INEEL/EXT-03-01169
Rev. 2

INEEL Subregional Conceptual Model Report

Volume 3: Summary of Existing Knowledge of Natural and Anthropogenic Influences on the Release of Contaminants to the Subsurface Environment from Waste Source Terms at the INEEL

September 2003

Idaho National Engineering and Environmental Laboratory
Bechtel BWXT Idaho, LLC

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Idaho National Engineering and Environmental Laboratory
Idaho Falls, Idaho 83415

Prepared for the
U.S. Department of Energy
Assistant Secretary for Environmental Management
Under DOE Idaho Operations Office
Contract DE-AC07-99ID13727

ABSTRACT

This source-term summary document is intended to describe the current understanding of contaminant source terms and the conceptual model for potential source-term release to the environment at the Idaho National Engineering and Environmental Laboratory (INEEL), as presented in published INEEL reports. The document presents a generalized conceptual model of the sources of contamination and describes the general categories of source terms, primary waste forms, and factors that affect the release of contaminants from the waste form into the vadose zone and Snake River Plain Aquifer. Where the information has previously been published and is readily available, summaries of the inventory of contaminants are also included. Uncertainties that affect the estimation of the source term release are also discussed where they have been identified by the Source Term Technical Advisory Group. Areas in which additional information are needed (i.e., research needs) are also identified.

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ACRONYMS

ABRA	Ancillary Basis for Risk Assessment
AEC	Atomic Energy Commission
ANL	Argonne National Laboratory
ANP	aircraft nuclear propulsion
ARA	Auxiliary Reactor Area
ASWS	Air Support Weather Shield
BORAX	Boiling Water Reactor Experiment
BRA	Baseline Risk Assessment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFA	Central Facilities Area
CIDRA	Contaminant Inventory Database for Risk Assessment
COC	contaminant of concern
DOE	U.S. Department of Energy
ECF	Expended Core Facility
EOCR	Experimental Organic-Cooled Reactor
EPA	Environmental Protection Agency
HDT	Historic Data Task
HTRE	Heat Transfer Reactor Experiment
ICDF	Idaho CERCLA Disposal Facility
ICPP	Idaho Chemical Processing Plant
IET	Initial Engine Test
INEEL	Idaho National Engineering and Environmental Laboratory
INTEC	Idaho Nuclear Technology and Engineering Center
ISB	in situ bioremediation
IWD	Industrial Waste Ditch

LLW	low-level waste
LOFT	Loss-of-Fluid Test (facility)
LPTF	Low Power Test Facility
LTCD	Long Term Corrosion/Degradation Test
MCL	maximum contaminant level
MNA	monitored natural attenuation
NAT	neutron-probe access tube
NEPA	National Environmental Policy Act
NRF	Naval Reactors Facility
OAC	Operating Area Confinement
OU	operable unit
PBF	Power Burst Facility
PCB	polychlorinated biphenyl
RFP	Rocky Flats Plant
RI/FS	remedial investigation/feasibility study
ROD	Record of Decision
RPDT	Recent and Projected Data Task
RSAC	Radiological Safety Analysis Computer
RWMC	Radioactive Waste Management Complex
RWMIS	Radioactive Waste Management Information System
SDA	Subsurface Disposal Area
SRPA	Snake River Plain Aquifer
TAN	Test Area North
TRA	Test Reactor Area
TRU	transuranic
TSF	Technical Support Facility

USGS	United States Geological Survey
VOC	volatile organic compound
WAG	waste area group
WIP	Water Integration Project
WRRTF	Water Reactor Research Test Facility

INEEL Subregional Conceptual Model Report Volume 3—Summary of Existing Knowledge of Natural and Anthropogenic Influences on the Release of Contaminants to the Subsurface Environment from Waste Source Terms at the INEEL (Draft)

1. INTRODUCTION

The National Research Council defines a conceptual model as "an evolving hypothesis identifying the important features, processes, and events controlling fluid flow and contaminant transport of consequence at a field site in the context of a recognized problem." Development of a conceptual model is a necessary first step in the evaluation of contaminant transport at any scale within a hydrologic system. Conceptual models of subsurface contaminant transport at the Idaho National Engineering and Environmental Laboratory (INEEL) require an understanding of the rate and mechanism of contaminant release from its source term, movement of the contaminant through the vadose zone to the Snake River Plain Aquifer (SRPA), and movement of the contaminant through the aquifer. A summary of key components of INEEL subregional and facility-specific conceptual models of flow and contaminant transport in the SRPA is presented in Volume I of this series of INEEL Subregional Conceptual Model (Orr et al. 2003). A summary of conceptual-model components of vadose-zone transport (in preparation) will be presented in Volume II.

After evaluating the conceptual models for contaminant flow and transport, the INEEL Water Integration Project (WIP) determined that definition of the contaminant source terms and estimation of release into the environment was a topic that deserved further evaluation. However, because of the considerable complexity of the subject, the technical advisory group recommended that an initial exploratory effort be conducted. A team of researchers made an exploratory effort early in 2003 to identify readily available source-term information and conceptual models of contaminant release at the INEEL.

This document is intended to summarize the present understanding of contaminant release from source terms at the INEEL. The document presents a generalized conceptual model for estimation of the release of key contaminants from the INEEL as a whole to the vadose zone or directly to the aquifer. The document focuses on processes that occur only in the area of the source term. The document describes the different types of source terms at INEEL, including buried waste, injection wells, contaminated soils, and releases of contaminated water (i.e., infiltration from ponds and ditches) or other process liquids.

This document presents results of the initial investigation and is intended only to provide the technical advisory group a starting point to plan a course forward. This material is by no means intended to be comprehensive or complete. It was beyond the scope of this effort to attempt to resolve any contradictions, unclear information, or other uncertainties in description of the various source terms.

2. SUBREGIONAL SOURCE TERM CONCEPTUAL MODEL

Operations at INEEL facilities have involved a wide variety of processes that changed over time as operations and missions of the facilities evolved. These processes resulted in the generation of a wide variety of waste forms, including liquids, solids, and solidified wastes. INEEL also received wastes from other government and private facilities for a period of time. These wastes were managed in a variety of ways, including disposal in injection wells, discharge to ponds, and burial in shallow surface soils. In addition, soils at some of these facilities became contaminated by spills, leaks, or other discharges from operations or waste-management activities.

In order to estimate the future impact of these past waste management activities on the quality of the aquifer, researchers are faced with the dilemma of developing conceptual models that provide reasonably accurate representation of the release, transport and fate of contaminants within this complex and difficult-to-characterize system. As discussed in the Volume 1 of the INEEL Subregional Conceptual Model Report, components of the conceptual model include source-term release, vadose-zone transport, and contaminant fate and transport in the aquifer. All three of these components are coupled to predict future potential levels of contamination and associated risk in the SRPA.

The Revised Preliminary Scoping Risk Assessment for the Radioactive Waste Management Complex (RWMC) (Burns et al. 1994) included a sensitivity study that is significant in that it is one of the only instances where a quantitative sensitivity study of the overall risk assessment model was performed. In the Burns et al. (1994) analysis, sixteen parameters were chosen which had been identified as uncertain values that were expected to have a measurable effect on the modeling results. Four contaminants (Am-241, C-14, I-129 and Sr-90) were selected as sensitivity indicators. Through a series of model runs, numerical values representing “scaled sensitivity” were computed for each input parameter. Scaled sensitivity was a measure defined as the ratio of input change to output change times the range of uncertainty in the input parameter, times the base case risk value. The results of the analysis indicated that the release rate had the greatest scaled sensitivity. Within the release rate calculations themselves, soil to water partition coefficients were identified as an important controlling parameter.

Other factors that influence the groundwater pathway include porosity, well screen thickness, infiltration rate, contaminant source-term inventory, source length and source area, and soil to water partition coefficients (as applied to retardation contaminant transport). The details of the analysis are not important, as the work was performed for a specific purpose, but what is important is the idea that WIP develop a methodology to identify those specific issues that are not only are uncertain with a wide range of possible values, but also impact the overall risk model in a meaningful way.

2.1 Waste Inventories

Operations at INEEL facilities involved a wide variety of processes that changed over time as operations and missions of the facilities evolved. These processes resulted in the generation of a wide variety of waste forms, including liquids, solids, and solidified wastes. These wastes were managed in a variety of ways, including disposal in injection wells, discharge to ponds, and burial in shallow surface soils. In addition, soils at some of these facilities became contaminated by spills, leaks, or other discharges from operations or waste management activities.

The major categories of wastes generated by activities at the INEEL and the disposal methods are shown in the center column of a process flow diagram (Figure 2-1). Subsets of these waste categories (i.e., solid, solidified, or aqueous waste) are shown as arrows that lead to the disposal avenue for that waste stream. Examples of disposal avenues include landfills and injection wells. With the exception of wastes that are disposed at off-site facilities or placed into dry storage, all types of wastes disposed at the INEEL have the potential to release contaminants that may eventually migrate into the SRPA.

Various waste disposal methods introduce wastes into different portions of the environment. Wastes are released at or near the ground surface by burial in landfills, and discharge into ditches and percolation ponds. Contaminated soils are another example of wastes near the ground surface. Most injection wells introduce wastewater directly into the SRPA, but some have also released wastewater into the vadose zone above the aquifer. Table 2-1 identifies the portion of the environment that wastes are released into by each disposal method. The waste form (aqueous, non-aqueous liquid, organic solid, or inorganic solid) disposed by each method is also identified. In this table, non-aqueous liquid refers to liquids that are immiscible with water, such as petroleum and chlorinated solvents. Organic solids are primarily sewage sludges, and inorganic solids include sediment and granular solid waste such as floor sweepings and debris from machining and cleaning operations.

Table 2-1 also identifies “important mechanisms” for contaminant release to the environment. These mechanisms are important either in the release of contaminants from the waste, or during migration of contaminants in the subsurface after release from the waste itself. Some mechanisms are important in both conditions. These mechanisms are discussed in the following section. Information about the type and amount of wastes disposed via each approach is provided in Section 3 of this document.

2.2 Contaminant Generation, Transport, and Fate Mechanisms

This section provides a high-level description of the mechanisms that are important in controlling the fate of contaminants that are constituents of waste disposed in the subsurface at the INEEL. These mechanisms include those involved in releasing contaminants from the original waste form and those affecting the transport and fate of contaminants after release from the original waste form. Figure 2-2 summarizes the types of mechanisms that typically govern the exchange of contaminants between the altered waste zone and the natural geochemical environment in the subsurface. It is appropriate to discuss mechanisms that are important after contaminant release from a waste because these mechanisms can, in some cases, act to form secondary contaminant sources that continue to release contaminants into water after the original waste has ceased to do so. Secondary contaminant sources may act as long-term sources of contaminated groundwater that may persist well beyond the lifetime of the original waste. This section provides a summary of the mechanisms that are believed to be important for different disposal methods and a description of each of the mechanisms identified.

2.2.1 Advection and Dispersion

Advection and dispersion affect all wastes released into the subsurface at INEEL. Advection is the movement of contaminants dissolved in water caused by the bulk movement of that water. Dispersion is the spreading of contaminants in water, and is caused by differences in the length of the flow path traversed by different parcels of water (and the contaminants dissolved in that water) and differences in flow velocity in different flow paths. Dispersion mixes a plume of contaminated groundwater or vadose zone water with surrounding uncontaminated water, and acts to reduce maximum contaminant concentrations.

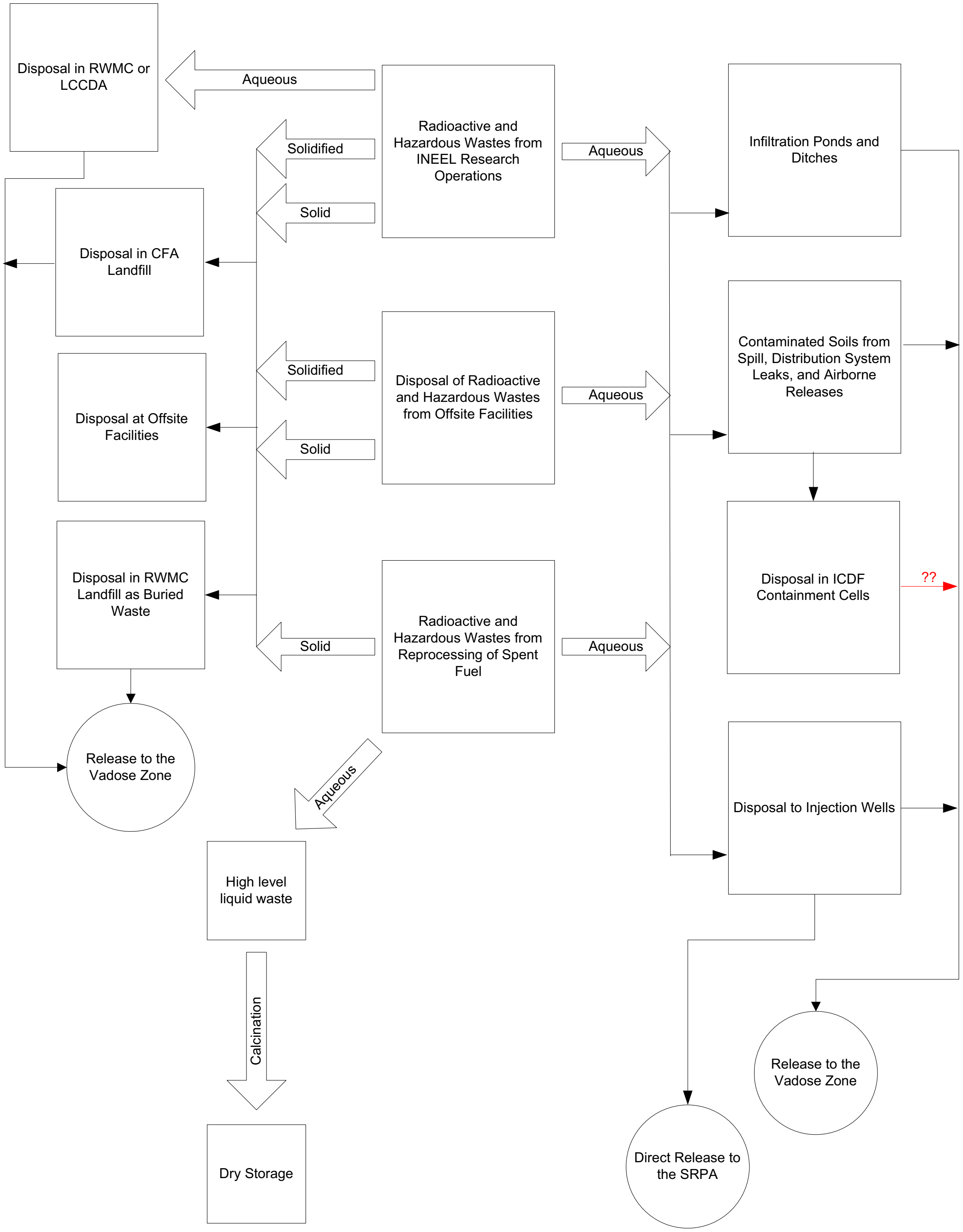


Figure 2-1. Summarized conceptual model for the Idaho National Engineering and Environmental Laboratory.

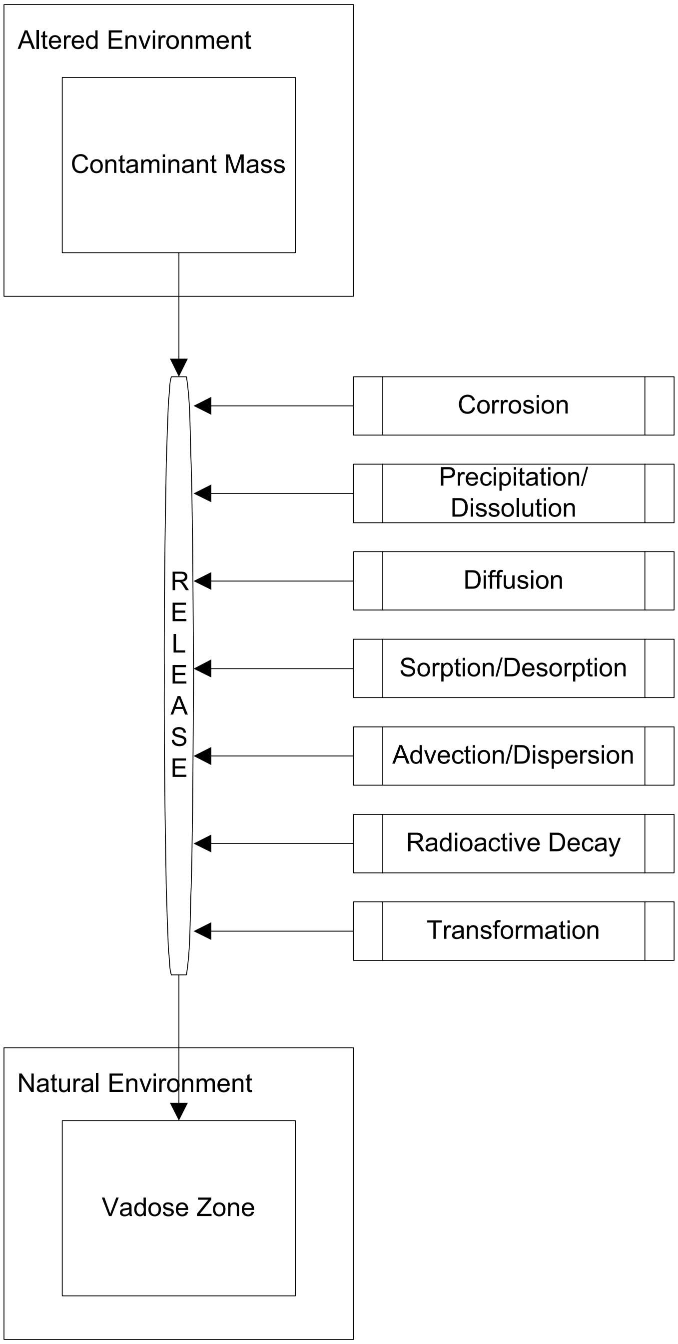


Figure 2-2. Mechanisms of release important in controlling fate of contaminants.

Table 2-1. Disposal location, waste form, waste constituents, and mechanisms associated with methods of contaminant disposal to the subsurface at the Idaho National Engineering and Environmental Laboratory.

DISPOSAL METHOD	**DISPOSAL LOCATION**			**WASTE FORM**				**WASTE CONSTITUENTS**			**IMPORTANT MECHANISMS**					
	Surface or Near-Surface	**Vadose Zone**	**SRP Aquifer**	**Aqueous**	**Non Aqueous Liquid**	**Organic Solid**	**Inorganic Solid**	**Radionuclides**	**Organics**	**Inorganics**	**Advection and Dispersion**	**Diffusion**	**Precipitation / Dissolution**	**Sorption / Desorption**	**Transformation**	**Radioactive Decay**
Injection Wells		+	++	++	+	+	+	+	+	+	++	+	+	++	+	+
Burial	++	+			+	++	++	++	+	+	+	+	++	+	+	+
Infiltration Ponds and Ditches	++	+		++		+	+	++		+	++			++		+
Contaminated Surface Soils	++	+					+	+		+	+			++		+

\+ indicates relative importance or magnitude.

Advection and dispersion are primary transport mechanisms for aqueous wastes that were discharged directly into the SRPA. Advection and dispersion are also important for aqueous contaminants discharged into percolation ponds. Typically, some of the water discharged into percolation ponds infiltrates and flows downward through the vadose zone until it reaches the underlying aquifer, or it may temporarily reside in perched water bodies in the vadose zone.

Water from precipitation that comes into contact with either buried waste or contaminated soils may become contaminated due to mass transfer of contaminants from these sources. This contaminated water can then move downward through the vadose zone and ultimately reach the underlying aquifer. Given the low average precipitation rate and the corresponding low infiltration rate, advection and dispersion are usually of minor importance relative to transport of contaminants from these waste sources. However, when these sources are flooded, advection and dispersion may become important mechanisms.

2.2.2 Diffusion

Diffusion is the movement of constituents as a result of spatial concentration gradients. Diffusion transports mass from areas where concentrations are high to areas where concentrations are low. Diffusion occurs in both water and air, and diffusion in both media is important relative to contaminant transport in the subsurface at INEEL.

Diffusion transports mass slowly in water. Nevertheless, diffusion is believed to be an important process in release of contaminants from porewater in sludge injected into the aquifer near well Technical Support Facility (TSF)-05 at Test Area North (TAN). Diffusion is also important during dissolution of minerals and organic liquids. Processes that increase the concentration gradient will increase diffusive mass transfer and hence the rate of dissolution.

Diffusion of contaminants in dual porosity media (media in which adjacent zones have very different permeability) serves both to retard contaminant transport and to act as a long-term secondary source after the primary source is depleted. While the primary source is generating contaminated water and contaminant concentrations are higher in water in the more permeable zones than in the less permeable zones, diffusion transports contaminant mass into the porewater of the less permeable zones. After the primary source is depleted, the concentration reverses and diffusion transports contaminant mass from porewater in the less permeable zones back into the more permeable parts of the system. This causes long tails on elution curves and can substantially extend the time needed to remediate a contaminated site compared to the time that would be needed if diffusion to/from less permeable parts of the system did not occur.

Diffusion in air is an important process if volatile contaminants are present. Diffusion transports mass much more quickly in air than in water because gas phase diffusion coefficients are much larger than aqueous phase diffusion coefficients. Volatile contaminants in the subsurface, such as organic solvents in buried waste at the RWMC, can diffuse substantial distances –many tens of feet – in a matter of months in the gas phase of an unsaturated material. Contaminants can partition from the gas phase into water, and the contaminated water can then recharge the aquifer thereby generating contaminated groundwater. The significance of diffusion in the gas phase is that it provides a mechanism for rapid migration of volatile contaminants in the vadose zone.

2.2.3 Dissolution and Precipitation

Dissolution refers to the conversion of a solid phase into dissolved constituents, and precipitation is the formation of a solid phase from dissolved constituents.

Dissolution is primarily important because it is one mass transfer process by which constituents of a waste can become dissolved contaminants. Dissolution is probably most important as a process that converts buried waste into dissolved contaminants. It could also be important if soluble solids were disposed via injection wells, or if aggressive solutions were injected into the subsurface. Situations in which dissolution would be important include soluble salts in buried waste, or metals in buried waste could corrode and release dissolved metal ions. Dissolution may also be important if contaminants are precipitated, coprecipitated, or sorbed to minerals that are stable under one set of geochemical conditions, and are subsequently released due to changes in geochemical conditions, such as change in pH or redox state. An example of this is dissolution of pre-existing minerals following disposal of acidic waste in the subsurface.

Precipitation could be an important contaminant sequestering mechanism if contaminants were precipitated or coprecipitated in minerals that formed after wastes were introduced into the subsurface. An example is incorporation of strontium or other metals in calcite via coprecipitation. A second example is precipitation of solid phases following release of concentrated solutions that have extreme pH. Neutralization of pH by reactions with native minerals in the subsurface could cause other minerals to precipitate and thereby sequester contaminants.

2.2.4 Sorption and Desorption

Sorption is the partitioning of dissolved constituents to solids. Desorption is the release of materials from solids to the aqueous phase. Sorption includes adsorption, in which the partitioning is a surface phenomenon, and absorption in which partitioning occurs in the interior of the solid. A common example of adsorption is cation exchange, in which positively charged ions are attracted to negatively charged mineral surfaces by electrostatic forces. An example of absorption is the incorporation of potassium ions in interlayer positions of illite, a clay mineral. In this discussion, sorption is used as a global term that includes both absorption and adsorption.

Sorption is important primarily because it can drastically retard the movement of ionic contaminants. Cations are generally retarded much more than anions because most common minerals have negatively charged surfaces, which attract positively charged ions. Ions that are sorbed move more slowly than water is moving through the medium. The ratio of the rate of water velocity to contaminant velocity is the retardation coefficient, R. An unretarded solute, which moves at the same velocity as water, has a retardation coefficient of 1. Retarded solutes, which move slower than water, have retardation coefficients greater than 1. Some strongly sorbed contaminants have retardation coefficients on the order of 10,000 and are essentially immobile. Most contaminants have much lower retardation coefficients, and although they move more slowly than water, they are not immobile.

Sorption of contaminants has several effects. First, as previously discussed, it retards the migration rate. Second, it reduces concentrations in water. Third, it increases concentrations in the sorbed phase.

At some point, waste in the subsurface will no longer release contaminants into water, and concentrations in water downgradient of the waste will decline. If contaminants have previously been sorbed by the solid phase downgradient of the waste, they will desorb as the flux of contaminants from the primary source decreases, and thus the minerals that have sorbed contaminants will act as a secondary source of contaminants. Hence, sorption/desorption is a two-edged sword. It slows contaminant transport and reduces concentrations during the active life of a primary source, but it can also extend the time that contaminants will enter groundwater and the time needed for remediation. Like diffusion in dual porosity media, desorption can cause long tails on elution curves during remediation activities.

Sorption can be an important mechanism that removes contaminants from water discharged to percolation ponds. Cationic contaminants can be sorbed by the surficial sediments in which the pond was constructed, or by sediments in interbeds transited by water that infiltrated from percolation ponds. Sorption is also probably an important mechanism that binds some contaminants, e.g. cesium, to mineral surfaces in contaminated soils.

Desorption may also be important in the generation of dissolved contaminants from buried waste, from sediments in percolation ponds, or from contaminated soils. If uncontaminated water comes in contact with these materials, desorption could transfer contaminant mass from the waste into water.

2.2.5 Transformation

Transformation refers to the change in chemical species. It includes both biologically-mediated and abiotic transformations.

Transformations can be very important relative to the mobility, toxicity, and solubility of contaminants. Common transformations change in speciation due to redox reactions (i.e., Fe-II ⬄Fe-III or Cr-VI ⬄ Cr-III), and due to change in pH (uranium). Hexavalent chromium forms oxyanions that are soluble, not significantly retarded, and toxic. Trivalent chromium forms cationic species that are much less mobile, less soluble, and less toxic than hexavalent chromium species. The characteristics of many metals vary with pH and redox state.

Biologically-mediated transformations are also important factors that affect release of contaminants from waste and subsequently the transport and fate of contaminants in the subsurface. For example, biodegradation of organic matter in landfills such as the RWMC can generate reducing conditions that are favorable to release of some metals, and the inorganic and organic acids produced by organic matter biodegradation may accelerate corrosion of waste metals. Biotransformation may be important in converting nitrate into innocuous nitrogen. Biotransformation is very important in the fate of organic contaminants in the subsurface. For example, tetrachloroethene (PCE) and trichloroethene (TCE) in the groundwater plume emanating from TSF-05 at TAN are being transformed by biological processes into less chlorinated ethenes and ultimately into innocuous products, ethene, chloride, bicarbonate, and water. Biodegradation also commonly degrades contaminants derived from petroleum into innocuous products.

2.2.6 Radioactive Decay

Radioactive decay is important in that it reduces the mass of radioactive contaminants. If the daughter products are stable or long-lived, then radioactive decay reduces the amount of contaminants present. Conversely, if the daughter products are radioactive with half-lives comparable to those of the parents, then the overall amount of activity does not change.

3. SOURCE TERM CATEGORIES

To explore the feasibility of developing an integrated site-wide inventory of contaminant source terms, a team of researchers investigated the available source term information. To facilitate this research, available source terms were grouped into four general areas: injection wells, buried waste, ponds and ditches, and contaminated soils. For each of these areas, an attempt was made to collate information regarding the inventory of contaminants, factors controlling release, and known uncertainties. Although the task to assemble an exhaustive description of all INEEL source terms was beyond the scope of this document, general descriptions and some readily available information were collated and are presented in the following sections. It is important to note that due to the large number of investigations conducted at the INEEL, some of which are still ongoing, it was not possible to cross reference and check the data. In all cases where the reader intends to use data regarding a particular source term, they are referenced back to the original documents and will need to verify that they use the most current information.

3.1 Injection Wells

Starting in the 1950s, injection wells were constructed at the INEEL for disposing wastewaters directly into the SRPA or in some situations into the vadose zone. These injection wells provided a historical means of waste disposal, but are no longer used; injections ceased in the 1980s. The types of waste and volumes injected into each well varied based on location of the injection well in relation to contaminant sources. Table 3-1 presents the eleven injection wells discussed in this chapter. Organization of this section begins with a discussion of inventory for each injection well (Section 3.1.1), followed by a section describing factors controlling release of source term (Section 3.1.2) and finally uncertainties in the available data (Section 3.1.3).

Table 3-1. Injection wells at the Idaho National Engineering and Environmental Laboratory.

Injection Well	Facility	Years of Operation
TSF-05	TAN	1953-1972
IET-06		1956 to mid-1960s and a few years around 1978
WRRTF-05		1957-1984
LOFT-04		Constructed in 1957, used until at least 1980
TRA-05	TRA	1964-1982
USGS-53		1960-1964
CPP-23	INTEC	1952-1984
USGS-50		During CPP-23 failure
PBF-05	PBF	1973-1984
PBF-15		1972-1978
EOCR-2	EOCR	Never used

3.1.1 Inventory

In this section, each injection well constructed at the INEEL is discussed, even if it was never put into service. For the injection wells with documented information, date of construction, well description, years of operation, and inventory and types of waste disposed are provided.

TSF-05

The TSF-05 injection well (TAN-330) was drilled in 1953 to dispose of liquid effluent generated from the aircraft nuclear propulsion (ANP) program. It is located in the southwest corner of TSF within TAN. The well is 310 ft (93 m) deep and the depth to groundwater is about 206 ft (63 m). It has a 12-in (30.5-cm) diameter casing to 305 ft (93 m) and is perforated from 180 to 244 ft (55 to 74 m) and 269 to 305 ft (82 to 93 m) below ground surface (bgs).

The TSF-05 well was used from 1953 to 1972 to dispose of liquid wastes generated at TAN. Following the last time the well was used as a primary disposal site, in September 1972, the wastewaters were diverted to the TSF disposal pond. Until the early 1980s, the well was available to be used for overflow from the sump at TAN-655 in the event of a power failure, an equipment failure, or equipment maintenance that precluded discharge to the pond, but there are no records that indicate such overflows actually occurred (Kaminski et al. 1994).

The wastes injected into TSF-05 mainly consisted of industrial and sanitary wastewater and also included organic, inorganic, and low-level radioactive constituents. Activities that generated these wastes included efforts to develop a nuclear-powered aircraft and tests simulating accidental loss of coolant from nuclear reactors (DOE et al. 2001). The hazardous wastes include corrosive and ignitable wastes from shop operations and potentially corrosive and toxic condensate from the intermediate-level waste disposal system evaporator (Kaminski et al. 1994).

Limited information is available for compiling an inventory of waste injected into TSF-05 because historical records provide little definitive information on the types and volumes of organic wastes disposed to the well. Based on characterization data for groundwater and sludge from the well, disposal of fairly large quantities of organics and radionuclides are indicated. Estimations show as little as 350 gal (1325 L) of TCE may have been disposed of in TSF-05 (DOE et al. 2001). A suggested estimate of as much as 35,000 gal (132,489 L) of TCE may have been disposed in the well. This may be an overestimate, since it is based on personnel interviews. Records of TCE use at TAN apparently show that the Decontamination Room and Pipe Laundry at TSF used 1,000 to 1,400 gal (3,785 to 5,300 L) of TCE per year and the Air Filter and Weld Shops at the Loss-of-Fluid Test (LOFT) facility and the 607 Valve Shop reportedly used 110 gal (415 L) of TCE per year. While TSF-05 was in operation from 1955-1972, these sources could have contributed 25,670 gal (97,161 L) of TCE (Kaminski et al. 1994). In 1999, numerical modeling was performed using TETRAD to assess the mechanisms of natural attenuation (dispersion, sorption, and biodegradation). This calibrated model was used to inversely estimate the initial source and a residual source rate. An initial amount of TCE injected was estimated to be 1,100 gal. The residual source rate was estimated as 0.0026 gal/day of TCE, which was injected after the disposal well closure in 1972 (Martian 1999).

Toxic heavy metals are also suspected of being discharged into TSF-05. Reasons for suspecting metals discharge include mercury contamination in the later 1950s and early 1960s, the use of a potassium chromate solution in decontamination activities after 1970, and the abundance of lead used for shielding materials that were decontaminated with corrosive solutions. Personnel interviews indicated that concentrated material from the evaporator system was discharged to the well from the late 1950s to the early 1960s.

The Radioactive Waste Management Information System (RWMIS) lists the number of curies of individual nuclide releases to the injection well from 1971 through August 1972. Records of the radioactivity released before 1971 are limited. Based on the estimated amount released from 1959 through 1970 (45 Ci) and the reported RWMIS amount released in 1971 through 1972 (8.534 Ci), the estimated total released to TSF-05 is 53.53 Ci. Estimated totals for some contaminants of concern (COCs) are reported in Table 3-2 (Kaminski et al. 1994).

Table 3-2. Estimated amount of contaminants injected into TSF-05.

Contaminant	Estimated Amount Injected
Trichloroethylene	350 to 35,000 gal (25,670 gal)
Tritium	5.32×10^{1} Ci
Strontium-90	5.42×10^{-2} Ci
Cesium-134	2.88×10^{-2} Ci
Cesium-137	1.37×10^{-1} Ci
Unidentified alpha	6.55×10^{-3} Ci
Unidentified beta and gamma	5.35×10^{-2} Ci
Yttrium-90	5.42×10^{-2} Ci

Disposal of contaminants to the TSF-05 disposal well resulted in an immediate release to the SRPA. Continued disposal also resulted in accumulation of sludge and contaminants in the aquifer around the disposal well. This sludge has provided an attenuated release of organic and radionuclide contaminants from this secondary source. Further discussion of the secondary source sludge remaining in TSF-05 is discussed in the factors controlling release section.

IET-06

Initial Engine Test (IET)-06, the IET Injection Well TAN-332, was drilled in 1953 to dispose treated sanitary sewage and process wastewater from the IET facility. It is located north of TSF within the former IET facility, which was decommissioned in 1985 and 1986. The well is 329 ft (100 m) deep with a 12-in diameter well casing that is perforated from 219 to 319-ft. Concrete was poured in the bottom of the casing for a total depth of 5 ft, and gravel was packed around the outside of the casing to the surface. In 1986 the well depth was documented as being 250 ft, indicating that the casing had collapsed or that the gravel pack had come through the casing perforations (INEEL 1994).

The IET-06 injection well was used from 1956 to the mid-1960s for the disposal of wastes generated from the Aircraft Nuclear Propulsion Program, the Space Nuclear Auxiliary Power Transient Program, and for a few years around 1978 for the Hallam Decontamination and Decommissioning Project (INEEL 1994).

The wastes consisted of treated sanitary sewage, process wastewater, and other noncontaminated wastes. The sanitary sewage wastes came from restrooms in the Control and Equipment Building (TAN-620) and the Change Room (TAN-656) via a septic tank and sand filter system (IET-08). The overflow from the septic tank was treated with chlorinated water, discharged to the septic tank and sand filter, and subsequently discharged to the IET-06 injection well. Both process wastewaters and noncontaminated wastes entered the injection well directly via drain lines connected to TAN-620 and TAN Test Cell Building located west of TAN-620 (INEEL 1994).

The process water discharges were mainly associated with the operation of a boiler and water treatment system. A boiler was located in Building 620 and it is probable that the water treatment for this boiler consisted of sodium sulfite, di-sodium phosphate, and tri-sodium phosphate as additives. These chemicals would have reached the injection well via the process wastewater drains during startups and shutdowns and periodic boiler blowdowns. The water treatment system included demineralizer columns, which were regenerated through the use of sulfuric acid and sodium hydroxide. The regenerants were routed through the process water drain, then to the injection well (INEEL 1994).

The process waters may also have included acidic ion exchange regenerants, waste diesel fuels and boiler blowdown from TAN-620 boilers, small amounts of engine coolant or fuels from the nuclear engine tests, and possibly wastes from a photo lab in TAN-620 (Kaminski 1994). Information to compile an inventory of waste injected into IET-06 is not available. During Track 1 investigations, identification of past releases was hampered by the fact that both the contractor that initially operated the facility and the firm that designed and built it were no longer at the INEEL (INEEL 1994).

Possible discharges of mercury were considered during Track 1 investigation, although no mechanism for mercury discharge to the well has been identified. Mercury was used in shield tanks on the ANP cores during maintenance work at TSF. The mercury was supposed to be removed and replaced with water before the cores were moved back to the IET Facility, but significant quantities of mercury were found on the test pad, the railroad tracks, and elsewhere throughout the IET Facility (INEEL 1994).

The IET-06 well was converted to a United States Geological Survey (USGS) monitoring well in 1980 and a pump was installed in 1982. As part of the IET facility decontamination and decommissioning in 1985 and 1986, the injection well was capped and all discharge pipes were sealed (INEEL 1994).

WRRTF-05

Water Reactor Research Test Facility (WRRTF)-05, also known as the WRRTF injection well or the Low Power Test Facility (LPTF) disposal well, was drilled in 1957. It is located about 280 ft south of WRRTF. The well is 313 ft deep and has a 10-in. diameter casing. In March 1984 the well became blocked and was taken out of service (Frederick et al. 1998). It was grouted and abandoned in September 1984 (Kaminski et al. 1994).

Wastewater disposals to WRRTF-05 included discharges of treated sanitary waste from 1957 to 1981 and discharges of liquid chemical waste from 1957 to 1984. The wastewater sources included waste softening activities and cooling water from boilers (Frederick 1998). In addition, based on facility operations, the well is thought to have received cooling water effluent, boiler blowdown, sanitary waste, small amounts of process wastewaters, and materials from laboratories and process drains. There are also indications that hydrazine from facility operations was disposed in the well. There is no evidence of large volumes of concentrated wastes being disposed of to this well (Kaminski et al. 1994).

Inventory information is known for contaminants listed in Table 3-3. TAN groundwater plume contour plots show the origination of chloride and sulfate plumes around the WRRTF injection well, separate from the plumes originating from TSF-05. Contours of 100, 50, and 20 mg/L for chloride and 40 and 30 mg/L for sulfate indicate the distribution of the injected wastewater from the injection well (Bukowski et al. 1998b, Wymore et al. 2000). Cobalt-60 and turbine oil were documented as being released in 1969 and 1967, respectively (Frederick et al. 1998).

Table 3-3. Estimated amount of contaminants injected into WRRTF-05.

Contaminant	Estimated Amount Injected
Chloride	21,821 lb
Phosphate	709 lb
Sodium	16,447 lb
Sulfate	6,029 lb
Sulfite	1,005 lb
Cobalt-60	50 mCi
Turbine Oil	56 gal

LOFT-04

The LOFT-04 Injection Well (TAN-333), was constructed in 1957 to dispose of noncontaminated or cold wastewater from LOFT operations. It is located northeast of the Liquid Waste Storage Tank Building (TAN-726). The well is 300 ft deep (Frindlund 1992). Between 1971 and 1980, the monthly discharge to the well ranged from less than 10,000 gal to over one million gal (Frederick et al. 1998). No evidence exists that hazardous constituents ever entered this system. The well was converted into USGS monitoring well and the inlet piping has been disconnected (Frindlund 1992).

TRA-05

The waste disposal well for Test Reactor Area (TRA), TRA-05, was drilled in 1962 and 1963 and put into service in 1964 to dispose cold wastewater. It is located 275 ft west of the Warm Waste Pond in the southeast corner of the TRA facility (Hull 1989). The well was drilled and cased to a depth of 1,271 ft with ¼ in by 6 in slot perforations in the casing from 1182 to 1267 ft. Between July 29 and August 3, 1964, additional perforations were added between 930 and 1070 ft. During this time, the casing was severed at a depth of 1005 ft. A third set of perforations was made in the interval between 512 and 697 ft on August 6, 1964. These additional sets of perforations were added following injection tests. The well intersects the SRPA at all perforated levels (Hull 1989). The top of the disposal well was in the bottom of the cold waste sampling pit and water flowed down the well under the influence of gravity. The top of the well was covered with a screen to prevent debris from entering the well (Frederick et al. 1998).

TRA-05 was used from November 1964 through 1982 (DOE et al. 1995). Disposal included wastewater from the cold waste sampling pit (TRA-764) (Burns et al. 1997), secondary cooling water from the reactors, and nonradioactive wastewater from the Hydraulic Test Facility (TRA-666), the Hot Cells (TRA-632), the Steam Plant (TRA-609), maintenance and support buildings, and pumphouses (Frederick et al. 1998).

An estimated 3.9×10^9 gal of liquid waste was discharged to the disposal well between 1965 and 1982. Minimal information is available on the quantity of chemicals discharged to the well, but the waste stream probably contained similar contaminants as the effluent to the cold waste pond (Table 6-6 in Frederick et al. 1998). The only difference from the ponds waste stream is that, prior to 1972, hexavalent chromium was used as a corrosion inhibitor in the secondary cooling water. The concentration in the cooling water was 11-14 mg/L and the concentration in the blowdown stream was about 4-5 mg/L (Frederick et al. 1998). It is estimated that 31,130 pounds of hexavalent chromium was discharged to the TRA disposal well (Hull 1989).

In 1982, disposals to TRA-05 ceased. At this time, TRA-05 was converted to a USGS monitoring well (DOE et al. 1995) and the wastewater was rerouted to the Cold Waste Pond (Hull 1989). Groundwater from TRA-05 contains tritium and chromium above maximum contaminant levels (MCLs). Injections into this well were the primary source of chromium since the water in the cooling towers was treated with chromate to inhibit corrosion (Burns et al. 1997).

USGS-53

USGS-53 is 90 ft deep and completed in the deep perched water zone. It was used for the injection of chemical waste (Hull 1989). It is located on the southeast corner of TRA near the warm waste pond. This well was used from November 1960 to January 1962, from June 1963 to August 1963, and from November 1963 to September 1964. Injection rates were estimated at about 1.48×10^5 gal/day. Through estimations using this flow rate and the periods of operation, approximately 2.2×10^8 gal of liquid were injected in USGS-53. No documented information is available on the type of waste disposed to the well, but it is believed to have been secondary coolant blowdown water (Dames and Moore 1992). This well has not been used for waste disposal since 1964 (Hull 1989).

CPP-23

The Idaho Chemical Processing Plant (ICPP) injection well (CPP-23) was drilled in 1950 to dispose low-level radioactive and chemical wastes. It is located north of building CPP-666 at Idaho Nuclear Technology and Engineering Center (INTEC), formerly the ICPP. The well was drilled to 212 ft (64.6 m) in 1950 and deepened to 597 ft (182 m) in 1951. It has a 24-in (61-cm) diameter borehole and was cased using 16-in (41-cm) nominal diameter carbon steel casing. The annular space between the borehole and casing was filled with gravel. The well casing is perforated from 412 to 452 ft (125.6 to 137.8 m) and 490 to 593 ft (149.4 to 180.7 m) below land surface (bls) (Rodriguez et al. 1997).

The ICPP injection well was used from 1952 to February 1984 to dispose of ICPP service wastewaters. ICPP processes generated large volumes of plant cooling waters and condensates that contained small quantities of radioactive and inorganic contaminants. The average discharge to this well was about 363M gal/yr (1.4BL/yr) or about 1M gal/day (3.8ML/day) (Rodriguez et al. 1997).

In June 1970, during an attempt to replace a defective measuring line, it was discovered that collapsed casing had plugged the well at a depth of 226 ft (68.9 m). Therefore, wastewater had been injected into the unsaturated zone above 226 ft (68.9 m). The wastewater discharged to the disposal well was warm (65 to 70o C) and salty (chloride content averaging approximately 200 to 250 mg/L) and apparently corroded the casing until it collapsed, allowing the gravel pack and intruding sediment to fill the well to the 226-ft (68.9-m) depth. Only fragmentary corroded pieces of the original 16-in (41-cm) casing were left, as indicated by caliper logs and first attempts at cleaning the well. Measurements made in 1966 showed that the casing was still intact; therefore, most of the collapse took place in 1967 or early 1968 (Rodriguez et al. 1997).

