SAND--91-0587C

DE91 017314

USE OF RISK TO RESOLVE CONFLICTS IN ASSESSING HAZARDS AT MIXED-WASTE SITES*

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* This work was performed at Sandia National Laboratories, which is supported by the U.S. Department of Energy under contract number DE-AC04-76D0078.

DISCLAIMER

Abstract

Two main issues contribute to the assessment of health hazards from mixed waste: the scientific methods to assess these materials and the legislative and regulatory control of these materials. This paper is primarily concerned with the scientific method of assessing hazards from mixed waste (i.e., carcinogenic chemicals, noncarcinogenic chemicals, and radioactive material). This paper discusses SRS, a Site Ranking System, and its use of risk concepts to avoid introducing new inconsistencies when ranking mixed-waste sites. SRS ranks each site by scoring factors that influence the human health risk. The factors are (1) the potentially exposed population, (2) the average amount of exposure to the waste, and (3) the toxicity of the waste. The relative risk of a release is measured as the product of these three factors. The third factor, toxicity, is indexed with a single score, but because methods of measuring toxicity differ for carcinogenic chemicals, noncarcinogenic chemicals, and radionuclides, comparison can be difficult; hence, this paper also summarizes the logic and assumptions used to make toxicity comparisons in SRS. As may be expected, results from a ranking scheme based on risk are different from results generated by the original Hazard Ranking System (HRS), used by the Environmental Protection Agency. This paper briefly discusses these differences for five Superfund sites (no mixed waste). The legislative and regulatory control of these materials to protect human health is also discussed.

Introduction

When seeking solutions for disposal of mixed waste, inconsistencies are immediately apparent in the way scientists choose to describe the hazards from radioactive, carcinogenic, and noncarcinogenic chemical waste. For example, hazards are described as dose factors, unit cancer risks, and allowable daily intake. This paper describes how risk-based principles were used not only to identify inconsistencies in descriptions of mixed-waste hazards, but also to avoid introducing new inconsistencies when developing a site ranking system for hazardous chemical and radioactive wastes. These principles were exercised in the development of a Site Ranking System, SRS [1]. Specifically, the measurement of toxicity is discussed.

SRS was developed as a simple, low-cost approach to rank sites. Its risk-based concept differs from the original ranking system used by the Environmental Protection Agency (EPA) and so the results also differ. A comparison of results is provided.

Finally, inconsistencies are also apparent in the way the laws treat radioactive and hazardous chemical waste. For example, in some regulations, Congress assumes that land disposal is the best available technology for disposal of high-level waste [2,3]. In other regulations, land disposal is the least desirable way to manage hazardous chemical waste [4,5]. A discussion of the resulting inconsistency when the waste contains both radioactive and hazardous chemicals is provided.

Risk Approach for Avoiding Inconsistencies in Ranking Schemes for Mixed-Waste Sites

The use of a risk approach in developing ranking schemes for mixed-waste sites helps to avoid introducing new inconsistencies when comparing mixed wastes. A specific example of a risk approach is SRS, a Site Ranking System, which was developed in 1986 [1]. SRS is a relatively simple method that uses readily available pertinent site data to provide an order-of-magnitude ranking of sites containing both carcinogenic and noncarcinogenic hazardous chemicals and radioactive wastes. The purpose of SRS and the personal computer (PC) version, SRS88 [6], is to rank hazardous waste sites according to the relative human health risk they pose (here risk refers to risk of deaths over a lifetime). SRS was developed to help decision-makers allocate investigative funds for sites with the greatest potential health risk.

Many mixed-waste sites in the United States need remedial action to protect public health, but the hazards differ by orders of magnitude among these sites. Although a manager can use a comprehensive risk assessment or extensive field monitoring to select sites posing the greatest health risk, these approaches are quite costly in time spent both to gather data and prepare the assessment. SRS provides an order-of-magnitude ranking of sites similar to that obtained with a detailed risk assessment.

SRS ranks each site by scoring factors that influence this risk. The factors are (1) the potentially exposed population, (2) the average amount of exposure to the waste, and (3) the toxicity of the waste. The relative risk of a release is measured as the product of these three factors. Later in this paper, the logic and assumptions used to index the third factor for toxicity comparisons are summarized.

Development Goals of SRS

Several goals guided the development of SRS. SRS was to (1) approximate the ranking obtained with a comprehensive risk assessment, (2) have a logic that is easy to understand and use, (3) require only readily available but relevant data with an order-of-magnitude effect on the health risk, consistent with the uncertainty of the preliminary data, (4) apply to most waste sites, and (5) score sites using hand calculations as an option.

Using SRS in Conjunction with Other Waste Site Decisions

A distinct advantage of basing a ranking approach for mixed-waste sites on a maturing probabilistic risk assessment (PRA) methodology is that the role and limitations are clearly defined [7]. The role of SRS ranking in a general program to reduce hazards at mixed-waste sites is as follows:

Check for Emergency Action. A site evaluation begins with the assessment of the acute hazards. If acute hazards are present and no barriers protect the public, immediate action must be taken. Because the assessment for acute hazards is made before screening with SRS, SRS does not account for acute risks.

