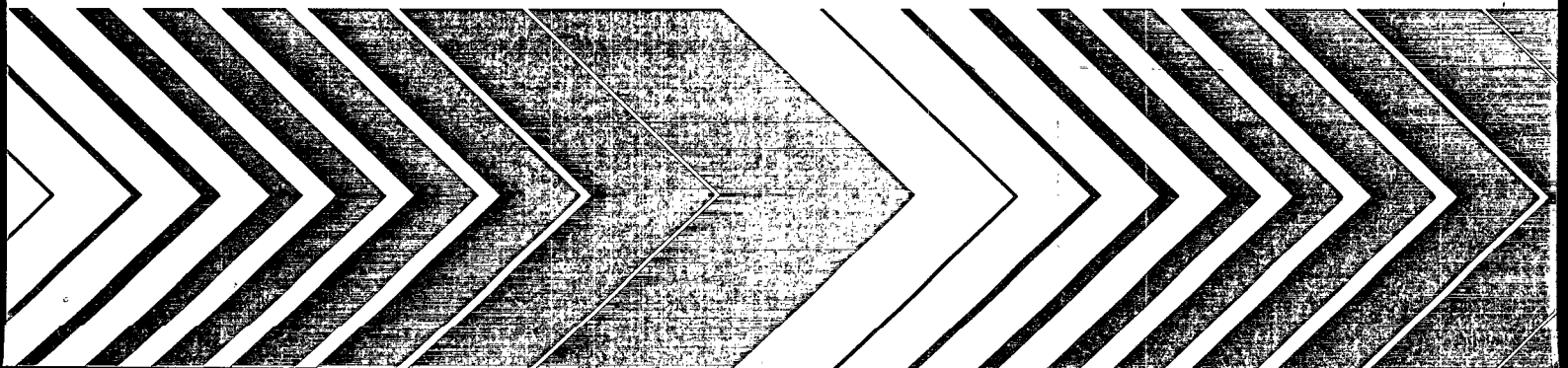
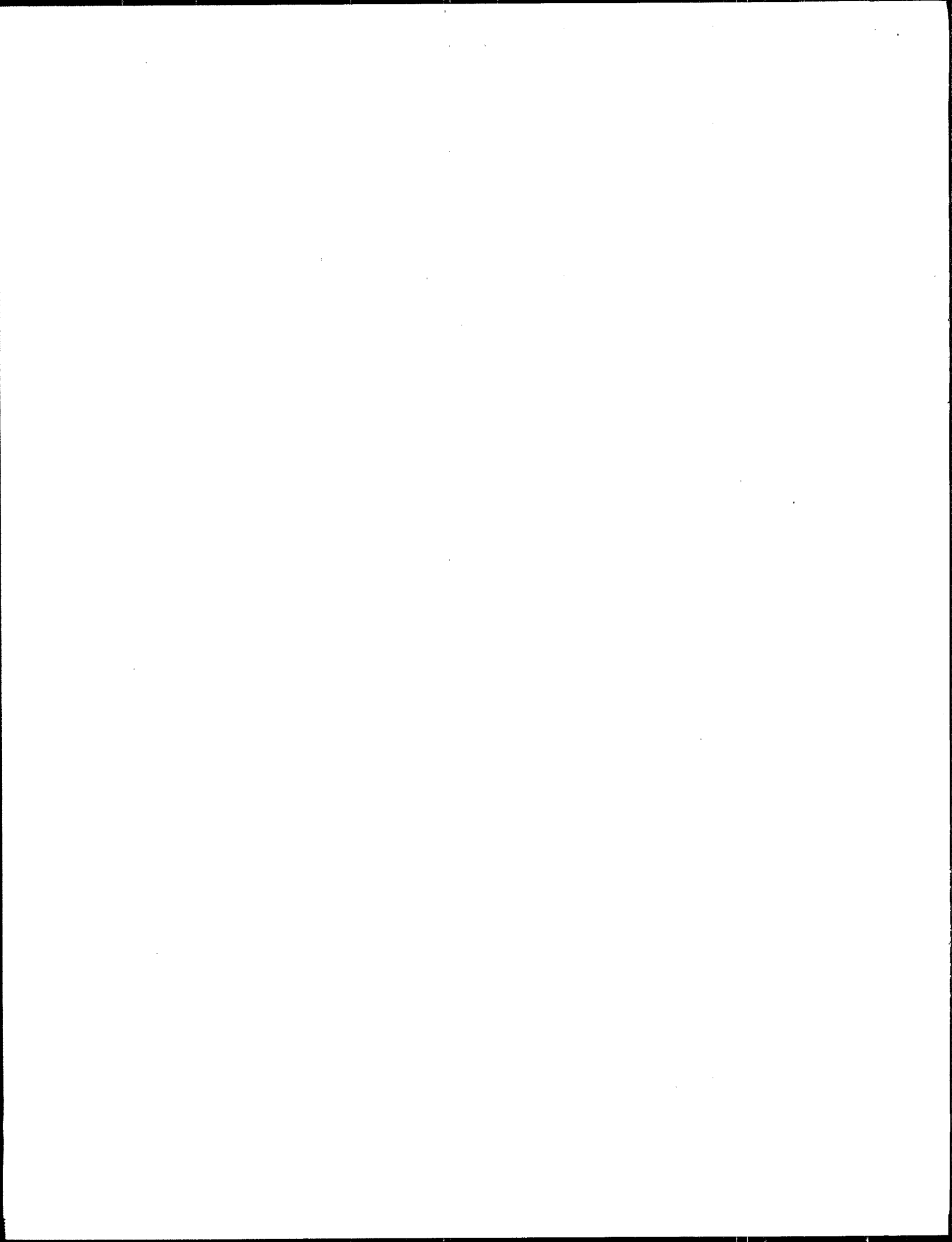




Methodology for Assessing Environmental Releases of and Exposure to Municipal Solid Waste Combustor Residuals





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April 1991
Final

**METHODOLOGY FOR ASSESSING ENVIRONMENTAL RELEASES OF
AND EXPOSURE TO MUNICIPAL SOLID
WASTE COMBUSTOR RESIDUALS**

Exposure Assessment Group
Office of Health and Environmental Assessment
U.S. Environmental Protection Agency
Washington, DC 20460



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FOREWORD

The Exposure Assessment Group (EAG) of EPA's Office of Health and Environmental Assessment (OHEA) has three main functions: (1) to conduct exposure assessments, (2) to review assessments and related documents, and (3) to develop guidelines for Agency exposure assessments. The activities under each of these functions are supported by and respond to the needs of the various EPA program offices. In relation to the third function, EAG sponsors projects aimed at developing or refining techniques used in exposure assessments.

The purpose of this document is to provide users with a methodology to assess the potential exposure to municipal solid waste (MSW) residuals, commonly known as ash. This document is designed to complement another OHEA document titled, *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (EPA, 1990b). The risk commonly associated with MSW combustors has been from direct exposure to combustor emissions. These two OHEA documents now allow for a more complete evaluation of risk from release of contaminants of MSW combustion. This document accomplishes the following: (1) summarizes existing information on MSW combustor design, types, and location of MSW facilities nationally; beneficial uses of ash; characteristics of ash; and contaminant concentrations of ash, (2) summarizes the management of MSW ash to identify points of environmental release from generation to disposal in a landfill, (3) provides methodologies to quantify these releases, and (4) directs the reader to other documents, particularly the companion document noted above, which detail fate and transport models, and exposure and risk algorithms. The document closes with an example of the methodologies applied to an organic contaminant, TCDD, and an inorganic contaminant, cadmium, both common in MSW combustor residuals.

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PREFACE

The Exposure Assessment Group (EAG) of the Office of Health and Environmental Assessment (OHEA) has prepared this document under the Research to Improve Health Research Assessment (RIHRA) program, Topic II: Integrated Exposure Assessment.

The purpose of this document is to provide users with a methodology to assess the potential exposure to municipal solid waste (MSW) residuals. MSW residuals, or ash, can be released to the environment from the point they are generated within the municipal waste combustor (MWC) facility to when they are disposed of in a landfill. This document identifies all such points of potential release and provides methods to quantify these releases. A comprehensive example is provided to demonstrate this methodology on an organic and an inorganic contaminant common in MSW ash.

This document does not describe or demonstrate methodologies to estimate the further transport of ash releases from the point of release to the point of the exposed individual, nor does it describe methodologies to quantify exposure and risk. However, it does provide overview guidance on these issues, and directs the user to appropriate materials.

AUTHORS AND REVIEWERS

The Exposure Assessment Group (EAG) within EPA's Office of Health and Environmental Assessment was responsible for the preparation of this document. The document was prepared by Technical Resources, Inc. under EPA Contract No. 68-02-4199, Work Assignment No. 21. EAG task managers included John Segna, Russell Kinerson, and Matthew Lorber. They provided overall direction and coordination as well as technical assistance and guidance.

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Matthew Lorber made changes to the text as a result of reviewers comments and otherwise. In particular, he made significant additions to Sections 5 and 7, including a more thorough discussion of AP-42 emission factor equations, the addition of two mechanisms of ash release (vehicular resuspension and drainage of moist ash), and the refinement and correction of all release estimates made in Section 7.

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1. INTRODUCTION

Incineration of municipal solid waste (MSW) is increasingly being used as a means of reducing the volume of waste that must ultimately be disposed in a landfill. However, the residuals that are generated by the combustion of such wastes must still be managed in some manner. In the past, risk assessments of MSW incineration have been concerned with human health and environmental impacts resulting from stack emissions. This document focuses on another fundamental issue, that of exposure to municipal waste combustion (MWC) residuals.

The objective of this document is to develop guidance for assessing exposure to MWC residuals or their chemical constituents that is consistent with methods that are already in use within the U.S. Environmental Protection Agency (EPA). Existing guidance includes methods for selection and use of media-specific fate and transport models, and methods for performing multi-media pathway-specific exposure assessments. To avoid redundancy with existing documents, this methodology will direct assessors to the appropriate EPA source materials and focus on issues germane to the assessment of risk posed by exposure to MWC residuals.

Another goal of this document is to complement the *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (EPA, 1990b). Therefore, this document not only provides guidance in defining and quantifying the sources of environmental release of MWC residuals, but also directs the user to the appropriate sections of the previous document that addressed MWC risk assessment methodology for completion of the assessment and evaluation of health risks.

The *Methodology for Assessing Environmental Releases of and Exposure to Municipal Solid Waste Combustor Residuals* is organized as follows. Section 2. is an overview of the incineration process. It describes some common facility types and the use of MWC in the United States. Section 3. reviews the subject of chemical and physical characterization of MWC residuals. It represents a summary of the state-of-the-science and recently completed efforts at characterizing ashes and leachates. Section 4. describes the MWC residuals management process and identifies potential sources of release during and following such management activities. This chapter enables the assessor to determine where releases are likely to occur during the residuals management process. Methods for determining the extent of these environmental releases are presented in Section 5. Section 6. refers the assessor to appropriate existing documents necessary to complete the exposure assessment process. Finally, Section 7. provides a demonstration of the methodology described in this document. An integral part of this demonstration is the step-by-step reiteration of the source analysis process. The output of an analysis of the type

demonstrated in Section 7. are release rates appropriate for use in subsequent fate and transfer assessment.

It should be emphasized that a number of uncertainties remain with regard to MWC residuals exposure/risk assessment. These are described in detail within the text and summarized in Section 7. Predictive methodologies for MWC residuals exposure/risk assessment are in their developmental infancy. Insufficient information exists, for example, to predict pollutant specific concentrations based on facility type or source composition. Limited information is available concerning the environmental impact if residuals are managed in a way other than disposal in a landfill. Subsequently, this document focuses only on that management option. Additionally, many of the predictive equations described in this document have not been adequately field tested. Such uncertainties should be considered when conducting an exposure/risk assessment for MWC residuals.

2. MUNICIPAL WASTE COMBUSTION IN THE UNITED STATES

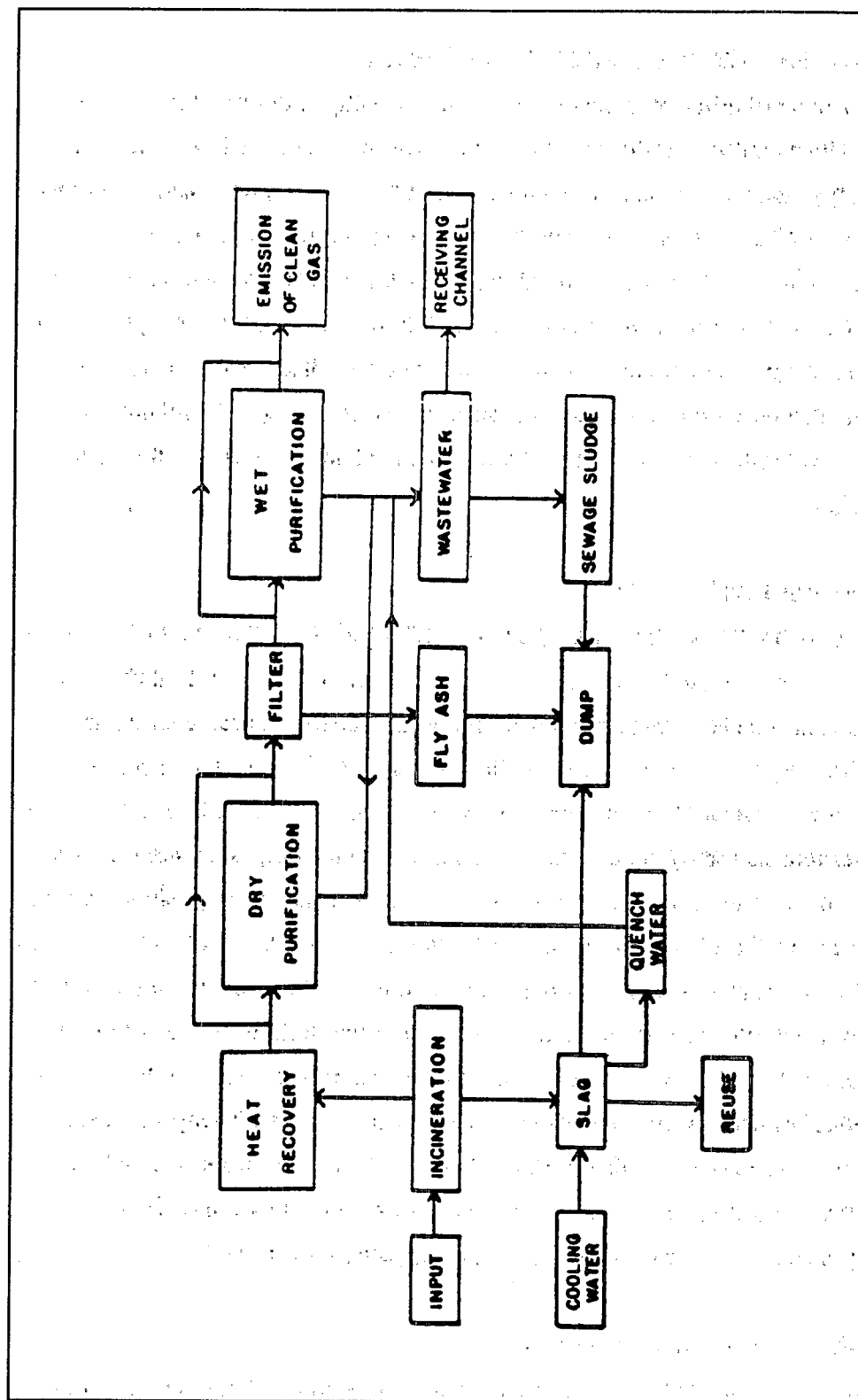
The goal of municipal solid waste (MSW) incineration is to reduce the volume and mass of material that ultimately needs to be disposed. During the process of incineration, the combustible portion of the solid waste is converted into gases and heat. The heat is often used for steam and electricity generation, while the gases are released to the atmosphere. The noncombustible or uncombusted fractions of the waste remain for disposal. These wastes are known as combustion residues or residuals. This section will describe the process of residual production, the types of facilities most often used, and the geographic distribution of these facilities across the United States.

2.1. MUNICIPAL SOLID WASTE RESIDUALS

The flow of materials within a typical municipal waste incineration facility is shown in Figure 2-1. The materials flowing out of a MSW incinerator are ashes, quench water, and gases. The ashes can be divided into two main categories: bottom ash and fly ash. Bottom ash consists of slags and cinders remaining in the combustion chamber after burning. These are generally noncombustible materials and materials with boiling points greater than the combustion temperature. The bottom ash is usually removed from the combustion chamber by a conveyor and then passed through a quench system to wet and cool the ash. Fly ash includes those particulates and fine particles that are collected from the stack and pollution control devices. The small particles that make up fly ash are noncombustible materials and may be carried by the combustion gases. Some of the products of incomplete combustion may be captured in the pollution control devices and some noncombustible materials with boiling points lower than the combustion temperature may also be entrained as volatilized vapors along with the combustion gases. As the combustion gases cool, these volatile constituents may condense and precipitate onto small particles in the stack, the stack itself, and pollution control devices within the stack.

The different effluents that result from MSW incineration may be managed in a number of ways. Bottom and fly ash may be mixed together or managed separately. Quench water may be completely or partially recycled within the facility, or discharged as effluent. The bottom and fly ashes make up the bulk of municipal waste combustion residuals, and their proper management represents a principal concern. The physical and chemical characteristics of these residuals will be discussed in detail in Section 3.

Figure 2-1
Flow Diagram of MSW Incineration



Source: Lisk, 1988

2.2. MUNICIPAL WASTE COMBUSTION FACILITIES

A number of different incinerator types are currently in operation in the United States. Common features of all combustion facilities include a mechanism to feed the MSW fuel, a combustion chamber, a heat (energy) recovery system, a pollution/emission control system, and mechanisms to collect and remove ash. Incineration facilities may be classified by the fuel type and combustor design (e.g., mass burn, processed refuse derived fuel, modular, starved air, dedicated stoker, co-firing, etc.), or by pollution control system (e.g., electrostatic precipitator, baghouse, scrubbers, sorbent injection). Each combination of fuel type, design, and pollution control system may potentially result in significant differences in the quantity and composition of the residual produced, and in consequent variations in exposure to specific constituents. Unfortunately, there exists a lack of research to statistically compare such characteristics by facility design.

2.2.1. Mass Burn Incinerators

Mass burn incinerators make up almost 90% of MWC facilities in operation in the United States (EPA, 1989a). A mass burn incinerator is so termed because it incinerates unprocessed municipal waste. The central component of a mass burn incinerator is the furnace. Newer units employ a waterwall furnace, while a refractory wall furnace is common in older designs. The furnace enclosure is positioned over the combustor grate. Older mass burn units introduce waste into the burn chamber using a gravity chute, while newer models use hydraulic rams. A number of grate designs are used in mass burn incinerators. All grates use one or a combination of forms of fuel agitation to produce uniform burning and to maximize the combustion of the waste. The grates are agitated in reciprocating, oscillary, or rotary motion. As the grates agitate, the waste is moved from the drying portion of the grate to the burn portion and finally toward the burnout grate for removal from the furnace. In mass burn units, combustion air is introduced below the grate and above the fire. The combustor air may be introduced in excess or it may be controlled (starved air combustors) to regulate the combustion temperatures and to ensure complete combustion. The noncombustible portion of the waste and the unburned carbon fall off the grate as bottom ash or are carried up by the flue gases as fly ash. Bottom ash usually drops off into a water filled bath for quenching. Fly ash is collected by the air pollution control devices present in the stack.

2.2.2. Refuse Derived Fuel Incinerators

Refuse derived fuel (RDF) combustion uses MSW that has been processed to some degree. RDF incinerators are less commonly used than mass burn incinerators in the United States (EPA, 1989a). RDF

has been classified according to the amount of processing it has undergone (Table 2-1). The advantages of using RDF rather than unprocessed MSW lie in its greater uniformity in composition and greater efficiency as a fuel. RDF combustors are generally able to achieve better combustion control than units using heterogenous wastes as fuels. In dedicated stoker boilers, fuel is injected into the furnace by air swept spouts. Traveling grates drop the bed ash into hoppers as they move towards the front of the boiler. Several levels of overfire air nozzles induce turbulence and provide the necessary mixing of partially combusted flue gas as it exits the grate bed.

Highly processed RDF (e.g., RDF 3, 4, or 5) is the least often used fuel for MWC facilities. However, when it is used, it is often co-fired with another fuel, such as coal, wood, or sewage sludge. Co-firing may be done in a spreader-stoker furnace or a utility steam generator. Fluff RDF (RDF-3) may be pneumatically injected into the furnace or stoker units.

2.2.3. Air Pollution Control Devices

Both mass burn and RDF incinerators use air pollution control devices in their stacks to capture fly ash as it is transported upward by the flue gases. Air pollution control devices used on MWC facilities include electrostatic precipitators (ESP), fabric filters, and scrubbers.

Electrostatic Precipitators

Electrostatic precipitators are efficient at removing particulate matter that is entrained with the gases. ESPs have been used alone, linked in series, or used with other pollution control devices such as scrubbers. ESPs function by inducing an electrical charge to the dust particles in the gas stream. The presence of an electric field in the gas space between the high voltage discharge electrodes and the collection plate propels the charge particulate matter towards the collection plate. The last step in the process involves the removal of the dust from the collection electrodes. A rapping device periodically hits the collection plates, and the dust is collected into a hopper. The fly ash is then conveyed to storage or disposal points.

Fabric Filters

In a fabric filter, dust particles impact onto a fabric as the gas is filtered and passes through the cloth and dust cake. The fabric used is typically woven or felt. The dust cake that forms on the filter plays a key role in improving the overall efficiency of the filter. The dust cake is periodically removed

Table 2-1. RDF Classification

RDF-1	MSW with minimal processing to remove bulky wastes.
RDF-2	Coarsely processed MSW, with or without ferrous metal separation such that 95% (by weight) passes through a 6 inch square mesh.
RDF-3	Shredded MSW with glass, metals and other inorganic materials removed. 95% by weight of the material passes through a 2 inch square mesh.
RDF-4	Powdered combustible fraction of MSW, such that 95% by weight of the material passes through a 10-mesh screen.
RDF-5	Combustible waste fraction compressed into pellets.
RDF-6	Combustible waste fraction processed into a liquid.
RDF-7	Combustible waste fraction processed into a gaseous fuel.

Source: Hickman, 1983

from the filter surface by shaking, reverse air cleaning, or blow-back of compressed air. Fabric filters are not usually used alone on MSW incinerators, but are normally downstream of lime injection scrubbers.

Scrubbers

Scrubbers come in three forms: wet, dry, and wet-dry. Wet scrubbers operate on the principle of intimate contact between a gas stream and a liquid that may contain absorbents or reagents for removal of acid gases. The main disadvantage of wet scrubbers is the generation of liquid waste effluent and a wet plume from the stack. Newer scrubbers are either of the dry or wet-dry configuration. In dry scrubbers, a powdered dry sorbent is typically injected into the gas stream. Intimate mixing of the sorbent with the gas occurs, then the dry gases are directed into a particulate removal device such as a fabric filter or an ESP. A dry scrubber may be preceded by a heat exchange or water-spray system to cool the gases. A wet-dry scrubber is also called a spray dryer or semi-dry scrubber. This kind of scrubber uses a liquid sorbent stream sprayed into the gas stream. The amount of liquid is controlled so all the liquid evaporates

into a gas stream, thereby yielding a dry fly ash product. These particulates are then removed by passing the gas stream through a baghouse or ESP.

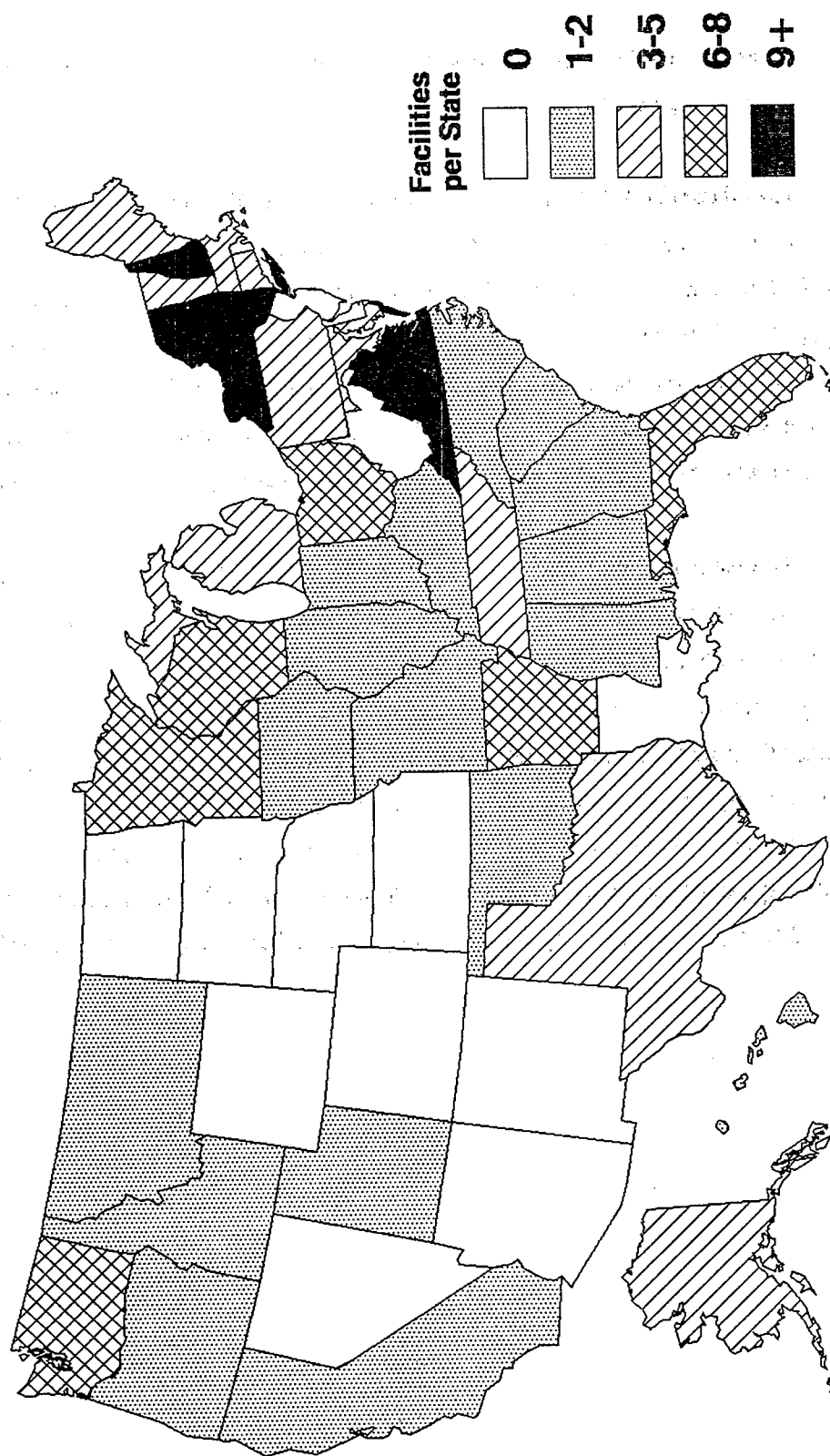
2.3. DISTRIBUTION AND CAPACITY OF MWC FACILITIES IN THE UNITED STATES

Approximately 140 to 155 MWC facilities were in operation in the United States in 1988 (EPA, 1989a; Ujihara and Gough, 1989). These facilities have an estimated total installed capacity of 78,700 tons per day (Ujihara and Gough, 1989). Currently, 14% of the municipal solid waste stream is incinerated. The greatest number of these facilities can be found in three states: New Hampshire, New York, and Virginia. These states each had more than 9 incinerators in operation in 1988 (Figure 2-2). The total number of incinerators in the United States has been projected to increase to 227 by 1992 (Levy, 1989).

The results of a survey conducted by the U.S. EPA Office of Solid Waste (EPA, 1989a) indicate that over 90% of the MWC facilities in the United States use some form of mass burn combustor. Most facilities operate 7-days-a-week, 24-hours-a-day. The average amount of municipal solid waste combusted in an MWC facility in the United States is about 9,355 tons per month on an annual basis, with facilities receiving 8% more waste in the summer than in the winter months. The typical mix of waste combusted is approximately 68% residential, 23% commercial and 9% industrial.

Municipal waste combustors in the United States produce about 2.8 to 5.5 million tons of ash per year, with fly ash comprising from 5-15% of the total. The total amount of ash produced could potentially increase 2 to 5 times, depending on how many new facilities are constructed (OTA, 1989).

Figure 2-2
Total Municipal Waste Combustor
Facilities by State



Source: EPA, 1989a

3. CHARACTERIZATION OF MUNICIPAL WASTE COMBUSTION ASH

Fly ash, bottom ash, and quench water make up the major waste streams exiting a MWC facility. Possible exposures to the chemicals contained in these ashes, and to the leachates that are generated following disposal, account for the major health and environmental risks posed by these residuals. An understanding of the chemical and physical characteristics of municipal incinerator residuals is therefore the first step in defining the potential exposures and risks. Information regarding the chemical and physical characteristics of MWC ashes and leachates can be found in:

- EPA, 1987a; 1988a. Characterization of Municipal Waste Combustor Ashes and Leachates from Municipal Solid Waste Landfills, and Co-Disposal Sites. Volumes I to VII and Addendum. Prepared by NUS Corporation for the EPA Office of Solid Waste (OSW).
- EPA, 1990a. Characterization of Municipal Waste Combustion Ash, Ash Extracts, and Leachates. Coalition on Resource Recovery and the Environment (CORRE) and the EPA Office of Solid Waste and Emergency Response (OSWER).

This section provides a general overview of the current information on ash and leachate characterization. The assessor is directed to the above documents for more detailed information.

3.1. FACTORS AFFECTING THE CHEMICAL COMPOSITION AND CONCENTRATION OF RESIDUALS

The concentrations of constituents found in the ashes, and their concentration in ash leachates, may be affected by a number of factors, including:

- **Sources of municipal solid waste:** A 1988 industry survey (EPA, 1989a) found that the average MWC facility incinerates waste from a combination of sources. At a typical facility, the primary waste stream is from residential sources (68.3%); however commercial sources contribute 23.1%. Industrial wastes contributed the smallest percentage to the MWC facility's waste stream (< 9%). Differences in ash characteristics might be explained in part by the proportion of the total combustible waste originating from each source.

- **Operating conditions:** Differences in operating conditions and combustion practices might also explain differences in ash characteristics. Fly ash from similar incinerators located in different countries (Norway and Canada) has been shown to contain different concentrations of organic contaminants, including polychlorinated dibenzo-*p*-dioxins (PCDDs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and polychlorinated dibenzofurans (PCDFs) (Viau et al., 1984).

- **Incinerator and pollution control equipment characteristics:** As described in Section 2.0, the specific incinerator design can greatly affect the quality of the residuals that are produced. Residuals vary with incinerator operating parameters and efficiency, and are dictated by the burn temperature, mixing of air with combustion gases, grate design, turbulence, and fuel characteristics. Residuals, especially fly ash, can also vary with the pollution control equipment used. The use of catalysts or adsorbents, such as lime, can also significantly affect the particle size distribution and the leachability of different components of the ash.

- **Type of residual (e.g., quench water, fly ash, bottom ash, and combined ash):** The exposure and risk characteristics of MWC residuals differ with residue type. The majority of MWC residuals are managed as combined bottom and fly ash. Approximately 72% of the MWC (disposal) facilities receive combined ash. The remaining 28% of disposal facilities receive only bottom ash or only fly ash (EPA, 1988c). The differences in the organic and inorganic chemical makeup of fly ash and bottom ash are further discussed in Sections 3.3. and 3.4.

- **Pretreatment of MSW:** Pretreatment of the MSW stream may affect the composition and physical characteristics of the residuals. A survey of industry practice (EPA, 1989a) found that 61.1% of the facilities removed some combustible materials and 86.4% removed some non-combustibles from the waste prior to incineration. The wastes that are removed from the waste stream may be diverted to recycling facilities. Removal of metallic waste and batteries may decrease the levels of cadmium, mercury, lead, and other metals from the ash.

3.2. QUALITY OF REPORTED DATA ON MWC CONSTITUENT CONCENTRATIONS

The ranges of concentrations of ash constituents that have been reported by various authors are often seen to vary by several orders of magnitude. This variation has been attributed to differences in sampling, analytical, and quality assurance/quality control protocols (Clement et al., 1984). For example, Van der Sloot et al. (1989) compared the composition of bottom ash and fly ash from an incinerator in the Netherlands with values reported in the literature for incinerators in Canada, Denmark, and other Dutch incinerators. Although there appeared to be general agreement between the elemental composition of ash from facilities in different countries, there was a wide range of variation in the results previously reported in the literature when compared to the authors' recently acquired data. The authors suggest that these differences in the magnitude of data variability between older and recent data may be the result of improved sampling and analytical techniques and due to more efficient, pretreatment of waste in the recent studies. Since many of the analytical techniques used to analyze MWC ash have not been standardized, it is not unusual to find differences in the data originating from different researchers. In addition, many researchers report data from leaching studies as compositional data, when, in fact, the techniques are designed to predict only the concentrations of ash constituents in landfill leachates.

