Attachment

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RISK ASSESSMENT OF OPTIONS FOR DISPOSITION OF EAF DUST FOLLOWING A MELTDOWN INCIDENT OF A RADIOACTIVE CESIUM SOURCE IN SCRAP STEEL

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Prepared for

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ABSTRACT

Keystone Steel & Wire Co., in Peoria, Illinois, receives and melts scrap steel in two electric arc furnaces, and uses the recycled steel in a broad array of wire and rod products. Sometime in early December 1992, an industrial gauge radiation source , containing the radioisotope cesium-137, inadvertently was included in a load of scrap steel charged to one of the arc furnaces. The melted and volatilized cesium was released into the off-gas system where it became distributed in the ducts and in the bag house. During a cleanup effort, more than 600 tons of contaminated dust were collected and placed into temporary storage.

Electric Arc Furnace (EAF) dust, known as K061 because of hazardous material content, with the addition of a radioactive component becomes "mixed waste." Regulations are not in place for handling this very low radioactivity level mixed waste. Work done during this study identifies the various options for disposition of the contaminated EAF dust and performs risk assessments of identified pathways for radiation exposure to workers and the public. The RESRAD computer code is used to calculate radiation dose.

All of the calculations are initially based upon a normalized cesium concentration in the dust of 20 picocuries per gram. In discussion, results are then extended to other concentrations. Direct gamma exposure to masses of contaminated material are first calculated for large exposure areas ("semi-infinite volume") and then adjusted for various actual areas for storage, shipping, and handling geometries. Pathways associated with disposal at a fully permitted (Part B) hazardous waste landfill, and processing at the Horsehead Resource Development Co. are considered. Groundwater pathways at the landfill are: 1) the leachate collection system remains intact and drinking water is drawn directly from the leachate, and 2) the leachate collection system degrades and drinking water is drawn from the aquifer below the landfill. It is found that in this last pathway, the cesium decays to negligible values long before migrating down to the aquifer. Treatment to remove cesium using a soil-washing process with ion-exchange resins is briefly discussed and the potential benefits are tabulated.

It is very important that regulatory relief be obtained to permit doing something with EAF dust contaminated with cesium, instead of continuing to store the material. The risk analysis results for the pathways considered indicate that Cs-137 concentrations in EAF dust up to 100 picocuries per gram or greater would not cause workers in unrestricted areas or members of the public to receive doses exceeding guidelines that are in general use. It may be that two levels of regulatory limits can be considered: one lower level for continuous control plus a higher limit for one incident or an occasional incident.

1. INTRODUCTION

Keystone Steel & Wire Company, in Peoria, Illinois receives and melts scrap steel in two electric arc furnaces. The recycled steel is used in a broad array of wire and rod products. Impurities in the scrap steel are removed during melting via slag and in the off-gas. The off-gas carries dust, containing a number of metals, to the bag house where the dust is trapped and recovered. Hazardous components: lead, cadmium, and chromium, cause the dust to be designated as U.S. EPA hazardous waste no. K061. This dust also contains about 20% zinc, a valuable byproduct for recovery. The EAF (Electric Arc Furnace) dust is either disposed at a landfill, licensed for disposal of hazardous wastes, or sent to the Horsehead Resource Development Co., for processing. The processing recovers zinc and other metals of value and removes the hazardous constituents. The remaining iron rich material is released for unrestricted use.

Sometime in early December 1992, an industrial gauge radiation source, containing the radioisotope Cs-137, inadvertently was included in a load of scrap steel charged to one of the arc furnaces. This source along with its encapsulation and holder were melted and the cesium was released into the off-gas system where it became distributed in dust in the ducts and in the bag house. Steel production was continued until a planned maintenance shutdown during the last two weeks of December. During the shutdown, an extensive cleanup effort was mounted and more than 600 tons of contaminated dust were collected and placed into temporary storage.

Unfortunately, a regulatory gap exists for disposition of the slightly radioactive contaminated EAF dust. The low activity does not justify the expense of handling as low-level waste, yet it is not of zero activity. The situation is further complicated by the presence of hazardous constituents, making the radioactively contaminated hazardous waste a "mixed waste," not yet covered by regulations except on a case-by-case basis.

The study reported here has two objectives: 1) identify and evaluate options for processing and/or disposal of contaminated EAF dust, and 2) perform a risk assessment of the significant environmental pathways as an aid in obtaining regulatory relief to permit processing and/or disposal of the contaminated EAF dust.

The format of this report is to provide background on the nature of cesium contamination and related regulations in Section 2, a statement of options in Section 3, risk assessment modeling and calculated results for direct exposure, disposal in a landfill, and processing at Horsehead in Sections 4 through 6, a discussion in Section 7 of treatment to remove cesium, and discussion of results and conclusions in Section 8.

2. CESIUM CONTAMINATION AND REGULATIONS

2.1 Cesium

Cesium is a volatile metal; the pure metal melts at 28.5 C and boils at 670 C. As a chloride in industrial sources, CsCl melts at 646 C and sublimes at 1,290 C. After being subjected to temperatures as high as 1,700 C in an arc furnace, the form of cesium in EAF dust is likely as one of the oxides. It is because of the volatility that the cesium in the melting incident left in the off-gas and ended up in the EAF dust; there was none detected in slag or in the steel product.

The radioactive isotope Cs-137, used in industrial sources, has a half-life of 30 years. This means that the level of activity decreases to one-half in 30 y, to one-fourth in 60 y, etc. After 100 y, the level is 0.1 of the initial level, after 200 y is 0.01, and after 300 y is 0.001.

The decay scheme of Cs-137 is as follows:

$$Cs^{137} \xrightarrow{30\ y} Ba^{137m} + \beta^{-}$$
$$Ba^{137m} \xrightarrow[0.3\ s]{} \underline{Ba}^{137} + (\text{X-ray})$$

The cesium decays with the 30 y half-life to an unstable "daughter": an isomer of barium, Ba-137m, with emission of a beta particle. The Ba-137m then decays with a half-life of only 0.3 second to the stable form of barium, Ba-137, with emission of a 0.662 Mev (Million electron volts) X-ray. It is the X-ray that is the penetrating emission from this decay process. While the emission is an X-ray, it behaves similarly to gamma radiation, and dose from exposure to this radiation is therefore loosely referred to as "gamma dose." Because of the almost immediate decay of the barium isomer, dose conversion factors combine the two decay steps into one for "Cs-137+D" (cesium plus daughter).

The specific activity of pure Cs-137 is 86.5 Ci/g (Curie per gram). One Curie represents an activity of 3.7×10^{10} disintegrations per second. The density of CsCl is 3.97 g/cm^3; the volume specific activity for this form becomes 305 Ci/cm^3 (Curie per cubic centimeter). The density of CsO is 4.36 g/cm^3; the volume specific activity for this form becomes 338 Ci/cm^3. The volume of 100 mCi (millicurie) of either form is therefore only about 0.3 cubic millimeter (a diameter as a sphere of only 0.6 mm)!

2.2 Industrial Cesium Sources

An industrial gauge radiation source, such as Cs-137, is used in conjunction with an ion chamber detector to measure the presence and density of intervening material. Applications include non-contacting measurement of level and/or density of liquids, solids, or slurries. Gamma (or X-ray) energy is absorbed by the mass of any material between the source and detector; the fraction of gamma energy absorbed increases with the mass of absorber in the path of the beam. The source material is doubly encapsulated in stainless steel and located in the center of a lead-filled welded steel holder. A shutter in the holder blocks a passage in the lead shielding, but is opened to permit emission of the beam for measurement use. The geometry of the shielding produces a highly collimated narrow beam of gamma energy. A source with source holder and shutter is an integral assembly without ready access to the source material.

The available sizes of industrial gauge cesium sources are typically up to 5 Ci, though the most popular sizes in use are in the 50-200 mCi (millicurie) range. It was initially assumed that the source involved in the incident at Keystone was 100 mCi. However, review of preliminary survey and analysis data for stored EAF dust suggests that a larger source was involved.

2.3 Cesium Contamination Concentrations

The incident at Keystone dispersed a small amount of radioactive cesium very finely throughout more than 600 tons of EAF dust. Because of the low resulting concentrations, the unit of picocurie is used throughout this report. A picocurie (pCi) is 1×10^{-12} Ci. This small unit represents only 0.037 disintegration per second or 2.22 disintegrations per minute (DPM). Concentrations in picocuries per milliliter (pCi/ml) and picocuries per gram (pCi/g) are used.

Box number 1 of stored dust (21.3 tons) contains an estimated 78% of the total cesium released and has varying concentrations up to a maximum of 16,200 pCi/g. Survey meter readings for this box ranged from 3 to 10 mR/h. Therefore, it is estimated that the average concentration in this box is (3 + 10)/(2 x 10) = 0.65 times the maximum, or 10,530 pCi/g. This value and measured concentrations for average samples from the other 50 boxes of contaminated dust collected through the end of the clean out operations are listed in Appendix A. Integration of concentrations over the 654 tons of dust indicates the quantity of cesium released was 260 mCi. The number of boxes in storage with various ranges of cesium concentrations are summarized in Table 1.

Table 1. Concentration Ranges in Boxes of Stored Dust

Concentration Range, pCi/g	Number of Boxes
10,530	1
742	1
340	1
201-300	3
151-200	2
101-150	8
51-100	11
41-50	5
31-40	6
26-30	1
21-25	2
16-20	2
11-15	1
5-10	3
< 5	4
TOTAL	51

2.4 Regulations

The EAF dust, contaminated with Cs-137, has a very low average level of radioactivity, but it is not zero. At the same time, it is much lower in activity than the category designated as "Low-Level Waste" (LLW). Also, the EAF dust contains the hazardous components: lead, cadmium, and chromium. Because of this, the dust is designated as U.S. EPA hazardous waste No. K061, and is regulated under RCRA (Resource Conservation and Recovery Act). Waste containing both radioactive and hazardous components is

"mixed waste." Regulations are not generally in place for very low levels of radioactivity, and particularly are not in place for mixed waste. Therefore, in the absence of specific regulatory requirements, consideration of options following a cesium source meltdown incident is not simply a matter of evaluating compliance with such requirements.

In the following subsections, various regulations and regulatory agency actions are briefly described. For each item, only the parts that relate to cesium contamination are presented.

2.4.1 Radiation Protection

"Standards for Radiation Protection" issued by the U.S. Nuclear Regulatory Commission (NRC) are contained in 10 CFR 20 [1]. The specified limits for release in effluents of Cs-137 in a soluble form to unrestricted areas are 20 pCi/ml in water and 0.002 pCi/ml (2 pCi/L) in air. This standard contains no reference to soil contamination.

The State of Illinois, through the Illinois Department of Nuclear Safety, adopted the same limits for water and air in the Illinois Code: Appendix A, Table II, Column 2, of Ill. Adm. Code 340 [2].

The State of Texas also adopted the 10 CFR 20 limits, but the Texas Bureau of Radiation Control also expanded the waste water limits to soil and vegetation. Section 21.108 of the Texas Regulations for Control of Radiation [3] apply the waste water limits to soil and vegetation by changing the units for that application from µCi/ml to µCi/g. This corresponds to 20 pCi/g for Cs-137.

DOE Order No. 5400.5: "Radiation Protection of the Public and the Environment," [4] contains a Radiation Protection Standard of 100 mrem/y for the general public.

2.4.2 Low-Level Wastes

The NRC "Licensing Requirements for Land Disposal of Radioactive Waste" (applies to Low-Level Wastes) are contained in 10 CFR 61 [5]. The lowest concentration category of waste addressed is Class A. This class applies for Cs-137 content up to 1 Ci/m^3, which is 10^6 pCi/ml. Various portions of the contaminated EAF dust have cesium contents lower than the Class A limit by a factors ranging from 60 to 500,000. Concentrations of radioactive materials released to the general environment in ground water, air, soil, etc., must not result in an annual dose exceeding 25 mrem whole body dose to any member of the public. Handling the EAF dust as LLW subjects it to excessively stringent repository requirements and high costs.

DOE policies and guidelines for radioactive waste management are covered in DOE Order 5820.2A [6]. Chapter III of this order is titled "Management of Low-Level Waste." It states: "Assure that external exposure to the waste and concentrations of radioactive material which may be released into surface water, ground water, soil, plants and animals results in an effective dose equivalent that does not exceed 25 mrem/y to any member of the public," and "Assure that the committed dose equivalents received by individuals who inadvertently may intrude into the facility after the loss of active institutional control (100 years) will not exceed 100 mrem/y for continuous exposure or 500 mrem for a single acute exposure."

Standards for LLW by the U.S. EPA are not expected for about two years.

2.4.3 High-Level Wastes

The NRC regulations "Disposal of High-Level Radioactive Waste in Geologic Repositories" are in 10 CFR 60 [7]. These regulations limit the total body dose equivalent to any member of the public outside of the controlled area to 25 mrem/y. Concentrations in

ground water are limited to that which would produce a total body dose equivalent of 4 mrem/y for an individual consuming 2 L/d.

The corresponding EPA standards "Environmental Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes" are in 40 CFR 191 [8]. The proposed re-promulgation of this rule limits any member of the general public to 15 mrem/y "committed effective dose", which because of differences in the method of calculation, is equivalent to the previously used 25 mrem/y "total body dose." Concentrations in ground water are limited to the Maximum Contaminant Levels (MCLs) in the National Primary Drinking Water Regulations (NPDWRs) in 40 CFR 141 [9]. For radionuclides such as Cs-137, the MCL is the concentration causing 4 mrem/y total body dose for an individual consuming 2 L/d (same as for NRC regulations). For a dose conversion factor of 5.0×10^{-3} mrem/pCi for Cs-137, the corresponding MCL is 0.11 pCi/ml. While this concentration is much lower than the 20 pCi/ml limit for effluents in 10 CFR 20, it should be noted that the MCL applies to water specified as used for drinking.

2.4.4 Soil and Groundwater Guidelines

NRC "Current Guidelines on Acceptable Levels of Contamination in Soil and Groundwater on Property to be Released for Unrestricted Use," dated January 1992, were released with an order establishing criteria for decommissioning a contaminated site in Bloomsburg, Pa (57 FR 6136-6141, February 20, 1992) [10]. For Cs-137, the maximum soil concentration was stated to be 15 pCi/g and for groundwater to be 0.2 pCi/ml.

2.4.5 Basic Dose Limit Guideline

The "U.S. DOE Guidelines for Residual Radioactive Material at Formerly Utilized Sites Remedial Action Program and Remote Surplus Facilities Management Program Sites," as revised March 1987 [11], states that the basic limit for the annual dose received by an individual member of the general public is 100 mrem/y. It may be noted that a full-time dose of 100 mrem/y corresponds to 23 mrem/y for a 40 hour work week, 50 weeks per year, close to the 25 mrem/y used for waste repository dose limits previously discussed.

