

Future risk from a hypothesized RCRA site disposing of carcinogenic metals should a loss of societal memory occur

David Okrent* and Leiming Xing

Department of Mechanical, Aerospace and Nuclear Engineering, 48-121 Engineering IV, University of California at Los Angeles, Los Angeles, CA 90024 (USA)

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Abstract

The future risk of a hypothesized Resource Conservation and Recovery Act (RCRA) site disposing of carcinogenic metals, arsenic, chromium, nickel, cadmium, and beryllium in the U.S. is assessed. Societal memory is assumed to be lost regarding the site. A human intrusion scenario on the site and a residential scenario one kilometer down-gradient of the groundwater flow direction from the site are assumed, starting at 1000 years after the site's closure. For the human intrusion scenario, the exposure pathways considered are fruit and vegetable intake, soil ingestion, and dermal contact with soil. The quantitative results obtained for the three pathways are as follows: lifetime excess cancer risk due to fruit and vegetable intake is 0.18; risk due to dermal contact with the soil is 0.12; and risk due to soil ingestion is 2.6×10^{-3} . For the residential scenario, only qualitative discussion of exposure via groundwater is presented due to the large uncertainties. The U.S. EPA (Environmental Protection Agency) attention to and requirements concerning long-term risk from RCRA sites containing metal carcinogens, which never change due to radioactive decay, stand in sharp contrast to the stringent requirements over 10,000 years posed by EPA for geologic disposal of high level radioactive wastes, and the long-term requirements posed by the U.S. Nuclear Regulatory Commission for low level radioactive waste disposal sites.

1. Introduction

Toxic metals are usually considered persistent in the environment in that, unlike radioactive isotopes, metals do not decay in the environment. This raises a serious concern about whether and how we can dispose of the wastes containing such toxic metals, given that society chooses to use them. As is well known, in 40 CFR 191 (remanded), EPA poses stringent requirements, including loss of societal memory, over 10,000 years for geologic disposal of high level radioactive wastes, despite the presence of markers at the site [1]. Also, the

*To whom correspondence should be addressed.

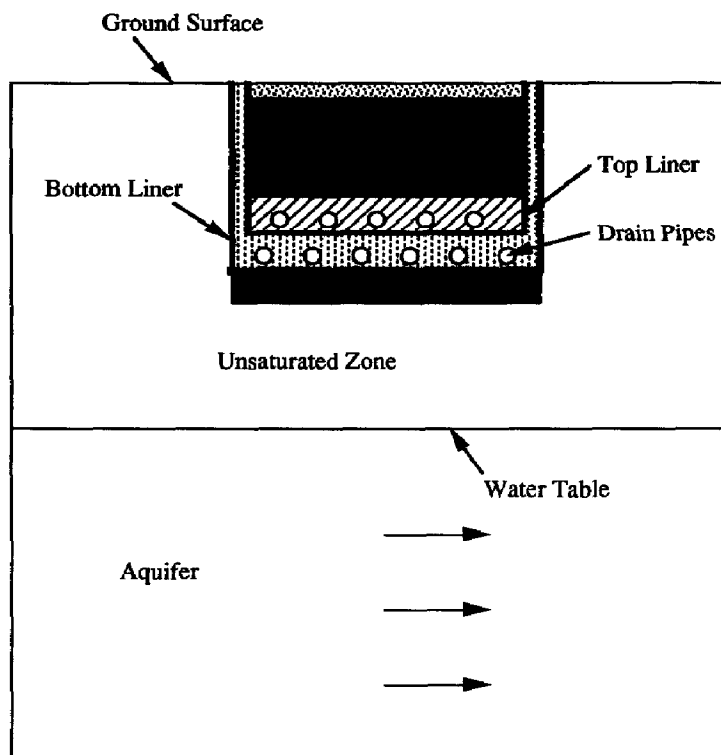
U.S. Nuclear Regulatory Commission (USNRC) requires long-term risk assessment for low level radioactive waste disposal sites [2]. This paper is to address the disparity between the EPA regulation regarding carcinogenic metals and the requirements posed on high level radioactive wastes.

Five metals with potential carcinogenicity are the focus of this paper. These are arsenic, cadmium, chromium, nickel, and beryllium. The future risks of a hypothesized Resource Conservation and Recovery Act (RCRA) site which disposed of these metals are assessed herein using similar scenarios and assumptions to those of a high level or a low level radioactive waste disposal site risk assessment [2]. Among the important assumptions, societal memory is assumed lost regarding the hypothesized RCRA site; human intrusion may occur at the site; and there may be a residential scenario down-gradient of the groundwater flow direction.

Section 2 is a description of the hypothesized RCRA site. Section 3 is a review of toxicity and current usage of five carcinogenic metals. Section 4 discusses the waste treatment and treatment standards imposed by EPA. Section 5 hypothesizes a RCRA site inventory that meets the current EPA requirements. Section 6 assumes scenarios for the future risk analysis; these include the human intrusion scenario and residential scenario. Section 7 discusses models of several exposure pathways as well as presents qualitative discussion on groundwater contamination. Section 8 presents results for the human intrusion scenario. Section 9 is a discussion concerning the implications of this analysis.

2. Site description

Figure 1 shows a hypothesized RCRA site. This site is assumed to be used to dispose of wastes containing several carcinogenic metals and their compounds. These wastes need to be treated before being disposed of. After treatment, different kinds of containment, such as Above Grade Mound, Above Grade Vault and Above Grade Building as well as Below Grade Landfill [3], can be used to hold the wastes. The difference between above grade and below grade containment is that the below grade containment holds the wastes below the ground surface. In this analysis, it is assumed that the RCRA site uses a Below Grade Landfill with double liners and leachate collection system. The basic design features of the hypothesized site are similar to those of the waste sites reported by the Minnesota Waste Management Board [3]. As shown in Fig. 1, the lower most layer is recompacted clay with at least three feet thickness. This layer of clay has low permeability. On top of the clay is a secondary flexible membrane liner which is at least 30 mils (0.03 inch) thick [3]. Above the liner is the secondary leachate collection system which is able to drain off fluid which would reach the liner. Above the secondary leachate collection system is the primary flexible membrane liner. It is also required to be at least 30 mils thick. On the top of this liner lies the primary leachate collection system. The



Legend:






	Caps		Wastes
	Drainage Materials		Drainage Materials
	Recompacted Clay		

Fig. 1. A hypothesized RCRA site.

primary leachate collection system is covered by a permeable membrane over which the waste is placed. Above the wastes, there are vegetative caps. The caps might consist of three layers, as used in a low level radioactive waste site [2]. The bottom layer is a low-permeability clay, the middle layer a silt loam, and the top layer is the original undisturbed soil. The total thickness of these layers might be 2 m.