Given the variability of the data reported in the literature, the prediction of potential concentrations

of chemical constituents of ash based on the factors described above is not generally possible. Ideally, it would be advantageous to be able to predict the concentrations of constituents in ash from those factors which effect ash composition, so that knowledge of facility type and location, combustion parameters, and fuel source would be sufficient to enable prediction of the concentrations of constituents of concern in ash. Methods for determining these relationships have been documented (Hasselriis, 1989). However, the state-of-the-science is such that predictions of this kind are not yet possible. There are two major causes for this limitation: the extreme heterogeneity of ash and deficiencies in the literature database.

Many factors affect ash composition; the result is that ash is a very heterogeneous substance (Ujihara and Gough, 1989). Therefore, the acquisition of a representative sample of ash is a very difficult task. Although EPA has recommended the use of standard sampling and Quality Assurance/Quality Control (QA/QC) protocols for MWC residuals (EPA, 1990a), the literature contains little or no consistency in the way ash samples are collected, handled, or prepared for analysis.

The heterogeneous nature of ash is reflected in the broad ranges of concentrations for constituents in ash that are reported in the literature. For example, Mika and Feder (1985) took 96 samples of bottom ash from a mass burn incinerator over a single 48-hour period. Using the EPA Extraction Procedure, they found the concentration range for lead to be from 0.08-15.08 mg/L, and for cadmium to be from below detection limits to 1.59 mg/L. A second series of 48 samples taken over another 48-hour period showed concentrations of lead to be from 0.04 to 33.42 mg/L, and cadmium to be from 0.01 to 0.53 mg/L. This variability is typical of that found in the literature. Tables 3-1 to 3-10 summarize the available data on the composition of ash from MSW combustors. The wide range of values reported in these tables for ash constituents underscores the heterogeneity of MWC ash.

The second major reason for the inability to predict ash constituent concentrations for facility parameters is the lack of a sufficiently strong database. Table 3-11 was compiled to provide a brief overview of the kind of research that has been conducted on MWC ash, and the quality and usefulness of the data that has been published. This literature was reviewed with the goal of developing a representative data set of ash constituent concentrations that could be used in developing general parameters for exposure assessments. However, it was quickly discovered that data from the literature are often of little help in establishing the interactive effects of the various factors affecting composition because of incomplete data on MSW fuel characteristics, MWC facility design and operating parameters, representativeness of ash samples, and accuracy/precision of analytical methods employed.

Table 3-1. Ranges of Concentrations of Some Inorganic Constituents of Fly, Bottom, and Combined Ash from Municipal Waste Combustors (concentration in µg/g or ppm)

Chemical	Fly Ash	Combined Ash	Bottom Ash
Aluminum	5,300-176,000	5,000-60,000	5,400-180,000
Antimony	4.4-760	20-<260	1-600
Arsenic	1-750	2.9-<1,200	1-80
Barium	80-9,000	79-2,700	40-2,000
Beryllium	ND-<4	ND-2.4	ND-<0.44
Boron	5-5,654	24-174	85
Bromine	21-250	NR	NR
Cadmium	0.3-2,100	0.18-100	0.2-442
Calcium	3,000-290,000	4,100-85,000	5,900-112,000
Carbon	17,000-74,000	NR	10,000-287,000
Cesium	2,100-12,000	NR	NR
Chlorine	1,160-253,000	NR	2,000-10,000
Chromium	110-13,000	12-1,500	13-10,000
Cobalt	2.3-5,000	1.7-91	3-62
Copper	69-3,100	40-28,460	80-26,000
Fluorine	1,500-3,100	NR	130-300
Iron	900-87,000	690-133,500	1,000-320,000
Lead	6-26,600	31-36,600	0.2-17,000
Lithium	7.9-34	6.9-37	7-19
Magnesium	2,000-40,000	700-16,000	400-18,000
Manganese	65-8,500	14-3,130	50-390,000
Mercury	ND-40	<0.020-1.5	ND-3.5
Molybdenum	9.2-700	2.4-290	2-500
Nickel	9.9-1,966	13-12,910	9-1,300
Nitrogen	ND	NR	ND-3,500
Phosphorus	1,000-12,000	290-5,000	400-17,800
Potassium	4,300-74,800	290-12,000	920-24,100
Selenium	0.48-16	0.10-<780	ND-7
Silicon	1,783-320,000	47,000-120,000	1,333-460,000
Silver	ND-77,500	0.05-93.4	ND-38
Sodium	477-80,000	1,100-33,300	1,800-69,000
Strontium	98-1,100	12-640	81-1,000
Sulfur	4,000-40,000	NR	1,750-20,000
Thorium	NR	NR	70-300
Tin	<100-12,500	13-1,000	40-1,300
Titanium	<50-42,000	1,000-28,000	400-11,400
Vanadium	22-298	13-150	36-90
Zinc	120-152,000	92-46,000	200-36,100

ND = not detected at the detection limit

NR = not reported

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hrudey et al., 1974; Giordano, et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-2. Ranges of Concentrations of Some Organic Constituents of Fly and Bottom Ash from Municipal Waste Incinerators (concentration in ng/g or ppb)

Constituent	Fly Ash	Bottom Ash
Acenaphthene	NR	28
Acenaphthylene	ND-3,500	37-390
Anthanthrene	NR	305
Anthracene	1-500	53
Benzanthrene	ND-300	NR
Benzo(k)fluoranthene	ND-470	ND-180
Benzo(a)pyrene	ND-400	ND-5
Benzo(g,h,i)perylene	ND-190	ND
Biphenyl	2-1,300	NR
Bis(2-ethylhexyl)phthalate	852,100	NR
Butyl benzyl phthalate	ND	180
Chlorobenzenes	80-4,220	17
Chlorophenols	50.1-9,630	ND
Chrysene	ND-690	ND-37
1,1-Dichloro-3-phenylpropane	80.2	NR
Di-n-butyl phthalate	ND	360-1500
Diethyl phthalate	6,300	105.2
Fluoranthene	ND-6,500	4.5-230
Fluorene	ND-115	ND-150
M. cyclopentaindene ^a	60	NR
Naphthalene	270-9,300	570-580
Normal alkanes	3,647-50,000	NR
Thio-PAHs ^b	50-75	NR
Triptycene	NR	85.0
Phenanthrene	21-7,600	500-540
3-Methyl phenanthrene	NR	2.5
Pyrene	ND-5,400	31.3-220

ND = not detected at the detection limit

NR = not reported

^a Methyl-diphenylhexahydrocyclopentaindene

^b Alkyl-substituted dibenzothiophene = 50 ng/g

2,5-Bis(p-Chlorophenyl)1,4-diithin = 75 ng/g

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Vian et al., 1984; Norheim et al., 1989; Hrudef et al., 1974; Giordano, et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-3. Ranges of Concentrations of PCDDs, PCDFs, and PCBs in Municipal Waste Combustor Ash (concentration in ng/g or ppb)

Constituent	Fly Ash	Combined Ash	Bottom Ash
MCDD	2.0	ND	NR
DCDD	0.4-200	ND-120	NR
T ₃ CDD	1.1-82	ND-33	NR
T ₄ CDD	ND-250	0.14-14	<0.04-410
PCDD	ND-722	0.07-50	ND-800
H ₆ CDD	ND-5,565	0.07-78	ND-1,000
H ₇ CDD	ND-3,030	0.07-120	ND-290
OCDD	ND-3,152	0.07-89	ND-55
2,3,7,8-TCDD	ND-330	0.02-0.78	<0.04-6.7
Total PCDD	5-10,883	6.2-350	ND-2,800
MCDF	41	1.1	NR
DCDF	ND-90	ND-42	NR
T ₃ CDF	0.7-550	ND-14	NR
T ₄ CDF	ND-410	2.3-9	10.1-350
PCDF	ND-1800	1.6-37	0.07-430
H ₆ CDF	Tr-2,353	1.2-35	ND-920
H ₇ CDF	Tr-887	0.62-36	ND-210
OCDF	ND-398	0.18-8.4	ND-11
2,3,7,8-TCDF	0.05-5.4	0.41-12	ND-13
Total PCDF	3.73-2,396	6.14-153.9	ND-1,600
Mono CB	0.29-9.5	ND	ND-1.3
Di CB	0.13-9.9	0.126-1.35	ND-5.5
Tri CB	ND-110	0.35-14.3	ND-80
Tetra CB	0.5-140	16.5	ND-47
Penta CB	0.87-225	ND	ND-48
Hexa CB	0.45-65	ND-39	NR
Hepta CB	ND-0.1	ND	NR
Octa CB	ND-1.2	ND	NR
Nona CB	ND	ND	NR
Deca CB	ND	ND	NR
Total PCB	ND-360	ND-32.15	ND-180

ND = not detected at the detection limit

NR = not reported

Tr = 0.01<Tr<0.1 ng/g

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Norheim et al., 1989; Hrudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-4. Ranges of Extract Concentrations of Some Inorganic Constituents from Municipal Waste Combustor Fly Ash (concentration in mg/L or ppm)

Chemical	SW-924 ^a	TCLP ^b	EP ^c
Aluminum	0.09-0.805	0.09-16.0	0.159-18.8
Arsenic	ND-0.14	ND-0.111	0.002-0.23
Antimony	0.080	0.154	NR
Barium	0.19-1.68	<0.1-1.86	<0.02-22.8
Beryllium	ND-0.01	ND-0.01	0.001-0.005
Boron	ND-2,100	1.36-7.3	0.7-8
Cadmium	ND-33	0.015-120	<0.005-120
Calcium	646-4,620	1,250-5,390	1,150-5,810
Chlorine	133-18,500	NR	NR
Chromium	0.0025-176	0.0025-0.544	0.0025-<0.20
Cobalt	ND-0.12	0.03-0.14	0.025-0.114
Copper	0.0025-1,240	0.0025-14.70	0.033-10.6
Iron	0.0025-12	0.0025-190	0.0025-38
Lead	ND-150,000	0.025-65	0.019-65.0
Lithium	0.27-0.38	0.25-0.55	0.261-0.455
Magnesium	0.03-37.6	0.04-171	0.093-149
Manganese	0.0005-0.052	0.01-14.7	0.005-61
Mercury	0.00002-0.02	<0.0002-0.004	ND-0.007
Molybdenum	0.22-0.34	0.10-0.31	0.10-0.229
Nickel	ND-420	0.0075-2.48	0.09-11
Potassium	19-2,530	574-2,780	616-2,170
Selenium	0.0025-0.108	0.002-0.1	0.003-0.62
Silver	0.02-0.05	<0.01-0.08	0.001-0.57
Sodium	16-971	474-2,500	506-821
Strontium	2.6-17.7	3.4-17.30	3.5-16
Sulfate	80-10,400	NR	NR
Tin	0.09	0.09	0.09
Titanium	0.05	0.05	0.05
Vanadium	0.004-0.02	0.02-0.464	0.015
Yttrium	0.05	0.05	0.05
Zinc	0.0015-2,000	0.151-885	3.36-768
TDS	484-10,900	NR	NR

TDS = total dissolved solids

ND = not detected at the detection limit

NR = not report

^a Monofilled Water Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hruday et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-5. Ranges of Extract Concentrations of Some Inorganic Constituents of Bottom Ash from Municipal Waste Combustors (concentration in mg/L or ppm)

Constituent	SW-924 ^a	TCLP ^b	EP ^c
Arsenic	NR	<0.02	<0.007-0.13
Barium	NR	<0.5-0.74	<0.02-<0.20
Cadmium	0.001-0.05	<0.01-0.034	<0.005-1.1
Chromium	<0.01	<0.05	0.011-<0.2
Cobalt	<0.02	NR	NR
Copper	0.04	NR	NR
Lead	0.01-1	0.067-6.4	<0.01-32
Mercury	ND	<0.0002	<0.0002-0.04
Nickel	<0.05	NR	NR
Selenium	NR	<0.02	<0.01-0.16
Silver	NR	<0.01	0.005-0.05
Zinc	0.1-5	NR	NR

ND = not detected at the detection limit

NR = not reported

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hrudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-6. Ranges of Extract Concentrations of Some Inorganic Constituents of Combined Fly and Bottom Ash from Municipal Waste Combustors (concentration in mg/L or ppm)

Parameter	SW-924 ^a	TCLP ^b	EP ^c
Aluminum	0.17-29.4	0.01-0.05	31.9-43.8
Arsenic	<0.001-0.54	0.005-0.10	ND-<3.8
Barium	0.15-2.83	<0.01-3.2	<0.002-11.8
Beryllium	ND-0.01	0.01	0.001-0.005
Boron	ND-0.95	1.75-2.89	0.56-4.7
Cadmium	ND-0.03	0.01-3.32	0.002-5.3
Calcium	122-536	362-1,990	77-1,740
Chlorides	209-960	302-625	78-952
Chromium	0.0025-<0.05	<0.003-0.439	<0.003-0.8
Cobalt	0.01-0.03	0.03-0.05	0.050-0.057
Copper	0.0025-0.19	0.0025-0.09	0.039-24.3
Iron	ND-0.09	0.828-230	0.8-143
Lead	0.021-2.98	<0.02-47	0.02-34
Lithium	0.01-0.05	0.06-0.10	0.063-0.093
Magnesium	ND-0.19	0.14-41.8	22.8-42.7
Manganese	ND-0.03	4.2-11.9	2.1-47
Mercury	ND-0.1	ND-0.10	ND-6.0
Molybdenum	0.07-0.1	0.10	0.10
Nickel	0.0075-0.09	0.09-0.805	0.18-2.9
Potassium	85.2-120	86.5-111	10-154
Selenium	0.002-0.07	<0.001-<0.1	ND-<0.5
Silver	ND-0.05	<0.01-0.05	<0.001-<0.11
Sodium	68.3-85.3	103-1,500	89.9-100
Strontium	0.58-3.19	3.94-5.47	2.45-4.9
Sulfate	156-571	260-1,450	85-1,150
Tin	0.02-0.09	0.09	0.09
Titanium	0.01-0.05	0.05-0.08	0.05
Vanadium	0.02-0.03	0.02-0.06	0.015
Yttrium	0.01-0.05	0.05	0.05
Zinc	0.0015-0.96	23.3-373	10-726

ND = not detected at the detection limit

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-7. Ranges of Extract Concentrations of Some Organic Constituents of Fly Ash from Municipal Waste Incinerators (concentration in ng/g or ppb)

Constituent	SW-924 ^a	TCLP ^b	EP ^c
Chlorobenzenes	ND	NR	NR
Chlorophenols	ND-675	NR	NR
Dimethyl Propdiol	ND	ND	ND
Naphthalene	ND	ND	ND
Methoxy Ethanol	ND	ND-10	ND
Methyl Naphthalene	ND	ND	ND
Methoxy Ethane	ND	ND-10	ND

NR = not reported

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-8. Ranges of Extract Concentrations of Some Organic Constituents of Bottom Ash from Municipal Waste Incinerators (concentration in ng/g or ppb)

Constituent	SW-924 ^a	TCLP ^b	EP ^c
Benzoic Acid	ND-46	ND	ND
Bis oxy Ethanol ^d	ND	ND	ND
Cycloocta Decone ^e	ND-150	ND	ND
Dimethyl Propdiol ^f	ND	NDND	
Ethoxy Ethanol ^g	ND	ND	ND
E. Dim Dioxane ^h	ND	ND	ND
M. Furan Dione ⁱ	ND-6	ND	ND
Methyl Naphthalene	ND	NDND	
Methoxy Ethane ^j	ND-10	ND-22	ND-12
Naphthalene	ND	ND	ND
Oleyl Alcohol ^k	ND	ND	ND
Phenol	ND-28	ND	ND

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

^d 2,2-[1,2-Ethanediyl]bis(oxy)bis-ethanol (CAS 112-27-6)

^e 1,4,7,10,13,16-Hexaoxacyclooctadecane (CAS 17455-13-8)

^f 2,2-Dimethyl-1,3-propanediol (CAS 126-30-7)

^g 2-[2-(Ethenyloxy)ethoxy]-ethanol (CAS 929-37-3)

^h 5-Ethyl-2,2-dimethyl-1,3-dioxane (CAS 25796-26-3)

ⁱ 3,4-Dimethyl-2-5-furandione (CAS 766-39-2)

^j 1-Methoxy-2-(methoxy methoxy)ethane (CAS 74498-88-7)

^k (2)-9-Octadecen-1-Ol (CAS 143-28-2)

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hrudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-9. Ranges of Extract Concentrations of Some Organic Constituents of Combined Fly and Bottom Ash from Municipal Waste Incinerators (concentration in ng/g or ppb)

Constituent	SW-924 ^a	TCLP ^b	EP ^c
Bis oxy Ethanol ^d	ND-96	ND	ND
Cycloocta Decone ^e	ND-1,200	ND	ND
Dimethyl Propdiol ^f	ND-160	ND-140	ND-190
E. Dim Dioxane ^g	ND-510	ND	ND
Ethoxy Ethanol ^h	ND-390	ND	ND
M. Furan Dione ⁱ	ND	ND	ND
Methoxy Ethanol	ND	ND	ND
Methyl Naphthalene	ND-80	ND	ND-18
Methoxy Ethane ^j	ND	ND	ND
Naphthalene	ND	ND	ND-8
Oleyl Alcohol ^k	ND-88	ND	ND-18
Phenol	ND-33	ND	ND

ND = not detected at detection limit

CAS = Chemical Abstract Services

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

^c Extraction Procedure

^d 2,2-[1,2-Ethanediyllbis(oxy)bis-ethanol (CAS 112-27-6)

^e 1,4,7,10,13,16-Hexaoxycyclooctadecane (CAS 17455-13-8)

^f 2,2-Dimethyl-1,3-propanediol (CAS 126-30-7)

^g 2-[2-(Ethenyloxy)ethoxy]-ethanol (CAS 929-37-3)

^h 5-Ethyl-2,2-dimethyl-1,3-dioxane (CAS 25796-26-3)

ⁱ 3,4-Dimethyl-2-5-furandione (CAS 766-39-2)

^j 1-Methoxy-2-(methoxy methoxy)ethane (CAS 74498-88-7)

^k (2)-9-Octadecen-1-ol (CAS 143-28-2)

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hruday et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-10. Ranges of Extract Concentrations of PCDDs, PCDFs, and PCBs from Fly and Combined Ash from Municipal Waste Incinerators (concentration in ng/L or ppt)

Constituent	Fly Ash SW-924 ^a	Fly Ash TCLP ^b	Combined Ash TCLP ^b
TCDD	NR	<0.056-<0.094	<0.038-<0.023
PCDD	NR	<0.040-<0.056	<0.023-<0.067
HCDD	NR	<0.019-<0.027	<0.015-<0.044
H ₇ CDD	NR	<0.038-0.11	<0.028-<0.120
OCDD	NR	<0.078-0.11	<0.035-<0.091
2,3,7,8-TCDD	NR	<0.056-<0.094	<0.038-<0.230
Total PCDD	ND	ND-0.188	ND-<0.091
TCDF	NR	<0.048-<0.120	<0.031-<0.200
PCDF	NR	<0.016-<0.026	<0.013-<0.042
HCDF	NR	<0.013-<0.020	<0.008-<0.025
H ₇ CDF	NR	<0.020-<0.063	<0.013-<0.043
OCDF	NR	<0.015-<0.089	<0.060-<0.054
2,3,7,8-TCDF	NR	<0.048-<0.012	<0.031-<0.200
Total PCDF	ND	ND-0.152	ND-0.054
Total PCB	ND		

ND = not detected at detection limits

NR = not reported

^a Monofilled Waste Extraction Procedure

^b Toxicity Characteristic Leaching Procedure

Sources: EPA, 1987a; EPA, 1988a; EPA, 1990a; Clement et al., 1984; Viau et al., 1984; Northeim et al., 1989; Hrudey et al., 1974; Giordano et al., 1983; Lisk, 1988; Belevi and Baccini, 1989; Van der Sloot et al., 1989; DiPietro et al., 1989; Kullberg and Fallman, 1989; Austin and Newland, 1985; Kuehl et al., 1985; Kuehl et al., 1986; Kuehl et al., 1987; Buser et al., 1978; Tanaka and Takeshita, 1987.

Table 3-11. Summary of Literature Database Quality

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^e	Analysis
Giordano et al. (1983)	Study mobility, plant availability, and attenuation of Pb and Cd from ash	Not reported	Reported only that ash received from 8 facilities in U.S. and Canada	Not reported	Methodology reported or referenced; some tests done in triplicate; • QA/QC unclear
Belevi and Baccini (1989)	Analyze bottom ash from combustor in Switzerland with respect to long-term leaching	Not reported	Not reported	Sample preparation reported; ash sampling protocol and number of samples not reported	Extraction procedure reported; analytical procedure and QA/QC not reported
Chesner (1990)	Characterize and compare ash with natural aggregate material	Not reported	Reported	Referenced	Methodology and QA/QC reported or referenced
DiPietro et al. (1989)	Evaluate effects of pH and redox potential on metal leaching	Not reported	Reported in detail	Sample preparation reported; sampling protocol not reported	Methodology and QA/QC reported
Eighmy et al. (1990)	Study factors affecting heavy metal leachability	Not reported	Reported	Sampling method and sample types reported, number of samples not reported	Methodology reported; QA/QC not reported
EPA (1990a)	Enhance the database on MWC ash characteristics, laboratory extracts of ash, and leachates from ash disposal facilities	Not reported	Reported in detail for 5 different facilities	Reported	Methodology and QA/QC reported or referenced
Bagchi and Sopcich (1989)	Characterize ash	Not reported	Reported	Sampling protocol, number of samples, and sample preparation reported	Methodology reported or referenced; QA/QC not reported
Holland et al. (1989)	Compare physical and leaching properties of stabilized ash	Table taken from 1988 reference that is not listed in reference section of paper	Reported	Ash sampling location, duration, and sample preparation reported; ash sampling protocol and number of samples not reported	Methodology and QA/QC reported or referenced

Table 3.11. Summary of Literature Database Quality (continued)

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^d	Analysis ^e
Kullberg and Fallman (1989)	Investigate physical, chemical, and technical properties of flue gas cleaning residues from Swedish facilities	Not reported	Not reported	Not reported	Methodology reported or referenced; QA/QC not reported
Mintott (1989)	Describe operating principles and environmental parameters of fluidized bed combustors	Not reported	Reported	Not reported	Analyses conducted by EPA-certified laboratory; methodology and QA/QC not reported
Mohamad et al. (1988)	Characterize coal-RDF fly ash	Reported	Name and location of facility only reported	Sampling protocol, number of samples, and sample preparation reported	Methodology and QA/QC reported
Norheim et al. (1989)	Evaluate laboratory leaching tests	Not reported	Not reported	Sample preparation reported; 2 ash sampling protocol referenced analysis	Methodology and QA/QC reported
Sawell and Constable (1989)	Evaluate ash streams from a RDF and a modular facility in Canada	Not reported	Reported or referenced	Sampling protocol, number of samples, and sample preparation reported	Methodology reported or referenced; QA/QC not reported
Van der Sloot et al. (1989)	Examine metal leaching behavior of ash from combustor in The Netherlands	Not reported	Facility name, location, capacity, and ash handling reported; operating parameters not reported	Ash sample preparation reported; ash sampling protocol and number of samples not reported	Methodology reported or referenced; some procedures performed in duplicate, QA/QC for others not reported
Clement et al. (1988)	Determine if combustion conditions could affect formation of PCDDs and PCDFs	Highly industrialized area	Reported	Reported	Methodology and QA/QC reported
Bleifuss et al. (1988)	Evaluate EP toxicity and ASTM water leach tests for fly ash	Not reported	Name and location of combustor only reported	Composite test lots of ash from 3 facilities prepared and distributed to participating laboratories; sample preparation reported	Methodology referenced; QA/QC varied between laboratories

Table 3-11. Summary of Literature Database Quality (continued)

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^d	Analysis ^e
EPA (1988a)	Characterize MWC ash and leachates from disposal facilities; summarizes data reported in literature	Not reported	Not reported	Not reported	Not reported
Kamada et al. (1988)	Measure amount of PCDDs in flue gas and ash from 7 Japanese incinerators	Not reported	Reported in detail	Reported in detail	Reported in detail
Pherson (1988)	Study ability of solidification to fixate Pb in ash	Not reported	Not reported	One 55-gal drum of mixed ash, sampling protocol not reported	Methodology and QA/QC reported
Schwind et al. (1988)	Analyze fly ash for PB/CDDs and PB/CDFs	Not reported	Not reported	Single ash sample used for analysis; sampling protocol not reported; sample preparation reported	Methodology and some QA/QC reported
Sovocool et al. (1988)	Isolate and identify brominated organics in fly ash	Not reported	Not reported	Sampling protocol and number of samples not reported; sample handling and preparation reported	Methodology and some QA/QC referenced or reported
EPA (1987a)	Characterize ash and leachate	Not reported	Reported	Reported or referenced	Methodology and QA/QC reported or referenced
Francis and White (1987)	Study leaching of toxic metals	Not reported	Location of facilities only reported	Referenced	Methodology reported or referenced
Karasek and Dickson (1987)	Study of dioxin formation from precursors in incinerators from Canada and Japan	Not reported	Location and combustor type only reported	Sampling protocol and sample number not reported; sample preparation reported	Methodology and QA/QC reported
Karasek et al. (1987)	Study leaching of organics from Canadian fly ash	Not reported	Location only reported	Single sample from ESP	Methodology and QA/QC reported

Table 3-11. Summary of Literature Database Quality (continued)

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^d	Analysis ^e
Kuehl et al. (1987)	Determine isomer dependent bioavailability of PCDDs and PCDFs in fly ash to freshwater fish	Not reported	Location of combustor only reported	Sampling protocol and number of samples not reported; sample handling and preparation reported	Methodology and QA/QC reported or referenced
Tanaka and Takeshita (1987)	Summary of research projects and literature on TCDDs and PCDDs in fly ash	Not reported	Combustor type only reported	Sampling protocol and number of samples not reported; sample preparation reported	Methodology briefly reported; QA/QC not reported
Carsch et al. (1986)	Study leaching behavior of organics in fly ash from incinerator in Germany (FRG)	Not reported	Not reported	Samples taken from 2 incinerators; sampling protocol and number of samples not reported	Methodology and QA/QC reported or referenced
Kuehl et al. (1986)	Study bioavailability of PCDD/PCDFs from fly ash	Not reported	Not reported	Not reported	Not reported
Tong and Karasek (1986)	Compare PCDD and PCDF isomer distribution pattern and concentrations from fly ash from different countries	Not reported	Location of combustors only reported	Sampling protocol number of samples not reported; sample handling preparation reported or referenced	Methodology and some QA/QC reported or referenced
Austin and Newland (1985)	Study leaching of Cd and Mn from combustor in The Netherlands	Not reported	Briefly reported	Single sample taken; sample preparation reported	Methodology and QA/QC reported
Kuehl et al. (1985)	Determine bioavailability of TCDD from fly ash to freshwater fish	Not reported	Regional location of combustor only reported	Sampling protocol not reported; sample handling and preparation reported	Methodology and QA/QC reported or referenced
Mika and Feder (1985)	Provide information on the nature of residue in the environment	Not reported	Facility location, operating conditions, APC, ash handling and treatment reported	Ash sampling protocol not reported;	Methodology and some QA/QC reported
Morselli et al. (1985)	Study effect of different technologies and working conditions on toxic organics in fly ash	Reported in fair detail	Reported in detail	2 samplings under same conditions, details not reported	Reported in detail

Table 3-11. Summary of Literature Database Quality (continued)

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^d	Analysis ^e
Wakimoto and Tatsukawa (1985)	Determine presence of PCDDs and PCDFs in fly ash from 9 Japanese incinerators	Not reported	Location of incinerator and type of APC device briefly reported	Not reported	Methodology and some QA/QC reported or referenced
Clement et al. (1984)	Compare analytical methods for analysis of organics in fly ash from Canada	Not reported	Not reported	Sampling protocol not reported; 3 grab samples, and sample handling and treatment reported	Methodology and QA/QC reported or referenced
Rghei and Eiceman (1984)	Characterize adsorption and chlorination reactions of DD and 1-MCDD on fly ash	Not reported	Not reported	Referenced	Referenced
Tong et al. (1984)	Demonstrate use of HPLC in quantifying PCDDs in fly ash	Not reported	Briefly reported	One grab sample from each of two facilities and some sample handling reported; sampling protocol not reported	Methodology and some QA/QC reported or referenced
Viau et al. (1984)	Evaluate source of hazardous organic compounds in fly ash from Canada and Norway	Not reported	Location of facility only reported	Not reported	Methodology and QA/QC reported or referenced
Ballschmiter et al. (1983)	Compare procedures used to analyze PCDDs and PCDFs in fly ash	Not reported	Not reported	Not reported	Methodology reported or referenced; QA/QC not reported
Eiceman and Vandiver (1983)	Measure adsorption of PAHs on fly ash	Not reported	Location of facility only reported	Sampling protocol and number of samples not reported; sample handling and preparation reported	Methodology and QA/QC reported or referenced
Karasck et al. (1983)	Study operating conditions of incinerator to minimize dioxin formation	Analyzed for PVC content only	Reported	Reported	Methodology and QA/QC reported

Table 3-11. Summary of Literature Database Quality (continued)

Reference ^a	Research Goal	MSW Fuel ^b	Facility ^c	Sampling ^d	Analysis ^e
Cavallaro et al. (1982)	Study PCDDs and PCDFs in incinerator effluents	Not reported	Not reported	Sampling protocol and sample preparation referenced; number of samples not reported	Methodology and some QA/QC reported or referenced
Clement and Karasek (1982)	Measure concentration of PCDDs on different size fractions of fly ash	Not reported	Reported	Sampling protocol not reported; number of samples and sample preparation reported	Methodology and some QA/QC reported or referenced
Karasek et al. (1982)	Analyze concentration of organic compounds on different size fractions of fly ash	Not reported	Reported	Reported in detail	Reported in detail
Olie et al. (1982)	Evaluate PCDDs and PCDFs in fly ash from the Netherlands	Not reported	Not reported	Not reported	Methodology and QA/QC reported or referenced
Buser et al. (1978)	Analyze PCDFs in fly ash	Not reported	Country location only reported	Sampling protocol and number of samples not reported; sample preparation referenced	Methodology and some QA/QC reported or referenced
Hrudey et al. (1974)	Analyze constituents of ash	Not reported	Not reported	Sample handling and preparation only reported	Methodology and some QA/QC reported or referenced

^a References are arranged by year of publication in reverse chronological order and alphabetized within years.

^b "Reported" or "referenced" in this column indicates that a characterization of the MSW combusted was conducted.

^c "Reported" in this column indicates that a description of the facility and combustion parameters was included in the referenced report.

^d This column indicates whether a description of where, when, and how many ash samples were acquired from the MWC facility prior to any analysis. In general, sample handling and preparation includes only those physical processes used to make the ash amenable to analysis (e.g., storage, grinding, or screening).

^e "Reported" or "referenced" in this column indicates that a description of methodology or QA/QC procedures was included in the report. "Not reported" does not necessarily indicate that proper methodology or QA/QC was not used, only that there is no mention of it in the text. Methodology includes experimental and control procedures (e.g., extraction, dilution, and column leaching tests) and analytical procedures (e.g., GC/MS and AA Spectroscopy). QA/QC includes replicate testing, instrument calibration, standards, controls, descriptive statistics, etc.

As shown in Table 3-11, virtually no studies have been undertaken that attempt to relate the source and composition of MSW fuel, incinerator type, location, operating parameters, and pollution control devices to ash composition. Much of the research on ash was performed for purposes other than providing representative data on ash composition. In most studies, no attempt was made to collect ash that is representative of the facility, or to determine intra-facility variability. Often, research was performed on single grab samples of ash provided to the researcher by facility operators or third parties. Thus, although most studies pay adequate attention to analytical QA/QC, few studies document sampling and sample handling protocols, or even the number of samples taken. Our intent is not to criticize the researchers or the research, since obtaining representative samples was usually not necessary to address their specific research objectives, but to point out the difficulty in using these data for an exposure/risk assessment for MWC residuals.

The use of a number of different procedures to characterize MWC ash can be found in the literature. There has been considerable confusion over the basis and the roles of some of these tests, in particular the TCLP (Toxicity Characteristic Leaching Procedure) and EP (Extraction Procedure). The TCLP (which recently replaced the EP) is used for identifying those wastes which may present a risk to human health and the environment when improperly managed. Wastes that fail this test are hazardous wastes under RCRA. In developing this test, EPA chose a reasonable worst case mismanagement scenario in order to ensure that wastes would be adequately controlled, regardless of the manner in which they are actually managed. The results of these tests were never intended to be compared with actual field leachates.

A number of other tests, such as the Monofilled Extraction Procedure (SW-924), have been developed to estimate the presence of leachable constituents in wastes under different disposal scenarios. Although these procedures may be useful for predicting the leachability of a waste under a particular disposal scenario, they carry no regulatory weight.

Nevertheless, these extraction procedures continue to be used to characterize ash in many studies. Although a comprehensive research, development, and demonstration program has been proposed by EPA to redress the deficiencies in the ash database (EPA, in preparation), it will still require several years and a great deal of resources before confident predictions of ash composition can be made from existing data, without actually sampling ash from individual facilities. For all of these reasons, values reported in this section should be viewed with extreme caution and only in the context of a demonstration of the ranges of constituent concentration reported in the literature.