Gather Site Data. The time-consuming portion of the SRS (or any ranking scheme) is collecting the pertinent data on waste properties, site features, and facility design. Of the data collected for SRS, the identity, quantity, and toxicity of waste placed at a site are the most influential; they determine the maximum potential health risk per exposed individual. Unfortunately, for many abandoned and existing sites, this information is difficult to find. SRS can be used when only the general types of wastes are known, but estimating waste properties is more difficult.

Screen with SRS. The third major step of a site evaluation involves scoring with SRS. Ranking the sites of concern is a part of the overall assessment and management of risk; thus, the basis of a ranking scheme must be similar to a risk assessment methodology to ensure a consistent ranking.

Because SRS is risk based, information from the ranking can guide further data collection efforts; hence, some interaction may occur between screening and collection of readily available relevant data.

Ranking. Factors such as ecologic hazards, costs of cleanup, technical feasibility, public concerns, and risk-management guidelines of a regulatory agency are not considered in SRS. Consequently, the ranked list is not a final action list; it provides only technical input about human health risk. The final action list is the result of risk management decisions based on the technical input of SRS and these other factors. The criteria used to make a final action list requires value judgments, which implies a political process [8]. The political decisions (value judgments) may very well overshadow any risk assessment results.

Many ranking schemes combine aspects of both risk assessment and risk management. In some instances, the risk-management decision criteria are rigid enough to incorporate into a ranking scheme, but, in general, this practice is not recommended [7]. Combining risk assessment and risk management frequently confuses industry and the public about decisions based on the ranking scheme, reduces the portability of the method among governmental agencies with different decision criteria, and reduces the flexibility of the risk manager to manage.

Risk Basis of SRS

Components of Risk. Several measures of risk are possible: risk to human health ($\hat{R}_h$), risk to other ecosystems ($\hat{R}_e$), and risk to economic institutions ($\hat{R}_\$$). In principle, all could be summed to obtain total risk, $\hat{R}$:

$$\hat{R} = \hat{R}_h + \gamma_1 \hat{R}_e + \gamma_2 \hat{R}_\$ \qquad (1)$$

where γ_n are subjective weights assigned to the ecosystem and economic risk components, based on value judgments.

Although other risk components could be added, currently SRS scores only $\hat{R}_h$; other risks are described but not scored. These latter risks can be brought into the decision process when the risk manager is selecting the priority sites.

The health risk ($\hat{R}_h$) can be subdivided into chronic risks, which occur over long periods of time, and acute risks, which occur suddenly:

$$\hat{R}_h = \hat{R}_{chronic} + \gamma \hat{R}_{acute} \tag{2}$$

where γ is a subjective weight assigned to acute risk according to the public's disposition to risk aversion.

SRS considers only the chronic health risk. Acute risks from fire, explosion, or direct contact require rapid response (e.g., separating certain chemicals or building a security fence around the site) as compared to long-term remedial response (e.g., stabilizing lagoon wastes). Consequently, of the four subsets of hazardous waste described by RCRA (toxicity, ignitability, corrosivity, or reactivity) [5], SRS evaluates only toxicity, a large subset of hazardous wastes, and of the toxicity category, only long-term potential risks (chronic). Hazards from the other three categories are acute risks and noted in an accompanying narrative of the site but not scored because of the usual need to respond immediately to these threats. Emergency response is begun before SRS screening.

Health risk ($\hat{R}_h$) is the product of the probability (P) of a mutually exclusive release scenario and the consequence (ψ) of that scenario. Here a scenario is a sequence of events that could lead to release and transport of hazardous chemicals from a waste site to some target (receptor) that could be endangered. Usually a large number of scenarios can be developed; $\hat{R}_h$ is the sum of these individual risks:

$$\hat{R}_h = \Sigma\, P_i \psi_i; \quad i=1, \text{ number of scenarios} \tag{3}$$

One Scenario with Three Pathways. In SRS, only one scenario is considered and has a probability of one; therefore, the ranking is based solely on the consequence. The scenario is composed of three generic migration pathways, identified by the manner in which hazardous waste leaves the site—either through an air ψ_a, surface-water ψ_{sw}, or groundwater ψ_{gw} route (release is not necessarily simultaneous or equal along all pathways):

$$\hat{R}_h = \sum_k \psi_k; \quad k = \text{pathway} \tag{4}$$

(An alternative is to estimate probabilities of release [or some index] based on qualitative metrics such as observed, very high, high, medium, or low.)

Although contaminant movement in the environment is complex (Eq. 4), SRS simplifies this movement to three pathways with minimal interaction. Furthermore, the contaminant is assumed to be directly ingested or inhaled. Intricate exposure pathways through food web or recycling of contaminants among surface water, groundwater, surface soil, and air are assumed to be minor components of the ultimate risk. (In some instances this may not be true, but data to evaluate these risks are normally not available during the screening phase.) Finally, the chronic health hazard of direct contact with the waste is evaluated with the air pathway.