In September 1970, drilling began to reline the injection well to its original depth. By October, deepening had progressed to about 500 ft (152.4 m) and the water level in the well resumed to its normal depth at about 455 ft (138.7 m). During this time, wastewaters were redirected to USGS-50 (see description below). It is assumed that during this period, the well collapsed again. It was reopened to the water table in 1982. At this time, a 1-in (2.5-cm) diameter high-density polyethylene line was placed in the well from ground level to the bottom of the well and was perforated from 450 ft (137 m) to the bottom of the well (Rodriguez et al. 1997).

On February 7, 1984, the injection well was taken out of routine service and wastewater was pumped from two parallel collection vaults to Percolation Ponds 1 and 2. After this time, the injection well served as an emergency overflow protection for buildings CPP-709, -734, and –797, which contain the vaults from which the service wastewater is monitored and pumped. This overflow protection was only required on a temporary basis if the operating and standby pumps from one of the parallel streams failed simultaneously. Volumes of service waste effluents transferred to the injection well during emergencies were about 7.8×10^7 gal (2.96×10^8 L) in 1984, 49,000 gal (185,490 L) in 1985, and 3,220 gal (12,190 L) in 1986 (Rodriguez et al. 1997).

In 1986, modifications were made to the injection well entry to decrease injections to approximately 3,220 gal (12,200 L). No releases have occurred to the well since 1986. In October and November 1989, the injection well was sealed by perforating the casing and pumping in cement. The well was sealed from 475 ft (145 m) bls to land surface to prevent hydraulic communication between the land surface, perched water, and the aquifer (Rodriguez et al. 1997).

A total of 22,200 Ci of activity (approximately 96% consisting of tritium) are estimated to have been contained in 1.1 × 1,010 gal (4.2 × 1,010L) of water released to CPP-23 during 1950-1984. A summary of the total curies discharged to the injection well for each radionuclide is shown in Table 3-4 (Rodriguez et al. 1997). In addition, a summary of total mass or activity for COCs and contamination sources for the injection well are listed in Table 3-3 of the Conceptual Model Report (Orr 2003). Nearly all of the radionuclides discharged were fission and activation products from the spent fuel processed at the ICPP. Liquid radioactive waste created during reprocessing was treated via evaporation and ion exchange prior to being routed to storage tanks. The condensate generated by the evaporation treatment was discharged to the well. This condensate contained relatively high concentrations of tritium because tritiated water readily evaporates (Frederick et al. 1998). The major radionuclides of concern, based on drinking water standards, are tritium, Sr-90, and Cs-137 (Rodriguez et al. 1997). In addition to radionuclides, chemical wastes were also discharged to the CPP-23. In 1969, the primary chemicals in the wastewater were sodium chloride, sulfuric acid, and sodium hydroxide. Sodium chloride was used for conventional water softeners, and sulfuric acid and sodium hydroxide were used to regenerate ion exchange units that produced demineralized water (Frederick et al. 1998). During the operational life of the well, known discharges include:

1. July 1953: The contents of a tank discharged to the wastewater flowing to the well. A post discharge analysis showed that 51 mCi of radioactive contaminants were released in 244,000 gal (923,640 L) of water.

2. December 1958: About 29 Ci of radioactive contaminants, including 7 Ci of Sr-90 were released to the well.

3. September 1969: Two separate releases resulted in 19 Ci of fission products released to the well. Releases included Cs-137, Cs-134, cerium (Ce)-144, and antimony (Sb)-125 in 3.28×10^6gal (12.4×10^6L) of wastewater.

4. December 1969: Two releases occurred in which the quantity of Sr-90 released was higher than expected. About 1 Ci, including 0.3 Ci Sr-90 was released.

5. March 1981: Mercury was detected during routine monitoring of the ICPP Service Waste System. Mercury in the form of mercuric nitrate released from CPP-601, through the ICPP Service Waste System to the ICPP injection well. An estimated 0.207 mg/L of mercury was detected in service waste (Rodriguez et al. 1997).

Table 3-4. Estimated curies injected to CPP-23.

Contaminant	Estimated Amount Injected[a]
Mercury	4.00E+02 kg
Am-241	1.23E-01 Ci
Co-60	1.24E+00 Ci
Cs-137	2.58E+01 Ci
H-3	2.01E+04 Ci
I-129	1.39E+00 Ci
Np-237	1.07E+00 Ci
Total Pu	8.22E-01 Ci
Sr-90	2.43E+01 Ci
Total U	2.69E-01 Ci

a. The time period over which the estimates are available (from RWMIS) varies by contaminant (information from Table 5-42 in Rodriguez et al. 1997).

USGS-50

USGS-50, located near CPP-23, was used as an injection well during failure of CPP-23. It is assumed that the average discharge to USGS-50 was the same as to CPP-23 of 363M gal/year (1.4BL/yr) or about 1M gal/day (3.8ML/day) and the same low-level radioactive and chemical wastes.

PBF-05

Power Burst Facility (PBF)-05, the PBF Reactor Area Warm Waste Injection Well (PBF-301), is a deep vadose zone injection well constructed in 1969 for the disposal of low-level radioactive waste and raw coolant waste. It is 110 ft (933.5 m) deep. Injections began in 1973 when low-level radioactive waste and raw coolant water were routed to the well. Discharges were limited in activity to 18,800 counts per minute above background. An approximate total of 0.48 Ci were released to the well from low-level radioactive waste and raw coolant water. Radionuclides with contributions greater than 0.01 Ci and half-lives greater than 10 years are Cs-137 at 0.30 Ci and tritium at 0.02 Ci. Raw coolant water was the only effluent from 1981 through 1984. Only Sr-90 was retained for risk analysis and no hazardous waste was discharged to the well. In 1984, discharge pipes to the well were sealed with concrete and the well was capped and sealed with concrete (Holdren et al. 1999).

PBF-15

PBF-15, PBF Reactor Area Corrosive Waste Injection Well (PBF-302), is a vadose zone injection well constructed in 1969. It is 116 ft (35 m) deep. This well received discharge from regeneration of the PBF Reactor demineralizers and from the PBF Reactor secondary coolant system from 1972 through 1978. In 1978, discharges to the well were rerouted to the PBF Evaporation Pond. Historical disposals included an average of 2.9E+05 gal/year (1.1E+06 L/year) of wastewater containing sulfuric acid, sodium hydroxide, chromium, hydrazine, and zinc (Holdren et al. 1999). More detailed information about waste quantities discharged is not available.

EOCR-2

The Experimental Organic-Cooled Reactor (EOCR)-2 injection well was constructed to dispose wastewater and industrial waste from the EOCR buildings. The buildings were completed in 1962, so it is assumed that the injection well was drilled at the same time. The well was approximately 258 ft deep. It was filled as part of a decontamination and decommissioning effort in 1981. The reactor was never used and no evidence exists that hazardous chemicals were injected in this well at any time (Paarmann 1993).

3.1.2 Factors Controlling Release

For the majority of injection wells, release occurred instantaneously from direct injection into the SRPA. Details of injection history are explained for each well in the above inventory section. For a few wells, a secondary source of sludge remains in the well and on the adjacent formation, contributing residual releases to the SRPA. Other wells injected wastewater into the vadose zone, presenting the possibility of source in the vadose zone that could leach into the SRPA. Figure 3-1 shows the injection wells and disposal location, along with other information about factors controlling release.

TSF-05

Initial contaminant release occurred through direct aquifer injection of liquid wastes into the SRPA. Presently the release of these contaminants is associated with long-term dissolution of nonaqueous contaminants and desorption of radionuclides. As a result of injections, a secondary source of organic sludge accumulated in TSF-05 and in the adjacent formation. The presence of this sludge complicates release of contaminants because it appears to trap even aqueous contaminants such as tritium for long periods of time (Sorenson et al. 2000).

Sludge removal efforts took place during January and February 1990 when 43.2 cubic ft (1.22 m^3) sediment and sludge (Bukowski et al. 1998b) was removed from inside the well casing from well depths between 250 and 305 ft (76.2 and 91.4 m) (Kaminski et al. 1994). This did not result in complete removal of the sludge as a video log of TSF-05 taken in 1996 showed dense sludge material occurring 9 ft (2.7 m) above the bottom of the well (Bukowski et al. 1998a). Analysis of the sludge material indicated approximately 3% TCE and 0.3% PCE concentrations in the sludge (Bukowski et al. 1998b), in addition to elevated concentrations of radiological, such as tritium and cesium, and metal constituents (Kaminski et al. 1994).

Contamination from waste injection in TSF-05 has resulted in nearly 1.9-mi (3.0-km) long TCE, PCE, tritium, chloride, and sulfate plumes. Plumes of cis-DCE, trans-DCE, Sr-90, nitrate, and alkalinity are localized around TSF-05 (Wymore et al. 2000). Tritium has never been measured outside of TSF-05 at concentrations greater than the drinking water standard of 20,000 pCi/L, despite concentrations measured in the sludge in 1990 that were almost two orders of magnitude higher. This suggests that tritium is trapped in the sludge pore water along with other contaminants that are subject to sorption within the sludge. It is thought that tritium and other contaminants slowly diffuse out of sludge pore water into the groundwater (Wymore et al. 2000). Present remedial efforts for TSF-05 groundwater remediation combines in situ bioremediation (ISB) for hot spot restoration, pump-and-treat technology for the medial zone, and monitored natural attenuation (MNA) for the distal zone of the plume. In situ bioremediation uses amendment injection to enhance dechlorination of TCE through anaerobic reductive dechlorination. Using pump-and-treat technology in the medial zone, contaminated groundwater is extracted, treated through an air stripper, and the treated groundwater is reinjected. The distal zone remedial strategy is MNA, which allows for the natural reduction of volume, toxicity, and mobility of contaminants without human intervention. Current estimates indicate that natural attenuation will reduce volatile organic compound (VOC) concentration to MCLs by 2095 (DOE et al. 2001).

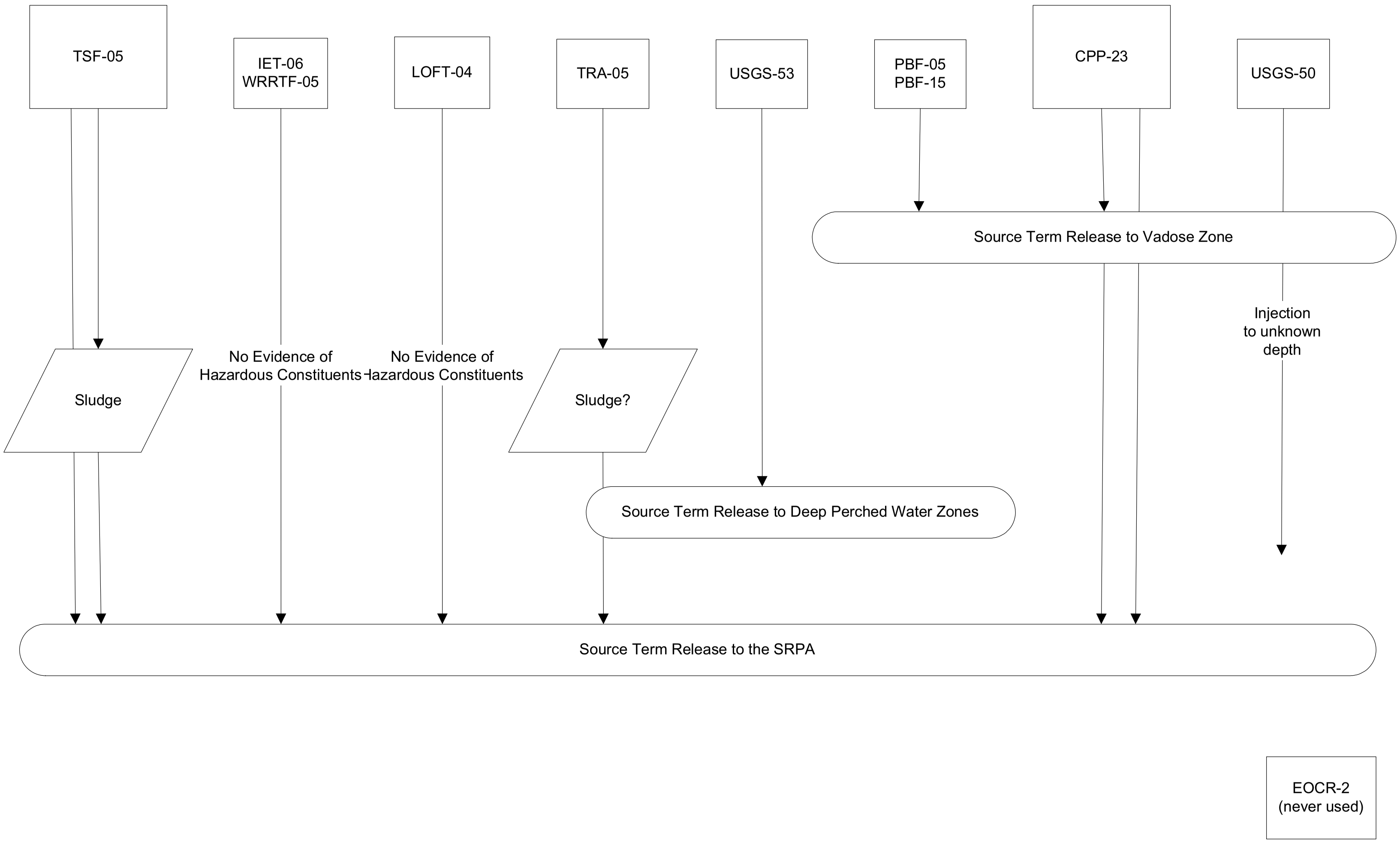


Figure 3-1. Idaho National Engineering and Environmental Laboratory injection wells and disposal location.

IET-06

Release occurred as direct injections to the SRPA. There is no evidence of other sources of contaminants remaining in the well. The well is currently used as a monitoring well. Conclusions from Track 1 investigations stated that the main wastes discharged were acid (sulfuric) and caustic (sodium hydroxide) solutions from regeneration of demineralizer columns, but these wastes are not considered to be a factor warranting concern because the acid and caustic solutions are expected to have largely neutralized each other and any portion not neutralized would have been diluted in the aquifer (INEEL 1994).

WRRTF-05

Release occurred as a direct injection, but the depth of release is unknown. Since there is no evidence of large volumes of concentrated wastes being disposed to the well, then there is no evidence of other sources of contaminants remain in the well.

LOFT-04

Release occurred as a direct injection to the SRPA, but there is no evidence that hazardous constituents ever entered the system.

TRA-05

Historical injections of cooling tower blowdown water resulted in direct release of chromium to the SRPA. During Track 2 evaluation of TRA-05, a concern was raised regarding potential releases of contaminants to the SRPA from possible chemical and radiological sludge. This sludge may have accumulated in the well casing below the perforated interval. Since the well collapsed at 980 ft bgs, a confirmatory sample of the sludge could not be collected during the scoping process. At the depth where the sludge would be present, the aquifer has low transmissivity and GWSCREEN modeling demonstrated that the risk based concentrations of potential COCs are in the acceptable range; therefore, the well does not pose an unacceptable risk even if sludge does exist. In addition to potential contaminants in the well as sludge, contaminants (such as arsenic and chromium) may be present in well sediments, but Track 2 investigations showed there are no risk drivers in the sediments (DOE et al. 1994).

USGS-53

Direct release was in the deep perched water zone; therefore, contaminants could potentially migrate to the SRPA.

CPP-23

This injection well released wastewater and contaminants directly to the SRPA. It also released wastewater and contaminants to the vadose zone during periods of casing collapse. Some of the contaminant inventory introduced to the vadose zone could be released slowly to the aquifer through vertical saturation and unsaturated flow.

USGS-50

This well was used during failure of CPP-23. The depth of injections is unknown.

PBF-05

This was a vadose zone injection well; therefore, injections could have and maybe will migrate into the SRPA.

PBF-15

This was a vadose zone injection well; therefore, injections could have and maybe will migrate into the SRPA.

EOCR-2

This well was never used.

3.1.3 Current Uncertainties

There are many uncertainties associated with injection wells since available information is limited. These uncertainties include inventory, contents of liquids injected, factors for release, and others. Some of these uncertainties have been previously documented and some were identified by the technical advisory group. This section summarizes, to the extent that the information is available, what elements of the conceptual model of release are important (sensitive and uncertain).

Some overall uncertainties that apply to all injection well information include: Does a distinction need to be made between primary and secondary sources? Are contaminants that have migrated from the source still considered a source?

Also, additional research required is discussed in relation to the uncertainties in order to allow more accurate quantification of the source term and the transport of contaminants from the source term to the vadose zone and SRPA.

TSF-05

- Inventory of waste is estimated as ranging from 350 to 35,000 gal of TCE. Records of TCE use at TAN show that 25,670 gal were used from 1955-1972.

- Suspected discharges of heavy metals.

- From 1972 to the early 1980s, the well could have been used for overflow, but no records are available.

- Release of contaminants from the secondary source sludge has been modeled.

- Additional research is being performed through and in support of remedial activities.

IET-06

- Although there is information about the types of waste and waste generators, no information is available to state an inventory.

- Additional research could include groundwater monitoring at IET-06.

WRRTF-05

- Additional research could take place to identify plumes other than chloride and sulfate originating from WRRTF-05.

LOFT-04

- No inventory information is available. No evidence that hazardous constituents ever entered this system.

TRA-05

- Unknown vertical distribution of contaminants since this is a deep well with three areas of perforations.
- Sludge may still be in the well.
- Additional research could be performed to see if sludge is present or if it is contributing contaminants to the SRPA.

USGS-53

- Estimated injection rates used to predict total gallons of liquid injected.
- Located in deep perched water zone, so contaminants may be migrating to SRPA.
- Additional research could be performed to see if migration to the SRPA has taken place.

CPP-23

- The well collapsed and wastewater was injected into the vadose zone. Migration of waste from vadose to the SRPA is unknown. Also, vertical distribution of contaminants is unknown.
- Inventory is estimated based on RWMIS. Early records used in RWMIS may not been accurate.

USGS-50

- Documentation about well construction was not found.
- Injection volumes were estimated.

PBF-05

- Injections were made in the vadose zone, so migration into the SRPA is unknown.

PBF-15

- Injections were made in the vadose zone, so migration into the SRPA is unknown.

EOCR-2

- This well was never put into service, so there are no uncertainties.
- Additional research is not needed.

3.1.4 References

Bukowski, J.M. and K.S. Sorenson, Jr., 1998a, *Site Conceptual Model: 1996 Activities, Data Analysis, and Interpretation Test Area North Operable Unit 1-07B*, INEEL/EXT-97-00556, Rev. 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Bukowski, J.M., H. Bullock, and E.R. Neher, 1998b, *Site Conceptual Model: 1997 Activities, Data Analysis, and Interpretation for Test Area North, Operable Unit 1-07B*, INEEL/EXT-98-00575, Rev. 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Burns, D.E., K.M. Davis, S.C. Flynn, J.F. Keck, N.L. Hampton, A.H. Owen, and R.L. VanHorn, 1997, *Comprehensive Remedial Investigation/Feasibility Study for the Test Reactor Area Operable Unit 2-13 at the INEEL*, DOE-ID-10531, Rev. 0, February 1997.

Dames and Moore, 1992, *Remedial Investigation Report for the Test Reactor Area Perched Water System (Operable Unit 2-12)*, EGG-WM-10002, Rev. 0, June 1992.

DOE, EPA, and IDEQ, 2001, *Record of Decision Amendment, Technical Support Facility Injection Well (TSF-05) and Surrounding Groundwater Contamination (TSF-23) and Miscellaneous No Action Sites, Final Remedial Action INEEL*, Idaho Falls, Idaho, DOE-ID-10139, Rev. 0, September 2001.

DOE, EPA, and IDHW, 1995, *Decision Statement for OU 2-11 Track 2 Summary Report for the Test Reactor Area (TRA) TRA-03, TRA-04, and TRA-05- No Action and No Further Action Determination*, 5859, Rev. 0, January 1995.

Frederick, D.B., A.E. Merrit, M. Fischer, and L.J. Campbell, 1998, "Wastewater Disposal at the Idaho National Engineering and Environmental Laboratory, Idaho," State of Idaho INEEL Oversight Program.

Frindlund, P. S., 1992, "No Action Site Documentation Package- LOFT Injection Well," 5111, Rev. 1, January 1992.

Holdren, K.J., Hiaring, C.M., Burns, D.E., Hampton, N.L., Broomfield, B.J., Neher, E.R., VanHorn, R.L., Stepan, I.E., Wells, R.P., Chambers, R.L., Schmeising, L., and Henry, R., 1999, *Waste Area Group 5 Operable Unit 5-12 Comprehensive Remedial Investigation/Feasibility Study, INEEL, Idaho Falls, Idaho*, DOE-ID-10607, January 1999.

Hull, L.C., October 1989, "Conceptual Model and Description of the Affected Environment for the TRA Warm Waste Pond (Waste Management Unit TRA-03)," EG&G Idaho, Inc., Informal Report EGG-ER-8644, Idaho National Engineering and Environmental Laboratory, EG&G Idaho, Idaho Falls, Idaho.

INEEL, 1994, "Track 1 for the IET Injection Well- No Further Action Determination," 5678, Rev. 0.

Kaminski, J.F., K.N. Keck, A.L. Schafer-Perini, C.F. Hersley, R.P. Smith, G.J. Stormberg, and A.H. Wylie, 1994, *Remedial Investigation Final Report with Addenda for the Test Area North Groundwater Operable Unit 1-07B at the Idaho National Engineering Laboratory*, EGG-ER-10643, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Martian, P., 1999, *Numerical Modeling Support of the Natural Attenuation Field Evaluation for Trichloroethene at the Test Area North, Operable Unit 1-07B, Idaho National Engineering and Environmental Laboratory*, INEEL/EXT-97-01284, Revision 1, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Morris, D.A, J.T. Barraclough, G.H. Chase, W.E. Teasdale, and R.G. Jensen, 1965, *Hydrology of Subsurface Waste Disposal, National Reactor Testing Station, Idaho, Annual Progress Report*, 1964, U.S. Geological Survey, IDO-22047-USG, U.S. Geological Survey.

Orr, B.R., Armstrong, A.T., Bates, D.L., Bukowski, J.M., Sorenson, K.S., and Whitmire, D.L., 2003, *INEEL Subregional Conceptual Model Report Volume1-Summary of Existing Knowledge of Natural and Anthropogenic Influences Governing Subsurface Contaminant Transport in the INEEL Subregion of the Eastern Snake River Plain*, INEEL/EXT-02-00987, Rev. 0, January 2003.

Paarmann, M. L., 1993, "No Action Site for the EOCR-02 Injection Well," 5581, Rev. 0, December 1993.

Rodriguez, R.R., VanHorn, R.L., Schafer-Perini, A.L., McCarthy, J.M., Martian, P., Burns, D.E., Raunig, D.E., Burch, N.A., 1997, *Comprehensive Remedial Investigation Feasibility Study (RI/FS) for ICPP OU 3-13 Part A; Remedial Investigation BRA (RI/BRA) Report*, DOE-ID-10534, Rev. 0, November 1997.

Sorenson, K.S., J.P. Martin, and H. Bullock, 2000, *Field Evaluation Report of Enhanced In Situ Bioremediation, Test Area North, Operable Unit 1-07B*, INEEL/EXT-2000-00258, Rev. 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Wymore, R.A., J.M. Bukowski, and K.S. Sorenson, Jr., 2000, *Site Conceptual Model: 1998 and 1999 Activities, Data Analysis, and Interpretation for Test Area North, Operable Unit 1-07B*, INEEL/EXT-2000-00188, Rev. 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

3.2 Buried Waste

Disposal of wastes by shallow land burial has been an accepted standard practice in industry. Waste burials in general pose a threat to groundwater because the wastes are typically placed directly into the vadose zone, and any contaminant release from the waste is then available for transport in the vadose zone and, potentially, to the underlying aquifer. Wastes have been disposed of through burial at the INEEL since the site was originally established in the early 1950s. The various aspects of buried waste disposal at the INEEL, including types of disposal, inventories of wastes disposed, factors affecting release of contaminants from disposal locations, and uncertainties associated with the disposals, are discussed in the following sections.

3.2.1 Waste Burial Facilities at the Idaho National Engineering and Environmental Laboratory

Burial of wastes can be roughly classified into two broad categories, convenience disposal and managed disposal. Convenience disposal typically occurred during short-term projects, such as construction activities. Managed disposal is typified by the presence of acceptance criteria for the wastes being disposed and active management (e.g., placement, cover) of the disposal area. Managed buried waste disposals were typically performed in planned locations and conducted over longer durations or, for specific projects, over a shorter time frame to meet clean up goals.

3.2.1.1 Convenience Disposal

Examples of convenience disposal, typically small burial pits for disposal of construction debris or other items, have been identified at INTEC (DOE-ID 1997a) and TAN (DOE-ID 1997b). These sites include:

- IET-1 Burial Pit - northeast of IET – this pit was constructed in the 1960s for the disposal and burning of debris from around the IET facility

- LOFT-15 – LOFT Buried Asbestos Pit – a construction materials (most likely concrete, metal, and wood) burn pit used irregularly from as early as 1957 to as late as 1979. Abandoned in 1979, it was covered with 2 – 4 ft of soil and most remaining debris was removed in 1992 and disposed of in the Central Facilities Area (CFA) landfills.

- LOFT-16 – LOFT Landfill - northeast of LOFT-02 drainage pond, this landfill was operational from 1973 to 1980 and was used for disposal of excess construction materials and equipment, such as manual power tools and debris, at the end of LOFT construction. The site was backfilled, compacted, and graded using heavy equipment when it reached capacity.

- CPP-84 – this site contains buried gas cylinders dating from the construction of the Idaho Chemical Processing Plant (ICPP, now know as INTEC) in the 1950s. Based on a review of construction records, these cylinders are believed to contain gases commonly used in construction (i.e., argon, carbon dioxide, nitrogen, oxygen, and acetylene).

- CPP-94 – three buried gas cylinders identified during a survey of INTEC - the contents of these cylinders is unknown, but are thought to contain liquid hydrofluoric acid or hydrofluoric gas based on small metal tags found in proximity to the cylinders.

These sites do not present a threat of potential contamination to the SRPA. The construction debris landfills were never used for disposal of hazardous or radioactive wastes. The gas cylinder disposals may present hazards due to potential asphyxiation, explosion (acetylene), corrosivity (hydrofluoric acid/gas), or energetic release of their contents. However, the contents themselves are of sufficiently low molecular weight and high vapor pressure that, in the event of release, migration and transport will be via the atmosphere and not downward into the vadose zone.

3.2.1.2 Managed Buried Waste Disposal

Managed waste burials are present at several locations at the INEEL. These burial grounds include the Stationary Low Power Reactor (SL-1) Burial Ground, the Boiling Water Reactor Experiment (BORAX)-I Burial Ground, the landfills at the Central Facilities Area (CFA), the Subsurface Disposal Area (SDA), and the Idaho Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Disposal Facility (ICDF).

3.2.1.2.1 SL-1 Burial Ground

The SL-1 Burial Ground is a 600 × 300 ft area containing approximately 95,000 ft^3 of debris from the SL-1 incident in January 1961. Wastes were disposed of in a trench (6 ft wide × 10 to 16 ft deep × 495 ft long) and two pits (Pit 1 [12 ft wide × 10 ft deep × 466 ft long] and Pit 2 [20 ft wide × 10 ft deep × 400 ft long]). Depth of waste burials in the trench was dependent on the depth to basalt under the trench. Following disposal, approximately 2 ft of clean backfill was placed over the trench and pits; additional backfill was mounded over the trench and Pit 2 to facilitate precipitation runoff (Holdren et al. 1995).

3.2.1.2.2 BORAX-I Burial Ground

The BORAX-I Burial Ground is an area approximately 18 × 32 × 11 ft consisting of the buried foundation of the BORAX-I reactor. The final excursion test of the BORAX-I reactor in July 1954, planned to result in the potential partial melting of fuel elements and limited release of fission products, resulted instead in the destruction of the reactor and the spread of contamination over 84,000 ft^2 of ground surrounding the reactor. After cleanup of the area, the top half of the shield tank for the reactor was removed and placed in the bottom half of the shield tank in the reactor foundation along with debris and activated metal scrap. The area was backfilled with clean soil and gravel was mounded over and around the reactor for additional shielding. The burial ground area has had additional soil placed to counter subsidence and been fenced to prevent intrusion. Contaminated soils in the area were removed to the extent possible and the entire area was covered by a minimum of 6 in. of clean gravel.

3.2.1.2.3 CFA Landfills

The CFA landfills include – three landfills ranging in size from 8.25 to 15 acres. The predominant waste types disposed in the landfills were construction, office, and cafeteria solid and liquid nonradioactive waste generated at INEEL facilities including Argonne National Laboratory-West (ANL-W), Auxiliary Reactor Area (ARA), CFA, INTEC, Experimental Breeder Reactor-II, Naval Reactors Facility (NRF), Special Power Excursion Reactor Test, TAN, and TRA. Waste inventory records indicate that the major types of waste disposed of in the landfills include trash sweepings, cafeteria garbage, wood and scrap lumber, masonry, concrete, scrap metal, weeds and grass, dirt and gravel, asphalt, and asbestos. The landfills include the following:

- Landfill I was used from the 1950s to 1984 and occupies an area of 8.25 acres. The landfill consisted of three subunits: 1) the rubble landfill, 2) the western waste trench, and 3) the northern waste trench. The rubble landfill was used by the U.S. Navy from 1942 to 1949 and for Site-wide waste disposal sometime after 1949. This was an area of approximately 5.5 acres, estimated to be 12 to 15 ft deep, and covered with approximately 1 to 5 ft of soil overlain by gravel. The western waste trench was approximately 2 acres in area consisting of smaller waste trenches, excavated 8 ft wide × 10 ft deep and 50 ft long and separated from each other by 15 ft of clean soil. Filled trenches were covered with 1 to 5 ft of clean soil. The northern waste trench covered approximately 0.75 acres. Information regarding the actual dimensions of this trench is limited.

- Landfill II covered an area of approximately 15 acres and was located in the corner of an abandoned gravel pit. This landfill received waste from 1970 until 1982. Wastes were disposed to an estimated depth of 12 to 28 ft and covered with 0.33 to 3.2 ft of soil.

- Landfill III consisted of six trenches covering approximately 12 acres and operated from 1982 until 1984. Wastes were disposed of in layers averaging approximately 13 ft deep. A soil cover ranging in thickness from 1 to 8 ft was placed after closure (DOE-ID 2002b).

Potentially hazardous wastes, including waste oil, solvents, chemicals, and paint, may have also been disposed of in the CFA landfills. Records indicate that wastes were not segregated by material type within the landfills. From 1970 to 1984, during operation of Landfills II and III, procedures were in place to screen wastes for radiological contamination prior to disposal. To prevent inadvertent disposal of radioactive waste, wastes were screened by radiological control technicians at the generating facility to determine if the waste was above background levels. Those wastes determined to be above background were prohibited from disposal at the landfills; however, it is generally acknowledged that up to one shipment per month of low-level waste may have been disposed accidentally in the landfills. In 1989, screening of waste for radioactivity was begun on a full-time basis (DOE-ID 2002b).

3.2.1.2.4 Subsurface Disposal Area

The SDA at the RWMC is an area of approximately 98 acres. Waste was disposed of at the SDA beginning in 1952 and continues at present. The SDA serves as the primary, centralized facility for burial of waste at the INEEL, and was established shortly after the INEEL was established in the early 1950s. Disposal units within the SDA include the following:

- Pits – unlined pits excavated to basalt and backfilled with 2 to 5 ft of soil to provide a level surface for waste placement.

- Trenches – narrow excavations averaging 7 ft wide by approximately 10 ft deep and up to 1,800 ft long.

- Soil vaults – unlined, vertical cylindrical borings designed for the disposal of high radiation level waste. Soil vaults were drilled in rows and ranged from 1.3 to 6.5 ft in diameter and averaged 12 ft deep.

- Acid Pit – a pit approximately 197 × 104 ft excavated to basalt and backfilled with approximately 1 to 2 ft of soil. Used to dispose of liquid wastes which were dumped directly into the pit; lime was occasionally added to the pit to neutralize acids.

- Pad A – an aboveground asphalt pad approximately 240 × 335 ft stacked with waste boxes and drums covered by plywood and plastic and with a soil embankment cover.

Table 3-5 presents a summary description and dates of use of the disposal units within the SDA.

Table 3-5. Disposal units at the Subsurface Disposal Area (DOE-ID 2002a).

Disposal Unit	Dates of Use		Volume (m^3)[a]
	Open	Closed	
Pits			
1	11/1/57	10/1/59	9,800
2	10/1/59	7/1/63	31,000
3	12/15/61	1/3/63	16,000
4	1/3/63	9/26/67	45,000
5	6/18/63	12/22/66	42,000
6	5/18/67	10/22/68	22,000
7	9/19/66	10/5/68	11
8	3/6/67	11/30/69	12,000
9	11/8/67	6/9/69	18,000
10	6/7/68	7/8/71	45,000
11	4/14/70	10/16/70	9,800
12	7/2/70	9/12/72	12,000
13	7/20/71	7/29/74	7,600
14	7/1/74	3/31/76	16,000
15	6/25/75	7/3/84	31,000
16	5/22/78	10/25/84	8,700
17	5/5/84	open	
18	3/1/88	open	
19		open	
20		open	
Acid Pit	1/1/54	1/1/61	8,400
Trenches			
1	7/8/52	10/1/54	3,100
2	10/1/54	12/21/54	3,100
3	12/22/54	4/22/55	3,100
4	4/22/55	11/21/55	3,400
5	11/4/55	3/29/56	3,100
6	3/22/56	9/4/56	3,100
7	8/14/56	12/20/56	3,100
8	12/13/56	5/7/57	3,100

Disposal Unit	Dates of Use		Volume (m³)[a]
	Open	Closed	
9	1/17/57	9/6/57	3,400
10	7/19/57	2/7/58	3,100
11	2/11/58	7/25/58	2,500
12	1/3/58	1/16/59	5,000
13	1/9/58	4/24/59	2,100
14	4/16/59	7/30/59	4,200
15	7/31/59	10/16/59	2,200
16	10/17/59	4/12/60	4,200
17	11/1/59	7/1/60	1,700
18	5/10/60	7/20/60	2,800
19	7/5/60	11/29/60	3,900
20	12/1/60	6/30/61	2,800
21	12/13/60	1/10/61	1,000
22	2/1/61	4/25/61	1,100
23	6/20/61	9/15/61	1,200
24	10/1/61	7/31/62	1,200
25	8/1/61	7/27/62	2,800
26	4/13/62	8/17/62	1,200
27	8/20/62	1/4/63	2,800
28	12/26/62	3/12/63	1,200
29	11/19/62	3/20/63	950
30	3/2/63	9/12/63	2,800
31	3/25/63	11/22/63	1,200
32	4/1/63	11/18/63	980
33	10/11/63	8/11/64	2,800
34	3/18/64	8/27/64	2,800
35	8/28/64	1/19/65	2,800
36	12/1/64	7/24/65	3,400
37	12/24/64	7/1/65	2,800
38	5/15/65	9/16/65	2,500
39	7/20/65	11/5/65	2,800

Disposal Unit	Dates of Use		Volume (m^3)[a]
	Open	Closed	
40	10/7/65	1/13/66	2,800
41	1/4/66	10/4/66	2,800
42	5/9/66	1/17/67	3,100
43	10/20/66	6/1/67	2,700
44	1/13/67	3/24/67	1,400
45	2/28/67	9/27/67	3,100
46	9/25/67	3/14/68	2,700
47	2/28/68	8/5/68	3,100
48	8/8/68	5/2/69	2,700
49	11/18/68	6/30/69	3,100
50	7/1/69	11/1/69	2,600
51	10/30/69	4/8/70	3,100
52	3/4/70	7/4/70	2,500
53	7/1/70	10/12/70	3,100
54	9/23/70	5/4/71	2,500
55	4/7/71	3/12/82	3,100
56	12/29/71	2/1/73	3,100
57	12/28/72	6/11/74	2,500
58	2/20/74	8/1/81	2,500
Soil vault rows			
1	3/3/77	9/7/77	3.3
2	3/16/77	10/28/77	0.57
3	3/14/77	10/27/77	0.85
4	3/17/77	11/1/77	0.65
5	3/15/77	9/27/77	0.42
6	3/25/78	2/27/81	7.1
7	5/8/78	1/23/79	70
8	2/12/78	9/8/81	14
9	2/6/79	7/15/81	170
10	6/9/81	11/3/82	35
11	8/7/81	12/12/84	9.3

Disposal Unit	Dates of Use		Volume (m^3)[a]
	Open	Closed	
12	5/5/82	7/27/83	25
13	4/4/83	12/21/84	110
14	10/16/84	5/8/89	28
15	10/16/84	3/17/87	180
16	10/16/84	2/1/90	180
17	4/24/87	3/1993	14
18	2/12/87	10/1993	23
19	1/22/90	8/28/91	250
20	8/29/91	8/1995	270
21	11/2/93	2/3/94	0.3
Storage Pads			
Pad A	9/26/72	11/17/78	

a. Including overburden.

Presently, the SDA is used for disposal of low-level waste (LLW) only. Historically, wastes disposed at the SDA have included radioactive (LLW and transuranic [TRU]), hazardous, and mixed wastes. Wastes disposed at the SDA were received from a large number of on-site and off-site generators, including other Department of Energy (DOE) facilities, universities, research laboratories, and commercial operations (primarily during the period from 1960 to 1963 before commercial radioactive waste disposal facilities were available). In particular, large amounts of TRU waste were received from the Rocky Flats Plant (RFP) (currently known as the Rocky Flats Environmental Technology Site) in Golden, Colorado and are buried in the SDA. Table 3-6 presents a listing of the principle offsite waste generators (excluding the RFP) that disposed of waste at the SDA.

3.2.1.2.5 Idaho CERCLA Disposal Facility

The ICDF is presently being constructed near the INTEC facility and is intended for the disposal of contaminated soils and debris generated during remediation activities at the various CERCLA sites located across the INEEL. The design for this disposal facility includes two lined disposal cells for disposal of contaminated soil and an evaporation pond for disposal of aqueous wastes. To eliminate the potential for migration of wastes away from the facility, wastes will be treated before disposal. These treatment techniques include microencapsulation of debris wastes and stabilization of soils with Portland cement (INEEL 2003). The ICDF is scheduled to open in 2003 and has not yet accepted any waste.

Table 3-6. Principle offsite generators disposing of waste at the Subsurface Disposal Area.

Generator	Dates	Estimated Amount (m^3)[a]
U.S. Bureau of Mines Reno, Nevada	1961	0.42
U.S. Bureau of Mines Albany, Oregon[a]	1960 – 1962	15.85
U.S. Naval Radiological Defense Laboratory San Francisco, California[a]	1963	8.04
Laboratory of Nuclear Medicine and Radiological Biology University of California Los Angeles, California[a]	1961	4.11
Radiobiology, Department of Anatomy University of Utah Salt Lake City, Utah[a]	1960 – 1962	20.11
U.S. Marine Corps Logistics Base Barstow, California[a]	1962	13.42
U.S. Army Chemical Center Dugway Proving Ground Dugway, Utah[a]	1961 – 1962	25.88
U.S. Army Chemical Center Mt. Ranier Depot Ft. Lewis, Washington[a]	1961	1.05
Babcock & Wilcox Lynchburg, Virginia[a]	1960	15.63
U.S. Air Force PM-1 Nuclear Power Plant Sundance, Wyoming[a]	1969	49.4
Colorado School of Mines Research Institute Golden, Colorado[a]	1960 – 1962	0.42
Teledyne Wah Chang Corporation Albany, Oregon[a]	1961	4.53
Chicago Bridge and Iron Co. Seattle, Washington[a]	1962 – 1963	0.26
Nuclear Regulatory Commission – Region V Walnut Creek, California[a]	1961	791.42
University of California Lawrence Berkeley Radiation Laboratory Berkeley, California[a]	1962 – 1963	262.71

Generator	Dates	Estimated Amount (m^3)[a]
University of California Lawrence Livermore Radiation Laboratory Livermore, California[a]	1962 – 1963	1978.77
Rockwell International Atomics International Division Canoga Park, California[a]	1960 – 1962, 1966, 1968	1392.3
General Atomic Company San Diego, California[a]	1961 – 1963	714.82
Coors Porcelain Company Golden, Colorado[a]	1964	5.45
IT Corporation Wilmington, California[a]	1962 – 1963	28.52
American Electronics, Inc. Fullerton, California[a]	1961	10.42
Oregon Metallurgical Corporation Albany, Oregon[a]	1961	7.33
San Antonio Air Logistics Center Kelly Air Force Base, Texas[a]	1960 – 1961	83.92
Radiological Safety Division University of Washington Seattle, Washington[a]	1960 – 1963	12.97
Washington State University Pullman, Washington[a]	1962	1.67
Thiokol Corporation Wasatch Division Brigham City, Utah[a]	1960	2.12
Nuclear Engineering Co., Inc. Louisville, Kentucky[a]	1961 – 1962	163.9
International Chemical and Nuclear Irvine, California[a]	1961 – 1963	79.96
Radiological Health, Public Health Service Department of Health, Education, and Welfare Rockville, Maryland[a]	1961	94.46
Memorial Hospital of Sheridan County Sheridan, Wyoming[a]	1963	0.14
Atlas Foundry and Machine Co. Tacoma, Washington[b]	NA	NA

Generator	Dates	Estimated Amount (m^3)[a]
Birdwell Division Seismograph Services Corporation Tulsa, Oklahoma[b]	NA	NA
California Salvage Co. San Pedro, California[b]	NA	NA
Fort Douglas Salt Lake City, Utah[b]	NA	NA
General Dynamics Fort Worth, Texas[b]	NA	NA
General Electric Co. Vallecitos Atomic Laboratory Pleasanton, California[b]	NA	NA
Isotope Specialties Co. Burbank, California[b]	NA	NA
Industrial X-Ray Engineers Seattle, Washington[b]	NA	NA
Metallurgical Engineers, Inc. Portland, Oregon[b]	NA	NA
Nuclear Engineering Co. Pleasanton, California[b]	NA	NA
Nuclear Power Field Office Fort Belvoir, Virginia[b]	NA	NA
Sacramento Signal Depot Sacramento, California[b]	NA	NA
U.S. Army Chemical Center Maryland[b]	NA	NA
U.S. Army Edgewood Arsenal Maryland[b]	NA	NA
USARAL Support Command and Fort Richardson Seattle, Washington[b]	NA	NA
U.S. Nuclear Corporation Burbank, California[b]	NA	NA

a. Information from Clements (1980)

b. Information from LITCO (1995a)

3.2.2 Inventory of Buried Waste

The inventory of contaminants present in buried waste and potentially available for transport is dependent on the amount of contaminant initially disposed and any processes that have occurred to reduce or change that amount. In the case of buried nonradionuclides, these processes include migration of contaminants from the burial site into the vadose zone (in the case of metals or VOCs that have vapor densities heavier than air) or to the atmosphere (for those VOCs having vapor densities lighter than air), depending, of course, on the depth of original disposal. In the case of radionuclides, radioactive decay over the span of time from initial disposal to present will determine not only the amount of the initial radionuclide present, but also the presence of additional radionuclides as daughter products of this decay. Additionally, the source term may be reduced by remediation activities that may have been conducted between the initial disposal and present. The following subsections describe the contaminant inventories in buried wastes at the INEEL, and modifications to these inventories since their initial emplacement.