About the location of the hypothesized site, the EPA requirement is the following according to Wagner [4]: “Currently, the only location restrictions are: The facility must be at least 200 feet from an active (during the last 10,000 years) Holocene fault; Facilities in a 100-year flood plain must be designed to prevent washout from 100-year floods.”

“More stringent location standards, as mandated by HSWA (Hazardous and Solid Waste Amendments), were expected to be issued in 1992. The purpose of these standards will be to create national requirements for the location of the

TABLE 1

Parameters for the hypothesized site (adapted from Table 4–1 of Ref. [2])

Parameter	Value
Average annual infiltration	25 cm/y
Thickness of unsaturated zone	24 m
Saturated hydraulic conductivity of unsaturated zone	115.3 m/y
Porosity of unsaturated zone	0.52
Thickness of aquifer	25 m
Hydraulic conductivity of aquifer	115.3 m/y
Hydraulic gradient of aquifer	0.02
Porosity of aquifer	0.52
Average pore velocity of aquifer	4.44 m/y
Longitudinal dispersivity of aquifer	2 m
Transverse dispersivity of aquifer	0.2 m

hazardous waste management facilities. These requirements will contain restrictions based on proximity to populations, vulnerable hydrogeology, seismic zones, 100-year flood plains, poor foundation areas, subsidence-prone areas, landslide-prone areas, wetlands, and karst terranes (limestones areas with fissures, sink-holes, underground streams, and caverns).”

While not violating the current site requirements, the site is assumed on a local topographic high point. The water table is located about 24 m below the land surface. The aquifer is 25 m thick and is confined from below by an impermeable bedrock.

Table 1 lists the geological parameters which could be used to characterize the hypothesized site if groundwater contamination were to be studied. The choice of this parameter set is consistent with a hypothesized site used for disposal of low-level radioactive wastes [2]. The characteristics are permissible under the current RCRA regulations [4]; of course, a wide variety of sites are in actual use.

Actual sites would vary in their characteristics from that of the hypothesized site. For example, porosity might be 0.35. However, such differences are not important herein since these site characteristics were not used in these calculational results. A groundwater source was not modeled quantitatively in the risk assessment.

3. Carcinogenicity and current usage of several metals

The contaminants considered in the sites are several metals and their compounds. Five metals which are carcinogenic are listed in Table 2 [5]. In the EPA weight-of-evidence classification system for carcinogenicity, A stands for human carcinogen, B1 stands for probable human carcinogen, and B2 stands

TABLE 2

Carcinogenic group of metals (adapted from Ref. [5])

Metal	EPA Group	Description
As	A	Human carcinogen, sufficient human evidence
Be	B2	Probable human carcinogen, inadequate human evidence, sufficient animal evidence
Cd	B1	Probable human carcinogen, limited human evidence
Cr(VI)	A	Human carcinogen, sufficient human evidence
Ni (subsulfide)	A	Human carcinogen, sufficient human evidence

TABLE 3

Annual production of the four metals (data collected from Ref. [6])

Metal	Total annual world production (metric tons)	Year of production
As	60,000	1975–1977
Be	10,000	not specified
Cd	12,000	1980
Cr	8,600,000	1976
Ni	660,000	1973

for probable human carcinogen with limited human evidence, but with sufficient animal evidence. Arsenic, chromium (VI), and nickel (subsulfide) are considered by EPA to be sufficient for them to be labeled as carcinogens. For cadmium, EPA considers human evidence as to its carcinogenicity to be limited; hence cadmium is ranked lower (category B1) in Table 2. Beryllium belongs to category B2.

Annual production of these five metals are listed in Table 3 [6]. It can be seen that the amount of metals used is large. After the technologically useful lifetime of those products containing these metals, one or more RCRA sites is needed to dispose of them. Recycle of all the metals is impractical. The costs to separate and encapsulate the vast tonnage of most of these widely used carcinogens would be very large, even more so for existing wastes.

Carcinogenicity data of these five metals are listed in Table 4. The slope factor is usually, but not always, the upper 95th percent confidence limit of the slope of the dose–response curve and is expressed as $(\text{mg}/\text{kg}\cdot\text{day})^{-1}$ [7]. It represents the lifetime excess cancer risk per unit dose. Usually, the slope factors are different for inhalation and ingestion. They are related to the unit risks as follows [7]:

$$\text{air unit risk} = \text{risk per } \mu\text{g}/\text{m}^3 = \text{slope factor} \times 1/70 \text{ kg} \times 20 \text{ m}^3/\text{day} \times 10^{-3}$$

TABLE 4

Unit risks and slope factors of five metals

Metal	Air unit risk ^a ($\mu\text{g}/\text{m}^3$) ⁻¹	Slope factor for inhalation ^b ($\text{mg}/\text{kg}\cdot\text{day}$) ⁻¹	Water unit risk ($\mu\text{g}/\text{L}$) ⁻¹	Slope factor for ingestion ($\text{mg}/\text{kg}\cdot\text{day}$) ⁻¹
As	4.3×10^{-3}	15	$5.1 \times 10^{-4\text{c}}$	18 ^d
Cr(VI)	1.2×10^{-2}	42	- ^e	- ^e
Ni (sulfide)	4.8×10^{-4}	1.7	- ^e	- ^e
Cd	1.8×10^{-3}	6.3	- ^e	- ^e
Be	2.4×10^{-3}	8.4	$1.2 \times 10^{-4\text{f}}$	4.3 ^f

^aData in this column is taken from IRIS [8].^bCalculated from air unit risk, using 20 m³/day as the inhalation rate.^cCalculated from slope factor for ingestion, using 2 L/day as the ingestion rate.^dCalculated based on data from Smith et al. [9].^eNo data is available.^fTaken from IRIS [8].

and

$$\text{water unit risk} = \text{risk per } \mu\text{g}/\text{L} = \text{slope factor} \times 1/70 \text{ kg} \times 2 \text{ L}/\text{day} \times 10^{-3}$$

Several aspects of these five metals are discussed in the following.