3.3. CHEMICAL CHARACTERISTICS OF BOTTOM AND FLY ASH

As discussed in Section 3.2, and shown in Tables 3-1 to 3-3, the ranges of concentrations of the ash constituents that have been reported by various authors are often seen to vary by several orders of magnitude. This section describes in more detail the inorganic and organic constituents that have been reported in bottom and fly ash from MWC facilities, and further underscores the reported intra- and inter-facility variability in the chemical makeup of MWC ash.

3.3.1. Inorganic Constituents

High concentrations of heavy metals and other inorganic constituents have been reported to be present in MWC residues (Healy et al., 1979; EPA, 1987a; EPA, 1988a). The concentration ranges of some inorganic constituents in bottom, fly, and combined ash from municipal solid waste incinerators are shown in Table 3-1. Higher concentrations of inorganic chemicals have generally been found in fly ash than in bottom ash. For example, fly ash usually has greater mass values of cadmium and lead than bottom ash. However, bottom ash has generally been found to have greater total mass values for silicon, aluminum, calcium, iron, copper, and zinc.

The partitioning of inorganic constituents between bottom and fly ash may be due to the deposition of metal oxides, hydroxides, or salts on fly ash particles by means of the volatilization-condensation reaction mechanism. This reaction is controlled by the individual vapor pressures and boiling points of the various metals. However, the reported partitioning of a number of elements cannot be explained by this reaction alone. In some cases, the concentration of these chemicals (copper, iron, lead, chromium, cadmium, tin, strontium, cobalt, barium, and phosphorus) has been reported to be higher in bottom ash than in fly ash. Many of these constituents volatilize under combustion conditions, and therefore could become oxides on particle surfaces. Some metals, such as cadmium, apparently concentrate on the surface of fly ash particles, while others, such as manganese, appear to occur as a matrix component of fly ash particles (Austin and Newland, 1985).

3.3.2. Organic Constituents

The ranges of concentrations reported in the literature of some organic constituents present in municipal waste combustor ash are shown in Table 3-2. Polycyclic aromatic hydrocarbons (PAHs), phthalates, chlorobenzenes, and chlorophenols are the most prevalent types of organic compounds found in municipal waste combustor ashes. The concentrations of organic constituents are generally greater in fly ash than in bottom ash, while the concentrations of organic constituents in combined ash have

intermediate values. The various data sets also indicate an absence of volatile organic compounds. Volatile compounds would not be expected in materials that are combusted at temperatures that are much higher than their boiling points, such as those temperatures present in municipal waste combustors. Also, they would tend to be lost during extraction and analysis procedures.

Like the inorganic constituents, the concentration ranges of organic constituents vary over several orders of magnitude in MWC ash. This may be due to the variability in combustion quality, difficulties in obtaining representative samples, and some of the same factors effecting inorganic constituent concentrations.

PCDDs, PCDFs, and PCBs, and their homologs, have been detected and quantified in MWC ash (EPA, 1987a; 1988a; Lisk, 1988; Kuehl et al., 1985, 1986, and 1987; Tanaka and Takeshita, 1987; Clement et al., 1984; Viau et al., 1984; Buser et al., 1978). Table 3-3 shows the reported concentration ranges of PCDDs, PCDFs and PCBs in fly, bottom, and combined ash. These compounds are usually found in greater concentrations among the smaller particle sizes, such as found in fly ash. This differential partitioning may be explained by the fact that smaller sized particles have larger surface areas relative to weight and therefore have a greater area for sorption per mass unit.

The variation in reported concentrations of PCDDs, PCDFs and PCBs is relatively greater than the variation in concentration of other organic compounds or inorganic compounds found in ash. This variability may be due to sampling location within the incinerator, operating conditions, and the incinerator or air pollution control system configurations and designs.

3.3.3. Intra- and Inter-Facility Variability

The concentration of residual components may vary in relation to the operating parameters of an incineration facility, and in relation to the variability of the feedstock. Since MSW is a heterogeneous product, a lack of homogeneity in the composition of the residuals produced is to be expected. A comparative analysis of the inorganic and organic constituents of the residues produced from four municipal waste combustor facilities has been conducted (EPA 1987a; EPA, 1988a). The four incineration facilities were all continuous feed, mass burn incinerators having different grate designs (one rotary, one traveling grate, and two with reciprocating grates), and ranging in operation dates from 1972 to 1986. The study examined the composition of residuals sampled over different shifts, days, and incinerator units. The results of one study indicated a wide variability in contaminant concentrations within the same facility over the different days, shifts, and units. These results implied that a slight change in feed material or operating parameters would significantly affect the composition of the residuals. Furthermore, intra-

facility comparisons found that the variability of contaminant concentrations between facilities was so large that the standard deviation of the sample means exceeded the average concentration of each contaminant (EPA, 1988a). In addition, results indicated a substantially higher variability within and between facilities for bottom ash alone and for combined ash, than the variability found in fly ash samples from different facilities.

3.4. LEACHATE CHARACTERISTICS

The direct contamination of groundwater, surface water, and soil by leachates from MWC residuals represents a potential route of exposure to the compounds that are found in ash. Leachate characteristics may also determine the extent of contaminant migration in environmental media. Two approaches have been taken to determine the composition of MWC residuals leachates: (1) the generation of simulated leachates, and (2) the study of field-generated leachates.

Although each approach has certain limitations, laboratory leachability simulation studies have been used to understand the potential leachability of organic and inorganic constituents in MWC ash. These extraction procedures were devised to simulate natural leaching conditions in the absence of actual field leachate data. The tests were designed to be conservative, i.e., they maximize the potential for leaching to occur, since the data generated from these tests are used for designing landfills and leachate treatment facilities, or to designate a waste as hazardous or not. Because a number of such leaching procedures exists, the data generated from these leachate tests have been criticized for the variabilities in experimental conditions, and for their inability to predict long-term leaching behavior at all disposal sites and for all types of residues (Van der Sloot et al., 1989; Northeim et al., 1989; Francis and White, 1987; Belevi and Baccini, 1989; EPA, 1989a; Kellermeyer and Ziemer, 1989; Ujihara and Gough, 1989). Furthermore, the characterization data available on ash extracts are more limited than data on ash itself, and relate mostly to the inorganic constituents rather than the organic content of ash extracts.

Of the commonly used laboratory leaching methods, two have regulatory significance for the U.S. EPA: the Extraction Procedure (EP) and the Toxicity Characteristic Leaching Procedure (TCLP). The TCLP replaced the EP in 1989 as the test used by EPA to determine if a waste exhibits hazardous characteristics, and should therefore be handled under hazardous waste regulations. The Monofilled Waste Extraction Procedure (also called the Deionized Water Extraction test method, and referred to as SW-924) has been used to estimate the presence of potentially leachable constituents in a solid waste, and to measure the concentration of these constituents in extracts. Simulated Acid Rain extraction is another commonly used test procedure that is a more aggressive variant of SW-924. However, SW-924 and the

Simulated Acid Rain extraction procedure are not appropriate for regulatory purposes. The conditions for the EP, TCLP, and SW-924 are summarized in Table 3-12.

The EP and TCLP tests were designed to identify wastes which may present a risk if mismanaged. The design of these tests is based on a scenario of co-disposal with MSW in an unlined sanitary landfill. EPA (1987a; 1988a) found that leachates produced under simulated laboratory conditions showed greater concentrations of lead and cadmium than actual leachates collected in the field. This is not unexpected given the design of these tests. However, it should be noted that these studies relied on the collection of a small number of leachates over a relatively short period of time. Therefore, sampling conditions may not adequately reflect the long-term leachability of these elements. As well, the EP Tox and TCLP tests have underestimated the presence of other elements, as seen in Table 3-13 for arsenic, barium, manganese, selenium, chloride, and sulfate.

3.4.1. Inorganic Constituents in Leachate

Several authors have investigated the factors that affect concentration of inorganic constituents in leachate. Ontiveros (1988) found that extract concentrations of cadmium were dependent on pH, while extraction concentrations of lead were enhanced by the addition of anions. Van der Sloot and colleagues (1989) also observed a relationship between pH and leachability for several metals. These findings underscore the need for a study of the long-term pH changes in disposal or construction sites that contain combustor residues. Under oxidizing conditions, Van der Sloot et al. (1989) observed minimum leaching of metals within the pH range of 8-10, but found large increases in metal leachability when the pH was decreased to 6 or 5. In addition, increasing the pH to above 10 increased the potential for leaching of lead, zinc, and copper. Under reducing conditions, however, metals were thought to be effectively retained as sulfides. DiPietro et al. (1989) also examined the effects of pH and redox potential on leachability of metals from combined bottom and fly ash. Lower pH levels increased the concentrations of metals, except for sodium and aluminum, in laboratory leachates. This finding has been attributed to a greater adsorption and precipitation of metals at moderate pH. Sodium concentrations were found to increase slightly at higher pH levels, possibly as a result of ion-exchange. Aluminum concentrations were higher at pH levels of 10 and 4 than at pH 7, reflecting aluminum's amphoteric nature (the pH minimum of aluminum hydroxides is generally around pH 6).

Table 3-12. Summary of Conditions for EP, TCLP, and SW-924 Leaching Methods

Conditions	EP ^a	TCLP ^b	SW-924 ^c
Liquid:Solid Ratio	20:1	20:1	10:1 per extraction
Extraction Medium	0.5 N acetic acid	0.01 N acetate buffer	Distilled/deionized water
pH Control	5	5 or 3	None
Extraction Time	24 hours	18 hours	18 hours per extraction
Agitation Method	Tumbler	Tumbler at 30 \pm 2 rpm	Tumbler
Temperature Control	20 - 40° C	22 \pm 3° C	25 \pm 1° C
Particle Size	<9.5 mm	<9.5 mm	<9.5 mm
Number of Extractions	1	1	4, sequentially

^a EP = Extraction Procedures (40 CFR 261, Appendix II), 1980

^b TCLP = Toxicity Characteristic Leaching Procedure (Revised 40 CFR 261, Appendix II), 1986

^c SW-924 = Monofilled Waste Extraction Procedure (A Procedure for Estimating Monofilled Solid Waste Leachate Composition. Technical Document SW-924. 2nd edition).

Source: EPA, 1987a.

Table 3-13. Concentration of 13 Inorganic Constituents in Ash, Laboratory Extracts, and Field Leachates from Five Municipal Waste Combustor Facilities and Their Associated Land Disposal Sites^a

Chemical ^c	CO2	Ash Extract by Six Methods (µg/L) ^b					Leachate µg/L
		SW 924	EP Tox	TCLP1	TCLP2	SAR	
Arsenic	10.3	11.0	10.4	8.5	14.5	8.5	79.5
Barium	298.0	599.2	221.0	491.7	373.2	695.0	1404.3
Cadmium	92.9	2.5	633.3	344.2	611.9	2.3	2.0
Chromium	3.4	5.4	35.9	3.1	150.6	4.2	7.4
Copper	160.0	123.2	1211.8	116.3	300.9	127.7	3.8
Lead	92.5	571.6	6621.1	1869.7	8188.0	584.2	21.5
Manganese	1068.8	3.3	2917.4	1476.3	3438.4	1.5	4217.5
Mercury	8.0	0.3	15.6	0.4	0.3	0.3	0.1
Selenium	2.5	2.5	2.5	2.5	2.5	3.9	5.6
Silver	2.6	2.0	2.0	2.0	2.0	2.0	2.0
Zinc	17705.9	257.5	48768.2	21150.3	58835.9	241.7	103.5
Chloride	666.6	695.9	711.3	744.3	716.9	1030.2	7571.4
Sulfate	628.8	267.9	852.1	644.1	845.3	544.6	1391.1

^a Extract values reported in this table are mean concentrations from five samples of ash extract from five facilities, i.e., 25 samples. Leachate mean was from 16 samples: 7 from one land disposal site, 3 from one site, and 2 each from 3 sites. When not detected, the value assumed to determine mean concentration was 1/2 the limit of detection. Data from EPA, 1990a, statistical analysis in MRI, 1990b.

^b CO₂: CO₂ saturated deionized water
 SW 924: Monofilled Waste Extraction Procedure
 EP Tox: Extraction Procedure (EP) toxicity method
 TCLP1: Toxicity Characteristic Leaching Procedure (TCLP) method #1
 TCLP2: TCLP method #2
 SAR: Simulated Acid Rain

^c Limit of Detections for ash extraction and leachates, µg/L:

Arsenic: 17	Barium: 1	Cadmium: 4
Chromium: 4	Copper: 4	Lead: 32
Manganese: 2	Mercury: 0.2	Selenium: 5
Silver: 4	Zinc: 2	Chloride: 0.5
Sulfate: 0.5		

Source: EPA, 1990a; MRI, 1990b.

Redox potentials dramatically affect the soluble metal concentrations of zinc, copper, nickel, iron, and lead (DiPietro et al., 1989). These metals are known to precipitate as insoluble metal sulfides under reducing conditions. DiPietro et al. (1989) noted that changes in redox potentials may affect metal solubility by directly changing redox sensitive metal oxidation states to more soluble/insoluble species, by changing the extent of redox sensitive metal surfaces available for adsorption, and by changing the degree of co-precipitation, precipitation, and complexation with other redox sensitive cations and anions. In general, iron concentrations in laboratory leachate experiments were found to be greatest at low pH and low redox potentials. The solubilities for zinc, copper, nickel, and aluminum were greatest at low pH and high redox potentials.

A study by Kullberg and Fallman (1989) indicates that differences in the flue gas cleaning processes can produce fly ash having different metal and salt leaching properties. Fly ash produced from four flue gas cleaning processes in Swedish incinerator facilities had little variation in inorganic concentrations. However, the flue gas cleaning process greatly affected the variation in leaching of inorganic constituents, especially chloride, lead, and cadmium. Greater leaching was observed in residues produced from semi-dry processes that use lime slurry scrubbers than from fabric filter processes or washing tower condensing processes.

The leachability of metals has been found to be less affected by particle sizes than would be expected to result from differential surface areas (Van der Sloot et al., 1989). Leaching test differences between crushed and untreated bottom ash have been found to vary by only 12-50%. This indicates that metal leaching may be dictated more by the ash matrix, and less by exposed surface area (Van der Sloot et al., 1989).

The ranges of concentrations reported in the literature of some inorganic constituents determined by SW-924, TCLP, and EP in fly ash, bottom ash, and combined bottom and fly ash extracts are shown in Tables 3-4, 3-5, and 3-6. As can be seen, there are wide variations in reported concentrations, within and between laboratory procedures.

There are also differences between laboratory procedures and field leachates in some limited comparison sampling that has been done. Table 3-13 compares the concentration of 13 inorganic constituents in combined ash, ash extracts by 6 methods, and in leachate collected from land disposal sites. This study was conducted by EPA and CORRE on 5 MWC facilities and their associated land disposal sites (EPA, 1990a). Five ash samples were taken from each facility and measured for the 13 constituents; all six extraction methods were also run on each of the five samples. Two leachate samples were taken from 3 of the associated land disposal sites; 3 and 7 samples were taken from the remaining 2 disposal

sites. Further details of this study can be found in EPA (1990a). The statistical evaluation and the summary provided in Table 3-13 was conducted by MRI (1990b). Although the sample sizes were small, particularly for the field leachates, some observations can be made. Generally higher field leachates were noted for arsenic, barium, manganese, selenium, and chloride, while the laboratory extraction procedures generally had higher concentrations for copper, lead, and zinc. Four of 6 laboratory procedures had substantially higher results for cadmium as compared to the field leachates, while only 1 of 6 laboratory results differed substantially for chromium. Without statistical rigor, reasonable agreement between laboratory results and field leachates were noted for other results (statistical correlation analysis performed in MRI, 1990b). Concentration of these inorganic compounds were provided on Table 3-13 for comparison.

3.4.2. Organic Constituents in Leachate

A range of the concentrations of some organic constituents from extracts of fly ash and bottom ash are shown in Table 3-7 to 3-10. Few data exist on the concentrations of organic constituents in fly ash and bottom ash extracts from laboratory leaching tests. The concentrations of most organic constituents analyzed in fly ash and bottom ash extracts are generally below detection limits or at trace levels. The concentration of chlorophenols in fly ash determined by SW-924 were detected only in incinerators not equipped with fabric filter dust collectors. Trace concentrations of several organic constituents were found in bottom ash extracts using the SW-924 method (e.g., benzoic acid, cyclooctadecane, dimethyl furandione, methoxy ethane, and phenol). As shown in Table 3-8, methoxy ethane was the only organic constituent found in bottom ash extracts using the TCLP or EP methods, and its concentration was only slightly above the detection limits.

Results of laboratory leaching studies for combined bottom and fly ash are shown in Table 3-9. As with the fly ash and bottom ash databases, this information is also extremely limited. Only dimethyl propdiol was detected in combined ash extracts using the TCLP method. Several organic constituents were detected at various concentrations in combined ash extracts using the SW-924 method.

Limited information exists on concentrations of PCDDs, PCDFs, and PCBs in fly ash extracts from simulated leaching tests (Table 3-10). No PCDDs, PCDFs, or PCBs have been found above the detection limit by the SW-924 method. Individual homolog concentrations of PCDDs, PCDFs, or PCBs were never found to exceed 0.12 ng/L (parts per trillion) in fly extracts determined by TCLP. Total PCDDs did not exceed 0.188 ng/L and total PCDFs did not exceed 0.152 ng/L in fly ash extracts determined by the TCLP method. No data were found for concentrations of these compounds in laboratory leaching test extracts

for bottom ash. The concentrations of PCDDs, PCDFs, and PCBs in combined bottom and fly ash extracts using the TCLP method are similar to those for fly ash extracts.

3.5. PHYSICAL PARAMETERS OF MWC ASH

The physical characteristics of MWC residuals represent an important consideration in analyzing their potential for environmental transport and human exposure. A source analysis must necessarily include a description of the physical characteristics of the released substances. This information is used to develop a better understanding of the environmental transport and fate characteristics of the containment and its exposure potential.

3.5.1. Particle Morphology and Mineralogy

As described below, the chemical and physical properties of the ash (e.g., particle morphology and mineralogy) from MWC facilities will largely determine the extent to which ash constituents will be released via leaching.

Ontiveros (1988) reported that the morphology of fly ash did not vary significantly between samples from different incinerators. Eighmy et al. (1990) examined the particle morphology and mineralogy of MWC ashes using scanning electron microscopy. In general, municipal waste incinerator residues were described as amorphous non-descript and mineralogically diverse. Ettringite or calcium silicate hydrate crystal morphologies were also not observed in bottom or combined ash, indicating that solidification of these ashes may be CaCO_3 - or $\text{Ca}(\text{OH})_2$ -based pozzolan-like cementation. A pozzolon is a siliceous material which reacts to the presence of moisture and alkali and alkaline earths to yield a cementitious product. However, Ontiveros (1988) reported crystals heavily dispersed on fly ash particle surfaces.

The "core" principal elements found in ash particles are aluminum, calcium, iron, sulfur, and silicon. In addition, calcium and iron aluminosilicates are observed to form the major solid phase of fly ash, bottom ash, and combined fly and bottom ash with scrubber residue (Ontiveros, 1988; Eighmy et al., 1990). A number of other minerals have been identified in ash, including NaAlBr_4 , FeCr_2O_4 , and CaCO_3 in bottom ash; CaSO_4 and CuFe_2S_3 in fly ash; and $\text{Ca}_2\text{B}_2\text{O}_5$, CaSO_3 , CaHPO_4 , CaCl_2 , and $\text{Ca}_2\text{P}_2\text{O}_7$ in scrubber residue. Lead and cadmium are found as PbSO_4 and CdSiO_3 , respectively, in bottom ash; CdSO_4 in fly ash; and $\text{Pb}_5(\text{PO}_4)_3\text{Cl}$ in scrubber residue. A number of other sulfate minerals, as well as chlorides and oxides are found in residues (Ontiveros, 1988; Eighmy et al., 1990).

3.5.2. Particle Size Distribution

Fly ash is made up of much finer particles than bottom ash. As shown in Figure 3-1, the grain size distribution of mixed ash has been determined by Forrester (1988). A particle size analysis for fly ash at one site was performed as part of the Red Wing RDF Ash Disposal Site Risk Assessment (Northern States Power Company, 1987). Only particles less than 50 μm diameter were considered to have wind erosion/fugitive emission potential. Less than 6.7% of mixed ash consists of such particles. The particle size distribution for this potentially fugitive ash fraction is shown in Figure 3-2.

3.5.3. Particle Sizes and Chemical Composition

The constituent elements of ash exhibit a differential sorption on to different sized particles. For example, more than 75% of the mass of the following elements was found on small particles ($<2\mu\text{m}$) from the incinerators (Greenberg et al., 1978): Na, Cs, Cl, Br, Cu, Zn, As, Ag, Cd, In, Sn, Sb, W, Pb, while the following elements had a predominant large particle distribution: Ca, Al, Ti, Sc, La, and Vanadium. Constituent elements such as Cr, Mn, Fe, Co and Se had mixed size distributions.

According to the EPA (1986a), data on the distribution of organic compounds adsorbed on different sized particles of ash is lacking.

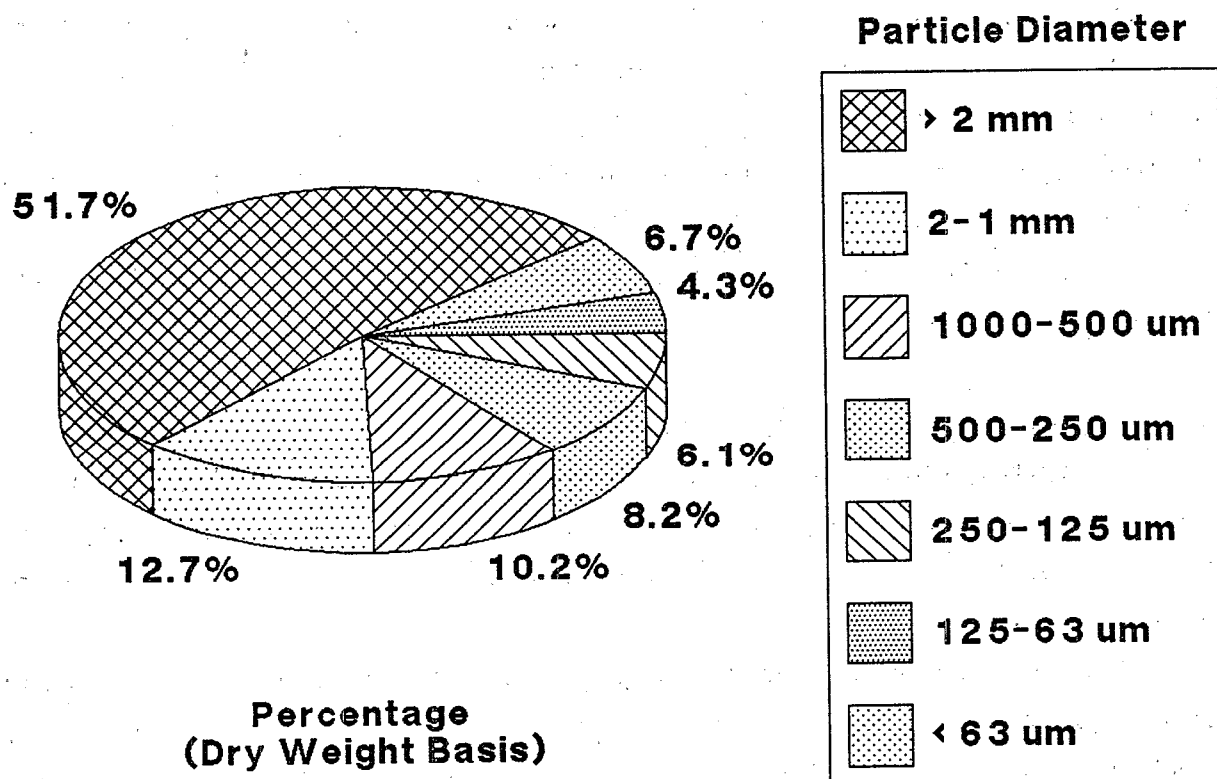
3.5.4. Engineering Characteristics

The aggregate characteristics of ash are important in determining its potential use as a substitute in construction and in landfill liner and cover systems. These characteristics are also important in determining potential exposure routes and contaminant release. Various beneficial uses of ash are described in Section 4.

Chesner (1990) reported interim results of physical and engineering tests of the non-ferrous portion of combined ash from three facilities. Ash from these facilities was described as having a "well graded" grain size distribution. This characteristic make the ash potentially frost-susceptible, and thus unusable for an unbound aggregate base, sub-base, or select fill material.

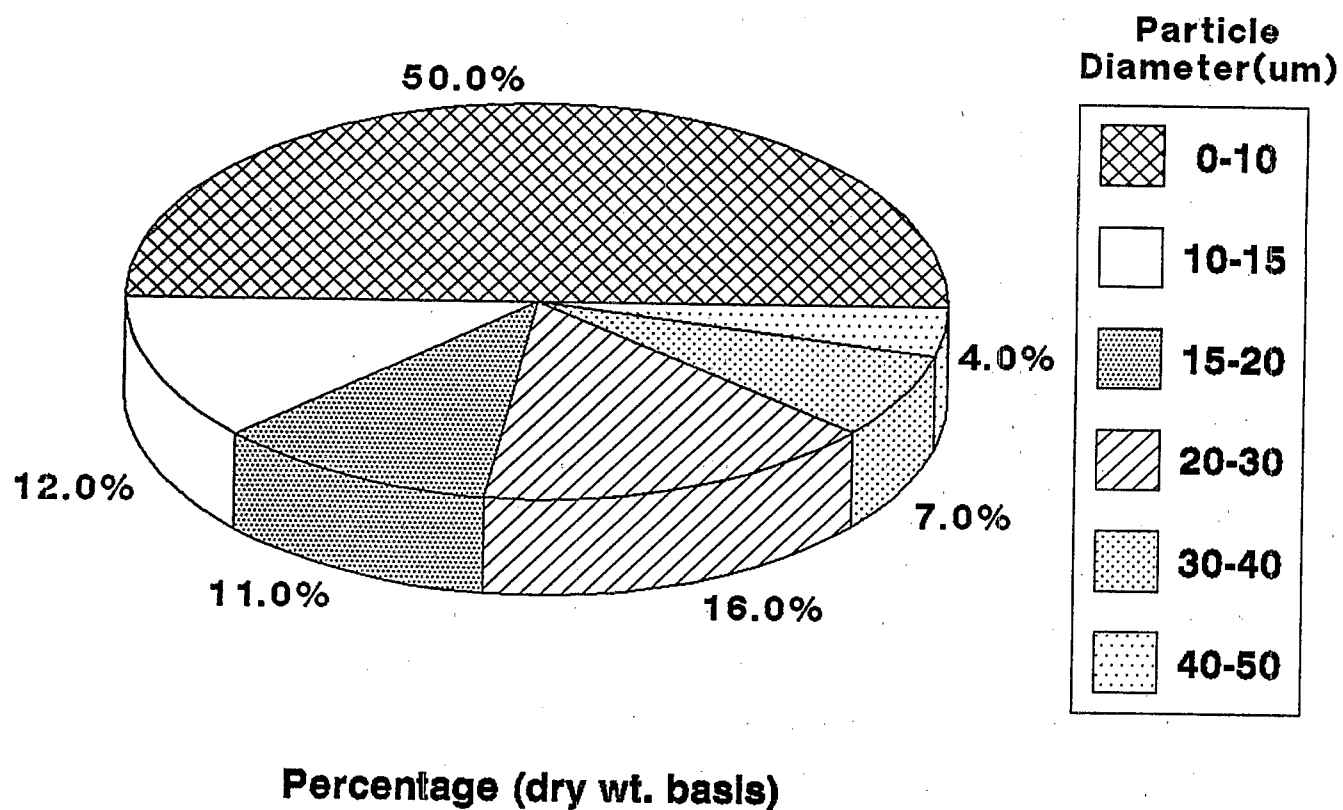
The moisture content of ash is greatly affected by the ash quenching and draining methods that are employed at a facility. The moisture content from the three facilities studied by Chesner (1990) ranged from 29.6-48.0% of total weight. This range is much too high for direct use of ash in many construction material applications. However, a decrease in moisture content has been reported after the ash had been stored over one month, resulting a more attractive material for reuse (EPA, 1988a).

Figure 3-1
Grain Size Distribution of Mixed Ash



Source: Forrester, 1988

Figure 3-2
Particle Size Distribution of Fly Ash



Source: Northern States Power Company, 1987

Gravimetric tests indicate that combined ash is more absorptive than natural aggregate materials, with values ranging from approximately 3.5-7.0% for coarse fraction material to 4.7-14.8% for fine fraction material. The frost-susceptibility of construction material incorporating ash is therefore potentially greater as a result of this property. The "less than two inch" ash fraction is a lightweight material when compared to natural material

Engineering tests of the durability of construction materials that incorporate ash show that such materials may be only marginally durable as compared to materials that use natural aggregates. Chesner (1990) concluded that the potential use of ash may be limited to such applications as an aggregate substitute in asphaltic concrete, portland cement concrete, stabilized base, granular base, sub-base, and fill applications.

3.6. SUMMARY AND CONCLUSIONS

Several constituents of ash are known to be important environmental pollutants, including certain heavy metals and organic contaminants. However, data on the presence and concentration of these constituents underscores the great variability that exists in the characteristics of the ash between facilities and residue types. This makes development of an exposure assessment for MWC residuals difficult, since input parameters such as concentrations and quantities of inorganic and organic constituents, and their leachability and physical properties, are numerous and variable both within and between sites. The differences in operating parameters and location of facilities clearly contribute to the observed variability in ash characteristics. In addition, much of the data reported in the literature is of little statistical use because of poor or unreported quality control/quality assurance, or because of insufficient information on the MSW fuel type, facility design, and facility operating parameters. Therefore, field research should be undertaken to develop information relating facility design and MSW fuel characteristics with ash characteristics. Such information would greatly support exposure assessments by clearly identifying and quantifying ash characteristics.

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4. MUNICIPAL WASTE COMBUSTION RESIDUAL MANAGEMENT

The analysis of the sources of potential release of MWC residuals into the environment is an essential first step in performing an exposure assessment of this material. This analysis begins with an understanding of the material flow from an incineration facility to the final disposal or reuse site. The following discussion provides an overview of common industry practices in the management of MWC residuals.

4.1. INTRAPLANT COLLECTION AND TRANSPORT OF RESIDUALS

The ash conveying systems in use at MSW incinerators were analyzed as part of a review of MWC systems operations and maintenance (EPA, 1987b). Bottom ash collected on or under the furnace is generally conveyed to a quench pit by a hopper. The residue hopper is normally discharged through a watertight gate valve into trucks or containers for storage or shipment to disposal sites.

Heavy fly ash may be collected in the mixing chamber, a vessel that holds volatile furnace gases at the proper temperature for a long enough period for complete combustion to occur. Cooling towers and air pollution control equipment also act as fly ash collection points. Conveyor systems may be employed to transport the collected ash from each of these collection points. Fly ash may also be combined with the bottom ash in the mechanical conveyor system prior to entering the quench pit, conveyed to a separate quench and combined with bottom ash after removal from the quench pit, or managed separately and not quenched.

The quench pit is a water-filled pit into which the ash falls after it exits the furnace. The quench serves to cool and wet the ash to reduce fugitive dust emissions. Ash dumped into the quench pit is normally removed by a drag conveyor submerged in the pit. This conveyor is constructed to permit drainage of water back into the pit and conveys wet ash (bottom, fly, or mixed) either to holding hoppers or directly into trucks.

Quench water is usually collected by draining basins or overflow tanks. In some facilities, quench water may be completely recycled within the system. In others, the quench water may be released as an effluent. Quench effluents usually require some form of treatment before being discharged, often to a publicly owned treatment facility.