2.4.6 Transportation

NRC regulations "Packaging and Transportation of Radioactive Material" in 10 CFR 71 [12] and DOT (Department of Transportation) regulations in 49 CFR 173.441 [13] apply to radioactive materials packaged in containers. While EAF dust shipped in boxes and trailers covered by tarpaulins does not represent packaging as required under these regulations, some of the provisions are of interest. The least stringent packaging is "Type A". For Cs-137, the maximum quantity in each Type A package is 10 Ci, much larger than the total in the stored EAF dust. The maximum dose rate at

the package surface is 200 mrem/h and at a distance of 1 m from the surface is 10 mrem/h. The maximum dose rate for any normally occupied position of a transport vehicle is 2 mrem/h. "Low specific activity material" per 10 CFR 71.4 is material in which the average concentration does not exceed 0.3 mCi/g (3×10^8 pCi/g).

2.4.7 Dose Conversion Factors

The term "dose" in this report, expressed in mrem, refers to effective dose equivalent for external exposure, and committed effective dose equivalent for internal exposure. A "dose conversion factor" (DCF), for the pathways considered, is the ratio of either of these doses to the concentration of a radionuclide in the ground (for external exposure), or the quantity ingested (for internal exposure). A variation of the DCF used in this report is annual dose rate per unit concentration of Cs-137 in the EAF dust, (mrem/y)/(pCi/g).

3. OPTIONS

There are several possible options for processing and/or disposal of the Cs-137 contaminated EAF dust. In the absence of regulatory relief, the only disposition available is to package and ship all of the contaminated material to an out-of-state low-level waste disposal site that is also licensed to receive mixed waste. This would be a very expensive alternate and would also utilize LLW repository capacity better used for higher activity material.

Other options stated below depend upon establishing acceptable concentrations for processing and/or disposal in a hazardous waste landfill. Acceptable concentrations may evolve to be different for processing at Horsehead than for disposal in a landfill, based upon risk analysis results for the differing environmental pathways. In addition, the landfill operator may or may not elect to impose an additional restriction beyond the regulatory limit.

3.1 No Treatment to Remove Cesium

Some units of the stored EAF dust have cesium concentrations that are lower than regulatory limits that can reasonably be expected to be established. Indeed, some units of the stored dust have concentrations of only a few picocuries per gram. Therefore, if no treatment to remove cesium is performed, the following option remains:

1. Process at Horsehead and/or dispose in landfill, that portion having concentrations less than the limit(s).

2. Package and ship balance to LLW repository.

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3.2 Treat to Remove Cesium

If treatment of stored dust that exceeds the concentration limit is performed, it will only be done with the expectation that "clean" material will result which has concentrations below the limit. Such treatment also produces a relatively small volume of ion-exchange resins with concentrated cesium. Therefore, if treatment to remove cesium is performed, the following can be done:

1. Process at Horsehead and/or dispose in landfill, that portion having concentrations less than the limit(s) without treatment.

2. Treat material having concentrations exceeding the limit to reduce cesium concentrations to below the limit. Process cleaned material at Horsehead and/or dispose in landfill.

3. Package and ship ion-exchange resin to LLW repository.

A variation of this option is to forego treatment for cesium removal of the highest concentration stored material if the costs of multiple treatment passes for this portion exceeds the cost of handling as LLW.

4. DIRECT EXPOSURE

Radiation dose from direct exposure to masses of EAF dust contaminated with Cs-137 is a pathway associated with temporary storage, transport to other facilities, and handling at the other facilities. In this section, a base case is developed and used to demonstrate the effect of cesium concentration, decay time, material density, depth dimension, and area and shape geometry factors. Subsequent sections extend the direct exposure analysis to introduce factors related to disposal in a landfill and processing at Horsehead. In this and subsequent sections, a cesium concentration of 20 pCi/g is assumed as a normalized base. Results scale up or down for other concentrations that may evolve as regulatory limits.

The starting point in the direct exposure pathway calculations is to use the RESRAD computer code [14] to obtain the radiation dose at a distance of 1 m from the surface of a semi-infinite volume of EAF dust. This refers to a volume of infinite lateral extent and infinite depth. This condition is reached for practical purposes with a radius of 20 m or larger. Because of self-shielding effects, it is found that for Cs-137 contamination, the outer 0.5 m of material contributes almost all of the radiation dose, with the outer 0.25 m contributing 92% of the total. For smaller and non-circular sources, a "shape/area factor", less than 1.0, is applied to the dose rate for a semi-infinite source to obtain the lower dose rate for a given finite source.

The density of EAF dust varies with the degree of settling. Dust as settled in collection bins and boxes is typically 2,800 to 3,000 lb/ft^3. For the base case, an average of 2,900 lb/ft^3 (specific gravity = 1.72) is used in the RESRAD code. Results of RESRAD calculations presented below are obtained with this density, except where noted. Figure 1 is a plot of the dose conversion factor (DCF), mrem/y per pCi/g concentration, versus time, for a semi-infinite mass of dust. Initially, the DCF is 5.05 (mrem/y)/-(pCi/g), decreasing with time as radiodecay progresses. For a contamination concentration of C_c = 20 pCi/g, this initial base dose rate becomes

$$D_{base} = DCF \times C_c = 101 \text{ mrem/y} = 101/8{,}760 = 0.0115 \text{ mrem/h}, \quad (1)$$

where 8,760 is the hours per year for full-time occupancy.

As density of the material is increased, the volume concentration of the contaminant increases, but the radiation absorbing shielding effect also increases, with the net effect that the DCF is found to decrease with increased density. The effect of density is shown in Fig. 2, in which the DCF (and base dose rate) at time zero is plotted versus density. At the lower end, density corresponds to

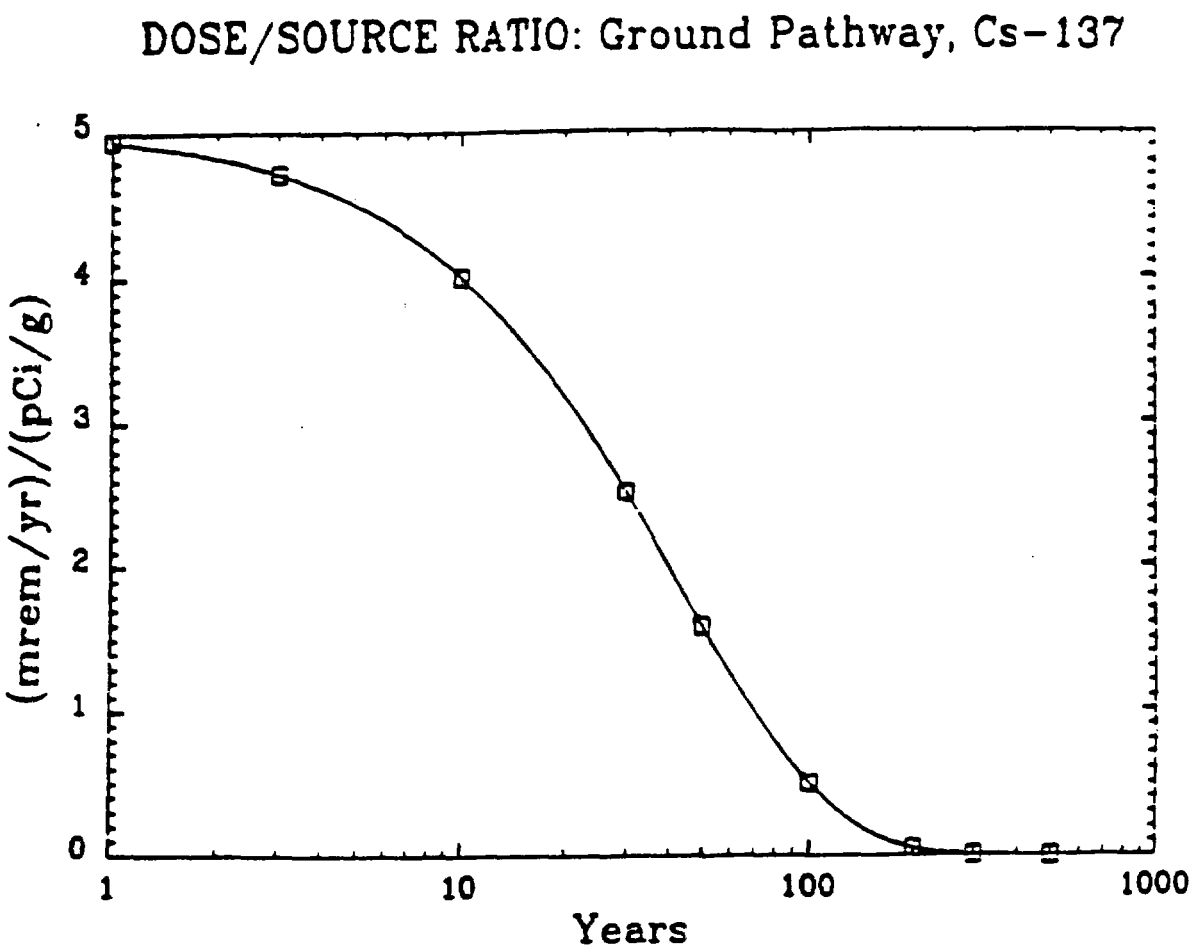

Fig. 1. Dose conversion factor versus time for EAF dust.

unsettled material with some aeration (2,200 lb/yd^3, 1.31 g/cm^3), and at the upper end, density corresponds to denser material from the horizontal roof duct (3,900 lb/yd^3, 2.32 g/cm^3). If it is necessary to adjust results from the base case which uses a density of 2,900 lb/ft^3, a density factor, F_d, derived from the data in Fig. 2 may be applied. Values for F_d are 1.0 for the base case, 1.04 for the less dense unsettled material, and 0.86 for the more dense settled roof duct material.

For material with large depths, radiation from material at the greater depths is absorbed by intervening mass and does not contribute to the direct exposure dose. Radiation from material at shallower depths is progressively less absorbed as the depth decreases. Figure 3 shows the initial DCF versus depth of contaminated material. It may be seen that for depths greater than about 0.5 m, the curve approaches the value for the semi-infinite source. For the pathways considered in this study, depths are generally greater than 1 m, but the values shown for lesser depths are useful for evaluating spills or residual deposits.

Finally, quantities of EAF dust we are considering do not cover large areas with radius greater than or even approaching 20 m. The base dose rate for semi-infinite extent must be multiplied by an area/shape factor, F_s, to obtain the actual dose expected from a finite dimensioned source. Using RESRAD methods, the curves in Fig. 4 are obtained. The upper curve shows the decreasing value for F_s as the diameter of a circular source decreases. The shape is also of importance. The bottom three curves show F_s for rectangles of widths 1, 2, and 3 m, as a function of rectangle length. Also shown in Fig. 4 are points representing dimensions of a typical box used for storage and transport.

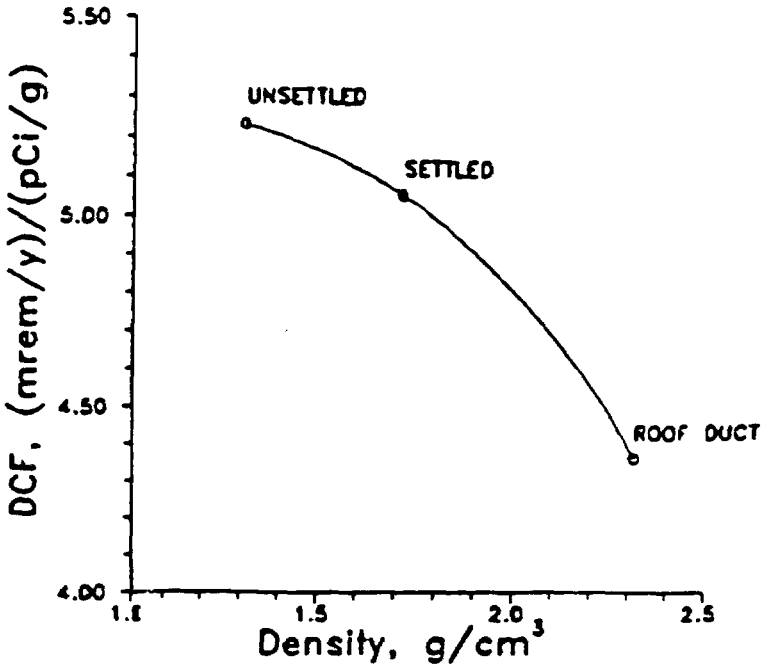

Fig. 2. Dose conversion factor versus density.

Table 2 lists values of F_s obtained for the sides and top of three containers in use. In each case, for the side height dimension, the container is assumed to be two-thirds full, due to weight limits.

Table 2. Area/shape factors for EAF dust containers.

Container	Dimensions, m	F_s
20 yd box, side	6.1 x 0.8	0.08
20 yd box, top	6.1 x 2.1	0.20
30 yd box, side	7.3 x 1.1	0.10
30 yd box, top	7.3 x 2.1	0.21
trailer, side	8.5 x 1.1	0.10
trailer, top	8.5 x 2.4	0.26

The direct exposure dose to a finite source of EAF dust, such as a storage or shipping box or a pile of material being processed, becomes

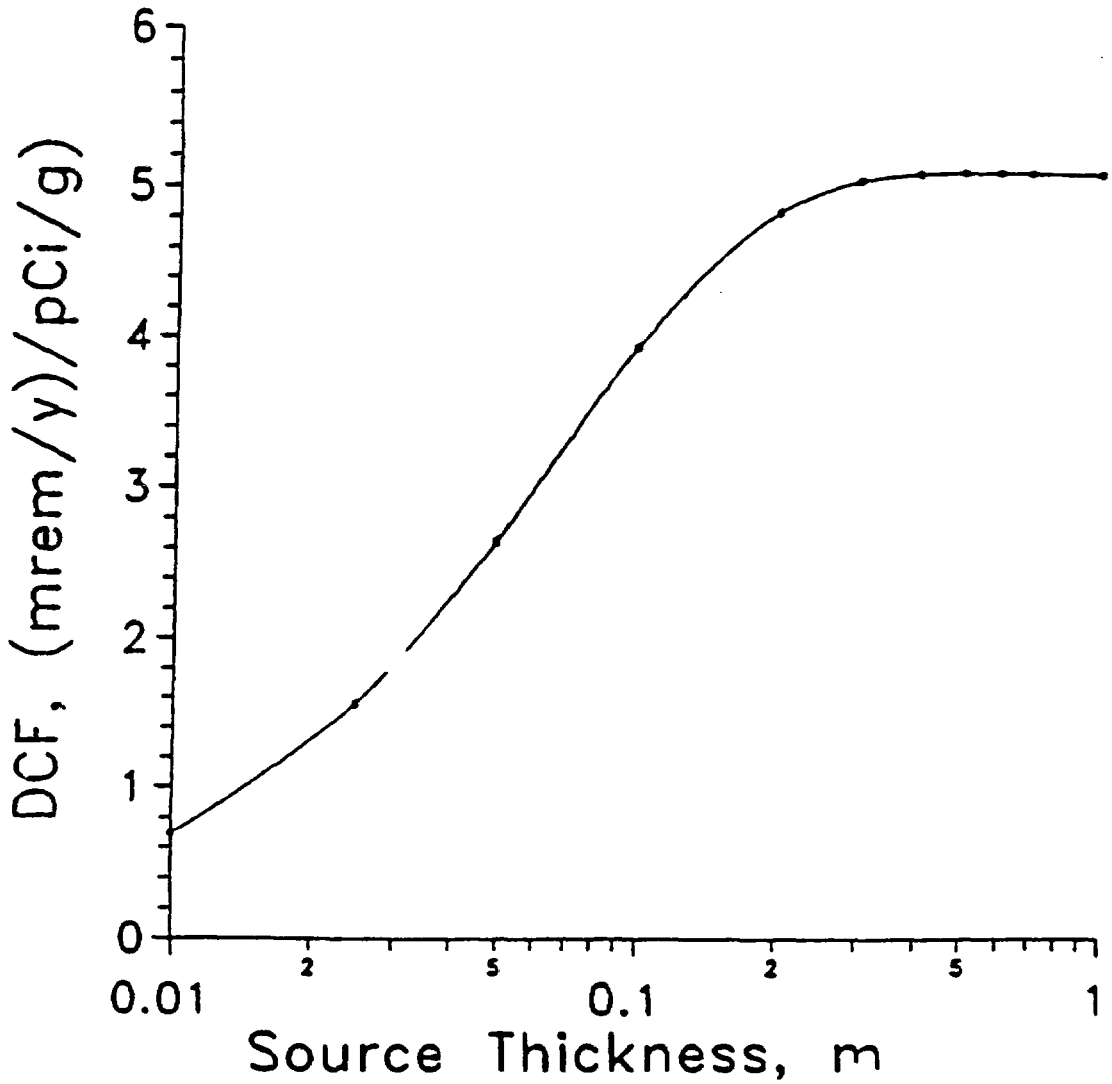

Fig. 3. Dose conversion factor and annual dose rate versus source thickness.

$$D = D_{base} \times F_s \times F_d \ . \tag{2}$$

For example, The dose rate for a person standing 1 m from the center of a "30 yd roll-off box" containing 20 pCi/g material of density 2,900 lb/ft^3 becomes

$$\begin{aligned} D &= D_{base} \times F_s \times F_d \\ &= 0.0115 \times 0.10 \times 1.0 = 0.0012 \text{ mrem/h.} \end{aligned} \tag{3}$$

Much of the currently stored material is at concentrations greater than 20 pCi/g; the above result scales up linearly with concentration.