3.1 Arsenic

Epidemiological studies have indicated that exposure via ingestion to inorganic arsenic compounds in drugs and drinking water is causally related to the development of skin cancer in humans. Exposure to arsenic trioxide by inhalation is clearly related to the development of lung cancer in certain smelter workers [5]. The Integrated Risk Information System (IRIS) [8], an EPA on-line data base, gives an air unit risk of 4.3×10^{-3} (mg/m^3)⁻¹. Therefore an inhalation slope factor of 15 ($\text{mg}/\text{kg}\cdot\text{day}$)⁻¹ can be calculated. The IRIS also gives a water unit risk of 5×10^{-5} ($\mu\text{g}/\text{L}$)⁻¹ for skin cancer due to ingestion of arsenic in water. A corresponding ingestion slope factor is 1.8 ($\text{mg}/\text{kg}\cdot\text{day}$)⁻¹. However, Smith et al. [9] reported that arsenic can also cause liver, lung, kidney, and bladder cancer. They estimated that at the current EPA drinking water standard of 50 $\mu\text{g}/\text{L}$, the lifetime risk of dying from cancer of the liver, lung, kidney, or bladder from drinking 1 L/day of water could be as high as 13 per 1000 persons. A ingestion slope factor of 18 ($\text{mg}/\text{kg}\cdot\text{day}$)⁻¹ can be calculated; this value is listed in Table 4. Also, Anderson [10] gave a ingestion slope factor of 14.0 ($\text{mg}/\text{kg}\cdot\text{day}$)⁻¹ for arsenic.

The major current uses of arsenic [6] are as pesticides, e.g., lead arsenate, calcium arsenate, and sodium arsenite, herbicides, and cotton desiccants. Elemental arsenic is utilized as an additive in the production of several alloys

to increase hardness and heat resistance. Gallium arsenide, an artificial crystal, has become an important material in the manufacture of integrated circuits.

3.2 Chromium (VI)

Epidemiologic studies of chromate production facilities in the United States, Great Britain, Japan, and Germany have established an association between chromium exposure and lung cancer [8]. According to IRIS, chromium-exposed workers are exposed to both chromium III and chromium VI compounds. However, only chromium VI has been found to be carcinogenic in animal studies. Therefore, only chromium VI is concluded as carcinogenic. The inhalation unit risk of chromium is $1.2 \times 10^{-2} (\mu\text{g}/\text{m}^3)^{-1}$ [8], which corresponds to a inhalation slope factor of $42 (\text{mg}/\text{kg}\text{-day})^{-1}$. No data is reported in IRIS regarding the ingestion slope factor for chromium. However, in the National Primary Drinking Water Regulations (40 CFR parts 141, 142, 143), EPA concludes that the presence of chromium (VI) in drinking water should be minimized in recognition of its biological reactivity including its potential for a carcinogenic hazard, and recommends that the uncertainty regarding chromium (VI) carcinogenic risk in drinking water warrants additional investigation. Cohen used a ratio of oral slope factor/inhalation slope factor of 1/10 to derive an oral slope factor for chromium [11].

The principal industrial consumers of chromium are the metallurgical, refractory and chemical industries [6]. The U.S. figures for consumption by these industries were 60%, 20% and 20%, respectively.

3.3 Nickel

Increased risks of lung and nasal cancer have been reported in humans exposed to nickel refinery dust, most of which was believed to have been nickel subsulfide [8]. The IRIS gives an air unit risk of $4.8 \times 10^{-4} (\mu\text{g}/\text{m}^3)^{-1}$, which corresponds to an inhalation slope factor of $1.7 (\text{mg}/\text{kg}\text{-day})^{-1}$ for nickel (subsulfide). No data has been reported regarding the ingestion slope factor.

About 40% of the nickel produced is used in steel production [6]. Nickel is also used in the production of other alloys, for which reason nickel can be found in coins and household utensils. Electroplating using nickel sulfate accounts for 20% of the nickel produced. Nickel hydroxide is used in nickel-cadmium batteries. Nickel carbonate serves in electronic components, such as vacuum tubes and transistor cans.

3.4 Cadmium

Limited evidence from occupational epidemiologic studies of cadmium is consistent across investigators and study populations [8]. Occupational exposure to cadmium (primarily as the oxide) increases the risk of prostate, respiratory, and genitourinary cancers in humans [5]. Excess cancer risk of lung cancer was observed in cadmium smelter workers [8]. There is sufficient evidence of carcinogenicity in rats and mice by inhalation and intramuscular and

subcutaneous injection [8]. Seven studies in rats and mice wherein cadmium salts (acetate, sulfate, chloride) were administered orally have shown no evidence of carcinogenic response [8]. The IRIS gives an air unit risk of $1.8 \times 10^{-3} (\mu\text{g}/\text{m}^3)^{-1}$, which corresponds to an inhalation slope factor of $6.3 (\text{mg}/\text{kg}\text{-day})^{-1}$. According to 40 CFR parts 141, 142, 143, “chronic oral animal studies with cadmium have shown kidney damage but no carcinogenic activity and ingestion-specific human data are not available”. Also, “those comments that conclude that cadmium is a carcinogen provide no new evidence that cadmium is carcinogenic via drinking water but rather, argue that it is prudent to assume that cadmium is carcinogenic via ingestion”. Cohen gives a ratio of oral slope factor/inhalation slope factor of 1/3 in his study [11].

Cadmium is used in a number of industrial processes, but for most of its uses there are alternatives of lower toxicity [6]. In the U.S., 60% of the cadmium produced or imported was used for plating, 11% in color pigments, 19% as stabilizers in plastics, 3% in accumulators, and 7% for other purposes.