Three Factors for Each Pathway. SRS assumes that a pathway consequence (e.g., ψ_{gw}) is influenced by three factors: the size of the population at risk (N), the exposure (E) of that population to the hazardous material, and the toxicity of the hazardous material (T). Further, SRS assumes the following: exposure is expressed as a daily rate (e.g., mg/kg/day); and toxicity is expressed as an age-adjusted annual risk per rate of exposure assuming an average 70-yr life (e.g., [1/70]•[lifetime risk per mg/kg/day]).

Given these assumptions, the consequence (i.e., annual number of chronic illnesses or cancer deaths) is

$$\psi \propto N \cdot E \cdot T \tag{5}$$

Furthermore, ψ of any pathway is the sum of the consequences from each hazardous chemical or radionuclide of the inventory at the waste site; hence the risk is proportional to (substituting Eq. 5 into Eq. 4).

$$\psi \propto \sum_k \sum_m N_{km} \cdot E_{km} \cdot T_{km} \; ; \quad k = \text{pathway}, \quad m = \text{chemical} \tag{6}$$

As shown in Eq. 6, if the toxicity hazard is for a critical subset of the population (such as children), then only children should be included in the population at risk (meaning that different population counts may be needed for different hazardous chemical and radionuclides). Although this complication can be included, SRS currently avoids it by assuming that the toxicity hazard is for an average population. The assumption is conservative provided the most restrictive toxicity is used (e.g. toxicity to children even though much of the average population may consist of adults).

$$\psi \propto \sum_k N_k \sum_m E_{km} \cdot T_{km} ; \quad k = \text{pathway}, \; m = \text{chemical} \tag{7}$$

Of these three factors, two (population and toxicity) are easily characterized by single "scores" in a ranking system. Only exposure E must be subdivided further into component factors.

Exposure Component. The specific component factors of E vary with each pathway. Generally, the E is evaluated from the quantity of waste initially deposited at the site W_0 and factors indexing the effectiveness of the engineered barrier, f_{eng}, and site features, f_{site}, in reducing the amount of hazardous waste reaching the potentially exposed population:

$$E \propto W_o \cdot f_{eng} \cdot f_{site} \tag{8}$$

The mathematical rigor used in evaluating the reduction factors, f_{eng} and f_{site}, determines the computational burden of SRS. A major effort was expended to reduce analytic or empirical expressions for f_{site} (using a combination of mathematical and heuristic arguments) to easily calculated terms involving readily available data. To increase the understanding of the system, the terms were also made as similar as possible in all three generic pathways. The f_{eng} reduction factor was evaluated qualitatively because the emphasis on using SRS was on sites where few engineered barriers existed (e.g. CERCLA sites [Comprehensive Environmental Response, Compensation, and Liability Act]) [9]. (The evaluation could change if SRS were used to evaluate sites with engineered barriers [e.g. new RCRA sites with engineered barriers].)

Substituting Eq. 8 into Eq. 7 yields (assuming that the toxicity T is path independent and the amount of waste W_o does not vary with the pathway)

$$\hat{R}_h \propto \sum_k N_k \sum_m T_m W_o \quad f_{eng,mk} f_{site,mk}; \quad m - \text{chemical}, \; k - \text{pathway} \tag{9}$$

Risk Argument for Comparing Toxicity Values

Toxicity Metrics

The human health effect of a unit dose exposure to a chemical is evaluated from the chemical toxicity [T]. Three bases of chronic toxicity are used:

- Acceptable daily intake (ADI) or reference dose (RfD) for noncarcinogens

- Unit carcinogenic risk (UCR), carcinogenic potency factor (CPF), or slope factors for carcinogens
- Effective dose equivalents for radionuclides.

ADI (Chemical Noncarcinogens). The acceptable daily intake (ADI) as defined in this paper means the highest human intake of a chemical, expressed as mass of chemical per individual body mass per day (mg/kg/day) that does not cause adverse effects when exposed for a lifetime. Although many early ADIs were originally developed for assessing acute toxicity, the practical application of ADIs over the past 25 years has been in setting food and drinking water standards (chronic toxicity) [e.g., 10]. Health Effects Assessment (HEA) documents by the EPA Office of Research and Development are recognized sources for ADIs on hazardous materials. Findings are summarized in data bases such as PHRED (Public Health Risk Evaluation Data Base) [e.g., 11].

Toxicologists evaluate the ADI for a chemical by applying a safety factor to an experimentally estimated threshold dose, referred to as NOAEL (no-observed-adverse-effect level). Toxicologists consider noncarcinogenic chemicals as having a threshold dose (NOAEL) below which a toxic effect will not occur (for some chemicals, a beneficial effect occurs below the NOAEL, but when making hazard comparisons with UCRs, we are concerned strictly with adverse effects). The safety factors vary: a 10-fold safety factor for long-term human experiments; a 100-fold safety factor for well-conducted, chronic, animal experiments; and a 1,000-fold safety factor for subchronic animal or lower quality experiments [12,13].

UCR (Chemical Carcinogens). In this paper, the unit carcinogenic risk (UCR), the cancer potency factor (CPF), or slope factor means the upper bound on lifetime risk of cancer per unit of dose, expressed as risk per mass of substance per individual body mass per day $(mg/kg/day)^{-1}$ [14]. For carcinogens, toxicologists generally assume there is no threshold dose below which the substance will not cause cancer: any exposure level poses some risk of cancer even though there may be beneficial effects at low doses as well. To estimate the cancer risk, toxicologists extrapolate from studies on animals at high doses to humans at low doses using a mathematical model, frequently linear [15].