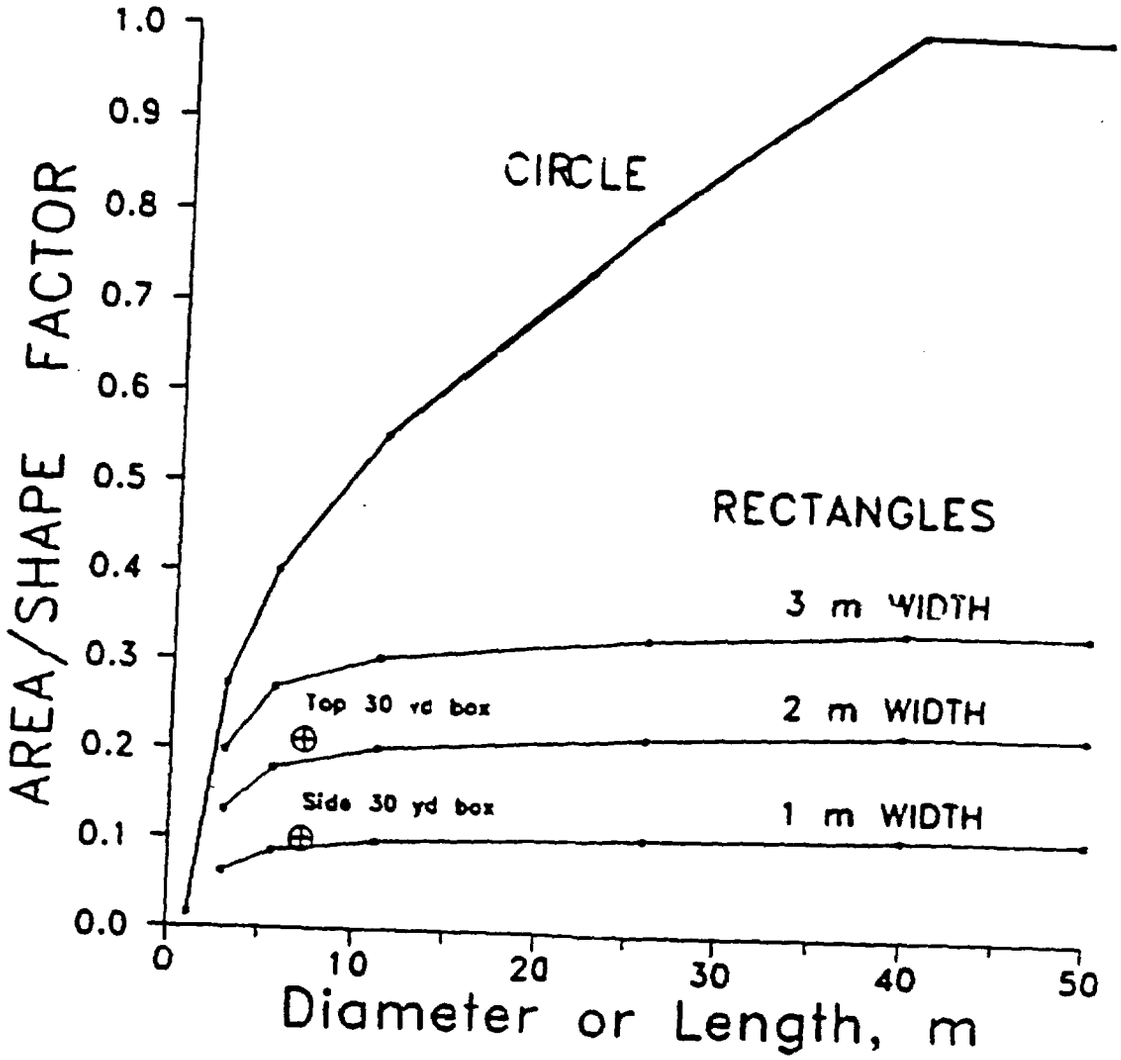

Fig. 4. Area/Shape factor versus diameter of circle or length of rectangle.

5. DISPOSAL IN LANDFILL

Some of the EAF dust (hazardous waste number K061) produced at Keystone is disposed in a hazardous waste landfill. This facility has a RCRA (Resource Conservation and Recovery Act) [15] permit to accept hazardous wastes and to dispose these wastes after immobilization treatment. Existing regulations and operational practice do not accommodate waste having radioactive contamination (mixed waste). It is expected that regulatory relief will be obtained that will establish a level of Cs-137 contamination acceptable for disposal in the landfill. If the landfill operator will accept the limit determined by regulatory agencies, or a modification of that limit to accommodate other restraints, disposal in the landfill will become an option for portions of the stored dust and/or additional portions after treatment for removal of cesium. In this section, the landfill is described, followed by modeling and the pathway analysis to evaluate risks associated with disposal in the landfill.

5.1 Description of Landfill

The landfill facility is located in the Midwest. Area C of this facility was designed with a double liner and leachate collection system in compliance with USEPA guidance for implementing RCRA Sections 3005(0) and 3015, as amended by the Hazardous and Solid Waste Amendments (HSWA) of 1984. Figures 5-7 are from the RCRA Part B Permit Application [16]. Figure 5 shows the Key Plan for Area C, which went into operation several years ago. Trench C-2 is currently operational. This trench has an area of approximately 23,800 m^2 and a disposal volume of 193,200 m^3 (252,700 yd^3), representing an average waste depth of 8.1 m. Figure 6 is a Typical Plan, showing an open trench prior to emplacement of waste and cover layers. Slopes in the various parts of the trench are indicated, along with the leachate collection system with leachate transport pipes leading to sump pits. A 4-foot diameter lined manhole provides leachate detection and pumping access to the sump pits.

Figure 7 is a typical cross-section of a perimeter section of a trench. The bottom preparation has a 3-foot thick low-permeability recompacted soil, covered by a 60-mil High Density Polyethylene (HDPE) Flexible Membrane Liner (FML). As shown in Detail A in Fig. 7, a drain net layer is placed above the lower FML, followed by a second FML, another drain net, and a filter fabric to inhibit waste material from clogging the drain net. The drain net is a molded HDPE grill-like material with drainage passages. In the bottom zone, 12 inches of sand serve as leachate drainage/bedding material, covered by 6 inches of coarser filter material. The HDPE pipe in this bedding material (Fig. 6) is slotted to collect and transport leachate.

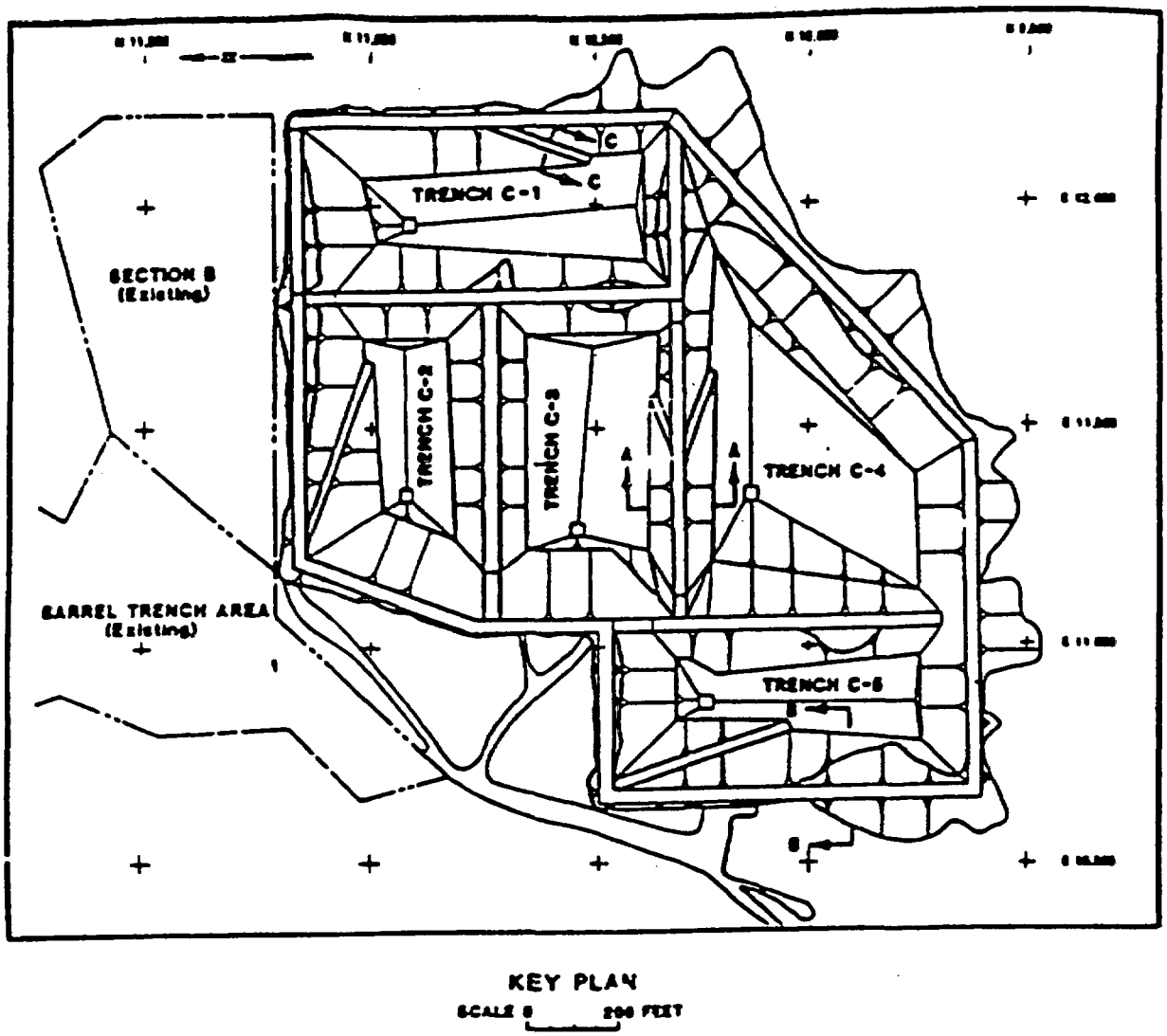

Fig. 5. Key plan for Area C.

Waste is emplaced to a depth of 25-50 feet, depending upon location in the sloped-bottom trench. An average of 35-40 feet is expected. Emplacement is in narrow bands, 2-3 feet wide, across the narrow dimension of the trench, building up from the bottom and tamped as they go. The waste is covered by a sequence of layers more than 9-feet thick: low-permeability fill, coarse filter material, random fill, and top soil. The surface is contoured and grassed to promote runoff of precipitation and minimize erosion.

The geologic setting of the landfill consists of three major geologic units. The surface formation is Illinoian Drift, which is predominantly silty clay till with interbedded sand, gravel, and silt. Below this is Shelbyville Outwash, predominantly sand with

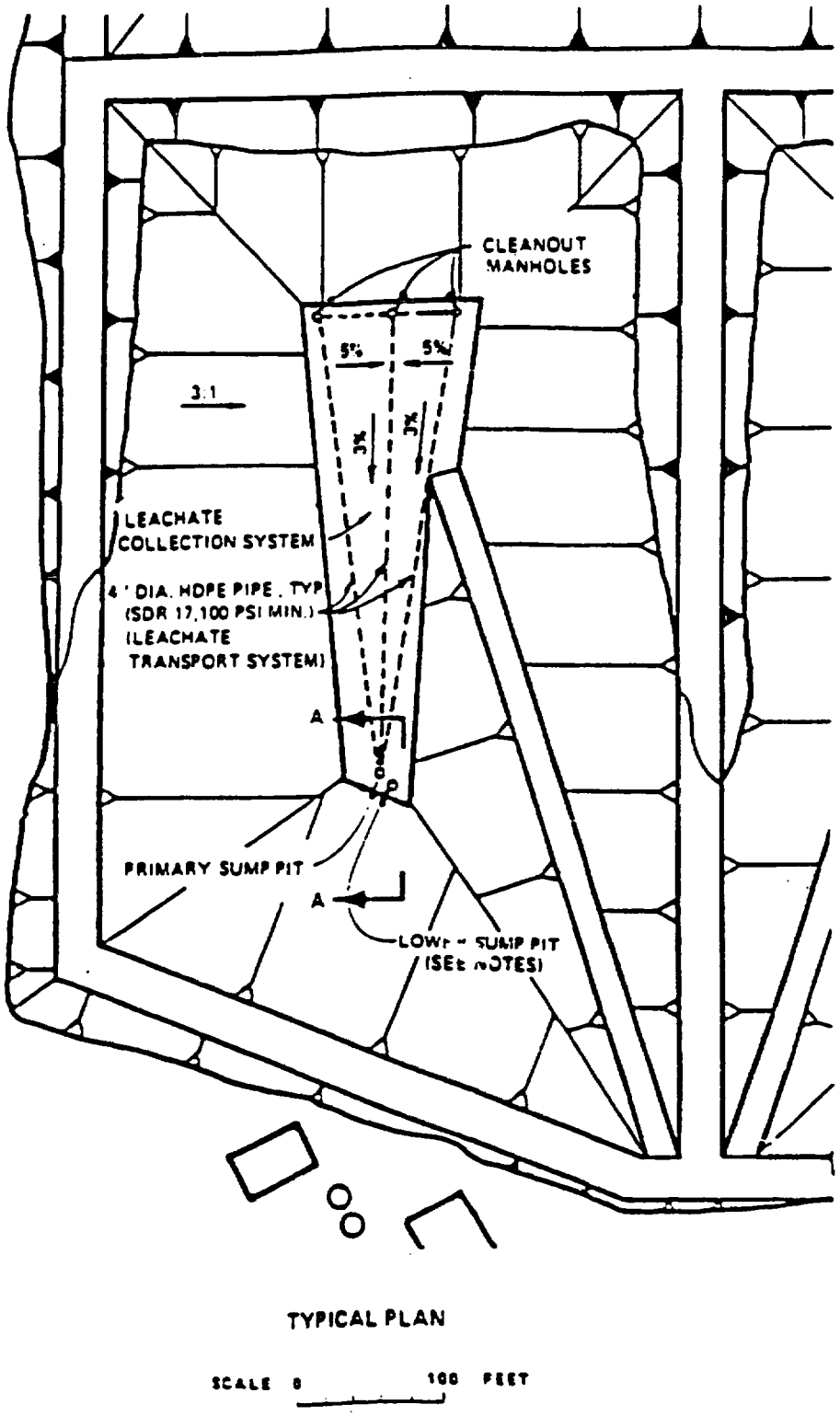

Fig. 6. Typical plan for a disposal trench.

some gravel and silty sand. Below this is Pennsylvanian Bedrock, predominantly gray shale with coal, limestone, and sandstone. The water table is within the Shelbyville Outwash. Groundwater flow under the site is predominantly from the west to the east. Recharge to the Shelbyville Outwash is mainly by subsurface inflow from the west and west-northwest and to a small extent by downward infiltration of precipitation through the Illinoian Drift. There are no bodies of water in the vicinity of the site. A small creek is more than two miles to the west and flows for about fifty miles before reaching a large river downstream of the facility.