3.5 Beryllium

Beryllium has been shown to induce lung cancer via inhalation in rats and monkeys and to induce osteosarcomas in rabbits via intravenous or intramedullary injection [8]. Human epidemiology studies are considered to be inadequate [8]. The IRIS gives an air unit risk of $2.4 \times 10^{-3} (\mu\text{g}/\text{m}^3)^{-1}$, which corresponds to an inhalation slope factor of $8.4 (\text{mg}/\text{kg}\text{-day})^{-1}$. The water unit risk of $1.2 \times 10^{-4} (\mu\text{g}/\text{L})^{-1}$ and ingestion slope factor of $4.3 (\text{mg}/\text{kg}\text{-day})^{-1}$ are both taken from IRIS. Anderson [10] reported the ingestion slope factor of $4.9 (\text{mg}/\text{kg}\text{-day})^{-1}$ derived from the linearized multistage model and $3.4 (\text{mg}/\text{kg}\text{-day})^{-1}$ derived from the one-hit model.

About 20% of the world production of beryllium is used as free metals. Beryllium alloys account for about 72% of the total production. The master alloy is 96% copper and 4% beryllium. The remaining 8% of beryllium produced is used as the oxide in ceramic formulations [6].

4. Wastes treatment and treatment standards

As mentioned, the hazardous waste has to be treated before land disposal. Stabilization and Solidification are commonly used treatment methods. For inorganic wastes stabilization/solidification, there are two recommended methods which are used for setting Best Demonstrated Alternative Technology (BDAT) standards for many wastes [12]. These are cement-based stabilization/solidification and pozzolanic stabilization/solidification. Descriptions about these two technologies are adapted from reference [12] as follows:

“Cement-based stabilization/solidification is a process in which waste materials are mixed with portland cement. Water is added to the mixture to ensure proper hydration reactions necessary for bonding the cement. The wastes are

incorporated into the cement matrix and, in some cases, undergo physical-chemical changes that further reduce their mobility in the waste-cement matrix. Typically, hydroxides of metals are formed, which are much less soluble than ionic species of the metals. The final product may vary from a granular, soil-like material to a cohesive solid, depending on the amount of reagent added and the types and amounts of wastes stabilized/solidified. The Cement-based stabilization/solidification has been applied to plating wastes containing various materials such as cadmium, chromium, copper, lead, nickel, and zinc, etc.”

“Pozzolanic stabilization/solidification involves siliceous and aluminosilicate materials, which do not display cementing action alone, but form cementitious substances when combined with lime or cement and water at ambient temperatures. The primary containment mechanism is the physical entrapment of the contaminant in the pozzolan matrix. Examples of common pozzolans are fly ash, pumice, lime kiln dusts, and blast furnace slag. Pozzolans contain significant amounts of silicates, which distinguish them from the lime-based materials. The final product can vary from a soft fine-grained material to a hard cohesive material similar in appearance to cement. Pozzolanic reactions are generally much slower than cement reactions. Waste materials that have been stabilized/solidified with pozzolans include oil sludges, plating sludges containing various metals (aluminium, nickel, copper, lead, chromium, and arsenic), waste acid, and creosote.”

Both cement-based and pozzolanic stabilization/solidification aim at immobilizing the metals in the wastes, instead of removing the metals permanently. The choice of treatment technology as well as achievable standards are much waste type dependent.

The land disposal restrictions, codified in 40 CFR 268, place stringent controls on the land disposal of hazardous wastes [4]. Congress set forth a schedule of land disposal restrictions in HSWA. The statute automatically prohibited the land disposal of hazardous wastes if EPA failed to set a treatment standard by the statutory deadline. The statute also required EPA to make determinations on prohibiting land disposal, within the indicated time frames, for the following:

1. At least one-third of all ranked and listed hazardous wastes by August 8, 1988.
2. At least two-thirds of all ranked and listed hazardous wastes by June 8, 1989.
3. All remaining ranked and listed hazardous wastes and all hazardous wastes identified by a characteristic by May 8, 1990.

Within the first third wastes, there are K101 and K102 wastes. K101 wastes are distillation tar residues from the distillation of aniline-based compounds in the production of veterinary pharmaceuticals from arsenic or organo-arsenic compounds (40 CFR 268.10). K102 wastes are residues from the use of activated carbon for dechlorination in the production of veterinary pharmaceuticals from arsenic or organo-arsenic compounds (40 CFR 268.10).

TABLE 5

BDAT Treatment standards for K101 waste [12] (nonwaste waters) (Low arsenic subcategory — less than 1% arsenic)

Constituent	Maximum for any single grab samples	
	Total composition (mg/kg)	TCLP (mg/L)
<i>Ortho</i> -nitroaniline	14	(1)
Cadmium	(1)	0.066
Chromium (total)	(1)	5.2
Lead	(1)	0.51
Nickel	(1)	0.32

(1) Not applicable.

On August 17, 1988, EPA promulgated treatment standards for hazardous wastes listed in 40 CFR 268.10. For K101 and K102 wastes (ash residues), the Best Demonstrated Available Technology chosen is stabilization [12]. EPA also established BDAT treatment standards. The BDAT standards for K101 waste are shown in Table 5. While these standards are derived based on BDAT, they are concentration standards. In other words, any technology satisfying this standard is acceptable. The BDAT standards for K102 wastes are the same as given in Table 5.

In Table 5, TCLP is the Toxicity Characteristic Leaching Procedure (Federal Register 1986). It presumably represents the maximum leaching concentration of the waste. It can be seen from the table, for metals cadmium, chromium, and nickel, only TCLP is specified. No limits are given on the total composition of these metals.

5. The waste inventory of the hypothesized RCRA site

Based on the above discussion, we can hypothesize a RCRA site containing considerable amounts of arsenic and other metals. For simplicity, it can be assumed that the waste consists mainly of distillation tar residues from the production of veterinary pharmaceuticals (i.e., K101 wastes), which can have up to 1% weight content of arsenic. Also, other kinds of waste which contains more of chromium, cadmium, nickel, and beryllium have been mixed with the K101 waste. The stabilization process has been chosen to treat the mixture of these wastes. The treated wastes are assumed fine grained, soil-like in form and meet the TCLP standards as discussed above. So conceivably, one can have a RCRA site of the inventory as shown in Table 6. The maximum allowed leachate concentration of arsenic is set to comply with the Toxicity Characteristic Level of 5.0 mg/L [4].