Dose Equivalents (Radionuclides). Cancer risk from radionuclides is usually reported by dose factors (rem/μCi/day) or (Sv/Bq/s) [e.g., 16,17,15].

Comparing UCR and ADI

As explained when defining ADI and UCR, toxicologists use different assumptions to evaluate measures of toxicity (i.e., carcinogenic and radionuclide, no threshold dose; noncarcinogenic, threshold dose). Often risk assessments will evaluate the carcinogenic and noncarcinogenic effects separately [e.g., 18], leaving comparisons strictly as a value judgment. However, both measures of toxicity attempt to evaluate chemical hazards (above NOAEL for noncarcinogens), and thus a toxicity comparison is certainly a proper subject of scientific opinion as well, guided perhaps by expert elicitation [19]. We argue that aligning the scales for each toxicity metric requires finding the point representing an acceptable *risk* for each scale.

Conversion of Dose Factors to UCRs. The first step in the comparison of the toxicity metrics is the conversion of dose factors for radionuclides to UCRs for chemical carcinogens. Currently, health physicists use committed effective dose equivalents over 50 yr [$H_{E,50}$] as described in ICRP Publication 26 and ICRP Publication 30 [20,17]. Maximum values for committed effective dose equivalents [$H_{E,50}$] were obtained from U.S. Department of Energy (DOE) tables [16]. The updated UCR values were calculated as

$$UCR = a \cdot C_R \cdot H_{E,50} \quad (10)$$

where

UCR = unit cancer risk for radionuclide (μCi/kg/day)

a = factor for converting committed effective dose equivalent [$H_{E,50}$] rem to rem/kg body mass/day (70 kg • $2.56x10^4$ day)

C_R = total risk per unit uniform whole-body irradiation ($1.65x10^{-4}$ rem^{-1})

$H_{E,50}$ = maximum committed effective dose equivalent for radionuclide for various compounds [16]

$= \Sigma_b \omega_b H_{50,b}$

$H_{50,b}$ = predicted dose equivalent to an tissue or organ b for radionuclide over 50 yr after an initial intake of radionuclide into the body (contributions from external dose not included) (rem/μCi)

ω = fractional contribution of organ b to the total risk under uniform whole-body irradiation [20]

The essential assumption in Eq. 10 is that the dose factor from one initial dose where effects are evaluated over 50 yr [$H_{E,50}$] (rem/μCi) is greater than or equal to a dose factor for uniform annual doses multiplied by 50 yr (rem/μCi/yr • 50 yr). The two dose factors (initial and 50 x annual) are about equal for short-lived radionuclides that are rapidly cleared from the body. The $H_{E,50}$ is probably greater than the dose factor for uniform annual doses for long-lived radionuclides that are slowly cleared from the body.

Carcinogenic risk factor for nonradionuclides are reported in cancer risk per mg/kg body mass/day rather than cancer risk per intake (mg) over an average person's lifetime; hence, to express radionuclide UCR in corresponding units, the conversion factor [a] multiplies $H_{E,50}$ by the average body weight (70 kg) and average life span (2.56×10^4 day) to obtain the cancer risk per intake per average body mass per average lifetime $(\mu Ci/kg/day)^{-1}$.

Acceptable Cancer Risk. SRS assumes a 10^{-5} lifetime risk of cancer is acceptable based on the following arguments. Two sources for acceptable cancer risk are considered: radiation protection standards and guidelines for carcinogenic chemicals. The negligible individual lifetime risk recommended by the National Council on Radiation Protection and Measurements is 10^{-5} risk of cancer health effects/individual [21], the approximate risk of background radiation.

The government in cooperation with international agencies has expended considerable effort to control radiation exposures of the public. Because laws and regulations ideally reflect society's value judgments, conceivably they could serve as a reasonable basis for evaluating society's judgment on acceptable cancer risk, also. Unfortunately, current U.S. regulations (even for regulations on one type of hazard such as radiation exposure) are quite inconsistent [22]. Similar to regulatory differences among waste types, the inconsistencies occur in part because they are not based on risk evaluations but rather on how familiar the source is, how well the hazard can be measured, which agency is setting the standard, and what can be reasonably achieved, technologically, at a particular time (i.e., when the regulations were drafted).

Although the radiation regulations are not entirely consistent, a 10^{-5} chance of contracting cancer approximates the middle ground for involuntary aggregate risks permitted in regulations. For example, 10 CFR 20 (radiation protection standard) limits involuntary, whole-body radiation exposure to the public to 0.5 rem/yr, and 10 CFR 61 and 40 CFR 190 (low-level nuclear waste and nuclear power regulations) to 0.025 rem/yr [23,24,25]. The latent cancer risk over a lifetime per rem of exposure for whole-body exposure is about 1.65×10^{-4} health effects risk/rem • 70 yr [20,26], thus the regulations correspond to lifetime risks of 6×10^{-3} and 3×10^{-4}, respectively, for involuntary exposure [see also 22]. At the other extreme, the restrictive disposal standard for high level radioactive waste, 40 CFR 191, limits the risk to about 0.1 total deaths/yr [27]. Averaged over the current U.S. population, this represents a lifetime risk of death of 3×10^{-8}. (An average does not completely describe the distribution of risks; thus an extreme such as the maximally exposed individual [MEI] may also be used. However, the MEI depends not only upon site characteristics, but also upon speculations about future human behavior and population distribution. International agencies [e.g., Nuclear Energy Agency] set limits around 10^{-6}.)