3.2.2.1 SL-1 Burial Ground Inventory

3.2.2.1.1 SL-1 Inventory as Disposed

The nature of the SL-1 incident and waste forms present preclude direct measurement of the inventory of contaminants buried in the SL-1 Burial Ground. The primary COCs buried at the SL-1 Burial Ground are radionuclides originating from the fuel present in the reactor at the time of the incident and as products of decay of that fuel over the intervening years. The inventory of these radionuclides present in the waste was calculated using the ORIGEN2 computer code.

The ORIGEN2 code calculates radionuclides in three classes as produced in operating reactor cores and associated hardware: fission products generated by the fission of U-235 atoms, activation products arising from neutron absorption in structural materials, and actinides produced by transmutation of U-238 in the fuel. Ingrowth products are also calculated. For this analysis, activation products were not calculated for the following reasons (Holdren et al. 1995):

- Due to the reactor's low operating history, insignificant quantities of activation products would have been produced relative to fission products

- Uncertainties in predicting the fate of all of the radionuclides is greater than the error introduced by not considering the quantity of activation products

- Activation products are generally not mobile in the environment (and consequently less of a threat to groundwater) because they are integral parts of the structural metals.

Table A-1, Appendix A, presents estimated activities and concentrations of each radionuclide in the buried SL-1 debris as calculated by the ORIGEN2 code. The concentration is based on a source volume of 99,000 ft^3 (2,800 m^3) and an assumed density of 1.5 g/cm^3 (Holdren et al. 1995).

Nonradionuclide contaminants present in the trench and pits at the site includes approximately 9,000 lb of lead sheet and 2,000 lb of miscellaneous lead items. Organic hazardous contaminants are not known to have been buried with the debris from the incident.

3.2.2.1.2 Modifications to the SL-1 Inventory

Although the surface soils have been remediated, no remediation of the buried wastes has occurred. Thus, the only modifications made to the waste inventory estimates have been through radioactive decay. This is reflected in Table 3-5.

3.2.2.2 BORAX-I Burial Ground Inventory

3.2.2.2.1 BORAX-I Inventory as Disposed

As with the SL-1 incident, the destructive nature of the final excursion of the BORAX-I reactor presents certain challenges in estimating the inventory of contaminants disposed of in the buried waste. At the time of the test, the reactor core contained an estimated 4.2 kg of U-235, of which approximately 478.6 ± 27.1 g was recovered in the subsequent clean up operations. The remaining 3.7 kg (approximately 88% of the original fuel inventory) of U-235 was either buried with the remaining waste or scattered in the 84,000 ft^2 of soil contaminated by the excursion. Based on contamination surveys and historical operating information for the reactor, the Radiological Safety Analysis Computer System Program (RSAC-5) was selected to estimate the fission product inventory for the BORAX-I reactor.

The RSAC-5 code estimates fission product inventory, but does not identify activation products or actinides. An Engineering Design File included as part of Holdren et al. (1995) concluded that the production of actinides by the BORAX-I reactor was insignificant relative to the activities of fission products. In addition, activation products were not considered in Holdren et al. (1995) because of the short operational life of the reactor (generation of sufficient quantities of activation products to influence the analysis requires much greater burnup and a longer operational life than that experienced by BORAX-I.) The RSAC-5 code was selected because it is applicable for reactor cores that have a U-235 enrichment greater than 25% and less than 60% burnup; BORAX-I had an enrichment of 95% and a burnup of less that 1%.

To perform the estimates, the inventory of uranium isotopes in the fuel is added to the fission product inventory to obtain the total radionuclide inventory for the RSAC-5 code. Activities of U-234, U-235, and U-238 were calculated using a technique developed for the INEEL Historical Dose Evaluation, and are based on the fuel parameters for BORAX-I. Results of the RSAC-5 analysis provide a list of radionuclides and level of radioactivity that may be present after various periods of time. These estimates are summarized in Table A-2, together with a concentration calculated by dividing the curie content by the volume of waste buried (6,336 ft^3 [180 m^3]) and an assumed density of 1.5 g/cm^3.

Organic contamination of the BORAX-I buried waste is not suspected. An unknown quantity of lead shielding from the reactor may be present in the buried waste.

3.2.2.2.2 Modifications to the BORAX-I Inventory

Remediation activities at the BORAX-I site have been limited to the initial collection of scattered fuel and reactor fragments following the final excursion and soil remediation efforts in the contaminated area. As with the SL-1 Burial Ground, the only modifications to the buried source term have been radioactive decay of the nuclides since the time of burial. This is reflected in Table 3-6.

3.2.2.3 SDA Inventory

3.2.2.3.1 SDA Inventory as Disposed

Wastes received at the SDA are highly heterogeneous with regards to waste type (e.g., radioactive, hazardous), generator, and disposal location. This has resulted in challenges in determining the types and amounts of waste constituents present in the waste, as sampling alone is insufficient to characterize the waste. Consequently, several efforts have been undertaken to determine the source term inventory represented by the SDA. The efforts were similar, but because the waste is regulated differently, the inventory effort was broken into two time periods, 1952 through 1983 and 1983 through present. The two major inventory efforts were the Historic Data Task (HDT) (DOE-ID 1995a) and the Recent and Projected Data Task (RPDT) (DOE-ID 1995b). The HDT looked at waste from 1952 (the beginning of operations) through 1983 (DOE-ID 1995a). This is the inactive portion of the landfill. The RPDT looked at waste currently regulated under DOE Order 435.1 for LLW disposal. This is the waste from 1984 to current disposals (DOE-ID 1995b). Over the period of operations, the disposal practices and SDA waste acceptance criteria have changed. These changes include:

- TRU waste burials ended in 1970
- Hazardous chemical burial ended in 1983
- Soil vault disposals ended in 1993
- Remote-handled LLW waste disposal is now limited to concrete vaults.

To compile an inventory of waste constituents disposed during the period from 1952 to 1983, the HDT was implemented. The intent of this task was to develop a comprehensive inventory by waste generator for wastes disposed during this period. Information for this effort was collected from the following sources (LITCO 1995a):

- Process knowledge and plant operating records
- Inventory and other technical reports
- Engineering and nuclear physics calculations
- Shipping and disposal records, particularly the RWMIS
- Interviews with plant operations personnel
- Other sources.

Data collected in this effort were entered onto review forms, technically reviewed, and entered into a database. Information collected in this database, referred to as the Contaminant Inventory Database for Risk Assessment (CIDRA), include the types of buried contaminants, amounts of individual contaminants contained in each waste stream, dates of waste stream disposal, waste stream generator, physical and chemical forms of the contaminants, and types of containers used for the disposal. Information entered into CIDRA included all of the disposal locations at the SDA, including the pits, trenches, and soil vaults; the Acid Pit; and Pad A (LITCO 1995a).

Since the development of the comprehensive inventory in the combined HDT and RPDT, some issues were uncovered with the inventory. The large amounts of activation products from one facility and the lack of activation products and actinides from other facilities prompted an inventory review. Also previous modeling showed an issue with the amount of VOCs disposed. In addition, in the years following the RDPT report, additional waste has been disposed so an update to summarize the additional disposals has been developed. The best estimate of the inventory is documented in the Ancillary Basis for Risk Assessment (ABRA) (DOE-ID 2002a). It is noted in this document that there are ongoing efforts to further refine the inventory from several key INEEL generators.

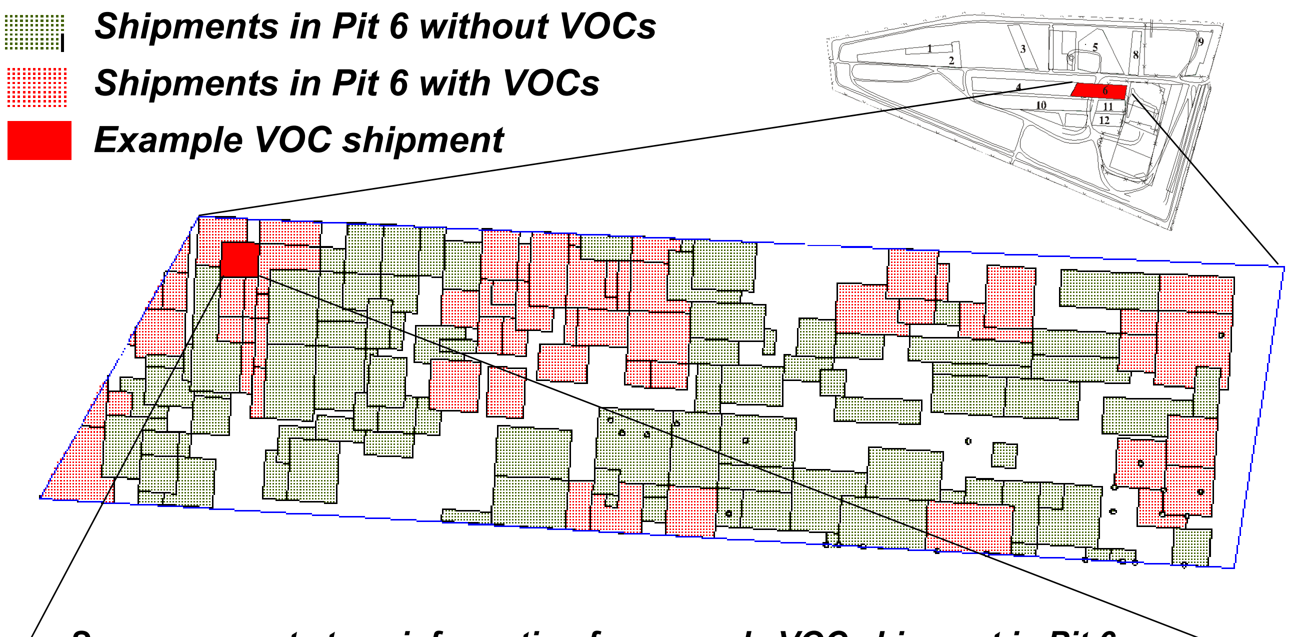
Because of uncertainties in the amount of organic compounds (i.e., carbon tetrachloride, 1,1,1-trichloroethane, trichloroethylene, and methylene chloride) in the disposal records and databases used, an effort was undertaken to refine the estimate of carbon tetrachloride disposed of in the SDA (Miller and Varvel 2001). This effort was based on additional information received from RFP in the form of waste disposal sheets that, combined with information from shipping manifests, allowed the separation of generic RFP waste into their respective waste codes. Additionally, the Organic Waste Treatment Process Logbook (also referred to as the Grease Plant Logbook) was received from RFP and provided information (e.g., individual drum serial numbers, individual drum weights, and other information) regarding the production of 743-series sludge (solidified organics, which contained most of the VOCs received from RFP) drums (Miller and Varvel 2001). An extension of this effort led to revisions in the estimates of tetrachloroethylene, 1,1,1-trichloroethane, and trichloroethylene buried in the pits and trenches at the SDA (Varvel 2001).

Estimated amounts of the non-radionuclide waste constituents disposed in the SDA are provided in Table A-3. This table reflects the current knowledge of the inventory as documented in Holdren (2002). These estimates are based on the efforts performed under the HDT and the RPDT, as modified by Miller and Varvel (2001) and Varvel (2001) and using the interim results of the ongoing INEEL-generated inventory reviews. The ABRA (DOE-ID 2002a) screened the list of non-radionuclide and radionuclide contaminants based on estimated human health risks. The resulting list of COCs includes 12 radionuclides and 4 non-radionuclides, as presented in Table 3-7. Estimated yearly inventories of radionuclide COCs are given in Table A-4. Table A-5 presents estimates of radionuclides and non-radionuclides located on Pad A.

Table 3-7. Contaminants of concern for the Subsurface Disposal Area (DOE-ID 2002a).

Radionuclides			Non-Radionuclides
Am-241	Pu-238	U-233	Carbon tetrachloride
C-14	Pu-239	U-234	Methylene chloride
I-129	Pu-240	U-235	Nitrate
Nb-94	Sr-90	U-236	Tetrachloroethylene
Np-237	Tc-99	U-238	

To complement the inventory information available in CIDRA, spatial information regarding the disposal locations of waste types and items is available through the WasteOScope system. WasteOScope is a customized ArcView Geopgraphic Information System that provides a method for merging physical spatial data and historical disposal information to visually represent the information on a computer or as hard copy graphics. The system was developed by combining historical disposal information (e.g., manifest shipping information, trailer load lists as received at the RWMC, types and number of waste containers, type of waste, radiological data, waste origin) and geophysical data (pit and trench boundaries from survey data, topographic, soil-vapor, surface cover elevations, basalt bedrock contours, aerial photographs). The result is a system that can provide users with detailed information and maps of the burial locations of particular waste types in the pits and trenches (DOE-ID 2002a). An example of the information that can be provided by WasteOScope is presented in Figure 3-2.



Summary waste type information for example VOC shipment in Pit 6

Rocky Flats Facility Code	Description of waste	Rocky Flats Plant Waste ID	Container Count	Total Weight (lbs)	Container Type
71	Wet and Dry Combustibles	RFO-DOW-4H & 9/10H	18	1,985	55-gal
743	Organic sludge	RFO-DOW-15H	35	19,269	55-gal
76	Combustibles	RFO-DOW-9/10H	5	433	55-gal
745	Nitrate salts	RFO-DOW-17H	10	5,505	55-gal
742	2nd stage wastewater sludge	RFO-DOW-3H	3	1,653	55-gal
741	1st stage wastewater sludge	RFO-DOW-13H	4	2,202	55-gal
746	Empty drums	RFO-DOW-4H	54	3,711	55-gal

Figure 3-2. WasteOScope presentation of buried waste information (example).

3.2.2.3.2 Modifications to the SDA Inventory

Over the years of operation at the SDA, the source term has been reduced through various efforts, including retrieval of waste containers and recovery of VOCs from soil gas.

During 1969 and the 1970s, five waste retrieval efforts were conducted. These ranged in size from the attempted retrieval of selected waste items to large scale retrieval of drums to demonstrate the potential for safe retrieval, packaging, and transfer of containers. These efforts were:

- **Pit 1 Retrieval** – Conducted in 1969, this effort was an attempt to recover some experimental equipment that may have been inadvertently sent to the landfill and disposed of in Pit 1. This effort was not well documented, and was apparently unsuccessful in locating the equipment of concern. Undamaged drums, which had been buried for approximately 10 to 12 years, were found to be in good condition in this effort, but boxes and similar waste containers were found to be badly deteriorated. The exact number of drums retrieved is unknown, but is estimated at around 25. The eventual disposition of the retrieved containers is not known (DOE-ID 2002a).

- **Solid Radioactive Waste Retrieval Test** – In 1971, the Atomic Energy Commission (AEC) requested a series of solid waste retrieval tests on the SDA to gain insight into the challenges of conducting a large-scale retrieval effort. The test involved excavating and retrieving waste containers from specific sites within the SDA to evaluate container and waste condition. In addition, AEC requested the retrieval of three specific drums suspected of containing excessive amounts of plutonium. Retrieval operations focused on Pits 2, 5, 10, and 11 (Thompson 1972). Specific details regarding the operations are:

    - Pit 2 – During excavation, drums were found along the edge of the pit and outside of pit boundary markers. The location of these containers was attributed to movement from their disposal location during earlier flooding (the SDA had been flooded in 1962 and 1969). Six of the stacked drums in this pit were exhumed and sent to the ARA-I Hot Cell for examination. These drums had been buried for approximately 12 years and were in the poorest condition of any encountered during the test. Some drums were corroded through and were leaking sludge. Some drums were open and were found not to contain plastic liner bags that were supposed to be in use at the time the waste was packaged. Soil cover over the drums ranged in depth from 12 to 18 in. Plutonium concentrations in the soil ranged from 1E-01 to 1E-03 nCi/g of soil, making retrieval operations difficult due to the need for respirators and frequent changes of protective clothing.Pit 5 – Initial attempts to excavate drums from this pit were unsuccessful due to the waste being located approximately 20 ft inside the pit boundary. Drums in this pit were randomly dumped and had large amounts of soil interspersed among the containers. Five drums, which had been buried for approximately 7 years, were removed from the pit and sent to the ARA-I Hot Cell for examination. A total of 16 drums were exhumed from the pit, 10 of these drums could not be specifically identified, and one-fourth of the drums were damaged or lidless. During excavation, several filters from RFP were found, as were the remnants of cardboard cartons that had contained filters. A soil sample in the area of the filters showed plutonium concentrations of 1.14 nCi/g. A wooden box was also excavated and found to be deteriorated to the point that the report concluded that “future excavations in dumped pits where wooden boxes are involved probably will require special attention and the contents of the wooden boxes removed piece-by-piece.” (Thompson 1972). Plutonium concentrations in the area of the wooden box ranged from 5.58E-04 to 4.51E-02 nCi/g. Overall concentrations of plutonium found in Pit 5 were low, approximately 1E-02 to 1E-03 nCi/g.

- Pit 10 – As with Pit 5, drums in this pit had been randomly dumped, which complicated excavation. The pit was opened in an attempt to recover a specific drum identified by the AEC. Although disposal records indicated that the drum had been disposed of in Pit 10, neither the drum nor any other containers from the shipment were found in Pit 10. Owing to the random placement of drums in the pit, many seriously damaged drums were found. Some of these drums were open and had the inner liners punctured. In addition, loose materials were found in the pit. Of the 24 drums exhumed, approximately one-half were without identification and about a quarter of the drums were lidless or crushed. A soil sample taken from excavated soil showed a plutonium concentration of 2.28E-01 nCi/g, compared with a pre-excavation soil sample of 6.22E-04 nCi/g. All of the containers were physically damaged by the heavy equipment used to level and cover the drums during disposal. Although corrosion of the containers was found, this did not appear to be a significant factor affecting drum condition.

- Pit 11 – This pit was entered in an attempt to retrieve two of the drums requested by AEC. One of these drums was found near the location identified in the disposal records, but the other drum could not be found. The waste in this pit was apparently stacked during disposal and had been in place approximately 18 months before being retrieved. Five drums were selected at random and removed for examination in the ARA-I Hot Cell. Plutonium concentrations in the soils between and under the drums were 2E-03 nCi/g or less.

- **Initial Drum Retrieval** – This project was conducted from July 1974 through June 1978 to demonstrate the safe retrieval, packaging, and transfer to interim storage of drums containing TRU waste buried at the SDA from 1968 to 1970. The project was successful, and resulted in the retrieval of a total of 20,262 drums (corresponding to a waste volume of 4,397 m^3) from Pits 11 and 12. Drums in these pits had been stacked and, because they had not been spread or compacted by heavy equipment, experienced very little damage during disposal. After retrieval and repackaging, 18,029 drums (including 1,015 drums of 743-Series sludge [organic setups]) of TRU waste were placed on the Transuranic Storage Area-Retrievable Storage; 2,232 drums of evaporator salts (Series 745 sludge, primarily sodium nitrate and potassium nitrate) and depleted uranium were placed on Pad A. A single drum containing Co-60 and Cs-137 sources was disposed in Pit 15 (McKinley and McKinney 1978).

  During this effort, most drums were found to be in good condition. External alpha contamination up to 120,000 cpm was found on 1,236 drums removed from Pit 11; drums from Pit 12 had no external contamination other than some fixed contamination. In addition, 486 recovered drums were breached and 162 of these drums leaked free liquids. This liquid was usually uncontaminated, but contamination levels up to 40,000 cpm were found in some liquids.

- **Early Waste Retrieval** – This project was conducted from May 1976 to 1978 to investigate the problems associated with retrieving wastes from belowground burial. During this effort, wastes were removed from Pits 1 and 2 and Trenches 1, 5, 7, 8, 9, and 10. Retrieval operations were conducted in an Operating Area Confinement (OAC) building located within an Air Support Weather Shield (ASWS) to provide confinement for potential contamination and protection from weather, respectively.

  Efforts in 1976 began west of the southeast corner of Pit 2 in May and ended in September. Eighty drums of waste and 25 m^3 of loose waste and soil were retrieved. Five drums were damaged or missing lids and 10 drums leaked liquids when removed from the pit. Of the leaking drums, four contained contaminated liquids, but only one incident of soil contamination resulted from the leakage. After relocating the OAC within the ASWS, 12 additional drums were retrieved. Two of the drums leaked a clear liquid that was contaminated with Pu-238 to 2E+06 cpm; these drums were bagged and placed into overpack containers.

In 1977, wastes were retrieved from Pit 2 and Trenches 8 and 10 that consisted of drums, loose waste, and contaminated soil. This waste included 240 drums (representing a total of 50 m^3 of waste), 21.2 m^3 of loose waste, 6.4 m^3 of contaminated soil, and 12.9 m^3 of waste generated by the retrieval operation (i.e., bubble suits, decontamination materials, and anti-contamination clothing). Seventy percent of the drums retrieved from Pit 2 and Trench 10 and all of the drums from Trench 8 were severely breached, with free liquid leaking from about 9% of the drums; alpha-contamination levels greater than 2E+06 were frequently encountered (Bishoff and Hudson 1979).

Also in 1977, near surface explorations were conducted of Pit 1 and Trenches 1, 5, 7, and 9 prior to moving the ASWS and OAC over these areas. This exploration was conducted by digging a trench perpendicular to the trenches down to the top of the buried waste. Drums in Pit 1 were stacked and were found to be in good condition. Numerous filters and frames were found mixed with the drums. The wooden frames were deteriorated and filters were found to have contamination ranging from 2,000 to 2E+06 cpm. Containers in Trench 1 appeared to be almost completely disintegrated. Waste retrieved included glass vials, metal filings, tubing, and gloves. Drums in Trench 5 had been randomly dumped and exhibited poor integrity; contents appeared to be graphite molds and filters. Few drums were found in Trench 7, with the majority of the waste consisting of uncontained sample bottles, tubing, plastic, and several lengths of pipe. Drums in Trench 9 had been randomly dumped but appeared to be in good condition. Loose items, including tubing, bottles, gloves, and a small cart, were mixed with the drums.

During 1978, operations retrieved 137 55-gal drums, of which approximately 65% were breached. Several drums fell apart during retrieval. Approximately 3% of the retrieved drums leaked fee liquid having alpha contamination ranging from about 2,000 to 60,000 cpm. Many drums were also externally contaminated, having fixed alpha contamination that ranged from 2,000 to greater than 2E+06 cpm. Loose waste, including glass vials and metal cylinders, contained in deteriorated wooden boxes was also retrieved (Bishoff and Hudson 1979).

- **Pad A Penetration and Inspection** – An effort to check the condition of the waste stored on Pad A was conducted in 1978 and 1979. Although the initial goal of this effort was to retrieve drums for inspection and to sample surrounding soils for evidence of contaminant migration, it was discovered that the wooden boxes stacked around the perimeter of the pad were in an advanced stage of deterioration. At least one drum was recovered, but because of safety concerns related to the deteriorated box conditions and concern over safe handling of other waste containers, with regard to preventing the spread of contamination, the project was halted (DOE-ID 2002a).

- **Organic Contamination in the Vadose Zone** - Discovery of VOCs in elevated amounts in soil gas at the SDA eventually led to establishment of Operable Unit (OU) 7-08, Organic Contamination in the Vadose Zone. The remedial action selected for remediation of this OU involved the construction of a vapor vacuum extraction system that began operation in 1996. This system consists of a series of five extraction wells and an associated treatment system. Between January 1996 and early 2000, this system removed an estimated total of 105,291 lb (2.32E+08 g) of VOCs from soil gas at the SDA, including:

  - 67,589 lb (1.49E+08 g) of carbon tetrachloride

  - 15,794 lb (3.47E+07 g) of trichloroethylene

- 4,212 lb (9.27E+06 g) of tetrachloroethylene
- 5,265 lb (1.16E+07 g) of 1,1,1-trichloroethane
- 12,431 lb (2.73E+07 g) of chloroform.

3.2.3 Waste Forms

Waste forms buried at the INEEL include various types of debris (ranging from building materials to heavy equipment and clothing), sludges, and cemented liquids. The particular form in which a waste item was disposed will help determine what release processes control release of contaminants from the buried waste. In turn, these mechanisms will determine the rate of release of contaminants and the contaminants available for release. For example, contamination present on the surface of a waste item is available for wash-off by infiltrating water; however, contamination present as activation products within the structure of the item is not available for wash-off, but may be released as the item corrodes. Likewise, liquid wastes are immediately available to migrate through the vadose zone when they are disposed (provided they are not in a container); liquid waste solidified into a monolith or sludge, on the other hand, must be released from the solid matrix before they are able to be transported in the environment.

3.2.3.1 SL-1 Waste Forms

Wastes disposed in the SL-1 Burial Ground include general building debris (walls, roof, portions of the fan room and operating room floors, piping, ducting, shielding gravel, and insulation); various pieces of equipment, including a large fan, coolant cooler, condensers, an airlock, storage tanks, a pressure vessel mock up, and instrument panels; contaminated equipment (desks, chairs, filing cabinets, office equipment, and bathroom fixtures) from a support facilities building; and soil and gravel contaminated by the incident. In addition, miscellaneous lead items and lead sheet used as shielding for heavy equipment operators were also disposed in the burial ground. According to Holdren et al. (1995), the COCs for the SL-1 Burial Ground are radionuclides and are in forms that are not likely to leach into the environment.

3.2.3.2 BORAX Waste Forms

Following the final excursion of the BORAX-I reactor, waste items were placed into the foundation of the installation. Activated metal scrap, fuel residue, and the top half of the reactor shield tank were placed into the bottom half of the shield tank in the foundation. The installation pump pit, various equipment, debris, and approximately 5,400 ft^3 of contaminated soil were also placed into the foundation before it was backfilled with clean soil and mounded over with gravel for additional shielding. As with the SL-1 Burial Ground, leaching of COCs (i.e., radionuclides) from the BORAX-I Burial Ground is not likely based on the waste form present (Holdren et al. 1995).

3.2.3.3 SDA Waste Forms

Wastes buried at the SDA are highly heterogeneous in form, including reactor components, structural metals, jet engines, heavy equipment, clothing, wood, other metal objects, glassware and laboratory equipment, biological wastes, production equipment (e.g., lathes, furnaces), wastewater treatment sludges, and solidified organics. These wastes were contained in various types of packaging, including cardboard boxes, carbon steel drums, plywood boxes, and shipping casks. Large or bulky waste items were not packaged, but were, in some instances, wrapped in plastic or sealed to minimize the chances for worker exposure and were placed directly into the disposal units. Some wastes, particularly those disposed of in trenches, were compacted in place after disposal to conserve volume in the disposal units (DOE-ID 2002a).

Of particular interest are disposals of activated metals, especially irradiated beryllium blocks and reactor components made of stainless steel and Inconel (high nickel content) alloy. Irradiated beryllium blocks used as neutron reflectors at the TRA contain a significant percentage of the total buried inventory of C-14 and tritium estimated to occur in the SDA (DOE-ID 2002a). Activation products in irradiated stainless steel and Inconel include both long-lived (C-14, Ni-59, Nb-94, and Tc-99) and short-lived (Co-60 and Ni-63). Some of the stainless steel was in the form of highly irradiated end pieces from Experimental Breeder Reactor-II fuel elements; these items were buried in scrap cask inserts that were open on top and perforated on the bottom, placing these disposed items in direct contact with the soil after burial (Salomon 2001).

Placement of waste into the disposal units varied with time. Drums of waste were stacked in some pits to conserve disposal volume; at other times, waste containers were randomly dumped into the pits to minimize worker exposure and labor costs. Generally, based on earlier retrieval efforts, wastes that were stacked are believed to have better container integrity as heavy equipment used to place fill over randomly dumped containers has resulted in breaching and/or crushing of waste containers, many of which were also damaged during dumping (DOE-ID 2002a).

To assist remediation planning, wastes streams containing the SDA COCs were identified from shipping and other disposal records. These records also contained a description of the waste form comprising the waste stream and an indication of the generator of the waste. These waste streams are presented in Table B-1, together with the mechanism by which the contaminants may be released from the waste. The information in Table B-1 comes from the ABRA and reflects the state of knowledge currently. Some waste streams may change as a result of the ongoing inventory evaluations.

3.2.4 Factors Controlling Release

The release of contaminants from buried waste sources to the vadose zone is controlled by water flow, physical and chemical interactions between contaminants, waste forms and soil moisture, and transport of the contaminants to the boundary of the waste zone. Simplified models that incorporate generalized rate coefficients have been developed to provide conservative estimates of contaminant flux into the vadose zone. However, governing processes such as fluid flow, container degradation, waste form leaching are quite complex, are typically not well characterized and have not been described by detailed conceptual or multiple dimension/multiple phase numerical models.

Release of contaminants from buried waste cannot be modeled precisely because the variability of the chemical and physical reactions that occur in the waste material. Contaminants are distributed across a wide variety of waste forms and are heterogeneous in chemical composition and concentration. Waste containers fail at different rates because of variability in the subsurface conditions and the method of emplacement. Variable quantities of radionuclide and chemical contaminants are dissolved or washed away by infiltrating precipitation. To support risk contaminant fate and transport modeling, generalized conceptual models have been implemented that provide a simplified methodology to estimate the input flux of contaminants into the vadose zone. The methodology and the individual parameters that control the estimated flux are chosen to be conservative, if not bounding to avoid underestimating future potential risk due to limited site-specific data.

Release of buried waste source terms has been modeled for the RWMC in support of several risk assessment studies including the LLW Performance Assessment (Case et al. 2000), *Interim Risk Assessment and Contaminant Screening for the Waste Area Group 7 Remedial Investigation* (Becker et al. 1998) and ABRA (DOE-ID 2002a). A methodical evaluation and selection process was undertaken to identify and validate an approach for source term modeling (Becker 1997). DUST-MS was ultimately selected for use as the source term simulator at the RWMC. The model was developed at Brookhaven National Laboratory for use in performance assessment for LLW disposal. The selection of DUST-MS has influenced the development of the conceptual model, and associated model parameters, for source term release of buried waste at the INEEL.

In this process diagram, adapted from Brookhaven's work to model release from low-level waste repositories (Sullivan 1993), information regarding the buried waste inventory, specific waste streams, and waste forms are first compiled. Each waste stream is then partitioned into general categories depending on its mode of release. For example, the inventory of C-14 may be split between irradiated metal waste forms and ion exchange resins. Each group of contaminants is then made available to the release and transport processes at varying times after disposal based on the waste form and/or expected rate of container degradation. Release rates, in terms of mass of a particular contaminant per year, are then estimated by first order rate equations that represent washoff, dissolution and diffusion. The rate equations each utilize release rate coefficients that represent, in conservative terms, the multitude of physical and chemical processes at work in the waste zone.

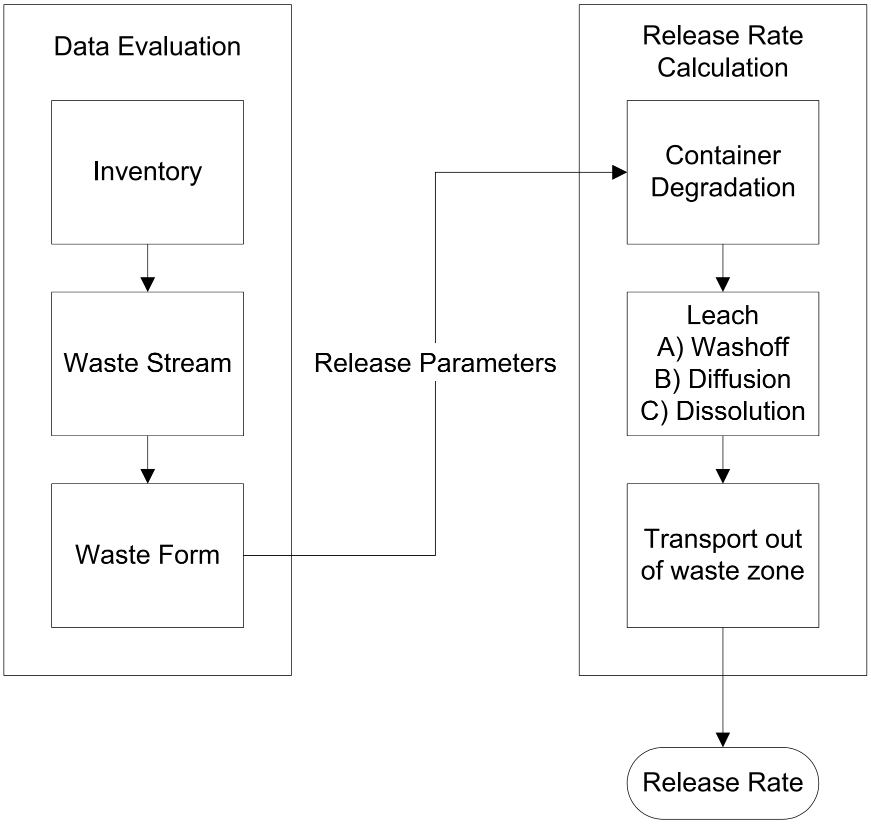


Figure 3-3. Source term release modeling process diagram.

The DUST-MS code was used to predict releases from buried waste into the shallow subsurface. The code was used to model container failure and eventual contaminant release from the waste. The DUST-MS code is a one-dimensional model that has three waste form release mechanisms: surface washoff, diffusion, and dissolution. These release mechanisms are described below:

- Surface washoff model can be used to estimate the release from general laboratory trash and is equivalent to the first-order leach model used in other codes such as GWSCREEN (Rood 1994).

- Diffusion model computes the diffusional release from different waste geometries based on user-supplied diffusion coefficients for each waste form. Diffusion of VOCs from sludge or cement-encased waste was estimated with the diffusion release model.

- Dissolution release model was used to estimate the release caused by general corrosion such as the release of activated metals from the corrosion of the base metal.

A number of key parameters govern the rate equations for each mechanism of release. As presented in the ABRA (Holdren et al. 2002), Table 3-8 summarizes the important values or source of values used in the most recent modeling work. Further discussion of a number of parameters follows.

Table 3-8. Release rate parameters.

Release Type	Rate	Comment
Container failure rate	Site-specific values	See Section 5.3.1.3
Corrosion of carbon steel	1 mm in 450 to 680 years	Nagata and Banaee (1996)
Corrosion of stainless steel	1 mm in 4,500 years (1.19E-05/year)	Adler-Flitton et al. (2001)
Beryllium corrosion	1 mm in 39.37 years (2.65E-03/year)	Adler-Flitton et al. (2001)
Metal corrosion	Contaminant solubility	Dicke (1997) (see Section 5.2)
Resin dissolution	Surface washoff model was used	
Surface washoff	Soil-to-water partition coefficients	Dicke (1997) (see Section 5.2)
Volatile organic compound diffusion	2E–03 cm^2/second (soil) to 5E–09 cm^2/second (saturated saltstone) 1.5E–08 cm^2/second is recommended.	Kudera and Brown (1996)

3.2.4.1 Dissolution

A number of key contaminants are associated with metal debris buried at the SDA. Tc-99, C-14, Nb-94, Ni-59 are contaminants that were produced in situ in metal reactor components by transmutation or activation. These activated elements are produced inside the metallic parts and are not surficial contamination. Activated elements are released to the environment by corrosion of the metal in which they were produced. To determine the release of the long-lived activated elements to the environment from the irradiated metals, it is necessary to determine the corrosion rates of the metals.

Corrosion of metal debris underground is a complex process. Soil oxygen, soil moisture, infiltrating precipitation, chemical species in the pore water, and soil bacteria contribute to faster rates of corrosion than observed in air alone. During spring thaw, the shallow RWMC soils may be in near saturated conditions, which can have high concentration of corrosive chemicals such as $MgCl_2$[1] (Nagata and Banaee 1996). In certain areas, residual moisture levels can remain high which can lead to development of bacterial colonies on the surface of the metals, which in turn can create localized areas of corrosion. Both soil clumps and microbial colonies on the metal surface can provide a mechanical barrier to soil gasses interacting with the steel. These localized areas are referred to as oxygen concentration cells where the metal beneath the barrier is exposed to less oxygen, becomes anodic with respect to the rest of the metal, and corrodes preferentially (Nagata and Banaee 1996).

The chemical composition of the buried metal is also a significant factor in estimating the rate of corrosion. Chromium, nickel, and molybdenum are included in stainless steel and Inconel alloys because they form a protective layer of oxidation on the metal surface that is resistant to most corrosive attacks and significantly reduces the corrosion rate for these metals. Conceptually, these and a host of other factors including compaction, resistivity, pH, temperature, etc. control the rate of corrosion of metallic waste forms

Prior to 1997, no corrosion data were available on metals in the RWMC soil. To support early modeling work, a comprehensive literature study and comparative analysis was performed to estimate corrosion rates for various metals buried at the RWMC (Nagata and Banaee 1996). Data from two U.S. National Bureau of Standards reports, published in 1957 and 1981, were of particular utility. By comparing reported soil properties to RWMC conditions, reasonable estimates of corrosion rates for stainless and carbon steels were identified. No literature data was found on corrosion behavior of Inconel alloys. From an examination of the alloy constituents, however, researchers concluded that Inconel corrosion behavior would be superior to stainless steel.

In addition, Nagata and Banaee (1996) reported results of a small scale corrosion test that was performed at INTEC (formerly ICPP) in 1994 to test the effects of $MgCl_2$. Researchers found that rates of corrosion for stainless steel were surprising low, even in a $MgCl_2$ solution. The low corrosion rate observed was attributed to the low temperature of the test (50°F).

The Nagata and Banaee (1996) document highlighted a number of complications in estimating corrosion rates. First, long term corrosion behavior is not known. Weight loss is probably described by power or logarithmic law that decreases with time and would not be constant as it is typically modeled. Second, many site-specific conditions are thought to influence rate of corrosion. For example, at the

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[1] Soil moisture at the RWMC can contain chlorides from a saturated aqueous solution of $MgCl_2$ that was used as a dust suppressant on RWMC roads.

RWMC the presence of roads would cause compaction, increased soil moisture, and decreased soil aeration, as well as the presence of corrosive chemicals that had been applied as dust suppressant; all of these factors would act to increase corrosion rates. This report highlighted the need for empirical data under RWMC conditions, concluding "laboratory and in situ testing should be performed to (1) develop long-term corrosion data for RWMC, and (2) confirm the validity of [the report's] approach" (Nagata and Banaee 1996).

Starting in 1997, the Long Term Corrosion/Degradation Test (LTCD) was initiated where specially prepared metal coupons of known composition were buried at varying depths in a test pit near the RWMC. Field conditions including precipitation, soil moisture, soil-water chemistry, soil-gas composition, and soil temperature were continuously monitored.

The LTCD test includes non-radioactive coupons of stainless steel (Types 304L, 316L, 316L welded), Inconel 718, Beryllium S200F, Aluminum 6061, and Zircaloy-4, and low-carbon steel. The selection of test materials was based on a previous study of the amounts and types of metal wastes present at the SDA that would have been activated after exposure in a neutron flux (Rood and Adler-Flitton 1997). The current schedule calls for corrosion measurements to be made after 1, 3, 6, and 11 years, with out years yet to have programmatic funding identified (Adler-Flitton et al. 2001)

Buried test coupons were retrieved from the soil test cell in 1998 and fall of 2000 (Figure 3-4) as reported in Adler-Flitton et al. (2001). The coupons were cleaned and analyzed for total weight loss. Pitting measurements were also made using a vertical scanning interferometer. Results from samples buried for three years showed measurable weight loss and pitting. However, the initial conclusion was that the actual corrosion rates for stainless steels at the RWMC might be considerably lower than the standard corrosion rates that were used in previous modeling studies such as the RWMC performance assessment.

Figure 3-4. Corrosion test coupons removed for analysis after 3 years of burial near the Radioactive Waste Management Complex (photo reproduced from Adler-Flitton et al. 2001).

For stainless steels and Inconel, the 3-year corrosion was not visible. Weight loss data were just above the reportable threshold and indicated that corrosion rates were 0.0001 to 0.0006 mils/year. These rates are two orders of magnitude lower than those specified in the RWMC performance assessment. Coupons for beryllium, carbon steel and aluminum showed more evidence of corrosion as would be expected. For carbon steel, the maximum corrosion rate was 0.4598 mils/year while for beryllium the maximum rate was 0.3267 mils/year. However, because both these metals exhibited significant pitting, the general corrosion rate calculated from weight loss data underestimates the actual rate of corrosion.

The initial LTCD results also demonstrated that microbial communities had developed on all of the metal samples. The presence of sulfate reducing bacteria on carbon steel, aluminum and beryllium coupons coincided with the highest corrosion rates indicating the sulfate reducing bacterial activity could have a significant impact on the calculated rate of metal corrosion in SDA soils. These results also indicated that the carbon steel, aluminum, and beryllium materials are susceptible to biocorrosion.

As summarized in Adler-Flitton et al. (2001), the initial results of the LTCD indicate that literature values for corrosion of stainless steel may overestimate the rate of corrosion, and thus would impact the estimate of future C-14 release. However, the corrosion of beryllium, for which no corrosion data exists, appears to be significant in the SDA environment, which could impact the estimates of release of C-14, tritium, and other isotopes inherent in the beryllium reflector blocks buried in the RWMC. Figure 3-5 shows cleaned beryllium coupons after 3 years of burial. Future recovery and evaluation of beryllium coupons will improve the knowledge of beryllium corrosion.

For implementation of source term release at RWMC in the most recent modeling study (DOE-ID 2002a), researchers modified the data reported in Adler-Flitton et al. 2002 to account for the potential effects of MgCl2 in the RWMC. In a letter report, Adler-Flitton, Nagata and Norby (2001) recommended a rate of 4,500 y/mm (2.22E-05 cm/year) for stainless steel in the presence of MgCl2 based on literature studies conducted by Nagata in 1997 and a rate of 39 y/mm (2.54E-03 cm/year) for beryllium in the presence of MgCl2. It should be noted that all estimates were accompanied with recommendations that specific corrosion testing be performed under RWMC conditions.

Figure 3-5. Beryllium coupon (front and back) recovered after 3 years of burial, before cleaning (photo reproduced from Adler-Flitton et al. 2001).

To calculate the release of contaminants from the metal waste forms, the geometry (surface area) also needs to be estimated. In development of the ABRA, investigators estimated the ratio of surface area to volume for typical INEEL-type reactor components to be 0.535/cm based on data provided in Oztunali and Roles (1985). Combining the corrosion rate and the geometry data provided a fractional release from the stainless steel of 1.19E-05/year. The ratio of surface area to volume of the beryllium blocks is 1.043 1/cm. Combining the geometry data with the corrosion rate provides a fractional release from the beryllium blocks of 2.65E-03/year.

3.2.4.2 Infiltration Rates

Buried waste disposal facilities at the INEEL are located above the water table in the unsaturated zone. Infiltration of water into a facility will involve many processes including precipitation, evapotranspiration, and surface run-off. This is further complicated by engineered barriers such as caps, soil covers, etc. that will minimize the fraction of precipitation that flows into the waste zone. Finally, the flow rate will vary over time due to short-term events (i.e., precipitation) and long-term events (i.e., barrier degradation and climatic changes). Even if the change in material properties and precipitation rates are known, water flow in the unsaturated zone is difficult to model due to the non-linearlity of the flow properties (Sullivan 1992). Because the complexities involved with modeling the evolution of unsaturated flow through a disposal unit, a simplified model has been adopted. The DUST-MS code models water flow through tabular input of the volumetric flow velocity versus time. This velocity can be represented by the annual precipitation rate (as an upper bound) or simulated based on site-specific moisture measurements.

Infiltration of precipitation through the surficial sediments is a key parameter in the conceptual model. Although permanent surface water features do not complicate the hydrology, infiltration is spatially and temporally variable. Infiltration of water involves many processes including precipitation, evapotranspiration and runoff. Water flow through the unsaturated waste zone is complicated by properties and distribution of the overlying and interstitial soils as well as the waste itself. A simple model for infiltration is utilized where average infiltration rates for different areas across the SDA were derived from soil moisture monitoring data. Although a constant flow rate is assumed in order to simplify the model, because the period of interest is so great, a time dependent infiltration model would probably not be meaningful.