TABLE 6

Inventory of the hypothesized RCRA site

Metal	Weight content (mg/kg)	Maximum allowed leachate concentration (mg/L)
Arsenic	1,000	5.0
Beryllium	100	N.A.
Cadmium	100	0.066
Chromium	100	5.2
Nickel	100	0.32

6. Scenarios for future risk assessment

It should be acknowledged that EPA indeed considers the near future situation of a RCRA site. For example, EPA currently requires that a site is not in a seismic active region and it should not be in a 100 year flood plain. Also, leachate collection and post-closure monitoring for groundwater contamination should last for 30 years.

However, in contrast to the stringent standards EPA imposed on the geological disposal of high level radioactive wastes [1] as well as the long term consideration of low level radioactive waste disposal [2], the requirement of long term risk assessment for RCRA site disposal of persistent carcinogenic metals is lacking. The authors believe there is an inconsistency in current regulation practices. Therefore in this text, an effort is made to do a future risk assessment of the hypothesized RCRA site, given that the societal memory has been lost in a distant future, e.g., a thousand years later.

Two scenarios are considered in this study. One is a human intrusion scenario, the other is a residential scenario down-gradient of the site.

6.1 Human intrusion

A human intrusion scenario similar to that analyzed for a low-level radioactive waste disposal site [2] is considered. It is assumed that 1000 years after the site closure, a group of farmer families intrude the site. During this 1000 years, there is a thousand-year flood which washes out the caps of the site. The farmers excavate the disposal facility area and build houses directly on the site as shown in Fig. 2. The farmers dig wells just down gradient of the waste disposal facility. The groundwater serves for their family's drinking water and is also used to irrigate the fruits and vegetables in case the weather is dry. The families consume part of the fruits and vegetables produced on the site. The farmers contact the soil during various activities.

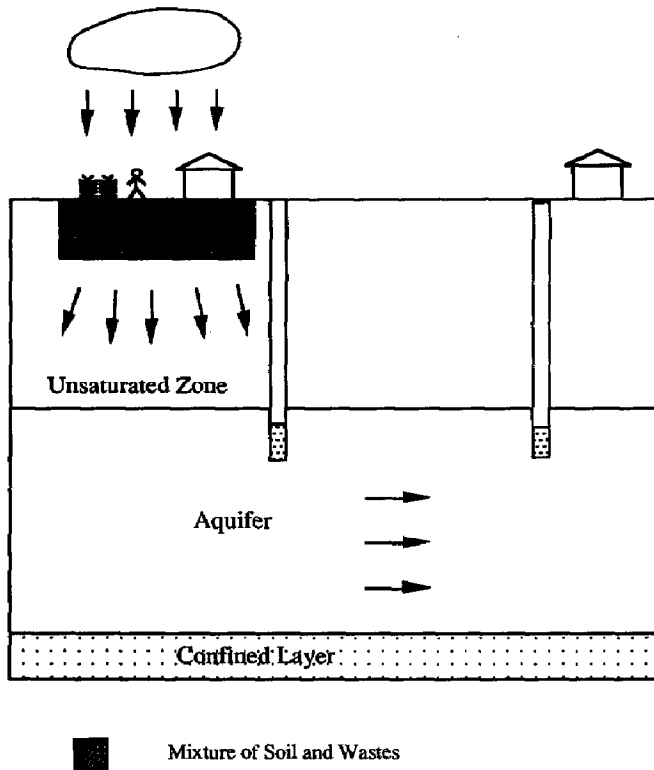


Fig. 2. Intrusion and residential scenarios.

6.2 Residential scenario (1000 m down-gradient of the site)

It is assumed that present day precipitation and infiltration data do not change dramatically in the future. The precipitation at the site will cause percolation of water through the cover system of the facility into the disposal units. The double liners and leachate collection system are assumed to lose functionality after 100 years of site closure. The wastes might be transported down into the aquifer and be further transported in the aquifer.

Suppose at 1000 years a community resides 1000 m down-gradient of the groundwater flow direction from the site and people use groundwater as their drinking water. The cancer risk should be considered.

7. Exposure pathway modeling

7.1 Inventory losses

We assume the site inventory can be approximated by an exponential loss equation as follows

$$m_i(t) = m_{i,0} e^{-\lambda t} \quad (1)$$

where $m_i(t)$ is the mass inventory of metal i at time t after the liners lose their functions. Here, i refers to five metals arsenic, chromium, cadmium, nickel, and beryllium. $m_i(t)$ accounts for all the forms of metal i ; $m_{i,0}(t)$ is the initial mass inventory of metal i . The inventory loss rate, λ , can be expressed as

$$\lambda = \ln 2 / T_{1/2} \quad (2)$$

where $T_{1/2}$ is the half time of inventory losses. The basic loss mechanism is leachate into the soil and then groundwater. If metal i changes from one chemical form to another, it is not counted as a loss.

It can be assumed that the total volume of the wastes does not change with time, so the metal concentration also observes an exponential law, i.e.

$$C_{s,i}(t) = C_{s,i,0} e^{-\lambda t} \quad (3)$$

where $C_{s,i}(t)$ (mg/kg) is the concentration of metal i at time t . The value of $C_{s,i,0}$ has been listed in Table 6.

The loss rate λ depends on many factors, such as pH level of the soil, the infiltration rate, total metal inventory in the site, the area of the site, etc. In this analysis, we assume that after liners lose their functions, the initial leachate concentration does not exceed that given in column 3 of Table 6. This assumption gives a very small loss rate λ . As a result, the metal concentration at 1000 years does not change very much from the initial concentration; this is effectively the case with $\lambda=0$. We thus use the initial weight concentration in column 2 of Table 6 for $C_{s,i}$ in the human intrusion scenario. It is noted that the treatment of λ herein neglects the groundwater contamination, which might be an important pathway. A realistic value of the loss rate λ is needed for more elaborate models on this topic.