5.2 Waste Treatment Prior to Disposal

Waste received at the facility is treated to obtain a waste form meeting RCRA immobilization requirements. A load of incoming K061 in a roll-off box on a truck is first dumped onto the floor in one of four bays in the treatment building. This material as received can have and preferably does have a moisture content for suspended dust control. A moisture content up to a thick slurry can be accepted. A front end loader transfers the material to a twin-shaft mechanical mixer-blender where additives are introduced. For K061, the additives are: 1) liquid reagent to adjust oxidation potential, and 2) dry powder additive (portland cement, fly ash, etc.) and if dry, add moisture. A hydraulic lift is used to transfer material from the mixer-blender to roll-off boxes. These boxes are temporarily stored in the yard until laboratory tests confirm that the batch meets treatment standards and then placed in the disposal trench. The pH of the mixture is typically 10.5. The additives result in some dilution; for K061 from Keystone, this typically amounts to approximately 30 g of additives per 100 g of dust, representing a dilution factor of 0.77. Density is typically 1.36 g/cm^3 (2,300 lb/yd^3).

The waste form must meet the TCLP (Toxicity Characteristic Leaching Procedure) requirements specified in 40 CFR 261, Appendix II [17]. The TCLP procedure for K061 is briefly as follows: 1) prepare sample with particle size less than 1 cm in its narrowest dimension or surface area per gram ≥ 3.1 cm^2, 2) add extractant = 20 x weight of sample (e.g. if 100 g sample, add 2000 g or 2 L of extractant), 3) agitate for 18 hours, 4) separate and analyze extractant, and 5) compare constituents with limits in 10 CFR 261.24, Table I. The extractant is reagent water with pH adjusted to either 4.93 or 2.88, with the latter pH the one likely required for K061. Disposal is done only after the standards in Table 3 are met.

The TCLP test provides a standardized means for obtaining a qualitative comparison of degree of immobilization. The value of milligrams per liter in the extractant is a combination of the concentration of the constituent and its leachability in the treated waste form to an excess of solvent. It is not directly a measure of leach rate expected in the repository. Also, the TCLP standards do not address cesium, and no data is available for samples with cesium contamination.

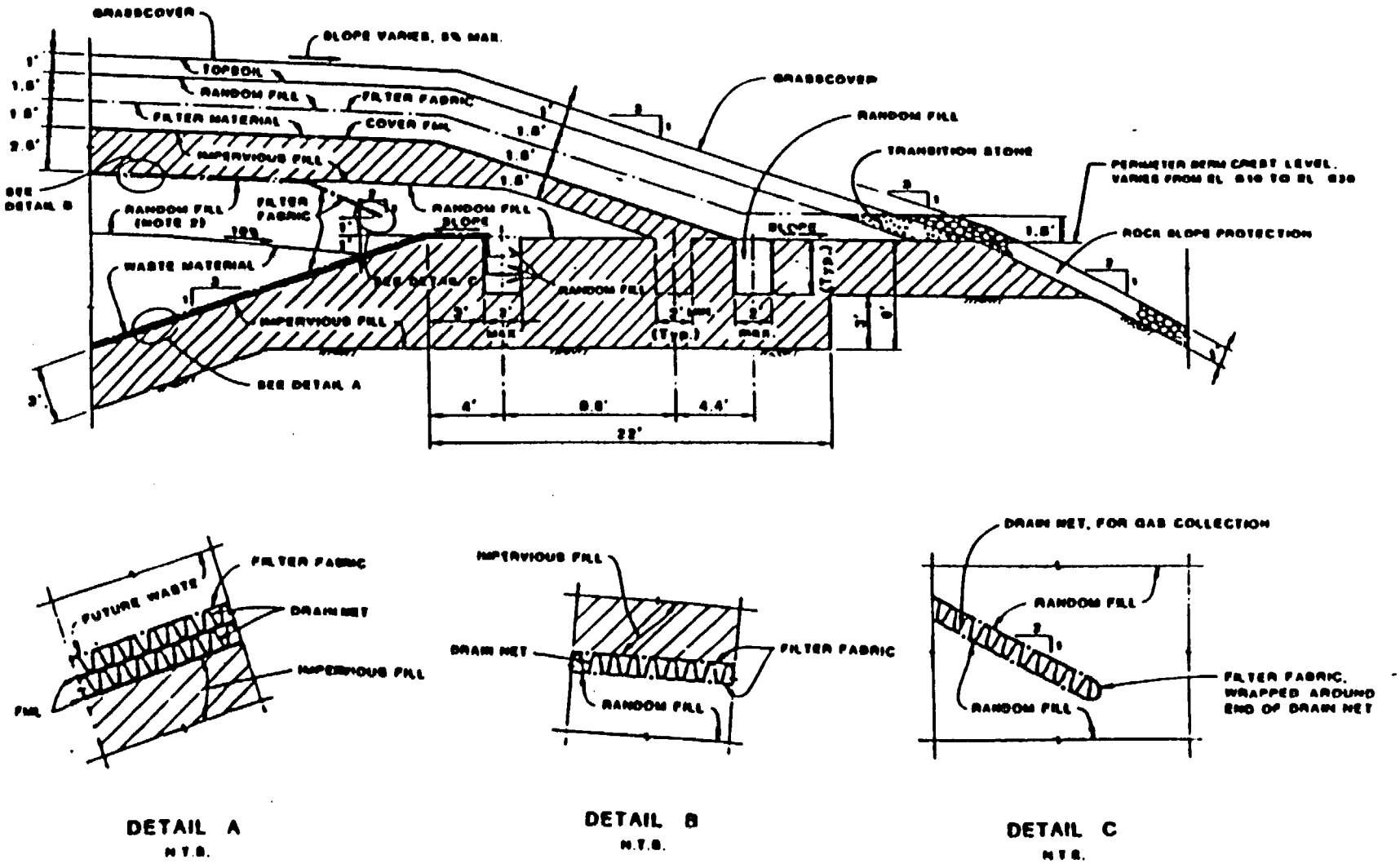

Fig. 7. Typical perimeter section of a disposal trench.

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Table 3. TCLP Standards for Disposal

Regulated Constituent	Maximum for TCLP Sample, mg/L
Antimony	2.1
Arsenic	0.055
Barium	7.6
Beryllium	0.014
Cadmium *	0.19
Chromium (Total)*	0.33
Lead *	0.37
Mercury	0.009
Nickel	5.0
Selenium	0.16
Silver	0.30
Thallium	0.078
Zinc	5.3

* hazardous constituents for which K061 is listed

5.3 Potential Release Pathways

A generalized list of potential pathways for release of radionuclides such as cesium from a shallow land burial site is as follows:

1. External Radiation
   a. Ground volume or surface source
   b. Airborne suspension of radionuclides
2. Inhalation of airborne radionuclides
3. Ingestion
   a. Food: Plant foods, meat, milk, aquatic foods
   b. Water: Groundwater (well), surface water
   c. Soil

Several of these can be eliminated immediately. There is no surface water nearby. The K061 material is shipped to the site in covered containers and is initially wetted in the treatment process, precluding suspension by winds. The treated material is emplaced in the form of a wet soil cement. After emplacement, more than six feet of soil cover is added. In the hundreds of years that would be required for erosion to expose the waste to winds, Cs-137, which has a relatively short half-life of 30 years, would decay to insignificant levels. Airborne radionuclide pathways for inhalation or external radiation need not be further considered. Food that could be grown on the site after the post-operation control period would be protected from root uptake by the cover thickness which exceeds root penetration depth. This also applies

to grazing feed for meat and milk production. The remaining pathways to be evaluated are:

1. Direct radiation exposure during treatment.
2. Direct radiation exposure to emplaced waste prior to covering.
3. Direct radiation exposure after waste is covered.
4. Ingestion of groundwater withdrawn via well.

The last of these, ingestion of groundwater, is divided into two risk scenarios:

1. Direct leachate drinking scenario:
The leachate collection system remains intact and drinking water is withdrawn via the manhole from the leachate sumps or via a well into the leachate drainage/bedding material.

2. Aquifer drinking water scenario:
The leachate collection system degrades, leachate moves down to the aquifer and drinking water is withdrawn via a well into the aquifer at the down-flow edge of the disposal trench.

Results obtained for these scenarios show that there is no risk beyond the site boundaries. Hence, no scenarios for transport to off-site locations need be considered.

5.4 Pathway Modeling for Landfill Disposal

5.4.1 Direct Exposure Pathways for Landfill Disposal

Modeling and results for direct exposure pathways, using the RESRAD Code, were previously described in Section 4. In the later subsection 5.5, the results calculated with RESRAD for direct exposure are extended to evaluate waste handling at the facility, the effect of adding cover material over the waste, and the effect of the concentration dilution by waste treatment.

5.4.2 Modeling Related to Both Groundwater Ingestion Scenarios

5.4.2.1 Modeling of Vertical Structure

A representative model for the sequence of layers for use in RESRAD calculations of leaching and downward movement of water was prepared using the geologic and hydrologic descriptions and data in the RCRA Permit Application [16]. This sequence is listed in Table 4.

The Drain Nets/FMLs layer is listed in Table 4, to relate it to the cross-section in Fig. 7, but it doesn't enter into the vertical flow calculations in RESRAD. For the direct leachate drinking scenario, it is assumed that the leachate collection system remains intact and blocks further downward movement. For the aquifer

drinking water scenario, the FMLs are assumed to degrade, permitting downward flow to continue.

In Table 4, the water table is at the interface between the unsaturated and saturated zones within the Shelbyville Outwash formation. The vertical distance from the bottom of the waste to the water table, as modeled, is 32.5 m (107 ft).

Table 4. Layering Model for Groundwater Scenarios.

Layer	Designation in RESRAD	Thickness, m
Disposal trench:		
Cover Layers	Cover	2.9
Waste	Contaminated Zone	8.1
Drain Layer	Unsaturated Zone 1	0.4
Drain Nets/FMLs	-----	----
Clay	Unsaturated Zone 2	0.9
Underlying geology:		
Illinoian Drift	Unsaturated Zone 3	16.0
Shelbyville Outwash	Unsaturated Zone 4	15.2
Shelbyville Outwash	Saturated Zone	12.2
Penn. Bedrock	-----	----

5.4.2.2 Infiltration

Leaching of emplaced waste and downward transport occurs by infiltrating water. The infiltration rate in the absence of irrigation is simply:

$$I = (1 - C_e)(1 - C_r)P_r \qquad (4)$$

where C_e is the evapotranspiration coefficient, C_r is the surface runoff coefficient, and P_r is the average annual precipitation rate. The evapotranspiration coefficient is the fraction of precipitation that evaporates or is lost through transpiration from vegetation following root uptake. This factor typically is 0.7 for non-arid regions; for conservatism, a value of 0.6 is used here. The runoff coefficient is the fraction of precipitation that runs off before penetrating the soil. Values for a non-wooded agricultural environment range from 0.2 for flat open sandy loam to 0.7 for hilly land of impervious clay. Considering the top slopes and "impervious fill" layer shown in Fig. 7, a value of 0.6 is used here. The average precipitation for the Midwest area, according the National Climatic Center [18], is 0.92 m/y (36.25 inches per year). The infiltration rate calculated and used in RESRAD then becomes 0.4 x 0.4 x 0.92 = 0.147 m/y. This infiltration rate leads

to a total leachate rate over the 23,800 m^2 area of Trench C-2 of 3,500 m^3/y (1.76 gpm).

5.4.2.3 Cover Erosion and Soil Texture Parameter

Erosion rates depend upon slopes and the nature of vegetation cover. With a top cover more than nine feet thick, plus grass and erosion protection, significant removal of cover by erosion is not credible and there can be no removal of waste material by erosion. For conservatism, a relatively high rate of 0.0005 m/y is used. Over the 500 year range of calculations, this rate leads to 0.25 m of the 2.9 m removed by erosion.

A soil texture parameter, b, is used by RESRAD in calculation of volumetric water content in the various layers. Values of interest here are 4.05 for sand, 7.75 for silty clay loam, and 11.4 for clay.

5.4.2.4 Distribution Coefficient

The most important parameter in leaching of Cs-137 from the waste and in movement in groundwater is the distribution coefficient, K_d. This parameter expresses the ratio of the species sorbed onto solid particles to the amount remaining in the fluid. The higher the value of K_d, the more the species tends to be sorbed onto solid material and have movement retarded relative to the water movement. The retardation factor is

$$R_d = 1 + (\rho \times K_d)/(p_t \times R_s) \qquad (5)$$

where ρ is bulk soil density in g/cm^3, p_t is total porosity, and R_s is the saturation ratio (equals 1.0 if saturated and <1.0 if unsaturated). For example, for a value of K_d of only 10 cm^3/g, a typical soil density of 1.6 g/cm^3, and porosity 0.4, the retardation factor for saturated conditions becomes equal to 41. The velocity of a species of interest would be less than water velocity by this factor. For cesium, values of K_d are typically greater than 100, ranging as high as 18,000 in tuff [14, 19]. Specific data for the waste form and soil layers at the site are not available. A conservative value for sand, K_d = 80, is used in this study for base cases. The effects of other values are discussed later.