A number of studies have been conducted by the DOE and USGS to estimate infiltration rates for purposes of vadose zone transport modeling. The results of these studies have also been applied to source term release modeling. In 1988, the USGS completed construction of a simulated waste trench several hundred yards north of the SDA. The test trench was designed to investigate infiltration in natural, undisturbed sediments; disturbed (excavated and backfilled) sediments; and simulated waste zones. Using tracers and extensive instrumentation (i.e., thermocouple psychrometers, lysimeters, tensiometers, and neutron moisture probes), the USGS quantified soil moisture and variability with depth, time, and temperature and estimated soil hydraulic conductivity, soil-moisture flux, and evapotranspiration rates (Pittman 1995).

The USGS also conducted 24-hr infiltration tests with small, 8-cm deep ponds. The objective of these infiltration tests was to quantitatively compare field scale measured hydraulic properties to the lab scale measurements. Despite the thoroughness of the soil property measurements, the data proved inadequate for predictive one-dimensional modeling. The USGS found that “even with a detailed characterization of a natural soil profile, undetected features may dominate the hydrologic behavior” (Nimmo et al. 1999). It is thought that layering and preferential paths influence flow in undisturbed soil. In disturbed, waste trench soil, initial infiltration is slower, but lack of significant layering is thought to permit water to move freely to depth (Nimmo et al. 1999). USGS test trench conclusions, as summarized by Perkins (2000), were that moisture content is variable in shallow depths but relatively constant at depths below 3 m, even under flood conditions. The USGS test trench work is further documented in Pittman (1989) and Davis and Pittman (1990). Additional infiltration studies were conducted by Cecil et al. (1992) using chlorine-36 as a surface water tracer.

A network of neutron-probe access tubes (NATs), with associated tensiometers, was installed across the SDA during the mid 1980s to measure soil moisture content and estimate infiltration rates (Bishop 1996). Monitoring started in 1986 and was performed in a number of discontinuous events across a variably sized network. McElroy (1993) and Bishop (1994; 1996) demonstrated that net infiltration is spatially and temporally variable. Net infiltration was found to range from 49 cm/yr (19.4 in./yr) to less than 0.3 cm/yr (0.1 in./yr) depending on available precipitation (snow depth) and variations in runoff and ponding (Bishop 1996). The greatest infiltration occurs in the spring, since snowmelt is a significant contributor (DOE-ID 2002a). Local topography and soil characteristics also control infiltration.

Martian (1995) analyzed the moisture data obtained from the NATs between 1986 and 1996 to estimate variable infiltration rates across the SDA. These estimated infiltration rates were used in the IRA and the ABRA. The background infiltration rate outside of the SDA was 1 cm/yr. Infiltration rates within the SDA ranged from 0.64 to 24.1 cm/yr (Martian 1995), with a spatial average of 8.5 cm/yr. The cumulative volume of modeled infiltration at the SDA is approximately 27.8 acre-ft/yr (Magnuson and Sondrup 1998).

To support recent modeling of source term release, the NAT data from across the SDA were used to estimate specific infiltration rates for a variety of waste streams, depending on their disposal location. Figure 3-6 (reproduced from DOE-ID 2002a) shows simulated infiltration rates distributed across the SDA. The colored grid blocks are associated with specific waste streams. The average infiltration rates for each waste stream are indicated at the bottom of the figure.

3.2.4.3 Container Failure Rates

Before a contaminant can be released from waste form to the environment, the container in which the waste is buried must first degrade. Waste containers at the RWMC include drums, steel boxes and canisters, wooden and cardboard boxes, and plastic bags. Mechanistic modeling of the degradation of container materials would require extensive knowledge of local chemistry and detailed inventory of waste containers. The data for mechanistic modeling of corrosion is not well known in a disposal environment and subject to large uncertainties. (Sullivan 1992?) Therefore, the approach used at INEEL has been to use the DUST-MS computer code, which accounts for container degradation by specifying time to failure and allowing fractional area of the container that has failed to increase with time until the entire container is breached. This provides a means of estimating releases due to localized failure that occur earlier in time than does general failure.

One of the primary forms of waste containment used at the RWMC was 55-gal steel drums. An estimate of the time at which drum failure can be expected is needed to estimate the rate of release of contaminants from the buried waste. Drum failure is not defined as complete degradation of a drum because small holes would allow diffusion of VOCs and infiltration of water through the drum contents. Therefore failure has been defined as any breach of containment (Becker 1997).

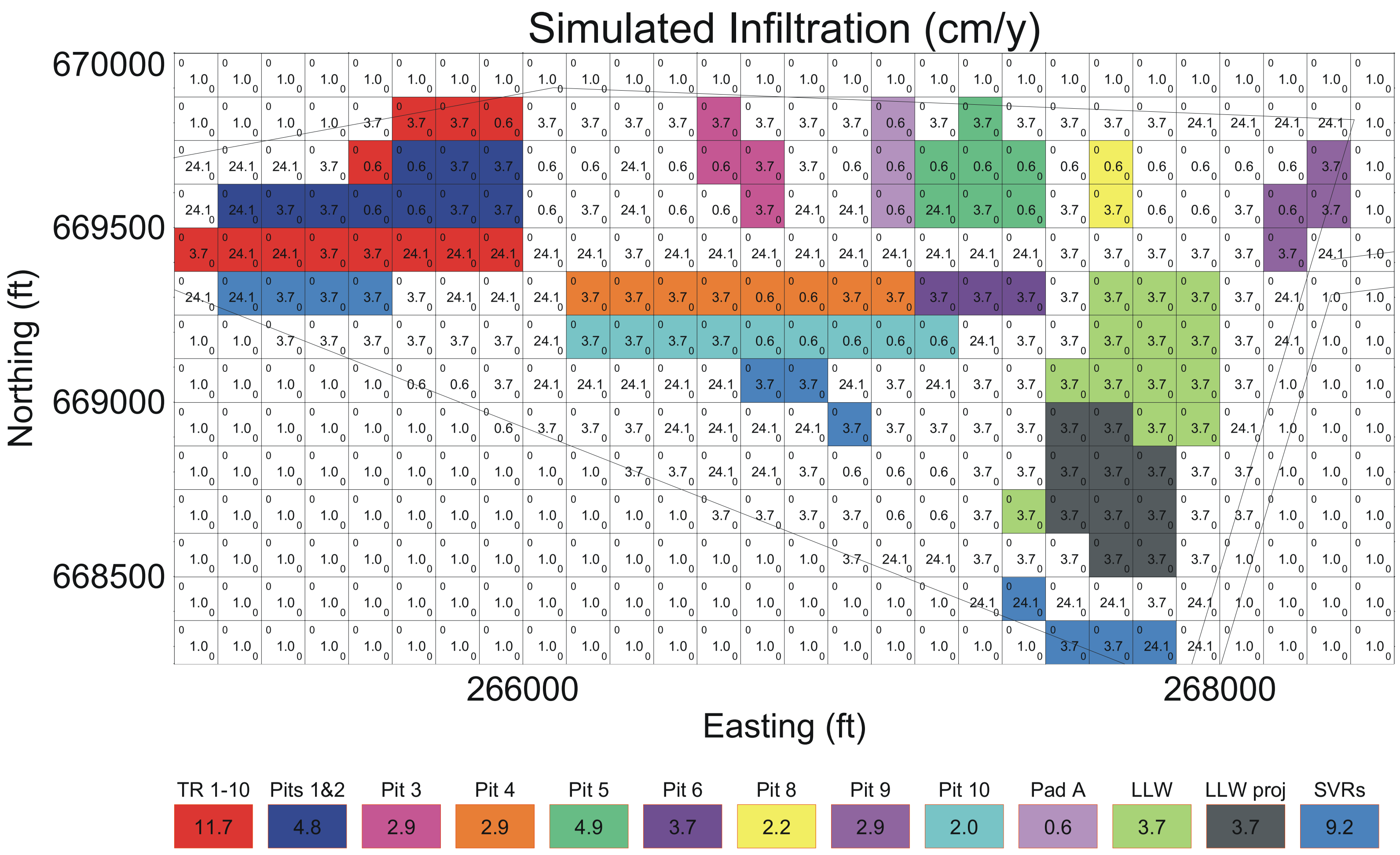


Figure 3-6. Infiltration rates assigned in the subsurface flow model and averages used for the various waste streams identified in the Ancillary Basis for Risk Assessment source term model (reproduced from DOE-ID 2002a).

Historic retrievals of buried waste conducted during the 1970s provided a substantial set of data related to the condition of drums buried for various lengths of time. Analysis of the drum condition information indicated that randomly dumped drums failed significantly sooner than did stacked drums (Figures 3-7 and 3-8) (Martian and Sondrup 1995). Assuming that drum failure would follow a normal distribution model, mean container life from dumped drums was estimated at approximately 12 years with a standard deviation of 5 years. Stacked drums, using the same method, showed mean container life of 23 years with a standard deviation of 10 years.

Time dependent container failures were implemented in the most recent source term modeling work by using DUST-MS. The DUST-MS code allows the user to specify the time of container failure. If the waste disposal was performed without containment, then the failure time in the code was set at zero and the release mechanisms would control the release rate of the contaminants from the waste. When the container is assumed to have been breached, the data will indicate that the waste was released to the subsurface in accordance with the release mechanisms that were appropriate for that waste stream (DOE-ID 2002a).

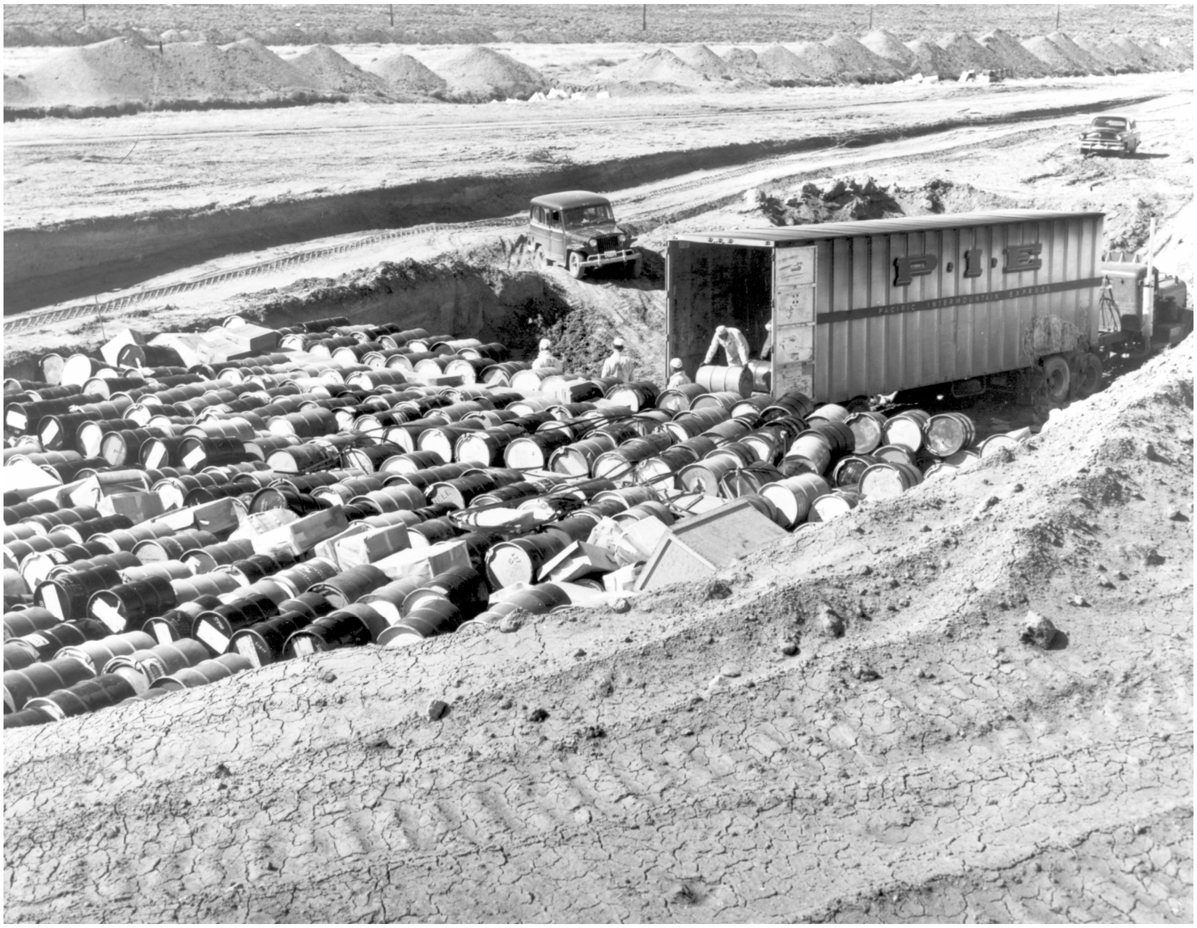


Figure 3-7. Early disposal practices included orderly stacking of drums (INEEL photo # 58-1450).

Figure 3-8. To reduce worker exposure times, random dumping practices were adopted in later years (INEEL photo # 69-6138).

Researchers used the yearly RWMC disposal information to assign containers for calculation of the release. A single contaminant might have been included in multiple types of containers buried in a given year because of the different waste form or different containment to be modeled. For example, if the disposal contents of a particular waste stream were buried in drums and cardboard boxes, the relative fractions would be used to model that year's disposal. Because cardboard boxes do not provide any barrier to contaminant movement, the fraction of waste disposed of in cardboard boxes was simulated as having the container fail at time of disposal. The fraction of a contaminant in drums is modeled with the distributed failure appropriate for the type of drum disposal (i.e., whether the drums were stacked or dumped).

In the ABRA (DOE-ID 2002a), individual waste streams were evaluated for the type of container used. The disposal contents of many waste streams were buried in wood or other readily degradable boxes. It was assumed that these readily degradable containers offered no impediment to contaminant movement; therefore, no delay of the contaminant release was assumed for the boxes in the model. Also, polyethylene bags were not accounted for in the release modeling. This was a conservative assumption for contaminants other than VOCs. Calculations by Kudera and Brown (1996) indicated that VOC transport through polyethylene bags is rapid compared to transport through sludge. Therefore, transport through the bags is not the limiting step in VOC releases. The 55-gal drums, concrete casks, and metal boxes offer a barrier to contaminant release accounted for in the source term model. Waste in the containers will be assumed as released only after the drums, casks, or metal boxes are assumed to have failed. Waste streams for which no specific container type was identified were modeled as having no containment to be conservative.

Figure 3-9. Corrosion observations during historic drum retrievals were used to estimate container failure times (INEEL photo #71-5417).

Metal boxes and canisters were assumed to have failed when one half of the wall thickness has corroded, using the carbon steel corrosion rate estimated by the recent LTCD (Adler-Flitton et al. 2001). Concrete casks were not modeled with an assumed failure time. Instead, the release of contaminant mass from within the casks was modeled as diffusion out of the cask. The casks were modeled as cylinders with a 6-in. (15-cm) wall thickness. This thickness assumption was conservative for early releases because it assumed that the waste was at the surface of the cask and was released readily. In addition, a conservative diffusion coefficient of 1.0E-06 cm^2/g was used. A diffusion coefficient of 1.0E–06 cm^2/g is typical for a metal ion in water and is conservative because it does not account for the possible partitioning of the contaminant with the waste form or the porous media that the contaminants must travel through. Any partitioning would slow the contaminant release.

3.2.4.4 Washoff

The washoff or surficial rinse model assumes that contaminants are available as soon as water contact occurs. Prior to water accessing the wastes, the contaminants may be held on the surface by adsorption, chemisorption, adhesion, and ion exchange processes (Sullivan 1992).

For modeling work done at the RWMC, the surface washoff release mechanism was used for waste streams that are generic laboratory trash. The surface washoff release mechanism provides the most conservative release rates. Similarly, contaminants identified as surface contamination of a base material (e.g., anticontamination clothing) were modeled with the surface washoff release model. The surface washoff model uses a partition coefficient to determine the release.

Distribution coefficients (i.e., Kds) are commonly used in computer modeling as a mathematically simple representation of sorption. Kd values encompass all processes that remove a contaminant from solution. They are a bulk term representing the ratio of adsorbed to dissolved concentrations, typically given in units of mL/g. Typically, the value is obtained by fitting a linear isotherm to results of batch or column experiments, neglecting the actual mechanisms responsible for contaminant removal (Dicke 1997).

However, where there are significant temporal or spatial heterogeneities in the geochemical environment, the use of Kds becomes problematic. For example, Hull (2001) comments that multiple water types have been identified in the RWMC subsurface, indicating that geochemical conditions are not uniform. Hence, the Kd for a particular contaminant would vary spatially.

There are a number of limitations with using the Kd approach. For example, spatial or temporal variability in chemical conditions and solubility potentials cannot be addressed with singular partition coefficients. Additionally, it is difficult to accurately represent actual field conditions in batch sorption experiments (Hull and Pace 2000). Or as stated by the USGS (1999), the measured Kd values are applicable only to the specific conditions under which the experiments were conducted. Other problems with the use of Kd values include poor design of batch experiments, competition between species for sorption sites, and non-linear response to increasing aqueous concentration. To date, facility-specific simulations have relied on the linear isotherm approach using a limited set of site-specific data. Although the results are generally considered conservative for purposes of risk assessment, a number of investigators have suggested that long-term evaluations be conducted to provide a basis for more representative geochemical modeling.

The literature regarding the use of Kd values is extensive. As a starting point, the reader is referred to the discussions developed in USGS (1999). These researchers reviewed the use of Kds at RWMC for four specific contaminants (Am, Np, Pu and U) and concluded that "the experimental conditions used to measure Kds for the IRA model do not adequately represent the aqueous and solid phase variability at [the RWMC]." The USGS comment was specific to interbeded sediments, but the same comment applies to the use of Kds in the release model. The reviewers proposed the use of a surface complexation model, as specific surface-complexation and ion-exchange constants are important parameters to utilizing Kd measurements. In order to support the simulation work specifically for the RWMC, a comprehensive set of Kds and solubility limits were compiled in Dicke (1997) for RWMC contaminants. This compilation utilized literature values and results of batch tests on RWMC cores where available. The paucity of site-specific data prompted testing on additional samples in the 1999 timeframe.

A key parameter controlling the sorption reactions is the oxidation state of the contaminant of interest. It has been shown that the transport and adsorptive characteristics of contaminants, particularly actinides, are strongly affected by their initial oxidation state and by oxidation/reduction reactions within the vadose zone (Hull and Pace 2000; USGS 1999). Oxidation states and associated potential for organic and inorganic complexation controls, in part, the fraction of contaminant entering and remaining in solution. Speciation of contaminants (i.e., their form as disposed and as altered by waste zone characteristics), as well as redox state of the water, is not well known and thought to be highly variable.

The RWMC contaminants display a wide range of geochemical properties. Chemical interactions with pore water, sediments, and basalt matrices are complex and supported by a small set of empirical data. Solubility calculations on a contaminant-by-contaminant basis indicate that the effect of oxidation/reduction potential and hydrogen potential on the solubility of these contaminants varies. However, for many contaminants (particularly actinides) the ambient SDA conditions, with moderately reduced environment and approximately neutral pH, may minimize solubility (Hull and Pace 2000). To simulate the release of metals in the ABRA (DOE-ID 2002a) the surface washoff release model has been used with an estimated appropriate solubility limit for INEEL soil water chemistry.

Release rates from ion exchange resins are somewhat uncertain. The surface washoff release model was used as the release model for the resins, which was chosen to be conservative (DOE-ID 2002a).

3.2.4.5 Diffusion

Experimental leaching data from solidified radioactive wastes indicate that diffusion is the predominant rate controlling process (Sullivan 1992). Diffusion controlled release is characterized by relatively high leach rates at early times that continually decrease in time. The diffusion model implemented for INEEL buried waste considers two geometries cylindrical (drums) and rectangular (boxes). A simplifying assumption is that the contacting solution has a concentration of zero, which provides a conservative, albeit unrealistic concentration gradient, resulting in a high release. Diffusion controlled release mechanism ahs only been applied to VOC waste forms and a few waste streams that were encased in concrete. The majority of the RWMC waste forms are thought to be better described by other, more rapid, release processes.

Sludge buried at the SDA contains VOCs, radioactive contaminants, and other hazardous constituents. The release of VOCs from the sludge is through vapor diffusion. In Kudera and Brown (1996) the average diffusion rate was estimated as 2E-06 cm^2/second, which was based on the VOC transport model calibration effort, but values can vary by several orders of magnitude. Release of metals and radioactive contaminants from sludge occurs through leaching, which was modeled with the surface washoff release model. The surface washoff release model provided a more conservative estimate of the release than with aqueous diffusion from the sludge.

3.2.5 Current Uncertainties

Two general types of uncertainties regarding the buried waste source term, those related to the source term inventory and those related to mechanisms for release to the vadose zone of contaminants in the inventory. Uncertainties related to the source term inventory result from the lack of information regarding the contaminant inventories that may have been disposed. Consequently, various efforts have been undertaken to quantify waste amounts and constituents in the various burial grounds. These efforts include reviewing disposal records, assay information, and sampling data related to the waste.

For the SL-1 and BORAX-I Burial Grounds, computer codes were used to estimate the inventory and concentrations of radionuclides in the buried waste. These models are based on steady-state operations of the reactors; however, major power excursions occurred in the final moments of operation for each reactor and resulted in their destruction. For each reactor, the majority of the radionuclide inventory is produced by fission products. Additional inventory would be added by the activation of U-238 to produce Pu-239 and the activation of reactor structural materials and reactor vessel. However, uncertainty regarding the fate of radionuclides after the destruction of the reactors themselves is much larger than the uncertainty associated with not including activation products in the structural materials. Also, it is possible that the final excursions of the reactors produced more or less fission products than predicted by the models (Holdren et al. 1995).

The mechanisms and factors controlling contaminant release from buried waste are also under investigation. The basic mechanisms, surface washoff, diffusion, and dissolution, have been identified. Research is currently being performed to understand and quantify factors, such as infiltration, corrosion, and container failure, that influence these mechanisms and affect the release of contaminants from buried wastes. Investigations are also underway to better understand the influence of near-field effects on the chemistry of infiltrating water and its effect on contaminant solubility.

3.2.5.1 SL-1 Source Term Uncertainties

The nature of the SL-1 incident and its cleanup did not permit accurate measurement of the radionuclides present in the waste that was disposed of in the burial ground. As a result, the ORIGEN2 computer model and operating history of the reactor were used to estimate inventories for the reactor. This required that certain assumptions be applied, as follows:

- The ORIGEN2 code is appropriate for estimating inventories of radionuclides produced by the reactor
- Activation products associated with structural materials of the reactor are minimal and were not considered
- The final excursion did not significantly impact the quantities of fission products predicted by the code
- The percentage of fission products present in the burial ground is the same as the percentage of U-235 (originally present in the fuel) not recovered during cleanup operations; for the SL-1, this amounted to 7%. Consequently, the results of the ORIGEN2 modeling were multiplied by 7% to give the amount estimated to be present in the buried waste.

The ORIGEN2 model is based on steady-state operation of the reactor. The SL-1 incident produced only a small fraction of the total power produced during the operating history of the reactor. Consequently, the potential deviation from the actual inventory to the modeled inventory is expected to be small (Holdren et al. 1995).

Uncertainties are also associated with the source term for non-radionuclide contaminants in the SL-1 Burial Ground. Anecdotal evidence indicates that lead sheeting used as shielding by the equipment operators during clean-up, fluorescent lights (possibly containing polychlorinated biphenyls [PCBs], cadmium, and mercury), and ceiling tiles containing asbestos may have been buried with the other wastes. Lead sheeting would be present in monolithic form and would not be readily leached by soil moisture. PCBs and metals (cadmium and mercury) are not readily mobile in the subsurface, so even if present would not represent a significant risk. Additionally, there is evidence that the light fixtures were not disposed of with the buried waste but were decontaminated and placed into storage (Holdren et al. 1995). Asbestos, if disposed in the burial ground, does not present a hazard as long as it remains buried and unavailable for suspension in the air.

3.2.5.2 BORAX-I Source Term Uncertainties

As with the SL-1 reactor, the final destructive excursion of the BORAX-I reactor precluded accurate measurement of the radionuclide source term that may be present in the buried waste. Buried radionuclide inventories for the BORAX-I Burial Ground were modeled using the RSAC-5 model and reactor operating history. Assumptions used in estimating the BORAX-I source term include:

- The RSAC-5 code is applicable to the BORAX-I reactor for accurately estimating radionuclide inventories.
- To conservatively estimate the amount of fission products present, the fuel in the reactor at the time of the final excursion was assumed to have been used during the entire operating history (2 years) of

the reactor. No documentation was available regarding the replacement history of the fuel elements, despite being used for tests that likely resulted in damage to fuel elements in the reactor core.

- Activation products associated with structural materials of the reactor are minimal and were not considered.

A major uncertainty results from the assumption that the percentage of radionuclides present in the waste is directly proportional to the percentage of U-235 in the fuel that was not recovered during clean-up operations. For BORAX-I, the amount of U-235 not recovered was 88%, so modeling results were multiplied by 88% to give the estimated buried inventory. However, the final excursion of the reactor resulted in contamination of approximately 84,000 ft^2 of soil surrounding the reactor. Even after several attempts to remediate this area, the soil remains radioactively contaminated, so assuming that the entire 88% of U-235 not recovered is buried with the reactor shield tank and other materials is considered conservative (Holdren et al. 1995).

3.2.5.3 CFA Landfills Source Term Uncertainty

Information regarding the types and quantities of hazardous and radioactive materials disposed with the wastes buried in the CFA landfills is nonexistent. Locations of these wastes in the landfills is also unavailable. Therefore, estimates of the radionuclide and non-radionuclide inventory within waste buried in these areas cannot be made.

3.2.5.4 SDA Source Term Uncertainties

Estimates regarding the quantities of contaminants in the buried waste at the SDA are uncertain. Likewise, information related to disposal locations of specific wastes and waste items are limited by the available records. Although disposals at the SDA were documented, at least at some level, the information available in this documentation remains open to interpretation, as can be seen in previous and on-going efforts to refine inventory estimates. As described previously, past efforts (i.e., HDT and RPDT) have focused on refining the inventory of wastes and contaminants based on review of historical disposal records. These efforts have culminated in the CIDRA and WasteOScope databases. Currently, efforts continue to refine inventory estimates and waste locations using existing documentation, but are focused on disposal inventories originating from INEEL generators (DOE-ID 2002a).

Uncertainties still exist in the location of particular waste disposals. Uncertainties also exist in the near-field conditions (i.e., alterations to natural soil conditions in the proximity of the waste from soil disturbance during excavation and burial) within the SDA. Consequently, it is desirable to have a means of verifying CIDRA and WasteOScope information to provide additional information on waste locations, and to provide information on potential migration of contaminants within the SDA. Direct physical sampling of the SDA disposal areas is not feasible due to the heterogeneous nature of the wastes, the presence of large or bulky waste items, and hazards present in the wastes. To provide this additional information and to collect monitoring data directly from the waste zone, 337 probes were installed in the SDA beginning in December 1999. These probes are intended to evaluate the following (DOE-ID 2002a):

- Locations of waste types, distributions of radionuclides in the buried waste near the probes, and thickness of the soil and waste layers

- Radiological fingerprints for identifying different waste streams

- Infiltration rates through the cover, buried waste, and underburden soils

- Release rate and solubility of uranium
- Release rate of C-14
- Mass of the VOC source term remaining in the buried waste.

Most of the probes were installed directly into the buried waste to provide monitoring capabilities next to or near the probes. The probes were of two general types, Type A and Type B. Type A probes are hollow tubes, sealed at the bottom, to allow safe access for a variety of nuclear logging instruments, including passive spectral gamma detectors, neutron activation instruments, neutron-neutron detectors, passive neutron detectors, and directional gamma detectors. Type B probes include a variety of probe types, as follows (DOE-ID 2002a):

- Tensiometers - provide data on the variability of moisture in the waste zone, quantify the amount and timing of moisture infiltration, and define the presence and extent of saturated conditions
- Soil moisture probes – measure the moisture content of soil using the relationship between the soil dielectric constant and the moisture content, perform resistivity measurements of the electrical contrasts between different geologic media, and provide temperature measurements of material surrounding the probe
- Suction lysimeters – collect soil moisture samples by applying a slight vacuum to a porous cup in contact with the soil
- Vapor ports – collect gas samples from the waste zones and the area surrounding the soil vaults
- Visual probes – allow direct observation and investigation of the soil overburden and the waste zone, including:
    - Location of the top and bottom of the overburden and underlying sediment
    - Thickness of sediment beneath the waste
    - Relative grain size (i.e., sand, pebbles, and cobble) of the geologic media
    - Stratification in the sediment beneath the waste or disturbance of the sediment
    - Color of the sediment (to indicate oxidation/reduction)
    - Amount of sediment versus waste adjacent to the probe
    - Visual indication of moisture movement
    - Evidence of how tightly the tube is sealing
    - Waste container condition
    - Void space caused by drum placement or lack of material
    - Presence of cellulosic material (e.g., cardboard, wood, and paper)
    - Waste form identification (e.g., sludge, graphite, metal, nitrate salts, combustibles, and noncombustibles).

The probes have been placed to provide information on specific areas (focus areas) of the SDA: the depleted uranium focus area (Pit 10), the organic sludge focus area (Pit 4), the americium and neptunium focus area (Pit 10), the uranium and enriched uranium focus area in Pit 5, and the activated metal investigations in Soil Vault Rows 12 and 20. The probe array is also providing data to support waste zone moisture monitoring and the Pit 9 study area. The SDA probe studies are presently underway and additional detail regarding the studies can be obtained in DOE-ID (2002a).

3.2.5.5 Release Factor Uncertainties

Uncertainties also exist in quantifying the various factors associated with the release of contaminants from buried waste. The release of contaminants is a key, rate-controlling element in the migration of contaminants to groundwater. Unfortunately, leach rates and release mechanisms for buried waste at INEEL have yet to be established and are not completely understood. Further, the geochemical conditions in buried waste are complex with variable oxidation/reduction environments, microbial activities, waste container and waste item corrosion rates, and rates of diffusion through monolithic waste forms. These uncertainties affect which contaminants may be released, as well as the rate of release of those contaminants, and are summarized below.

Infiltration - One of the basic drivers for modeling of releases from buried waste is the rate of water infiltration through surficial soils placed over the waste and soil in the waste interstitial spaces. Annual precipitation at the INEEL averages approximately 9 in./year, with the highest rate of precipitation coming during the months of May and June, with infiltration of snowmelt also being a significant contributing factor. Local topography can greatly influence infiltration, particularly at the SDA, where ditches, roads, and small depressions channel runoff and snowmelt to localized ponding areas within the SDA. Within the buried waste, layering and preferential flow paths result from the waste pits and trenches.

Studies have been undertaken at the SDA to understand the various influences affecting infiltration. Infiltration rates have been estimated for various generalized waste streams based on the characteristics of the waste, local soils, and burial types. However, such detailed efforts have not been performed for the SL-1 and BORAX-I Burial Grounds, resulting in greater uncertainty for any estimates of contaminant release from these sites.

Corrosion rates – Corrosion rates will affect not only the release of contaminants from sealed containers, but also the release of activation products from irradiated metals. Container corrosion rates have been estimated for both stacked and randomly dumped containers to model the release of VOCs and other waste constituents from failed containers. Paint or other coatings on the containers will slow the rate of corrosion, except in areas where the coating may be damaged. These damaged areas will serve as an anode and corrosion will occur at an increased rate. Container corrosion estimates are based on several studies and experiments, but are not measurable for actual releases. Thus, although the estimates are based on the best available data, they remain estimates and as such are subject to error.

For activated metals such as stainless steel and Inconel alloy, it is difficult to estimate the corrosion rates because 1) the period of interest is so long that climatic changes could alter corrosion rates; 2) corrosion weight losses on small specimens for short term specimens are very small, which are difficult to accurately measure; and 3) the corrosion rate is probably not constant, and would be expected to decrease over time which is not discernable in short term tests. Additionally, effects of site-specific conditions such as compaction, soil texture, moisture content, etc. were not well known and contribute to uncertainity in the estimates (Nagata and Banaee 1996). Although corrosion of stainless steel is conservatively assumed to occur at a constant rate, the formation of an oxidation layer by chromium and nickel form on the surface of the metal inhibits further corrosion (Nagata and Banaee 1996).

Also, all tests on metal corrosion were performed on unirradiated metals. The difference between unirradiated and irradiated metal, especially beryllium may have a significant effect on the corrosion rates (Adler-Flitton, Nagata, and Norby 2001). The characteristics of beryllium metal are greatly affected by irradiation, which causes the blocks swell and crack, increasing the effective surface area. Also, although corrosion is usually measured by differences in weight of specimens, the susceptibility of beryllium to corrode via pitting of the surface may be an important mechanism that increases potential releases of activation products versus what one might expect in estimating potential releases and the degree of corrosion based on weight loss alone. All estimates of corrosion rates have been consistently caveated with the need to gather empirical data to substantiate "reasonable guesses" of corrosion rates.

Complexing agents – The presence of complexing agents in waste at the INEEL has not been evaluated. The formation of aqueous complexes in soil solutions can keep contaminants in solution and inhibit solution-solid reations that retard the migration of contaminants. Decontamination solutions containing complexing agents such as citrate, oxalate, and EDTA (present in RFP waste as Versenes) were used and disposed of in the SDA. Inorganic anions such as fluoride, carbonate, and phosphate can form aqueous complexes with contaminants. Exact amounts of complexing agents disposed at the SDA are not known, and anions are not always measured because they are not considered contaminants. Thus, the potential impact of complexing agents on release rates from buried waste at the SDA is not known.

Solidified waste forms – Releases of contaminants from solidified waste forms has been modeled as a simple diffusion process. This is estimated by assuming diffusion through the surface area of the waste as placed in the SDA. However, cracks may be present in the waste form, increasing the surface area over which diffusion can occur. This cracking also increases the area of which contaminants may be washed off the item and provide increased opportunity for dissolution of the waste form.

Temporal and spatial variability - Extreme spatial variability in the waste present at a given location results from the heterogeneous nature of the wastes themselves, as well as the disposal techniques used in placing the wastes. Also, because the waste changes over time (e.g., through radioactive decay, changes to oxidation states resulting from oxidation/reduction reactions in the soil), chemical conditions and solubility potentials cannot be addressed using singular, constant partition coefficients. Measured partition coefficients, referred to as K_ds, are often determined through experiments; however, the experimental conditions analyzed may not adequately represent conditions present in the field in the buried waste. These difficulties arise from poor experimental design, competition between species for sorption sites, and non-linear response to increasing contaminant concentrations (Hull and Pace 2000, USGS 1999).

Solution chemistry in the buried waste - Application of K_ds to buried wastes is also complicated by uncertainities related to the burial grounds themselves. Typically, the chemical form of the contaminants as disposed (i.e., oxidation state of metals) is not known and must be an assumed input to any release modeling performed. Also, anions/cations in soil will also affect the water chemistry and affect of the water on the waste items, an issue that is further complicated by near-field effects of the burials and, in the case of the SDA, the presence of different soil types (i.e., native soils and soils from the Big Lost River Spreading Areas have been used as backfill in the different disposal units fill and in recontouring the SDA to counter subsidence and enhance drainage).

3.2.6 References

Adler-Flitton, M.K., P.K. Nagata, and B. C. Norby, 2001, “$MgCl^2$ Enhanced Underground Corrosion Rate Estimates of Austenitic Stainless Steels and Beryllium,” mkaf-01-01, Interoffice Memorandum to K. J. Holdren, September 27, 2001, Idaho National Engineering and Environmental Laboratory.

Adler-Flitton, M. K., C. W. Bishop, R. E. Mizia, L. L. Torres, and R. D. Rogers, 2001, *Long Term Corrosion/Degradation Test Third-Year Results*, INEEL/EXT-01-00036, Revision 0, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory.

Becker, B. H., 1997, *Selection and Development of Models Used in the Waste Area Group 7 BRA*, INEEL/EXT-97-00391, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory.

Becker, B. H., J. D. Burgess, K. J. Holdren, D. K. Jorgensen, S. O. Magnuson, and A. J. Sondrup, 1998, *Interim Risk Assessment and Contaminant Screening for the Waste Area Group 7 Remedial Investigation*, DOE-ID-10569, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, U.S. Department of Energy, Idaho Operations Office.

Bishoff, J.R. and R.J. Hudson, 1979, “Early Waste Retrieval, Final Report,” TREE-1321, Idaho Falls, Idaho, Idaho National Engineering Laboratory, EG&G Idaho, Inc.

Bishop, C.W., 1994, *Expansion of Moisture Monitoring Network at the Subsurface Disposal Area*, INEL-94/0144, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company, Idaho Falls, Idaho.

Bishop, C.W., 1996, *LMITCO Internal Report, Soil Moisture Monitoring Results at the Radioactive Waste Management Complex of the Idaho National Engineering Laboratory, FY-96, FY-95, and FY-94*, INEL-96/297, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company.

Case, M. J., A. S. Rood, J. J. McCarthy, S. O. Magnuson, B. H. Becker, and T. K. Honeycutt, 2000, *Technical Revision of the Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment for Calendar Year 2000*, INEEL/EXT-2000-01089, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Bechtel BWXT Idaho, LLC.

Cecil, L.D., J.R. Pittman, T.M. Beasley, R.L. Michel, P.W. Kubik, P. Sharma, U. Fehn, and H. Gove, 1992, “Water Infiltration Rates in the Unsaturated Zone at the Idaho National Engineering Laboratory Estimated from Chlorine-36 and Tritium Profiles, and Neutron Logging,” in Proceedings of the 7th International Symposium on Water-Rock Interaction - WRI-7, Y. K. Kharaka and A. S. Meest (eds.), Park City, Utah.

Clements, T.L., 1980, “Buried Waste Characterization: Nonradiological Hazards Study – Offsite Waste Generators,” PR-W-80-027, Idaho Falls, Idaho, Idaho National Engineering Laboratory, EG&G Idaho Inc.

Davis, L.C. and J.R. Pittman, 1990, *Hydrological, Meteorological, and Geohydrological Data for an Unsaturated Zone Study near the Radioactive Waste Management Complex, Idaho National Engineering Laboratory, Idaho—1987*, U.S. Geological Survey Open-File Report 90–114 (DOE-ID– 22086), Idaho Falls, Idaho, U.S. Geological Survey.

Dicke, C.A., 1997, *Distribution Coefficients and Contaminant Solubilities for the Waste Area Group 7 BRA*, INEL/EXT-97-00201, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company, Idaho Falls, Idaho.

DOE-ID, 2002a, *Ancillary Basis for Risk Analysis of the Subsurface Disposal Area*, INEEL/EXT-02-01125, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, U.S. Department of Energy, Idaho Operations Office.

DOE-ID, 2002b, *Central Facilities Area Landfills I, II, and III Five-Year Review Supporting Documentation*, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, U.S. Department of Energy, Idaho Operations Office.

DOE-ID, 1997a, *Comprehensive Remedial Investigation/Feasibility Study for the Idaho Chemical Processing Plant Operable Unit 3-13 at the INEEL – Part B FS Report*, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, U.S. Department of Energy, Idaho Operations Office.

DOE-ID, 1997b, "Comprehensive Remedial Investigation/Feasibility Study for the Test Area North Operable Unit 1-10 at the Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho," Idaho National Engineering and Environmental Laboratory, U.S. Department of Energy, Idaho Operations Office.

Holdren, K.J., R.G. Filemyr, and D.W. Vetter, 1995, "Remedial Investigation/Feasibility Study Report for Operable Units 5-05 and 6-01 (SL-1 and BORAX-I Burial Grounds)," Idaho Falls, Idaho, Lockheed Idaho Technologies Company, Idaho National Engineering Laboratory.

Hull, L., 2001, Interoffice Memorandum to Theodore J. Meyer, Doug Burns, Swen Magnuson, and James McCarthy, "Approach to Simulation of Actinides for the OU 7-13/14 BRA," LCH-04-01, April 4, 2001, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Bechtel BWXT Idaho, LLC.

Hull, L., and M.N. Pace, 2000, *Solubility Calculations for Contaminants of Potential Concern at OU 7-13/14*, INEEL/EXT-2000-00465, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory.

INEEL, 2003, "Statement of Work for the ICDF Complex Implementation Project," Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory.

Kudera, D. E. and B. W. Brown, 1996, *Volatile Organic Compounds Disposed of from 1952 Through 1983 at the Radioactive Waste Management Complex: Quantities Forms, Release Mechanisms and Rates*, ER-WAG7-90, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company.

LITCO, 1995a, *A Comprehensive Inventory of Radiological and Nonradiological Contaminants in Waste Buried in the Subsurface Disposal Area of the INEL RWMC During the Years 1952 – 1983*, INEL-95/0310, Rev. 1, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Co.

LITCO, 1995b, *A Comprehensive Inventory of Radiological and Nonradiological Contaminants in Waste Buried in the Subsurface Disposal Area of the INEL RWMC During the Years 1984 – 2003*, INEL-95/0135, Rev. 1, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Co.

Magnuson, S.O. and A.J. Sondrup, 1998, *Development, Calibration, and Predictive Results of a Simulator for Subsurface Pathway Fate and Transport of Aqueous- and Gaseous-Phase Contaminants in the Subsurface Disposal Area at the Idaho National Engineering and Environmental Laboratory*, INEEL/EXT-97-00609, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company.

Martian, P., 1995, *UNSAT-H Infiltration Model Calibration at the Subsurface Disposal Area, Idaho National Engineering Laboratory*, INEL-95/0596, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company.

Martian, P. and A. J. Sondrup, 1995, *SDA Buried Drum Failure Rate Data Compilation*, Engineering Design File ER-WAG7-68, INEL-95/126, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company.

McElroy, D.L., 1993, *FY-1993 Soil Moisture Monitoring Results at the Radioactive Waste Management Complex of the Idaho National Engineering Laboratory*, EGG-WM-11066, EG&G Idaho, Inc., Idaho Falls, Idaho.

McKinley, K.B. and J.D. McKinney, 1978, “Initial Drum Retrieval, Final Report,” TREE-1286, Idaho Falls, Idaho, Idaho National Engineering Laboratory, EG&G Idaho, Inc.

Miller, E.C. and M.D. Varvel, 2001, *Reconstructing the Past Disposal of 743-Series Waste in the Subsurface Disposal Area for Operable Unit 7-08, Organic Contamination in the Vadose Zone*, INEEL/EXT-01-00034, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Bechtel BWXT Idaho, LLC.

Nagata, P.K. and J. Banaee, 1996, *Estimation of the Underground Corrosion Rates for Low-Carbon Steels; Types 304 and 316 Stainless Steels; and Inconel 600, 601, and 718 Alloys at the Radioactive Waste Management Complex*, INEL-96/098, Idaho Falls, Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Co.

Nimmo, J.R., S.M. Shakofsky, J.F. Kaminsky, and G.S. Lords, 1999, *Laboratory and Field Hydrologic Characterization of the Shallow Subsurface at an Idaho National Engineering and Environmental Laboratory Waste-Disposal Site*, U.S. Geological Survey Water-Resources Investigations Report 99–4263 (DOE-ID-22163), Idaho Falls, Idaho, U.S. Geological Survey.

Oztunali, O. I. and G. W. Roles, 1985, “Update of Part 61 Impacts Analysis Methodology,” NUREG/CR-4370, Vol. 1, U.S. Nuclear Regulatory Commission.

Perkins, K.S., 2000, *Hydrological and Meteorological Data for an Unsaturated Zone Study Near the Radioactive Waste Management Complex, Idaho National Engineering Laboratory, Idaho 1997-1999*, U.S. Geological Survey Open File Report 00-248, (DOE-ID-22171), Idaho Falls, Idaho, U.S. Geological Survey.