4.2. POTENTIAL SOURCES OF RELEASE DURING TRANSPORT AND DISPOSAL ACTIVITIES

Disposal of MWC residual in a landfill is by far the most common fate for such wastes. (Other fates, including commercial uses of MWC ash, are outlined in Section 4.3.). The sources of potential environmental release of residuals during disposal activities include:

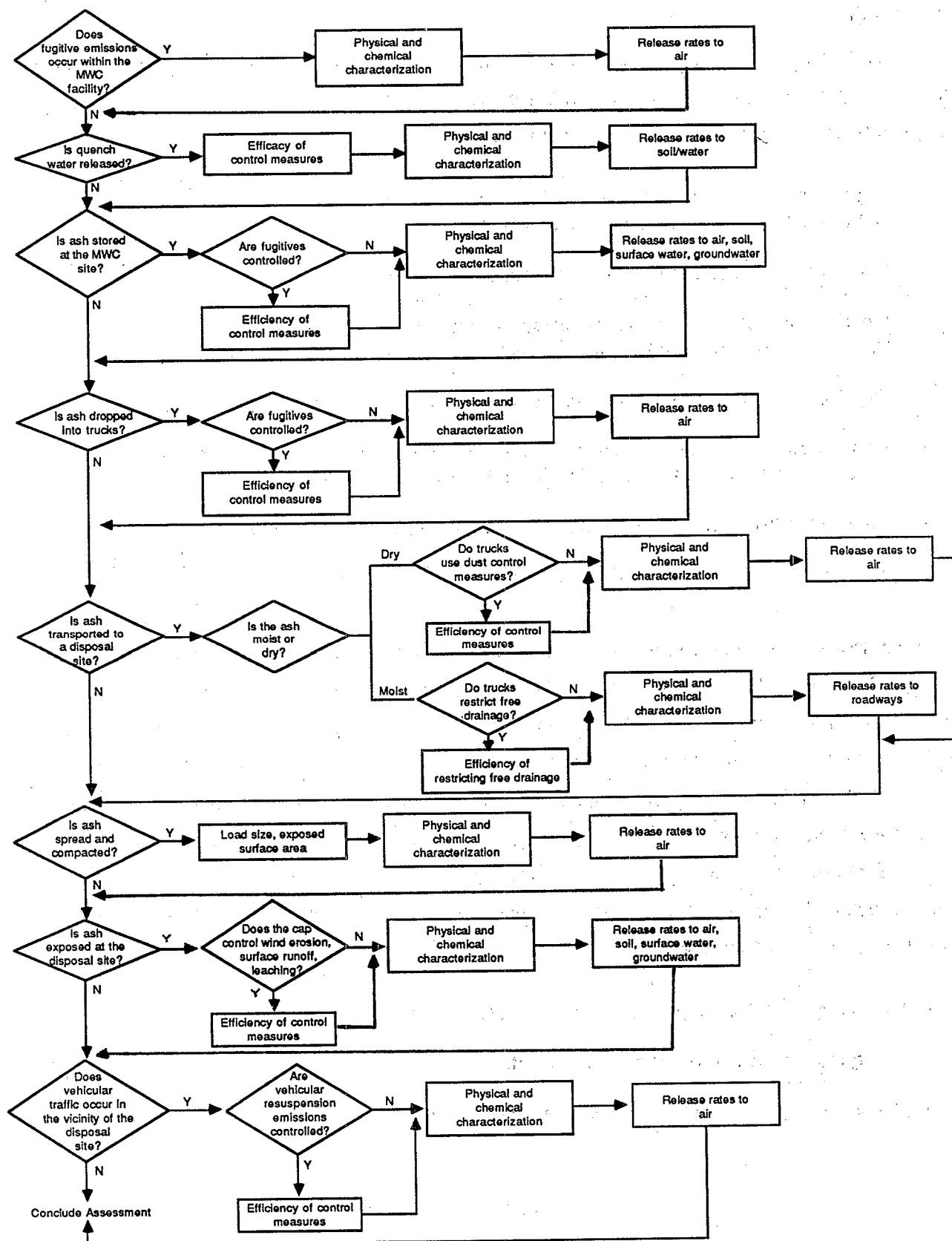
- Vaporization and fugitive emissions within the incinerator conveyor system during quenching and movement of ash to the storage/transport containers,
- Contaminated water releases from spray and quench water,
- Fugitive emissions, ground contamination, and runoff when the ash is dropped into the transport/storage containers or dumped into temporary storage piles or pits,
- Fugitive emissions or excess moisture drainage during truck loading and travel,
- Fugitive emissions due to deposition of ash on roadways and haul routes near storage and disposal points and subsequent vehicular traffic over these roadways and routes
- Fugitive emissions during unloading and spreading operations at the disposal site, and
- Fugitive emissions, runoff, and leachate generation at the disposal site.

These processes are represented in the source analysis schematic presented as Figure 4-1.

A particularly good source of information on operating practices common in the MWC industry, with respect to the transport and disposal of ash, is the survey conducted by the EPA Office of Solid Waste (OSW) in 1989 (EPA, 1989a; hereafter referred to as the OSW survey). EPA received responses from 121 of the 139 facilities surveyed, or a response rate of 87%. All of the statistics quoted in this report are based upon properly completed or "valid" responses. The number of valid responses varied by question. Some questions, such as "name of the facility" had a very high rate of valid response. Other questions, such as "amount of liquid residue generated" had very low valid response rates. Through this survey, EPA received information on:

- the design and operating practices of facilities used to incinerate MSW,
- the types of waste received by these facilities,
- the establishment of any recycling activities at these facilities, and

Figure 4-1
Source Analysis for MWC Residuals Release



- how MWC residues are generated, handled, transported, stored, and disposed of at these facilities.

The data obtained in this survey are necessary to evaluate the MWC waste management techniques currently employed in the United States, and subsequently to assess the potential for individuals in the population to become exposed to the toxic constituents of MWC residuals.

Because of the high response rate in this survey (87%), the results are assumed to be fairly representative of the operating practices of MSW incinerator facilities in the U.S. As a result, the OSW survey is able to accurately represent the potential sources of release of MWC residuals during transport and disposal. Therefore, the following discussion of potential sources of release of MWC residuals is based largely on the results of the OSW survey.

With regard to disposal practices, the OSW survey reported that 46.2% of the responding MWC operators stored ash on-site prior to disposal. This stored ash consisted of combined ash (66% of the cases), only fly ash (7%), only bottom ash (20%) and separate fly and bottom ash storage areas (7%). The average capacity of the storage area of the respondents to the survey was found to be 876 tons, while the average quantity of ash stored was 380 tons.

Storage areas can act as sources of release of residues to the environment. These release mechanisms may be through runoff, leaching, or fugitive dust emissions. The OSW survey found that various contaminant control measures were used by MWC operators (Figures 4-2 and 4-3).

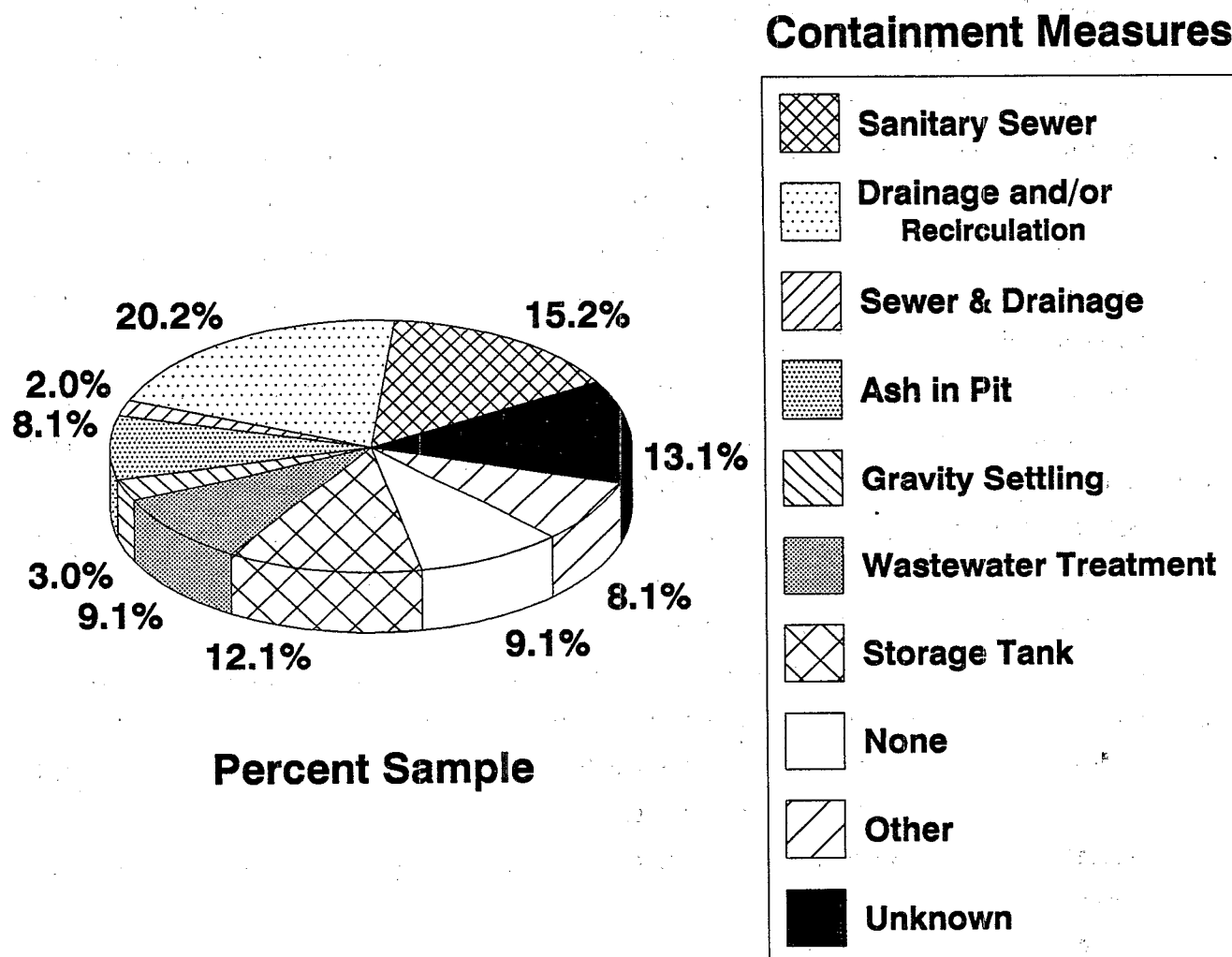
Ash is exclusively transported by truck from the incinerator/storage facility to the disposal site. Distance between the incinerator and respective disposal site were found in the OSW survey to range from 0 to 700 miles, with a mean distance of 32 miles, and a median distance of 8 miles.

Fugitive dust emissions can occur during transport of the residues to the disposal site. The majority (92%) of respondents to the OSW survey reported that precautions were taken at their facility to avoid fugitive dust releases during transportation. The relative use of various means to control fugitive dust emissions that may occur during transportation is shown in Figure 4-4.

Another means of limiting fugitive dust emissions, and possibly reducing leachates, is to treat the ash prior to disposal. The OSW survey found that only 10.2% of MWC operators treat the ash prior to disposal. This treatment can include neutralization or stabilization.

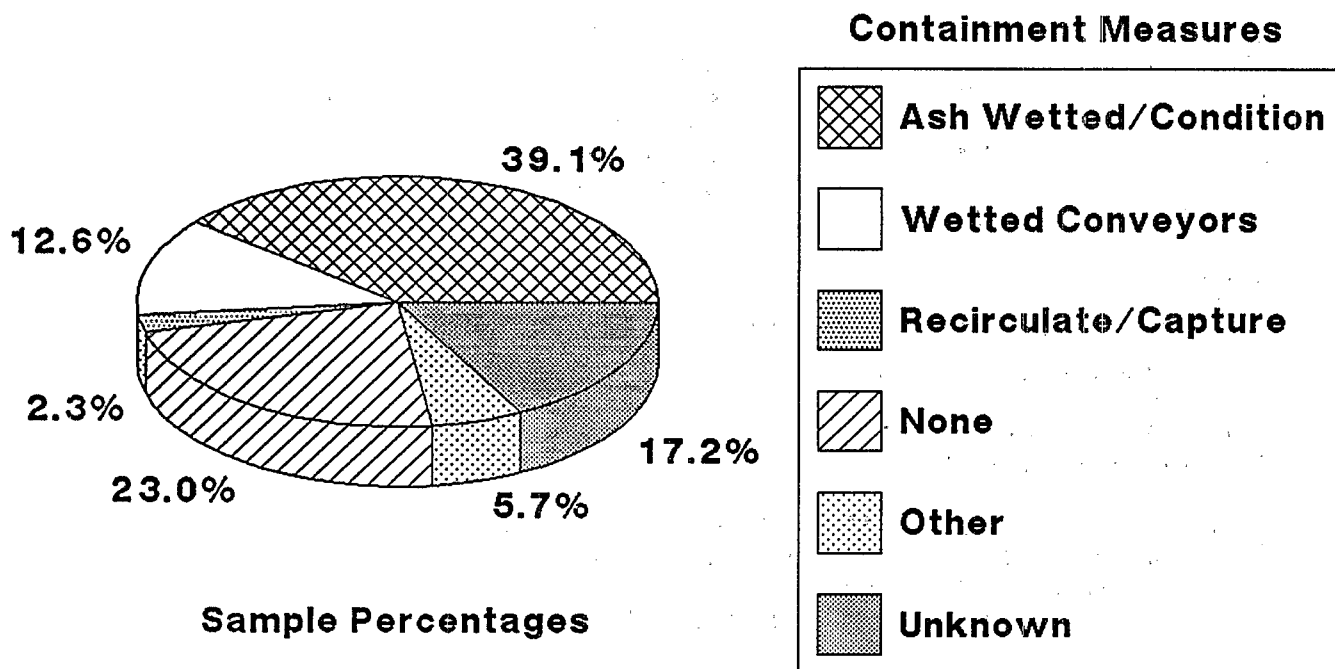
Fugitive dust emissions may also occur at a landfill during dumping and smoothing operations. No information was identified in the literature regarding precautions used to prevent or limit fugitive emissions at disposal sites.

Figure 4-2
Runoff Containment Measures
Used by MWC Operators



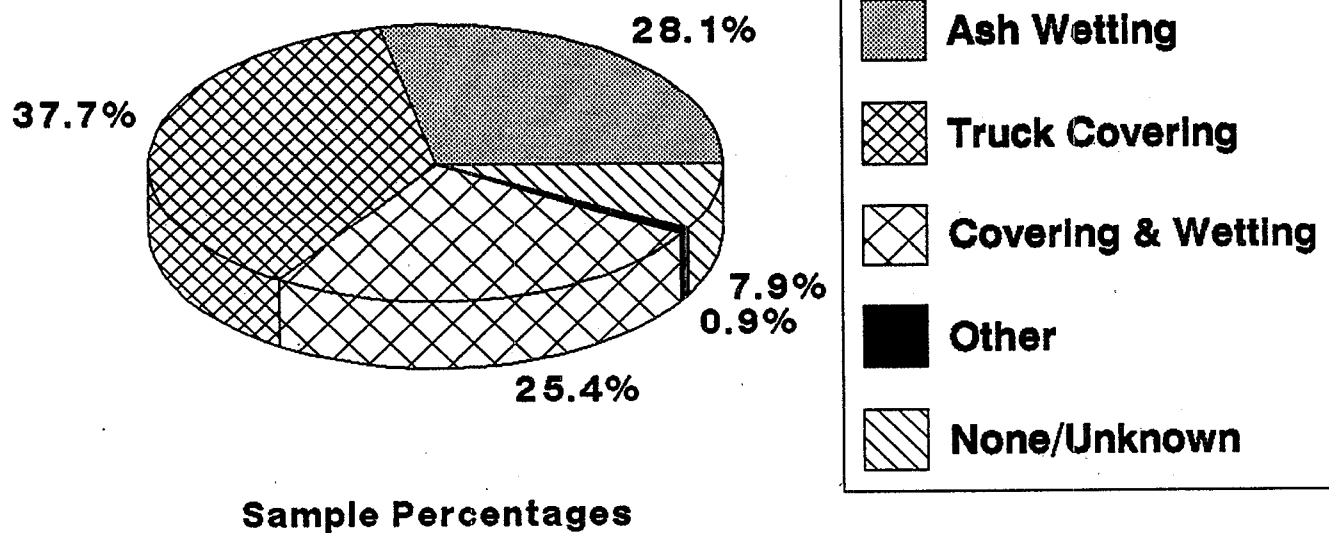
Source: EPA, 1989a

Figure 4-3
Fugitive Dust Containment Measures
Used by MWC Operators



Source: EPA, 1989a

Figure 4-4
Precautions to Control Fugitive Dust Emissions
During Transportation



Source: EPA, 1989a

The majority of ash is disposed of in municipal solid waste landfills where it may be segregated from other wastes or mixed with other trash. The frequency of use of the various sites is shown in Figure 4-5.

Protective liners may be used at onsite or offsite disposal facilities to prevent leachate migration. In the OSW survey, the MWC facilities were asked to provide information, to the extent possible, on the types of protective liners used at the land disposal site accepting their ash, on-site or off-site. This question had a low response rate of 42%. Of those responding, 96.1% used protective liners, the majority of which (54.9%) were clay liners. The balance of respondents used facilities with composite or synthetic liners (13.7% each), both clay and composite (2.0%), did not specify liner type (11.8%), or did not use liners (3.9%). A related question on this survey was on the use of a leachate collection system. This question also had a low response rate of 30%. Of those responding, 68% did employ a leachate collection system.

An EPA Office of Research and Development (ORD) laboratory conducted a follow-up survey to the OSW survey on the very same disposal facilities. In their case, however, they queried the land disposal facility in contrast to the MWC facility. Of 72 facilities they queried, including both off-site and on-site facilities, 40.3% used no liner material. Other respondents indicated: 38.9% used natural clay liners, 11.1% used synthetic liners, and 9.8% used a combination of liners. Asking also about leachate collection, they found that 50% of the facilities employed a leachate collection system and 34.7% of all facilities treated their leachates (EPA, 1988c).

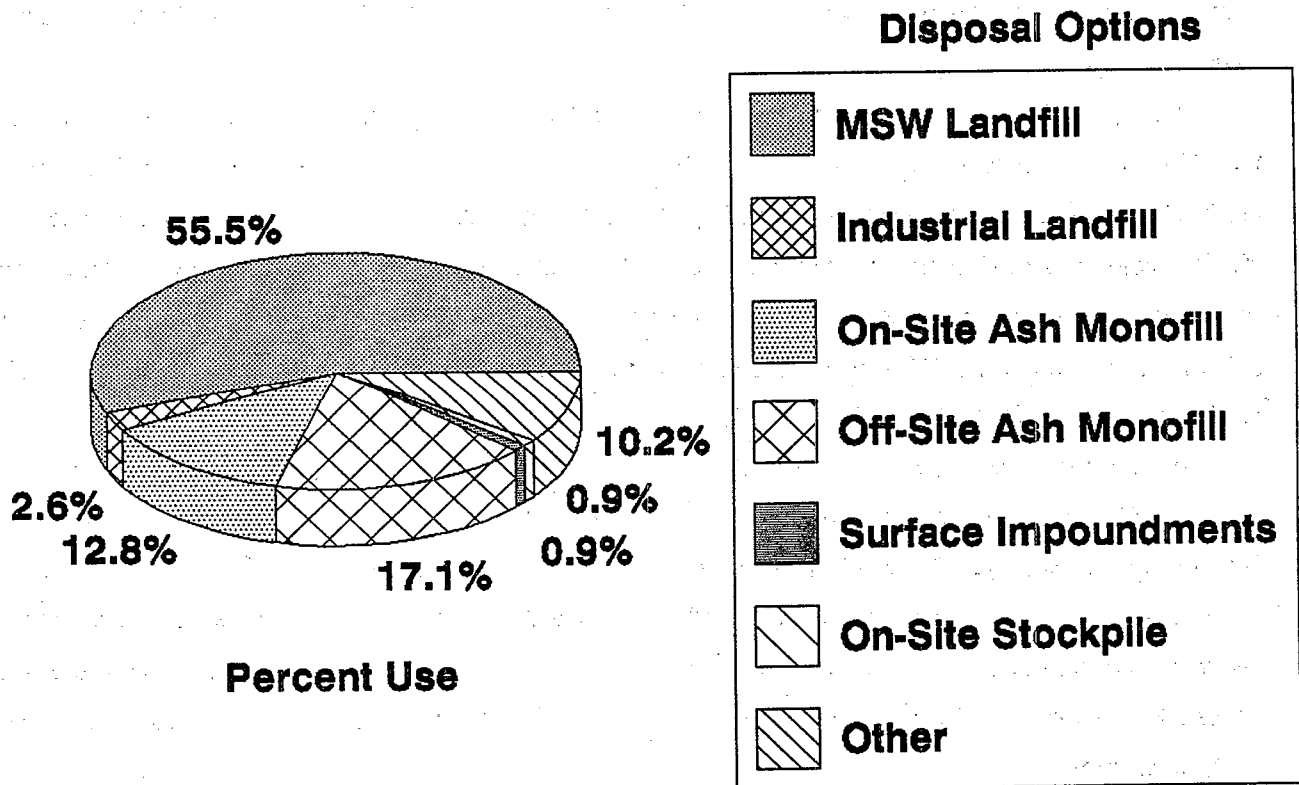
If improperly managed, this leachate could also be a point of contaminant release into the environment. Management options include sewage or on-site treatment, placing back on top of the landfill, and spray irrigation onto large land areas to promote evaporation.

The data summarized in this section, which were obtained in the OSW survey conducted in 1989, not only provide an overview of MWC operating practices common in the U.S., but also allow the exposure assessor to identify likely sources by which MWC residuals can be released into the environment.

4.3. USES OF RESIDUALS

Although disposal is presently the favored method of MWC ash management in the United States, increased landfilling costs and the potential for beneficial re-use may encourage alternatives in the treatment and use of MWC residuals. Previous documents have reviewed such potential treatments and beneficial uses of combustion residues (EPA, 1988a). It is not clear that potential re-use mitigates

Figure 4-5
Ash Disposal Options
Frequency of Use



environmental contamination. A discussion of environmental considerations can be found in Denison (1988). MWC ash has been used for several purposes (EPA, 1988a), including:

- construction fill material for minor roads, turning and parking areas,
- landfill cover,
- aggregate for highway construction,
- glassy frit after fusion of ashes via glassification or vitrification,
- soil enhancer for grass, ornamental plants, and non-edible foliage,
- substitute for gravel and cobble stones,
- substitute for artificial reefs,
- reclaim abandoned land,
- sub-base material, and
- building blocks for furnaces.

Municipal waste combustion bottom ash has been used as road construction material in the past. In the early 1970s, several streets in Harrisburg, PA, were constructed using ash from the local waste-to-energy combustor (Strauss, 1989). A demonstration project using incinerator residue as a substitute for road construction aggregate has been conducted by the Massachusetts Department of Environmental Management since 1982 (EPA, 1988a). Bottom ash was used to fill the road bed.

4.3.1. Stabilization of MWC Ash

Research has been conducted to determine the effectiveness of stabilization in reducing the potential for hazardous constituents to leach from MWC ash. Stabilization/solidification techniques that have been used to process hazardous solid waste may be amenable for treating MWC ash. These processes include thermoplastic, encapsulation, glassification (vitrification), and cement- and lime-based fixation.

The effectiveness of cement- and lime-based fixation agents to retard the release of metals from MWC ash matrices depends on both the physical and chemical properties of the binding mechanism and on the environmental conditions (Holland et al., 1989). For example, Kullberg and Fallman (1989) reported that lower concentrations of lead were found in leachate from semi-dry fly ash residue stabilized by mixing with water, cement, bentonite, sodium metalilicate, or vermiculite, than in leachate from unstabilized fly ash. However, other research (Holland et al., 1989) has shown that combined ash amended with cement was not successful in stabilizing aluminum, lead, and, to a lesser extent, nickel.

Similarly, Holland et al. (1989) has reported that combined fly and bottom ash amended with lime failed to immobilize manganese and zinc. All other metals that were tested for in this research were effectively stabilized by these two treatments.

In an effort to gather additional data on innovative ash stabilization technologies, EPA's ORD initiated a project to investigate the effectiveness of solidification/stabilization and other technologies in eliminating or reducing the release of toxic constituents from MWC ash and its leachate. A technical Advisory Panel, consisting of members from the private sector, environmental advocacy groups, incineration vendors, academia, and state, federal, and foreign regulatory agencies, assisted EPA in developing the program, evaluating proposals, and selecting vendors. The results of this study are expected to be available in the spring of 1991.

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5. EXPOSURE PATHWAYS

The analysis of exposure pathways addresses the transport of MWC residuals and their constituents from the source of release to their point of exposure with the affected individuals or populations. This analysis is concerned with the qualitative and quantitative examination of the releases which affect the environmental fate and transport of residuals and their constituents and which ultimately influence exposure point concentrations. The initiation of an exposure pathway analysis begins with the identification of the relevant environmental transport media. This step is followed by a quantitative estimation of exposure point concentrations of the residuals and residual components using appropriate environmental fate and transport models. The components of an exposure pathway analysis are:

- a qualitative and quantitative description of the scenarios and mechanisms of release of the residuals to the environment, including a description of the environmental transport media responsible for the transport of ash, and of the mechanisms of contaminant transfer from one medium to another, and
- a description of the points of potential receptor contact with contaminated media (exposure points), and the relative contribution of exposure routes at exposure points.

The release of MWC ash residuals to the environment can occur through fugitive dust emissions and leaching of ash components, to eventual transport to soil, groundwater and surface water bodies. This chapter focuses on these two mechanisms of release of MWC residuals and their components. The exposure points and relative exposures are described in Section 6.0.

5.1. MECHANISMS OF RELEASE: FUGITIVE EMISSIONS

Municipal waste combustion residues can be emitted into the air at a number of points along the route from the incinerator to the landfill/disposal site. In addition, residues can be released into the air after they have been disposed in a landfill. These fugitive emissions may occur through normal handling of MWC residuals through activities such as:

- conveying residues to storage piles,
- on-site storage,
- conveying and loading residues at the incinerator site,
- release along roadways while trucks are traveling to the disposal sites,
- unloading the trucks,

- spreading and smoothing operations at the landfill,
- wind erosion of the exposed ash surface at the landfill, and
- vehicular traffic in the vicinity of storage and disposal sites.

Methods to estimate the release of MWC residuals into the environment at these points is discussed below. Most of the methodologies are commonly known as "AP-42" emission factors. These have been developed by EPA's Office of Air Quality Planning and Standards (EPA, 1985a; EPA, 1988d), and are empirical equations. They are commonly used to estimate fugitive dust emissions, including applications described below for ash management (MRI, 1990a; EPA, 1988e; Kellermeier and Zeimer, 1989).

5.1.1. Conveying Residues to On-Site Storage

The emission of particulates during conveying operations from the MWC to the on-site storage site offers one possible pathway of release. Whenever the MWC facility uses a quenching system for both fly and bottom ash, air emissions would be expected to be limited. The high moisture content of the ash exiting a quenching system causes fine particles to adhere to the surfaces of larger particles with a resulting dust suppression effect. As mentioned in Section 3., over 50% of facilities use wetting or conditioning steps to limit particulate emissions during conveying operations.

If the moisture content is very low, the processes of loading and unloading, and transporting to the storage pile, can result in fugitive emissions. The loading and unloading emissions may be quantified in a manner described in Section 5.1.3.; emission from trucks is described in section 5.1.4.

5.1.2. On-Site Storage

The amount of fugitive ash released during on-site storage prior to final disposal can be estimated in a manner described in Section 5.1.6., on wind erosion.

5.1.3. Loading and Unloading of Trucks

Loading and unloading operations at MWC and disposal sites may result in the release of fugitive dust. The emission factor equation is developed to provide emission factors for kilograms of particulate emitted per megagram (metric ton) of soil loaded and unloaded (EPA, 1988d):

$$Elu = k(0.0016)(U/2.2)^{1.3}(M/2)^{-1.4} \quad (5-1)$$

where:

Elu = emission factor for loading and unloading(kg fugitive dust/Mg ash)
 k = particle size multiplier (dimensionless)

- U = wind speed (m/s)
M = material moisture content (%)

5.1.4. Transportation to Landfill

Fugitive emissions from trucks carrying ash can be minimized by wetting the ash or the use of truck covers. Since no emission factors have yet been developed for estimating emissions from open trucks, EPA (1988e) used the following approach:

1. Estimate daily number of truck loads transported
2. Estimate surface area of ash in each truckload capacity
3. Estimate emission of ash per time travel period (minute) by multiplying the surface area by the emission factor
4. Estimate the total particulate emission rate by multiplying the value of the particle emission rate by the travel time and by the number of daily loads.

The emission factor was one earlier developed for emissions from aggregate piles. (EPA, 1985a; a more refined methodology for emissions from aggregate piles is described below):

$$Ew_1 = 1.9(s/1.5)[(365-P)/235](f/15) \quad (5-2)$$

where:

- Ew_1 = emission factor 1 for wind erosion from stationary piles (kg/day/hectare)
s = silt content (%)
P = number of days with >0.25 mm precipitation per year (use 365 if ash is wet prior to transport)
f = percentage of time that the obstructed wind speed exceeds 5.4 m/s (assumed to be 100% when truck travels faster than 12 mph)

As this equation was developed for stationary piles rather than moving trucks, it is likely to be a source of uncertainty. Still, transport of ash of very low moisture content without proper truck covering (i.e., poor management practices) is likely to result in quantifiable fugitive emissions. Such a release should be considered in some manner, and equation 5-2 might give a reasonable first approximation.

The application of this equation to a situation where the ash is properly wetted prior to transport would show that no fugitive emissions of dust are expected to occur (e.g., $P=365$, $E=0$).

5.1.5. Spreading and Compacting at Disposal Sites

Fugitive dust emissions from spreading and compacting ash at disposal sites have been estimated in more than one way, although the different ways found in the literature to estimate emissions from these

processes are all based on AP-42 emission factor equations. The differences in the approaches were due to assumptions as to which processes the spreading and compacting of ash was most analogous to, and used the AP-42 factor developed for that process. MRI (1990a) used an AP-42 emission factor developed for dozer moving of overburden in western surface coal mines. This is given in EPA (1988d) as:

$$E_b = 0.34(s)^{1.5}/(M)^{1.4} \quad (5-3)$$

where:

E_b = emission factor for bulldozing of overburden (kg/hr)
 s = silt content of overburden (%)
 M = moisture content of overburden (%)

Kellermeyer and Ziemer (1989) assumed that the spreading and compaction of ash was analogous to vehicular transport on unpaved surfaces, and used the emission factor for that process (EPA, 1988d):

$$E_{up} = k[1.7(s/12)(V_s/48)(W/2.7)^{0.7}(nw/4)^{0.5}(365-P)/365] \quad (5-4)$$

where:

E_{up} = emission factor for unpaved surfaces (kg/VKt - vehicle kilometer traveled)
 k = particle size multiplier (dimensionless)
 s = silt content (%)
 V_s = mean vehicle speed (km/hr)
 W = mean vehicle weight (kg)
 nw = mean number of wheels
 P = number of days with at least 0.254 mm (0.01 inch) precipitation per year.

In applying this equation, Kellermeyer and Ziemer (1989) made the following assumptions:

- vehicle wt = 19 tons
- vehicle velocity = 2 mi/hr
- fraction of time on exposed ash surface = 65% of an 8 hr day
- number of wheels = 2 (tracked vehicle).

EPA (1988e) separated the processes of spreading and compacting. For compacting, they assumed the process of vehicular transport over unpaved surfaces. For spreading, they assumed that the process was analogous to agricultural tillage. That emission factor equation is (EPA, 1985a):

$$E_{at} = k(5.38)(s)^{0.6} \quad (5-5)$$

where:

E_{at} = emission factor for agricultural tillage (kg/ha)
 k = particle size multiplier (dimensionless)
 s = silt content (%).

The best approach to take is somewhat arbitrary, although some site-specific considerations might lead to a rational selection. Visual inspection of the process can indicate whether fugitive emissions are occurring at all; spreading and compacting of very moist ash may lead to no fugitive emissions. If portions of the disposal area are uncovered for long periods of time such that the ash can dry out, then any traffic associated with spreading and compacting over these areas can mimic vehicular traffic over unpaved surfaces. One rational approach might be to iteratively use combinations of all three equations presented above, so that potential emissions from spreading and compacting can be bounded by a sensitivity analysis approach. Another consideration is that, since equations 5-4 and 5-5 were not developed as a function of the moisture content of the ash, a wetness coefficient might be introduced to provide a more realistic estimate of emissions. Alternatively, if the ash is often, but not always, moist when spreading and compacting traffic occurs, the value of P in equation 5-4 can be set to a number near 365.

5.1.6. Wind Erosion

Equation 5-2, used above for estimating emissions from trucks, was developed for fugitive emissions from stationary piles (EPA, 1985a). This AP-42 emission factor was replaced by a more sophisticated emission factor (EPA, 1988d):

$$E_w = k \sum_{i=1}^N P_i \quad (5-6)$$

where:

E_{w2} = emission factor 2 for wind erosion from stationary piles, g/m²
 k = particle size multiplier (dimensionless)
 N = number of disturbances per year
 P_i = erosion potential corresponding to the observed (or probable) fastest mile of wind for the i th period between disturbances, g/m²

EPA (1988d) further presents equations to estimate P_i as a function of wind speed, friction velocity, and threshold friction velocity. Means to estimate these variables are non-trivial, and it is only applicable on a site-specific and event basis. Further details are supplied in EPA (1988d).

Another method of estimating fugitive dust emissions was developed in EPA (1985c) based on Gillette's (1981) field measurements of highly erodible soils. Application of this method to ash piles or

uncovered portions of disposal sites assumes that the exposed surface is uncrusted and consists of finely divided particles. Under these conditions, an essentially "unlimited reservoir" of ash exists. This may not be an appropriate assumption in all circumstances. For example, fly ash which has been through a lime scrubber is thought to take on a pozzolonic nature which may reduce wind erosion (a pozzolon is a siliceous material which reacts to the presence of moisture and alkali and alkaline earths to yield a cementitious product). Also, and like some of the emission factors above, there is no moisture term in this approach.

The flux of dust particles from an unlimited reservoir is given as (EPA, 1985c):

$$E = 0.036 (1-V)(U_m/U_t)^3 F(x) \quad (5-7)$$

where:

- E = total wind erosion flux of particles <10 μm ($\text{g}/\text{m}^2 \text{ hr}$)
- V = fraction of vegetation cover
- U_m = mean wind speed (m/s)
- U_t = threshold wind speed (m/s)
- $F(x)$ = a dimensionless ratio function.

