563	-5.12
	87.7	9.396	0.0657	-2.36
Xiong 1998	40.9	8701	18.3	-495
Helgesen and Larsen 1998	89.9	0.2311	0.00537	0.089
Carbonell-Barrachina et al. 1999	4.8	20.89	2.52	16.4
	1.6	10.58	-0.73	13
Jinadasa et al. 1997	97.1	0.03553	0.505	0.0015
	0.3	0.2624	-0.015	0.177
Garcia et al. 1981	60.1	1.412	0.00451	2.22
	2.2	0.8875	0.0189	0.543
	19.4	18.4	0.0128	17.3
	0.5	0.6037	-0.000113	0.695
	0	0.226	-0.00037	0.158
	0.3	7.015	0.00054	8.92
De Pieri et al. 1997	0.3	0.3328	0.06	0.314
	17.2	0.03978	0.00317	0.459
	55.1	0.0606	0.1713	0.205
	25.7	0.1812	0.0223	-0.24
	4.8	7.545	0.652	68.053
Sloan et al. 1997	88	2.059	1.12	-0.89
	9	21.91	-0.002	0.99
	41	15.09358	0.13	9.25
Davies 1978	15.8	66.7641	0.582	-0.275
Haghiri 1973	98.6	2.059	2.35	4.01
	85.7	3.914	1.3	3.87
	78.1	1.832	0.47	1.59
Alloway et al. 1990	12	0.2999	0.506	15.1

[a] *S* is the standard error about the regression line and is reported in the regression output. Where values are missing, R^2 values were reported in the study and *S* values were unavailable.

Table 13: *R*, R^2, Slope and Y-Intercept Values Combined from Like Plant-Metal Groups and the Number of Studies in Each Group.

Plant-Metal Combination	***R***	R^2**(%)**	**Slope**	**Y-Intercept**	***N*** [1]
Arsenic-Lettuce	0.772	60.2	0.035	2.6	2
Arsenic-Radish	0.242	9.6	0.037	7.033	3
Arsenic-Carrot	0.853	72.9	0.0045	-0.266	2
Cadmium-Lettuce	0.922	86.3	0.396	22	8
Cadmium-Cabbage	0.148	4.1	0.0045	5.197	3
Cadmium-Radish	0.02	1.5	0.008	12.88	6
Lead-Lettuce	0.511	26.3	0.0175	-6.465	6
Lead-Cabbage	0.415	17.2	2.945	-247.27	2
Lead-Radish	0.074	0.9	0.021	8.332	5
Zinc-Lettuce	0.441	25.8	0.0245	25.9	4
Zinc-Radish	0.055	0.3	0.033	77.46	2

[1] *N* indicates the number of studies of each type that were combined to yield the *R*, R^2, Slope and Y-Intercept values.

APPENDIX B

Risk Assessment Data

Table 14: Values used in the Calculation of Cancer Risk for Populations Exposed to Arsenic

Population	IR (kg/meal)	FI (unitless)	EF (meals/year)	ED (years)	BW (kg)	AT (days)
Boys 1-6 years old						
Carrot	0.043	0.4	100	70	16.95	25500
Lettuce	0.065	0.4	100	70	16.95	25500
Radish	0.00078	0.4	52	70	16.95	25500
Girls 1-6 years old						
Carrot	0.043	0.4	100	70	16.23	25500
Lettuce	0.065	0.4	100	70	16.23	25500
Radish	0.00078	0.4	52	70	16.23	25500
Average Male						
Carrot	0.043	0.4	100	70	78.1	25500
Lettuce	0.065	0.4	100	70	78.1	25500
Radish	0.00078	0.4	52	70	78.1	25500
Average Female						
Carrot	0.043	0.4	100	70	65.4	25500
Lettuce	0.065	0.4	100	70	65.4	25500
Radish	0.00078	0.4	52	70	65.4	25500
55+ Men						
Carrot	0.043	0.4	100	70	76.8	25500
Lettuce	0.065	0.4	100	70	76.8	25500
Radish	0.00078	0.4	52	70	76.8	25500
55+ Women						
Carrot	0.043	0.4	100	70	67.25	25500
Lettuce	0.065	0.4	100	70	67.25	25500
Radish	0.00078	0.4	52	70	67.25	25500

Table 15: Values Used in the Calculation of Hazard Quotients for Populations Exposed to Cadmium and Lead.