Although not entirely consistent, in general, the Food and Drug Administration (FDA) and EPA regulations consider lifetime cancer risks from *individual* chemicals of less than 10^{-6} as insignificant and risks between 10^{-6} and 10^{-4} as possibly significant but acceptable if reducing them is either technically infeasible or too costly (e.g., polychlorinated biphenyls [PCBs] and dioxin) [12]. Consistent with this generality, EPA policy permits the *aggregate* cancer risk after cleanup at Superfund sites to range anywhere from 10^{-4} to 10^{-7} [11]. Thus, 10^{-5} represents a "very small" aggregate cancer risk from a waste site [28].

Acceptable Noncancer Risk. SRS assumed that the health risk from a lifetime ADI exposure is also 10^{-5}. An ADI for a noncarcinogen means that the allowable dose poses a very small risk of adverse effects. Unfortunately, we have no means of quantifying this "very small" risk. If one assumes that noncarcinogenic materials have the no-observed-adverse-effect level (NOAEL), the risk is very small indeed.

However, we present the following two arguments to support our assumption of 10^{-5}. First, we simply argue that regulators have established 10^{-5} as an acceptable risk and ADI levels are defined as acceptable. Second, we give the following heuristic argument, although it is admittedly more conservative than the threshold concept. If, as an example, an animal study includes 1,000 animals in the test group and at a certain extrapolated dose, one animal is adversely affected in comparison to a control group, then there is a 0.1% risk near the NOAEL. If a 100-fold safety factor is applied to the NOAEL, the risk is 10^{-5} (assuming a 1-to-1 dose response curve).

Aligning scales. If the assumptions concerning UCR and ADI health risks are accepted, the UCR and ADI scales can be aligned. For example, if a cancer risk of 10^{-5} is acceptable, then a mild carcinogen with a UCR of 10^{-5} per mg/kg body weight/day has an equivalent ADI of 1 mg/kg/day; $UCR \cdot ADI = 10^{-5}$ cancer risk. A more conservative approach would be to add a factor of safety of 10 or define 10^{-6} simply as negligible risk. However, the important point to note is that risk concepts help clarify this comparison and that alignment of the toxicity scales should probably occur between 10^{-6} and 10^{-5} ($UCR \cdot ADI = 10^{-5}$ or 10^{-6}).

Comparison of SRS with other Ranking Schemes

Qualitative Comparison

Currently, the most widely used ranking scheme is the Hazard Ranking System (HRS) used by the EPA for placing sites on the National Priorities List (NPL) [9]. HRS as first promulgated (1982) had drawbacks in areas such as scoring the population, ignoring important characteristics of the pathway, differing in its approach from general procedures of risk assessment, and scoring waste properties [1,29]. SRS was first proposed as a way to more clearly (1) identify deficiencies in HRS, (2) identify differences with general risk assessment methods, and (3) possibly serve as a starting point for developing an alternative to HRS for the DOE.

In recognition of the deficiencies of HRS, the Superfund Amendments and Reauthorization Act required that EPA modify HRS for scoring future (i.e., not already scored) sites to more "accurately assess the relative degree of risk to human health and the environment posed by these sites" [30]. After receiving extensive comments during various phases of modifications and incorporating several concepts from SRS, EPA promulgated the revised HRS (rHRS) [31]. In general, there are revisions to every factor. Also, the degree of complexity and amount of data needed are much greater in rHRS than in HRS [32].

Although differences in methods of scoring site characteristics remain between SRS and rHRS, because of the fundamental difference in developing the scoring schemes, many of the same site characteristics are now indexed in both systems. For example, chemical toxicity in rHRS now depends on chronic toxicity rather than indirectly on acute toxicity through SAX scores [32].

The remaining differences between SRS and rHRS include the following: (1) rHRS scores risks to other ecosystems ($\hat{R}_e$), (2) SRS combines both chemical and radioactive hazardous waste scores, (3) SRS evaluates the health hazard from the quantity of each constituent that comprises the waste, and (4) SRS separates risk assessment from risk management, whereas rHRS remains a management screening tool, with both risk assessment and risk management attributes.

Comparison of SRS with HRS Results

Unfortunately, data for mixed-waste sites were not readily available; hence SRS scores were compared to HRS and estimated rHRS scores for five Superfund sites from a list of 40 that the EPA used to benchmark ranking schemes during preliminary development of rHRS. Because we did not have the original HRS scores, the HRS scores were reported from the published NPL (40 CFR 300); the rHRS scores are preliminary estimates by hand without benefit of a software package. The purpose of the comparison is to glimpse how a ranking scheme based solely on risk concepts differs from HRS and rHRS, schemes that include a mix of risk management and risk assessment concepts and were developed as a political process and thus include value judgment.