Pittman, J.R., 1995, *Hydrological and Meteorological Data for an Unsaturated Zone Study Near the Radioactive Waste Management Complex, Idaho National Engineering Laboratory, Idaho 1988 and 1989*, U.S. Geological Survey Open File Report 95-112, (DOE-ID-22118), Idaho Falls, Idaho, U.S. Geological Survey.

Pittman, J.R., 1989, *Hydrological and Meteorological Data for an Unsaturated Zone Study near the Radioactive Waste Management Complex of the Idaho National Engineering Laboratory, Idaho 1985-1986*, U.S. Geological Survey Open File Report 89-74, (DOE-ID-22079), Idaho Falls, Idaho, U.S. Geological Survey.

Rood, A. S., and M. K. Adler Flitton, 1997, *Identification of Metal Alloys for Inclusion into Corrosion Test*, RWMC-EDF-931, INEL/INT-97-00038, Idaho Falls, Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Co.

Salomon, H., 2001, *Field Sampling Plan for Monitoring of Type B-Probes in Support of the Integrated Probing Project Operable Unit 7-13/14*, INEEL/EXT-2000-01435, Rev. 0, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Bechtel BWXT Idaho, LLC.

Sullivan, T.M., 1993, "DUST-Disposal Unit Source Term Data Input Guide," NUREG/CR-6041, BNL-NUREG-52375, Upton, New York, Brookhaven National Laboratory.

Sullivan, T.M., 1992, "Development of DUST: A Computer Code That Calculates Release Rates from LLW Disposal Unit," BNL-NUREG-47118, Upton, New York, Brookhaven National Laboratory, U.S. Nuclear Regulatory Commission.

Thompson, R.J. 1972, "Solid Radioactive Waste Retrieval Test," ACI-120, Allied Chemical Corp.

USGS, 1999, "Review of the Transport of Selected Radionuclides in the Interim Risk Assessment for the Radioactive Waste Management Complex, Waste Area Group 7 Operable Unit 7-13/14, Idaho National Engineering and Environmental Laboratory, Idaho (Draft)," U.S. Geological Survey Administrative Report, Prepared for Idaho Operations Office U.S. Department of Energy under IAG DE-A107-97ID13556, Idaho Falls, Idaho, U.S. Geological Survey.

Varvel, M.D., 2001, *Mass Estimates of Organic Compounds Buried in the Subsurface Disposal Area for Operable Unit 7-08 and 7-13/14*, INEEL/EXT-01-00277, Rev. 0, Idaho Falls, Idaho, Idaho National Engineering and Environmental Laboratory, Bechtel BWXT Idaho, Inc.

3.3 Infiltration Ponds and Ditches

Wastewater generated by INEEL operations has been discharged since the early 1950s to a series of unlined ponds and ditches. These unlined ponds and ditches have been used at most of the major INEEL facilities for disposal of wastewater that was generated from INEEL facility operations. Wastewater disposed to these surface-disposal facilities has contained small concentrations of dissolved radioactive and/or nonradioactive chemical constituents.

The release of contaminants from ponds and ditches is controlled by the hydraulic and geochemical properties of the geologic materials that make up the pond or ditch bottom. Typically, the hydraulic conductivity of pond bottom materials is large and wastewater disposed to ponds and trenches rapidly infiltrates. Under these conditions, the release of wastewater and chemically conservative contaminants to the underlying vadose zone nearly approximates the rate of discharge to the ponds. Less conservative constituents may be sorbed to pond-bottom sediments. These sorbed constituents may provide a later source of release to the vadose zone under altered geochemical conditions. Shallow perched groundwater bodies commonly form in alluvial deposits directly beneath the pond in response to infiltration. These perched groundwater bodies also constitute a point of attenuated contaminant source-term release to the vadose zone.

3.3.1 Inventory

Infiltration ponds, ditches, and other surface-disposal facilities have been used routinely since the early 1950s to dispose of aqueous wastes at TAN, NRF, TRA, INTEC, and ANL-W. Subsequent sections will summarize the operational histories of those surface-disposal facilities that have constituted a point of release of hazardous and radioactive contaminants to the subsurface.

The summary also includes a description of the inventory of contaminants as available in remedial investigation/feasibility study (RI/FS) documents. Where available, discussion focuses on the contaminants identified in previous RI/FS documents as risk drivers, rather than those screened out in the process. The summary will describe any estimates of residual source terms in sludge or sediments that may remain as a secondary source for contaminant release. Sites are identified where residual contaminated soils have been remediated so the inventory for this source term category is not overestimated.

Perched-water zones associated with surface-disposal facilities are described from the perspective of near field chemistry and as potential secondary source term. Source-term release mechanisms will be discussed as described in RI/FS documents.

3.3.1.1 Test Area North

Contaminant source terms to the subsurface at TAN are described in the TAN Remedial Investigation Report (Kaminsky et al. 1994). The TSF-05 injection well was the primary source of subsurface contamination until 1972. Subsequently, the TSF-07 disposal pond and associated perched water in sediments have provided a point of release for radioactive, inorganic-chemical and organic-chemical contaminants to the subsurface (Sorenson et al. 1996). Water and dissolved contaminants infiltrate through the pond bottom and migrate downward through the vadose zone to the SRPA.

TSF-07 Disposal Pond—The TSF-07 disposal pond is an unlined, diked area encompassing approximately 35 acres. The disposal pond, first put into service in 1972, historically received wastewater containing low-level radioactive waste, and continues to receive cold process water and treated sewage effluent under a Wastewater Land Application Permit. The sections that follow summarize the operational history and contaminant inventories that have been disposed to this pond.

3.3.1.1.1 Operational History

The TSF-07 Disposal Pond was constructed during 1971-72 to replace the TSF-05 injection well. Since 1972, wastewater from the sewage treatment plant and other sources has been disposed to the TSF-07 pond from the common sump (see Figure 3-10).

TSF Sewage Treatment Plant – The TSF sewage treatment plant is a trickling filter sewage treatment facility that treats sanitary wastewater and other liquid wastes prior to transfer to the common sump and eventual disposal in the TSF-07 pond. Wastewater streams to the sewage treatment plant have included hazardous liquids that were generated by TAN facilities. From 1957 to 1985, liquid wastes generated by the TAN Maintenance and Paint Shops were disposed to the sewage treatment system. This wastewater contained chemical cleaners, paint thinners, and paint strippers. The TSF sewage treatment plant also may have received other potentially hazardous wastes that were generated by operations related to the Actuator Building, Hot Cell Annex, and 607 Assembly and Maintenance Shops (Kaminsky et al. 1994). Lightly radioactive borated wastewater was transferred from LOFT and disposed to the TSF-07 pond.

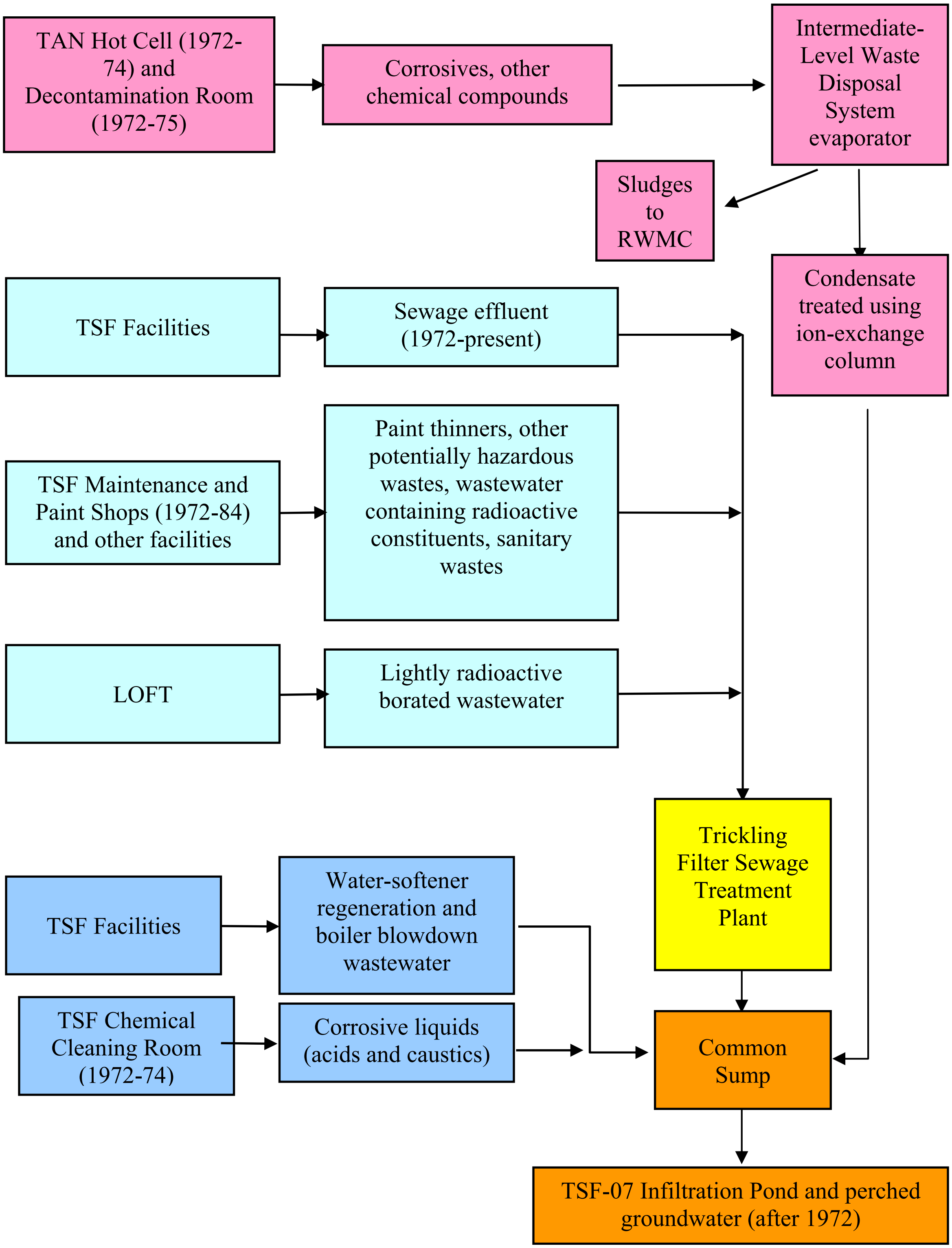
A series of three clarifier pits is used to remove suspended solids from wastewater streams prior to treatment and disposal. The clarifier pits are rectangular, concrete settling basins with a total capacity of 3,158 gal (Kaminsky et al. 1994).

TSF Cleaning Room – Wastewater was generated by the TSF Cleaning Room (pipe laundry) until 1974. Liquids were corrosive and consisted of acids and caustics. This wastewater was transferred to the common sump during 1972-74 and was disposed to the TSF-07 pond.

TSF Intermediate-Level Waste Disposal System – Other sources of wastewater disposed to the TSF-07 pond included condensate from two evaporator systems connected to the modified Intermediate-Level Waste Disposal System during 1972-75. Intermediate-level wastes were treated in an evaporator and the separated solids were transferred to RWMC for disposal. Condensate resulting from this evaporation process then was passed through an ion-exchange column to remove radioactive constituents. This condensate was transferred to the common sump for disposal in the TSF-07 pond. During the early 1970s, a leak in the evaporator steam jacket allowed radioactive contamination to be discharged to the process waste system. This leak took place during transition from the injection well to the TSF-07 pond and it is unclear to which system contamination was discharged. The evaporator system was discontinued in 1975 because of operational difficulties and subsequent wastes were transported by tanker truck to INTEC for treatment.

Other Sources of Wastewater – Wastewater from regeneration of water softeners and boiler blowdown operations has been disposed to the TSF-07 pond since 1972. This wastewater contained chemicals used to regenerate ion-exchange columns and as corrosion inhibitors.

Perched Water Body—Perched water has been detected in the sediments beneath the TSF-07 pond. Contaminants in perched water constitute an attenuated source of contaminant release to the vadose zone. Release is controlled by continued disposal to the pond, hydraulic properties of the sediments, and geochemical characteristics of the contaminant and sediments.



(Release to the vadose zone is controlled by the hydraulic and geochemical properties of surficial alluvium and the pond bottom materials)

Figure 3-10. Contaminant waste streams contributing to the Test Area North TSF-07 disposal pond source term.

Contaminant inventory

During routine sampling of water supply wells in 1987, trichloroethene (TCE) was detected in the groundwater at TAN. Subsequent investigations delineated an extensive aqueous TCE plume at the facility. It was determined that injection of liquid and sludge wastes (both organic and radioactive) into well TSF-05 was the most probable source of TCE in the nearly 2-mi long plume (Kaminsky et al. 1994). In addition to TCE, six other COCs were identified: tetrachloroethene (PCE); 1,2-dichloroethene (DCE); strontium-90 (Sr-90); tritium (H-3); cesium-137 (Cs-137); and uranium-234 (U-234).

The use of the TSF-07 disposal pond has provided an additional source of contaminants to the subsurface and has locally affected water levels and direction of groundwater flow. The inventory of COCs discharged to the pond includes radionuclides and other chemical constituents.

Radionuclides released to the TSF-07 disposal pond during 1972-85 are listed in Table 3-9. Three of these radionuclides were identified as COCs. These included strontium-90 (3.923×10^{-2} Ci), tritium (1.072×10^{1} Ci), cesium-137 (2.748×10^{-2} Ci).

Table 3-9. Radionuclides released to the TSF-07 pond during 1972-1985.

Radionuclide	Curies Released
Cobalt-58	4.063×10^{-2}
Cobalt-60	1.973×10^{-2}
Cesium-134	2.588×10^{-3}
Cesium-137	2.748×10^{-2}
Hafnium-181	2.046×10^{-3}
Molybdenum-90	1.228×10^{-2}
Ruthenium-106	1.915×10^{-5}
Strontium-89	3.358×10^{-3}
Strontium-90	3.923×10^{-2}
Tritium	1.072×10^{1}
Unidentified alpha	4.566×10^{-3}
Unidentified beta and gamma	2.124×10^{-1}
Yttrium-88	2.757×10^{-4}
Yttrium-90	3.923×10^{-2}
Total	**11.124**

The TSF clarifier pits are used to remove suspended solids from wastestreams flowing into the sewage treatment plant. Pit bottoms contain as much as 11 in. of cohesive sludge. Organic compounds were detected in these sludges (Kaminsky et al. 1994). Sludge samples from May 1988 and June 1989 contained acetone, methylene chloride, methyl ethyl ketone, toluene, and xylene. The amounts of these organic chemicals that may have leached from the pit sludge and reached the disposal pond is not known. These organic compounds have not been detected in the TSF-07 disposal pond. The volume of these organic chemicals may have been reduced by bacterial degradation or volatilization in the trickling filter at the sewer plant.

Water samples were collected from two wells completed in perched water-bodies in sediments below TSF-07 in 1990 and 1991 (Kaminsky and others, 1994). Selected trace elements were present in concentrations exceeding Environmental Protection Agency (EPA) drinking water standards in both wells on several occasions. Gross alpha activities ranged from 3.2 to 120 pCi/L. Gross beta activities ranged from 0 to 120 pCi/L. Strontium-90 activities ranged from 1 to 136 pCi/L. It is believed that metals and radionuclides present in perched water generally are being attenuated through adsorption and cation-exchange in pond bottom sediments and in the vadose zone and do not pose a threat to the SRPA. Organic compounds were not detected in perched water.

3.3.1.2 Test Reactor Area

Wastewater disposed at the TRA has contained radioactive and chemical contaminants. Contaminant release to the vadose zone at the TRA has occurred through several unlined infiltration ponds (Figure 3-11). These contaminant sources are described in the Comprehensive RI/FS for the TRA (Burns and others 1997). Infiltration of wastewater through pond bottoms has provided a mechanism for release of these contaminants to the subsurface. Water and contaminants have migrated downward from these percolation ponds, forming a series of shallow and deep perched water bodies. More reactive contaminants may be temporarily or permanently attached to sites within the vadose zone.

A series of infiltration ponds has been used at the TRA since 1952 to dispose wastewater containing radioactive and other chemical constituents. TRA infiltration ponds have included the warm-waste, chemical-waste, cold-waste, and sanitary waste ponds.

3.3.1.2.1 Operational history

Warm-Waste Ponds—Low-level radioactive wastewater generated by test reactor facilities, including cooling tower blowdown water, wash water from hot cell drains, laboratory solutions, and floor drainage from the Advanced Test Reactor, was discharged to the warm-waste ponds during 1952-1993. The TRA warm-waste ponds consisted of three unlined cells that were constructed between 1952 and 1964. The first cell, 150 × 250 × 15 ft deep, was constructed in 1952 for disposal of all wastewater except sanitary wastewater. In 1957, a second cell, 125 × 230 × 15 ft deep, was constructed adjacent to the first cell because of decreased infiltration through the pond bottom from buildup of chemical precipitates and algae. The second cell was separated from the first cell by an earthen dike and received discharge from the first cell. In 1964, a third cell, 250 × 400 × 6 ft deep, was constructed next to the first two cells to alleviate continued pond bottom plugging problems.

During 1952 –62, all non-sanitary wastewaters were disposed in the warm-waste ponds. After 1964, the warm-waste ponds were used principally for disposal of wastewater containing radioactive constituents. Disposal was discontinued in 1993 when radioactive wastewater was diverted to lined evaporation ponds. Until 1979, the annual discharge averaged 24 million ft^3. Subsequently, the annual average was approximately 4 million ft^3. A total of about 708 million ft^3 of wastewater were disposed to the warm-waste ponds during their operational history (1952-93). In 1993, the warm-waste ponds were decommissioned and wastewater was discharged to lined evaporation ponds.

Radioactive wastewater was distributed to the warm-waste ponds through a concrete-lined retention basin. Throughout its operational history, leaks from this basin contributed to a shallow perched water body in the surficial alluvium.

Chemical-Waste Pond—The chemical-waste pond was constructed in 1962 for disposal of nonradioactive wastewater that contained industrial chemicals associated with ion-exchange columns and water softeners. The chemical-waste pond is an unlined disposal unit 170 ft × 170 ft × 15 ft deep, surrounded by 2 to 3 ft high berm. During 1962-94, an annual average of 3.2 million ft^3 of wastewater was disposed to the chemical-waste pond.

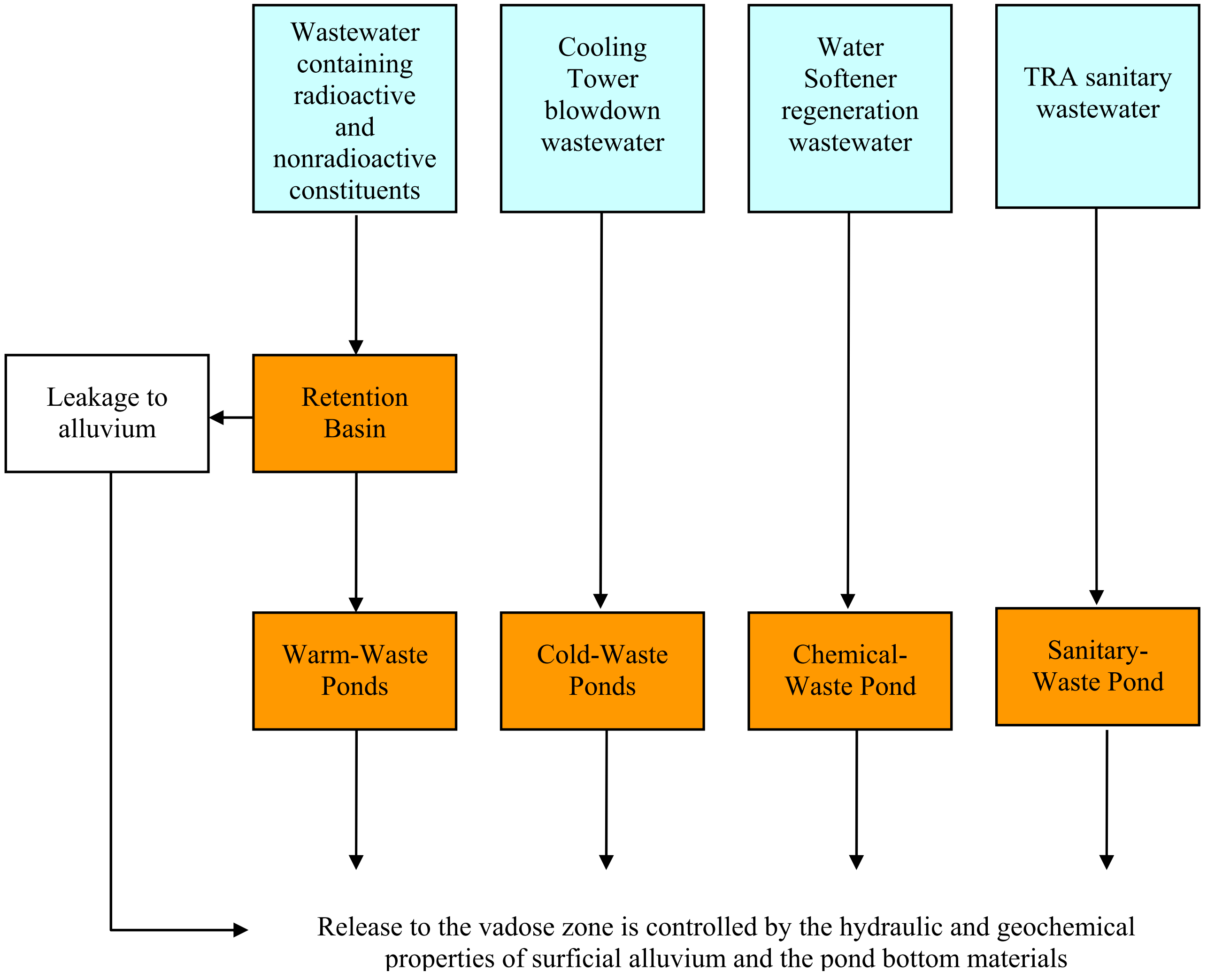


Figure 3-11. Contaminant waste streams contributing to Test Reactor Area infiltration-pond source term.

Cold-Waste Pond—The cold-waste pond was constructed in 1982 for disposal of cooling-tower blowdown water. This pond accounts for the second-largest source of water from surface-disposal at TRA. The cold-waste pond is an unlined pond composed of two cells, each 150 × 400 ft. The pond was constructed to replace the TRA injection well. The average annual discharge to the cold-waste pond is 30 million ft^3. Discharges alternate between the two cells of the pond on an annual basis.

Wastewater consists of cooling water from blowdown during reactor operations, discharge from air conditioning units, secondary system drains, floor drains, and other non-radioactive drains throughout TRA. Wastewater discharged into the CWP percolates through the soil and merges with subsurface water from adjacent ponds. The combined waters form the shallow and deep perched water system beneath TRA.

Sewage-Leach Ponds— The sewage leach ponds are two unlined pond cells constructed to accept all sanitary wastewaters generated at TRA. The northern cell was constructed in 1965 and measures approximately 250 ft × 50 ft × 14 ft deep. The southern cell was constructed in 1950 and measures approximately 250 ft × 80 ft × 18 ft deep. This system has been used continuously since 1952 to receive effluent from TRA sanitary sewer drains. The average discharges range from 20 to 30 gpm. The total volume of wastewater disposed to the sewage-leach ponds through 1994 was 56 million ft^3, with an annual average of approximately 1.3 million ft^3.

Perched Water Bodies—Infiltration from wastewater disposal ponds has resulted in formation of shallow and deep perched water bodies in alluvium and underlying basalts in the immediate vicinity of the disposal ponds. Shallow perched water forms at the interface of the alluvium and the underlying basalt at a depth of approximately 50 ft bls. Deep perched groundwater, attributed to sedimentary interbeds or sediment infilling fractures in the interbedded basalt-sediment sequence, occurs at depths from 140 to 200 ft. The areal dimensions of the deep perched water body are approximately 6,000 × 3,000 ft.

3.3.1.2.2 Contaminant inventory

The RI/FS (Burns et al. 1997) addressed perched groundwater and water in the SRPA, although the RI/BRA determined that neither presents risks exceeding 10^{-4} to receptors for considered scenarios. Contaminants discharged to the TRA ponds since the early 1950s have included radionuclides, inorganic chemicals, and organic chemicals. Concentrations of two contaminants, tritium and chromium, in groundwater exceeded MCLs, but these concentrations were predicted to decrease to less than MCLs within 20 years. Other radionuclides and chemicals disposed to infiltration ponds at TRA included strontium-90 (Sr-90), cesium-137 (Cs-137), cobalt-60 (Co-60), chromium-51 (Cr-51), sodium (Na), chloride (Cl), and sulfate (SO_4^{2-}).

The total activities of tritium, Sr-90, Cs-137, Co-60, and Cr-51 as disposed to the TRA warm-waste ponds during 1952-90 and the total activity corrected for half-life are shown in Table 3-10. The more conservative radionuclides probably were transported to the aquifer, with attenuation in the multiple perched-water bodies beneath the ponds. Less conservative radionuclides are believed to have sorbed to sediments or decayed.

During 1952 to 1964, approximately 24,222 lb of Cr^{6+} are estimated to have been disposed to the warm-waste ponds in cooling water discharges. Detection of chromium in perched water and in the SRPA in 1966 resulted in abandonment of chromates as a corrosion inhibitor.

Table 3-10. Radionuclide disposal to the Test Reactor Area warm-waste ponds and correction for radioactive decay (from Doornbos et al. 1991).

Radionuclide	Total activity released as of January 1991 (units)	Corrected for radioactive decay (units)
Tritium	9,880	3,700
Strontium-90	111	67.8
Cesium-137	157	91.3
Cobalt-60	480	19.1
Chromium-51	11,900	26.5

Chromium and selected radionuclides strongly adsorb to surficial sediments in the warm-waste pond bottoms. Analysis of pond-bottom material indicates that contamination generally extends to depths less than 2 ft. Trivalent chromium concentrations in sediment were 340 mg/kg, indicating that some of the hexavalent chromium may have been converted to the less trivalent form and remained in the pond bottom sediments. Cesium-137 activities ranged from 2.9 to 39,400 pCi/g. Cobalt-60 activities ranged from 0.2 to 27,100 pCi/g.

The warm-waste ponds were decommissioned in 1993. Decommissioning operations included excavation of sediments from sidewalls and base of 1964 cell and disposal into the 1952 cell. Materials from a 1992 windblown-soil cleanup activity were also placed into the 1952 cell to bring it to the level of the land surface. The cell was covered with 1 ft of clean soil and was graded to a 2% slope. Sidewall sediments from the 1957 cell were scraped to the base of the cell and a contaminated wooden structure associated with the warm-waste ponds was demolished and placed into the cell.

Contaminated soils from other areas also were placed into cells for disposal. These soils included 788 yd^3 from ANL-W that contained 800 pCi/g of Cs-137, 1,178 yd^3 from the BORAX ditch that contained 95.4 pCi/g of Cs-137, 1,947 yd^3 from the TRA-North Storage Area that contained 7,755 pCi/g Sr-90, 913 pCi/g Eu-152, 146 pCi/g Eu-154, 684 pCi/g Am-241, 404 pCi/g Cs-137, and 74 pCi/g Co-60. Disposed soils also included 2,737 yd^3 from TAN Area B that contained 75 pCi/g Cs-137 and 160 pCi/g Sr-90, and 2,208 yd^3 from TSF-26 that contained 39 pCi/g Cs-137 and 405 pCi/g Sr-90.

Pond bottom sediments contain concentrations of trace elements and other contaminants. These include barium averaging 300 mg/kg, with a maximum concentration of 3,830 mg/kg, and mercury, averaging 0.50mg/kg, with a maximum concentration of 133 mg/kg, concentrated in shallow sediments and suspected to be insoluble or relatively insoluble. Sediments also contain PCBs, (0.33 mg/kg) detected in surface sediments but not in subsurface samples.

Unlined service-waste disposal ponds at INTEC have been used to dispose of aqueous wastes containing radioactive and chemical contaminants. Contaminant releases from these ponds to the vadose zone at INTEC are described in the Comprehensive RI/FS for the ICPP OU 3-13 (DOE-ID 1997). Infiltration of wastewater through pond bottoms has provided a mechanism of release of these contaminants to the subsurface (Figure 3-12). Water and contaminants have migrated downward from these percolation ponds, forming a series of shallow and deep perched water bodies. More reactive contaminants may be temporarily or permanently attached to sites within the vadose zone. Wastewater discharge began in 1984 to two service-waste ponds. A third service-waste pond replaced these ponds in 2002. Release to the vadose zone is controlled by the hydraulic and geochemical properties of surficial alluvium and the pond bottom materials.

3.3.1.2.3 Operational history

Service-Waste Percolation Ponds—The service-waste ponds were constructed in 1984 directly south of INTEC (then known as the Idaho Chemical Processing Plant). The purpose of these ponds was to replace the CPP-23 injection well for disposal of service wastewater. These ponds were designed to allow infiltration of water to the subsurface. Perched groundwater bodies formed beneath the service-waste ponds in response to infiltrating wastewater. Perched water contained radionuclides and other chemical constituents and attenuated release of these contaminants to the vadose zone.

In 2002, a new service-waste percolation pond was constructed several miles to the east of INTEC to replace the two ponds. This replacement pond was part of a plan to limit local recharge in response to continued wastewater infiltration in proximity to contaminated areas in the vadose zone beneath INTEC.

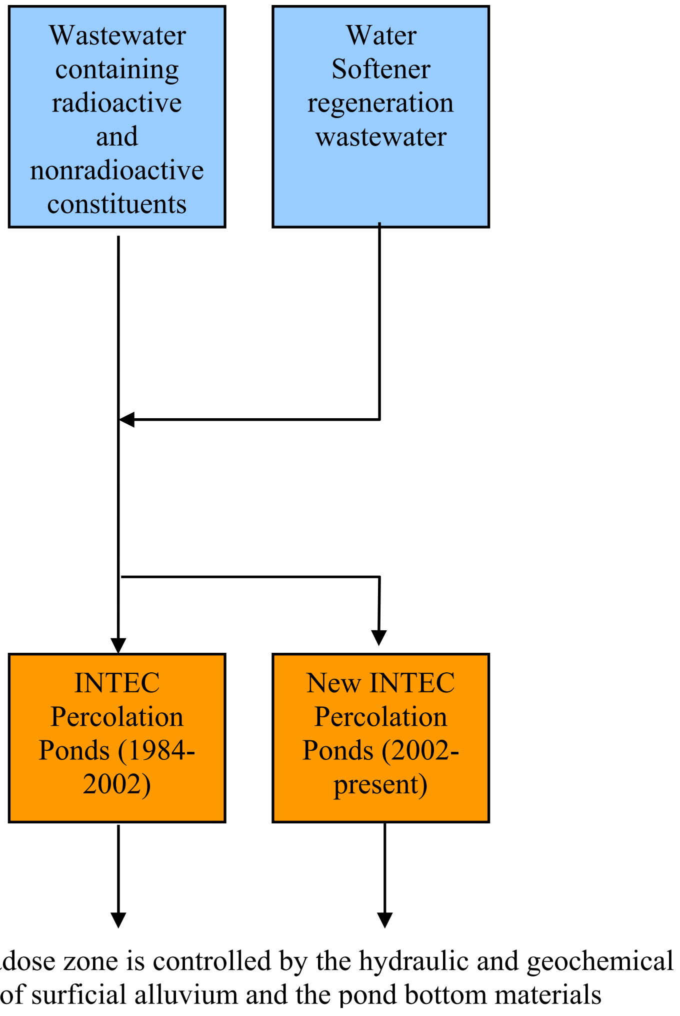


Figure 3-12. Contaminant waste streams contributing to the Idaho Nuclear Technology and Engineering Center percolation waste stream.

Other Sources—The CPP-603 fuel storage basins have been operational since 1952. These basins store spent fuel assemblies prior to processing of the assemblies. The basins consist of three reinforced-concrete, unlined fuel-storage basins connected by a transfer channel. Leakage from the fuel storage basins has been estimated to be from 500 to 700 L/day (DOE-ID 1997a).

3.3.1.2.4 Contaminant inventory

Radioactive, inorganic chemical, and organic chemical contaminants derived from INTEC operations have been disposed to the INTEC service-waste ponds. Based on pond-sediment characterization data and the chemistry of the waste streams to these disposal points, these two sources do not contribute significantly to the inventory of contaminants. However, water movement from these ponds may affect the migration of contaminants in other areas of the INTEC.

Contaminants released to the subsurface since the early 1950s have included radionuclides, inorganic chemicals, and organic chemicals. Table 3-11, modified from Table 5-42 of Appendix F, INTEC Remedial Investigation report (DOE-ID 1997a), summarizes the total mass or activity of contaminants of potential concern and the point of disposal. Masses and activities presented in Table 3-11 are based on RWMIS and Industrial Waste Management Information System (IWMIS) data and, in several cases, on estimates using limited analytical data. This table provides a comparison of the magnitude of radioactive and trace-elements discharged at INTEC. Tritium was the only constituent of consequence disposed to the service-waste ponds, with a total activity of about 1,000 Ci.

Leaking water from the CPP-603 fuel storage basins contained concentrations of radionuclides. The principle contaminant in tank leakage water is cesium-137. This inventory probably is strongly sorbed to alluvial sediments beneath the storage basins.

Table 3-11. Summary of total mass or activity for contaminants of concern and contamination sources at the Idaho Nuclear Technology and Engineering Center.

Contaminant of Potential Concern	Units	Known Release		Service Waste		Soil Contamination	Total
		Tank Farm	Other	Injection Well	Service-Waste Ponds		
Arsenic	Kg	0	0	0	0	457	457
Chromium	Kg	16.0	0	0	0	120	136
Mercury	Kg	28.1	0	400	0	596	1,024
Americium-241	Ci	110	0	0.123	0	0.905	111
Cobalt-60	Ci	68.4	0	1.24	0.0112	106	176
Cesium-137	Ci	26,800	309	25.8	0.534	2,680	29,815
Tritium	Ci	18.5	378	20,100	999	0	21,496
Iodine-129	Ci	0.00705	0	1.39	0.0820	0.0389	1.52
Neptunium-237	Ci	0.216	0	1.07	0	0.133	1.42
Total Plutonium	Ci	1,180	0	0.822	0.0692	10.2	1,191
Strontium-90	Ci	18,000	309	24.3	0.295	1,110	19,444
Technetium-99	Ci	2.58	0	0	0	0.106	2.69
Total Uranium	Ci	0.747	0	0.269	0.0701	0.940	2.03

3.3.1.3 ARA/PBF

3.3.1.3.1 Operational history

- ARA-I Chemical Evaporation Pond was a shallow, unlined pond that was used to dispose of wastewater generated in the ARA-1 Shop and maintenance building. This pond was operated during 1970-1988. Wastewater contained small quantities of radionuclides, acids, bases, and VOCs.

- ARA-III Radioactive Leach Pond was an unlined pond in a natural depression with dimensions about 377 × 164 ft. This pond received low-level waste from reactor research operations as well as facility surface runoff during 1959-1987.

- PBF SPERT-I Leach Pond was a 15 × 45 ft diked, unlined pond that received sporadic amounts of radiologically contaminated and nonradioactive overflows from the SPERT-1 reactor pit during 1955-1964.

- PBF SPERT-II Leach Pond was an unlined pond with dimensions about 230 × 167 ft. During 1959-1964, this pond received effluent from a demineralizer, water softener waste, emergency shower drain water, and discharges from reactor-building floor drains.

- PBF SPERT-III Large Leach Pond was used during 1958-1968 to discharge primary coolant water and sump pump discharges.

- PBF SPERT-IV Leach Pond was an unlined pond used from 1961 to the early 1980s to receive effluent from the SPERT-IV reactor. Effluent included radioactively contaminated wastewater, emergency shower water, and primary coolant effluents.

- PBF SPERT-IV Lake was a 6-million gal surface impoundment used in 1961-1970 to receive uncontaminated secondary cooling water from the SPERT-IV reactor and in 1985-1992 to receive uncontaminated effluent from the Three Mile Island studies.

3.3.1.4 BORAX/EBR-1

A leach pond and drainage ditch were constructed and used for disposal of wastewater from the BORAX II through V reactor experiments during 1954-1964. All of the BORAX facilities were remediated, the subsurface structures were entombed with soil and concrete, and the surface structures were excavated and covered. Wastewater discharges from the various BORAX reactors ended in 1964 (DOE-ID 2001). Because no significant local sources of recharge exist and there are no production wells, water levels in the BORAX area are unaffected by anthropogenic recharge or discharge.

3.3.1.5 ANL-W

Potential surface-release sites at ANL-W include wastewater structures and leaching ponds (DOE et al. 1998).

3.3.1.5.1 Description and operational history

Industrial Waste Pond—The industrial waste pond, located at the northwest corner of ANL-W, has been operational since 1964. The pond is an unlined evaporative seepage pond that is fed by a system of drainage ditches (Argonne National Laboratory 2002). Water disposed to the pond prior to shut down of the EBR-II reactor in 1994 predominantly consisted of blowdown effluent from the main cooling tower. The average discharge to the pond during 1979 through 1994 was 39 million gal/yr; discharge rates ranged from 1.42 to 4.22 million gal/month (Argonne National Laboratory 2002). Since then, discharge to the pond has been reduced greatly. Remaining sources of discharge to the pond have consisted of blowdown effluent from the auxiliary cooling tower and auxiliary boiler, water from air-conditioning systems, and non-contact cooling water from other sources. Reductions in discharge to the pond have resulted in drying of the pond.

ANL-W Ditches and Canals—Ditches have been used to transport surface water from runoff and facility drains at ANL-W to the industrial waste pond. These unlined ditches, which include the main cooling tower blowdown ditch, ditches A, B, and C, the industrial waste discharge ditch, and the interceptor canal, are described below.

The main cooling tower blowdown ditch runs north to the industrial waste pond. This unlined ditch is approximately 700-ft long. During 1962 through 1996, the ditch was used to convey industrial wastewater from the cooling tower to the industrial waste pond. Contaminants principally consisted of water treatment chemicals derived from regeneration of ion exchange resin beds. During 1962 through 1980, contaminants included a chromate-based corrosion inhibitor.

Ditch A conveys industrial wastewater from the EBR-II power plant auxiliary cooling tower and storm water runoff to the main cooling tower blowdown ditch. Wastewater from this unlined ditch eventually flows to the industrial waste pond (DOE et al. 1998). Ditches B and C have been used to convey storm water runoff and wastewater from the power plant and fire station to the industrial waste pond through the cooling tower blowdown ditch (DOE et al. 1998).

The industrial waste discharge ditch is approximately 500 ft long (DOE et al. 1998). The unlined ditch receives industrial wastewater derived from cooling water, photo processing wastes (containing photo developers, fixers, stabilizers, and acids), and retention tank overflows containing ethanol, sodium hydroxide, and some radionuclides.

The interceptor canal has been used to convey industrial waste to the industrial waste pond (DOE et al. 1998). This unlined canal also is used to divert surface runoff. Industrial wastewater is delivered to this canal through two pipes, one from the cooling tower and the other from the industrial waste lift station. That pipeline also has been used to convey radioactive wastes to the EBR-II leach pit. Discharge of industrial wastes and cooling tower blowdown water was discontinued in 1973 and 1975, respectively.

Sanitary Sewage Lagoons—The sanitary sewage lagoons are located north of ANL-W. These three lagoons were constructed in 1965 and 1974 and cover approximately 2 acres. The lagoons receive most of the sanitary wastewater generated at ANL-W. They also have received some photo processing solutions. The bottoms of these lagoons are sealed with bentonite.

Leach Pit—The EBR-II leach pit was constructed into basalt and used to dispose of liquid waste containing cooling tower blowdown, sanitary effluent, cooling condensates, and radioactive effluent until 1973 (DOE et al. 1998). Approximately 90,000 gal of wastewater were disposed annually.

Contaminant transport in the subsurface occurs as contaminant-bearing wastewater, which contains contaminants that infiltrate pond, ditch, and pit bottoms. This transport mechanism requires water volumes sufficient to sustain flow through the vadose zone. Sorption tends to inhibit transport; these processes are evident in subsequent discussion of contaminant distribution in the subsurface at ANL-W.

3.3.1.5.2 Contaminant inventory

Wastewater in the main cooling tower blowdown ditch contained water-treatment chemicals and hexavalent chromium from a chromate-based corrosion inhibitor. During 1962 through 1980, wastewater also contained concentrations of sulfuric acid and sodium hydroxide. Following a pH measurement of 1.86 in 1986, a neutralization tank was installed to treat water prior to ditch disposal.

Based on the presence of cesium-137 in sediments of the interceptor canal, some radioactive wastes were transported with the industrial wastes in the canal.

Based on contaminant concentrations in sediments of ditches A, B, and C, these other ditches transported wastewater containing mercury, chromium, and zinc. The industrial waste pond received wastewater from these different ditches. Contaminants present in pond sediments indicate that wastewater contained concentrations of cesium-137, trivalent chromium, mercury, selenium, and zinc (DOE et al. 1998).

Photo processing solutions were discharged to the sewage lagoons. Also, no known radioactive or hazardous substances have been released to the lagoons.

Wastewater disposed to the lined EBR-II leach pit contained 10.4 Ci of radioactivity derived primarily from cesium-137, strontium-90, cobalt-60, and uranium-238. Other sources of contaminants included the main cooling tower riser pits. The presence of arsenic, trivalent chromium, lead, and mercury in pit sediments indicates that these inorganic contaminants were contained in wastewater disposed to the pits.

3.3.1.6 NRF

Contaminant source terms at the NRF are described in the Final NRF Comprehensive RI/FS (Westinghouse Electric Corporation 1996). Water and dissolved contaminants infiltrate through the ditch and pond bottoms and migrate downward through the vadose zone to the SRPA. Surface disposal sites for NRF wastewater have included the Industrial Waste Ditch (IWD), S1W wastewater disposal facilities, A1W/Expended Core Facility (ECF) wastewater disposal facilities, and sewage lagoons (see Figure 3-13).