5.4.2.5 Summary of Model Data

Based upon data in the RCRA Permit Application [16], RESRAD guidelines [14], and other references, a conservative set of model data was compiled as shown in Table 5. Some of these values were previously discussed. The times used here for RESRAD calculations are 0, 1, 3, 10, 30, 50, 100, 200, 300, and 500 years. By the end of 500 y, Cs-137 will have decayed to only 10^{-3} of its initial value.

5.4.3 Leaching Model in RESRAD

The RESRAD code uses a sorption-desorption, ion-exchange leaching model. The fraction of available radionuclide leached out per year is the leach rate constant, L. The radionuclide release rate is

$$R = LAT\rho S \ , \ Ci/y \qquad (6)$$

where the last four factors express the radionuclide inventory. The emplaced waste has area A m^2, thickness T m, density ρ kg/m^3, and contaminating radionuclide concentration S Ci/kg. The leach rate constant is

$$L = I/\theta T R_d \qquad (7)$$

where I is the infiltration rate from Eq. (5), and R_d is the retardation factor from Eq. (4). The volumetric water content θ, is

$$\theta = p_t \ R_s \qquad (8)$$

Where p_t is total porosity as before, and R_s is the saturation ratio

$$R_s = (I/K_{sat})^{(1/2b+3)} \qquad (9)$$

where K_{sat} is the saturated hydraulic conductivity, and b is a soil-specific parameter. Table 4 lists values for T, ρ, p_t, K_{sat}, b, and K_d, used for each layer, including the contaminated (waste) layer.

The leach rate is dominated by the distribution coefficient, K_d, which is contained in the retardation factor, as discussed earlier. The release rate per unit area is approximately

$$R/A \approx IS/K_d \ . \qquad (10)$$

While the release rate increases with the infiltration rate (I), the corresponding increased water volume keeps the radionuclide concentration in the leachate almost constant for a given K_d, but the release rate and concentration are inversely proportional to K_d.

5.5 Results of Pathway Risk Analysis

5.5.1 Direct Exposure Pathways

As discussed in Section 5.3, direct radiation exposure at the landfill divides into several pathways: 1) shipment receival and treatment operations, 2) emplaced treated waste before covering, and 3) emplaced treated waste after covering. Two Cs-137 concentrations are involved for the base cases. The base case concentration in EAF dust is assumed to be 20 pCi/g. The dilution

Table 5. RESRAD Input Parameters ,/A facility

Parameter	Cover	Contaminated Zone	Unsaturated Zone 1 Drain	Unsaturated Zone 2 Clay	Unsaturated Zone 3 Illinoian	Unsaturated Zone 4 Shelby-	Saturated Zone
Thickness, m	2.9	8.1	0.4	0.9	16.	15.2	12.2
Density, g/cm^3	1.45	1.36	1.6	1.6	1.6	1.6	1.6
Total porosity	---	0.50	0.36	0.42	0.35	0.40	0.40
Effective porosity	---	0.30	0.28	0.05	0.05	0.15	0.15
"b" parameter	---	7.75	4.05	11.40	7.75	4.05	4.05
Cesium K_d, cm^3/g	---	80	80	80	80	80	80
Hyd. conductivity, cm/s		$2x10^{-4}$	$3.17x10^{-2}$	$1x10^{-7}$	$5x10^{-7}$	$3x10^{-3}$	$3x10^{-3}$
m/y		63.1	10,000	0.0315	0.1577	946.	946.

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factor during treatment is 0.77 (see Section 5.2). Therefore, incoming dust at 20 pCi/g results in corresponding treated waste of 20 x 0.77 = 15.4 pCi/g. For simplicity, this is rounded up to 16 pCi/g.

5.5.1.1 Exposure During and After Treatment

In Section 4, the base dose rate for a semi-infinite mass of EAF dust with Cs-137 contamination of 20 pCi/g would be 0.0115 mrem/h. The area/shape factor, F_s, for a 30 yd roll-off box, from Table 1, is 0.10 for the side, and 0.21 for the larger area of the top. This indicates, using Eq. 1, that a worker located 1 m from an incoming load, depending on whether along side or above the load, would receive a dose of 0.0012 to 0.0024 mrem/h. For continued receival of such loads, and 40 hours per week, 50 weeks per year, (2,000 h/y) the corresponding annual dose would be 2.4 to 4.8 mrem/y. It may be assumed that the pile of material during treatment has an exposure area comparable to the top of the shipping box (7.3 m x 2.1 m), with F_s = 0.21, and worker dose rate at 2,000 h/y is therefore not more than 4.8 mrem/y. Because of work patterns in which all of the time is not spent in the vicinity of waste being treated, and because all shipments would not be contaminated, the expected dose rates are less than the values calculated here.

After treatment, with the concentration diluted to 16 pCi/g, and density 1.36 g/cm^3 (2,300 lb/yd^3), the initial value of the DCF for the base case semi-infinite volume, by the RESRAD Code, is 5.23 (mrem/y)/(pCi/g). The DCF for the treated waste is plotted versus time in Fig. 8. By Eq. 1, the initial base case dose rate is 16 x 5.23 = 83.7 mrem/y for full-time exposure to a semi-infinite volume (0.0096 mrem/h). Again using F_s = 0.21, a worker who spends all working hours near piles or boxes of the affected waste (2,000 h/y) would be exposed to up to 4.0 mrem/y, slightly less than for the untreated EAF dust.

5.5.1.2 Exposure to Emplaced Treated Waste Before Covering

Emplacement of treated waste in a disposal trench is done in bands less than one meter wide. For conservatism, it is assumed here that successive contaminated shipments result in a large area (greater than 20 m radius) of contaminated emplacement, for which the area/shape factor, F_s = 1.0. For this geometry, and contamination diluted to 16 pCi/g by treatment, the DCF is as plotted in Fig. 8, with an initial value of 5.23 (mrem/y)/(pCi/g). The corresponding exposure dose rate is 0.0096 mrem/h. If contaminated waste occupies only one emplacement band, one meter wide, F_s = 0.12 (see Fig. 4), and the dose rate reduces by this factor to 0.0012 mrem/h (2.3 mrem/y for 2,000 h/y). Work patterns relative to the contaminated zone, plus partial to complete

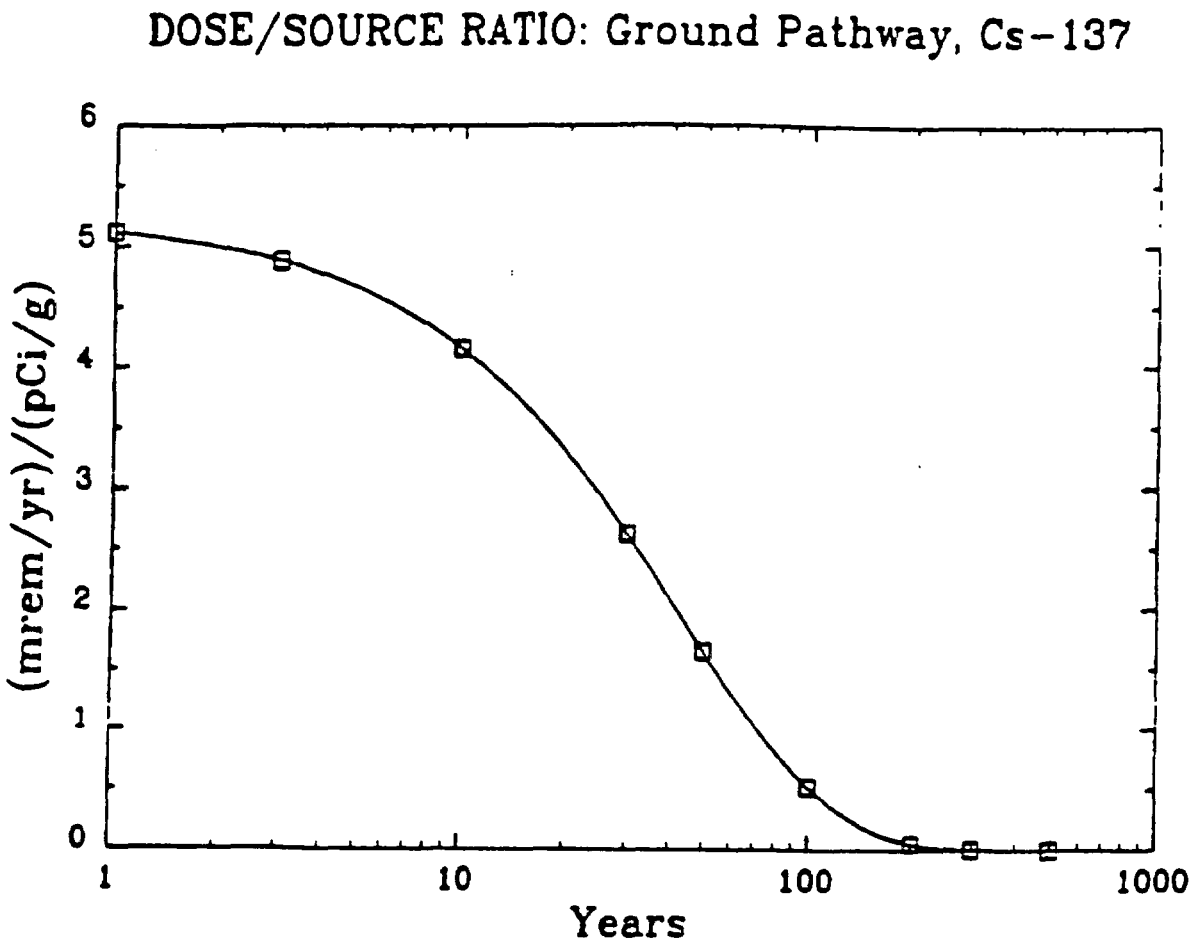

Fig. 8. Dose conversion factor for treated waste.

covering of emplaced waste, further reduce expected dose rates for the base case concentration.

5.5.1.3 Exposure to Emplaced Treated Waste With Cover

The shielding provided by the 2.9 m (9.5 ft) thickness of cover layers reduces the initial direct exposure rate for a large contaminated area by more than fourteen orders of magnitude to the negligible level of 1.6×10^{-13} mrem/y! Figure 9 shows how the initial very low dose conversion factor, DCF = 9.85×10^{-13} (mrem/y)/(pCi/g), decreases further with time. Figure 10 is a semi-log plot of the initial dose conversion factor (DCF) and annual dose rate for 16 pCi/g treated waste versus thickness of cover. Only the first 0.7 m of cover is included in the plot because the dose rate decreases to less than 1 mrem/y with only 0.4 m of cover.

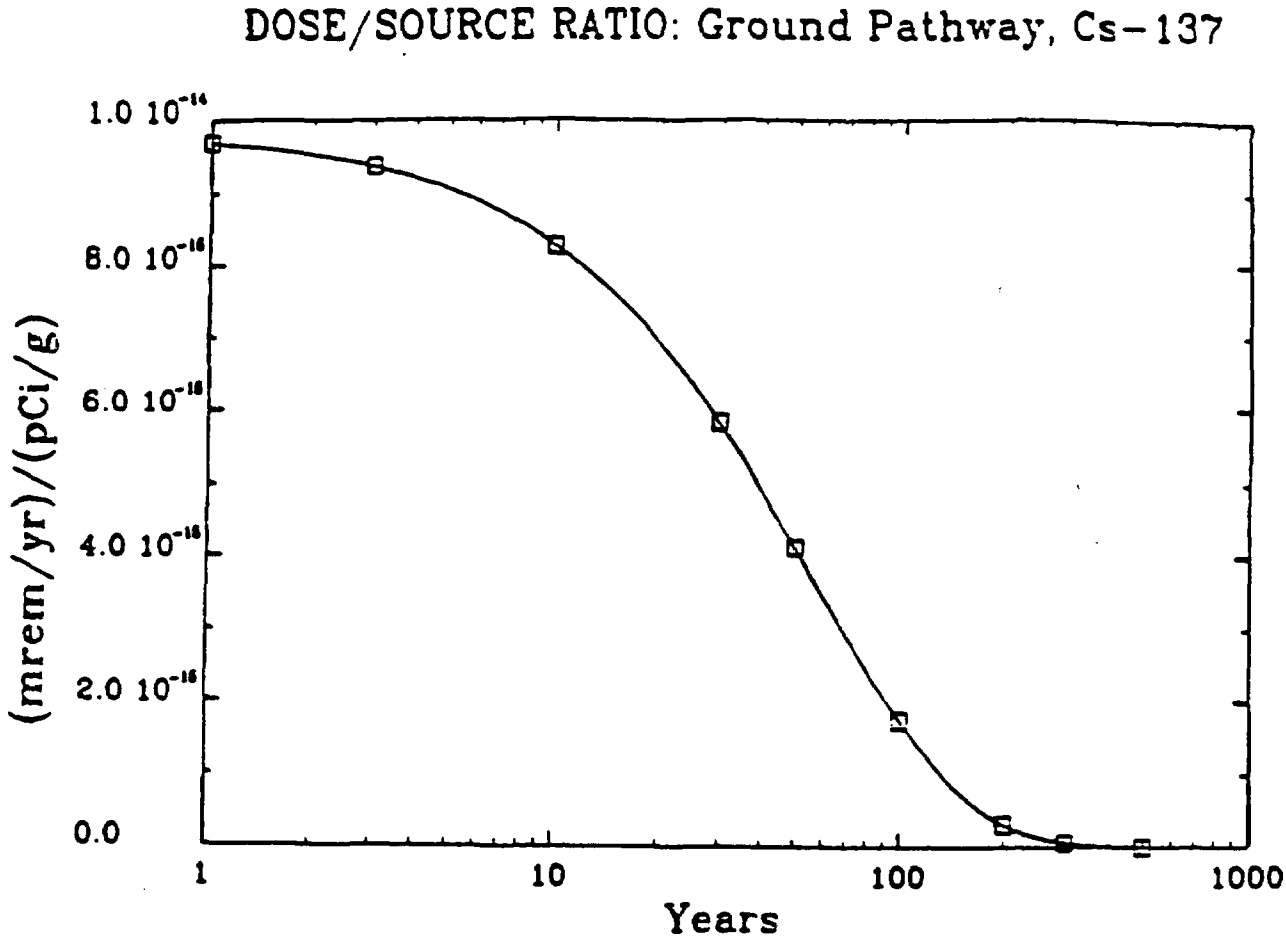

Fig. 9. Dose conversion factor for treated waste with top cover versus time.