As may be expected, the results of SRS, a ranking scheme base on risk, differ from HRS and rHRS results (Table 1). However, the ranking changes themselves were minor. For example, in HRS, Tyson's Dump would be ranked first, and Bunker Hill, second; in SRS, Bunker Hill Mining is ranked first, and Tyson's Dump, second. However, the SRS scores reflect the $\log_{10}$ of the potential consequences; thus SRS perceived dramatic differences in the hazards at the site. For example, Bunker Hill Mining was more than two orders or magnitude greater in relative potential risk than Tyson's Dump.

Certainly, the intended use as reflected by the development process of HRS and rHRS ultimately explain these differences but because this paper intends to touch on several uses of the risk concept, it does not explore in depth the reasons for the differences (e.g., whether differences were from value judgments, risk management decisions, or errors in risk evaluations). What follows is a brief description and history of these sites up to 1980, followed by a description of the SRS scoring. The sites are discussed in the order that SRS ranked them.

Bunker Hill Mining. The Bunker Hill Mining and Metallurgy Site in Idaho is one of the largest and most complex Superfund sites. The assumed area of the site was 80 acres, although some sources report up to 13,400 acres. In 1980, it consisted of four communities with a population of about 7,600, but since the mine and smelter shutdown in 1981, the population has dropped. The site has surface impoundments, waste piles, and contaminated soils. The contamination consists of heavy metals from mining and lead and zinc smelter emissions over a 70-yr period. Surrounding hills are devoid of vegetation because of overharvesting and forest fires, and new growth may be retarded by heavy metals.

The carcinogen, arsenic, is the primary heavy metal out of six that dominate the SRS score for this site along both the groundwater and surface pathways.

Tyson's Dump. Tyson's Dump site in Pennsylvania is an abandoned facility, on about 4 acres, located within an old sandstone quarry that operated between 1962 and 1970. Highly fractured bedrock allowed contact with aquifers and Schuylkill River. In 1973, the state ordered the site closed. The lagoons were drained and backfilled, and the site vegetated. Numerous hazardous chemicals have been identified in the subsurface of the site area. Groundwater in the Schuylkill River floodplain is contaminated with heavy metals.

The results show a dramatic change from the original ranking provided by HRS for Tyson's Dump. HRS recorded observed releases along all three pathways, which greatly influenced the ultimate score, especially because of the large population around the area. Out of 18 chemicals scored, carcinogens such as arsenic and several organics (e.g., 1,3-dichlorupropene and tetrachloroethylene) are the chemicals that dominate the score for SRS. The primary pathway was seepage from saturated soils (which were used to fill in past lagoons) through sandy loam soil to the Schuylkill River and then to city water intakes. In

SRS, a high but not excessively high leach rate was used. An excessively high leach rate, possibly justified by the highly fractured bedrock, would increase the score.

Hudson River PCBs. The Hudson River PCBs site consists of five dump sites in and along the river at Ft. Edward, NY, plus 40 "hotspots" (sandbars) down to Troy, NY. Liquid waste PCBs from two General Electric capacitory manufacturing plants were dumped directly into the Hudson River. For SRS88, the surface water pathway dominates the scoring for this site.

Whitehouse Waste Oil Pits. The Whitehouse Waste Oil Pits site, consisting of seven oil pits on about 5 acres near Whitehouse, FL, was established in 1958 to dispose of waste oil and acid sludges generated by Allied Petroleum Company, a waste oil recycler. The site was closed and some remedial action began in 1968. In 1980, the sludges were stabilized and the pits covered.

Out of six chemicals, arsenic and lead are the primary hazardous chemicals dominating the SRS score. The score is highly dependent on the aquifer source from nearby private wells. If the deep Floridan Aquifer is used, the score drops significantly.

Baxter/Union Pacific. The Baxter/Union Pacific Tie Treating site, located about 1-1/2 miles from downtown Laramie, WY, consists of about 700 acres. The relatively flat surface drains to the Laramie River. The wastes are from railroad tie treatment: primarily zinc chloride (1886-1931), creosote oil and asphalt residuum (1928-1983), and pentachlorophenol (PCP) (1956-1983), but over 30 chemicals, some possibly dumped in the 1970s, are present in subsurface samples.

For SRS88, the chemicals dominating the score are carcinogens: phenanthrene, anthracene, napthalene, and arsenic. All three pathways contribute to the score, but surface water runoff and air volitization are somewhat more significant [6]. (In 1983, a river levee was built to remove the site from the 100-yr flood plain, but this was not scored.)

The score for Baxter/Union Pacific would drop slightly if the pathway in which groundwater is withdrawn from the alluvial aquifer, as opposed to deeper formations, could be eliminated. Also, if air measurements for volitization of organics from surface ponds show only limited release, the score would drop significantly. On the other hand, anthracene was reported in extremely high concentrations (higher than its water solubility) under the site, because of possible mixture with oil. If the oil is mobile, the site score would increase substantially.