7.2 Human intrusion scenario

In principle, there are at least five pathways considered possible for the human intrusion scenario. They are: ingestion of contaminated groundwater, inhalation of dust containing metal carcinogens, ingestion of contaminated fruits and vegetables, dermal contact with contaminants in the soil, and ingestion of metals in the soil.

There are large uncertainties in modelling groundwater contamination for carcinogenic metals. These include the effect of retardation, uncertainties about the influence of chemical reactions, and uncertainties in geological configuration. The groundwater ingestion pathway is not modeled herein because: (1) we use a zero loss rate λ ; (2) the heavy metals are believed to have low mobility in the ground. But groundwater contamination may be a significant contributor to risk for the human intrusion scenario. The inhalation of dust pathway is not considered quantitatively, either, due to large uncertainties.

7.2.1 Fruit and vegetable ingestion

Homegrown vegetables and fruits are classified into three groups, i.e. leafy vegetables, exposed produce, and protected produce [5]. Leafy vegetables present a broad, flat leaf surface for direct interception of depositing pollutant. Exposed produce such as tomatoes, apples, etc. present edible portions for direct deposition pathway, but edible portions generally have reproductive functions and are associated with significantly different soil-plant uptake parameters than leafy vegetables. Protected produce such as potatoes and citrus fruits do not have edible portions exposed to direct atmospheric deposition. Like exposed produce, edible portions are not vegetative in nature. Therefore, the soil-plant transfer coefficient of a pollutant for vegetative portions is applied to the soil-plant uptake for leafy vegetables while the transfer coefficient of the pollutant for reproductive portions is applied to the soil-plant uptake for exposed or protected produce.

Neglecting the atmospheric deposition, the concentration of contaminant in homegrown fruits and vegetables can be calculated as [5]

$$C_{P,i} = (B_v \text{ or } B_r) C_{s,i} \quad (4)$$

where $C_{s,i}$ (mg/kg) is the contaminant concentration as calculated in (3). B_v is the soil-plant elemental transfer coefficient for vegetative portions (unitless), B_r is the soil-plant elemental transfer coefficient for reproductive portions (unitless). Table 7 [5] represents the soil-to-plant transfer coefficient for different metals.

The fruit and vegetable intake can be estimated as [5]

$$IT_{FV,i} = \frac{C_{P,i} \times IR \times FI \times EF \times ED}{BW \times AT} \quad (5)$$

where $IT_{FV,i}$ is the fruit and vegetable intake of contaminant i (mg/kg·day), $C_{P,i}$ (mg/kg) is contaminant concentration in fruits and vegetables as calculated in (4), and IR (kg/day) is the ingestion rate. Total daily consumption rate for each category of vegetable and fruit is assumed to be 38 g/day for leafy vegetables, 82 g/day for exposed produce and 153 g/day for protected produce,

TABLE 7

Soil-to-plant transfer coefficient (adapted from Ref. [5])

Pollutant	B_v	B_r
As	0.040	6.0×10^{-3}
Be	0.01	1.5×10^{-3}
Cd	0.55	0.15
Cr	7.5×10^{-3}	4.5×10^{-3}
Ni	0.060	0.060

respectively [5]. Therefore, the ingestion rate for vegetative portions is 0.038 kg/day (leafy vegetables and fruits), and the ingestion rate for reproductive portions is 0.235 kg/day (sum of exposed produce and protective produce). FI is fraction ingested from a contaminated source. The home grown portion of total vegetable and fruit consumed is assumed to be 25% regardless of the category [5]. Therefore FI is taken 0.25 for both vegetative and reproductive portions. EF (days/year) is the exposure frequency, for which a value of 365 day/y is assumed. $ED=70$ years is exposure duration; $BW=70$ kg is body weight; and $AT=70\text{ y} \times 365\text{ days/y}$ is the averaging time. It is noted that no allowance is made for contaminated water in this calculation.

To calculate the lifetime cancer risk, a proper dose-response model should be used. For extrapolating from high dose to low dose, "EPA's guidelines recommend that the linearized multistage model be employed in the absence of adequate information to the contrary. Among the other models available are the Weibull, probit, logit, one-hit, and gamma multihit models, as well as various time-to-tumor models" [7].

Since the intake in this analysis might be high, we choose the one-hit equation for risk calculation. The lifetime excess cancer risk due to fruit and vegetable intake can be calculated as [7]

$$R_{FV,i} = 1 - \exp(-IT_{FV,i} \times SF_i) \quad (6)$$

where $R_{FV,i}$ is the cancer risk posed by metal i ; and SF_i is the ingestion slope factor for metal i , as shown in Table 4.

7.2.2 Dermal contact with metals in soil

The absorbed dose due to dermal contact with chemicals in soil $IT_{AD,i}$ (mg/kg day) can be calculated as [7]

$$IT_{AD,i} = \frac{C_{s,i} \times CF \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \quad (7)$$

where $C_{s,i}$ is the chemical concentration in soil (mg/kg) as calculated in (3); $CF=10^{-6}$ kg/mg is the conversion factor; SA is the skin surface area available for contact (cm^2/event), the 50th percentile body part-specific surface areas for males is 0.23 m^2 for arms, 0.082 m^2 for hands and 0.55 m^2 for legs, hence SA is assumed to be the sum of these areas, i.e., 8620 cm^2 ; AF is the soil to skin adherence factor (mg/cm^2). It is assumed that a layer of soil with particle size 0.01 mm forms on the skin, the soil density is assumed to be 2 g/cm^3 , the product of thickness and soil density gives an AF value of 2 mg/cm^2 ; ABS is the absorption factor, a value of 0.1 is assumed; EF is the exposure frequency, a value of 100 events/year is assumed; ED , BW , and AT have the same values as discussed above.