5.5.2 Direct Leachate Drinking Scenario

This scenario assumes that the leachate collection system remains intact and drinking water is either withdrawn from the leachate sumps or via a well into the leachate drainage/bedding material. Modeling in RESRAD uses the layers in Table 4 only down to the FMLs (Flexible Membrane Layers), and the Drain Layer is considered to be a saturated layer instead of "Unsaturated Zone 1." This scenario assumes that the well is not continued down to the aquifer in the Shelbyville Outwash, and instead only seeks to make use of infiltrating water from precipitation. It is assumed that the entire disposal trench is filled with the Cs-137 contaminated waste material, and the calculations consider a 1,200 m^2 area within the trench. Actual emplacement of a contaminated load or sequence of contaminated loads would likely occupy only a small portion of a disposal trench. The base case for this scenario assumes a conservative value of 80 for K_d. The annual dose rate for an

corresponding leachate are inversely proportional to the value of K_d. For example, in the unlikely case that K_d is 40 instead of 80, the concentration and dose doubles. In the more likely case that K_d is 200, concentration and dose decreases by a factor of 2.5. If the thickness of contaminated waste is varied, the dose is not affected. This is because leaching and downward transport is limited by the sorption on the solid material. However, if a reduced contaminated material thickness is due to it being underlain by uncontaminated material, the resulting dose would be greatly reduced by sorption and retardation in the "clean" material. If the infiltration rate is increased, the dose rate is not affected. An increase in infiltration increases the leach rate in direct proportion, but the corresponding increase in water flow volume maintains the same concentration and hence the same dose rate from drinking 2 L/d. A 50% increase in infiltration would result if the precipitation increased by 50%, or if the runoff factor is reduced from 0.60 to 0.40 (see Eq. 4).

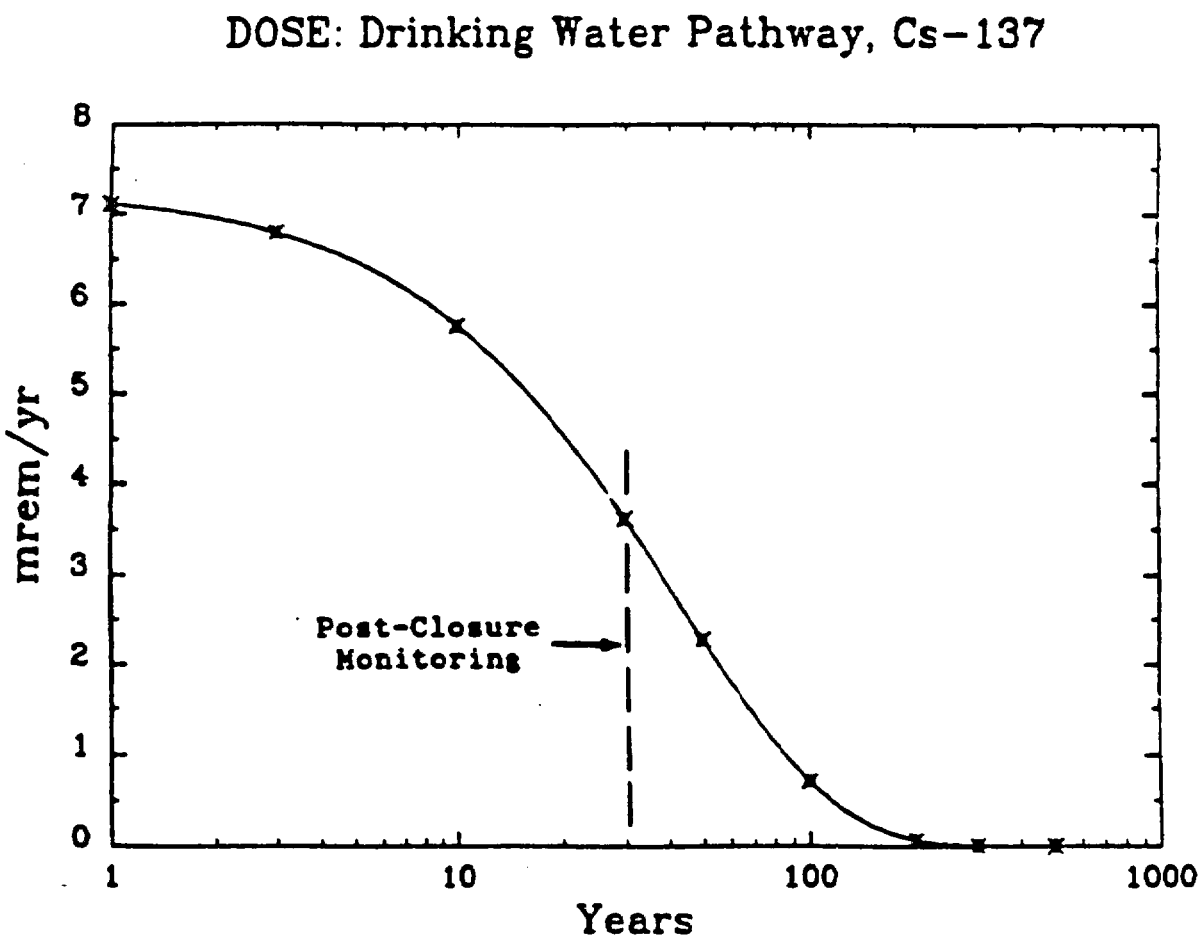

Fig. 11. Dose rate from direct drinking of leachate.

5.5.3 Aquifer Drinking Water Scenario

This scenario assumes that the leachate collection system degrades, permitting the leachate to move down to the aquifer, and drinking water is withdrawn via a well into the aquifer at the down-flow edge of the disposal trench. Primarily, this assumes that the FMLs (Flexible Membrane Liners) (Fig. 7 and Table 4) disintegrate or sufficiently crack and split that their barrier properties fail. Again, as a severe case, it is assumed that the entire trench is filled with Cs-137 contaminated waste. For this scenario, the layering model presented in Table 4 is used, in which the vertical distance from the bottom of the waste to the water table is 32.5 m (107 ft).

Results show that with reasonable values of K_d, such as 80 cm^3/g, the Cs-137 decays to negligible levels long before the time when contaminated leachate reaches the water table. This is best illustrated by discussing results obtained by using a value of K_d = 2, which is much lower than ever encountered for cesium in a non-brine environment. The dose rate versus time for this very conservative case is shown in Fig. 12. The maximum dose rate is 1.83 mrem/y, reached after 214 y. The corresponding concentration of Cs-137 reaching the well is 0.05 pCi/ml. The drinking water standards limit the concentration to 0.11 pCi/ml for 4 mrem/y dose rate. "Breakthrough" occurs at 196 y, and the rise time to the maximum takes another 18 y. The tail of the curve represents the combined effect of radiodecay and completion of inventory leaching. The retardation factors and vertical travel times for the K_d = 2 case are listed in Table 4. The sum of the travel times for all layers is the breakthrough time into the aquifer. The well is assumed to be immediately adjacent to the disposal trench and no delay results for horizontal travel time in the aquifer. The rise time represents the time for the plume depth to build up from further upflow. The corresponding time required for water to flow down without retardation (K_d = 0) is 13 y. A value for K_d of only about 10 would move the curve off-scale in Fig. 12 (residual radioactivity essentially zero).

Results for K_d = 80 are also listed in Table 6. Now the breakthrough time is 7,320 y and the Cs-137 will undergo 10 half-lives and decay to only 0.001 times the initial level before breaking through the clay layer over 30 m (over 100 ft) above the aquifer. This means that the cesium does not reach the aquifer and basically zero dose results from this scenario, regardless of initial concentration in the waste material. This also indicates that this pathway does not lead to transport of cesium off site.

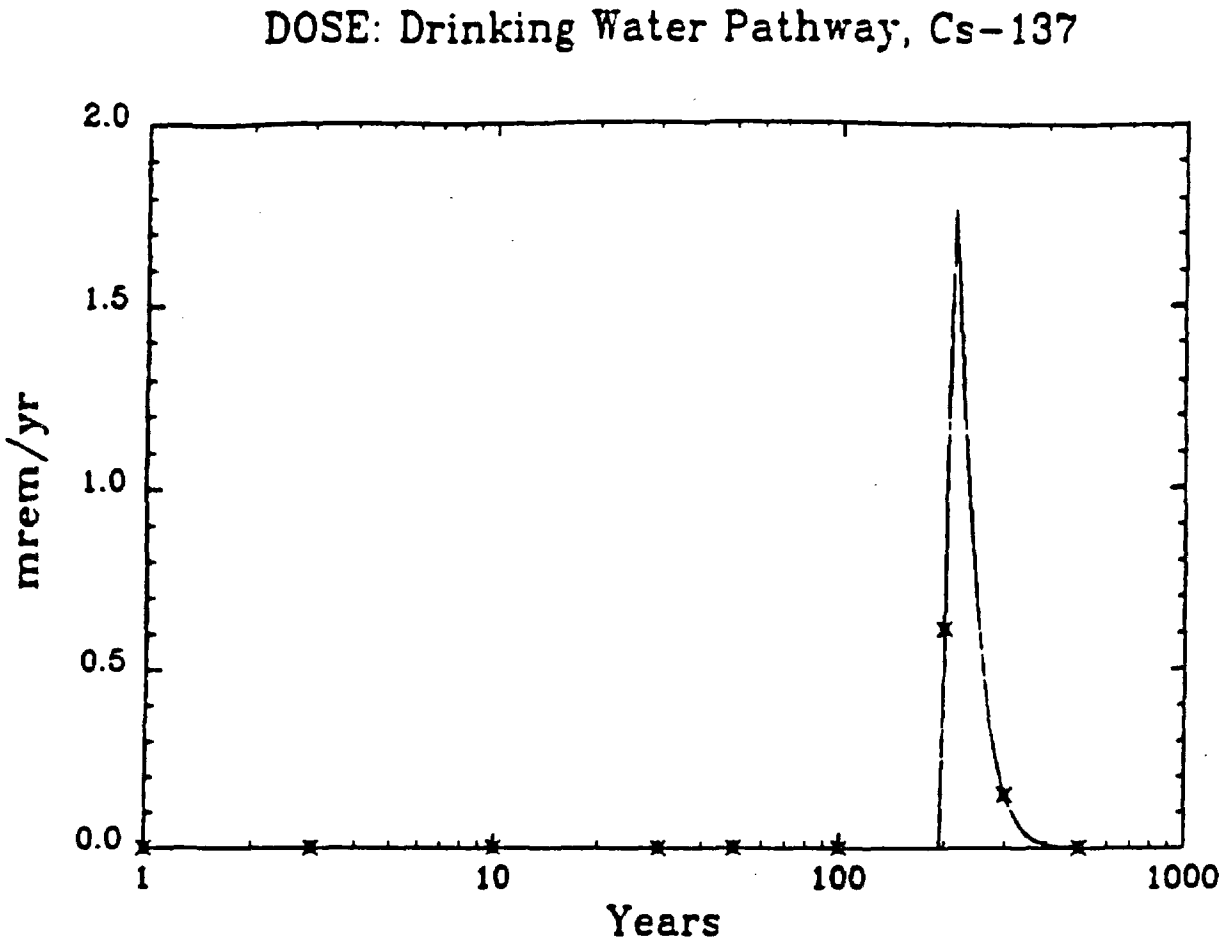

Fig. 12. Dose rate from aquifer drinking water source.

Table 6. Retardation and Travel Times to Aquifer and Well.

	K_d = 2 *		K_d = 80	
Layer	R_d	time, y	R_d	time, y
Drain	25.2	7.0	970	271
Clay	86.2	2.6	306	93
Illinoian	10.2	55.1	368	1,993
Shelbyville	18.6	130.9	706	4,964
Breakthrough Time		195.7		7,321
Rise Time		18.3		654

* value of 2 used for demonstration only

6. PROCESSING AT HORSEHEAD

6.1 Description of Processing at Horsehead

Dust from electric arc furnaces (EAF dust) is processed at Horsehead Resource Development Co., Inc. to remove lead, zinc, copper, and cadmium. In addition to removing hazardous components, processing recovers valuable metals such as zinc. A residual material called "IRM" (Iron Rich Material) has had hazardous materials removed, and is released for unrestricted use in the construction industry.

The processing sequence is shown in the flow chart in Fig. 13. The top box: "Subject EAF Dust," refers to a load being tracked such as material contaminated with a radioisotope. An incoming load of Subject EAF dust at the Waelzing Plant at Chicago, Illinois is blended at typically 10:1 with material from other sources (10 parts blended mixture to 1 part incoming load considered) and conditioned with addition of water. The mixture is conveyed in enclosed transfer via the Feed Building to Waelzing rotary kilns, operated at temperatures up to 1,300 C. Zn, Pb, and Cd volatilize and with suspended dust in the off gas go to the bag room where the collected material is called "Crude ZnO." Heavy unreacted material in the off gas is collected in a settling chamber and recycled. Air flow to the kiln is 100,000 actual cfm for a capacity of 275 tons of EAF dust per day. Solid material discharged from the kiln is the IRM, and air discharged from the bag room is released through a stack.

The Crude ZnO is shipped by rail to the Palmerton Plant at Palmerton, Pennsylvania for upgrading. An incoming load is blended at typically 5:1 with material from other shipments (5 parts blended mixture to 1 part incoming load considered) and conditioned with addition of water. The mixture is conveyed in enclosed transfer to rotary calcining kilns, operated at temperatures up to 1,100 C. Pb and Cd volatilize along with 5-10% of the zinc, under oxidizing conditions, and go to the bag room where the collected material is called "lead chloride concentrate." Again, a settling chamber is used to collect unreacted material for recycling. The air flow rate to the kiln is 68,000 actual cfm for a capacity of 210 tons of Crude ZnO per day. Most of the zinc is discharged from the kilns in the calcined material.

The lead chloride concentrate is packaged in "supersacks" (4 ft diameter by 8 ft tall) and shipped in covered gondola rail cars to a plant in Oklahoma for wet processing to recover $PbSO_4$, $ZnCo_4$, Cu sponge, and Cd sponge. The remaining solution after removal of metals goes to a waste water tank at the plant waste water treatment facility. 2,200 gal of waste water is produced per ton of feed to wet processing. This corresponds to 110 gal per ton of original EAF feed. An incoming shipment of lead chloride

concentrate is diluted by an undetermined amount of material from other sources.

6.2 Fate of Cesium in EAF Dust

Cesium is a volatile metal, melting at 28.5 C and boiling at 670 C. The form of cesium in EAF dust is as one of the oxides, which decompose at temperatures between 360 and 600 C. It is because of the volatility that the cesium in the melting incident ended up in the EAF dust; there was none detected in slag or the steel product. The likely fate of the cesium during processing at Horsehead is to 1) volatilize in the Waelzing kiln and be collected in the Crude ZnO, 2) revolatilize in the calcining kiln and be collected in the lead chloride concentrate, and 3) dissolve into wet processing solutions and end up in the waste water tank. There are some variations covered in the pathway analysis in the next section.

6.3 Pathways Considered

In the pathway analysis, the level of Cs-137 contamination in EAF dust is assumed to be 20 pCi/g. Settled EAF dust as shipped is assumed to have a typical density of 2,900 lb per cubic yard (1.72 g/ml). Results scale linearly to other levels of contamination. The pathways considered are:

1. Direct exposure to contaminated EAF dust.
2. Direct exposure to materials during processing.
3. Release to atmosphere at Chicago facility.
4. Release to air at Palmerton facility.
5. Release to waste water tank at Oklahoma facility.