Summary. Although SRS cannot pinpoint the absolute health risk, its parallels to the risk assessment method enable it to identify gaps in the data and suggest important chemicals and pathways at a site. For example, at the five sites selected, carcinogens dominate the scores. Furthermore, arsenic appears as an important contributor at all sites except Hudson River PCB.

Risk-Based Standards for Resolving Regulatory Conflicts Concerning Mixed Waste

Although professionals have been aware for several years of the inconsistencies in laws and regulations that concern risks from noncarcinogenic, carcinogenic, and radioactive wastes, the conflicts become more than academic when the subject is mixed waste.

This section discusses several points of potential confusion and subsequent conflict between the underlying philosophies in the EPA standards for regulating hazardous chemical waste (RCRA) and high-level and transuranic radioactive waste (40 CFR 191). The Waste Isolation Pilot Plant (WIPP), a DOE facility for the disposal of defense-related transuranic waste co-contaminated with hazardous chemicals, is used as an example. According to the EPA notification on radioactive mixed wastes [34] and a DOE interpretive rule codified as 10 CFR 962 ("by-product" standard), all DOE radioactive wastes that contain hazardous wastes are subject to regulation under both RCRA and the Atomic Energy Act (AEA) [35,36]. Thus, although the mission of the WIPP is to provide a safe, permanent disposal site for radioactive wastes, it is also a "treatment, storage, or disposal facility" for hazardous waste.

Summary of High-Level and Hazardous Waste Regulations. Compliance procedures under 40 CFR 191 somewhat resemble a PRA. However, because data for an actual PRA would contain extreme uncertainty (e.g., location of future populations in 10,000 yr), the EPA uses radionuclide release limits at a specified boundary for the disposal system. Also, the cumulative discharge, rather than the concentration variation with time, is required.

In contrast, in the RCRA (§§3004[d] through [n]), Congress established stringent requirements to prohibit land disposal of hazardous waste. However, a petitioner can demonstrate to a "reasonable degree of certainty that there will be no migration of hazardous constituents from the disposal unit for as long as the waste remains hazardous." This request for variance from required waste treatment is submitted in a no-migration petition [37].

The following discussion compares 40 CFR 191 and RCRA in five major areas: (1) containment, (2) monitoring and predictive modeling, (3) post-closure time frame, (4) waste migration, and (5) implementation.

Containment. 40 CFR 191 has an explicit containment philosophy, because radioactive waste cannot be "treated" to remove its radioactivity (except by transmutation, which has unknown but potentially large costs and yet undetermined licensing and technological hurdles). The best demonstrated available technology (BDAT) is deep, geologic disposal with engineered barriers within a natural geologic barrier. The natural barrier is defined by a maximum 5-km boundary.

The RCRA has an implicit containment philosophy. Usually, the hazardous waste must be treated according to standards established by EPA. For each waste, EPA usually determines the BDAT or in some cases the maximum allowable concentration after treatment. Once treated, the waste (for example, residual ash, chemical reactant, etc.) can be disposed in a licensed surface landfill that relies entirely upon an engineered barrier for containment. Thus, the boundary of the facility is usually the engineered barrier.

Two potential sources of conflict are the differences in boundaries and the extent of knowledge about the waste. 40 CFR 191 uses a performance measure for the entire disposal system (both engineered and natural barriers). Hence, the extent of waste characterization depends upon the effect of the estimated amount of waste on the disposal system.

For RCRA, each waste component must be known to select the proper BDAT for treatment. For previously stored waste, sampling to determine waste components can be more hazardous than storing the waste. For example at the WIPP, the waste consists of laboratory and production trash such as glassware, metal pipes, disposable clothing, gloves, cleaning rags, solidified sludges, contaminated with transuranic radionuclides. Sampling exposes individuals to real radiation hazards (not just hypothetical hazards over 10,000 yr), which is in direct conflict with AEA's requirements to keep radiation exposures as low as reasonably achievable

(ALARA). Currently discussions between DOE and EPA on the necessary level of sampling continue.

Monitoring and Predictive Modeling and Associated Uncertainty. Both 40 CFR 191 and RCRA establish two regulatory time periods: operation and post-closure. During the operational phase, both require extensive monitoring, which in turn is used to demonstrate regulatory compliance.

For post-closure, 40 CFR 191 requires predictive modeling to provide a reasonable expectation that the facility will comply. The EPA assumes that predictions about physical phenomena will be adequate for evaluating compliance with the regulations and making comparisons among sites since a priori land disposal had been selected as the best treatment option. Monitoring is desired but not required and must be nonintrusive. Furthermore, the treatment of uncertainty in scientific predictive modeling is carefully defined in 40 CFR 191 but not in RCRA.

Although predictive modeling is used to obtain a land disposal permit, RCRA implies that monitoring at the engineered barrier will be used to verify in audits that the closed facility still complies during an undefined performance period.

Post-Closure Time Frame. The time frame for post-closure in 40 CFR 191 is explicit (10,000 yr). The RCRA statute has no explicit time frame. It specifies only that untreated waste cannot migrate from a land disposal facility for as long as the waste remains hazardous. The time frame is a potential conflict in terms of waste migration, discussed below.