The risk due to dermal contact with chemical i in soil $R_{DC,i}$ is

$$R_{DC,i} = 1 - \exp(-IT_{AD,i} \times SF_i) \quad (8)$$

7.2.3 Ingestion of metals in soil

The intake of metal i , $IT_{Si,i}$ (mg/kg day), due to ingestion of soil can be expressed as [7]

$$IT_{Si,i} = \frac{C_{s,i} \times IR \times CF \times FI \times EF \times ED}{BW \times AT} \quad (9)$$

where $C_{s,i}$ is the chemical concentration in soil (mg/kg) as calculated in (3); $IR = 100$ mg/day is the ingestion rate for the age group greater than 6 years old; $CF = 10^{-6}$ kg/mg is the conversion factor; FI is the fraction ingested from contaminated sources, a value of 0.1 is chosen for FI ; $EF = 365$ days/year is the exposure frequency. ED , BW , and AT have the same meaning and values as discussed before.

The risk due to soil ingestion can be calculated as

$$R_{Si,i} = 1 - \exp(-IT_{Si,i} \times SF_i) \quad (10)$$

7.3 Residential scenario

For the residential scenario, it is assumed that the groundwater intake is the primary concern.

According to Wagner [4], RCRA section 3004(d) requires that petitioner for a RCRA site demonstrate, to a reasonable degree of certainty, that there will be no migration of hazardous constituents from the disposal unit or injection zone for as long as the waste remain hazardous. It seems to the authors that this requirement might not be met, given the current RCRA regulation imposed by EPA. First, carcinogenic metals are persistent, their toxicity might remain forever. Secondly, many compounds of these metals are soluble, thereby it is conceivable that the metal compound may be transported down into the aquifer by infiltration if some conditions are met (e.g., pH becomes small), and they might be transported further down gradient in the aquifer. Table 8 lists some metal compounds with high solubility [13].

In a report to the Congress [14], EPA assessed the potential health effects of the waste disposal sites of the U.S. coal power plants. These power plants generate fly ash, bottom ash, boiler slag, and flue gas desulfurization wastes which contains the carcinogenic metals we have discussed. According to the report, "while most of the laboratory results indicated that coal combustion wastes do not possess RCRA hazardous characteristics, in some instances, data on actual field observations indicates that migration of potentially hazardous constituents from utility wastes disposal sites has occurred. For example, observed concentrations of contaminants found in groundwater down-gradient from the sites exceed the primary drinking water standards about 5% of the time".

Based on the above discussion, groundwater contamination down-gradient of our hypothesized site might be possible. However, due to large uncertainties about models and available data, a quantitative assessment of the residential scenario has not been performed in this analysis.

TABLE 8

Some soluble metal compounds [13]

Name	Formula	Solubility (g/dL)
<i>Ortho</i> -arsenic	$H_3AsO_4 \cdot \frac{1}{2}H_2O$	302
Arsenic tri-iodide	AsI_3	6.0
Arsenic penta-oxide	As_2O_5	150
Arsenic trioxide	As_2O_3	3.7
Chromate	$(NH_4)_2CrO_4$	40.5
Dichromate	$(NH_4)_2Cr_2O_7$	30.8
Chromium chloride, hexahydrate	$[Cr(H_2O)_4Cl_2]Cl \cdot 2H_2O$	58.5
Chromium sulfate	$Cr_2(SO_4)_3 \cdot 18H_2O$	120
Nickel bromide, trihydrate	$NiBr_2 \cdot 3H_2O$	199
Nickel perchlorate	$Ni(ClO_4)_2 \cdot 6H_2O$	222.5
Nickel chloride, hexahydrate	$NiCl_2 \cdot 6H_2O$	254.0
Nickel nitrate, hexahydrate	$Ni(NO_3)_2 \cdot 6H_2O$	238.5
Cadmium borotungstate	$Cd_5(BW_{12}O_{40})_2 \cdot 18H_2O$	1250
Cadmium chlorate	$Cd(ClO_3)_2 \cdot 2H_2O$	298
Cadmium nitrate	$Cd(NO_3)_2$	109
Cadmium sulfate hydrate	$3CdSO_4 \cdot 8H_2O$	113
Beryllium oxalate	$BeC_2O_4 \cdot 3H_2O$	38.22
Beryllium oxide	BeO	0.00002
Beryllium selenate	$BeSeO_4 \cdot 4H_2O$	56.7
Beryllium sulfate, hydrate	$BeSO_4 \cdot H_2O$	42.4

8. Results for human intrusion scenario

This section presents the calculational results of lifetime excess cancer risk for the human intrusion scenario. In order to see the influence of carcinogenicity data, we show results for two cases herein, one is denoted as the "base case", the other is denoted as the "revised base case". These two cases differ in the use of oral slope factors, as shown in Table 9. The oral slope factors in the base case are the same as listed in Table 4. In the revised base case, we add the arsenic oral slope factor of $1.8 \text{ (mg/kg-day)}^{-1}$ due to skin cancer [8] to the base case arsenic oral slope factor of $18 \text{ (mg/kg-day)}^{-1}$, which accounts for cancer in liver, lung, kidney, and bladder, and get an arsenic oral slope factor of $19.8 \text{ (mg/kg-day)}^{-1}$ for all cancer sites. Also in the revised case, the oral slope factor of chromium is assumed one tenth of its inhalation slope factor. And the oral slope factor of cadmium is assumed one third of its inhalation slope factor. These ratios are taken from a study by Cohen [11]. EPA itself professes uncertainty about the ingestion carcinogenicity path for chromium. For cadmium, EPA's non-carcinogenic conclusion via the ingestion is not based on strong, conclusive data; hence, we use Cohen's estimate for illustrative purposes.

TABLE 9

Slope factors for ingestion pathway

Metal	Oral slope factor for the base case (mg/kg-day) ⁻¹	Oral slope factor for the revised base case (mg/kg-day) ⁻¹
As	18	19.8 ^a
Cr (VI)	—	4.2 ^b
Ni (subulfide)	—	—
Cd	—	2.1 ^c
Be	4.3	4.3

^aSummation of the base case slope factor (18) and the slope factor for skin cancer (1.8), the latter is calculated from water unit risk of $5 \times 10^{-5} (\mu\text{g/L})^{-1}$, which is taken from IRIS [8], an ingestion rate of 2 L/day is used in the calculation.

^bValue taken as one tenth of the slope factor for inhalation, this ratio is taken from Cohen [11].