It is expected that conditioning of the dust by addition of water, coupled with enclosed transfer and low Cs-137 concentrations, reduces inhalation and ingestion hazards to negligible levels. These pathways are not examined further here.

Nomenclature:

W_c = daily receipt of contaminated EAF dust, tons
W_w = daily total feed at Waelzing Plant, tons
C_c = Cs-137 concentration in contaminated dust, pCi/g
C_w = Cs-137 concentration in Waelzing Plant feed, pCi/g
C_p = Cs-137 concentration in Palmerton Plant feed, pCi/g
C_{Pb} = Cs-137 concentration in lead concentrate, pCi/g
DCF = direct exposure dose conversion factor, (mrem/y)/(pCi/g)
D = direct exposure dose rate, mrem/h

Unit conversion factors:

8,760 = h/y (hours/year)
2,000 = lb/ton
454 = g/lb
24 = h/d (hours/day)
28.32 = L/ft^3 (Liters/ft^3)
3,785 = ml/gal (milliliters/gallon)

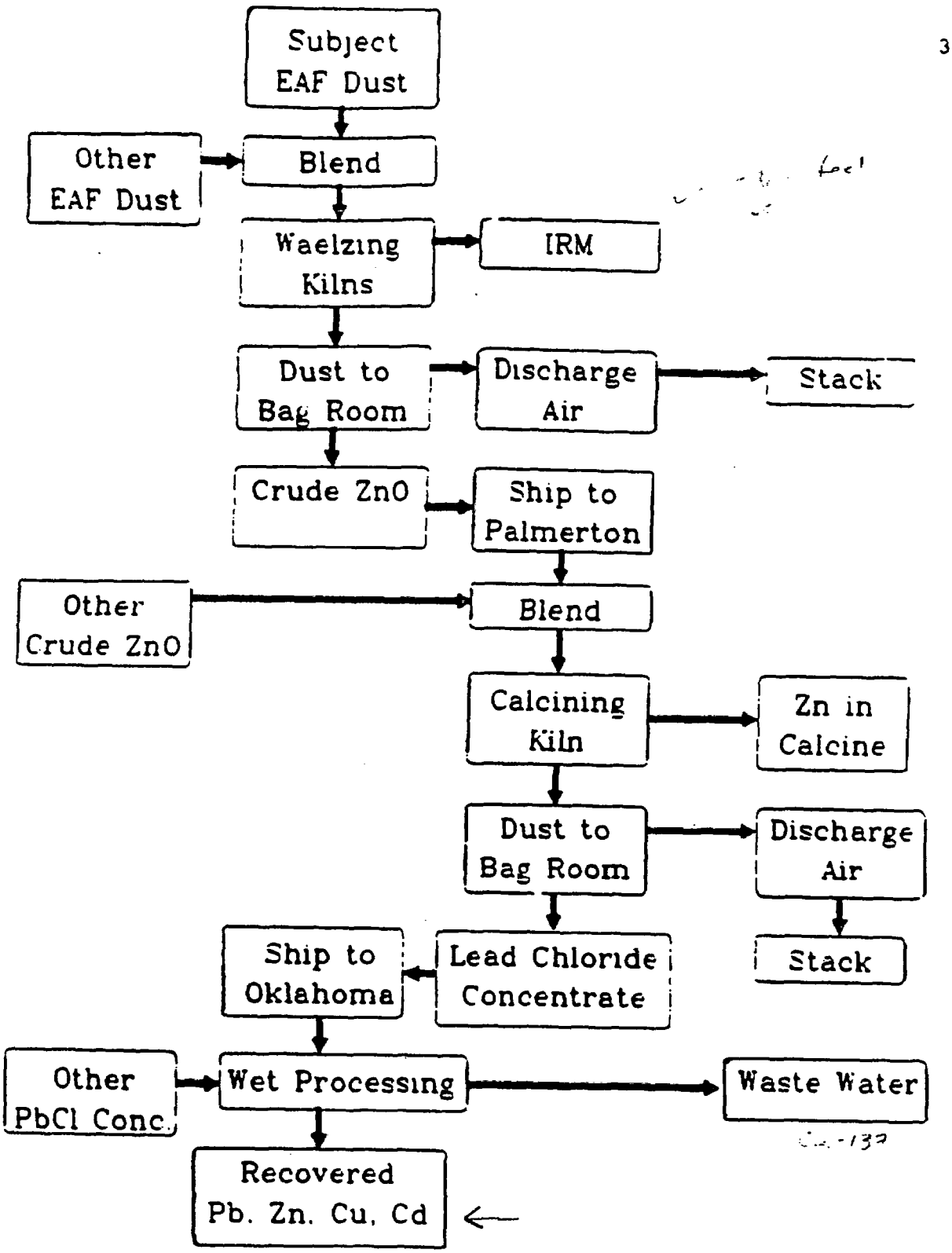

Fig. 13. Horsehead schematic flow chart.

6.3.1. Direct Exposure to Contaminated EAF Dust

The starting point in pathway calculations is to use the RESRAD computer code to obtain the radiation dose at a distance of 1 m from the surface of a semi-infinite volume of EAF dust. This refers to a volume of infinite lateral extent and infinite depth. This condition is reached for practical purposes with a radius of 20 m or larger. Because of self-shielding effects, the outer 0.5 m of material contributes almost all of the radiation dose, with the outer 0.25 m contributing 92% of the total. For smaller and non-circular sources, a "shape/area factor", less than 1.0, is applied to the dose rate for a semi-infinite source to obtain the lower dose rate for a given finite source. The RESRAD code yields a dose conversion factor (DCF) of 5.05 (mrem/y)/(pCi/g). For a contamination concentration of 20 pCi/g, this base dose rate becomes

$$D_{base} = DCF \times C_c/8{,}760 = 5.05 \times 20/8{,}760 = 0.0115 \text{ mrem/h.} \quad (11)$$

A 30 ton shipment of contaminated EAF dust in the shipping box or trailer, or when dumped on the receiving building floor can be represented by a rectangular exposure area of approximately 7 m by 2 m. The shape/area factor, F_g, using the method of the RESRAD code is 0.18. Direct exposure dose to this finite source becomes

$$D = D_{base} \times F_g = 0.0115 \times 0.18 = 0.0021 \text{ mrem/h.} \quad (12)$$

A worker standing next to this source for 40 hours per week, 50 weeks per year would receive 0.0021 x 40 x 50 = 4.1 mrem/y. Expected work patterns of intermittent presence and greater than one meter average distance reduce the expected dose rate.

A daily total feed at the Waelzing Plant of 300 tons dilutes the contaminated material during blending by a factor of 10:

$$C_w = C_c \times W_c/W_w = 20 \times 30/300 = 2 \text{ pCi/g.} \quad (13)$$

A corresponding pile of blended feed can be represented by a rectangle 15 m by 4 m. The shape/area factor for this source is 0.40, and the direct exposure dose for this larger, but lower concentration, pile becomes

$$D = 0.0115 \times (2/20) \times 0.4 = 4.6 \times 10^{-4} \text{ mrem/h.} \quad (14)$$

A worker standing next to this pile for 40 hours per week, 50 weeks per year would receive only 0.92 mrem/y. Again, expected work patterns reduce the expected dose rate.

The receiving building can have several thousand tons of blended material on hand. Additional material generally would lead to

lower average contaminant concentrations with a somewhat larger shape/area factor, but with decreased net dose rate to a worker.

6.3.2. Direct Exposure to Materials During Processing

The base exposure rate for a large exposure area (F_s = 1.0) was previously shown to be D_{base} = 0.0115 mrem/h. The corresponding base rate for blended feed became 0.1 x D_{base}. For each ton of EAF feed, 0.4 ton of Crude ZnO leaves Chicago. The base exposure rate (neglecting reduction for finite dimensions) for Crude ZnO becomes D_{base} x 0.1 x (1/0.4) = 0.25 D_{base}.

Feed at Palmerton is blended with feed from other sources typically at 1 ton from Chicago per 5 tons total. The base rate for Palmerton Crude ZnO feed becomes 0.1 x (1/0.4) x (1/5) D_{base} = 0.05 D_{base}. Finally, 1 ton of lead concentrate leaves Palmerton per 8 tons of Crude ZnO. The corresponding base exposure rate for the lead concentrate becomes 0.1 x (1/0.4) x (1/5) x (8/1) x D_{base} = 0.40 D_{base}.

During processing, the net effect of dilution and concentration at each stage is that the direct exposure dose to a large mass of material ranges from 0.05 to 0.40 times the base rate for the original incoming contaminated EAF dust. At each stage, various values of the shape/area factor, less than 1.0 and likely less than 0.4, apply to reduce the finite source dose rate.

6.3.3. Release to Atmosphere at Chicago facility

The Cs-137 volatilized in the Waelzing kiln in Chicago is trapped in the dust collected in the bag room. As a severe case, assume that <u>none</u> of the cesium is trapped and <u>all</u> of it escapes up the stack to the atmosphere. The kiln air flow rate is 100,000 acfm for a process rate of 275 tons per day. The air concentration for this extreme case becomes (for C_v = 2 pCi/g)

$$C_v \times 275 \times (1/100{,}000 \times 60) \times 2000 \times 454 \times (1/24) \times (1/28.32) = 0.122 \text{ pCi/L}. \quad (15)$$

The limit for unrestricted release of Cs-137 in air, per 10 CFR 20 [1], is 2.0 pCi/L, a factor of 16.3 times the value for the complete release to air. Any actual release can be expected to be limited to a few percent of the value obtained above.

6.3.4. Release to Air at Palmerton Facility

The Cs-137 concentration in Palmerton feed, based upon 0.4 ton of Crude ZnO per ton of EAF dust, and 5 tons of Palmerton blended feed per ton of Crude ZnO from Chicago is

$$C_p = C_v \times (1/0.4) \times (1/5) = C_v \times 0.5 = 2 \times 0.5 = 1 \text{ pCi/g}. \quad (16)$$

For this scenario, assume all of the cesium is trapped in the dust in Chicago and subsequently all of it escapes up the stack to the atmosphere at Palmerton. The kiln air flow rate is 68,000 acfm for a Crude ZnO process rate of 210 ton per day. The air concentration for this extreme case becomes

$$C_p \times 210 \times (1/68{,}000 \times 60) \times 2000 \times 454 \times (1/24) \times (1/28.32) = 0.069 \text{ pCi/L}. \quad (17)$$

The limit for unrestricted release (2.0 pCi/L) is 29.1 times the value for the complete release to air. Any actual release would be limited to a few percent of the above value and would also be reduced by any fraction released to the atmosphere at the Chicago Plant.

6.3.5. Release to Waste Water Tank at Oklahoma Facility

One ton of lead concentrate is produced from 8 tons of Crude ZnO at the Palmerton Plant. The concentration of Cs-137 in the lead concentrate becomes

$$C_{Pb} = C_p \times 8. \quad (18)$$

There are 2,200 gal of waste water produced per ton of lead concentrate processed. The concentration of Cs-137 in the waste water going to the waste water tank at the Oklahoma facility after removal of other metals is

$$C_{ww} = C_{Pb} \times (1/2{,}200) \times 2{,}000 \times 454 \times (1/3{,}785) = 0.87 \text{ pCi/ml}. \quad (19)$$

The limit for unrestricted release of Cs-137 in water, per 10 CFR 20 [1], is 20. pCi/ml. The limit is a factor of 22.9 times the concentration in the waste water if all of the cesium passes through to that point. This calculation does not take credit for the dilution of the wet process feed with material from other sources.

6.4 Discussion of Pathways at Horsehead

Results indicate that radiation dose from direct gamma exposure (actually x-rays from the barium daughter of cesium) is the greatest for the incoming contaminated EAF dust. Considering a reasonable geometry, the dose to a worker standing next to a shipping trailer or pile of dumped EAF dust, assuming 20 pCi/g, was obtained by the RESRAD code to be 0.0021 mrem/h. If a worker remained in such a position for the entire working year, the dose would be 4.1 mrem/y. This would be reduced an undetermined amount by actual expected work patterns of intermittent presence and greater than one meter average distance from the material. The dose rate for other concentrations of Cs-137 are proportional and would be $0.0021/20 = 1.05 \times 10^{-4}$ (mrem/h)/(pCi/g).

The net effect of dilution and concentration at each stage through the multi-plant processing is that the direct exposure dose to a large mass of material ranges from 0.05 to 0.40 times that of the incoming contaminated EAF dust. At each stage, various values of the shape/area factor less than 1.0 apply to reduce the dose rate for the finite sources. For comparison, it may be noted that the 10 CFR 60 regulations for nuclear waste repositories [7] limit the radiation dose rate to any member of the public to 25 mrem/y. The average dose rate from background sources in the United States, including radon, is approximately 300 mrem/y.

Extreme scenarios for total releases to the atmosphere, involving no bag room trapping at the Chicago or Palmerton Plants, result in air concentrations less than the limit by factors of 16.3 and 29.1, respectively. This means that these scenarios would produce marginal air concentrations with 325 or 580 pCi/g of Cs-137 in the original EAF dust, or higher contamination values for partial releases to the atmosphere.

If all of the cesium passes through to wet processing waste water, the Cs-137 concentration in the waste water tank is less than the limit by a factor of 22.9. This implies that the waste water would become marginal for an EAF dust concentration of 22.9 x 20 = 450 pCi/g.

7. TREATMENT TO REMOVE CESIUM

Treatment to remove cesium from the EAF dust may be feasible, using a soil-washing process in which the cesium is dissolved into solution and sorbed in canisters of ion-exchange resin. All of the cesium cannot be removed by such a process, but the objective is to remove enough of the radioactive component to reduce concentrations to limits expected in new regulations which will permit processing at Horsehead or disposal in a hazardous waste landfill. A treatment flow diagram is shown in Fig. 14.

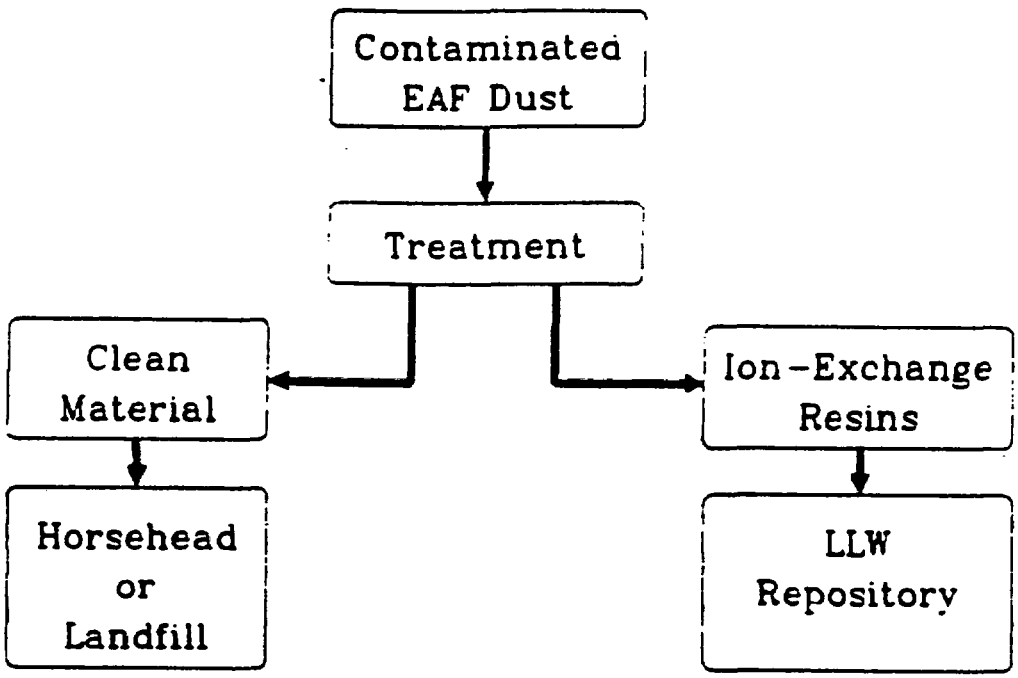

Fig. 14. Cesium removal treatment flow diagram.