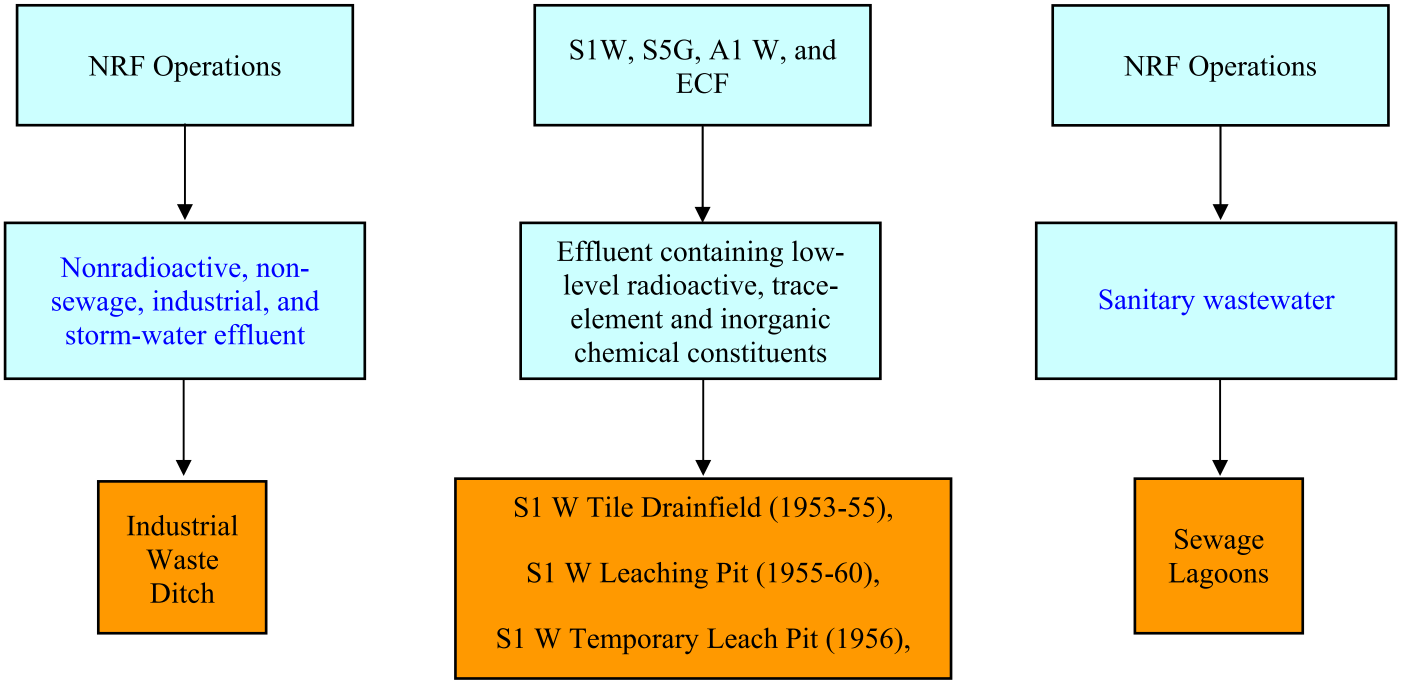
3.3.1.6.1 Operational History

Industrial Waste Ditch—The IWD is about 3.2 mi long and occupies parts of two meander channels. The IWD has been used for more than 30 years to dispose of nonradioactive, non-sewage, industrial, and storm-water discharge. Water presently flows in the first 0.5 to 1.2 mi. Prior to shutdown of the S1W, A1W, and S5G plants, water flowed through the entire reach of the ditch. Present discharges range from 50 to 150 gal/minute for an annual discharge of about 53 million gal (Westinghouse Electric Corporation 1996). An estimated 2.1 million gal of water evaporates annually from the ditch; the remaining ~51 million gal infiltrates.

S1W Wastewater Disposal Facilities—Beginning in 1953, low-level radioactive effluent was sent to a drainfield known as the Sl W Tile Drainfield (Unit 8-08-l1). The drainfield became plugged in 1955 (presumably with oil).

In 1955, the S1W Leaching Pit (Unit 8-08-12), constructed to the west and south of the original drainfield, consisted of an open trench 8- to 10- ft deep, 8-ft wide, and 50-ft long, excavated at the end of the discharge header to maintain an adequate discharge capacity. Unit 8-08-12A is the piping portion of the leaching field. Unit 8-08-12B is the trench or pit portion of the unit. In 1960, a new leaching bed was constructed for SlW discharges and the open trench was backfilled. The new leaching bed was constructed south of SlW in 1960 (SlW Industrial Waste Lagoons [Unit 8-08-14]). This leaching bed utilized seepage and evaporation of the water by allowing ponding within the bed. A similar bed was constructed adjacent to this bed in 1963. These discharge facilities (commonly known as the S1W Leaching Beds) received SlW effluent from 1960 to 1979.

The S1W Temporary Leaching Pit (Unit 8-08-13) was a basin constructed in 1956 for a one-time discharge of radioactive effluent that contained oil (Westinghouse Electric Corporation 1996). The basin was used to prevent the other leaching beds from being plugged because of the oil content in the liquid. The basin was filled in with the soil excavated immediately after the discharge.



(Release to the vadose zone is controlled by the hydraulic and geochemical properties of surficial alluvium and the ditch, pit, drainfield, and lagoon bottom materials)

Figure 3-13. Contaminant waste streams contributing to the Naval Reactors Facility ditch, pit, and lagoon source term.

A1W/ECF Wastewater Disposal Facilities—The Al W Leaching Bed (Unit 8-08-19) constructed to the west of NRF to receive the effluent from ECF and Al W, began operation in 1958 (Westinghouse Electric Corporation 1996). The Al W leaching bed was used sporadically after problems were noted in 1964, which were attributed to oil inhibiting the seepage capacity of the bed. Due to inadequate seepage capacity of the Al W leaching bed, the Sl W leaching beds started receiving ECF effluent in 1964 and Al W effluent in 1965. S5G operations began in 1965 and also sent low-level radioactive effluent to the Sl W leaching beds. During 1965 through 1972, the Al W leaching bed was used a few months of the year only for overflow. The last discharge to the Al W leaching bed was in May 1972.

Sewage Lagoons—Two sewage lagoons are located directly northeast of NRF. These infiltration lagoons have been used to dispose of an estimated 10 to 23 million gal/yr. Historical cross-contamination associated with the Old Sewage Treatment Plant, Seepage Basin Pump Out Area, and Sewage Lagoons. Other sites include the potentially radiologically-contaminated Old Ditch Surge Pond and S5G Basin Sludge Disposal Bed.

3.3.1.6.2 *Contaminant inventory*

Radioactive, inorganic, and organic chemicals were contained in wastewater disposed to the subsurface at NRF. Contaminants of potential concern at NRF primarily are radionuclides and trace elements (Westinghouse Electric Corporation 1996).

The prototype facilities at NRF generated wastewater effluent that contained small amounts of radioactivity as a result of normal operations. From June 1953 until April 1979, a total of 345 Ci of radioactivity were released to the NRF leaching pits, ponds, lagoons, basins, and drainfields. This activity was contained in almost 390 million gal of wastewater. Radioactive constituents with half-lives exceeding 5 years included cesium-137 (23.8 Ci), cobalt-60 (75.2 Ci), tritium (112.3 Ci), and strontium-90 (1.65 Ci). An additional 84 Ci were attributed to unidentified beta and gamma emitters.

In 1968, the total volume of effluent streams containing low concentrations of radioactive contaminants was minimized by removing nonradiological waste streams from discharge to the leaching beds and developing procedures for recycling radioactive liquid. By 1972, the ECF was a completely self-contained facility that recycled all radioactive liquids and no-longer-discharged low-level radioactive effluent to the environment. During the 1970s, recycling processes were instituted at all prototypes; the last low-level radioactive discharge was made in April 1979. The S1W and A1W/ECF wastewater disposal facilities were the primary source of contaminants during that period.

The IWD has been used primarily for disposal of wastewater from regeneration of ion-exchange units. During 1953-1992, regeneration wastewater contained 19.6 million lb of sulfuric acid, 5.1 million lb of sodium hydroxide, and 79 million lb of sodium chloride (Frederick et al. 1998). Lesser amounts of other chemical compounds and oil were disposed to the IWD from prototype-reactor and cooling-system waste streams. These included corrosion inhibitors containing chromium. Approximately 29 Ci of cesium-137 and cobalt-60 discharged to the IWD were derived from cross-contamination of reactor cooling systems.

Trace elements that were disposed to the IWD include chromium, lead, mercury, and silver. Inorganic constituents having elevated concentrations at NRF included sodium, chloride, and nitrate.

Hexavalent chromium was used as a corrosion and algal inhibitor in NRF reactor secondary water systems until 1965. In the 1950s and 1960s leakage from two spray ponds used to cool reactor secondary water released hexavalent chromium to the subsurface. In 1964, concentrations of hexavalent chromium were detected in NRF drinking water.

3.3.2 Factors Controlling Release

Chemically conservative contaminants (e.g., tritium) passing through pond bottoms are transported with infiltrating wastewater as vadose-zone flow. In certain cases, formation of a shallow perched zone beneath the pond attenuates release of contaminants.

Less chemically conservative contaminants (e.g., cesium-137) are immobilized in the sediment column beneath the pond. Some contaminants may be strongly sorbed to sediments; others may be released with changes in geochemical conditions in the shallow subsurface.

Some contaminants (e.g., chromium) are conservative in the hexavalent form. However, near-source chemical conditions may result in transformation in chemical species (e.g., trivalent chromium) that more strongly sorb to geologic media.

3.3.3 Current Uncertainties

- **TSF-07—**The contaminant inventory associated with the TSF-07 disposal pond is contained in sediments beneath the pond. Contaminants are radionuclides and selected trace elements. The range of contaminant concentrations in sediments from two coreholes indicates some uncertainty about the magnitude of inventory. It is believed that these contaminants do not pose a threat to the SRPA because they are immobilized through adsorption and cation-exchange in pond bottom sediments and in the vadose zone.

- **TRA Warm-Waste Ponds—**Contaminant inventory that was discharged to the warm-waste ponds included hexavalent chromium and selected radionuclides. The detection of hexavalent chromium in groundwater during the 1960s indicates that some of the chromium may have been transported directly to the SRPA. Sediment sample analyses indicate that some chromium probably was converted to trivalent chromium in the pond bottom and may have strongly adsorbed to sediments in the upper 2 ft of those sediments. Cesium-137 and cobalt-60 detections indicate that these radionuclides also have been sorbed strongly to sediments. The inventory remaining is not well identified.

- **INTEC Service-Waste Ponds—**Tritium was the only contaminant of consequence that was discharged to the service-waste ponds. The effect of continued wastewater disposal to the ponds on other near-surface inventories associated with contaminated soils resulted in construction of a new pond.

- **ARA/PBF ponds—**Soils beneath the seven aqueous surface-disposal sites associated with ARA and PBF probably contain small concentrations of sorbed radionuclides. The remaining inventory is unknown but the source-term contribution to contaminant transport in the vadose zone and SRPA is minimal because of the small quantities disposed and the immobility of sorbed constituents in surficial sediments.

- **ANL-W—**Radionuclides and trace elements detected in pond bottom sediments indicate that contaminant inventories remain in sediments. The inventory is uncertain, but is believed to be small and immobile.

- **NRF—**The amount of hexavalent chromium that may have been converted to trivalent chromium and sorbed to sediments is uncertain. Chromium concentration data in water from the SRPA indicates that whatever inventory remains is immobilized in surficial sediments.

- Research needs include:

  - Determination of actual inventories

  - Evaluation of potential for mobilization under different geochemical conditions.

3.3.4 References

Bartholomay, R.C., B.J. Tucker, L.C. Davis, and M.R. Greene, 2000, *Hydrologic Conditions and Distribution of Selected Constituents in Water, SRPA, Idaho National Engineering and Environmental Laboratory, Idaho, 1996 through 1998*, U.S. Geological Survey Water-Resources Investigations Report 00-4192 (DOE/ID-22167), U.S. Geological Survey.

Burns, D.E., K.M. Davis, S.C. Flynn, J.F. Keck, N.L. Hampton, A.H. Owen, and R.L. Van Horn, February 1997, *Comprehensive Remedial Investigation/Feasibility Study for the Test Reactor Area Operable Unit 2-13 at the Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Company*, DOE/ID-10531, Revision 0, U.S. Department of Idaho Operations Office Idaho Operations Office, Idaho Falls, Idaho.

Doornbos, M.H., et al., August 1991, *Environmental Characterization Report for the Test Reactor Area, Volumes I and II*, EGG-WM-9690, Rev. 0, Idaho National Engineering and Environmental Laboratory, EG&G Idaho, Idaho Falls, Idaho.

Frederick, D.B., A.E. Merritt, M. Fischer, and L.J. Campbell, 1998, Wastewater disposal at the Idaho National Engineering and Environmental Laboratory: State of Idaho INEEL Oversight Program.

Kaminski, J.F., K.N. Keck, A.L. Schafer-Perini, C.F. Hersley, R.P. Smith, G.J. Stormberg, and A.H. Wylie, 1994, *Remedial Investigation Final Report with Addenda for the Test Area North Groundwater Operable Unit 1-07B at the Idaho National Engineering Laboratory*, EGG-ER-10643, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Sorenson, K. S., Jr., A. H. Wylie, and T. R. Wood, 1996, *Test Area North Site Conceptual Model and Proposed Hydrogeologic Studies Operable Unit 1-07B*, INEL-96/0105, Revision 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Westinghouse Electric Corporation, 1996, “Final NRF Comprehensive Remedial Investigation/Feasibility Study, Waste Area Group 8, Naval Reactors Facility, Idaho Falls, Idaho, Part 1 – Remedial Investigation Report: Volumes 1 and 2,” prepared for the U.S. Department of Energy.

3.4 Contaminated Surface Soils

Contaminated soil sites are a common feature at many historical industrial facilities, including the INEEL. Past practices including surface discharges of contaminated liquids, fallout from air emissions, as well as accidental discharges resulting from failed underground piping, leaking underground basins and tanks, or simple spills, created numerous areas where surface and subsurface soils contained hazardous and radiological contaminants at varying concentrations.

Accidental releases generally were cleaned up at the time of the incident to protect facility workers. Those sites that were not cleaned up immediately, or that were a part of the facility operations, were incorporated into the environmental restoration program during the early 1990s. In general, the contaminated soil sites were evaluated as part of facility scale remedial investigations according to the CERCLA process. Most of those that were deemed to pose unacceptable risk have been remediated. In only a few cases are the CERCLA investigations ongoing.

Although literally hundreds of sites have been identified, only contaminated soils at the INTEC tank farm have been determined to have the potential to impact aquifer water quality. Section 3.4.1 provides a brief description of soil contamination at each of the major INEEL facilities, but is not intended to be an exhaustive compilation of soil sites. Inventory estimates for INTEC contaminated soils only are summarized and presented in Appendix A. Section 3.4.2 describes how the release of contaminants from soil sites has been conceptualized and modeled. Uncertainties associated with the contaminated soil conceptual models are discussed briefly in Section 3.4.3.

3.4.1 Inventory

A consolidated and comprehensive inventory of contaminated soil sites at the INEEL has not been developed and is not readily available. Such a compilation of contaminant mass or activity by site is beyond the scope of this document. Even if such a task were undertaken, the results would be of marginal utility, as conservative modeling conducted to date has not indicated a substantial groundwater risk resulting from contaminated soil sites at the INEEL. A general description of the various soil sites, and their contaminants, is provided below.

3.4.1.1 TAN—Nearly 100 contaminated soil sites have been previously investigated at TAN. A number of sites were contaminated by historic operations such as the TSF-07 Clarifier Pits, TSF-03 and WRRTF-01 Burn Pits, and the TSF-29 Acid Pond. There also exist larger contaminated soil areas associated with TSF surface ponds, drainage ditches, railroad tracks and areas used for equipment storage. There were occasional unintentional releases of radioactive and hazardous materials such as at the IET Valve Pit, and the mercury spill sites associated with the Heat Transfer Reactor Experiment (HTRE)-III engine. Most of these areas were remediated or screened out due to the low risk presented. One site that was investigated for potential impact to groundwater was the TSF Disposal Pond TSF-07 and associated sediments, which were found to be contaminated by low levels of radionuclides, corrosive wastewater, chromium and lead. The operation of the pond and the wastewater discharged to the pond are described in Section 3.3. Soil sampling conducted during the early 1990s indicated the presence of radionuclides and metals, but not organic constituents. The results of the TSF-07 sampling are provided in Medina and Hersley (1993). In the TAN RI/FS, investigators estimated the contaminant inventories at each site by averaging soil sample concentrations over specific volumes of contaminated soil. The conclusion of the remedial investigation was that the residual contamination in the sediments was not considered to be a significant source of groundwater contamination (Kaminsky et al. 1994).

3.4.1.2 TRA—The comprehensive RI/FS for TRA (Burns et al. 1997) describes 15 sites with surficial soil contamination. These soil areas were primarily associated with radiological waste tanks, including the Hot Waste Tanks, the former radionuclide-contaminated tank TRA-614, and Radiological Tanks at TRA-630. Sediments from the cold waste pond, the sewage leach pond, the warm waste pond and soils surrounding the warm waste retention basin were also evaluated in the RI/FS, as was the North Storage Area, a 2.5 acre area where equipment and materials were temporarily stored. The investigation even included the Hot Tree Site where an anonymous spruce was allegedly involved in the rupture of a low-pressure acidic transfer line originating from the gamma facilities building. The nature and extent of the contamination was described by a comprehensive soil sampling campaign. Radionuclides including Co-60, Cs-137, Eu-152, -154, Sr-90, Pu-238, -239/240, Am-241, and U-234, -235, -238 were commonly identified. In addition, a number of organic and heavy metal contaminants were also detected at varying concentrations. As common with most soil evaluations, the estimated dimensions of each contamination area and average contaminant concentrations were used to estimate contaminant inventories. Detailed tabulations of each contaminant concentration by area are provided in the RI/FS (Burns et al. 1997).

3.4.1.3 INTEC—The contaminant source term associated with contaminated soil sites at INTEC is presented in the comprehensive RI/FS (DOE-ID 1997). As with most INEEL soil sites, many of the INTEC soil sites could not be quantified based on known releases but rather were estimated based on sampling. Notable exceptions were three releases related to tank farm operations and two other releases related to facility drains. Contaminant source terms for these five known releases were estimated by the recorded volume of the liquid released and an examination chemical and radiological composition of each waste stream. The detailed release estimates for these five known releases are tabulated in Section 5 of the RI/FS (DOE-ID 1997). The known releases estimates for INTEC are considered to be bounding estimates.

The remaining contaminated soil sites at INTEC, more than 30, did not have detailed records available to derive exact sources, but were identified by sampling. For each of these sites, investigators calculated total mass or activity of contaminants present by estimating the contaminated soil volume and mass, estimating representative concentrations for each contaminant at each site, and multiplying the contaminant concentration by the soil mass. This is a typical approach applied at many DOE sites to derive conservative soil source terms for risk assessment purposes. The results of this analysis, presenting contaminant specific mass or activity for each of the soil sites is presented in Section 5 of the RI/FS.

The tank farm known releases present the vast majority of the Am-241, Cs-137, I-129, total Pu, Sr-90, and Tc-99. Most of the mercury, arsenic, and Co-60 are contained in the remaining soil sites. The total mass or activity of each of the COCs is presented in Appendix A. In summary, the known releases from the tank farm comprise the majority of the source term and, because the releases were recorded, are estimated with some certainty.

3.4.1.4 CFA—A variety of waste generating processes contributed to the contamination at this site. In addition to industrial support facilities such as vehicle maintenance, welding and other crafts, an industrial waste landfill, radiological laboratories, and radiological laundry facilities also operated at the CFA. During the RI/FS, individual sites were evaluated to identify contamination areas that had the potential to pose future human health risk. Thirteen individual contaminated soil sites were characterized and the nature and extent of contamination was described. Doing so allowed investigators to define source terms for individual sites to support separate modeling efforts for each area. In general, the source terms were comprised of soils contaminated by heavy metals including lead, arsenic, and mercury; radionuclides including Am-241, Ra-226, C-137, Ag-108, and U-235/238; and organic compounds associated with petroleum products. The mass of each contaminant of interest in the CFA surface soils was estimated primarily by results of soil sampling. Individual source volumes were estimated and the concentration indicated by sampling was distributed evenly across the element volume. Estimated masses or activities for each contaminant of interest are presented in Section 6 of the RI/FS (Burgess et al. 2000).

Because these sites were not considered to be a source of groundwater contamination, detailed contaminant information is not presented here.

3.4.1.5 ARA/PBF**—**Fifty-five individual contamination sites were originally identified at the ARA/PBF. The sites consisted primarily of localized surface and subsurface soils that were contaminated by a range of past practices, such as small unlined impoundments used to dispose of research wastewater, septic leach fields, reactor coolant water blow down pits, and windblown contaminated soils resulting from the 1961 SL-1 reactor accident. Based on the results of the RI/FS site characterization and screening process, 14 of the 55 sites were retained for risk analysis. The primary contaminants identified were heavy metals and fission products from the reactor operations. Detailed information regarding the contamination sources at the ARA and PBF, including the volumes of the contaminants, is found in the Record of Decision (ROD) (DOE-ID 2000a). In support of the RI/FS for OU 5-12, the mass of contaminants of interest in the surface soils was estimated primarily by results of soil sampling. Individual source volumes were estimated and the concentration indicated by sampling was distributed evenly across the element volume. Estimated masses or activities for each contaminant of interest are presented in the RI/FS (Holdren et al. 1999). Because these sites were not considered to be a source of groundwater contamination, detailed contaminant information is not presented here. The contaminated areas at ARA/PBF were previously remediated.

3.4.1.6 BORAX **—**The BORAX source term consists of soils and buried debris contaminated primarily by heavy metals and fission products from the reactor operations. Detailed radiological inventories were estimated in Holdren et al. (1995) based on the reactor operating history. For release modeling purposes, the inventory was assumed to be uniformly distributed across the source region. Uranium isotopes comprise a significant fraction of the inventory. Because these sites were not considered to be a significant source of groundwater contamination, detailed contaminant information is not presented here. The contaminated areas at BORAX were previously remediated.

3.4.1.7 RWMC **—**Because of the long-term use of the subsurface disposal area at the RWMC, contaminated soils do exist at the RWMC. However, as described in the buried waste section of this document, the potential impact to groundwater from the SDA has been assessed in terms of release directly from the various waste forms. There has generally been no need to estimate the mass of contaminants associated with the surrounding soils, because the total inventory of contaminants is already included in the conceptual model as buried waste. One exception, however, was the acid pit. The acid pit was an excavated area near the center of the SDA roughly 200 × 100 ft in size. The pit received liquid wastes, primarily organic solvents and radiologically contaminated acids and cleaning solutions. Initial screening investigations indicated a potential for significant groundwater risk from mercury. However, subsequent analysis of the mercury in soil samples indicated that the chemical form was relatively immobile and mercury was screened from further evaluation (Becker et al. 1998).

3.4.1.8 ANL-W**—**As described in the comprehensive RI/FS (Lee et al. 1997) and the subsequent ROD (DOE et al. 1998), soil and sediment samples collected from the industrial waste pond as part of four different investigations during 1986 through 1994 contained concentrations of cesium-137, trivalent chromium, mercury, selenium, and zinc. The area of surficial contamination was estimated to be approximately 200 × 250 ft and contamination was contained in the upper 0.5 ft of pond sediments. Mercury was contained along the entire length of Ditch A and in the upper 0.5 ft of sediments. Trivalent chromium and zinc were contained in sediments in Ditch B. Mercury, which was contained in sludge samples from the sewage lagoons, was contained in sediments along the entire reach of Ditch C. Sediment samples from the entire length of the main cooling tower blowdown ditch contained mercury and chromium. Most of the contaminants have been retained in the top 0.5 ft of sediment. Because these sites were not considered to be a source of groundwater contamination, detailed contaminant information is not presented here.

3.4.1.9 NRF—Accidental radiological discharges have occurred at NRF resulting from corroded underground piping, leakage from underground basins, and releases from radiological tanks (Westinghouse Electric Corporation 1996). Surface releases generally were cleaned up so that any radioactive material above background levels was retrieved. Underground releases were remediated to the maximum extent practical so that background radiation levels were not affected. Accidental release sites where contaminants may not have been completely removed include the Sl W Retention Basins, the Sl W Radiography Building Collection Tanks, the Hot Storage Pit, the ECF Water Pit Release, the Al W/S1 W Radioactive Line Near BB19, and the Al W Processing Building Area Soil. Other sites include the Old Sewage Treatment Plant, Seepage Basin Pump Out Area, and Sewage Lagoons. Other sites include the potentially radiologically-contaminated Old Ditch Surge Pond and S5G Basin Sludge Disposal Bed. Because these sites were not considered to be a source of groundwater contamination, detailed contaminant information is not presented here.

3.4.2 Factors Controlling Release

This section presents two conceptual models for release from contaminated soil sites. First is a description of the simplified, yet effective screening process typically applied at INEEL soil sites. The second part of the discussion is focused on releases at INTEC, where investigators were faced with a unique situation and developed a conceptual approach that supported a bounding estimate of groundwater risk.

Because most INEEL soil sites did not pose significant groundwater problems, detailed conceptual models for release and transport have generally not been developed. The contaminant fate and transport work performed in support of the various RI/FS documents utilized a one dimensional plug flow model and generalized parameters from guidance documents. A comprehensive set of site-specific characterization data was not needed and not compiled. In lieu of developing detailed site-specific geochemical data, DOE screened sites using a simplistic and conservative approach to bound the potential groundwater risk.

As the sources of contamination are contaminated soil sites, the mechanism of release is leaching of contaminants by precipitating soil moisture. Release rates from contaminated soil sites have not been directly measured. Instead, contaminant flux into the vadose zone is estimated using contaminant specific distribution coefficients and average infiltration rates.

To estimate future contaminant concentrations for purposes of evaluating human health risk for the groundwater pathway, investigators developed transport models using GWSCREEN (Rood 1994). GWSCREEN provides a simplified conceptual model for sites where groundwater impacts were expected to be very low. A conservative and simplistic modeling approach was considered adequate and allowed investigators to evaluate impacts from specific sites with minimal site characterization data.

In the GWSCREEN model, the vadose zone is considered to be a homogeneous, isotropic porous medium with constant unidirectional (downward) flow. Unsaturated flow was modeled through sediments only. The unsaturated basalt sequences were excluded from the model, as theoretical descriptions of fracture network flow were not considered defensible at that time. The sedimentary interbeds and surficial sediment thicknesses were summed and assigned conservative sorption coefficients for contaminants of interest using values from the Track 2 guidance (DOE-ID 1994) and other literature such as in Dicke (1997). Infiltration was considered spatially uniform.

Solubility limits were typically not incorporated into the release and transport modeling. To develop conservative estimates, solubility limits were assumed to be essentially infinite; assigning unrealistically high values (e.g., 1.0 E+6 mg/L in Holdren et al. 1995) allowed leachate concentrations to exceed the actual solubility limit of the contaminants.

Results of the fate and transport modeling indicated that peak groundwater concentrations would be low and that the time of peak concentration would be thousands of years in the future for most contaminants. The estimated risk contributed by the groundwater pathway fell below regulatory thresholds for concern for most INEEL contaminated soil sites. The model predictions are thought to overestimate the actual concentrations due to the conservative assumptions made during the modeling efforts.

The release of contaminants from contaminated soil sites at INTEC has been conceptualized a little differently than that at other INEEL soil sites. In the INTEC conceptual site model, the soil contaminants were still applied directly to the vadose zone transport model without a separate "release rate" model, but a more complex vadose zone transport model was used (TETRAD). As described in the RI/FS (DOE-ID 1997), the vadose zone model accounts for the complex layering of sedimentary units including the surficial alluvium by grouping them into five "effective" sedimentary layers. Default hydraulic properties identified for surficial sediments in Rood (1994) were assigned to the sedimentary layers including the top "alluvium" layer. The hydraulic parameters were adjusted as needed to achieve calibration of the flow component. To simulate transport through the vadose zone, contaminant specific sediment partition coefficients, based on the default Track 2 Guidance (DOE-ID 1994) were applied uniformly to each effective sedimentary layer.

Recall from the previous section that the source term at INTEC was divided into two groups. The first was comprised of the five known releases from the tank farm and other facility drains (primarily the tank farm) for which inventory could be reconstructed from operating records. The second source term was comprised of all the other contaminated soil sites, which were estimated by sampling activities.

The first source term, the known release sites, are conceptualized as releasing at a uniform rate over a fixed period. Each contaminant was assigned a mass per day for the number of days of the release. For example, mercury was estimated to have been released from the tank farm at a rate of approximately 1 kg/day for a period of 29 days (DOE-ID 1997). For the known release sites, the source term was conceptualized as being released directly to the uppermost basalt layer, bypassing the top effective sedimentary layer (the alluvium). Because the soil geochemistry is thought to have been altered by the extreme acidity of the released solutions, the conceptual model was that fast drainage paths through the alluvium could have been formed. In order to produce a conservative risk assessment model and avoid the need to characterize a very complex physical and geochemical situation, the tank farm contaminant solutions were input directly to the upper basalt (an un-retarded fast flow unit) bypassing the surficial sediments. Because the contaminants are introduced directly into the vadose zone model domain, the problem becomes an unsaturated transport problem through a layered sedimentary and fractured basalt system.

The second INTEC soil source term, the other contaminated soil sites, was also conceptualized as a vadose zone transport phenomena. The mass of each contaminant was applied, not at the upper basalt sequence as with the known liquid releases, but into the top sedimentary layer. Transport through this and the other sedimentary layers were calculated using default partition coefficients and hydraulic properties.

Many of the soil contaminants were considered to be either (a) present in much smaller amounts than could conceivably pose a risk to the aquifer, (b) have Kd values sufficiently large so that they will bind to the alluvium and not be transported to the aquifer, or (c) decay before they reach the aquifer (DOE-ID 1997). However, based on the conservative modeling approach used for the tank farm soils, it was concluded that aquifer quality would be impacted in the INTEC area by a few contaminants. Contaminants of potential concern resulting from soil contamination included Arsenic, Chromium, Co-60, U-235, Pu-238, -239/240.

3.4.3 Current Uncertainties

The question of how realistic the conceptual model of contaminant release and transport needs to be, or can be given available data and modeling capabilities, remains unresolved. However, with the exception of INTEC, contaminated soils are not considered to have a significant impact on aquifer quality. Using very conservative screening estimates, some groundwater contamination is typically predicted, especially with long-lived radionuclides, but the associated risks are generally below CERCLA guidelines. In addition, the majority of the historical sites were remediated to cut off the surface exposure pathway for facility workers.

Source term inventories at contaminated soil sites have been estimated based on the results of extensive soil sampling. Although this approach is thought to bound the inventory, it can result in overestimates of contaminant mass. Additional work to refine the inventory estimates would probably not be warranted for most sites where the ground water risk from contaminated soil sites is negligible.

Perhaps more significant than the inventory, is the conceptual model of release from soil sites. The actual release mechanism is a leaching process driven by infiltration and geochemical properties of the soil, soil moisture, and the contaminants themselves. A detailed geochemical model has not been developed to describe this process. At all INEEL contaminated soil sites, the estimated inventory of contaminants was input directly into the transport model (GWSCREEN or TETRAD). By applying default (literature derived) partition coefficients and other model parameters, conservative estimates of future groundwater contamination were made.

With the GWSCREEN approach, there are many “uncertainties,” due to the fact the entire approach is intended to provide bounding estimates rather than realistic predictions of groundwater contaminant concentrations. That is, the impact to groundwater from contaminated soil sites, as estimated by the simplistic conceptual model, is probably overestimated. For example, with the one-dimensional conceptual model, all of the infiltrating water passes through the contamination. Another conservative assumption typically used is that contaminant solubility is unlimited and infiltrating water is allowed to dissolve all of the contaminant mass. These and other model conditions are unrealistic, but conservative.

The interesting thing is that although the risk model made the conservative assumption that the source was transported directly to the vadose zone, i.e. not retained in the surface soils, the soils at the tank farm are significantly contaminated to the extent that they could pose a radiological exposure risk to remediation workers. Although the known releases (and the associated inventory of contaminants) are not technically a “source term,” the contamination retained in the surface and subsurface soils presents a potential release for future contamination. The DOE and the regulatory agencies have required additional characterization and evaluation of the tank farm soils (DOE-ID 2000b). The remedial investigation at this site is ongoing.

3.4.4 References

Becker, B.H., J.D. Burgess, K.J. Holdren, D.K. Jorgensen, S.O. Magnuson, and A.J. Sondrup, 1998, *Interim Risk Assessment and Contaminant Screening for the Waste Area Group 7 Remedial Investigation*, DOE-ID-10569, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Burgess, J., S. Flynn, P. Jessmore, J. Keck, S. McCormick, M. Rohe, I. Stepan, R. VanHorn, S. Fu, and L. Trozollo, July 2000, *Comprehensive Remedial Investigation/Feasibility Study for the Central Facilities Area Operable Unit 4-13 at the Idaho National Engineering and Environmental Laboratory*, DOE-ID-10680, Rev 1., Idaho National Engineering and Environmental Laboratory, Prepared for the Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Burns, D.E., K.M. Davis, S.C. Flynn, J.F. Keck, N.L. Hampton, A.H. Owen, and R.L. Van Horn, February 1997, *Comprehensive Remedial Investigation/Feasibility Study for the Test Reactor Area Operable Unit 2-13 at the Idaho National Engineering and Environmental Laboratory, Lockheed Idaho Technologies Company*, DOE-ID-10531, Revision 0, U.S. Department of Idaho Operations Office Idaho Operations Office, Idaho Falls, Idaho.

Department of Energy (DOE), Idaho Department of Health and Welfare (IDEQ), and Environmental Protection Agency (EPA) Region 10, 1998, “Final Record of Decision, Argonne National Laboratory-West,” U.S. Department of Energy.

Dicke, C.A., 1997, *Distribution Coefficients and Contaminant Solubilities for the Waste Area Group 7 BRA*, INEL/EXT-97-00201, Idaho National Engineering and Environmental Laboratory, Lockheed Martin Idaho Technologies Company, Idaho Falls, Idaho.

DOE-ID, 2000a, *Record of Decision for the Power Burst Facility and Auxiliary Reactor Area*, DOE-ID-10700, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

DOE-ID, 2002b, *Operable Unit 3-14 Tank Farm Soil and Groundwater Phase I Remedial Investigation/Feasibility Study Work Plan*, DOE/ID-10676, Revision 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

DOE-ID, 1997, *Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL-Part A, RI/BRA Report Final*, Appendix F, DOE-ID-10534, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

DOE-ID, January 1994, *Track 2 Sites: Guidance for Assessing Low Probability Hazard Sites at the INEL*, DOE-ID-10389, Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Holdren, K.J., C.M. Hiaring, D.E. Burns, N.L Hampton, B.J. Broomfield, E.R. Neher, R.L VanHorn, I.E. Stepan, R.P. Wells, R.L Chambers, L. Schmeising, and R. Henry, January 1999, *Waste Area Group 5 Operable Unit 5-12 Comprehensive Remedial Investigation/Feasibility Study*, DOE-ID-10607, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Holdren, K.J., R.G. Filemyr, and D.W. Vetter, March 1995, *Remedial Investigation Feasibility Study (RI/FS) Report For OU 5-05 and 6-01 SL-1 and BORAX I Burial Grounds*, INEL-95/0027, Revision 0, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Kaminski, J.F., K.N. Keck, A.L. Schafer-Perini, C.F. Hersley, R.P. Smith, G.J. Stormberg, and A.H. Wylie, 1994, *Remedial Investigation Final Report with Addenda for the Test Area North Groundwater Operable Unit 1-07B at the Idaho National Engineering Laboratory*, EGG-ER-10643, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Lee, S.D., M.J. Rohe, A.S. Rood, and I.E. Stepan, 1997, *Comprehensive Remedial Investigation/Feasibility Study for Argonne National Laboratory-West Operable Unit 9-04 at the Idaho National Engineering and Environmental Laboratory*, W7500-0000-ES-02, Rev 2, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Medina, S. M., and C. F. Hersley, 1993*, Evaluation of Historical and Analytical Data on the TAN TSF-07 Disposal Pond*, EGG-ERD-10422, August.

Rood, A.S., 1994, "GWSCREEN: A Semi-Analytical Model for the Assessment of the Groundwater Pathway from Surface or Buried Contamination," Version 2.0, Theory and User's Manual, EGG-GEO-10797, Idaho National Engineering and Environmental Laboratory, EG&G Idaho, Idaho Falls, Idaho.

Westinghouse Electric Corporation, 1996, "Final NRF Comprehensive Remedial Investigation/Feasibility Study, Waste Area Group 8, Naval Reactors Facility, Idaho Falls, Idaho, Part 1 – Remedial Investigation Report: Volumes 1 and 2," prepared for the U.S. Department of Energy.

4. SOURCE-TERM DISCUSSION BY CONTAMINANT

Contaminants disposed to the subsurface at the INEEL during 1952-2003 include radionuclides, inorganic chemicals, organic chemicals, and trace elements. These contaminants were disposed in aqueous or non-aqueous liquid wastes to injection wells, infiltration ponds, and ditches, were contained in surface spills and leaks, or were contained in solid or solidified wastes buried in landfills. Baseline risk assessments (BRAs) conducted for each of the Waste Area Groups (WAGs) identified those contaminants of potential concern (COPCs) and evaluated those contaminants to develop finalized lists of COCs. Table 4-1 lists COCs as listed for each INEEL facility or WAG and delineates contaminants that are connected to groundwater exposure pathways. Other contaminants, while not identified as COCs, function as chemical tracers and provide valuable information about groundwater flow and the fate of contaminants in the subsurface. Subsequent sections describe locations, quantities, source-term release mechanisms, and rates of release by contaminant, grouped as radionuclides, organic chemicals, trace elements, and inorganic chemicals.

4.1 Radionuclides

Radionuclide COCs at the INEEL were disposed to the subsurface at most of the INEEL facilities. Some of these contaminants are detected at small concentrations in water from the SRPA. Others are not detected. The rate of release from the waste material depends on the integrity of any container (or encapsulating material) surrounding the material, and on the physical and chemical properties of the material that is directly associated with the contaminant (e.g., contaminant crud on the surface of steel, or carbon-14 within the steel matrix). Subsequent detection in the aquifer is dependent on the mobility of the specific radionuclide away from the source term, rate of disposal, and length of half-life. Contaminants include tritium, strontium-90, cesium-137, cobalt-60, technetium-99, iodine-129, chlorine-36, carbon-14, uranium isotopes, and plutonium and americium isotopes. Except for those radionuclides that form volatile compounds (i.e., C-14 and H-3), all radionuclides migrate from the waste material and to the aquifer as ions dissolved in the liquid phase.

The rate of release of contaminant by injection of wastewater directly into the SRPA through disposal wells generally is considered to be the same as the rate of disposal. The rate of release from ponds, ditches, pits, trenches, and other near-surface sources is controlled primarily by the flux of water from local recharge as it comes into contact with the waste, and in part by the capacity of the geologic materials adjacent to the contaminant source to sorb the specific contaminant. Contaminant transport away from the source commonly generally is estimated using an estimated flux and a sorption coefficient (kd) as determined from site-specific studies or from generalized values.

Selected radionuclides have been identified as COCs in BRAs that were conducted as part of Remedial Investigations activities at the INEEL. Table 4-1 lists COCs identified by each WAG. Radionuclide COCs include actinium-227, americium-241, carbon-14, cesium-134, cesium-137, chlorine-36, cobalt-60, europium-152, europium-154, iodine-129, neptunium-237, niobium-94, plutonium 238, and -239/240, protactinium-231, silver-108m, strontium-90, technetium-99, thorium-228, thorium-229, thorium-230, tritium, and uranium-233, -234, -235, -236, and -238. Of these contaminants, actinium-227, carbon-14, cesium-137, chlorine-36, iodine-129, neptunium-237, niobium-94, total plutonium, protactinium-231, strontium-90, technetium-99, thorium-229, thorium-230, and uranium isotopes have been determined to be associated with groundwater exposure paths. The INEEL source terms for each of the radionuclide COCs, including location, activity, mechanism, and rate of release, are summarized in subsequent sections.

Table 4-1. Contaminants of concern as identified by baseline risk assessments for the waste area groups at the Idaho National Engineering and Environmental Laboratory.

Contaminant	***INEEL WASTE AREA GROUP***								
	[a]WAG-1	[b]WAG-2	[c]WAG-3	[d]WAG-4	[e]WAG-5	[f]WAG-6/10	[g]WAG-7	[h]WAG-8	[i]WAG-9
RADIONUCLIDE CONTAMINANTS OF CONCERN									
Actinium-227							G		
Americium-241		X	X				X	X	
Carbon-14			D				G	D	
Cesium-134		X							
Cesium-137	G	X		X			X	X	
Chlorine-36							G		
Cobalt-60		X							
Europium-152		X	X						
Europium-154		X	X						
Iodine-129			G				G		
Neptunium-237			G				G	X	
Nickel-63								X	
Niobium-94							G		
Plutonium(total)			G						
Plutonium-238		X	X				X	X	
Plutonium-239/240		X	X				X		
Plutonium-244								X	
Protactinium-231							G		
Radium-226				X			X		
Silver-108m		X							
Strontium-90	G	G	G				X	X	
Technetium-99		D	X				G		
Thorium-228		X							
Thorium-229							G		
Thorium-230							G		
Tritium	G	G	G						
Uranium-233							G		
Uranium-234	G						G		
Uranium-235			X				G	X	
Uranium-236							G		
Uranium-238		X					G		
ORGANIC CHEMICAL CONTAMINANTS OF CONCERN									
Trichloroethene	G								
Tetrachloroethylene	G						G		
Cis-1,2-DCE	G								
Trans-1,2-DCE	G								
Acrylonitrile		X							
Carbon tetrachloride							G		
Methylene chloride							G		

Contaminant	***INEEL Waste Area Group***								
	WAG-1	**WAG-2**	**WAG-3**	**WAG-4**	**WAG-5**	**WAG-6/10**	**WAG-7**	**WAG-8**	**WAG-9**
TRACE ELEMENT CONTAMINANTS OF CONCERN									
Arsenic		**G**			S				
Beryllium		**X**		**G**					
Chromium		**G**	**G**						
INORGANIC CHEMICAL CONTAMINANTS OF CONCERN									
Nitrates							**G**		

G Contaminant of concern for groundwater exposure pathway
X Contaminant of concern for exposure pathways other than groundwater
D Not identified as a contaminant of concern but disposed to the subsurface at the facility
[a] TAN COC information from Kaminsky et al. (1994), Table 7-6
[b] TRA COC information from Lewis et al. (1992) and Burns et al. (1997), Table 11-5.
[c] INTEC COC information from Lockheed Martin Idaho (1997), page 2-16.
[d] CFA COC information from Keck et al. (1994 p. v) and from Burgess et al. (2000 p. 8-1).
[e] No contaminants of concern were identified for groundwater exposure pathways at ARA/PBF (Holdren et al. 1999, Table 8-9).
[f] No contaminants of concern were identified for groundwater exposure pathways at EBR-1/BORAX and WAG-10 sites (DOE 2001)
[g] RWMC COC information from Holdren et al. (2000) Table 7-1.
[h] NRF COC information from Westinghouse Electric Corporation (1996), Table 20-3.
[i] ANL-W COC information from Lee et al. (1997)

4.1.1 Actinium-227

Actinium-227 was determined in BRAs to be a COC at the RWMC (Table 4-1). The radioactive decay half-life of actinium-227 is 21.773 years.

4.1.1.1 Actinium-227 source-term locations and the activities of actinium-227 disposed to the subsurface at the INEEL

Actinium-227 is listed as a groundwater ingestion COC at the RWMC in the WAG 7 OU 13/14 BRA (Holdren et al. 2000, Table 7-1). A total activity of 5.12×10^{-7} Ci of actinium-227 are associated with subassembly hardware, resins, fuel end pieces, beryllium waste, core structural pieces, and other miscellaneous waste streams.

4.1.1.2 Mechanisms and rates of actinium-227 release from source terms to the subsurface

Actinium-227 is available for release from RWMC wastes through dissolution and surface washoff. The best-estimated site-specific k_d for sediments at the RWMC was 400 mL/g. Based on this k_d and a peak simulated risk of 3×10^{-6}, actinium-227 is not considered to be a significant groundwater exposure pathway COC at the RWMC.

4.1.2 Americium-241

Americium-241 was determined in BRAs to be a COC associated with liquid and solid waste disposals to the subsurface at TRA, RWMC, and NRF (Table 4-1). The combined activity disposed from these facilities since 1952 is estimated to be about 1.83×10^{5} Ci, with almost all of the activity attributed to wastes buried at the RWMC. The radioactive decay half-life for americium-241 is 432.2 years.

4.1.2.1 Americium-241 source-term locations and the activities of americium-241 disposed to the subsurface at the INEEL

Americium-241 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Five soil-contamination sites were identified as containing quantities of americium-241. These sites (and activities, as used in GWSCREEN modeling activities) include soils surrounding Hot Waste Tanks 2, 3, and 4 at TRA 613 (1.91×10^{-4} Ci), pond sediments in the sewage leach pond (1.65×10^{-3} Ci), surface soil associated with the sewage leach pond berm and soil contamination area (2.55×10^{-3} Ci), soil and pond-bottom sediments of the warm waste pond 1952 and 1957 cells (5.74×10^{-0} Ci), and soils associated with the retention basin, cold-waste sampling pit, and cold-water sump (activity not described). The combined activity as disposed (not including the retention basin), and not corrected for radioactive decay, was approximately 5.75 Ci (Burns et al. 1997, Table B-28). Exposure pathways for each of these surface- and subsurface soil sites may include groundwater infiltration (Burns et al. 1997, Figure 5-1).