^cValue taken as one third of the slope factor for inhalation, this ratio is taken from Cohen [11].

TABLE 10

Excess cancer risk due to fruit and vegetable intake — vegetative portion

Metal	Soil concentration (mg/kg)	C_{pv} (mg/kg)	Intake (mg/kg day)	Risk
As	1,000	40	5.43×10^{-3}	9.31×10^{-2}
Be	100	1	1.36×10^{-4}	5.85×10^{-4}
Cd	100	55	7.46×10^{-3}	—
Cr	100	0.75	1.02×10^{-4}	—
Ni	100	6	8.14×10^{-4}	—

8.1 Base case results

Table 10 and Table 11 represent the risk calculation for fruit and vegetable intake. Table 10 shows the risk due to the vegetative portion intake and Table 11 shows the risk due to the reproductive portion intake. Total risk for the vegetative portion is 9.37×10^{-2} and that for the reproductive portion is 8.72×10^{-2} . Total risk due to fruit and vegetable intake is 0.18.

Table 12 represents the result for dermal contact with metals in soil. The total risk for this pathway is 0.12.

Table 13 represents the risk due to ingestion of metals in soil. The total risk of this pathway is 2.6×10^{-3} .

Table 14 lists the summary results for the base case. The total risk of the human intrusion scenario is 0.30. It is clear that the risk for the human

TABLE 11

Excess cancer risk due to fruit and vegetable intake—reproductive portion

Metal	Soil concentration (mg/kg)	C_{pr} (mg/kg)	Intake (mg/kg day)	Risk
As	1,000	6.0	5.04×10^{-3}	8.67×10^{-2}
Be	100	0.15	1.26×10^{-4}	5.42×10^{-4}
Cd	100	15	1.26×10^{-2}	—
Cr	100	0.45	3.78×10^{-4}	—
Ni	100	6.0	5.04×10^{-3}	—

TABLE 12

Excess cancer risk due to dermal contact with contaminants in soil

Metal	Soil concentration (mg/kg)	Absorbed dose (mg/kg day)	Risk
As	1,000	6.75×10^{-3}	0.114
Be	100	6.75×10^{-4}	2.90×10^{-3}
Cd	100	6.75×10^{-4}	—
Cr	100	6.75×10^{-4}	—
Ni	100	6.75×10^{-4}	—

TABLE 13

Excess cancer risk due to ingestion of metals in soil

Metal	Soil concentration (mg/kg)	Intake (mg/kg day)	Risk
As	1,000	1.43×10^{-4}	2.57×10^{-3}
Be	100	1.43×10^{-5}	6.15×10^{-5}
Cd	100	1.43×10^{-5}	—
Cr	100	1.43×10^{-5}	—
Ni	100	1.43×10^{-5}	—

intrusion scenario is large, and might be significantly enhanced by drinking contaminated groundwater and inhalation of the dust containing metals.

8.2 Results for the revised base case

Results for the revised base case are summarized in Table 15. The total risk of the revised case is 0.38. It can be observed that arsenic dominates the calculated risk in this study.

TABLE 14

Risk summary for base case

Metal	Fruit and vegetable ingestion (vegetative)	Fruit and vegetable ingestion (reproductive)	Dermal contact with soil	Ingestion of metals in the soil	Total risk for each metal
As	9.31×10^{-2}	8.67×10^{-2}	0.114	2.57×10^{-3}	0.296
Cr	—	—	—	—	—
Ni	—	—	—	—	—
Cd	—	—	—	—	—
Be	5.85×10^{-4}	5.42×10^{-4}	2.90×10^{-3}	6.15×10^{-5}	4.09×10^{-3}
Total risk	9.37×10^{-2}	8.72×10^{-2}	0.117	2.63×10^{-3}	0.30

TABLE 15

Risk summary for revised base case

Metal	Fruit and vegetable ingestion (vegetative)	Fruit and vegetable ingestion (reproductive)	Dermal contact with soil	Ingestion of metals in the soil	Total risk for each metal
As	0.102	9.50×10^{-2}	0.125	2.83×10^{-3}	0.325
Cr	4.28×10^{-4}	1.59×10^{-3}	2.83×10^{-3}	6.01×10^{-5}	4.91×10^{-3}
Ni	—	—	—	—	—
Cd	1.55×10^{-2}	2.61×10^{-2}	1.42×10^{-3}	3.00×10^{-5}	4.31×10^{-2}
Be	5.85×10^{-4}	5.42×10^{-4}	2.90×10^{-3}	6.15×10^{-5}	4.09×10^{-3}
Total risk	0.119	0.123	0.132	2.98×10^{-3}	0.38

9. Discussion

While there is uncertainty in modeling the various pathways, this study indicates that the future risk for the human intrusion scenario for a hypothesized RCRA site is potentially intolerable, if one adopts the general rationale used in 40 CFR Part 191 for high level radioactive waste disposal [1]. The risk of the residential scenario has not been obtained quantitatively due to the large uncertainty, but potentially significant risk for this scenario can not be excluded. Although the risk calculated is one thousand years away in the future, as in 40 CFR 191, it is not assumed that the future generations would be more advanced in technology. Furthermore, societal memory regarding the site is assumed to be lost. This degree of conservatism is along the same lines as that chosen by U.S. EPA for high level radioactive waste disposal in a geological repository [1] and by the U.S. NRC for disposal of low level radioactive wastes [2].

In Superfund cleanup programs, consideration of risk into the far future is also lacking. Doty and Travis [15, 16] reviewed 50 EPA Records of Decisions made in Fiscal Year 1987. Among them, less than half (22) of the Records of Decisions documented quantitative future risk assessment. It is obvious that risk a thousand years away did not play a role in the superfund decision making process.

In summary, the U.S. EPA requirements concerning long-term risk from RCRA sites containing metal carcinogens which never change due to radioactive decay stand in sharp contrast to the stringent requirements over 10,000 years posed by EPA for high level radioactive waste disposal in 40 CFR 191 (remanded), and the long-term requirements posed by U.S. Nuclear Regulatory Commission for low level radioactive waste disposal sites.

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