Population	IR (kg/meal)	FI (unitless)	EF (meals/year)	ED (years)	BW (kg)	AT (days)
Boys 1-6 years old						
Cabbage	0.068	0.4	37	30	16.95	10950
Lettuce	0.065	0.4	100	30	16.95	10950
Radish	0.00078	0.4	52	30	16.95	10950
Girls 1-6 years old						
Cabbage	0.068	0.4	37	30	16.23	10950
Lettuce	0.065	0.4	100	30	16.23	10950
Radish	0.00078	0.4	52	30	16.23	10950
Average Male						
Cabbage	0.068	0.4	37	30	78.1	10950
Lettuce	0.065	0.4	100	30	78.1	10950
Radish	0.00078	0.4	52	30	78.1	10950
Average Female						
Cabbage	0.068	0.4	37	30	65.4	10950
Lettuce	0.065	0.4	100	30	65.4	10950
Radish	0.00078	0.4	52	30	65.4	10950
55+ Men						
Cabbage	0.068	0.4	37	30	76.8	10950
Lettuce	0.065	0.4	100	30	76.8	10950
Radish	0.00078	0.4	52	30	76.8	10950
55+ Women						
Cabbage	0.068	0.4	37	30	67.25	10950
Lettuce	0.065	0.4	100	30	67.25	10950
Radish	0.00078	0.4	52	30	67.25	10950

Table 16: Values Used to Calculate Hazard Quotients for Populations Exposed to Zinc.

Population	IR (kg/meal)	FI (unitless)	EF (meals/year)	ED (years)	BW (kg)	AT (days)
Boys 1-6 years old						
Lettuce	0.065	0.4	100	30	16.95	10950
Radish	0.00078	0.4	52	30	16.95	10950
Girls 1-6 years old						
Lettuce	0.065	0.4	100	30	16.23	10950
Radish	0.00078	0.4	52	30	16.23	10950
Average Male						
Lettuce	0.065	0.4	100	30	78.1	10950
Radish	0.00078	0.4	52	30	78.1	10950
Average Female						
Lettuce	0.065	0.4	100	30	65.4	10950
Radish	0.00078	0.4	52	30	65.4	10950
55+ Men						
Lettuce	0.065	0.4	100	30	76.8	10950
Radish	0.00078	0.4	52	30	76.8	10950
55+ Women						
Lettuce	0.065	0.4	100	30	67.25	10950
Radish	0.00078	0.4	52	30	67.25	10950

VITA

TRENT D. LECOULTRE

Personal Data:	Date of Birth: November 8, 1975 Place of Birth: Knoxville, Tennessee Marital Status: Married
Education:	Gibbs High School, Corryton, Tennessee East Tennessee State University, Johnson City, TN Geography/Geology, B.S., 1999 East Tennessee State University, Johnson City, TN Environmental Health, M.S.E.H., 2001
Professional Experience:	Geographic Information Systems Project Manager, Dept. of Geography, Geology and Geomatics, East Tennessee State University, Johnson City, TN. 1998-1999 Graduate Research Assistant, Environmental Health Sciences Laboratory, East Tennessee State University, Johnson City, TN. 2000-2001
Presentations:	LeCoultre, T.D., Scheuerman, P.R., 2001. A Meta-Analysis and Risk Assessment of Heavy Metal Uptake in Common Garden Vegetables. Society of Environmental Toxicology and Chemistry, Baltimore, MD, 2001. Poster Presentation.
Honors:	Epsilon Nu Eta—Environmental Health Honor Society Vice President—Geo-Science Society