Americium-241 is not listed as a COC for INTEC. A total of 1.23×10^{-1} Ci of americium-241 were injected directly into the SRPA through injection well CPP-23 during 1953-86 (Fredericks et al. 1998, Table 3-1).

Americium-241 is listed as a COC in the WAG 7 OU 13/14 BRA (Holdren et al. 2000, Table 7-1). The risk primarily is through soil ingestion, inhalation, external exposure, and crop ingestion. Approximately 1.83×10^{5} Ci of americium-241 were associated with solid and semi-solid wastes that were disposed to the RWMC (Holdren et al. 2000, p. 4-39). About 80% of this amount was in the form of uncemented sludges from the RFP.

Americium-241 is listed in the OU 8-08 BRA as a COC at NRF (Westinghouse Electric Corporation 1996, Table 20-3). Americium-241 was identified in soil contamination sites.

4.1.2.2 Mechanisms and rates of americium-241 release from source terms to the subsurface

Americium-241 contained in water disposed to the TRA warm waste pond probably remains sorbed to sediments beneath the pond as a potential secondary contaminant source. Release from these sediments was simulated by GWSCREEN (Burns et al. 1997, p. C-69) using a Track-2 guidance sorption coefficient of 3.40×10^{2} ml/g (EG&G Idaho 1994, Table F-1) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that americium-241 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-69). Based on these results, americium-241 that has sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths.

Americium-241 contained in soils at the NRF was determined not to be a groundwater exposure pathway COC.

Americium-241 was directly disposed to the SRPA through INTEC injection well CPP-23. The rate of release was proportional to the concentration in wastewater and to the rate of disposal.

Americium-241 associated with solid wastes buried in RWMC pits and trenches is available for release from those wastes through surface washoff (Holdren et al. 2000, Table 5-3). Surface washoff requires that locally infiltrating water come in contact with the solid wastes. Transport of americium-241 away from these source terms was modeled using a best-estimate value of 4.5×10^{2} mL/g (Holdren et al. 2000, Table 5-13) as determined by Dicke (1997) rather than conservative Track-2 screening values. Based on the slow release and on a simulated peak risk of 3×10^{-5}, americium-241 was determined not to be a COC with respect to the groundwater exposure pathway at the RWMC. Baseline-case studies of release from source terms in the RWMC SDA did not include the potential for facilitated transport (Holdren et al. 2000, p. 5-30). However, studies conducted at the SDA have indicated that small fractions of americium-241 may move in a facilitated manner. Although americium-241 is not considered as a groundwater COC because of its immobility, this contaminant may provide a long-term secondary source term for its more mobile daughter product, neptunium-237.

Americium-241 detections in groundwater at the INEEL typically have been sporadic, non-repeatable, and near the instrument detection limit. These detections suggest that release from the source term typically is limited and that americium-241 readily sorbs to sediments near the source term. In general, because of the strong tendency to sorb to sediments, americium-241 has been determined not to be an INEEL COC with respect to groundwater exposure pathways.

4.1.3 Carbon-14

Carbon-14 was determined to be a COC at the RWMC (Table 4-1). The radioactive decay half-life of carbon-14 is 5,715 years.

4.1.3.1 Carbon-14 source-term locations and the activities of carbon-14 disposed to the subsurface at the INEEL

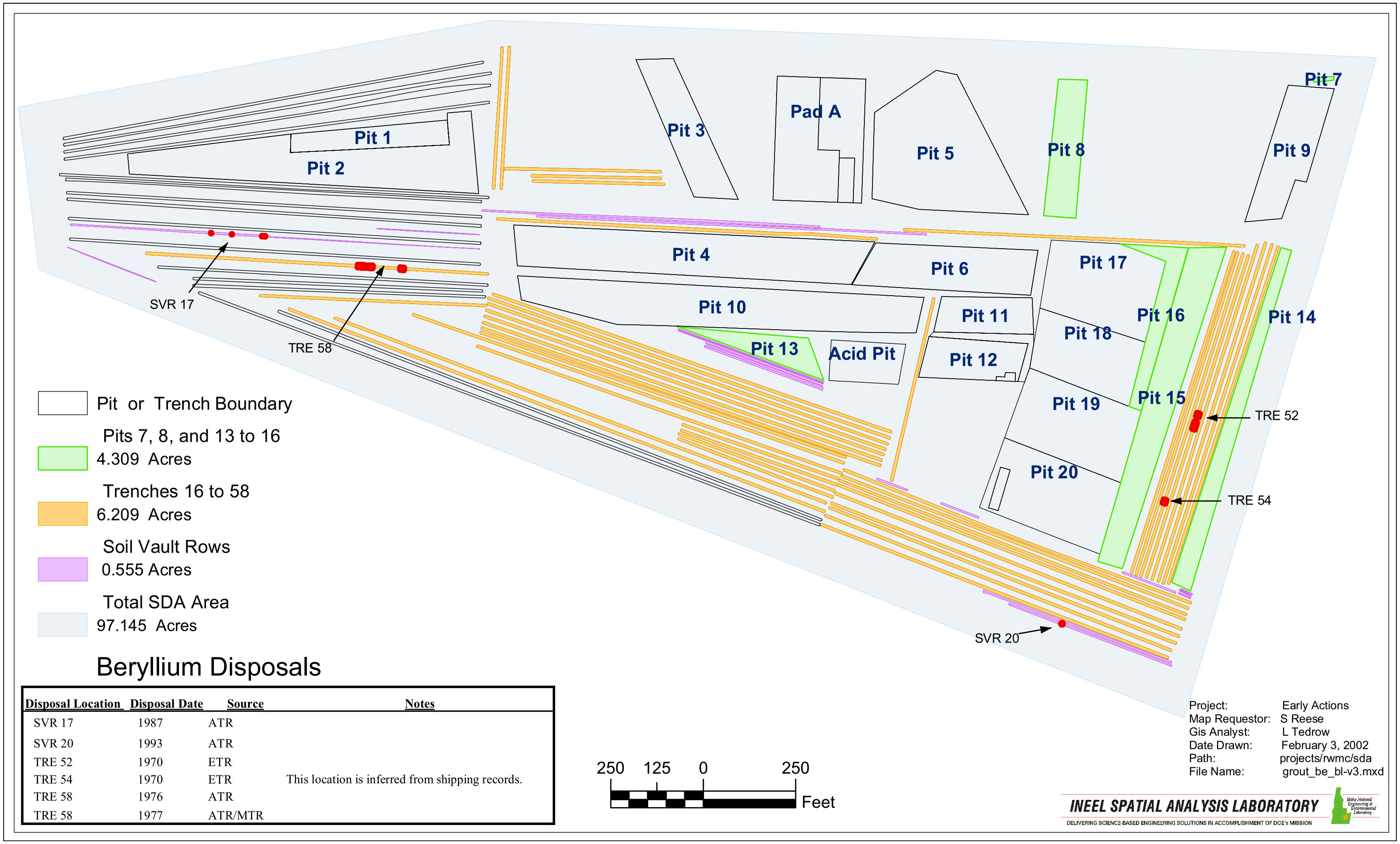
Carbon-14 is listed as a groundwater ingestion COC in the WAG 7 OU 13/14 BRA (Holdren et al. 2000, Table 7-1). The total activity of carbon-14 that is associated with solid wastes buried in pits and trenches at the RWMC is estimated to be 5.00×10^2 Ci (Holdren et al. 2000, Table 5-3).

Most of the INEEL C-14 source term and inventory is associated with activated beryllium and steel that is located in the SDA at the RWMC. The inventory in activated steel is over 70% of the total (based on the ABRA), and the activated beryllium contains about 18% of the total C-14 inventory. However, C-14 is released at a relatively high rate from the beryllium, so beryllium is considered to be the dominant C-14 bearing wasteform with respect to the source term. Beryllium was buried in a small number of discrete (i.e., isolated from other disposals and other materials), known locations, and the total volume of beryllium disposals is a very small fraction of waste disposed of in the SDA. The locations of activated beryllium disposals are shown in Figure 4-1. Table 4-2 (from the ABRA) summarizes the C-14 inventory information.

Approximately 500 Ci of C-14 were disposed of in the SDA. Waste streams containing the majority of C-14 activity are identified in Table 4-2. Gamma spectral logging in the waste zone provides no information about C-14.

A total of 9.56×10^{-7} Ci of carbon-14 was reported disposed to injection well CPP-23 at INTEC in 1985 (Frederick et al. 1998 Table 3-1). No other information is available about disposal of carbon-14 at INTEC.

Carbon-14 is not listed as a COC at NRF. However, an activity of 1.32×10^0 Ci was estimated for GWSCREEN evaluation of transport (Westinghouse Electric Corporation 1996, Table 18-5).



Disposal Location	Disposal Date	Source	Notes
SVR 17	1987	ATR	
SVR 20	1993	ATR	
TRE 52	1970	ETR	
TRE 54	1970	ETR	This location is inferred from shipping records.
TRE 58	1976	ATR	
TRE 58	1977	ATR/MTR	

Figure 4-1. Beryllium disposals in the Subsurface Disposal Area of the Radioactive Waste Management Complex.

Table 4-2. Waste streams containing C-14.

Waste Stream Code or Generator	Waste Stream Description	Activity (Ci)	Proportion of Total Activity (%)
Test Reactor Area (TRA)	Activation products.	2.08E+02	41.7
NRF-616-3H, 4H, 8H	Core structural pieces.	1.07E+02	21.3
TRA	Beryllium waste.	9.26E+01	18.5
CPP-603-1H	Fuel end pieces.	4.58E+01	9.2
Argonne National Laboratory-West	Subassembly hardware.	1.66E+01	3.3
Miscellaneous	Miscellaneous minor streams.	1.45E+01	2.9
TRA-603-1H	TRA resins.	7.81E+00	1.6
ANL-785-1	Subassembly low-level waste from the Hot Fuel Examination Facility experiments.	7.51E+00	1.5
	Total Disposals	5.00E+02	100

4.1.3.2 Mechanisms and rates of carbon-14 release from source terms to the subsurface

A total activity of 4.78×10^2 Ci of carbon-14 is estimated to be available for dissolution from buried wastes at the RWMC. Rate of release through dissolution is radically different depending on whether it occurs in association with low-solubility stainless steel fuel rods or larger-solubility beryllium blocks. Another 2.23×10^1 Ci is estimated to be available for surface washoff from buried solid wastes (Holdren et al. 2000, Table 5-3).

Numerical simulation of carbon-14 release and transport at the RWMC used a k_d of 0.10 mL/g (Holdren et al. 2000, p. 5-72) that was derived from site specific studies (Dicke 1998). Both dissolution and surface washoff of carbon-14 require a local source of infiltrating surface water that can come into contact with the solid waste. Local recharge was simulated with a variable flux across the SDA as determined from field data. Numerical modeling results overpredicted carbon-14 concentrations in groundwater when compared to measured concentrations. These results suggested that simulated release rates and other parameters were conservatively estimated.

The chemical form of C-14 in activated beryllium is not known. Beryllium in contact with INEEL soil corrodes at about the same rate as carbon steel[2]. Corrosion destroys the metal grain structure and is the dominant release mechanism. In situations where activated beryllium is the only material in the source region (e.g., in soil vault row disposals), it is unlikely that the C-14 is present in forms other than CO_2. If the beryllium is buried in a location where the chemical environment is reducing, a relatively large fraction of the C-14 may be present in organic form, particularly as $^{14}CH_4$.

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[2] Results of corrosion testing of buried carbon steel and beryllium coupons show a maximum thickness loss rate of 12 μm/y (0.460 mils/y) and 8.3 μm/y (0.327 mils/ y), respectively, after 3 years of burial in typical INEEL soil (Adler-Flitton et al. 2001).

A large fraction of the C-14 inventory is located in activated steel. It is assumed that this C-14 inventory is relatively immobile, and does not contribute significantly to the source term. A small fraction of the C-14 is associated with other, non-metallic waste forms such as ion exchange resins. Based on source term modeling for the current risk assessment, this C-14 contributes a very minor fraction of the total risk associated with the SDA C-14 inventory.

4.1.4 Cesium-134

Cesium-134 was determined to be a COC at the TRA (Table 4-1). The half-life of cesium-134 is 2.062 years.

4.1.4.1 Cesium-134 source-term locations and the activities of cesium-134 disposed to the subsurface at the INEEL

Cesium-134 was not determined to be a COC at TAN. A total of 2.88×10^{-2} Ci of cesium-134 was released to the TAN TSF-05 injection well during 1959-1972 (Kaminsky et al. 1994, Table 3-4). A total of 2.588×10^{-3} Ci of cesium-134 was released to the TSF-07 disposal pond during 1972-1985 (Kaminsky et al. 1994, Table 3-3).

Cesium-134 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Seven soil-contamination sites were identified that contained cesium-134 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 4.31×10^{-1} Ci (Burns et al. 1997, Table B-28).

4.1.4.2 Mechanisms and rates of cesium-134 release from source terms to the subsurface

Cesium-134 was released directly to the aquifer through the TAN TSF-05 injection well during 1953-72. Approximately 2.88 × 10-2 Ci were disposed to the well during 1959-72 (Frederick et al. 1998, Table 5-3). Gradual release also occurred from the secondary source term as cesium-134 may have leached from the sludges in the basalt around the well. The significance of the cesium-134 as a groundwater contaminant source term by direct injection is minimal because of the short radioactive decay half-life and the initially small activities disposed through injection.

Cesium-134 disposed to infiltration ponds at TAN and TRA has strongly sorbed to sediments beneath the ponds. Because of the 2.062-year half-life of cesium-134, the activity associated with these source terms has been reduced to an undetectable level since disposal. Release from TRA pond-bottom sediments was represented in GWSCREEN modeling activities (Burns et al. 1997, p. C-72) using a Track 2 k_d at the source of 5.00×10^2 ml/g (EG&G Idaho 1994, Table F-1) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that cesium-134 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-72). The TAN Remedial Investigation Final Report concluded that no evidence exists that direct migration of contaminants in surface water, sediment/soils, and the vadose zone contributes to groundwater contamination at TAN (Kaminsky et al. 1994, p. 5-1). Based on these results and on the short half-life, cesium-134 that has sorbed to surficial sediments at the TRA and TAN does not constitute a contaminant source term for groundwater exposure paths.

4.1.5 Cesium-137

Cesium-137 was determined to be a COC at TAN, NRF, and TRA (Table 4-1). Cesium-137 was disposed at these facilities and at INTEC and RWMC. The radioactive decay half-life of cesium-137 is 30.2 years.

4.1.5.1 Cesium-137 source-term locations and the activities of cesium-137 disposed to the subsurface at the INEEL

Cesium-137 is listed in the OU 1-07B BRA as a groundwater COC at TAN (Kaminsky et al. 1994, Table 7-6). A total of 1.37×10^{-1} Ci of cesium-137 was released to the TAN TSF-05 injection well during 1959-1972 (Kaminsky et al. 1994, Table 3-4). A total of 2.748×10^{-2} Ci of cesium-137 were reported released to the TSF-07 disposal pond during 1972-1985 (Kaminsky et al. 1994, Table 3-3).

Cesium-137 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Eleven soil-contamination sites were identified that contained cesium-137 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 8.17×10^{1} Ci (Burns et al. 1997, Table B-28). Cesium-137 was listed as a COC for the perched water system (Lewis et al, p. xviii).

Cesium-137 is not listed at INTEC as a COC. However, cesium-137 has been disposed to the subsurface at INTEC. Frederick et al. (1998, Table 3-1) reported approximately 32 Ci of cesium-137 were contained in aqueous wastes disposed to the CPP-23 injection well at INTEC during 1953-1986, with approximately 25 Ci disposed during 1956-1970. Robertson et al. (1974, Table IX) reported 52 Ci of cesium-137 contained in aqueous wastes disposed to the subsurface at INTEC during 1952-1970. Of this activity, an estimated 34 Ci were disposed to the waste disposal pit near building CPP-603. The remainder was disposed to the CPP-23 injection well.

Cesium-137 is listed in the CFA OU 4/13 BRA as a COC at CFA (Burgess et al. 2000, Table 8-4). However, the exposure routes are determined to be ingestion of soil, dermal contact with soil, external radiation exposure, and ingestion of homegrown produce (Burgess et al. 2000, Table 8-1). Groundwater pathways were determined to have a calculated risk less than 1×10^{-4}.

Cesium-137 is listed as an external exposure COC at the RWMC (Holdren et al. 2000, Table 7-1). A total of 6.17×10^{5} Ci of cesium-137 have been contained in solid wastes disposed to RWMC pits and trenches (Holdren et al. 2000, Table 5-3).

Cesium-137 is listed in the OU 8-08 BRA as a COC at NRF (Westinghouse Electric Corporation 1996, Table 20-3). A total of 23.8 Ci of cesium-137 was released to the S1W leach pit, S1W industrial waste lagoons, and A1W leaching bed during 1953-1979 (Frederick et al. 1998, Table 4-8).

A total of 1.671 Ci of cesium-137 are reported disposed to the EBR-II leach pit during 1960-1975 (LATA 1990) (Frederick et al. 1998, Table 2-1).

4.1.5.2 Mechanisms and rates of cesium-137 release from source terms to the subsurface

Cesium-137 that was contained in wastewater disposed to the TAN injection well TSF-05 during 1953-1972 and the INTEC injection well CPP-23 during 1952-1984 was released directly to the SRPA. Release from the source term took place at the rate of disposal. Some release of cesium-137 may have occurred from the secondary source term formed by the sludges injected into fractures in the basalt around TAN injection well TSF-05.

Cesium-137 has been detected in shallow perched groundwater beneath TRA infiltration ponds. Numerical modeling of contaminant migration from perched groundwater at TRA indicated that release of cesium-137 would be retarded to the extent that peak concentrations in groundwater would not be reached within 125 years (Lewis et al. 1992, p. xvi).

Cesium-137 disposed to ponds and other surface or shallow subsurface sites at TAN, TRA, INTEC, CFA, and NRF passed through surficial sediments. Transport is dependent on contaminant sorption to these sediments and on the flux of water available as local recharge. Most of this cesium-137 probably remained within a small distance of the point of infiltration because of a strong tendency to sorb to these sediments. Cesium-137 release from this secondary source term at TRA was estimated with GWSCREEN (Burns et al. 1997, p. C-73) using a Track 2 guidance k_d at the source of 5.00×10^2 ml/g (EG&G Idaho 1994, Table F-1) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that cesium-137 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-73). Additionally, the TAN Remedial Investigation Final Report concluded there is no evidence that direct migration of contaminants in surface water, sediment/soils, and the vadose zone contributes to groundwater contamination at TAN (Kaminsky et al. 1994, p. 5-1). Based on these results, cesium-137 sorbed to surficial sediments or in the vadose zone does not constitute a contaminant source term for groundwater exposure paths.

Cesium-137 that was associated with solid wastes disposed to pits and trenches at the RWMC is available for surface washoff. Release through surface washoff occurs as locally infiltrated water moves through surficial sediments and comes into contact with those wastes. Cesium-137 strongly sorbs to clay minerals, iron oxides, and other materials present in surficial sediments, subsurface sedimentary interbeds, and minerals lining and filling basalt fractures at the INEEL. Consequently, transport from points of disposal at the RWMC to groundwater exposure pathways is limited. Risks primarily are derived through external exposure.

4.1.6 Chlorine-36

Chlorine-36 was determined to be a COC at the RWMC. Chlorine-36 was disposed in aqueous wastes at INTEC and was contained in solid wastes buried at the RWMC. The half-life of chlorine-36 is 301,000 years. While not determined to be a COC at the INEEL, chlorine-36 in groundwater can act as a useful tracer to describe the movement of groundwater and transport of contaminants because of its long half-life and because it is relatively conservative in rock/water systems.

4.1.6.1 Chlorine-36 source-term locations and the activities of chlorine-36 disposed to the subsurface at the INEEL

Chlorine-36 is not listed as a groundwater COC at INTEC. A total of 7 Ci of chlorine-36 was disposed to the INTEC injection well CPP-23 during 1953-1984 (Cecil et al. 2000, Table 1).

Chlorine-36 is listed as a groundwater COC at the RWMC (Holdren et al. 2000, Table 7-1). A total of 1.11×10^0 Ci of chlorine-36 was associated with solid waste buried in RWMC pits and trenches (Holdren et al. 2000, Table 5-3). This amount included 6.62×10^{-1} Ci contained in beryllium blocks.

4.1.6.2 Mechanisms and rates of chlorine-36 release from source terms to the subsurface

Chlorine-36 contained in wastewater disposed to the INTEC injection well was released directly to the SRPA. The rate of release is equivalent to the rate of disposal.

At the RWMC, chlorine-36 may be released by dissolution from beryllium blocks or by surface washoff from solid wastes (Holdren 2000, Table 5-3). Numerical analyses of chlorine-36 release and transport used a best-estimated site-specific k_d of 0. This k_d assumes that chlorine-36 is completely conservative with respect to transport. Based on a simulated peak risk of 6×10^{-6} (Holdren et al. 2000, Table 6-6), chlorine-36 was determined not to be a significant groundwater exposure path COC at the RWMC.

4.1.7 Cobalt-60

Cobalt-60 was determined to be a COC at the TRA (Table 4-1). The half-life of cobalt-60 is 5.271 years.

4.1.7.1 Cobalt-60 source-term locations and the activities of cobalt-60 disposed to the subsurface at the INEEL

Cobalt-60 is not listed as a groundwater COC at TAN. A total of 1.973×10^{-2} Ci of cobalt-60 was released to the TSF-07 disposal pond during 1972-1985 (Kaminsky et al. 1994, Table 3-3).

Cobalt-60 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Eleven soil-contamination sites were identified that contained cobalt-60 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 2.53×10^{0} Ci (Burns et al. 1997, Table B-28).

Cobalt-60 is not listed at INTEC as a COC. However, 1.24×10^{0} Ci were injected with aqueous wastes directly into the SRPA through the CPP-23 injection well during 1953-1986 (Frederick et al. 1998, Table 3-1).

A total of 0.2709 Ci of cobalt-60 are reported disposed to the EBR-II leach pit during 1960-1975 (LATA 1990) (Frederick et al. 1998, Table 2-1).

Cobalt-60 is not listed at NRF as a COC. However, a total of 75.25 Ci of cobalt-60 were disposed to the S1W leaching pit, S1W industrial waste lagoons, and A1W leaching bed during 1953-1979 (Frederick et al. 1998, Table 4-9).

4.1.7.2 Mechanisms and rates of cobalt-60 release from source terms to the subsurface

Direct injection of wastewater containing Cobalt-60 to the SRPA occurred during 1953-1984. Release was related to the concentration and rate of wastewater discharge.

The TAN Remedial Investigation Final Report concluded that no evidence exists that direct migration of contaminants in surface water, sediment/soils, and the vadose zone contributes to groundwater contamination at TAN (Kaminsky et al. 1994, p. 5-1).

Cobalt-60 release from the secondary source term in surficial sediments at TRA was estimated with GWSCREEN using a k_d at the source of 1.00×10^{1} ml/g (Burns et al. 1997, p. C-71) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that cobalt-60 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-71). Similarly, the Baseline Risk Analysis for the NRF determined that the short half-life of cobalt-60, limited local recharge, and the long travel time through the vadose zone limited groundwater exposure pathways for this radionuclide. Based on these BRA results, cobalt-60 sorbed to surficial sediments at the INEEL does not constitute a contaminant source term for groundwater exposure paths.

4.1.8 Europium-152

Europium-152 was determined to be a COC at the TRA (Table 4-1). The half-life of europium-152 is 13.6 years.

4.1.8.1 Europium-152 source-term locations and the activities of Europium-152 disposed to the subsurface at the INEEL

Europium-152 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Four soil-contamination sites were identified that contained europium-152 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 5.78×10^0 Ci (Burns et al. 1997, Table B-28).

4.1.8.2 Mechanisms and rates of Europium-152 release from source terms to the subsurface

TRA GWSCREEN sorption coefficient at source 6.50×10^2 ml/g (Burns et al. 1997, p. C-74) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that europium-152 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-74). Based on these results, europium-152 sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths.

4.1.9 Europium-154

Europium-154 was determined to be a COC at the TRA (Table 4-1). The half-life of europium-154 is 8.8 years.

4.1.9.1 Europium-154 source-term locations and the activities of Europium-152 disposed to the subsurface at the INEEL

Europium-154 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Five soil-contamination sites were identified that contained europium-154 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 1.45×10^0 Ci (Burns et al. 1997, Table B-28).

4.1.9.2 Mechanisms and rates of Europium-154 release from source terms to the subsurface

TRA GWSCREEN sorption coefficient at source 6.50×10^2 ml/g (Burns et al. 1997, p. C-75) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that europium-154 in surficial sediments will not reach the SRPA (Burns et al. 1997 p. C-75). Based on these results, europium-154 sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths.

4.1.10 Iodine-129

Iodine-129 was determined to be a COC at INTEC and the RWMC (Table 4-1). The half-life of iodine-129 is 17 million years.

4.1.10.1 Iodine-129 source-term locations and the activities of iodine-129 disposed to the subsurface at the INEEL

Iodine-129 is listed as a COC at INTEC (Lockheed Martin Idaho Technologies Company 1997, p. 2-16). Frederick et al. (1998; Table 3-1) reported disposal of approximately 1.39 Ci of iodine-129 to injection well CPP-23 during 1953-1986. Iodine-129 is listed as a groundwater ingestion COC in the WAG 7 OU 13/14 BRA (Holdren et al. 2000, Table 7-1). A total activity of 1.58×10^{-1} Ci of iodine-129 was associated with solid and semi-solid wastes that were disposed to the RWMC (Holdren et al. 2000, Table 5-3).

4.1.10.2 Mechanisms and rates of iodine-129 release from source terms to the subsurface

Most of the iodine-129 disposed at INTEC was injected directly into the aquifer under pressure through the injection well (INEEL 1998, p. B 4-1). The rate of release to the SRPA was equivalent to the rate of disposal of wastewater containing iodine-129. Part of the iodine-129 continues to be released slowly from the vadose zone. Simulations indicate that this secondary source of iodine-129 will be reduced by removal of recharge from percolation ponds. These simulations assume the Track 2 guidance k_d of 0 (EG&G Idaho 1994, Table F-1).

Iodine-129 associated with solid wastes buried at RWMC is available for release through surface washoff from solid wastes. Release takes place as locally infiltrating water comes into contact with the solid wastes. Numerical modeling of iodine-129 transport in the subsurface at the RWMC used a best-estimate k_d of 0.1 mL/g (Holdren et al. 2000, Table 5-13). Simulated iodine-129 concentrations for SRPA wells were overpredicted, suggesting that release rates and other factors were conservatively estimated. Based on a simulated peak risk of 6×10^{-5} (Holdren et al. 2000, Table 6-6), iodine-129 was determined not to be a significant groundwater exposure path COC at the RWMC.

4.1.11 Neptunium-237

Neptunium-237 was determined to be a COC at the RWMC and NRF (Table 4-1). The half-life of neptunium-237 is 2.14 million years.

4.1.11.1 Neptunium-237 source-term locations and the activities of neptunium-237 disposed to the subsurface at the INEEL

Neptunium-237 is not listed as a groundwater COC at INTEC. Frederick et al. (1998; Table 3-1) reported that approximately 1.066 Ci of neptunium-237 were disposed to the INTEC injection well CPP-23 during 1953-1986.

Neptunium-237 is listed in the WAG 7 OU 13/14 BRA as a groundwater ingestion COC (Holdren et al. 2000 Table 7-1). An estimated activity of 2.64×10^{0} Ci is associated with solid wastes buried in pits and trenches (Holdren et al. 2000, Table 5-3). Of this activity, 0.64 Ci is attributed to ingrowth from plutonium-241 and americium-241 decay.

Neptunium-237 is listed in the OU 8-08 BRA as a COC disposed at two NRF sites (Westinghouse Electric Corporation 1996, Table 20-3). An activity of 6.67×10^{-2} Ci was estimated in GWSCREEN simulations for contaminant transport (Westinghouse Electric Corporation 1996, Table 18-5).

4.1.11.2 Mechanisms and rates of neptunium-237 release from source terms to the subsurface

The mechanism for neptunium-237 releases at INTEC was through direct disposal to the aquifer from injection well CPP-23.The rate of release was directly proportional to the rate of disposal of wastewater containing neptunium-237.

Neptunium-237 contained on RWMC solid wastes is available for surface washoff as locally recharged water infiltrates and comes into contact with those wastes. Numerical transport simulations used a best-estimated site-specific k_d of 8 mL/g for neptunium-237 (Holdren et al. 2000, Table 5-13).

A long-term secondary source of release of neptunium-237 will occur through decay of americium-241. Americium-241, with a half-life of 432 years, will decay in place because it will remain sorbed to sediments. The neptunium-237 daughter product will be much more mobile, as indicated by the small $k_{d,}$ and will move readily from this secondary source term. Based on a simulated peak risk of 4×10^{-4}, neptunium-237 was determined to be a significant groundwater exposure pathway COC at the RMWC (Holdren et al. 2000, Table 6-6).

4.1.12 Nickel-63

Nickel-63 was determined to be a COC at NRF (Table 4-1). The half-life of nickel-63 is 100.1 years.

4.1.12.1 Nickel-63 source-term locations and the activities of nickel-63 disposed to the subsurface at the INEEL

Nickel-63 is listed in the OU 8-08 BRA as a COC at the NRF (Westinghouse Electric Corporation 1996, Table 20-3). Amounts of nickel-63 disposed at the NRF are not listed, but the activity is estimated to be greater than 1×10^{-2} Ci (Westinghouse Electric Corporation 1996, p. 1-15). Nickel-63 was contained in soil contamination sites at the NRF.

4.1.12.2 Mechanisms and rates of nickel-63 release from source terms to the subsurface

GWSCREEN simulations of transport of Nickel-63 used a k_d of 1 × 102 mL/g (Westinghouse Electric Corporation 1996, Appendix F). These simulations demonstrated that nickel-63 is relatively immobile and is not a COC with respect to groundwater exposure pathways.

4.1.13 Niobium-94

Niobium-94 was determined to be a human-health COC at the RWMC (Table 4-1). The half-life of niobium-94 is 20,300 years.

4.1.13.1 Niobium-94 source-term locations and the activities of niobium-94 disposed to the subsurface at the INEEL

Niobium-94 is listed in the WAG 7 OU 13/14 BRA as an external exposure through groundwater ingestion COC at the RWMC (Holdren et al. 2000, Table 7-1). A total activity of 1.00×10^3 Ci is associated with solid wastes buried in pits and trenches (Holdren et al. 2000, Table 5-3).

4.1.13.2 Mechanisms and rates of niobium-94 release from source terms to the subsurface

Niobium-94 is available for release from the RWMC buried-waste source term primarily through dissolution from solid waste as locally derived water comes into contact with the waste. The best-estimated site-specific k_d at the RWMC of 500 mL/g (Holdren et al. 2000, Table 5-13) indicates that niobium-94 is not mobile in sediments. Based on the peak simulated risk of 8×10^{-5}, niobium-94 is not considered to be a significant groundwater exposure pathway COC at the RWMC (Holdren et al. 2000, Table 6-6).

4.1.14 Plutonium Isotopes

Plutonium isotopes were determined to be COCs at TRA, INTEC, RWMC, and NRF (Table 4-1). Half-lives of plutonium-238, -239, -240, and -244 are 87.75 years, 24,131 years, 6,537 years, and 82.6 million years, respectively.

4.1.14.1 Plutonium source-term locations and the activities of plutonium disposed to the subsurface at the INEEL

Plutonium-238 and -239/240 isotopes are listed in the OU 2-13 BRA as COCs at TRA (Burns et al. 1997, Table 11-5). Three soil-contamination sites were identified that contained plutonium isotopes with total initial plutonium-238 and –239/240 activities as used in TRA GWSCREEN simulations that was estimated to be 5.74×10^{-1} Ci and 6.10×10^{0} Ci, respectively (Burns et al. 1997, Table B-28).

Total plutonium is listed as a COC at INTEC (Lockheed Martin Idaho Technologies Company 1997, p. 2-16). Frederick et al. (1998; Table 3-1) report an estimated total plutonium activity of 2.57×10^{-4} Ci disposed to INTEC injection well CPP-23 during 1953-1986.

Plutonium-238, -239, and -240 were listed as soil and crop ingestion COCs at the RWMC (Holdren et al. 2000, Table 7-1). Total activities of 1.71×10^{4} Ci of plutonium-238, 6.49×10^{4} Ci of plutonium-239, and 1.71×10^{4} Ci of plutonium-240 are associated with solid wastes.

Plutonium-238 and -244 are listed in the OU 8-08 BRA as COCs at NRF (Westinghouse Electric Corporation 1996, Table 20-3). Activities estimated in numerical transport simulations using GWSCREEN were 6.31 × 10-2 and 6.71 × 10-4 Ci for plutonium-239 and -244, respectively (Westinghouse Electric Corporation 1996, Table 18-5).

4.1.14.2 Mechanisms and rates of plutonium-isotope release from source terms to the subsurface

Conceptual models of plutonium-isotope release from contaminated surficial sediments at the INEEL to the vadose zone generally include the attenuated release of plutonium isotopes from contaminated surficial sediments beneath disposal sites or adjacent to buried waste. This release is dependent on the solubility of the contaminant, the availability of water flux through the source, and the retardation of plutonium, which is a function of the partitioning coefficient (k_d).

TRA GWSCREEN contaminant transport modeling studies used a k_d at the source for Pu-238 and for Pu-239/240 of 1.50×10^{3} mL/g (Burns et al. 1997 p. C-77 and C-78) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that plutonium isotopes in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-77-78). Based on these results, plutonium isotopes sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths. Plutonium isotopes were not identified as COCs in TRA perched groundwater.

Plutonium isotopes associated with solid wastes disposed to pits and trenches at the RWMC are available for release through surface washoff as locally derived infiltrating water comes into contact with those wastes. The k_d was used in numerical analyses to evaluate release from the contaminated soil source term to the vadose zone. In the Pre-Draft RI/BRA, a plutonium isotope k_d of 5.1×10^{3} mL/g had been selected as the best available base site-specific value (Holdren et al. 2000, p. 5-113). Sensitivity analyses were conducted in which the k_d for the source release and for vadose-zone transport was varied from the extreme low end of measured k_ds of 22 mL/g (obtained from crushed Hanford basalts) to EPA midrange guidance values of 3.2×10^2 and 1.7×10^3 mL/g to the base value of 5.1×10^3 mL/g. Results provided a range of effects that could be compared to measured activities in sedimentary interbeds samples from a triad of wells at the RWMC. Results indicated that the midrange values (3.2×10^2 and 1.7×10^3 mL/g) did not contradict measured values (Holdren et al. 2000, Table 5-19), but that the low end value of 22 mL/g resulted in much larger plutonium activities in interbed sediments than were actually measured. Based on the best-estimated site-specific k_d of 5.1×10^3 mL/g, plutonium isotopes are not considered to be significant groundwater exposure pathway COCs at the RWMC (Holdren et al. 2000, Table 6-6).

4.1.15 Protactinium-231

Protactinium-231 was identified as a COC at the RWMC (Table 4-1). The half-life of protactinium-231 is 32,760 years.

4.1.15.1 Protactinium-231 source-term locations and the activities of protactinium-231 disposed to the subsurface at the INEEL

Protactinium-231 is listed as a groundwater ingestion COC at the RWMC (Holdren et al. 2000, Table 7-1). An activity of 8.64×10^{-4} Ci is associated with contaminated metal and debris in RWMC pits and trenches (Holdren et al. 2000, Table 5-3).

4.1.15.2 Mechanisms and rates of protactinium-231 release from source terms to the subsurface

Protactinium-231 associated with RWMC buried wastes is available for surface washoff. The best-estimated k_d of 8×10^0 mL/g indicates that protactinium-231 is readily released from the source term. However, release of protactinium-231 to groundwater exposure pathways is considered to be insignificant based on a peak simulated risk of 3×10^{-6} (Holdren et al. 2000, Table 6-6).

4.1.16 Radium-226

Radium-226 was determined to be a COC at CFA and the RWMC (Table 4-1). The half-life of radium-226 is 1,600 years.

4.1.16.1 Radium-226 source-term locations and the activities of radium-226 disposed to the subsurface at the INEEL

Radium-226 is listed as a COC at CFA. However, it was determined that radium-226 is naturally occurring.

Radium-226 is listed in the WAG 7 OU 13/14 BRA as an external exposure COC at the RWMC (Holdren et al. 2000, Table 7-1). A total activity of 6.00×10^1 Ci is associated with solid wastes buried in pits and trenches (Holdren et al. 2000, Table 5-3).

4.1.16.2 Mechanisms and rates of radium-226 release from source terms to the subsurface

Radium-226 in RWMC buried wastes is available for release from the source term primarily through surface washoff from solid waste as locally derived water comes into contact with the waste. The BRA determined that the risk from radium-226 primarily occurs through external exposure and that radium-226 does not constitute a contaminant source to groundwater at the RWMC.

4.1.17 Silver-108m

Silver-108m was determined to be a COC at the TRA (Table 4-1). Silver-108m is one of two forms of silver with an atomic mass of 108. Silver-108 has a half-life of 2.37 minutes. Silver-108m has a half-life of 127 years.

4.1.17.1 Silver-108m source-term locations and the activities of silver-108m disposed to the subsurface at the INEEL

Silver-108m is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). Five soil-contamination sites were identified that contained silver-108m with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 5.51×10^{-2} Ci (Burns et al. 1997, Table B-28).

4.1.17.2 Mechanisms and rates of silver-108m release from source terms to the subsurface

GWSCREEN numerical simulations of silver-108m transport from surface contamination sites at TRA (Burns et al. 1997 p. C-68) used a conservative screening sorption coefficient at the source of 9.00×10^{1} (EG&G 1994, Table F-1) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that silver-108m in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-68). Based on these results, silver-108m sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths.

4.1.18 Strontium-90

Strontium-90 was determined to be a COC at TAN, TRA, INTEC, RWMC, and NRF (Table 4-1). Dissolved strontium-90 was contained in wastewater disposed to the INTEC injection well and to INTEC and TRA infiltration ponds. The only measurable concentrations of strontium-90 in SRPA water have been associated with INTEC injection-well disposals. Processes that probably affect the release of strontium-90 to the subsurface from the source term include the relatively short half-life (29.1 yr) and sorption capabilities.

4.1.18.1 Strontium-90 source-term locations and activities of strontium-90 disposed to the subsurface at the INEEL

Strontium-90 is listed in the TAN OU 1-07B BRA as a COC (Kaminsky et al. 1994, Table 7-6). An estimated 5.42×10^{-2} Ci of strontium-90 were reported to be released to the TAN TSF-05 injection well during 1959-1972 (Kaminsky et al. 1994, Table 3-4). During 1972-1985, approximately 3.923×10^{-2} Ci of Sr-90 were disposed to the TSF-07 pond (Kaminsky et al. 1994, Table 3-3). The TAN TSF-07 disposal pond is an unlined, diked area encompassing approximately 35 acres. The disposal pond, constructed in 1971, has historically received wastewater containing low-level radioactive waste, and continues to receive cold process water and treated sewage effluent under a wastewater Land Application Permit.

Strontium-90 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997 table 11-5). Eight soil-contamination sites were identified that contained strontium-90 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 1.24×10^{2} Ci (Burns et al. 1997, Table B-28).

Strontium-90 is listed in the OU 3-13 BRA as a COC at INTEC (Lockheed Martin Idaho Technologies Company 1997, p. 2-16). Wastewater disposed to the injection well CPP-23 at INTEC during 1953-1984 contained about 24 Ci of strontium-90. Approximately 33 Ci of strontium-90 were reported by Robertson et al. (1974; p. 119) to have been disposed during 1962-1963 to a pit west of building CPP-603. Robertson referred to this pit as the waste disposal pit (alternatively as the covered pit, p. 116). Approximately 0.33 Ci were contained in wastewater discharged to the INTEC service-waste ponds during 1984-1998.

Strontium-90 is listed in the WAG 7 OU 13/14 BRA as a crop ingestion COC at RWMC (Holdren et al. 2000, Tables E-1 and E-2). Approximately 6.44×10^5 Ci of strontium-90 are associated with solid waste buried in RWMC pits and trenches.

Strontium-90 is listed in the OU 8-08 BRA as a COC disposed to four NRF sites (Westinghouse Electric Corporation 1996, Table 20-3). A total of 1.65 Ci were disposed to the S1W leaching pit, temporary leaching pit, and industrial waste lagoons, and the A1W leaching bed during 1953-1979 (Frederick et al. 1998, Table 4-9).

A total of 9.199 × 10-4 Ci of strontium-90 were reported disposed to the EBR-II leach pit during 1960-1975 (LATA 1990) (Frederick et al. 1998, Table 2-1).

4.1.18.2 Mechanisms and rates of strontium-90 release from source terms to the subsurface

The gradual release of strontium-90 from TAN injection well TSF-05 has been assumed to occur from the sludges around the well. Release is controlled by the tendency of strontium-90 to sorb to sediments. Release rates were simulated using a strontium k_d of 127.7 mL/g (Kaminsky et al. 1994, p. 6-7).

Strontium-90 was released directly to the SRPA by wastewater disposal through the INTEC CPP-23 injection well. Direct release of strontium-90 limited contact with sediments and reduced sorption to clay minerals. The activity was released at the rate of disposal.

Disposal of strontium-90 to infiltration ponds and pits at TAN, TRA, INTEC, NRF, and EBR-II enhanced sorption as strontium-90 came into contact with pond-bottom sediments, surficial alluvium, and sedimentary interbeds in the vadose zone. The capacity of pond-bottom sediments to sorb strontium-90 may attenuate release to underlying sediment and basalt. Factors that may affect subsequent releases may include changes in the flux and chemistry of disposed wastewater and other sources of infiltrating surface water. Release from pond-bottom sediments requires a flux of water that can come into contact with strontium-90 to expedite ion exchange processes. TRA GWSCREEN numerical simulations used a k_d at the source of 2.40×10^1 (Burns et al. 1997, p. C-79) to control release from these secondary sources and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that strontium-90 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-79). The TAN Remedial Investigation Final Report concluded that no evidence exists that direct migration of contaminants in surface water, sediment/soils, and the vadose zone contributes to groundwater contamination at TAN (Kaminsky et al. 1994, p. 5-1). Based on these results, strontium-90 sorbed to surficial sediments at TAN, TRA, INTEC, NRF, and EBR-II does not constitute a contaminant source term for groundwater exposure paths.

Strontium-90 associated with solid wastes buried at the RWMC is believed to be released from the buried-waste source term through dissolution and through surface washoff. Approximately 6.088×10^5 Ci of strontium-90 are estimated to be available for release through dissolution. Another 3.575×10^4 Ci are estimated to be available for release through surface washoff. Both mechanisms require flux of infiltrating water that comes into contact with the buried solid wastes. Strontium-90 is principally considered to be a COC for crop ingestion pathways at the RWMC and does not constitute a groundwater exposure COC.

4.1.19 Technetium-99

Technetium-99 was determined to be a COC at the RWMC (Holdren et al. 2000, Table E-1). The half-life of technetium-99 is 213,000 years.

4.1.19.1 Technetium-99 source-term locations and the activities of technetium-99 disposed to the subsurface at the INEEL

Technetium-99 was disposed to the warm waste pond at TRA. A total activity of 8.91 Ci was released during 1972-1990. No record of disposal is available during 1962-1971 and 1991-1993.