EPA (1985c) provides details allowing for the application of this equation under a variety of circumstances. The following is offered as guidance specific to ash applications:

E: As developed in EPA (1985c), this approach is most appropriate for particles of grain size less than 10 μm (for more discussion on grain size, see section 5.1.8 below).

V: A value of 0.0 is appropriate for uncovered portions of storage piles of disposal sites.

U_m : Table 4-1 in EPA (1985d) gives mean annual wind speeds for key cities around the United States. Values range from 2.7 to 6.3 m/s .

U_t : The threshold wind velocity is the wind velocity seven meters above the ground surface that is needed to initiate erosion. Determining U_t according to the methodology in EPA (1985c) is a multi-step procedure:

Step 1. Determine wind erosion threshold friction velocity. This is a function of the aggregate size distribution of the erodible material. Figure 3-2 implies an average grain size of 1.3 mm for combined ash. Figure 3-4 in EPA (1985c) shows that a grain size of 1.3 mm translates to a threshold friction velocity of 72 cm/sec .

Step 2. Determine the roughness height. Figure 3-6 in EPA (1985c) graphically shows the roughness height for the range of possible conditions. Included in this range are a roughness height of 0.1 cm for natural snow, 1.0 cm for a plowed field, 4.0 cm for a wheat field, an up to 1000 cm for high rise buildings (30+ floors). For ash surfaces, 1.0 cm is suggested.

Step 3. Determine the ratio of wind speed at 7 m to friction velocity. Figure 4-1 in EPA (1985c) gives this as a function of roughness height. For a height of 1.0 cm, this ratio is 18.

Step 4. Solve for u_t . This is simply the product of the ratio given in step 3 and the friction velocity: $18 \times 72 \text{ cm/sec} = 13 \text{ m/s}$.

F(x): The value of x is given as $0.886(U_t/U_m)$. Assuming a mean annual wind velocity of 5.0 m/s, x is solved as 2.3. Figure 4-3 in EPA (1985c) gives $F(x)$ as a function of x , up to a value of x of 2.0. For values of x greater than 2.0, $F(x)$ is approximated as: $0.18(8x^3 + 12x)\exp(-x^2)$. With $x = 2.3$, $F(x)$ is 0.11.

Whether ash dries out sufficiently for wind erosion is dependent on many factors. The amount of time ash is exposed to wind and the wind speeds necessary to "initiate" wind erosion are critical issues, which were addressed in the Red Wing Ash disposal risk assessment (Northern States Power Company, 1987). *In situ* drying tests showed an average daily drying rate of 1.2% for exposed ash. The wind speeds required to initiate erosion were determined to vary with the moisture content of the ash. A 10-minute exposure to winds with an average speed of 17 mph can initiate wind erosion at 1% moisture content, while winds in excess of 22-24 mph are needed to initiate erosion of ash containing 9-19% moisture.

5.1.7. Vehicular Traffic in the Vicinity of Storage and Disposal Areas

Ash residues can build up on roadways that are near storage areas for ash within combustor facilities as well as on haul routes in landfills where ash is disposed. Such build-ups can result from the tire track-out of trucks hauling ash away or just after dumping it, from the dripping of excess moisture from haul trucks through "weep holes" or cracks in truck beds, from wind erosion of storage piles or uncovered disposal areas, and so on. Vehicular traffic on these roadways, which can be paved or unpaved, can stir up and resuspend dust and this dust represents a source of release of ash contaminants into the air.

This point of release of MWC residuals is distinct from all others described here in that it is "indirect"—the release is not directly from the ash piles that reside in containment, transport, or disposal points. Still, the quantities released into air and subject to downwind transport can be significant because of the potential for heavy vehicular traffic on these roadways other than trucks transporting ash. Common practices to contain dust along haul routes, particularly necessary for visibility on unpaved routes that are common in landfills, include vacuuming, wetting, sweeping, and chemical dust suppression. At some MWC facilities, storage areas for ash piles are partially housed, with an opening for trucks to load and transport ash. These housed areas provide protection against wind erosion and transport of fugitive emissions. Containment measures at disposal areas, such as compacting or wetting, also reduce the transport of fugitive ash onto nearby roadways. One key difficulty in estimating the quantity of contaminant release through this indirect pathway is in estimating the length of roadway in the vicinity

of the storage and disposal areas that is impacted. A second difficulty is estimating the concentration of contaminant on impacted dust. It would be less than that of the ash because it mixes with clean dust, and also because of dust containment on roadways and sanitary practices near ash containment areas. Sampling of dust along roadways is the best way to resolve these difficulties. Such sampling was done in MRI (1990a).

The emission factor equations for vehicular traffic on unpaved roads was given above in equation 5-4. For paved roads, the emission factor is given in EPA (1988e) as:

$$E_p = k(sL/12)^3 \quad (5-8)$$

where:

- E_p = emission factor from paved surfaces (kg/VKt) (vehicle kilometer traveled)
- k = particle size multiplier (dimensionless)
- sL = mass of silt-sized material per unit of paved road material (g/m²)

An application of these equations to estimate emission from paved and unpaved roads impacted by ash management and disposal, within MWC facilities and at disposal sites, used the following values for these parameters (sL and s values derived from sample collection at 6 MWC and associated disposal sites) (MRI, 1990a):

- mass of silt-sized material, sL , combustor site: 0.38-17 g/m² (median = 2.5, $n = 12$)
- mass of silt-sized material, s , disposal site: 0.19-10.0 g/m² ($n=2$)
- silt content, unpaved road: 6.7-20.1 % ($n=2$)
- vehicle velocity = 8 - 30 km/hr
- vehicle weight = 14 - 40 Mg
- number of wheels = 6 - 14

Application of these equations further requires estimates of the number of vehicle kilometers traveled. Two considerations are relevant for this estimate: the number of vehicles which traverse over a specific length of roadway assumed to contain fugitive dust, and this specified length. Both issues were examined in depth in MRI (1990a).

5.1.8. Application of AP-42 Emission Factor Equations

Site-specific considerations are critical for application of these equations. These include not only parameter values (silt content, moisture content, etc.) for the equations, but also additional reductions in estimated emissions because of control practices associated with ash management. The AP-42 emission

factor equations were developed with no controls in place. At MWC facilities and disposal sites, control measures are taken to reduce fugitive emissions. These were summarized in Section 4. Estimations using the emission factors must be reduced assuming some efficiency of control measures. This "control efficiency" reduction is demonstrated in Section 7.

Three additional considerations are relevant for application of the AP-42 emission factor equations. These considerations are summarized on Table 5-1 and discussed below.

One is the qualitative rating of A through E, with A being the best. They are subjective and reflect the quality and amount of data available in the development of the equations. The developers indicate that they should be considered as an indicator of the accuracy and precision of a given factor when used to estimate emissions from a large number of sources, and perhaps could be used to infer error bounds around estimates (EPA, 1985a).

The second consideration is the range of parameter values for which application of the equations is considered appropriate. These were derived from the specific values of the parameters from the sites studied for development of the equations. The ratings described above are only relevant for parameter values within these ranges. Application of the equations for parameter values outside the specified range would have a high uncertainty. The guidance given in EPA (1985a, 1988d) is to recommend they not be used, or, in some cases, reduce the rating by a letter.

The final consideration is the appropriate selection of the dimensionless particle size multiplier, k . This multiplier is defined according to particle size: calculated emissions are relevant for all particles that size and less. An important consideration in the development of the k values is that the fugitive emission process is only expected to generate emissions of particles that are as high as the highest k presented. For example, agriculture tillage k values are as given for "total suspended particulates", while paved road k are only defined for particle sizes less than 15 μm . From an exposure standpoint, selection of k values among the ones available can vary according to the intent of the exercise. If the objective is only to evaluate the risk due to inhalation of "inhalable fraction" size particles, then k values corresponding to particle sizes of 10 μm or less are appropriate. Emission estimations made with this strategy are commonly referred to as "PM-10" (particulate matter - 10 μm) emissions. If instead the objective is to model total emissions, perhaps to then incorporate these emissions into air models for downwind deposition onto soil, biota, or surface water, the highest k value presented would be appropriate.

Equation 5-7 is not an AP-42 factor equation; the only qualifier given in its development is that is appropriate for 10 μm size particles and less.

Table 5-1. Considerations for Application of AP-42 Emission Factor Equations^a

Equation/Description	Rating ^b	Parameter limits ^c	Particle Size Multiplier ^d	
Equation 5-1 Loading and Unloading	A	M: .25-4.8 U: 0.6-6.7	<30 µm	0.74
			<15 µm	0.48
			<10 µm	0.35
			<5 µm	0.20
			<2.5 µm	0.11
Equation 5-2 Truck transport ^e	C,D	None given	TSP	
Equation 5-3 Dozer overburden	B	s: 3.8-15.1 M: 2.2-16.8	≤15 µm	0.45
			≤10 µm	0.34
			≤2.5 µm	0.05
Equation 5-4 Unpaved roads	A	s: 4.3-20 Vs: 21-64 W: 2.7-142 nw: 4-13	≤30 µm	0.80
			≤15 µm	0.50
			≤10 µm	0.36
			≤5 µm	0.20
			≤2.5 µm	0.095
Equation 5-5 Agricultural tillage	A or B ^f	s: 1.7-88	TSP	1.0
			<30 µm	0.33
			<15 µm	0.25
			<10 µm	0.21
			<5 µm	0.15
			<2.5 µm	0.10
Equation 5-8 Paved roads	A	sL: 2-240 MVW ^g : 6-42	≤15 µm	0.28
			≤10 µm	0.22
			≤3.5 µm	0.081

^a Equations 5-6 and 5-7 are not included in this table; equation 5-6 is a complex event-oriented wind erosion AP-42 equation - refer to EPA (1988d) for more details; equation 5-7 is not an AP-42 equation

^b A - best rating, E - worst rating; See Section 5.1.8. for discussion of ratings

^c See appropriate sections of Section 5.1. for definition of and units of variables.

^d The particle size multiplier, k, is unitless; see Section 5.1.8. for discussion of k; TSP = total suspended particulates.

^e Equation 5-2 was developed for erosion from open piles and not truck transport; see Section 5.1.4. for discussion.

^f Rating is A if used to estimate total particulate emissions, and B if used for a specific particle size.

^g MVW = mean vehicle weight, Mg (metric tons); although equation 5-8 does not have vehicle weight in it, this was identified in EPA (1988d) as the range of vehicle weights for which the equation was developed, and for the which the A rating is retained.

5.2. MECHANISMS OF RELEASE: LEACHATE AND RUNOFF

The production of leachates and runoff of contaminated water at an ash disposal facility, or from reused ash, represents the second source of release of MWC ash components into the environment. Leachates and runoff can contaminate surrounding soils, surface waters, or groundwater, and eventually represent a pathway of exposure to humans.

Leachates from an ash landfill or storage site can potentially enter the ground water following loss of liner integrity. The total leachate generation from an ash landfill can be calculated as:

$$L = 0.01 Q [F_w(A_w) + F_c(A_c)] \quad (5-9)$$

where:

- L = leachate volume (m³/yr)
- Q = potential percolation (cm/yr)
- F_w = fraction of potential percolation, Q, which results from precipitation falling on the working (exposed) portion of the landfill
- A_w = working landfill area (m²)
- F = fraction of potential percolation, Q, which results from precipitation falling on the covered portion of the landfill
- A_c = covered landfill area (m²).

F_w and F_c have been conservatively estimated by Kellermeyer and Ziemer (1989) as being 1.0 (100% of the potential percolation for the uncovered portion of a landfill) and 0.15 (15% of the potential percolation for a covered landfill). Kellermeyer and Ziemer (1989) also conservatively assumed that Q (potential percolation) was equal to annual precipitation.

While more detailed models of percolation are available (e.g., Lu et al., 1985), a screening level estimate can be obtained by applying a water balance equation (EPA, 1985b):

$$Q = P + I - E \quad (5-10)$$

where:

- Q = annual percolation (cm)
- I = annual irrigation (cm)
- P = annual precipitation (cm)
- E = annual evapotranspiration (cm).

This model relies on estimates of evapotranspiration and precipitation. In their document that describes screening techniques for determining the presence of pollutants in groundwater or surface water,

the EPA Environmental Research Laboratory-Athens (EPA, 1985b) presents a map of the United States with gradients of mean annual potential evapotranspiration minus precipitation.

Equation 5-8 neglects surface runoff, and/or conservatively assumes it is zero. For uncovered portions of landfills, surface runoff can transport contaminants in exposed ash to surface water bodies. Methods to estimate site-specific and event-oriented surface runoff are described in EPA (1985c). Alternately, regional estimates of annual surface runoff can be obtained in the Water Atlas of the United States (Geraghty et al., 1973). Annual totals of runoff need to be converted to volume of runoff water:

$$RV = 0.01 R A_w \quad (5-11)$$

where:

- RV = runoff volume (m³/yr)
- R = annual runoff (cm/yr)
- A_w = working (exposed) landfill area (m²).

5.3. MECHANISMS OF RELEASE: DRAINAGE OF MOIST ASH

Drainage can occur in the management of very moist ash. The moisture content of ash at the MWC load-in point, the point of temporary storage after quenching where it is loaded onto trucks, was measured at between 8 and 42%, with a median of about 25% (7 samples from 6 MWC study sites; MRI, 1990a). Such high moisture contents can lead to drainage loss prior to loading, spillage when loading onto trucks, leakage from trucks, and so on. For example, one facility (studied in MRI, 1990a) discharged ash directly from the quench tank into dump trucks. Weep holes in the trucks were opened during loading to allow drainage of excess water, which was collected and recirculated to quench tanks. While this particular management minimizes environmental release, mismanagement or poor containment in trucks carrying very moist ash can result in leakage along the truck's route.

Leakage of water along a truck's route can be estimated as follows:

$$El = 10(MC_1 - MC_o)(TT/TD)C(1-E_r/100) \quad (5-12)$$

where:

- El = amount of leakage during a truck haul, L
- MC₁ = moisture content of ash at truck loading, % by volume
- MC_o = moisture content of ash below which free drainage will not occur, % by volume
- TT = truck travel time, hr
- TD = total time to drain from MC₁ to MC_o, hr
- C = truck capacity, m³
- E_r = efficiency of truck at restricting free drainage, %

Such an approach to estimating leakage from trucks has never been attempted, and hence should be used with caution.

The grain size distribution of ash as shown in Figure 3-2 implies an average grain size of 1.3 mm. This grain size would be classified as a "coarse sand" according to USDA soil classifications (Brady, 1984). Drainage from this type of soil is very rapid, in a matter of hours, and the higher moisture contents that were noted at the load-in in MRI (1990a), as high as 42%, are unlikely to be representative of moisture contents of ash when transport to a disposal site occurs, even when the ash is moist enough so that this mechanism of release should be considered. At the site studied by MRI (1990a), where ash was directly discharged from the quench tank into the trucks, this mechanism of release might be considered, but even there the truck filling and drainage from the weep holes took between 12 and 16 hours. Another consideration is that very moist ash is very difficult to handle. It is probably not unreasonable that an MC_i of 20-30% is the highest that should be considered for this mechanism of release.

The comparison of ash grain size to that of a coarse sand provides a means to estimate MC_o . The "field capacity" of coarse sand, or moisture content at and below which free drainage is restricted, is in the neighborhood of 5% (Brady, 1984). This is a reasonable value for MC_o . The assignment of a time to drain from MC_i to MC_o , TD, can be thought of as the time it would take to drain if the truck weep holes were open, and no resistance, other than the truck bed, were offered to free drainage. The truck bed or any impermeable surface offers resistance, however. This resistance can be understood by visualizing the drainage that would occur if very moist ash (or very moist soil) were suspended in air by a fine wire mesh. Selection of TD is speculative, but a value of 24 hours might not be unreasonable.

Totally open weep holes in transit implies an E_i of 0%, whereas an airtight truck would have 100% efficiency. A truck reasonably airtight, but through which dripping can occur, might have an 80 or 90% efficiency, whereas a truck with cracks allowing more than dripping might have an efficiency less than 50%.

5.4. ESTIMATING ENVIRONMENTAL CONCENTRATIONS

The preceding discussion has focused on methods used to estimate the volume of leachate or runoff water, or the mass of fugitive ash emissions. It should be recognized that the mass release of the contaminant of concern is a simple product of the volume of leachate or runoff, or mass of ash, containing the contaminant and the concentration of the contaminant in the matrix of concern. The best way to ascertain this concentration is through actual sampling of ash or leachate water as part of a site-specific assessment. If this is not possible, however, simple assumptions must be made. It can be assumed, for example, that the concentration of the contaminant in the leachate or runoff water is equal to concentrations from laboratory leachate testing. Much information also exists concerning the

concentrations of contaminants on ash (Section 3). It can be further assumed, as a first approximation, that the concentration of contaminants in ash or leachate water is not affected by degradation processes. However, it should be recognized that these assumptions are likely to result in conservative estimates of the concentration of most contaminants in ash or leachate water. Likewise, the ultimate fate of the contaminant includes chemical and physical transformations, dilution, and other processes reducing the concentrations estimated at the point of release.

5.5. MECHANISMS OF RELEASE: INTERMEDIA TRANSPORT

Intermedia transfers refer to the ability of some substances to transfer from one environmental medium to another. Some common intermedia transfer processes have already been discussed. For example, fugitive dust generation from ash waste piles transfers dust from the ground to the air. Leachate generation and surface runoff may result in transfers of ash constituents from the soil to surface water or groundwater. An additional mechanism by which intermedia transfer can occur is through the uptake by biota of ash contaminants in air, water, soil, or other biota through direct contact, ingestion, or inhalation pathways. These processes are demonstrated in greater detail in Figures 5-1 to 5-4.

Intermedia transfer mechanisms are an important factor to consider when estimating media specific contaminant concentrations. The intermedia transport parameters that may be of concern when performing an exposure assessment for MWC residuals include:

- resuspension of dust into the atmosphere;
- dust deposition into aquatic receptors, on soil, or on plants;
- leaching of contaminants from the soil or disposal site to groundwater, or transport to surface water bodies; and
- uptake of ash components by flora, aquatic biota, wildlife, or livestock followed by bioaccumulation or bioconcentration.

As noted in the above list, particularly in the last bullet, an endpoint of concern is "biota" that can be ingested by humans. This endpoint is also noted in Figures 5-1 to 5-4 in the branches titled, "Go to biota fate/food chain analysis". Most multi-media exposure assessments estimate contaminant concentrations in biota as a simple product of two factors: the concentration of the contaminant in the media in contact with the biota and an empirical factor such as a "bioconcentration factor." Alternately, there are complex food chain models (e.g., for fish), deposition models, and plant uptake models. Methodologies for estimating biota concentrations appropriate for the screening level approach of this document are given in EPA (1990b).

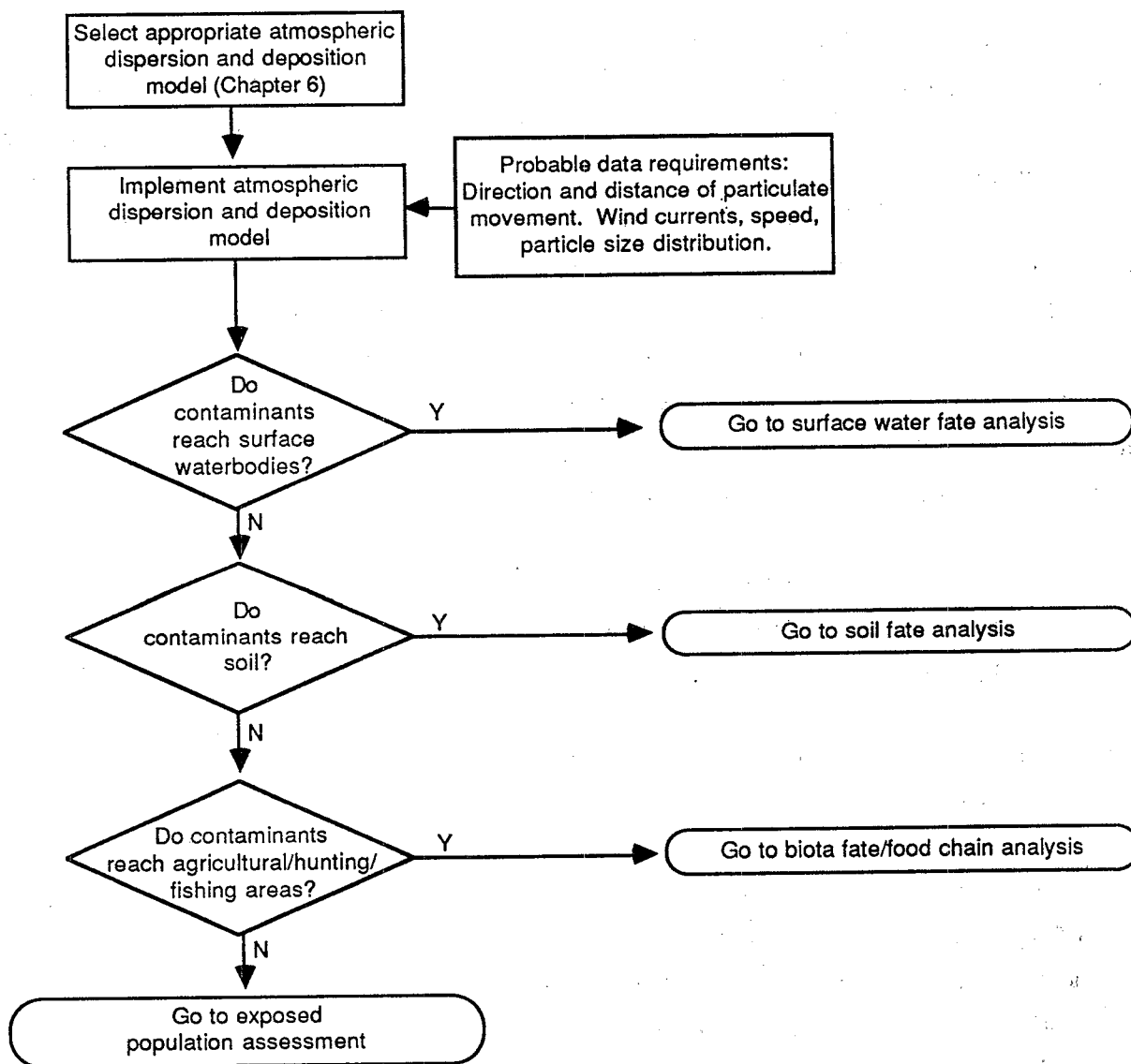
Figure 5-1 Atmospheric Fate Analysis for MWC Ash Exposure

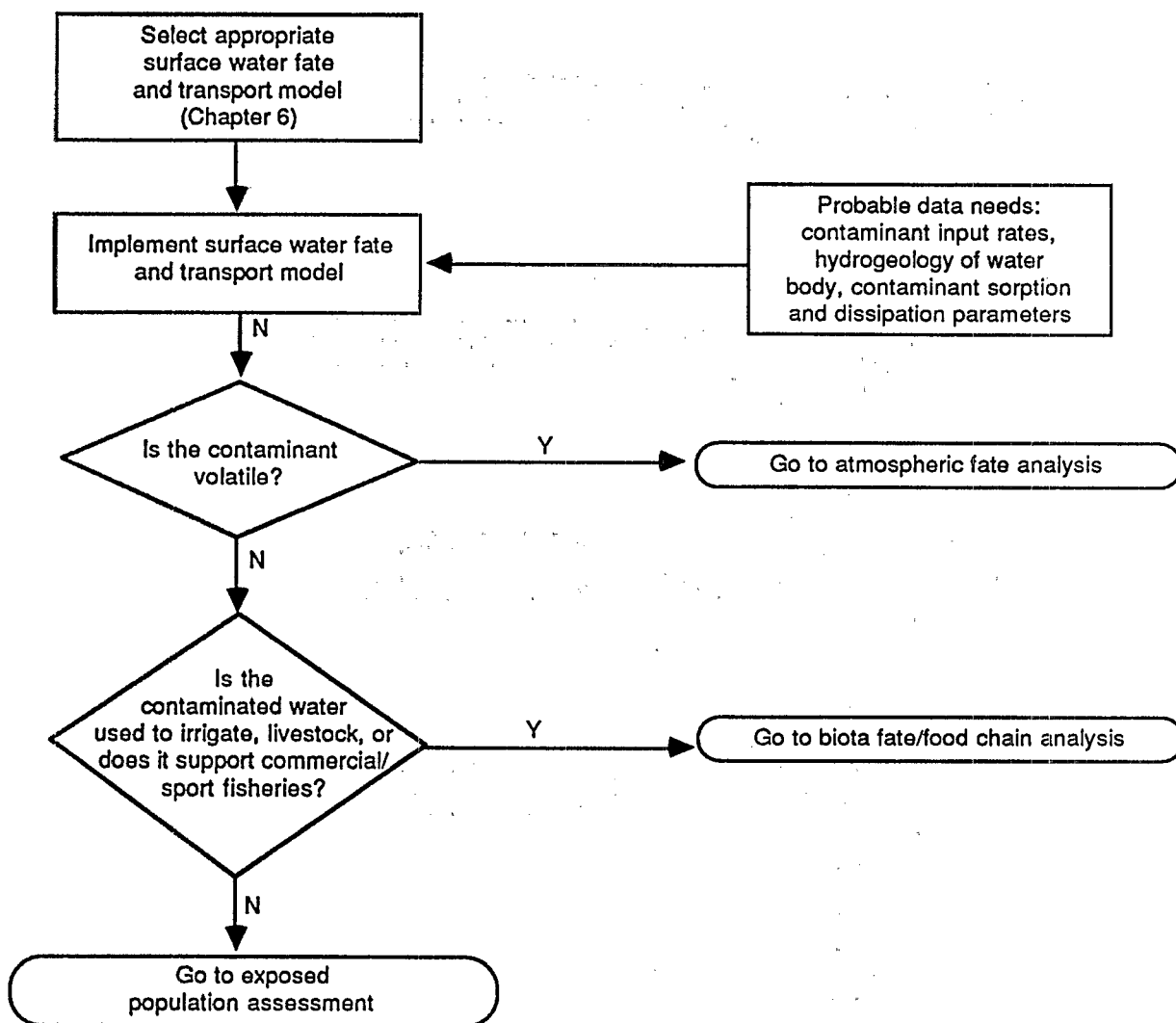
Figure 5-2 Surface Water Fate Analysis for MWC Ash Exposure

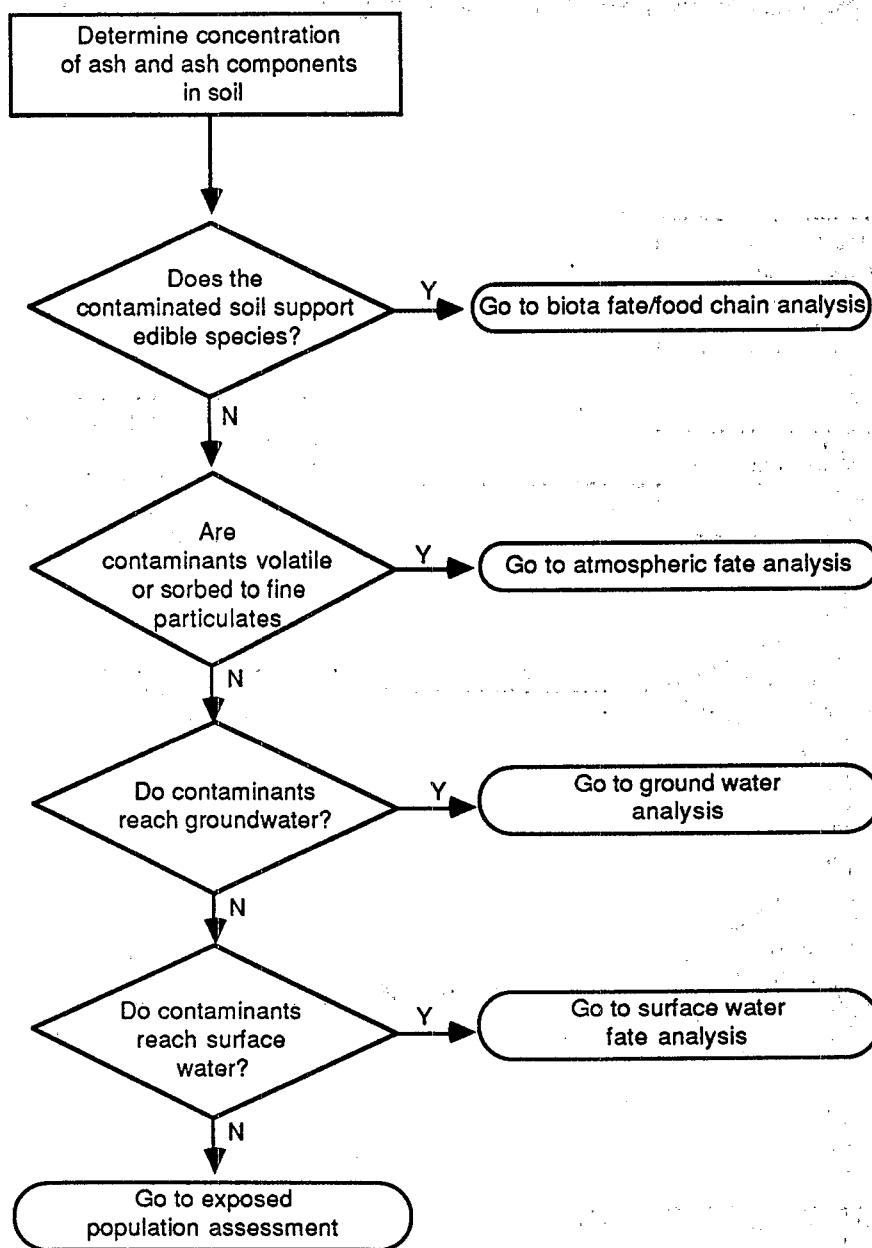
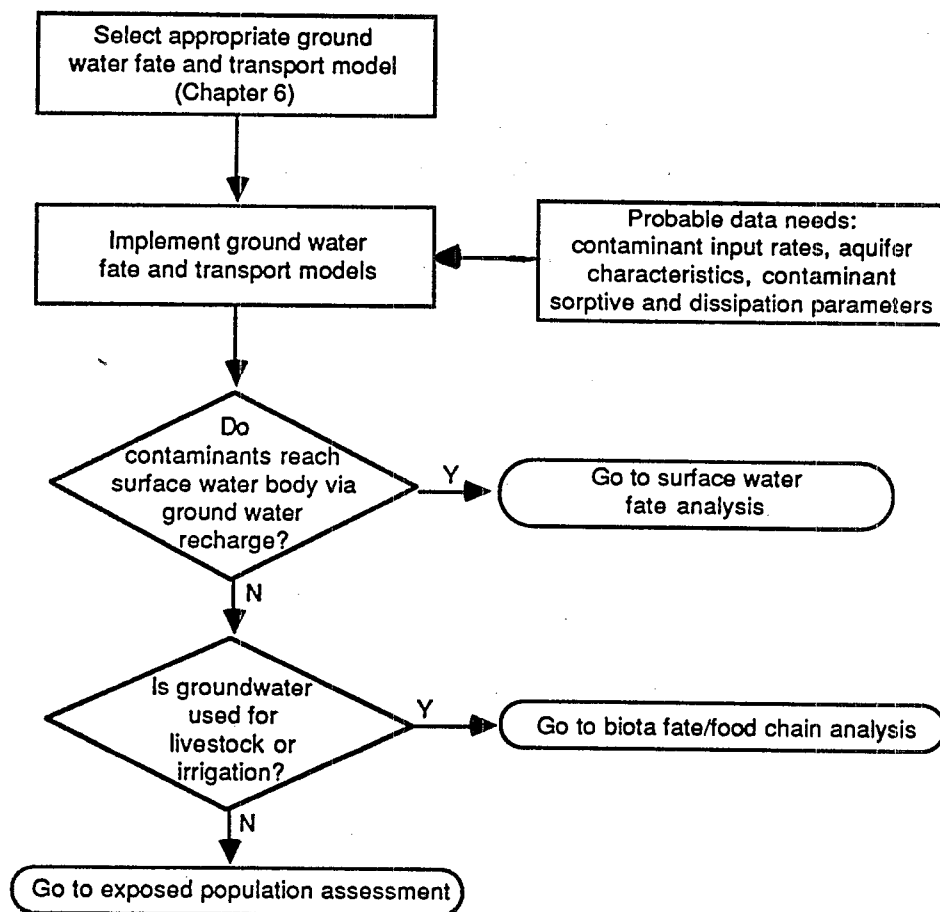
Figure 5-3 Soils Fate Analysis for MWC Ash Exposure

Figure 5-4 Groundwater Fate Analysis for MWC Ash Exposure

6. EXPOSURE PATHWAY AND EXPOSED POPULATION ANALYSES

An exposure pathway and exposed population analysis consists of a quantitative and qualitative description of exposure points, the routes of exposure at each point, and the population facing exposure from each route. This analysis is generally implemented following the determination and description of the potential releases of contaminants to the various environmental media.