Treatment may involve one or more than one pass to achieve the concentration levels required for the "clean" material. The ion-exchange resins end up with concentrated Cs-137 and must be disposed in a low-level-waste repository. Laboratory scale tests are being done by Lockheed Environmental Systems & Technologies Company to determine processing parameters and capabilities.

8. DISCUSSION AND CONCLUSIONS

The EAF dust is designated as hazardous waste No. K061, and with Cs-137 contamination is "mixed waste." Regulations are not generally in place for very low levels of radioactivity, and particularly are not in place for mixed waste. The lowest class of Low-Level Waste (Class A) permits up to 10^6 pCi/ml of Cs-137. This is almost 62 times as great as the maximum concentration in the most contaminated container of stored dust at Keystone (Box No. 1). It is also 10,000 times as great as the average concentration in the other 50 boxes of stored waste from the clean-out operations. Clearly, handling the contaminated EAF dust as LLW would be "overkill" with unnecessary expense and using repository space better used for higher activity material.

Section 2.4 of this report discusses various agency regulations and guidelines. Generalized radiation protection standards limit radiation dose to the public to 100 mrem/y. Requirements for LLW and HLW repositories tighten the limit for those applications "to any member of the public" to 25 mrem/y. The concentration limit for Cs-137 release in water to unrestricted areas is 20 pCi/ml. NRC guidelines for release of land for unrestricted use limit Cs-137 to 0.2 pCi/ml in groundwater and 15 pCi/g in soil. Drinking water standards limit Cs-137 further, to 0.11 pCi/ml, corresponding to a dose rate of 4 mrem/y for ingestion at the rate of 2 L/d. The limit for release of Cs-137 to air for unrestricted use is 2 pCi/L. Regulations for packaging and shipping radioactive material limit the dose rate at the package surface to 200 mrem/h (and 10 mrem/h at a distance of 1 m). Using the dose conversion factor, 5.05 (mrem/y)/(pCi/g), obtained by the RESRAD code, the package surface limit corresponds to a Cs-137 concentration of 346,930 pCi/g (0.347 microcurie per gram)!

Considering the range of regulatory requirements for various circumstances, and the lack of regulations for extremely low-level radioactive and mixed wastes, the risk analysis of pathways in this study is not targeted at clearly defined regulatory limits. For this reason, the analyses are normalized to a Cs-137 concentration of 20 pCi/g in the contaminated EAF dust. This level was chosen as a possible reasonable value for new regulations, but noting that the reference level and corresponding calculated results may be scaled either up or down as new regulations evolve.

When considering an annual dose limit, it should be recognized that this does not necessarily involve full-time exposure. Activity patterns of workers and members of the public may involve intermittent exposure and varying distances. For example, the transportation limit of 10 mrem/y at a distance of 1 m, implies that no member of the public is expected to spend more than 10 hours in close proximity to such a package, or equivalent longer

periods of time at greater distances, in order to meet a 100 mrem/y limit.

8.1 Discussion of Calculated Results

All of the calculations, using the RESRAD code [14], were made with the Cs-137 concentration in EAF dust normalized to 20 pCi/g. Corresponding to this level, the concentration in treated waste is 16 pCi/g, and various concentrations apply during processing at Horsehead. Any of the numbers in the results scale linearly for other initial EAF dust concentrations.

8.1.1 Direct Exposure Pathways

A starting point is to summarize results for direct exposure to semi-infinite volumes of contaminated materials, followed by discussion of area/shape factors and exposure time patterns. These results are listed in Table 7.

Table 7. Direct Exposure to Semi-Infinite Volumes of Material Normalized to 20 pCi/g in EAF Dust.

	mrem/y	mrem/h
EAF Dust Storage and Transportation:		
Settled Bag House Dust	101.	0.0115
Denser Roof Duct Dust	86.7	0.0099
Disposal in Landfill:		
Before Treatment	101.	0.0115
After Treatment	83.7	0.0096
Emplaced Without Cover	83.7	0.0096
Emplaced With Cover	0	0
Processing at Horsehead:		
Receival of EAF Dust	101.	0.0115
Waelzing Plant Feed	10.1	0.0012
Crude ZnO	25.3	0.0029
Palmerton Plant Feed	5.1	0.0006
Lead Concentrate	40.4	0.0046

The Area/shape factors estimated for EAF containers range from 0.08 to 0.10 for the sides and 0.20 to 0.26 for the larger top areas. Factors for a load dumped at the landfill or Horsehead for treatment are estimated at 0.18. The factor for the larger pile of Waelzing Plant feed is estimated at 0.4, but this applies to the material diluted by 10:1. Therefore, the exposure dose to finite volumes of contaminated material ranges from about one-tenth to one-quarter of the values listed in Table 7 for semi-infinite volumes.

During processing at horsehead, the net effect of dilution and concentration is to reduce the potential dose relative to conditions at the time of receival. In fact, the EAF dust while in storage and before treatment at either the landfill or Horsehead, represents the greatest potential dose from direct exposure.

Exposure times from work patterns and patterns of the public related to transportation reduce the dose from the full-time mrem/y values in Table 7, in addition to the area/shape factors. The annual exposure time for a worker at 40 hours per week, 50 weeks per year is 0.23 y. Potential annual exposure time for a member of the public during shipment of contaminated loads is much less than for a full-time worker. The 10 mrem/h at a distance of 1 m limit in transportation regulations suggests a maximum of 10 hours per year to a member of the public to meet a 100 mrem/y guideline. Assuming a worker spends full-time in the vicinity of the contaminated material, and using the 0.26 maximum area/shape factor for shipping containers, the maximum dose to a worker (greater than to the public) appears to be 101 x .23 x .26 = 6.04 mrem/y. This suggests that the Cs-137 concentration could be 16 times the assumed 20 pCi/g level before exceeding a 100 mrem/y guideline, or a factor of 4 if 25 mrem/y is used. The corresponding concentration limits would become 320 pCi/g and 80 pCi/g, respectively.

8.1.2 Groundwater Ingestion Pathways

The groundwater pathways for disposal at the landfill assume the severe condition that Trench C-2 is filled entirely with the contaminated material. This serves as a starting point for considering fractions expected to actually be filled with such material. The groundwater pathway results are summarized in Table 8 for ingestion of affected water at the rate of 2 L/d.

Table 8. Ingestion Dose for Groundwater Pathways, with K_d = 80.

	mrem/y	pCi/ml
Direct Leachate Drinking:		
Initial, at time zero	7.28	0.20
After 30 y control period	3.63	0.10
Drinking from Aquifer	0	0

Results indicate that a person drawing water directly from the leachate collection system after the end of the post-operations 30-year control period would find water meeting drinking water standards (less than 4 mrem/y and 0.11 pCi/ml Cs-137). If the leachate collection system is assumed to degrade and fail, water would travel down from the waste zone to the aquifer in 13 y, but retardation would delay the cesium transport to a well drawing water from the aquifer for from hundreds to thousands of years, far

longer than the time required to decay to negligible levels. Two other factors act to reduce the consequences of both of these scenarios. First, the value of 80 for K_d is lower than reasonably expected. Higher values increase retardation and reduce potential dose. Secondly, it is unlikely that waste contaminated with radioactive components would occupy more than a few percent of the trench volume, greatly reducing potential dose. While higher K_d values are expected for the waste form and soil layers, laboratory tests would be required to measure actual values. Because of retardation of movement using the conservative value for K_d, there is no upper limit for cesium contamination for the aquifer drinking scenario. Concerning the direct leachate drinking scenario, if as much as 10% of the trench contains contaminated waste material, the Cs-137 content of the EAF dust could be a factor of 10 greater than 20 pCi/g, or 200 pCi/g, before the average leachate would exceed drinking water standards after the post-operations control period expires.

8.1.3 Pathways to Air and Water During Processing at Horsehead

The likely fate of cesium during processing at Horsehead is to 1) volatilize in the Waelzing kiln and be collected in the Crude ZnO, 2) revolatilize in the calcining kiln and be collected in the lead chloride concentrate, and 3) dissolve into wet processing solutions and end up in the waste water tank. The three air and water pathways considered assume that all of the cesium does end up in the waste water or that it all somehow escapes into the stack and atmosphere at one or the other kilns. Any combination of these three scenarios would reduce the impact of all three. Table 9 lists the results.

Table 9. Maximum Releases to Air and Water During Processing at Horsehead

	pCi/L	Allowable pCi/L
Maximum Air Releases:		
Waelzing Plant	0.122	2.0
Palmerton Plant	0.069	2.0
	pCi/ml	pCi/ml
Maximum Release to Waste Water	0.87	20.

Results indicate that Cs-137 contamination could be 16 to 29 times the assumed 20 pCi/g before exceeding allowable concentrations for release to unrestricted areas for these air and water pathways. A factor of 15 represents 300 pCi/g.

8.2 Conclusions

It is very important that regulatory relief be obtained to permit doing something with EAF dust contaminated with Cs-137 instead of continuing to store the material. Any level of pCi/g that is established by the Illinois Department of Nuclear Safety permits at least some of the stored dust to be routed immediately to Horsehead for processing or to the landfill for disposal. This is particularly important for quantities of dust having only a few pCi/g or less, collected since the clean-out operations. Depending upon the level established, additional quantities, and perhaps all, of the dust in storage from the clean-out can be sent to Horsehead and/or the landfill after soil washing treatment for cesium removal. If the established regulatory level is not sufficiently high, the quantity that can be treated by soil washing with cost effectiveness is lowered, and the only recourse for the remainder of material is to ship it to an out-of-state LLW repository that is also licensed to receive mixed waste. This would be an expensive alternate that would utilize LLW repository capacity better used for higher activity material. Also, packaging and out-of-state shipment of substantial quantities would likely incur greater risk than associated with the in-state options and pathways considered in this study.

Table 10 provides a summary of the number of boxes among the 51 boxes of stored dust from clean-out operations that can be sent to Horsehead or the landfill with and without soil washing treatment to remove cesium. Removals of 80% and 90% are assumed for illustration.

Table 10. Number of Boxes of EAF Dust Released to Horsehead or Landfill with Various Regulatory Limits

Reg. Limit	No Treatment	Cesium Removal Treatment			
		80% Removal		90% Removal	
pCi/g	No. Boxes	pCi/g	No. Boxes	pCi/g	No. Boxes
10	7	50	24	100	35
15	8	75	29	150	43
20	10	100	35	200	45
25	12	125	40	250	47
30	13	150	43	300	48
40	19	200	45	400	49
50	24	250	47	500	49
80	29	400	49	800	50
100	35	500	49	1,000	50

Results are not yet available from Lockheed laboratory tests that are being conducted to determine process parameters and attainable removal effectiveness. The one or several boxes having the highest cesium concentrations likely require more than one treatment pass to meet reasonable regulatory limits.

The risk analysis results discussed earlier for the pathways considered indicate that Cs-137 concentrations in EAF dust up to 100 pCi/g or greater would not cause workers in unrestricted areas or members of the public to receive doses exceeding guidelines that are in general use. It should be noted that the analyses are based upon a continual flow of contaminated material. It may be that two levels of regulatory limits can be considered: one lower level for continuous control plus a higher limit for one incident or an occasional incident.

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APPENDIX A

CESIUM IN KEYSTONE DUST THROUGH END OF CLEAN OUT KEY0404A.XLS

Box No	Tons	pCi/g	pCi	mCi	
1	21.3	10530	2.02E+11	202.31	Avg = 0.65 x Max
2	17.7	742.3	1.19E+10	11.93	in Box 1
3	19.7	340.5	6.09E+09	6.09	
4	19.3	203.9	3.57E+09	3.57	
5	18	94.4	1.54E+09	1.54	
6	18	94.8	1.55E+09	1.55	
7	12	152.1	1.66E+09	1.66	
8	12	48.5	5.28E+08	0.53	
9	12	47.2	5.14E+08	0.51	
10	12	38.9	4.24E+08	0.42	
11	12	34.5	3.76E+08	0.38	
12	12	31.3	3.41E+08	0.34	
13	12	33.8	3.68E+08	0.37	
14	12	24.1	2.63E+08	0.26	
15	12	19.6	2.14E+08	0.21	
16	12	15.8	1.72E+08	0.17	
17	12	115.3	1.26E+09	1.26	
18	12	71.7	7.81E+08	0.78	
19	12	85.2	9.28E+08	0.93	
20	12	20.8	2.27E+08	0.23	
21	12	160.3	1.75E+09	1.75	
22	12	118.4	1.29E+09	1.29	
23	12	146.7	1.60E+09	1.60	
24	12	62.4	6.80E+08	0.68	
25	12	2.6	2.83E+07	0.03	
26	12	43	4.69E+08	0.47	
27	12	91	9.92E+08	0.99	
28	12	110.6	1.21E+09	1.21	
29	12	34.8	3.79E+08	0.38	
30	12	26.8	2.92E+08	0.29	
31	12	45.5	4.96E+08	0.50	
32	12	37.5	4.09E+08	0.41	
33	12	47	5.12E+08	0.51	
34	12	134.8	1.47E+09	1.47	
35	12	122.1	1.33E+09	1.33	
36	12	278.1	3.04E+09	3.04	
37	12	13.9	1.51E+08	0.15	
38	12	4.4	4.79E+07	0.05	
39	12	8.7	9.48E+07	0.09	
40	12	54.3	5.92E+08	0.59	
41	12	1.9	2.07E+07	0.02	
42	12	76.4	8.32E+08	0.83	
43	12	127.9	1.39E+09	1.39	
44	12	71	7.74E+08	0.77	
45	12	233.3	2.54E+09	2.54	
46	12	102.1	1.11E+09	1.11	
47	12	3.6	3.92E+07	0.04	
48	12	90.7	9.88E+08	0.99	
49	12	9.6	1.05E+08	0.10	
50	12	6	6.54E+07	0.07	
51	12	50.5	5.50E+08	0.55	
TOTAL	654		2.60E+11	260.29	
Boxes 1-14	210		2.31E+11	231.47	
	Fraction in Box 1		0.78		
	in 1-14		0.89		