Technetium-99 is listed in the WAG 7 OU 13/14 BRA as a groundwater ingestion COC at the RWMC (Holdren et al. 2000, Table 7-1). An activity of 6.05×10^1 Ci is associated with reactor operations waste, subassembly hardware, facility decontamination and demolition, and other wastes buried in pits and trenches at the RMWC (Holdren et al. 2000, Table 5-3).

4.1.19.2 Mechanisms and rates of technetium-99 release from source terms to the subsurface

Release of technetium-99 from solid wastes occurs through surface washoff as water infiltrates surficial sediments as local recharge and comes into contact with buried wastes. The best-estimated site-specific kd was determined to be 0 at the RWMC (Holdren et al. 2000, Table 5-13). Based on the peak simulated risk of 4×10^{-4} (Holdren et al. 2000, Table 6-6), technetium-99 is considered to be a groundwater exposure pathway COC at the RWMC.

4.1.20 Thorium-228

Thorium-228 was determined to be a COC (Table 4-1) at the TRA (Burns et al. 1997, Table 11-5). The half-life of thorium-228 is 1.913 years.

4.1.20.1 Thorium-228 source-term locations and the activities of thorium-228 disposed to the subsurface at the INEEL

Thorium-228 is listed in the OU 2-13 BRA as a COC at TRA (Burns et al. 1997, Table 11-5). One soil-contamination site was identified that contained thorium-228 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 9.33×10^{-3} Ci (Burns et al. 1997, Table B-28). Because of the 1.913-year half-life and the strong sorbing tendency of thorium-228, the source term activity is expected to be undetectable.

4.1.20.2 Mechanisms and rates of thorium-228 release from source terms to the subsurface

TRA GWSCREEN Sorption coefficient at source 1.00×10^2 (Burns et al. 1997, p. C-80) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that thorium-228 in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-80). Based on these results, thorium-228 sorbed to surficial sediments at the TRA does not constitute a contaminant source term for groundwater exposure paths.

4.1.21 Thorium-229

Thorium-229 was determined to be a COC (Table 4-1) at the RWMC (Holdren et al. 2000, Table 7-1). The half-life of thorium-229 is 7,340 years.

4.1.21.1 Thorium-229 source-term locations and the activities of thorium-229 disposed to the subsurface at the INEEL

Thorium-229 is listed in the OU 7-13/14 BRA as a groundwater ingestion COC at RWMC (Holdren et al. 2000, Table 7-1). An activity of 6.81×10^{-6} Ci of thorium-229 is associated with contaminated structural materials, compacted wastes, beryllium blocks, and other wastes in pits and trenches (Holdren et al. 2000, Table 5-3).

4.1.21.2 Mechanisms and rates of thorium-229 release from source terms to the subsurface

Thorium-229 is available for release from RWMC buried wastes through surface washoff. The best-estimated site-specific kd for thorium-229 of 500 mL/g indicates that thorium-229 strongly sorbs to sediments and is not mobile (Holdren et al. 2000, Table 5-13). The peak simulated risk of 4×10^{-7} (Holdren et al. 2000, Table 6-6) indicates that thorium-229 is not a significant groundwater exposure pathway COC at the RWMC.

4.1.22 Thorium-230

Thorium-230 was determined to be a COC (Table 4-1) at the RWMC (Holdren et al. 2000, Table 7-1). The half-life of thorium-230 is 77,000 years.

4.1.22.1 Thorium-230 source-term locations and the activities of thorium-230 disposed to the subsurface at the INEEL

Thorium-230 is listed in the OU 7-13/14 BRA as a groundwater ingestion COC at the RWMC (Holdren et al. 2000, Table 7-1). An activity of 3.13×10^{-2} Ci of thorium-230 soil, scrap metal, equipment, contaminated debris, and noncompacted TAN hot shop wastes that are buried in pits and trenches (Holdren et al. 2000, Table 5-3).

4.1.22.2 Mechanisms and rates of thorium-230 release from source terms to the subsurface

Thorium-230 is available for release from RWMC buried wastes through surface washoff. The best-estimated site-specific kd for thorium-230 of 500 mL/g indicates that thorium-230 strongly sorbs to sediments and is not mobile (Holdren et al. 2000, Table 5-13). The peak simulated risk of 7×10^{-7} (Holdren et al. 2000, Table 6-6) indicates that thorium-230 is not a significant groundwater exposure pathway COC at the RWMC.

4.1.23 Tritium

Tritium was determined to be a COC at TAN and at INTEC (Table 4-1). Tritium has been contained in liquid and solid wastes that have been disposed to the subsurface at several INEEL locations. An estimated activity exceeding 32,000 Ci has been disposed in liquid form since 1952 through INEEL injection wells and infiltration ponds. Rates and methods of disposal and the relatively short radioactive decay half-life of tritium (12.38 years) have controlled the distribution and amount of tritium releases to the subsurface and eventually to the SRPA.

4.1.23.1 Tritium source-term locations and the activities of tritium disposed to the subsurface at the INEEL

Tritium is listed in the TAN OU 1-07B BRA as a COC (Kaminsky et al. 1994, Table 7-6). Approximately 5.32×10^1 Ci of tritium were released to the TAN TSF-05 injection well during 1959-1972 (Kaminsky et al. 1994, Table 3-4). A total of 1.072×10^1 Ci of tritium were released to the TSF-07 disposal pond during 1972-1985 (Kaminsky et al. 1994, Table 3-3). Tritium is considered as a groundwater exposure pathway COC at TAN.

Tritium was disposed in aqueous wastes to the warm-waste ponds at TRA during 1952-1993 and to the service-waste infiltration ponds at INTEC during 1984-2002. Approximately 1.032×10^4 Ci of tritium were disposed in aqueous wastes to the TRA warm-waste ponds during 1952-1993 (Frederick et al. 1998). Tritium is considered as a COC with respect to groundwater exposure pathways at TRA.

Tritium was contained in wastewater disposed directly to the SRPA through the CPP-23 injection well at INTEC during 1953-1984. A total of 2.01×10^4 Ci was disposed to the injection well in 1.1×10^{10} gal of wastewater during this period (Rodriguez et al. 1997, Table 4-1).

Tritium is listed in the INTEC OU 2-12 BRA as a COC (Lockheed Martin Idaho Technologies Company 1997, p. 2-16). Approximately 8.8×10^2 Ci of tritium were disposed to the INTEC service-waste ponds during 1984-1996 (Frederick et al. 1998).

Tritium is contained in activated beryllium reactor components buried at RWMC between 1970 and 1993. A total activity of about 1,000,000 Ci is estimated to remain of the original activity disposed. The amount of activity retained in the activated beryllium is not known. This activity represents an estimated 97% of the total inventory of tritium disposed to the subsurface at the INEEL.

Tritium was disposed to in aqueous wastes to pits at NRF. A total of 112.33 Ci were reported disposed to the S1W industrial waste lagoons and the A1W leaching bed during 1953-1979 (Frederick et al. 1998, Table 4-9).

An estimated 3,574 Ci of tritium were disposed to the industrial waste pond at ANL-W during 1973-1994 (Frederick et al. 1998, Table 2-2). An additional 0.1806 Ci was disposed to the EBR-II leach pit during 1960-1975 (Frederick et al. 1998, Table 2-1, (derived from LATA 1990)).

4.1.23.2 Mechanisms and rates of tritium release from source terms to the subsurface

Tritium disposed to the CPP-23 injection well was in an aqueous form. Release from the injection well to the SRPA took place directly in response to disposal rates. Tritium moved from the injection well into local and subregional flow systems unaffected by chemical interactions between rock and water or between wastewater and groundwater.

Tritium was disposed in aqueous form to the TRA warm-waste ponds and INTEC service-waste ponds. Tritium is considered to be a conservative constituent that is not attenuated by geochemical processes. Release of tritium from the ponds to the subsurface occurred directly with infiltrating wastewater and was dependent on the hydraulic properties of pond-bottom sediments. Commonly, plugging of pond bottoms reduced infiltration rates and required construction of additional ponds. Subsurface movement of wastewater away from the ponds took place primarily as vertical, saturated or unsaturated flow. Source term release to the subsurface (vadose zone and SRPA) was directly related to disposal rates with the exception of residual tritiated water that remained in the vadose zone after disposals ceased. Release of tritium from this secondary source term is controlled by local recharge and is much smaller then the rate of release during disposal. Radioactive decay will account for most of this residual tritium source before it reaches the SRPA.

The annual rate of release at the surface was directly in response to disposal and infiltration rates. Tritium moved with flow through the vadose-zone, moving through fractured basalts and sedimentary interbeds to the SRPA. Cessation of disposal to the warm waste pond in 1994 reduced the flux through the vadose zone. Some residual water containing tritium probably remains as a secondary source term. However, downward movement of tritium to the aquifer from this source term probably is slow and much of the residual source term will be removed through radioactive decay before reaching the SRPA.

Tritium in activated beryllium is contained as tritiated hydrogen gas (T_2) at interstitial sites between metal grain boundaries. Beryllium in contact with INEEL soil corrodes at about the same rate as carbon steel[3]. Corrosion destroys the metal grain structure and is the dominant release mechanism. Tritiated hydrogen is rapidly oxidized in surface soil (McFarlane et al. 1979). The ratio of tritiated hydrogen to tritiated water vapor has been measured in soil gas samples taken near the beryllium (Ritter and McElroy 1999). The results indicate that essentially all the released H-3 is oxidized to form tritiated water near the point of release in the deeper subsurface environment.

Tritiated water can migrate in gaseous form and liquid form. Advection of liquid water through the subsurface causes similar advection of tritiated water – tritium is a good tracer for water movement in the liquid phase. Tritiated water vapor diffuses through the soil gas, driven by the concentration gradient of tritiated water rather than the concentration gradient of total water. However, vapor diffusion is strongly affected by partitioning of tritiated water into the liquid phase. Vapor diffusion generally would cause very slow migration in all but the driest soils (e.g., dry sands). Advective transport of tritiated water vapor can cause relatively high migration rates, but is not well understood.

4.1.24 Uranium Isotopes

Uranium isotopes were determined to be COCs at TAN, TRA RWMC, and NRF (Table 4-1). The half-life of uranium-234 is 245,000 years; the half-life of uranium-235 is 704,000,000 years; the half-life of uranium-236 is 23,400,000 years; and the half-life of uranium-238 is 4,468,000,000 years.

4.1.24.1 Uranium source-term locations and the activities of uranium isotopes disposed to the subsurface at the INEEL

Uranium-234 is listed in the TAN OU 1-07B BRA as a COC (Kaminsky et al. 1994, Table 7-6). Uranium-234 concentrations of 5 to 7 pCi/L were detected in water samples from injection well TSF-05 (Kaminsky et al. 1994, Table 7-6). The source-term activity is not known.

Uranium-234 is listed in the OU 2-13 BRA as COC at TRA (Burns et al. 1997, Table 11-5). Three soil-contamination sites were identified that contained uranium-234 with a total initial activity as used in TRA GWSCREEN simulations that was estimated to be 1.10×10^{-1} Ci (Burns et al. 1997, Table B-28). Additionally, uranium-235 and –238, while not determined to be COCs, were identified at two sites, with total initial activities as used in TRA GWSCREEN simulations of 1.18×10^{-3} Ci, and 4.73×10^{-2} Ci, respectively.

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[3] Results of corrosion testing of buried carbon steel and beryllium coupons show a maximum thickness loss rate of 12 μm/y (0.460 mils/y) and 8.3 μm/y (0.327 mils/ y), respectively, after 3 years of burial in typical INEEL soil (Adler-Flitton et al. 2001).

Uranium-233, -234, -235, -236, -238 are listed in the WAG 7 OU 13/14 BRA as groundwater ingestion COCs (Holdren et al. 2000, Table 7-1). Activities of 1.51×10^0 Ci of uranium-233, 6.74×10^1 Ci of uranium-234, 5.54×10^0 Ci of uranium-235, 2.86×10^0 Ci of uranium-236, and 1.17×10^2 Ci of uranium-238 are associated with depleted uranium, rubble, and other wastes.

Uranium-235 is listed as a COC in the OU 8-08 BRA at NRF (Westinghouse Electric Corporation 1996 20-3). Uranium-234 is not listed as an NRF COC. GWSCREEN used activities of 2.77×10^{-2} and 1.88×10^{-3} Ci for uranium-234 and -235, respectively (Westinghouse Electric Corporation 1996, Table 18-5).

ANL-W A total of 1.33×10^{-6} Ci of uranium-238 reported disposed to the EBR-II leach pit during 1960-75 (Frederick et al. 1998, Table 2-1).

4.1.24.2 Mechanisms and rates of uranium-isotope release from source terms to the subsurface

Release of uranium-234 from the TAN injection well was directly related to the rate of injection and the concentration. Some uranium-234 may be contained in the sludges filling voids around the injection well.

The conceptual model of uranium-isotope release from TRA source terms consists of leaching from contaminated soils and pond-bottom sediments. This release was simulated by GWSCREEN (Burns et al. 1997, p. C-83-85) using a sorption coefficient of 6.00×10^0 (EG&G Idaho 1994, Table F-1) and a natural infiltration rate of 10 cm/year (Burns et al. 1997, Table 5-2). TRA GWSCREEN results indicated that uranium isotopes in surficial sediments will not reach the SRPA (Burns et al. 1997, p. C-83 through C-85). Based on these results, uranium isotopes sorbed to surficial sediments at the TRA do not constitute a contaminant source term for groundwater exposure paths.

Uranium isotope release from RWMC buried wastes is attributed primarily to surface wash off. From Holdren et al. (2000 p. 6-24), RWMC solubility limits for uranium (base case 5.98×10^{-4} g/cm^3 and sensitivity cases 9.3×10^{-7} g/cm^3 and 9.3×10^{-11} g/cm^3, based on work by Hull and Pace (2000)) were used in simulations to address uncertainties in release of uranium. The best-estimated RWMC-specific k_d of 6 mL/g (Holdren et al. 2000, Table 5-13) indicates that the uranium isotopes are relatively mobile in sediments. The peak simulated risks for uranium-234, -235, -236, and -238 all exceeded 10^{-4} (Holdren et al. 2000, Table 6-6). These radionuclides are considered to be groundwater exposure pathway COCs at the RWMC.

4.2 Organic Chemicals

Organic chemicals have been disposed to the subsurface through injection wells, infiltration ponds, and as buried waste in landfills. These organic chemicals primarily consist of solvents in wastes generated at the INEEL or offsite facilities. Organic species identified as COCs in INEEL BRAs include acrylonitrile, carbon tetrachloride, methylene chloride, trichloroethane (TCE), PCE, cis-1,2-DCE, and trans-1,2-DCE (Table 4-1).

4.2.1 Acrylonitrile

Acrylonitrile was identified as an organic COC at the TRA (Table 4-1).

4.2.1.1 Acrylonitrile source-term locations and the amounts of acrylonitrile disposed to the subsurface at the INEEL

Acrylonitrile is listed as a COC in the OU 2/13 BRA (Burns et al. 1997, Table 11-5). Acrylonitrile has been identified in contaminated soils around the retention basin, cold waste sampling pit, and cold waste sump pit. A total of 2.54 × 10^7 g of acrylonitrile was estimated for the initial mass in the GWSCREEN transport modeling studies.

4.2.1.2 Mechanisms and rates of acrylonitrile release from source terms to the subsurface

The simulated risk associated with acrylonitrile was calculated to be less than 10^{-6}. Subsequently, through the BRA, acrylonitrile was determined not be a groundwater exposure pathway COC (Burns et al. 1997, Table B-45).

4.2.2 Carbon Tetrachloride

Carbon tetrachloride was identified as a COC at the RWMC (Table 4-1).

4.2.2.1 Carbon tetrachloride source-term locations and the amounts of carbon tetrachloride disposed to the subsurface at the INEEL

Carbon tetrachloride is listed in the OU 7-13/14 BRA as a groundwater ingestion COC (Holdren et al. 2000, Table 7-1). Carbon tetrachloride was contained in semisolid wastes disposed in drums to pits and trenches at the RWMC. A total mass of 8.21 × 108 g of carbon tetrachloride was disposed to the RWMC SDA during 1952-1978 (Holdren et al. 2000, Table 4-2).

4.2.2.2 Mechanisms and rates of carbon tetrachloride release from source terms to the subsurface

Release of carbon tetrachloride to the subsurface at the RWMC has occurred as waste drums have deteriorated. The rate of release is directly related to drum failure and occurs by vapor diffusion out of the sludge.

4.2.3 Methylene chloride

Methylene chloride was identified as a COC at the RWMC (Table 4-1).

4.2.3.1 Methylene chloride source-term locations and the amounts of methylene chloride disposed to the subsurface at the INEEL

Methylene chloride is listed as a groundwater exposure pathway COC at RWMC (Holdren et al. 2000, Table 7-1). A total mass of 1.41 × 10^7 g of methylene chloride was disposed in sludges to the RWMC SDA during 1952-1970 (Holdren et al. 2000, Table 4-2).

4.2.3.2 Mechanisms and rates of methylene chloride release from source terms to the subsurface

Methylene chloride is assumed to move by vapor diffusion from buried sludges. Methylene chloride is soluble and readily mobile in the subsurface.

4.2.4 Trichloroethylene

Trichloroethylene (TCE) was identified as a COC at TAN (Table 4-1).

4.2.4.1 Trichloroethylene source-term locations and the amounts of trichloroethylene disposed to the subsurface at the INEEL

Trichloroethylene is listed as the primary COC at TAN (Kaminsky et al. 1994, Table 7-6). TCE was injected directly into the SRPA through the injection well TSF-05 during 1953-1972. TCE was as contained in a liquid waste mixture that included organic sludge, treated sanitary sewage, process wastewaters, and low-level radioactive waste streams (Kaminsky et al. 1994). The TAN TSF-05 injection well was drilled in 1953 to 305 ft in depth, and was completed with 12-in. perforated casing with a gravel pack filling the annulus between the casing and the 20-in. borehole. This well was designed for the disposal of liquid wastes and concentrated evaporator sludges generated during the operations of the various TAN facilities.

Historical records provide little definitive information on the types and volumes of organic wastes disposed via the injection well; however, from as little as 350 gal to as much as 35,000 gal of TCE have been estimated to have been disposed in the well during its period of operation. Recent inverse numerical modeling studies have been used to estimate that the amount of TCE disposed to the injection well may have been 1,100 gal.

4.2.4.2 Mechanisms and rates of TCE release from source terms to the subsurface

Two mechanisms of TCE release to the aquifer from the injection well have been identified. First, TCE has been released directly to groundwater from injection. The rate of direct release was related to the rate of disposal and to the concurrent development of the sludge ring around the injection well.

Second, release is occurring from a secondary source that is derived from injected wastes that formed a cylinder of sludge in the fractured basalts around injection well TSF-05. These sludges have reduced permeability adjacent to the injection well and have retained TCE, which is released gradually from this secondary source as groundwater comes into contact with the sludge. TCE slowly leaches from this residual source that continues to contaminate groundwater at TAN (Kaminsky et al. 1994). Geophysical data indicate that this secondary source extends more than 100 ft from the injection well (Bukowski and Sorenson 1998). This residual source area continued to contaminate groundwater prior to remediation activities in 1996. In-situ bioremediation presently is being used to reduce the mass of TCE available for groundwater transport.

4.2.5 Tetrachloroethylene

Tetrachloroethylene (PCE) was identified as a COC at TAN (Table 4-1).

4.2.5.1 PCE source-term locations and the amounts of PCE disposed to the subsurface at the INEEL

PCE is listed in the TAN OU 1-07B BRA as a COC because it has been detected in groundwater adjacent to the TAN injection well (Kaminsky et al. 1994, Table 7-6). The amount of PCE disposed to the injection well is not known.

4.2.5.2 Mechanisms and rates of PCE release from source terms to the subsurface

Release of PCE from TAN injection well TSF-05 occurred directly to the SRPA and was dependent on the rate of injection and the concentration of PCE in the injection water. A secondary source has continued as PCE gradually has leached from the sludges around the injection well.

4.2.6 Cis-1,2-DCE

Cis-1,2-DCE was identified as a COC at TAN (Table 4-1).

4.2.6.1 Cis-1,2-DCE source-term locations and the amounts of cis-1,2-DCE disposed to the subsurface at the INEEL

Cis-1,2-DCE is listed in the TAN OU 1-07B BRA as a COC because it has been detected in groundwater adjacent to the TAN injection well (Kaminsky et al. 1994, Table 7-6). The amount of cis-1,2-DCE disposed to the injection well is not known.

4.2.6.2 Mechanisms and rates of cis-1,2-DCE release from source terms to the subsurface

Release of cis-1,2-DCE from TAN injection well TSF-05 occurred directly to the SRPA and was dependent on the rate of injection and the concentration of PCE in the injection water. A secondary source has continued as cis-1,2-DCE gradually has leached from the sludges around the injection well.

4.2.7 Trans-1,2-DCE

Trans-1,2-DCE was identified as a COC at TAN (Table 4-1).

4.2.7.1 Trans-1,2-DCE source-term locations and the amounts of trans-1,2-DCE disposed to the subsurface at the INEEL

Trans-1,2-DCE is listed in the TAN OU 1-07B BRA as a COC because it has been detected in groundwater adjacent to the TAN injection well (Kaminsky et al. 1994, Table 7-6). The amount of trans-1,2-DCE disposed to the injection well is not known.

4.2.7.2 Mechanisms and rates of Trans-1,2-DCE release from source terms to the subsurface

Release of trans-1,2-DCE from TAN injection well TSF-05 occurred directly to the SRPA and was dependent on the rate of injection and the concentration of PCE in the injection water. A secondary source has continued as trans-1,2-DCE gradually has leached from the sludges around the injection well.

4.3 Trace Elements

Trace elements identified in INEEL BRAs as COCs include arsenic, beryllium, and chromium (Table 4-1). Subsequent sections describe these trace element source terms, including location, amount, mechanism, and significance as groundwater pathway COCs.

4.3.1 Arsenic

Arsenic was determined to be a COC at TRA (Table 4-1).

4.3.1.1 Arsenic source-term locations and the amounts of arsenic disposed to the subsurface at the INEEL

Arsenic was listed as a COC for groundwater exposure pathways at TRA (Burns et al. 1997, p. C-42). An estimated mass of 6.41×10^8 mg of arsenic was used in GWSCREEN contaminant modeling activities.

Arsenic was not listed as a COC at NRF. GWSCREEN simulations used an estimated mass of 9.75×10^9 mg of arsenic (Westinghouse Electric Corporation 1996, Table 18-5).

4.3.1.2 Mechanisms and rates of arsenic release from source terms to the subsurface

Arsenic disposed to the surface at TRA was determined to be relatively mobile with respect to groundwater exposure pathways. TRA simulations used a kd of 3×10^0 mL/g (EG&G Idaho 1994, Table F-1).

4.3.2 Beryllium

Beryllium was determined to be a COC at TRA and CFA (Table 4-1).

4.3.2.1 Beryllium source-term locations and the amounts of beryllium disposed to the subsurface at the INEEL

Beryllium is listed as a COC in the OU 2/13 BRA (Burns et al. 1997, Table 11-5). Beryllium was contained in soil contamination sites at the TRA. The mass of beryllium used in GWSCREEN simulations was 1.88×10^7 mg (Burns et al. 1997, p. C-44).

Beryllium was listed as a COC in the OU 4/12 BRA. An estimated 1 m^3 of beryllium chips was disposed to CFA landfill II.

4.3.2.2 Mechanisms and rates of beryllium release from source terms to the subsurface

GWSCREEN simulations for release of beryllium to groundwater exposure paths used a k_d of 2.50×10^2 mL/g (EG&G Idaho 1994, Table F-1). Based on these simulations, it was determined that beryllium does not constitute a groundwater exposure pathway COC at the TRA.

The BRA for OU 4-13 identified no CFA contamination sites that provided groundwater exposure pathways. Beryllium was not identified as a groundwater exposure pathway COC at CFA.

4.3.3 Chromium

Chromium is listed as a COC at TRA and INTEC (Table 4-1).

4.3.3.1 Chromium source-term locations and the amounts of chromium disposed to the subsurface at the INEEL

Chromium is one of the principal groundwater COCs at the TRA (Burns et al. 1997, p. 11-9). Hexavalent chromium was used at the TRA as a corrosion inhibitor in cooling-tower blowdown water. Wastewater from this process was disposed to the TRA warm-waste pond during 1952-1964 and to the

TRA injection well during 1964-1972. A total of 31,131 lb (1.41×10^4 kg) of hexavalent chromium is estimated to have been disposed to the TRA injection well during 1965-1982 (Hull 1989).

TETRAD modeling results for INTEC included an estimated chromium mass within the vadose zone at TRA of 8.07×10^3 kg (Burns et al. 1997, p. 5-24). GWSCREEN estimates of hexavalent and total chromium

Chromium is not listed as a COC at NRF. GWSCREEN used a mass of 4.36×10^{10} mg (Westinghouse Electric Corporation 1996, Table 18-5).

4.3.3.2 Mechanisms and rates of chromium release from source terms to the subsurface

Chromium in the hexavalent state does not sorb strongly and tends to remain in solution. Hull (1989; p. 115) states that hexavalent chromium that was initially disposed to the TRA warm-waste pond was transported conservatively to the aquifer. However, as organic matter accumulated on the pond bottom from algal growth, it probably interacted with the strongly oxidizing hexavalent chromium, converting it to the trivalent form. In the trivalent form, chromium would precipitate, be sorbed, or exchanged on solid surfaces, removing it from solution. Hull states that most of the chromium disposed to the infiltration ponds during 1952-1964 probably remains in the upper several feet of pond bottom sediments.

In contrast, chromium disposed in wastewater directly to the SRPA through the TRA injection well probably remained in its hexavalent form and was conservatively transported with flow in the aquifer as it was released from the screened section of the injection well.

4.4 Inorganic Chemicals

Inorganic chemicals have been disposed to the subsurface at the INEEL. Wastewater containing inorganic chemicals has been disposed to infiltration ponds, ditches, and injection wells. Solid wastes have been buried at INEEL landfill facilities. These inorganic chemicals, primarily derived from industrial and water-conditioning processes at INEEL and offsite facilities, include sodium, chloride, sulfate, and nitrate. Concentrations of these chemicals have been detected in water from the SRPA and from perched-water bodies beneath facility infiltration ponds. The only inorganic chemical identified in INEEL BRAs as a COC has been nitrate (Table 4-1).

4.4.1 Nitrate

Nitrate was identified as a COC at the RWMC (Table 4-1).

4.4.1.1 Nitrate source-term locations and the amounts of nitrate disposed to the subsurface at the INEEL

Nitrates are listed in the WAG 7 OU 13/14 BRA as groundwater ingestion COCs (Holdren et al. 2000, Table 7-1). A total of 1.03×10^9 mg of nitrate are associated with acidic aqueous liquids and nitrate salts (Holdren et al. 2000, Table 5-3).

4.4.1.2 Mechanisms and rates of nitrate release from source terms to the subsurface

Nitrates associated with RWMC buried wastes are available for surface wash off (Holdren et al. 2000,Table 5-3). Based on a best-estimated site-specific k_d of 0, nitrate is mobile in the subsurface and will move with available local recharge.

4.5 References

Burns, D.E., K.M. Davis, S.C. Flynn, J.F. Keck, N.L. Hampton, A.H. Owen, R.L. VanHorn, February 1997, *Comprehensive Remedial Investigation/Feasibility Study for the Test Reactor Area Operable Unit 2-13 at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10531, Rev. 0, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Bukowski, J.M., and K.S. Sorenson, 1998, *Site Conceptual Model: 1996 activities, data analysis, and interpretation – Test Area North Operable Unit 1-07B*, INEL/EXT-97-00556, Prepared for the U.S. Department of Energy Idaho Operations Office.

Burgess, J., S. Flynn, P. Jessmore, J. Keck, S. McCormick, M. Rohe, I. Stepan, R. VanHorn, S. Fu, L. Trozollo, July 2000, *Comprehensive remedial investigation/feasibility study for the Central Facilities Area Operable Unit 4-13 at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10680, Rev. 1, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Cecil, L.D., J.A. Welhan, J.R. Green, S.K. Frape, and E.R. Sudicky, 2000, “Use of chlorine-36 to determine regional-scale aquifer dispersivity, eastern Snake River Plain aquifer,” *Idaho: Nuclear Instruments and Methods in Physics Research B*, v. 172, p. 679-687.

Department of Energy, August 2001, *Comprehensive Remedial Investigation/Feasibility study for Waste Area Group 6 and 10 Operable Unit 10-04*, DOE/ID 10807, Rev. 0, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Dicke, C.A., 1997, *Distribution coefficients and contaminant solubilities for the Waste Area Group 7 Baseline Risk Assessment*, INEL/EXT-97-00201, Idaho National Engineering and Environmental Laboratory. Lockheed Martin Idaho Technologies Company.

Dicke, C.A., 1998, *Carbon-14 distribution coefficients measured from batch experiments on SDA sediments*, INEEL/INT-98-00068 Design File EDF-RWMC-1011, Lockheed Martin Idaho Technologies Company, Engineering.

EG&G Idaho, Inc., January 1994, *Track 2 sites: Guidance for assessing low probability hazard sites at the INEL*, DOE/ID 10389, Rev. 6, EG&G Idaho, Idaho Falls, Idaho.

Holdren, K.J., C.M. Hiaring, D.E. Burns, N.L. Hampton, B.J. Broomfield, E.R. Neher, R.L. Van Horn, I.E. Stepan, R.P. Wells, R.L. Chambers, L. Schmeising, R. Henry, January 1999, *Waste Area Group 5 Operable Unit 5-12 Comprehensive Remedial Investigation/Feasibility Study*, DOE/ID-10607, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Holdren, K.J., B.H. Becker, N.L. Hampton, L.D. Koeppen, S.O. Magnuson, T.J. Meyer, G.L. Olson, A.J. Sondrup, April 2000, “Waste Area Group 7 Operable Unit 7-13/14 Pre-Draft Remedial Investigation and BRA (Draft),” DOE/ID-10995, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Hull, L.C., October 1989, *Conceptual model and description of the affected environment of the TRA Warm Waste Pond (Waste Management Unit TRA-03)*, EGG-ER-8644, EG&G Idaho, Inc., Idaho Falls, Idaho.

INEEL, October 1998, “Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL—Part B,” FS Supplemental Report, vols 1 and 2, DOE/ID-10619, U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Kaminsky, J.F., K.N. Keck, A.L. Schafer-Perini, C.F. Hersley, R.P. Smith, G.J. Stormberg, A.H. Wylie, January 1994, *Remedial Investigation Final Report with addenda for the Test Area North Groundwater Operable Unit 1-07B at the Idaho National Engineering Laboratory*. EGG-ER-10643, EG&G Idaho, Inc., Idaho Falls, Idaho.

Keck, K.N., I. Porro, A.J. Sondrup, S.H. McCormick, and S.M. Lewis, November 1994, *Remedial Investigation/Feasibility Study Report for Operable Unit 4-12: Central Facilities Area Landfills I, II, and III at the Idaho National Engineering Laboratory*, INEL-94/0124, Idaho National Engineering Laboratory, Idaho Falls, Idaho.

Lee, S.D., M.J. Rohe, A.S. Rood, I.E. Stepan, December 1997, “Remedial Investigation/Feasibility Study for Argonne National Laboratory-West Operable Unit 9-04 at the Idaho National Engineering and Environmental Laboratory (FINAL),” W7500-0000-ES-02, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Lewis, S.M., P.O. Sinton, M.J. Condran, J.W. Gordan, June 1992, *Remedial Investigation Report for the Test Reactor Area perched water system (Operable Unit 2-12)*, EGG-WM-10002, Rev. 0, EG&G Idaho, Inc., Idaho Falls, Idaho.

Lockheed Martin Idaho Technologies Company, November 1997, *Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL—Part B, FS Report*, DOE/ID-10572 (on CD), Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho, , U.S. Department of Energy Idaho Operations Office, Idaho Falls, Idaho.

Rodriguez, R.R., A.L. Schafer, J. McCarthy, P. Martian, D.E. Burns, D.E. Raunig, N.A. Burch, R.L. Van Horn, November 1997, “Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL—Part A, RI/BRA Report (Final),” DOE/ID-10534, Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho.

Westinghouse Electric Corporation, 1996, *Final NRF Comprehensive Remedial Investigation/Feasibility Study, Waste Area Group 8, Naval Reactors Facility, Idaho Falls, Idaho*, prepared for the U.S. Department of Energy.

5. SUMMARY AND IMPLICATIONS

Many contaminant source terms have been identified at the INEEL. Although INEEL attempted to consolidate its waste disposals and does not have numerous burial grounds like other DOE sites, the use of injection wells, infiltration ponds, and other surface releases of contaminated liquids has been widespread. The primary buried waste site, the RWMC, received waste from numerous onsite and offsite generators, resulting in a large and varied disposal inventory. In addition to the historic releases described in this document, there are many other stored or in process wastes that need to be evaluated as potential future source terms.

Extensive investigations of potential source-term release sites have been completed at the INEEL. Hundreds of individual sites have been investigated under the CERCLA program. These sites ranged in scope from the TRA "hot tree" to the SDA burial grounds. INEEL consistently and comprehensively examined known release sites as well as sites that were identified by sampling or unexpected detection of radioactivity. The majority of sites that posed risk for workers or future users of the site have been remediated.

The most uncertain and the most important parameter affecting the prediction of groundwater impact is the initial estimate of the mass of contaminants available to the environment. This document confirms that, where inventory information was insufficient to support modeling, investigators typically made bounding assumptions to evaluate the maximum potential risk. No source-term sites that were identified had detailed information available regarding release. All inventories described in the RI/FS documents were developed with a series of simplifying assumptions, each of which tended to increase the total contaminant mass. The cumulative result is that the risk to the aquifer is probably overestimated.

It is notable that significant research has been performed at the RWMC to recreate the disposal inventory based on generator process knowledge. At least for weapons production wastes, the best estimates used in the most recent modeling (DOE-ID 2002a) are expected to be reasonably accurate. Work with other generators (Navy and other reactor operations) is ongoing to better refine non-weapons waste streams.

Another conclusion is that a specific conceptual model of release been developed only at RWMC. By adopting DOE code for evaluating releases from low-level radioactive repositories, researchers have developed way to calculate two-dimensional, annual releases of contaminants from the SDA. Other sites, however, used the simplistic, but conservative approach of assuming that all inventory was immediately released for transport upon disposal. For example, at the INTEC tank farm, several known distribution system leaks released large volumes of contaminated solutions to the surficial sediments. In the comprehensive RI/FS, these releases were modeled as though they had been applied directly into the vadose zone transport system (at the uppermost basalt layer) with no attenuation in surficial soils.

Not only is the question of the mass or activity of each contaminant initially available for release highly uncertain, the mass and activity of contaminants already in the aquifer and contaminants yet to be released and transported also is not well characterized. For purposes of the WIP, researchers may want to limit future studies to the residual, active source terms. It could be assumed that the impact of past releases can be estimated simply by taking the current concentration profiles within the aquifer. In the cases of injection wells, infiltration ponds, and other points of release, it may be irrelevant to attempt to recreate the original disposal history to calculate the impact to groundwater, as the impact may be able to be directly measured. This approach is complicated where the full inventory has been retained somewhere in the waste zone/vadose zone system and has yet to be released. There is a need to distinguish between historical source terms and contemporary source terms which continue to exhibit some level of attenuated release.

Another observation that was made during this activity was that, with the exception of the use of DUST-MS to calculate leaching, diffusion, and dissolution of contaminants from the various RWMC waste forms, source term release models have not been developed separately from models of fate and transport in the vadose zone. Clearly, an important area of research is the development of conceptual models and associated numerical techniques to estimate flux of specific contaminants from their current waste form (whether imbedded in beryllium reflector blocks or adhered to carbonate minerals in the surface soils) to the transport environment. Models to address surface complexation, solubility limits, valence states, etc., will provide more accurate representations of release that perhaps may be compared to observed contaminant concentrations in the environment. However, it is unclear whether more detailed release models, separate from vadose zone transport models, are needed. The answer to this question depends on how realistic or conservative the results need to be, which is related to the charter of the WIP.

This evaluation also focused on CERCLA documentation. Additional sources, including composite analyses for DOE Orders and National Environmental Policy Act (NEPA) sitewide analysis, need to be incorporated and compared. As a result, stored or in-process waste streams have yet to be examined. A sitewide source term analysis should include these potential future sources as well.

In addition to this general summary, specific implications exist regarding conceptual models of each type of source-term release (injection wells, buried waste, ponds and ditches, and contaminated soils).

Injection Wells—Eleven wastewater injection wells were constructed at the INEEL. Of these eleven, LOFT-04 never received hazardous constituents and EOCR-2 was never used. IET-06 and WRRTF-05 injections occurred directly into the SRPA and no evidence for a secondary source of contamination is present. USGS-53, CPP-23 (when the casing collapsed), USGS-50, PBF-05, and PBF-15 all injected wastes above the water table; therefore, there is the possibility of migration of contaminants that remain in the vadose zone into the SRPA. TSF-05 contains secondary source sludge in the well and in the adjacent basalts; residual sludge also may be present in the injection interval of TRA-05.

Two estimates of INTEC source term inventory were presented in the comprehensive RI/FS. The upper bounding estimate of contaminant mass and activity was used for risk modeling purposes, but uncertainties remain as some contaminants may have been overestimated. The inventory estimate of TCE, the primary contaminant at TAN, is highly uncertain, but researchers have inversely calculated source-term estimates using measured hydraulic properties of the aquifer and known contaminant concentrations.

Buried Waste—Uncertainties with regard to buried wastes is most pronounced for SDA disposals. Although some uncertainty exists regarding the inventories at the SL-1 and BORAX Burial Grounds, the locations of the wastes are well documented. Contaminant inventories for these two sites are limited to radionuclides and have been modeled based on the best available information.

Within the SDA, locations of individual waste disposals are not exact, but location information is likely developed as well as is possible given the available information. Current efforts are focused on refining inventory estimates for INEEL-generated wastes and are continuing. Probes have been installed into the SDA to provide additional information on infiltration rates, disposal locations, and other factors. These probes have been installed to collect information on four main focus areas by type of waste (i.e., depleted uranium, organic sludge, americium and neptunium, and enriched uranium). The probe array also is providing information about moisture monitoring and Pit 9. With regard to the probe effort, several questions remain unanswered:

- How long will data be collected from the probes?
- When will adequate information be considered to have been collected?
- How will it be determined when adequate information has been collected?
- Are additional probes needed to address other focus areas within the SDA?

Primary uncertainties with regard to SDA buried waste (other than those related to the waste itself) are related to mechanisms of contaminant release from the wastes. Water infiltration rates vary across the SDA by location and by season. Estimates of infiltration have been made on generalized areas within the SDA, but additional information may be needed before the effects of the infiltration are known because of the highly heterogeneous nature of the waste.

Infiltration, in turn, will drive leach rates and other release mechanisms (e.g., container and activated metal corrosion rates). Solution chemistry of the infiltrating water will affect the release mechanisms, and will vary across the SDA based on existing natural soils, near-field effects in proximity to the waste, and the chemistry of other soils that have been relocated to the SDA for use in covering the waste and recontouring the burial ground. Because data on anions in the soils have, to some degree, been neglected in sampling efforts, and because their impact on the solution chemistry of infiltrating water within the SDA is unclear, this remains an area for further investigation.

Ponds and ditches—Unlined surface impoundments have been used since 1952 to dispose of aqueous wastes that contained concentrations of radioactive and chemical contaminants. The TSF-07 disposal pond at TAN has been used for disposal of an estimated 11 curies of radionculides. Most of these radionuclides probably are immobilized in pond-bottom sediments and associated shallow, perched groundwater. This pond is not considered to be a significant source of contaminants to the SRPA, but is believed to modify the local hydraulic gradient and direction of TCE transport from the TSF-05 secondary contaminant source. Unlined disposal ponds at TRA and NRF have contributed to direct release of conservative contaminants, including tritium and hexavalent chromium, to the vadose zone and SRPA. Again, less conservative contaminants remain immobile in surficial sediments beneath these ponds. Contaminant inventories associated with disposal to the unlined service-waste ponds at INTEC have been small. Again, the effect of disposal to these ponds has been more important in the context of formation of perched groundwater and contaminant migration from other INTEC sources. Contaminant inventories associated with other INEEL ponds and ditches are considered to be no threat to the SRPA.

Aqueous wastes discharged to ponds and ditches infiltrate pond bottom materials. Wastewater accumulates as shallow, perched groundwater until the hydraulic head is sufficient to cause wastewater to move downward at the rate of disposal. Release of contaminants contained in wastewater is hydraulically attenuated for a short time as the wastewater accumulates in these shallow perched groundwater bodies.

Some contaminants may be conservative with respect to sorption and may be transported rapidly with the wastewater to the aquifer. Typically, these contaminants do not remain in the shallow subsurface and are considered to be released at wastewater disposal rates.

Chemical conditions in the pond-bottom materials and shallow, perched groundwater may affect contaminants. Some contaminants may strongly sorb to sediments. Analysis of INEEL pond and ditch bottom sediments indicated that certain species remain immobilized in those sediments. Over an extended time, large inventories of immobile contaminants may be accumulated within small depth intervals beneath the ponds. Additionally, the near-field chemical environment may alter chemical species so that the contaminant becomes more or less mobile. If changes occur in the near-field chemical environment, contaminant release mechanisms may be altered.

While estimates are available of activities and masses of radiochemical and chemical contaminants discharged in aqueous wastes to most INEEL ponds and ditches, residual inventories in pond sediments and the near-field chemical environment are not well characterized. Research is needed to describe the effect of infiltration-pond operational changes on near-field chemical environment and residual contaminant inventory.

Contaminated Soils—Literally hundreds of contaminated soil sites have been investigated across the INEEL. The vast majority of them have been remediated, primarily to ensure ongoing safety of the facility workers. The impacts to groundwater from infiltration and leaching by precipitation and subsequent transport through the vadose zone have also been investigated in a number of CERCLA remedial investigations. The simplified conceptual models were used at most contaminated soil sites to develop bounding estimates of risk for the groundwater pathway. One notable exception is the known release sites at the INTEC Tank Farm. Here highly radioactive aqueous solutions were released to the surficial soils. To provide conservative estimates, investigators did not include the surficial soils in the conceptual model, but simply applied the entire mass of contaminants directly into the vadose zone transport model where their travel was governed by hydraulic properties and distribution coefficients applied to the sedimentary interbeds. Even with such a conservative approach, groundwater risk associated with the groundwater pathway was not unacceptable beyond the boundaries of the INTEC facility. However, to better evaluate remedial alternatives, additional soil characterization and evaluation work is ongoing at this site.

Notwithstanding the unresolved questions, a few recommendations can be made at this time for all source terms. These activities, regardless of the direction of the program, would undoubtedly result in meaningful and useful data and should be evaluated by the program:

- Explore the feasibility of developing a process knowledge (mass balance) approach to compile inventory estimates contaminant by contaminant, waste stream by waste stream across the INEEL.
- Develop methodology to differentiate between historical source terms which have already impacted the aquifer to their maximum extent and those contemporary source terms which will exhibit some level of attenuated release now and into the future.
- Development of methods to directly measure source term release (e.g. leachate collection and analysis). Including continuation of probing program at RWMC. Similar approaches may be applicable at other INEEL sites.
- Development of more complex release models to take advantage of near-field chemistry data being collected by Type B probes at RWMC.
- Continuation of metal corrosion studies at the RWMC.
- Continued characterization of tank farm soils for contaminant concentration data as well as geochemical properties.
- Evaluation of stored or in process waste streams for potential as future source terms. Include review of composite analyses and site wide NEPA evaluations.
- Scoping of a detailed soils release model at tank farm to account for sorption of contaminants and extreme near field soil chemistry.