The potential for fugitive releases of ash dust into the air can occur as a result of loading and unloading activities, vehicular resuspension near storage, handling, and disposal areas, transport, spreading and compaction, and wind erosion at a landfill. As fugitive ash enters the atmosphere, its concentration is diluted by meteorological mixing processes. In addition, some constituents can react physically or chemically with other airborne pollutants. Finally, the ash can be deposited on the ground by fallout or rainout.

A number of computer models are available to EPA for simulating the atmospheric dispersion of MWC residues. The models vary in sophistication and in their ability to incorporate simulations of certain processes important to determining the atmospheric fate of ash. The models recommended by EPA are evaluated and described in two documents:

- EPA, 1988b. *Air Dispersion Modeling as Applied to Hazardous Waste Incinerator Evaluations, An Introduction for the Permit Writer*. OSW, Waste Treatment Branch
- EPA, 1986b. *Guidelines on Air Quality Models*. (EPA-450/2-78-207R).

Releases of ash constituent chemicals into groundwater can follow the disposal of ash in a landfill, deposition of ash on a land surface, or reuse of ash in concrete or road material. Leachates of ash chemicals from MWC monofills or co-disposal facilities can enter the environment if there is no liner, following catastrophic liner failure, or through a liner leakage extending over a long period of time. Mismanagement of leachate collected from a landfill can potentially impact soil or ground water. Leachates can reach the groundwater or contaminate the soil following deposition of fugitive ash on the ground, a process that might occur along a transport corridor, or in the vicinity of storage or uncovered (or partially covered) disposal areas.

Leachates reaching groundwater can be diluted by mixing processes. Metals in leachates may adsorb to sediment, or speciate into forms that exhibit differing transport or toxicological properties. Organic compounds existing in the ash leachate are also prone to adsorption onto the soil matrix of the aquifer, and to undergo transformation or degradation into different compounds.

There exists an abundance of models for simulating groundwater contaminant transport. Selection criteria and a summary of these models is available in:

- EPA, 1988f. *Selection Criteria for Mathematical Models Used in Exposure Assessments: Ground-Water Models.* (EPA/600/8-88/075).

Surface water bodies may be contaminated by MWC ash following direct deposition of fugitive dust into a body of water, surface runoff of ash constituents (e.g., leachate), mismanagement of quench water or collected leachate water, or influx of contaminated groundwater. As ash constituent chemicals enter a surface water body, their concentration may be diluted by mixing processes. Furthermore, ash particulates and constituents may be influenced by various physical, chemical, and biological processes that take place in the particular water-body. Exposure to ash-contaminated surface waters is of concern if the water is used recreationally or for household water supplies, or supports fish that are consumed.

When surface bodies of water are contaminated by ash, it may be necessary to apply an appropriate aquatic fate and transport model in order to estimate exposure point concentrations. A number of models are available for this purpose. Selection criteria and descriptions of surface water contaminant transportation models can be found in:

- EPA, 1987c. *Selection Criteria for Mathematical Models Used in Exposure Assessments: Surface Water Models.* (EPA/600/8-87/042).

In all of the possible situations that result in the contamination of environmental media by MWC ash and residuals, three significant direct contributions to exposure exist. These are:

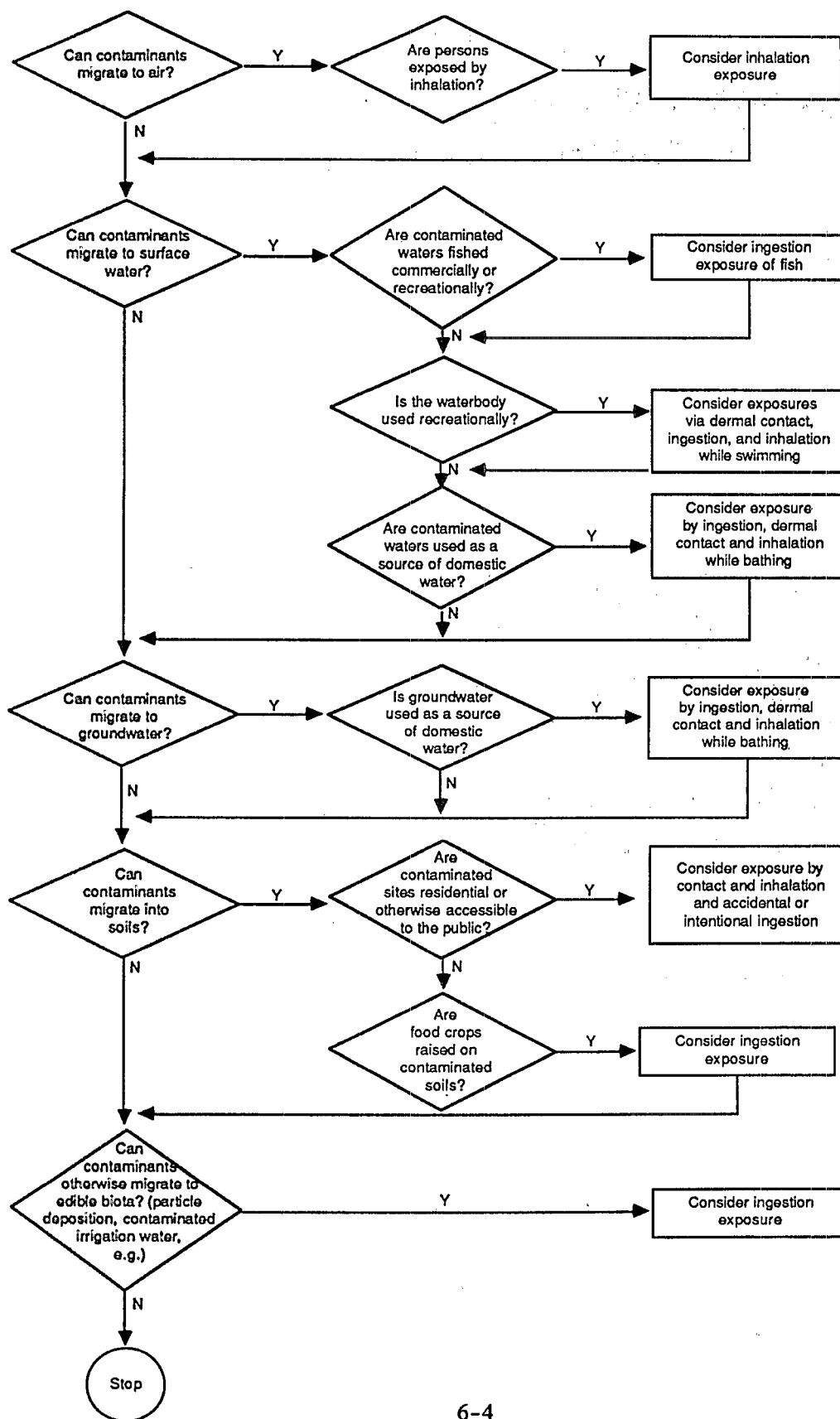
- inhalation of air contaminated by dust particles,
- skin absorption through contact with contaminated soil or contaminated water while swimming or bathing, and
- ingestion of contaminated soil (as dust or garden soil, e.g.) or water, or ingestion of food which is contaminated directly or indirectly through contact with ash components deposited from contaminated air or water.

The analysis of exposed populations requires that environmental contamination data be linked with population data. A quantitative analysis of exposed populations consists of an identification, enumeration, and characterization of exposed populations. The goal of this analysis is to determine the likelihood of human contact with the ash or ash constituents through one of the pathways outlined above. The decision network for performing an exposed population analysis is illustrated in Figure 6-1.

Guidance for performing an exposed population analysis and calculating exposure through each possible exposure pathways is provided in a number of EPA documents:

- EPA, 1989b. *Risk Assessment Guidance for Superfund. Human Health Evaluation Manual, Part A.* (9285.701A)
- EPA, 1988g. *Superfund Exposure Assessment Manual.* (EPA/540/1-88/001)
- EPA, 1990c. *Interim Guidance for Dermal Exposure Assessment*
- EPA, 1989c. *Exposure Factors Handbook* (EPA/600/8-89/043).

Figure 6-1 Exposed Populations - Identification of Relevant Exposure Routes



7. DEMONSTRATION OF METHODOLOGY

The purpose of this section is to demonstrate the methodology described in this document. This demonstration occurs in the context of a comprehensive example where all mechanisms of release are described and key parameters quantified. The example uses two contaminants, cadmium and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), to illustrate the methodologies. The selection of these two was made in order to demonstrate the methodology on an organic and an inorganic contaminant found in ash. Also, the related OHEA document, *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*, used the same approach by demonstrating its methodology on cadmium and a different organic compound, benzo(a)pyrene.

7.1. QUANTIFICATION OF RELEASE RATES OF MWC RESIDUALS

The first step in a comprehensive exposure/risk assessment is the quantification of rates at which MWC residuals are released into environmental receptor media (air, soil, water). Knowledge of the concentration of specific chemical contaminants in these ash releases enables the exposure assessor to estimate release rates for these contaminants. Once deposited in the environmental media, the exposure assessor then needs to consider further fate and transport of ash contaminants until they reach a point where populations are exposed: in the air they breathe, water they drink, food (or soil) they ingest, and so on.

It is important to understand that the example presented in this section is not meant to be an exposure assessment itself, or the beginnings of an exposure assessment. It is presented for the purpose of demonstration only. Assumptions and parameter values are justified where possible, otherwise they are assigned "in the absence of better information." Users of this methodology are encouraged to base their exposure assessments on data from specific sites they are evaluating, including any analysis of ash or leachate samples to estimate site-specific contaminant concentrations.

That being said, it is also fair to say that the key scenario definitions, amount of ash generated and contaminant concentrations, are high in comparison to industry averages. On the other hand, assumptions and parameters in the equations estimating release rates were determined from real and reasonable sources when possible. For example, "control efficiencies," or the parameters reducing maximum possible emissions because of control practices associated with a release, were high if information indicated that control practices were common. Key assumptions for this framework include:

- **Ash generation:** The hypothetical facility produces 1.82×10^8 kg, or 200,000 tons, ash per year. This is less than the maximum noted in the OSW survey of 287,000 tons (EPA, 1989a), but greater than

the 95th percentile tonnage generated of 137,679 tons/year (as statistically evaluated from the OSW survey in MRI, 1990a).

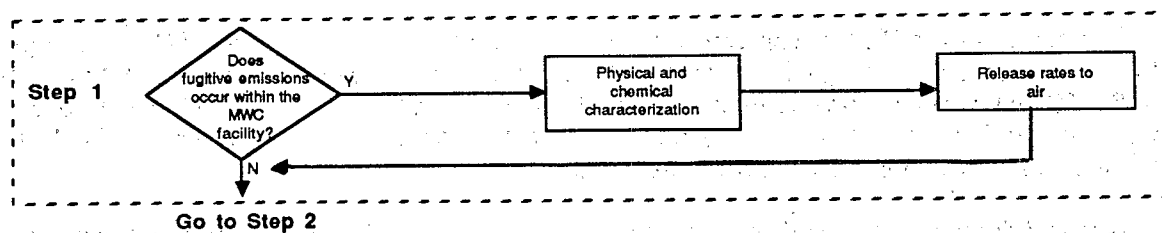
- **Cadmium and TCDD concentrations:** The concentration of cadmium and TCDD in fugitive ash emissions is assumed to be equal to the upper-bound reported concentrations of Cd, 100 µg/g, and TCDD, 7.8×10^{-4} µg/g, in combined bottom and fly ash. The concentration in water releases is assumed to be equal to the EP-Tox limit for Cd, 1000 µg/L, and the upper-bound reported concentration for TCDD, 2.3×10^{-4} µg/L, in extract from combined ash using the TCLP.

- **General framework:** Ash is temporarily stored at the MWC facility before it is trucked to an industrial solid waste (ISW) landfill located 32 miles from the facility. Transport in 20 truckloads per day is required to dispose of the ash, and this transport occurs 260 days/year (landfills are not open 365 days per year, although ash is generated 365 days/yr). The assumption of disposal in an industrial solid waste landfill is made for two reasons: it justifies the somewhat long trip of 32 miles (an ash monofill is likely to be much closer to the MWC facility; travel distances have been noted as high as 700 miles (EPA, 1989a)), and the size of an ISW landfill is typically larger than an ash monofill. This impacts one key point of release: resuspension of contaminated dust along the haul route in the vicinity of ash disposal. Length of an ISW landfill haul route and the daily vehicle transactions are both higher for an ISW in comparison to an ash monofill (MRI, 1990a). The total area in the landfill which contains ash is 8.25 hectares, although ash is only being disposed of on 0.25 ha at any time. The area not worked on is assumed to be capped, which eliminates wind erosion and surface runoff, and reduces leachate losses. When estimating fugitive ash emissions into air, the conservative assumption of 0.25% ash moisture content is made. In estimating the potential loss in leaking from the trucks in transit, the conservative estimate of 20% moisture content is made.

All other assumptions are detailed in their appropriate context.

The source analysis decision process for MWC residuals was described in Figure 4-1, and is repeated in this chapter (Figures 7-1 through 7-8) to facilitate discussion of the example. Portions of this figure are repeated as the example progresses through the decision framework. It is noted that at the beginning of each "step", and at places within steps, a possible "No emissions will occur..." decision sends the assessor to the next step. The methodology described in this document does not address how to determine whether or not an emission is likely to occur. As pointed out in parts of this example, however, quite often visual inspection of the specific site being assessed under normal conditions of operation can be informative as to whether an emission will occur or not.

Figure 7-1
Source Analysis for MWC Residuals Release - Step 1



STEP 1: DO FUGITIVE EMISSIONS OCCUR WITHIN THE MWC FACILITY?

RATIONALE

Anywhere ash moves within a MWC facility, such as along hoppers and conveyors, the possibility exists for emissions of small particulates. Dust films on surfaces are an indication that such emissions are occurring. Without complete combustion, volatile organics or metals, such as mercury, can also be emitted into the air. This document has not provided guidance on quantifying potential releases within a MWC facility. However, such releases can result in exposures to MWC workers.

NO, FUGITIVE EMISSIONS DO NOT OCCUR WITHIN THE MWC FACILITY.

Go to Step 2.

YES, FUGITIVE EMISSIONS OCCUR WITHIN THE MWC FACILITY.

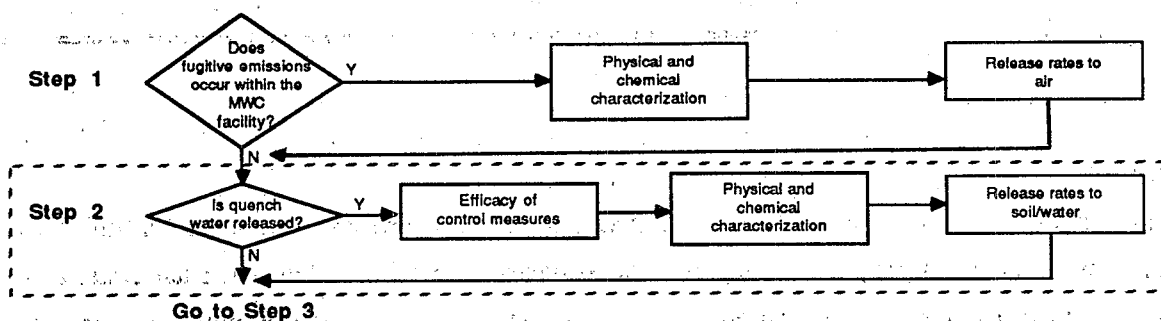
The following data must be obtained:

- Physical and chemical characterization of residuals at the point of escape,
- Release rates and frequency.

Fugitive emissions of vapors or particulates will not only introduce these contaminants into the air, but may also result in deposition of the compounds on surfaces. It is generally not necessary to use fate and transport models in this step, since the releases occur at the exposure point within the facility; however, it is necessary to identify relevant exposure routes (see Figure 6-1) and to quantify exposures.

Go to Step 2.

Figure 7-2
Source Analysis for MWC Residuals Release - Step 2



STEP 2: IS QUENCH WATER RELEASED?

RATIONALE

A quench water system used in a MWC serves two purposes. It provides a seal to the furnace, and quickly cools and wets the bottom (and/or mixed ash) prior to its transfer to a conveying system and then to be either temporarily stored or transported to a disposal area. The wetting of ash tends to suppress fugitive dust emissions. Sudden quenching also tends to alter the physical characteristics of ash: it tends to break up large masses into smaller particles. With the subsequent increase in surface area, there is the greater potential for desorption of contaminants from ash particles, and hence a greater potential for leaching of various chemicals. Quench water can be treated on-site, recycled, or otherwise managed so as not to be released to the environment. Mismanagement can result in environmental release.

NO, QUENCH WATER IS NOT RELEASED.

Go to Step 3.

YES, QUENCH WATER IS RELEASED.

Step 2a: Determine the following:

- Physical and chemical characteristics of the quench water.
- Release rates and frequencies.

The mean volume of quench water generated by a facility is 3.64×10^7 gallons per year (1.37×10^8 liters/year) (EPA, 1989a). Information is generally unavailable regarding the disposal methods for quench water. However, one might assume that facilities that collect and/or treat leachates will also handle quench water in a similar way. In the EPA (1988c) survey, it was found that 77.8% of the operators implemented some means of runoff collection and treatment, while no such measures were identified in 22.2 % of the cases.

Assumptions for the scenario:

- Volume of quench water generated by a facility = 1.37×10^8 L/year (EPA, 1989a).
- It will be assumed that some method of containment of quench water (recirculation, on-site treatment, etc.) is in place such that not all quench water generated is released to the environment. Based on the survey results noted above, it will be assumed that 25% of that generated gets released to the environment.

The volume of quench water = 1.37×10^8 L/yr \times 0.25 = 3.43×10^7 L/yr

Mass loadings to the environment of contaminant in the scenario from the release of untreated quench water:

The concentration of the example contaminants in the quench water is assumed to be equal to the EP-Tox limit for Cd (1000 μ g/L) and upper-bound reported concentration for TCDD in extract from combined ash using the TCLP (2.3×10^4 μ g/L).

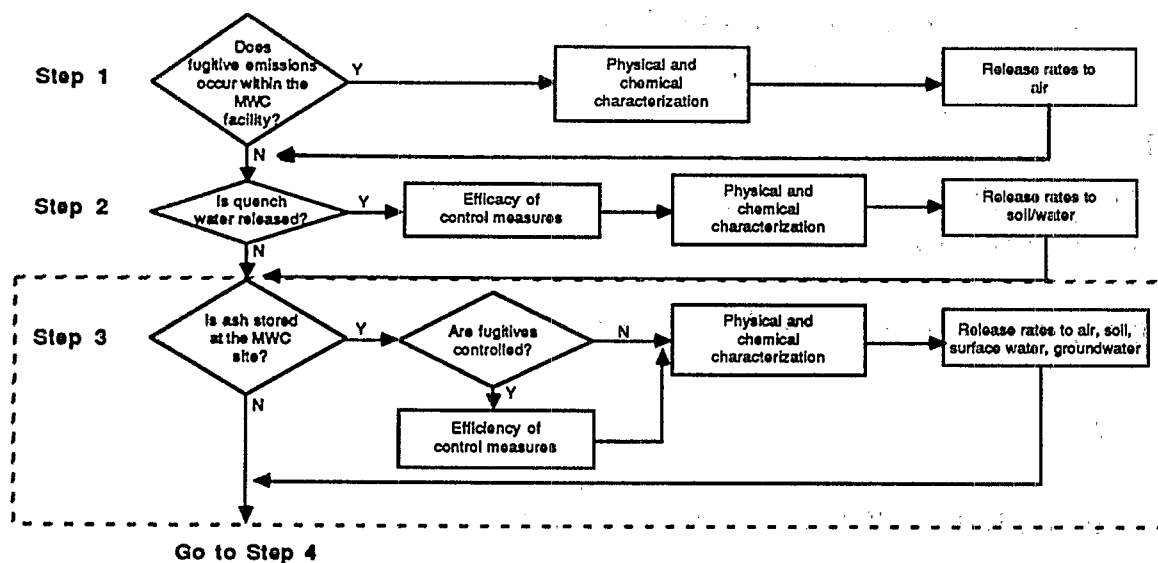
Cd: 3.43×10^7 L of quench water released/year \times 1000 μ g Cd/L of leachate = 3.43×10^{10} μ g/year

TCDD: 3.43×10^7 L of quench water released/year \times 2.3×10^4 μ g TCDD/L of leachate = 7.88×10^3 μ g/year.

Since the environmental release of quench water can occur into groundwater, soil, and surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3) and identify relevant exposure routes (Figure 6-1).

Go to Step 3.

Figure 7-3
Source Analysis for MWC Residuals Release - Step 3



STEP 3: IS THE ASH STORED AT THE MWC SITE?

RATIONALE

Some facilities store ash on-site prior to transport to a disposal facility. These storage areas can act as sources of runoff and of fugitive emissions.

NO, ASH IS NOT STORED AT THE MWC SITE.

Go to Step 4.

YES, ASH IS STORED AT THE MWC SITE.

On-site storage occurs at approximately 46% of MWC facilities (EPA, 1989a). An average of 380 tons (3.5×10^5 kg) are stored at each such site.

Step 3a: Are fugitive emission control measures used at the ash storage site?

NO, FUGITIVE EMISSION CONTROL MEASURES ARE NOT USED.

Assume a control efficiency of 0%.

Go to Step 3a(i).

YES, FUGITIVE EMISSION CONTROL MEASURE ARE USED.

The OSW survey (EPA, 1989a) revealed that 77% of MWC operators that store ash on site also use dust containment measures. The effectiveness of such measures varies. Some measures, such as wetting, may be highly effective in suppressing fugitive dust emissions; others, such as use of recirculation/capture systems, may be somewhat less effective. Of the operators using dust containment measures, 51.7% wet the ash or use a combination of wetting and covered conveyors (EPA, 1989a).

The efficiency of fugitive dust controls must be used when calculating release rates from the storage site. This can be done by calculating a control efficiency index:

- If wetting is used all the time, and the amount of wetting is judged sufficient to suppress fugitive emissions, assume a control efficiency of 100%.
- By way of example, and in the absence of other information, a control efficiency of 40% is assumed.

Control efficiency = 0.40; $[100(1-0.40) = 60\% \text{ release}]$

Go to 3a(i).

Step 3a(i): Determine the following:

- Physical and chemical characteristics of the ash.

Possible changes in the physical and chemical character of ashes depend on the exact conditions and duration of storage. Storage can lead to a reduction in the moisture content of the ash, to the formation of a surface crust, and to the cementation of particles into larger masses. All these changes may affect the fugitivity and erodibility of ash particles.

- Release rates and frequency.

Wind generated releases of ash from a storage site can be calculated using equation 5-7:

$$E = 0.036 (1-V)(U_m/U_t)^3 F(x)$$

where:

E = total flux of particles $<10 \mu\text{m}$ ($\text{g}/\text{m}^2 \text{ hr}$)

V = fraction of vegetation cover

U_m = mean wind speed (m/s)

U_t = threshold wind speed (m/s)

$F(x)$ = a dimensionless ratio function.

The following assumptions and default values are used in this example scenario to estimate wind-generated release rates of ash from a storage site.

Assumptions for the scenario:

Application of equation 5.7 was described in section 5.1.6., including recommendations for parameter values. Those parameter values will be used in this example, and they are:

V: No vegetative cover, or $V = 0.0$

Um: A mean annual wind speed of 5.0 m/s was used in the discussion in section 5.1.6.

Ut: This was solved for as 13.0 m/s

F(x): Given Ut and Um, this was solved for as 0.11

The unit emission solved for by equation 5.7 is units of $\text{g/m}^2 \text{ hr}$. To convert to an annual basis, the following assumptions are made:

Exposed Area: The exposed surface area for an ash storage site at a MWC facility must be determined on a site-specific basis. This illustration will use a conservative assumption of an area equal to 0.25 hectare, or 2500 m^2 .

Hours: For this example, it will be assumed that the ash is stored prior to loading onto trucks for transport to the landfill. It will be assumed that the storage area is covered during non-working hours. Since transport occurs during 260 work-days/yr, the number of hours per year equals: $260 \text{ wd/yr} \times 8 \text{ hr/wd} = 2080 \text{ hrs}$.

Using these assumptions, one can calculate the wind generated release of ash from a storage site (E) as follows:

$$E = 0.036 (1-V)(U_m/U_t)^3 F(x)$$

$$E = 0.036 (5/13)^3 (0.11)$$

$$E = 2.25 \times 10^{-4} \text{ g/m}^2\text{-hr}$$

For an area of 2500 m^2 and for 2060 hours/yr, the annual emission equals: $2.25 \times 10^{-4} \times 2500 \times 2060 = 1158 \text{ g/yr}$

Step 3a(ii): Apply the control efficiency factor

Determine net fugitive releases in the presence of controlling factors (E_{net}) using the following equation:

$$E_{net} = E [1 - (\text{control efficiency}/100)]$$

The control efficiency index in our scenario is assumed to be 0.6. Note that the storage area was assumed to be covered during non-working hours. Therefore, this efficiency can be interpreted as the efficiency of other measures not captured in the estimation, for example, wetting of ash in storage areas. The net fugitive release of ash from the storage site is:

$$1158 \text{ g/yr} \times 0.6 = 695 \text{ g/yr}$$

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions at the storage site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3-1 and 3-3).

Cd: 695 g fugitive ash released/year \times 100 µg Cd/g ash = 6.95×10^4 µg Cd released/year

TCDD: 695 g fugitive ash released/year \times 7.8×10^{-4} µg TCDD/g ash = .5 µg TCDD released/year.

Go to Step 3b.

Step 3b: Are Leachate or Runoff Control Measures Used?

YES, LEACHATE OR RUNOFF CONTROL MEASURES ARE USED.

Runoff containment measures are used by approximately 75% of MWC operators that store ash on the premises (EPA, 1989a). These measures include drainage and release to sanitary sewers, wastewater treatment, or storage.

The efficiency of the control measures must be included in any calculation of leachate or runoff release into the environment. This can be accomplished by using a control efficiency factor.

- If runoff and/or leachate containment measures are used, and if it is reasonable to assume that the measures are effective, assume a control efficiency of 100%.

Go to Step 3c.

If it is not possible to determine if effective containment measures are used, some efficiency must be assumed. For the sake of this example, it will be assumed that the containment efficiency equals the frequency of use of leachate control measures, i.e. 75%.

Go to Step 3b(i).

NO, LEACHATE OR RUNOFF CONTROL MEASURES ARE NOT USED.

- Assume a control efficiency of 0%.

Go to Step 3b(i).

Step 3b(i): Determine the following:

- Physical and chemical characteristics of the runoff or leachate.
- Leachate generation rate.

The rate of leachate or runoff water must be estimated; doing so requires an assessment of the leaching and/or runoff potential of the soil within the storage area. If the storage area is in an area that contains generally sandy soils, than excess water can be assumed to percolate; if otherwise the soil is a clay soil with poor drainage, much of the excess water will run off the storage area. For this example, it is

assumed that the soil has good drainage, and therefore, equation 5-10 will be used to estimate the amount of percolation:

$$Q = P + I - E$$

where:

- Q = annual percolation (cm)
- I = annual irrigation (cm)
- P = annual precipitation (cm)
- E = annual evapotranspiration (cm)

Assumption for the scenario:

Irrigation: Irrigation will not be considered to exist at the landfill site, therefore I will be set at 0.

Precipitation and Evapotranspiration. EPA (1985c) presents a map showing gradients of mean annual potential evapotranspiration minus precipitation. These values range from +70 inches in the desert regions of the southwest to -50 inches in the Pacific Northwest. A value similar to those found on the East Coast will be used: i.e. -20 inches = -50.8 cm.

The total leachate generated from the storage area can then be calculated from equation 5-9:

$$L = 0.01 Q [F_w(A_w) + F_c(A_c)]$$

where:

- L = leachate volume (m³/yr)
- Q = potential percolation (cm/yr)
- F_w = fraction of potential percolation, Q, which results from precipitation falling on the working (exposed) portion of the landfill
- A_w = working landfill area (m²)
- F_c = fraction of potential percolation, Q, which results from precipitation falling on the covered portion of the landfill
- A_c = covered landfill area (m²).

Assumption for the scenario:

Percolation: The value for mean annual potential evapotranspiration minus precipitation, from equation 5.10, will be used: 50.8 cm/year.

Exposed Area: In this simple example, the storage area is either entirely covered or entirely bare. Therefore, as a surrogate for exposed area, we can multiply the total area by the fraction of time it is exposed, or $2500 \times (2060 / (365 \times 24))$, or 588 m².

Covered Landfill Area: This is then equal to the total area minus the surrogate above for exposed area: $2500 - 588 = 1912$ m².

Working Area Percolation: Similar to Kellermeyer and Ziemer, 1989, the conservative assumption of 1.0 is assumed.

Exposed Area Percolation: The cover cannot be assumed to be 100% effective, or $F_c = 0.0$. Some leakage through a canvas tarpaulin, if that were used as a cover, or sideways water intrusion is to be expected. Similar to Kellermeyer and Ziemer, 1989, a value of 0.15 is assumed for F_c .

Calculation of leachate generation:

$$L = 0.01 Q [F_w(A_w) + F_c(A_c)]$$

$$L = 0.01 (50.8 \text{ cm/yr}) [1.0(588 \text{ m}^2) + 0.15(1912 \text{ m}^2)]$$

$$L = 444 \text{ m}^3 \text{ per year or } 4.44 \times 10^5 \text{ liters/year.}$$

Step 3b(ii): Apply the containment control efficiency factor to the release rate to determine net

The assumption was earlier made that the containment is 75% efficient. Therefore, 25% is assumed to be available for release. The net volume of leachate produced at this site (L_{net}), can be calculated by:

$$L_{\text{net}} = 0.25 (L) = 0.25 \times 4.44 \times 10^5 \text{ liters/year} = 1.11 \times 10^5 \text{ liters/year.}$$

Mass loadings into the environment of contaminants in the scenario from leachate release at the storage site:

The concentration of the example contaminants in the leachate is assumed to be equal to the EP-Tox limit for Cd (1000 µg/L) and the upper-bound reported concentration for TCDD in extract from combined ash using the TCLP (2.3×10^{-4} µg/L).

Cd: 1.11×10^5 L of leachate released/year \times 1000 µg/L of leachate = 1.11×10^8 µg/year.
TCDD: 1.11×10^5 L of leachate released/year \times 2.3×10^{-4} µg TCDD/L of leachate = 25.5 µg/year.

The environmental release of this leachate is into the soil or groundwater, which can migrate to surface water. The exposure assessor should choose the appropriate environmental fate and transport models (Figures 5-2 and 5-3), and identify relevant exposure routes (Figure 6-1)

Go to Step 3c.

Step 3c: Are emissions from vehicular resuspension controlled?

YES, CONTROL MEASURES TO SUPPRESS VEHICULAR RESUSPENSION ARE USED.

Measures to remove or contain ash residues which have been deposited on roadways near storage areas include vacuuming, sweeping, wetting, and chemical dust suppression. Use of any of these measures will provide some protection against vehicular resuspension of contaminated dust particles from roadways. In many facilities, storage areas are located near quench pits and within partially enclosed housing, with openings to allow for trucks to load and transport the ash. This housing, in combination with dust containment measures, is likely to provide the optimum protection.

The efficiency of dust containment or suppression measures must be included in any calculation of dust resuspension by vehicular traffic on roadways near storage piles. This can be accomplished by using a control efficiency factor.

It should be noted that, even when storage does not occur at an MWC facility, roadways near where ash exits the quench tank and in fact, anywhere along the route of the trucks carrying ash to a disposal site, can become contaminated with ash residues. For this example, dust resuspension by vehicular traffic over roadways is calculated near storage areas, and near disposal areas because this is likely where the release will be highest and of most concern, if it does occur.

- If containment and/or suppression measures are used, and these can be evaluated as fully effective, assume a control efficiency of 100%.

Go to Step 4.

If it is not possible to evaluate the effectiveness of dust resuspension or containment measures, then some estimate of containment efficiency must be assumed. In the absence of specific information on efficiency of different measures, a control efficiency of 25% will be used in this example.

Go to step 3c(i).

NO, DUST CONTAINMENT OR SUPPRESSION MEASURES WHICH WOULD REDUCE VEHICULAR RESUSPENSION ARE NOT USED.

- Assume a control efficiency of 0%.

Go to Step 3c(i).

Step 3c(i): Determine the following:

- Length of impacted roadway.
- Vehicle passes per day.

The length of impacted roadway times the vehicles passes per day equals the vehicle kilometers travelled per day.

- Proportion of impacted roadway that is paved.

Significantly more dust would be suspended if the roadway were unpaved. Unpaved roadways in MWC facilities are uncommon (MRI, 1990a) and are not assumed here.

The rate of dust resuspension due to vehicular traffic on paved roads near storage piles can be determined by applying equation 5-8 (EPA, 1985a):

$$E_p = 220(sL/12)^3$$

where:

- E_p = emission factor from paved surfaces (kg/VKt) (vehicle kilometer traveled)
 sL = mass of silt-sized material per unit of paved road material (g/m^2)

Assumptions for the scenario:

Length of impacted roadway: MRI (1990a) evaluated six MWC facilities for this mechanism of ash release. The haul roads for these facilities ranged in length from 100 to 400 meters, with an average of 238 meters. They determined that 25% of the haul length was commonly travelled by trucks transporting ash and other vehicles. What is likely to be equally, if not more, important than commonly traveled roadway is the length of roadway in the vicinity of storage pile that is impacted by fugitive emissions from the pile. Lacking specific information, the same 25% of roadway, or 60 meters, will be assumed for the length of impacted roadway.