Waste Migration. 40 CFR 191 explicitly recognizes the physical reality that the waste will migrate or be moved (geologically) given enough time. It also requires that migration by unintentional human intrusion be examined and sets an arbitrary 5-km boundary around the waste at which the cumulative release is predicted after 10,000 yr.

RCRA relies primarily upon treatment. When waste is disposed without treatment, hazardous waste migration from a land disposal facility is not allowed. EPA has interpreted this to mean waste migration that exceeds health-based standards beyond the facility boundary (undefined). (For RCRA, the predictive modeling requires knowledge of release rates and dispersivities, unlike 40 CFR 191; furthermore, associated uncertainty is not acknowledged.)

Implementation. 40 CFR 191 is implemented by a federal agency. For the WIPP, 40 CFR 191 may be implemented internally by DOE and externally by EPA with specific public comment by the State of New Mexico oversight group and National Academy of Science.

As desired by Congress, RCRA is implemented by each state. (However, EPA has indicated a desire to maintain at least some control in the review of no-migration petitions.) A potential conflict also exists, because the EPA cannot freely interpret the HSWA, since the legislation was so specific that the law was essentially codified in the regulations [e.g., 37]. Because different agencies implement the regulations for hazardous chemicals and radioactive wastes, there is always the possibility of jurisdictional conflicts [8].

Summary. Although most points mentioned above do not create an impenetrable impasse (in fact, extensive negotiations between EPA and DOE have resolved some conflicts), inconsistencies remain. Furthermore, conflicts require a negotiated position for each generator site with different EPA regions and states. Finally, although WIPP is not regulated by the NRC, sites such as Yucca Mountain are regulated by this agency and therefore must include it as a third party. Because Congress has created the inconsistencies through legislation, they could remove these inconsistencies by clarifying the relationship between hazardous waste laws. Furthermore, we believe that when establishing limits on chemical hazards, the standards should be based on risk evaluations and should acknowledge uncertainty in scientific predictions, as stated in 40 CFR 191. Doing so would diminish most potential disputes and should be a long-term goal of Congress and the regulatory agencies.

Overall Summary

In this paper, we have discussed how risk-based principles can be used to rank mixed-waste sites and compare hazards with differing metrics (e.g., toxicity). In addition, we have described the inconsistencies within legislation concerning hazardous and radioactive wastes and the difficulties they present in the scientific assessment of hazards.

Specifically SRS, developed in 1986, uses risk-based principles to provide a consistent framework for developing a ranking scheme for mixed-waste sites. Its benefits are (1) the logic remains clear so that tradeoffs between accuracy and computational burden are not obscured, (2) the role of the ranking scheme as an assessment tool remains clear, (3) the analyst evaluates the quality of, and identifies gaps in, existing data; and (4) because SRS parallels the risk assessment method, SRS suggests important release mechanisms and thereby identifies remedial possibilities during screening to explore in later investigations.

Within SRS, risk-based principles were used to compare different toxicity metrics. The method allows the components of the mixed waste to be compared instead of scored separately. Results from the SRS, HRS, and rHRS were compared to offer a glimpse of possible consequences from using risk-based principles and not mixing risk assessment, risk management and value judgments in one step.

Finally, inconsistencies in legislation for hazardous and radioactive waste may produce conflicts when trying to identify the hazards of a mixed-waste disposal site. A need exists for better definition in the legislative arena to clarify the role of hazardous waste laws when licensing facilities used for the land disposal of mixed waste. A much bigger task will be to factor in risk when establishing legislative and regulatory limits on hazards instead of basing them solely on familiarity, how well the hazard can be measured, and what can be reasonably achieved, technologically, when the regulations are drafted. Concerning this latter task, since EPA has stated a desire to use risk concepts in establishing regulations, it behooves the various professions involved to aggressively use risk concepts in their work as a form of encouragement and to establish evidence of its general applicability in dealing with both scientific and legislative mixed-waste problems.

Table 1. Comparison of SRS with Original and Revised HRS Composite Ranks for Five Superfund Sites

Superfund Site Name	SRS88			Original HRS		Revised HRS		
	Path[1]	Score[2]	Rank	Score[3]	Rank[4]	Path	Score	Rank
Bunker Hill Mining	GW,SW	8.9	1	55.3	110	GW,SW,A	87.5	1
Tyson's Dump	GW	6.3	2	67.2	25	SW,A	78.2	2
Hudson River PCBs	SW	5.3	3	55.2	111	SW,A	70.7	3
Whitehouse Waste Oil Pits	GW	5.0	4	52.5	147	GW,SW	54.2	5
Baxter/Union Pacific	SW,A	2.8	5	37.2	564	SW	59.3	4

[1]Dominant pathway, where GW = groundwater, SW = surface water, A = air

[2]Log_{10} of consequence (ψ)

[3]Score estimated from NPL rank

[4]NPL Rank as of February 1990

Acknowledgments

The authors appreciate the technical review by J. M. Phelan, C. D. Leigh, and L. S. Gomez of Sandia, the computational support of J. D. Schreiber of Science Applications International Corporation, Albuquerque, NM, and the editorial support of J. Chapman, Tech Reps., Inc, Albuquerque, NM.

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