Vehicles per day: MRI (1990a) determined, with actual counts of vehicles passes in their test facilities and other experience, that between 100 and 1000 vehicle passes per day occur over haul routes at MWC facilities, with the higher estimate for large capacity facilities. Since the facility in this example is a large capacity facility, the upper estimate of 1000 vehicle passes per day is assumed.

Silt loading: MRI (1990a) took 17 samples of roadway dust at their six facilities. The median sL , $2.5 g/m^2$, is assumed for this example.

Using these assumptions, one can calculate the unit resuspension of dust due to vehicular traffic as follows:

$$E_p = 220(sL/12)^3$$

$$E_p = 220(2.5/12)^3$$

$$E_p = 137.42 \text{ kg/VKt}$$

Step 3c(ii): Apply the control efficiency factor

Determine net dust resuspension in the presence of controlling factors (E_{net}) using the following equation:

$$E_{net} = E_p[1-(\text{control efficiency}/100)]$$

Since the control efficiency in this example is assumed to be 25%, the net dust resuspension per vehicle kilometer traveled is: $137.42 \times 0.75 = 103.07 \text{ kg/VKt}$. The total annual dust resuspension is:

$$103.07 \text{ kg/VKt} \times 1000 \text{ g/kg} \times (.06 \times 1000) \text{ VKt/d} \times 365 \text{ d/yr} = 2.26 \times 10^9 \text{ g.}$$

Mass loadings to the environment of each contaminant in the scenario from the resuspension of contaminated particulates in the vicinity of storage sites:

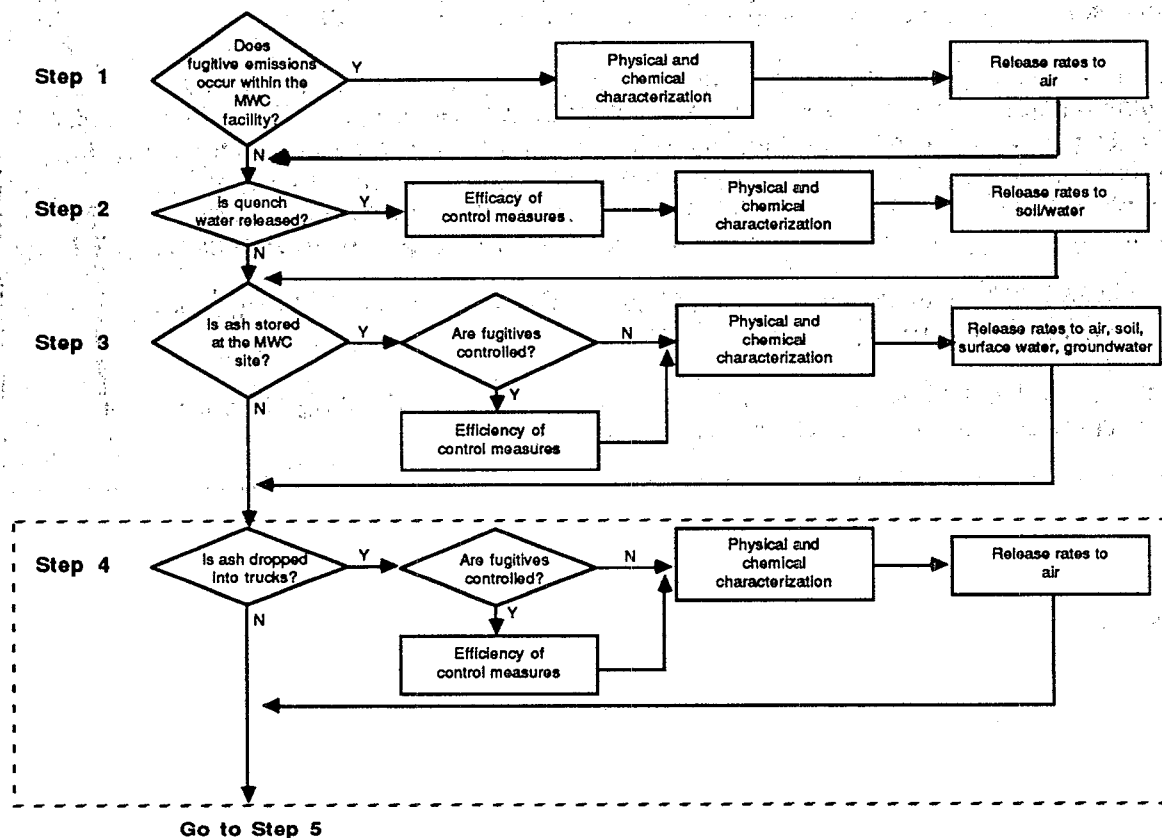
As noted in Chapter 5, the concentration on roadway particulates will be less than the concentration on ash itself because of mixing with clean dust particles, etc. MRI (1990a) took samples of ash as well as roadway particulates in their six sites and measured both for cadmium (and other metals), but not TCDD, concentrations. The ratio of concentrations (dust concentration/ash concentration) was wide, between 0.1 and 1.8, but the median ratio was 0.35. This ratio will be applied to both cadmium and TCDD (in the absence of better data for TCDD).

Cd: 2.26×10^9 g silt-sized dust resuspended/yr \times 100 μ g Cd/g ash \times 0.35 dust conc/ash conc = 7.91×10^{10} μ g Cd released/year

TCDD: 2.26×10^9 g silt-sized dust resuspended/yr \times 7.8×10^{-4} μ g TCDD/g ash \times 0.35 dust conc/ash conc = 6.17×10^5 μ g TCDD released/yr.

Go to Step 4.

Figure 7-4
Source Analysis for MWC Residuals Release - Step 4



STEP 4: IS THE ASH DROPPED INTO TRUCKS?

RATIONALE

According to the OSW survey (EPA, 1989a), ash is transported in the U.S. solely by trucks from MSW incinerators to disposal sites. Ash may be loaded directed onto trucks at the MWC facility using a conveyor dropping ash into a truck, or equipment may be employed to move the ash from on-site storage sites to the trucks.

NO, ASH IS NOT DROPPED INTO TRUCKS.

Go to Step 5.

YES, ASH IS DROPPED INTO TRUCKS.

Step 4a: Are fugitive emissions controlled?

Fugitive emissions may be controlled by wetting the ash or by enclosing the conveyor and hopper system. Purposefully wetting the ash for control of emissions may not be necessary if the trucks are loaded very shortly after exiting the quench tank. In that case, the ash is likely to wet enough so that fugitive emissions into air are unlikely. Alternately, wetting may be performed intentionally prior to transporting ash, if ash has been stored on-site.

YES, FUGITIVE EMISSIONS ARE CONTROLLED.

- If ash is sufficiently wetted prior to dropping into truck, assume 100% control of fugitive emissions.

Go to Step 5.

- If control measures other than wetting are used, or if ash is incompletely wetted, assume a lower level of control. Lacking such data, this scenario will use a level equal to 75% control.

NO, FUGITIVE EMISSIONS ARE NOT CONTROLLED.

Go to Step 4a(i).

Step 4a(i): Determine the following:

- Physical and chemical characteristics of the residuals
- Release rates and frequency.

Fugitive release rates can be estimated using equation 5-1:

$$Elu = k(0.0016)(U/2.2)^{1.3}(M/2)^{-1.4}$$

where:

- Elu = emission factor for loading and unloading(kg fugitive dust/Mg ash)
- k = particle size multiplier (dimensionless)
- U = wind speed (m/s)
- M = material moisture content (%)

Assumptions for the scenario:

Particle size multiplier. Total emissions will be estimated. Therefore, the maximum k value, for particle diameters <30 µm, 0.74, will be used.

Wind speed. The application of this equation is limited to situations where the wind speeds range between 0.6-6.7 m/s. In step 1, the wind speed was assumed to be 5.0 m/s for calculation of wind erosion. That same wind speed will be used here.

Moisture Content. The application of this equation is limited to materials with a moisture content ranges between 0.25-4.8%. The example scenario will use a conservative estimate of 0.25 %.

Using these assumptions, the fugitive release rates for ash loaded onto trucks is:

$$Elu = k(0.0016)(U/2.2)^{1.3}(M/2)^{-1.4}$$

$$Elu = .74(0.0016)(5/2.2)^{1.3}(.25/2)^{-1.4}$$

$$Elu = 6.32 \times 10^{-2} \text{ kg/Mg ash dropped.}$$

If fugitive emission control measures are applied, then E_{net} is calculated by:

$$E_{net} = Elu(1 - [\text{control efficiency}/100])$$

$$6.32 \times 10^{-2} \text{ kg/Mg } (0.25) = 1.6 \times 10^{-2} \text{ kg/Mg load.}$$

Finally, total annual emissions from loading prior to transport equals:

$$1.6 \times 10^{-2} \text{ kg/Mg} \times 1.82 \times 10^5 \text{ Mg/yr} = 2.91 \times 10^3 \text{ kg/yr, or } 2.91 \times 10^6 \text{ g/yr}$$

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions from loading ash onto trucks at the storage site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3.1 and 3.3).

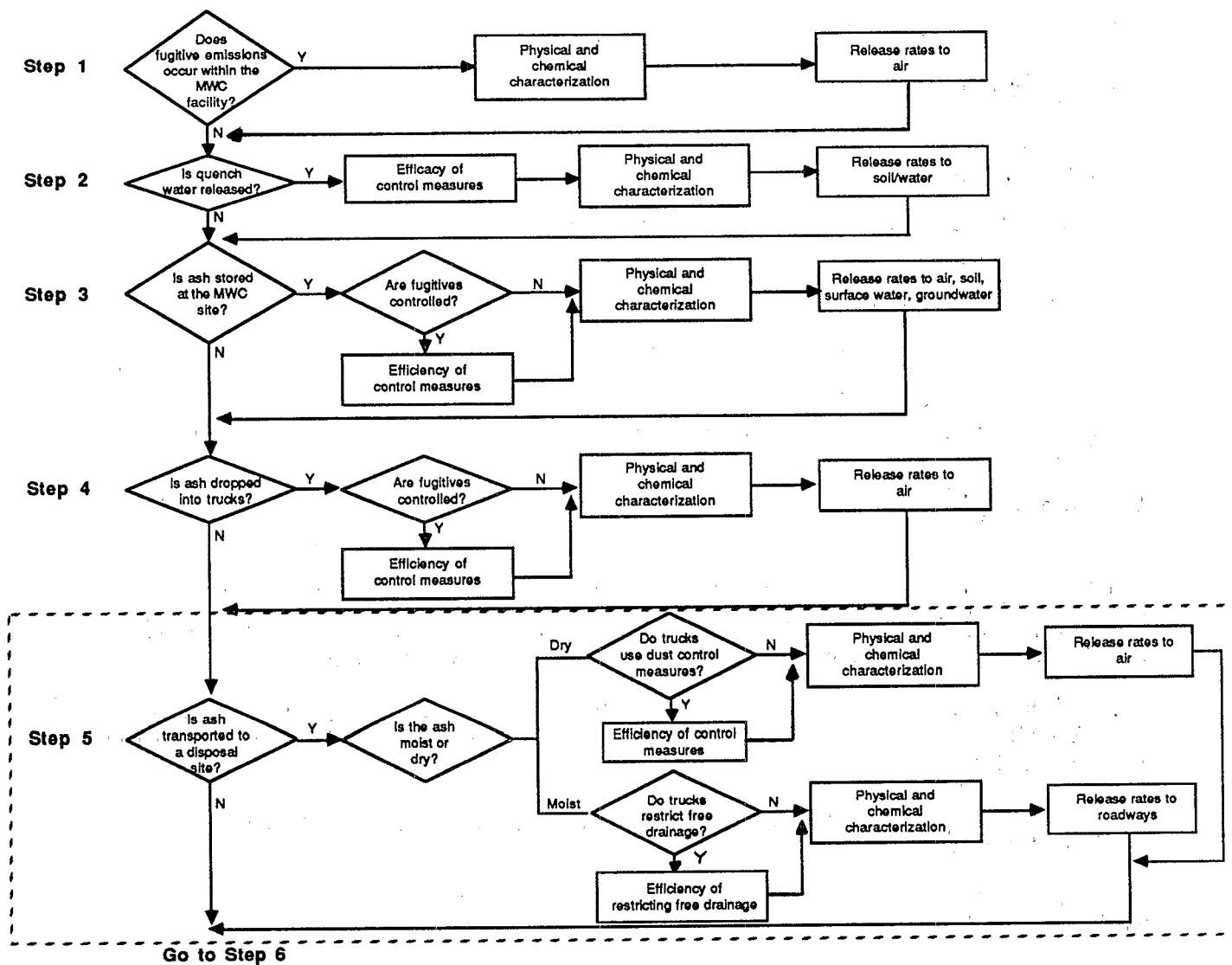
Cd: $2.91 \times 10^6 \text{ g fugitive ash released/yr} \times 100 \text{ µg Cd/g ash} = 2.91 \times 10^8 \text{ µg Cd released/yr.}$

TCDD: $2.91 \times 10^6 \text{ g fugitive ash released/yr} \times 7.8 \times 10^{-4} \text{ µg TCDD/g ash} = 2.27 \times 10^3 \text{ µg TCDD released/yr.}$

Since environmental release occurs into air and onto soil and surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3), and identify relevant exposure routes (Figure 6-1).

Go to Step 5.

Figure 7-5
Source Analysis for MWC Residuals Release - Step 5



STEP 5: IS ASH TRANSPORTED TO A DISPOSAL SITE ?

RATIONALE

Based on the results of the OSW survey (EPA, 1989a), it is assumed that ash is transported exclusively by trucks from the MWC facility to the various disposal sites. Distances for such transport can be as long as 700 miles. Two types of releases can occur, depending on whether the ash is very moist or very dry. Fugitive dust emissions can occur during transport if the ash is dry, and leaking of excess moisture can occur if the ash is very moist. Both possibilities are examined in this section, although obviously both possibilities cannot exist simultaneously.

NO, ASH IS NOT TRANSPORTED TO A DISPOSAL SITE.

Go to Step 6.

YES, ASH IS TRANSPORTED TO A DISPOSAL SITE.

Step 5a: Are either of the two possible releases of concern: high moisture content such that leakage from the trucks is possible, or low moisture content such that fugitive emissions are possible?

As indicated in Section 5., wind erosion, which was the surrogate process used to estimate fugitive emissions from trucks in transit, is a function of both moisture content and wind speed. At higher wind speeds, which can be expected for trucks, wind erosion can occur at higher ash moisture content. It might be surmised that wetting alone will not provide 100% protection for wind erosion, and the key for determining whether fugitive emissions from trucks is likely is in the use and effectiveness of truck coverings. The possibility that leakage from trucks will occur is best ascertained by visual inspection just before the truck begins transporting the ash.

With certainty, neither fugitive emissions or truck leakage will occur in transport of ash to a disposal site.

Go to Step 6.

Fugitive emissions should be considered.

Go to Step 5a.

Truck leakage should be considered.

Go to Step 5b.

Step 5a: Do trucks use dust control measures?

The majority of MWC operators (92%) responding to the OSW survey (EPA, 1989a) reported precautions to avoid fugitive dust releases during transportation. These precautions include covering the truck and/or wetting the ash.

NO, TRUCKS DO NOT USE DUST CONTROL MEASURES.

Go to Step 5a(i).

YES, TRUCKS USE DUST CONTROL MEASURES.

Although trucks use dust control measures, these measures may not be totally effective. Therefore, one should first estimate the efficiency of the control measures. If ash is covered or wetted, and it can be evaluated that such measures are fully effective, then assume 100% control efficiency. If no such data are available, then something must be assumed. Given that the frequency of use of such measures was high according to the OSW survey, 92%, it seems reasonable to also assume a high efficiency in the absence of information. In this example, a 90% efficiency will be used.

Go to Step 5a(i).

Step 5a(i): Determine the following:

- Physical and chemical characteristics of the residuals
- Release rates and frequency.

Assuming some release of ash does occur during transport, the release rates can be calculated using equation 5-2:

$$Ew_1 = 1.9 (s/1.5) [(365-P)/235](f/15)$$

where:

- Ew_1 = total suspended particulate emission factor (kg/day/hectare)
s = silt content (%)
P = number of days per year with more than 0.25 mm of rain
f = percent of time that wind speed is greater than 5.4 m/s (assumed to be 100 when the truck travels at a velocity ≥ 12 mph)

Assumptions for the scenario:

Silt content: A number of studies have provided values for the silt content of mixed MWC ash. These values range from 6.7% (Forrester, 1988) to 18.2% (Wells et al., 1988). Based on the scenarios developed by Wells et al. (1988), a conservative estimate of 19% silt content is used.

Precipitation: EPA (1988d) presents a map of the United States showing the number of days with more than 0.25 mm (0.01 inch) of rain. Regional values range from:

- 150 to 170 days for the Northeast
- 110 to 120 days for the South East
- 70 to 110 days for the Mid and South West
- 40 to 30 days for the Southern West Coast
- 90 to 150 days for the Northern West Coast

A value of 121 days will be applied to the example scenario (i.e., 365/3). Since most of the MWC facilities in the U.S. are in the Northeast, Southeast, and Pacific Northwest, this seems to be a reasonable estimate for our scenario.

Wind speeds: The trucks will be assumed to travel faster than 12 mph for the complete distance from the MWC facility to the disposal site.

Using these assumptions, the release rate of ash during transit can be calculated by:

$$Ew_1 = 1.9 (s/1.5)[(365-P)/235](f/15)$$

$$Ew_1 = 1.9 (19/1.5)[(365-121)/235](100/15)$$

$$Ew_1 = 166.6 \text{ kg/day/ha.}$$

Then, apply the emission control factor estimated for this scenario ($1 - 0.9 = 0.1$) to the emission rate to obtain the net emission rate (E_{net}):

$$E_{net} = Ew_1 \times 0.1 = 166.5 \text{ kg/day/ha} \times 0.1 = 16.7 \text{ kg/day/ha.}$$

This is equivalent to an hourly emission rate of .07 g/hr/m².

Assumption for the scenario:

- The approximate duration of a trip is 2 hours, assuming a mean distance of 32 miles (51.2 km) and an average truck velocity of 25 km/hr.
- The hypothetical facility produces 1.82×10^8 kg of ash per year (200,000 tons/year, 5×10^5 kg/day), and needs to dispose of the same volume that is produced.
- The average density of mixed ash can vary from 55 to 80 lb/ft³ (Wells et al., 1988) or 893 to 1300 kg/m³. A value of 894 kg/m³ will be used for the scenario.
- The average volume of a truck transporting such ash has been estimated to be 40 yd³ (Wells et al., 1988). The scenario will use a value of 40 m³, with a surface area of 40 m². With this volume, each truck can haul 35760 kg of ash per trip.
- In the example facility, hauling occurs 260 days per year. Therefore, it is necessary to load 20 truckloads of ash per day ($(1.82 \times 10^8 \text{ kg/yr}) / (260 \text{ d/yr} \times 35760 \text{ kg/load})$).

Using these assumptions and the estimated hourly ash emission rate, the total emission of ash for such a trip is:

$$0.07 \text{ g/hr/m}^2 \times 40 \text{ m}^2/\text{truck} \times 2 \text{ hr/trip} \times 20 \text{ trip/day} \times 260 \text{ workday/yr} = 2.91 \times 10^4 \text{ g/yr.}$$

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions from trucks enroute to the disposal site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3-1 and 3-3)

Cd: 2.91×10^4 g fugitive ash released/year \times 100 µg Cd/g ash = 2.95×10^6 µg Cd released/year.

TCDD: 2.91×10^4 g fugitive ash released/year \times 7.8×10^{-4} µg TCDD/g ash = 22.7 µg TCDD released/year.

In considering further environmental fate of ash emitted from trucks in transit to a landfill, one might also estimate a rate of emission per length of trip. In this example, 2.91×10^6 µg/yr of Cd is emitted along 51.2 km, resulting in a release of 5.68×10^4 µg/km/yr. Since environmental release of ash may occur onto soil or surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3) and identify relevant exposure routes (Figure 6-1).

Go to Step 6.

Step 5b: Are truck bottoms water-tight to avoid leakage?

As earlier indicated, visual inspection of the trucks as they are about to leave for disposal sites is the best way to ascertain whether leakage will occur. The only way unrestricted drainage would be likely to occur is if truck weep holes were open, or the truck is so poorly insulated against leakage such that unrestricted drainage would occur despite closed weep holes. For this example, it will be assumed that there is some restricted drainage.

Determine the following:

- Chemical and physical characteristics of the residuals
- Release rates and frequency

Assuming some truck leakage does occur during transport, the release rates can be calculated using equation 5-10:

$$El = 10(MC_1 - MC_o)(TT/TD)C(1-E/100)$$

where:

- El = amount of leakage during a truck haul, L
- MC₁ = moisture content of ash at truck loading, % by volume
- MC_o = moisture content of ash below which free drainage will not occur, % by volume
- TT = truck travel time, hr
- TD = total time to drain from MC₁ to MC_o, hr
- C = truck capacity, m³
- E_t = efficiency of truck at restricting free drainage, %

Assumption for the scenario:

Moisture content at transport: As discussed in Chapter 5, the moisture content of ash shortly after exiting the quench tank can be very high, over 40%. However, transport would most likely not begin with moisture contents this high; some drainage will be allowed to occur through weep holes, or drainage occurs during storage. For this example, 20% moisture content will be assumed.

Moisture content when no drainage will occur: Also as discussed in Section 5, a value of 5% seems reasonable for this parameter.

Truck travel time: As noted earlier, the assumed travel time is 2 hours.

Total time to drainage: A value of 24 hours was suggested in Chapter 5.

Truck capacity: The truck capacity for this example is 40 m³.

Truck efficiency: Since the weep holes will be closed, it will be assumed that the truck is 75% efficient in restricting free drainage.

Using these assumptions, the amount of water leakage that would occur per trip to the disposal site is:

$$El = 10(MC_1 - MC_o)(TT/TD)C(1-E/100)$$

$$El = 10(20 - 5)(2/24)40(1-75/100)$$

$$El = 125 \text{ L}$$

Using this value, and multiplying by 20 trips/day and 260 days/year, leads to an annual release of 6.5×10^5 L.

Mass loadings to the environment of each contaminant in the scenario from the leakage from trucks during transport to disposal.

The concentration of the example contaminants in the leakage is assumed to be equal to the EP-Tox limit for Cd (1000 µg/L) and upper-bound reported concentration for TCDD in extract from combined ash using the TCLP (2.3×10^{-4} µg/L).

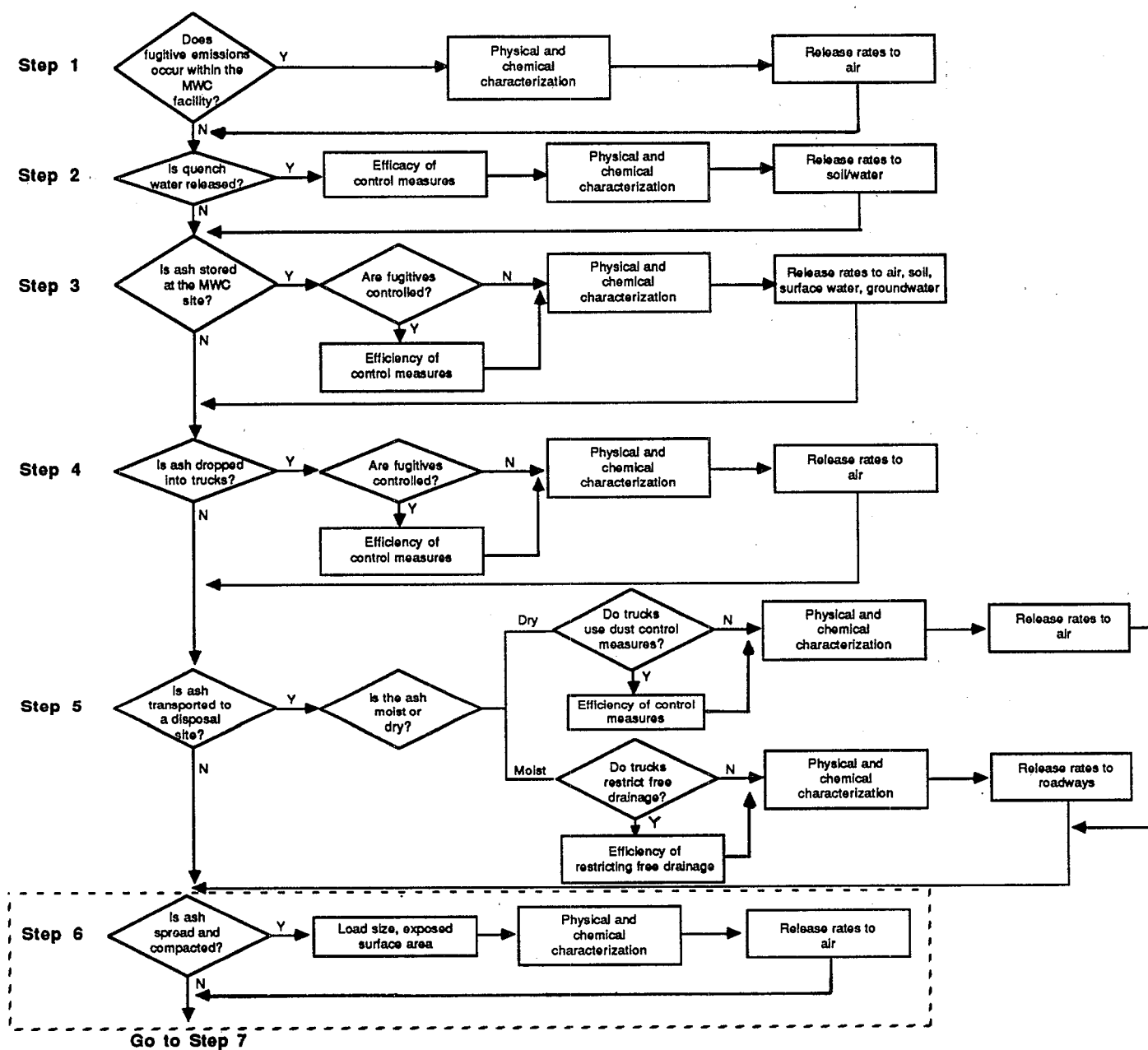
Cd: 6.5×10^5 L of leakage/year \times 1000 µg Cd/L of leakage = 6.5×10^8 µg/year

TCDD: 6.5×10^5 L of leakage/year \times 2.3×10^{-4} µg TCDD/L of leakage = 149.5 µg/year

The environmental release of the leakage occurs along the travel route, and can end up in storm sewers, road shoulders, otherwise transported by other cars via sorption to their tires, and so on. Determining the ultimate fate of such leakage needs to be site-specific, and would likely involve some creativity on the part of the exposure assessor.

Go to Step 6.

Figure 7-6
Source Analysis for MWC Residuals Release - Step 6



STEP 6: IS THE ASH SPREAD AT THE DISPOSAL SITE AND DOES VEHICULAR TRAFFIC OCCUR OVER PREVIOUSLY SPREAD ASH?

RATIONALE

At a landfill, dumping and spreading activities, and vehicular traffic over previously spread ash (e.g., during compaction), can result in fugitive emissions. The frequency of use of these operations has not been documented.

NO, THE ASH IS NOT SPREAD.

Go to Step 7.

YES, THE ASH IS SPREAD.

Go to Step 6a.

Step 6a: Emissions from Unloading.

Fugitive release rates can be estimated using equation 5-1:

$$Elu = k(0.0016)(U/2.2)^{1.3}(M/2)^{-1.4}$$

where:

- Elu = emission factor for loading and unloading (kg fugitive dust/Mg ash)
- k = particle size multiplier (dimensionless)
- U = wind speed (m/s)
- M = material moisture content (%)

Assumptions for the scenario:

Particle size multiplier. Total emissions will be estimated. Therefore, the maximum k value provided in EPA (1988d), for particle diameters < 30 μm , will be used: $k = 0.74$.

Wind speed. The application of this equation is limited to situations where the wind speeds range between 0.6-6.7 m/s. In step 1, the wind speed was assumed to be 5.0 m/s for calculation of wind erosion. That same wind speed will be used here.

Moisture Content. The application of this equation is limited to materials with a moisture content ranges between 0.25-4.8%. The example scenario will use a conservative estimate of 0.25 %.

Using these assumptions, the fugitive release rates for ash loaded onto trucks is:

$$\text{Elu} = k(0.0016)(U/2.2)^{1.3}(M/2)^{-1.4}$$

$$\text{Elu} = .74(0.0016)(5/2.2)^{1.3}(.25/2)^{-1.4}$$

$$\text{Elu} = 6.32 \times 10^{-2} \text{ kg/Mg ash unloaded.}$$

Next, determine net emissions if wetting or other fugitive controls are used. For this example, it will be assumed that the ash is unloaded at the landfill site with no control measures to reduce fugitive emissions:

$$6.32 \times 10^{-2} \text{ kg/MG ash unloaded} \times 1.82 \times 10^5 \text{ MG ash unloaded/yr} \times 1000 \text{ g/kg} = 1.15 \times 10^7 \text{ g/year.}$$

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions from unloading operations at the disposal site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 $\mu\text{g/g}$) and TCDD ($7.8 \times 10^{-4} \mu\text{g/g}$) in combined bottom and fly ash (see Tables 3-1 and 3-3)

Cd: $1.15 \times 10^7 \text{ g fugitive ash released/year} \times 100 \mu\text{g Cd/g ash} = 1.15 \times 10^9 \mu\text{g Cd released/year}$

TCDD: $1.15 \times 10^7 \text{ g fugitive ash released/year} \times 7.8 \times 10^{-4} \mu\text{g TCDD/g ash} = 8.97 \times 10^3 \mu\text{g TCDD released/year.}$

Go to Step 6b or 6c as necessary.

Step 6b: If the ash is spread after unloading, determine the emissions from spreading operations.

The emission rate for fugitive dust released from spreading operations can be calculated by using equation 5-5:

$$E_{at} = k(5.38)(s)^{0.6}$$

where:

E_{at} = emission factor for agricultural tillage (kg/ha)
 k = particle size multiplier (dimensionless)
 s = silt content (%)

Assumptions for the scenario:

Particle size multiplier. Total emissions will be estimated. Therefore, the maximum k value provided in EPA (1988d), for particle diameters $< 30 \mu\text{m}$, will be used: $k = 0.33$.

Silt content: A number of workers have provided values for the silt content of mixed MWC ash. These values range from 6.7% (Forrester, 1988) to 18.2% (Wells et al., 1988). Based on the scenarios developed by Wells et al. (1988), a conservative estimate of 19% silt content is used.

Using these assumptions and equation 5-5, the emission rate for spreading operations can be calculated in this scenario by:

$$E_{at} = k(5.38)(s)^{0.6}$$

$$E_{at} = 0.33 (5.38)(19)^{0.6}$$

$$E_{at} = 10.4 \text{ kg/ha}$$

Converting the emission rate to an hourly emission rate requires assumptions about the velocity at which the ash is spread, the spreader width, and the length of time per day that ash is spread.

Assumptions for the scenario:

- Spreader velocity is 3.2 km/hr (3.2×10^{-3} m/hr).
- Spreader width is 5 m.
- Spreader works for 8 hr per day.

Since ash is spread at a rate of:

$$(1\text{ha}/10^4 \text{ m}^2)(3.2 \times 10^3 \text{ m/hr})(5 \text{ m}) = 1.6 \text{ ha/hr},$$

then the emission factor, in units of kg/hr, can be calculated by:

$$\text{Eat (kg/hr)} = (\text{Eat kg/ha})(\text{spreading rate})$$

$$\text{Eat (kg/hr)} = 10.4 \text{ kg/ha} \times 1.6 \text{ ha/hr}$$

$$\text{Eat} = 16.6 \text{ kg/hr},$$

Therefore, assuming an 8 hour workday, the total emissions are:

$$16.6 \text{ kg/hr} \times 8 \text{ hours/day} \times 260 \text{ workdays/yr} \times 1000 \text{ g/kg} = 3.45 \times 10^7 \text{ g/year}$$

This value likely overestimates of the emissions, since the formula does not consider the moisture content of the ash that is being spread. This would be less true for this example, however, since the ash is assumed to have a very low moisture content of 0.25 %.

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions from spreading the ash at the disposal site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3-1 and 3-3)

Cd: 3.45×10^7 g fugitive ash released/year \times 100 µg Cd/g ash = 3.45×10^9 µg Cd released/year

TCDD: 3.45×10^7 g fugitive ash released/year \times 7.8×10^{-4} µg TCDD/g ash = 2.69×10^4 µg TCDD released/year

Step 6c: If vehicles traverse over spread ash, as in compacting activities, determine the emissions that result from vehicular traffic over the spread ash.

The magnitude of fugitive emissions occurring during vehicular traffic over previously spread ash (e.g., compacting activities) can be estimated using equation 5-4:

$$Eup = k[1.7(s/12)(Vs/48)(W/2.7)^{0.7}(nw/4)^{0.5}((365-P)/365)]$$

where:

- Eup = emission factor for unpaved roads (kg/VKt) (vehicle kilometer traveled)
- k = particle size multiplier (dimensionless)
- s = silt content (%)
- Vs = mean vehicle speed (km/hr)
- W = mean vehicle weight (Mg)
- nw = mean number of wheels
- P = number of days with at least 0.254 mm (0.01 inch) precipitation per year.

In the absence of actual data, the following assumptions and default values will be used in our scenario.

Assumptions for the scenario:

Particle size multiplier: Total emissions will be calculated. Therefore, the maximum k value, for particles diameters < 30 μm , will be used: $k = 0.80$.

Silt Content: The silt content of mixed ash has been discussed in previous steps. This value will be set at 19%.

Vehicle Speed: Kellermeyer and Ziemer (1989) used a value of 2 miles/hr. This value, which converts to 3.2 km/hr, will be used in the scenario.

Vehicle Weight: Kellermeyer and Ziemer (1989) used a value of 19 tons. This value, which converts to 1.73×10^4 kg, will be used in the scenario.

Number of Wheels: Compacting vehicles are typically two tracked, therefore, two wheels are used in the formula.

Precipitation: As in previous steps, a value of 121 days will be applied to the example scenario.

If the ash is wetted and maintained wet throughout the compaction period, the value of P should be set at 365.

Using these assumptions, and equation 5-4, a value for the release of ash during compacting operations can be estimated:

$$E_{up} = k[1.7(s/12)(V_s/48)(W/2.7)^{0.7}(nw/4)^{0.5}((365-P)/365)]$$

$$E_{up} = 0.80[1.7(19/12)(3.2/48)(1.73 \times 10^4/2.7)^{0.7}(2/4)^{0.5}((365-121)/365)]$$

$$E_{up} = 31.3 \text{ kg/km travelled.}$$

Using this value, and the assumption that the compactor travels at 3.2 km/hr for 8 hr/day (or 25.6 km/day), the daily emissions from compacting equal E_{up} x total daily km travelled, or

$$E_{up} = 31.3 \text{ kg/km} \times 25.6 \text{ km/day}$$

$$E_{up} = 801.3 \text{ kg/day, or } 2.08 \times 10^8 \text{ g/year (for a 260 day work-year).}$$

Mass loadings to the environment of each contaminant in the scenario from the release of fugitive ash emissions by vehicular transport over spread ash at the disposal site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3.1 and 3.3)

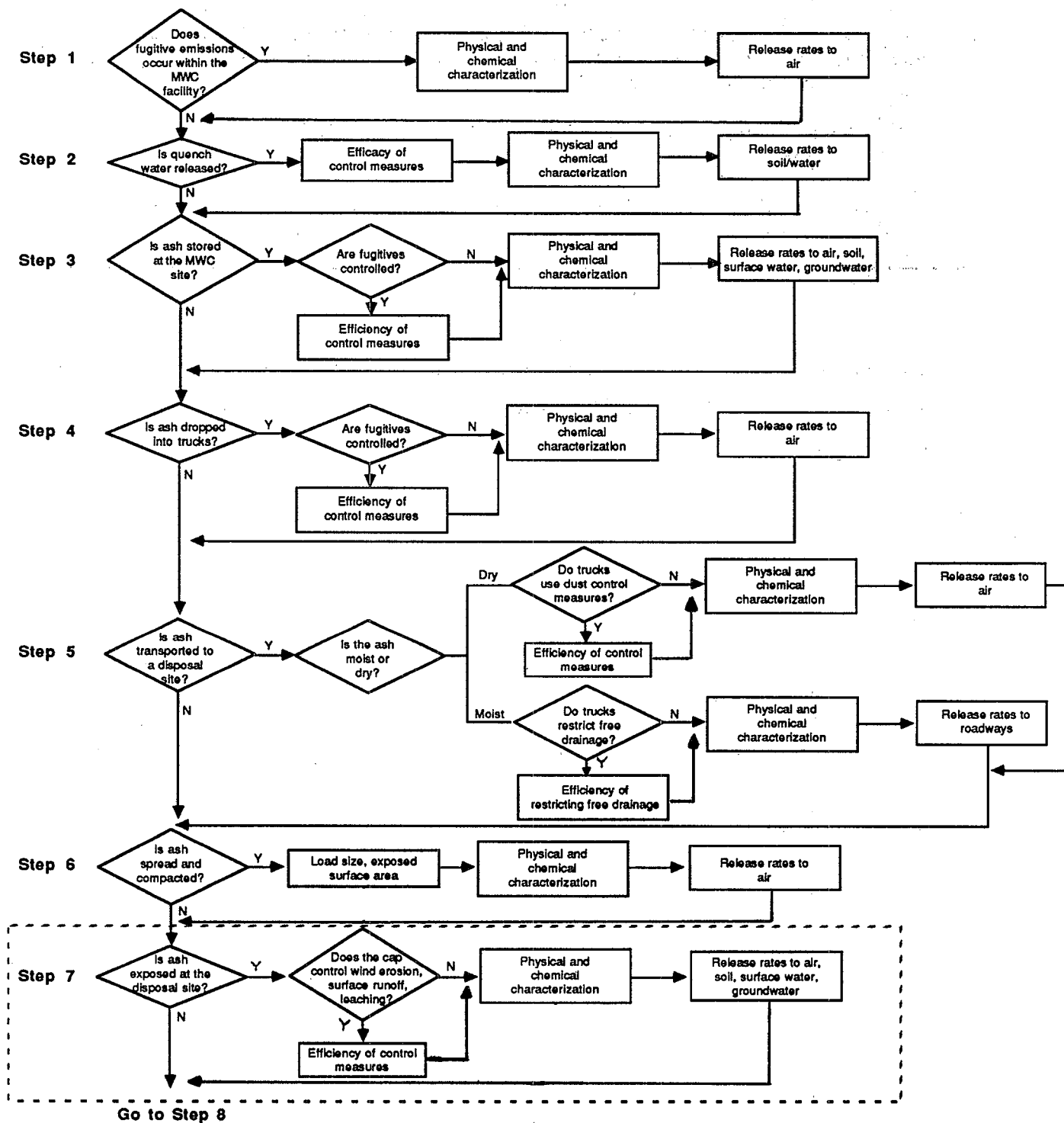
Cd: 2.08×10^8 g fugitive ash released/year \times 100 µg Cd/g ash = 2.06×10^{10} µg Cd released/year

TCDD: 2.08×10^8 g fugitive ash released/year \times 7.8×10^{-4} µg TCDD/g ash = 1.62×10^5 µg TCDD released/year.

Since environmental release may occur into air, soil, and surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3) and identify relevant exposure routes (Figure 6-1).

Go to Step 7.

Figure 7-7
Source Analysis for MWC Residuals Release - Step 7



STEP 7: IS ASH EXPOSED AT THE DISPOSAL SITE ?

RATIONALE

It might be reasonable to assume that an inactive, capped landfill is no longer susceptible to natural processes of release of ash and ash contaminants. These processes include wind erosion, surface runoff, and leaching. In this example, it is assumed that 0.25 ha of a total of 8.25 ha is active, and susceptible to these processes. Further, it will be assumed that the capped portion, 8 ha, is not fully protective of water which could leach to ground water carrying ash contaminants.

NO, ASH IS NOT EXPOSED AT THE DISPOSAL SITE.

Go to Step 8.

YES, ASH IS EXPOSED AT THE DISPOSAL SITE.

Step 7a: Determine the following:

- Physical and chemical characteristics of the residuals.
- Release rates and frequency.

Two types of releases will be estimated from the working, or uncapped, portion of the landfill. One is wind erosion resulting in fugitive emissions, and the other is surface runoff. A third type of release, leachate, is estimated for both capped and uncapped portions of the landfill.

Go to Step 7b, 7c, or 7d, as necessary.

Step 7b. Estimate fugitive emissions from wind erosion from uncapped portion of landfill.

Wind generated releases of ash from uncovered portions of a landfill site can be calculated using equation 5-7:

$$E = 0.036 (1-V)(U_m/U_t)^3 F(x)$$

where:

- E = total wind erosion flux of particles <10 μm ($\text{g}/\text{m}^2 \text{ hr}$)
- V = fraction of vegetation cover
- U_m = mean wind speed (m/s)
- U_t = threshold wind speed (m/s)
- F(x) = a dimensionless ratio function.

The following assumptions and default values are used in this example scenario to estimate wind-generated release rates of ash.

Assumptions for the scenario:

Application of equation 5-7 was described in Section 5.1.6., including recommendations for parameter values. Those parameter values will be used in this example, and they are:

V: No vegetative cover, or $V = 0.0$

U_m : A mean annual wind speed of 5.0 m/s was used in the discussion in Section 5.1.6.

U_t : This was solved for as 13.0 m/s

F(x): Given U_t and U_m , this was solved for as 0.11

The unit emission solved for by equation 5-7 is units of $\text{g}/\text{m}^2 \text{ hr}$. To convert to an annual basis, the following assumptions are made:

Exposed Area: The exposed surface area for a disposal site must be determined on a site-specific basis. This example will assume that the area in which ash is spread on at any given time is equal to storage area at the MWC facility, or 0.25 hectare, or 2500 m^2 .

Hours: For this example, it will be assumed that the disposal area is covered during non-working hours. Since disposal occurs during 260 work-days/yr, the number of hours per year equals: 260 wd/yr \times 8 hr/wd = 2080 hrs.

Control Efficiency: It will be assumed there are no controls to reduce wind erosion during working hours.

Using these assumptions, one can calculate the wind generated release of ash from a disposal site (E) as follows:

$$E = 0.036 (1-V)(U_m/U_t)^3 F(x)$$

$$E = 0.036 (5/13)^3 (0.11)$$

$$E = 2.25 \times 10^{-4} \text{ g/m}^2\text{-hr}$$

For an area of 2500 m² and for 2060 hours/yr, the annual emission equals: $2.25 \times 10^{-4} \times 2500 \times 2060 = 1158 \text{ g/yr}$

Mass loadings to the environment of each contaminant in the scenario from the release of windblown fugitive ash emissions at the disposal site:

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 µg/g) and TCDD (7.8×10^{-4} µg/g) in combined bottom and fly ash (see Tables 3-1 and 3-3)

Cd: $1158 \text{ g fugitive ash released/year} \times 100 \text{ µg Cd/g ash} = 1.16 \times 10^5 \text{ µg Cd released/year}$

TCDD: $1158 \text{ g fugitive ash released/year} \times 7.8 \times 10^{-4} \text{ µg TCDD/g ash} = .9 \text{ µg TCDD released/year}$

Note that this wind erosion emission is distinct from the emissions estimated for unloading, spreading and compacting. Since the environmental release of ash may occur into air, soil, and surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3) and identify relevant exposure routes (Figure 6-1).

Step 7c. Estimate surface runoff from uncapped portion of landfill.

The volume of runoff water can be estimated using equation 5-11:

$$RV = .01 R A_w$$

where:

RV = runoff volume (m^3/yr)

R = annual runoff (cm/yr)

A_w = working (exposed) landfill area (m^2).

Assumptions for the scenario:

Exposed landfill area: The working area of 0.25 ha, or 2500 m^2 , is susceptible to runoff.

Annual runoff: EPA (1985c) describes methods to estimate runoff and also includes a figure (Figure III-9) which contains isolines of mean annual row crop runoff for the eastern part of the United States. The isolines for annual row crop runoff for the soil highest in runoff potential (soil type "D") are most appropriate for this application, even though the ash and the soil beneath it may have good drainage and be less susceptible to runoff. This is because a landfill is more appropriately "fallow" rather than "row crop" and that runoff for fallow conditions greatly exceed those for row crop conditions. From that figure, annual runoff for a "D" soil on the eastern seaboard range from 5 to 17 inches. A value of 11 inches/year, or 28 cm/yr , is assumed for this example.

The runoff volume for this example then equals:

$$RV = .01 (28) (2500)$$

$$RV = 700 \text{ m}^3/\text{yr}, \text{ or } 7 \times 10^5 \text{ L}/\text{yr}$$

Mass loadings to the environment of contaminant in the scenario from runoff release at the disposal site:

The concentration of the example contaminants in the leachate is assumed to be equal to the EP-Tox limit for Cd (1000 $\mu\text{g}/\text{L}$) and upper-bound reported concentration for TCDD in extract from combined ash using the TCLP (2.3×10^{-4} $\mu\text{g}/\text{L}$).

Cd: 7×10^5 L of leachate released/year \times 1000 μg Cd/L of leachate = 7×10^8 μg Cd released/year

TCDD: 7×10^5 L of leachate released/year \times 2.3×10^{-4} μg TCDD/L of leachate = 161 μg TCDD released/year.

STEP 7d: Estimate leaching from capped and uncapped portions of landfill.

The use of landfill liners is important in determining the possibility of groundwater contamination. In the ORD/EMSL study survey (EPA, 1988c) it was reported that liners were used by 60% of 72 respondents. Unfortunately, no data have been developed regarding liner failure rates in ash landfills. However, three different leachate loss scenarios have been used in an evaluation of leachate release from an MWC ash landfill (Kellermeyer and Ziemer, 1989). This illustration scenario will be applied to liner failure resulting in loss of 100%, 10% and 1% of the total leachate.

The rate of percolation can be determined by applying equation 5-10. (EPA, 1985c):

$$Q = P + I - E$$

where:

- Q = annual percolation (cm)
- I = annual irrigation (cm)
- P = annual precipitation (cm)
- E = annual evapotranspiration (cm).

Assumptions for the scenario:

Irrigation: Irrigation will not be considered to exist at the landfill site, therefore I will be set at 0.

Precipitation and Evapotranspiration: EPA (1985c) presents a map showing gradients of mean annual potential evapotranspiration minus precipitation. These values range from +70 inches in the desert regions of the Southwest to - 50 in the Pacific Northwest. A value similar to those found on the East Coast will be used: -20 inches = -50.8 cm.

The total leachate generation can be calculated from equation 5-9:

$$L = .01 Q [Fw(A_w) + Fc(A_c)]$$

where:

- L = leachate volume (m^3/yr)
- Q = potential percolation (cm/yr)
- F_w = fraction of potential percolation, Q , which results from precipitation falling on the working (exposed) portion of the landfill
- A_w = working landfill area (m^2)
- F_c = fraction of potential percolation, Q , which results from precipitation falling on the covered portion of the landfill
- A_c = covered landfill area (m^2).

Assumptions for the scenario:

Percolation: The value determined using equation 5.8 will be used: 50.8 cm/year

Exposed and covered landfill area: The exposed landfill area will be estimated to be approximately the same as the area of the on-site storage facility, i.e., 0.25 ha (2500 m^2). By default, the covered landfill area is assumed to be 8 ha ($80,000 \text{ m}^2$).

Working Area Percolation: Similar to Kellermeyer and Ziemer, 1989, the conservative assumption of 1.0 is assumed.

Covered Area Percolation: Similar to Kellermeyer and Ziemer, 1989, the assumption of 0.15 is assumed.

Using these assumptions, and equation 5-9, the total leachate rate can be calculated by:

$$L = .01 Q [F_w(A_w) + F_c(A_c)]$$

$$L = .01 (50.8) [1.0(2,500) + 0.15(80,000)]$$

$$L = 7.37 \times 10^3 \text{ m}^3/\text{year}, \text{ or } 7.37 \times 10^6 \text{ L/year.}$$

Applying the liner failure rates gives:

$$\text{At 100\% failure: } 7.37 \times 10^6 \text{ L/year}$$

$$\text{At 10 \% failure: } 7.37 \times 10^5 \text{ L/year}$$

$$\text{At 1\% failure: } 7.37 \times 10^4 \text{ L/year}$$

Mass loadings into the environment of contaminant in the scenario from leachate release of the disposal site with varying degrees of liner failure:

The concentration of the example contaminants in the leachate is assumed to be equal to the EP-Tox limit for Cd (1000 µg/L) and upper-bound reported concentration for TCDD in extract from combined ash using the TCLP (2.3×10^{-4} µg/L).

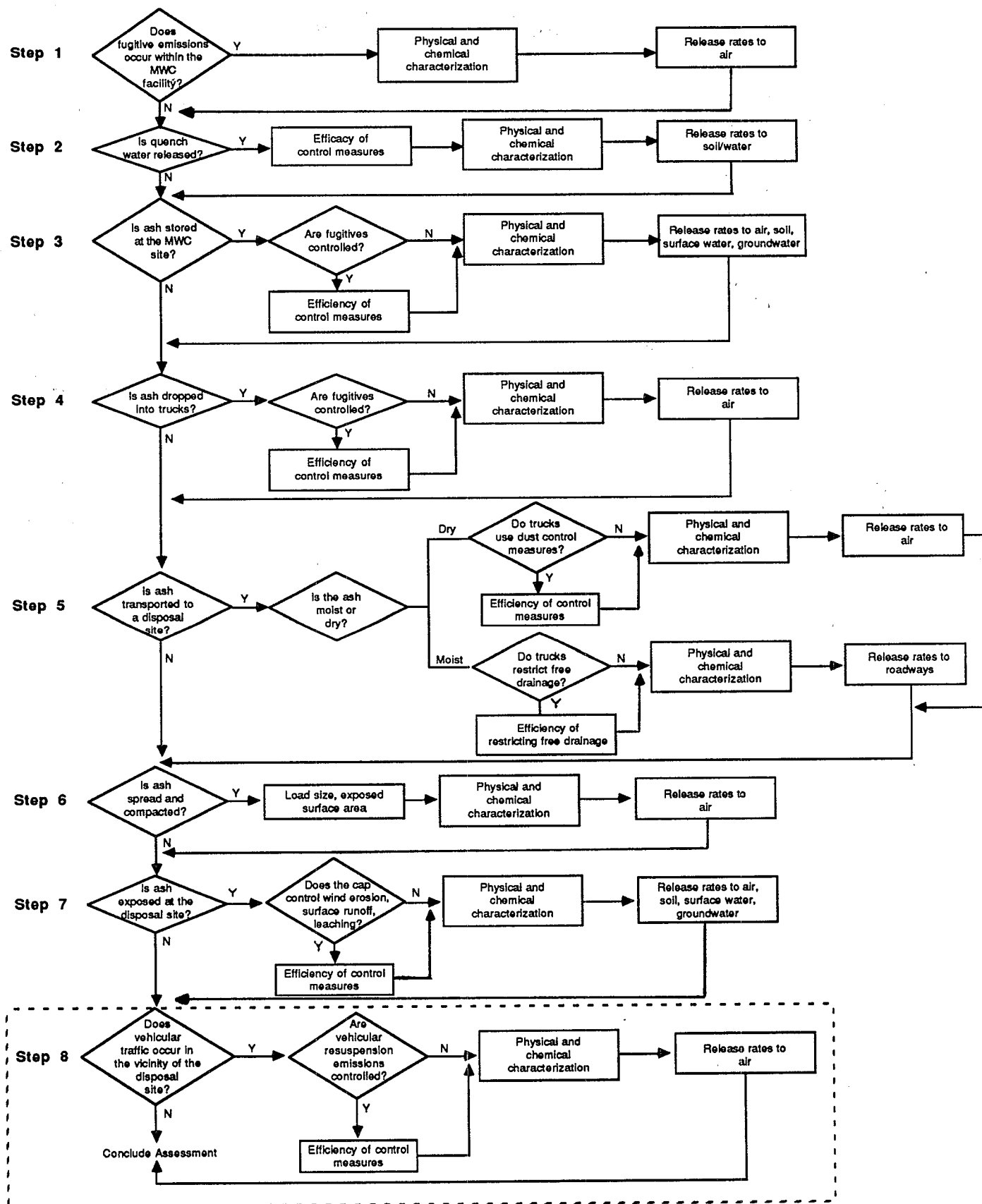
Cd: 100% failure: 7.37×10^6 L of leachate/yr \times 1000 µg Cd/L of leachate = 7.37×10^9 µg/year
10% failure: 7.37×10^8 µg/year
1% failure: 7.37×10^7 µg/year

TCDD: 100% failure: 7.37×10^6 L of leachate/yr \times 7.8×10^{-4} µg TCDD/L of leachate = 1.7×10^3 µg/year
10% failure: 1.7×10^2 µg/year
1% failure: 17 µg/year

Since the environmental release of the leachate occurs into the groundwater or soil, and subsequently to surface water, the exposure assessor should choose appropriate environmental fate and transport models (Figures 5-2 and 5-3) and identify relevant exposure routes (Figure 6-1).

Go to Step 8.

Figure 7-8
Source Analysis for MWC Residuals Release - Step 8



STEP 8: DOES VEHICULAR TRAFFIC OCCUR IN THE VICINITY OF DISPOSAL

RATIONALE

Often, haul routes in landfills are unpaved, which can lead to significant resuspension of dust due to vehicular traffic. Such traffic includes the trucks hauling ash and other vehicles. Roadside particulates can become contaminated because of fugitive emissions from the disposal site, drainage from trucks, vehicle track-out from spreading and compacting vehicles, and so on.

NO, VEHICULAR TRAFFIC DOES NOT OCCUR IN THE VICINITY OF DISPOSAL SITES.

Conclude Assessment

YES, VEHICULAR TRAFFIC OCCURS IN THE VICINITY OF DISPOSAL SITES

Step 8a: Are vehicular resuspension emissions controlled?

The principal measures to remove or contain ash residues which have been deposited on haul routes within disposal facilities are wetting and chemical dust suppression. Use of these can become critical for driver visibility if the haul route is unpaved, which it most often is within a landfill.

- If containment and/or suppression measures are used, and these can be evaluated as fully effective, assume a control efficiency of 100%.

Conclude assessment.

If it is not possible to evaluate the effectiveness of dust resuspension or containment measures, then some estimate of containment efficiency must be assumed. In the absence of specific information on efficiency of different measures, a containment measure of 90% will be used in this example.

Go to step 8(i).

NO, DUST CONTAINMENT OR SUPPRESSION MEASURES WHICH WOULD REDUCE VEHICULAR RESUSPENSION ARE NOT USED.

- Assume a containment efficiency of 0%.

Go to Step 8(i).

Step 8(i): Determine the following:

- Length of impacted haul route.
- Vehicle passes per day.

The length of impacted roadway times the vehicles passes per day equals the vehicles travelled per day.

- Proportion of impacted roadway that is unpaved.

Significantly more dust would be suspended if the roadway were unpaved. Unpaved roadways are not uncommon in MSW landfills, particularly in the active portions of the landfill. Unpaved roadways are assumed for this example.

The magnitude of fugitive emissions occurring during vehicular traffic over unpaved roadways can be estimated using equation 5-4:

$$Eup = k[1.7(s/12)(Vs/48)(W/2.7)^{0.7}(nw/4)^{0.5}((365-P)/365)]$$

where:

- Eup = emission factor for unpaved roads (kg/VKt) (vehicle kilometer traveled)
- k = particle size multiplier (dimensionless)
- s = silt content (%)
- Vs = mean vehicle speed (km/hr)
- W = mean vehicle weight (Mg)
- nw = mean number of wheels
- P = number of days with at least 0.254 mm (0.01 inch) precipitation per year.

In the absence of actual data, the following assumptions and default values will be used in our scenario.

Assumptions for the scenario:

Particle size multiplier: Total emissions will be estimated. Therefore, the k value for the largest sized particles, those with diameters $<30 \mu\text{m}$, will be used: $k = 0.80$.

Silt Content: MRI (1990a) took two samples on unpaved haul routes in two landfills, and obtained silt contents of 6.7 and 20.1%. The mean of these two values, 13.4%, will be assumed.

Vehicle Speed: MRI (1990a) assumed a vehicle speed of 24 km/hr (15 mph) in their assessment of dust resuspension. This will be assumed in this example.

Vehicle Weight: A range of vehicle weights for vehicles disposing materials in ISW landfills was given as $14\text{--}40 \times 10^3 \text{ kg}$ in MRI (1990a). The midpoint of this range, $27 \times 10^3 \text{ kg}$ will be assumed here.

Number of Wheels: A range for the number of wheels was given as 6-14 in MRI (1990a). The midpoint of this range, 10, will be assumed here.

Precipitation: As in previous steps, a value of 121 days will be applied to the example scenario.

Length of Haul Route: MRI conducted a survey of MWC ash disposal sites, including ISW landfills (described in MRI, 1990a). The median haul route length from 46 ISW landfills was 407 m (1320 ft). Like the assessment of resuspended dust in the vicinity of storage areas, it will be assumed that 25% of this length, or 102 m, is impacted by the ash disposal activities.

Vehicle Passes per Day: In this same survey, MRI determined that the median number of daily vehicle transactions, excluding ash haulers, was 26. However, not all 26 vehicles will pass over the same impacted roadway, since different portions of the landfill could be active. Assuming half of them pass over the impacted roadway, or 13 trucks, this translates to 26 passes. This is added to the 20 ash haulers per day, or 40 passes, for a total of 66 vehicle passes per day for this example.

Using these assumptions, and equation 5-4, a value for the release of ash due to vehicular traffic and resulting resuspension of contaminated dust operations can be estimated:

$$Eup = k[1.7(s/12)(Vs/48)(W/2.7)^{0.7}(nw/4)^{0.5}((365-P)/365)]$$

$$Eup = 0.80[1.7(13.4/12)(24/48)(2.7 \times 10^4/2.7)^{0.7}(10/4)^{0.5}((365-121)/365)]$$

$$Eup = 506.4 \text{ kg/km travelled.}$$

Step 3c(ii): Apply the control efficiency factor

Determine net dust resuspension in the presence of controlling factors (E_{net}) using the following equation:

$$E_{net} = Eup[1-(\text{control efficiency}/100)]$$

Since the control efficiency in this example is assumed to be 90%, the net dust resuspension per vehicle kilometer traveled is: $506.4 \times 0.10 = 51 \text{ kg/km}$.

The total annual dust resuspension considers the 102 m, or .1 km, assumed for impacted haul route, the 260 days/year, and 66 vehicle passes per day: $51 \text{ kg/km} \times 1000 \text{ g/kg} \times (.1 \times 66) \text{ km/d} \times 260 \text{ d/yr} = 8.75 \times 10^7 \text{ g/yr}$

Mass loadings to the environment of each contaminant in the scenario from the resuspension of contaminated particulates in the vicinity of storage sites:

As noted in Section 5., the concentration on roadway particulates will be less than the concentration on ash itself because of mixing with clean dust particles, etc. MRI (1990) took samples of combined ash as well as haul route particulates in four of their six sites and measured both for cadmium (and other metals), but not TCDD, concentrations. The ratio of concentrations was wide, between 0.01 and 0.45, but the mean ratio was 0.24. This ratio will be applied to both cadmium and TCDD (in the absence of better data for TCDD).

The concentration of the example contaminants in this scenario is assumed to be equal to the upper-bound reported concentrations of Cd (100 $\mu\text{g/g}$) and TCDD ($7.8 \times 10^{-4} \mu\text{g/g}$) in combined bottom and fly ash (see Tables 3-1 and 3-3).

Cd: $8.75 \times 10^7 \text{ g silt-sized dust resuspended/yr} \times 100 \mu\text{g Cd/g ash} \times 0.24 \text{ dust conc/ash conc} = 2.1 \times 10^9 \mu\text{g Cd released/year}$.

TCDD: $8.75 \times 10^7 \text{ g silt-sized dust resuspended/yr} \times 7.8 \times 10^{-4} \mu\text{g TCDD/g ash} \times 0.24 \text{ dust conc/ash conc} = 1.64 \times 10^4 \mu\text{g TCDD released/yr}$.

Conclude Assessment

7.2. COMPARISON OF THE EXAMPLE SCENARIO TO A FIELD STUDY OF FUGITIVE ASH EMISSIONS

A study was conducted to estimate the amount of fugitive dust generated at the ash monofill located adjacent to the Ogden Haverhill Associates (OHA) Resource Recovery Facility in Haverhill, Massachusetts (Hahn et al., 1990). However, in addition to simply using predictive equations and default values, field monitoring data and site-specific meteorological information were obtained at the OHA landfill. It was used to examine the ability of two AP-42 emission equations of the type presented in Section 5. and 7. of this document to predict ash emissions at the site. One was an early version of the emission from batch drop operations (EPA, 1985a) that has been since updated to be of the form presented as equation 5-1, and the other was equation 5-4 to estimate emissions from unpaved roads used in this application to estimate emissions from spreading.

The authors of the OHA study report that fugitive ash emissions measured in the field for unloading and spreading operations were significantly less than those predicted by the use of these equations when actual data on moisture content of the ash, wind speed and quantity of ash dumped were used in place of conservative default values. However, the actual moisture content in this study was 25%. This is significantly higher than the 0.25-0.70% range for moisture content recommended in EPA (1985a) for use with the "batch drop" equation. Also, equation 5-4 does not have a moisture content term, but the related term $(365-P)/365$ (where P is the number days where rainfall exceeds 0.01 mm) certainly implies that high moisture content reduces emissions; it also implies that this equation is most appropriately applied on an average basis, rather than an event basis.

The example in this section did assume a conservative moisture content of 0.25%. Noting this two-order-of-magnitude difference between the example moisture content and the measured moisture content at Haverhill, it is likely that predicted annual fugitive ash emissions in the example presented in Section 7.1. of this document are higher than those that can be found at many actual facilities. However, the purpose of this example is not to provide a relative indication of the amount of ash emissions that result under average operating conditions, nor is it necessarily intended to provide default values for the input parameters of the equations listed in Section 5., but rather to take the exposure assessor through a step-by-step process for estimating the release of MWC ash into the environment. As described below, subsequent steps are required to quantify the health risk posed by exposure to MWC residuals.

7.3. EXPOSURE PATHWAYS

The analysis of exposure pathways addresses the transport of MWC residuals and their constituents from the source of release to their point of exposure with the affected individuals or populations. The

initiation of an exposure pathway analysis begins with the identification of the relevant environmental transport media. The decision network for such identification was described in Figures 5-1 through 5-4.

An exposure pathway analysis further requires that the exposure point concentrations of the chemicals in question be known or estimated. They are often estimated using fate and transport models which take the contaminant from the point of entry into a media until it reaches an exposed individual. Before using fate and transport models, one must first convert emission rates to a chemical-specific basis, as described in Section 5.3. Section 6. described the source documents for selection of appropriate fate and transport models.

7.4. INTEGRATION OF SOURCE TERMS INTO AN EXPOSURE/RISK ASSESSMENT

The release rates of the MWC residuals and mass loadings of two environmental pollutants, cadmium and TCDD, were estimated in this example scenario under a defined set of conditions. The source terms for this example illustration are summarized in Table 7-1. These release rates and mass loadings can be used in an exposure assessment by following the procedures outlined documents identified in earlier sections of this report. In addition, the source terms can be used as input parameter in the equations in the *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (EPA, 1990b). This methodology provides risk assessors with the guidance necessary to estimate health risks following indirect human exposures to contaminants in soil, vegetation, and water bodies. Furthermore, the methodology guides the assessor to determine exposure through the various pathways, and to a final risk determination. Although the methodology does not address the inhalation pathway, guidance for determining human exposure by that route can be obtained in a number of publications such as:

- EPA (1989b). *Risk Assessment Guidelines for Superfund*.
- EPA (1988g). *Superfund Exposure Assessment Manual*.

Table 7-1. Summary of Source Terms for Example Illustration

Step	Description	Residual Release Rate	Cadmium Release Rate (µg/yr)	TCDD Release Rate (µg/yr)	Receiving Medium
1	Vaporization	n/a	n/a	n/a	air, surface deposition
2	Quench release	3.43×10^7 L/year	3.43×10^{10}	7.88×10^3	surface water, groundwater, soil
3	Ash storage				
3a	fugitive emissions	695 g/year	6.95×10^4	0.5	air, soil, surface water
3b	leachate release	1.11×10^5 L/year	1.11×10^8	25.5	groundwater, surface water, soil
3c	vehicular resuspension	2.26×10^9 g/year	7.91×10^{10}	6.17×10^5	air, soil, surface H ₂ O
4	Truck loading	2.91×10^6 g/year	2.91×10^8	2.27×10^3	air, surface water, soil
5	Ash transport				
5a	fugitive emissions	2.91×10^4 g/year	2.91×10^6	22.7	air, surface water, soil
5b	truck leakage	6.5×10^5 L/year	6.5×10^8	149.5	soil
6	Landfill operations				
6a	unloading	1.15×10^7 g/year	1.15×10^9	8.97×10^3	air, surface water, soil
6b	spreading	3.45×10^7 g/year	3.45×10^9	2.69×10^4	air, surface water, soil
6c	compacting	2.08×10^8 g/year	2.08×10^{10}	1.62×10^5	air, surface water, soil
7	Landfill releases				
7a	wind erosion	1158 g/year	1.16×10^5	0.9	air, surface water, soil
7b	run off	7×10^5	7×10^8	161	soil, surface water
7c	percolation				
	100% liner failure	7.37×10^6 L/year	7.37×10^9	1.7×10^3	soil, groundwater, surface water
	10% liner failure	7.37×10^5 L/year	7.37×10^8	1.7×10^2	groundwater, surface water
	1% liner failure	7.37×10^4 L/year	7.37×10^7	17	groundwater, surface water
8	Vehicular resuspension	8.75×10^7 g/year	2.1×10^9	1.64×10^4	air, soil, surface water

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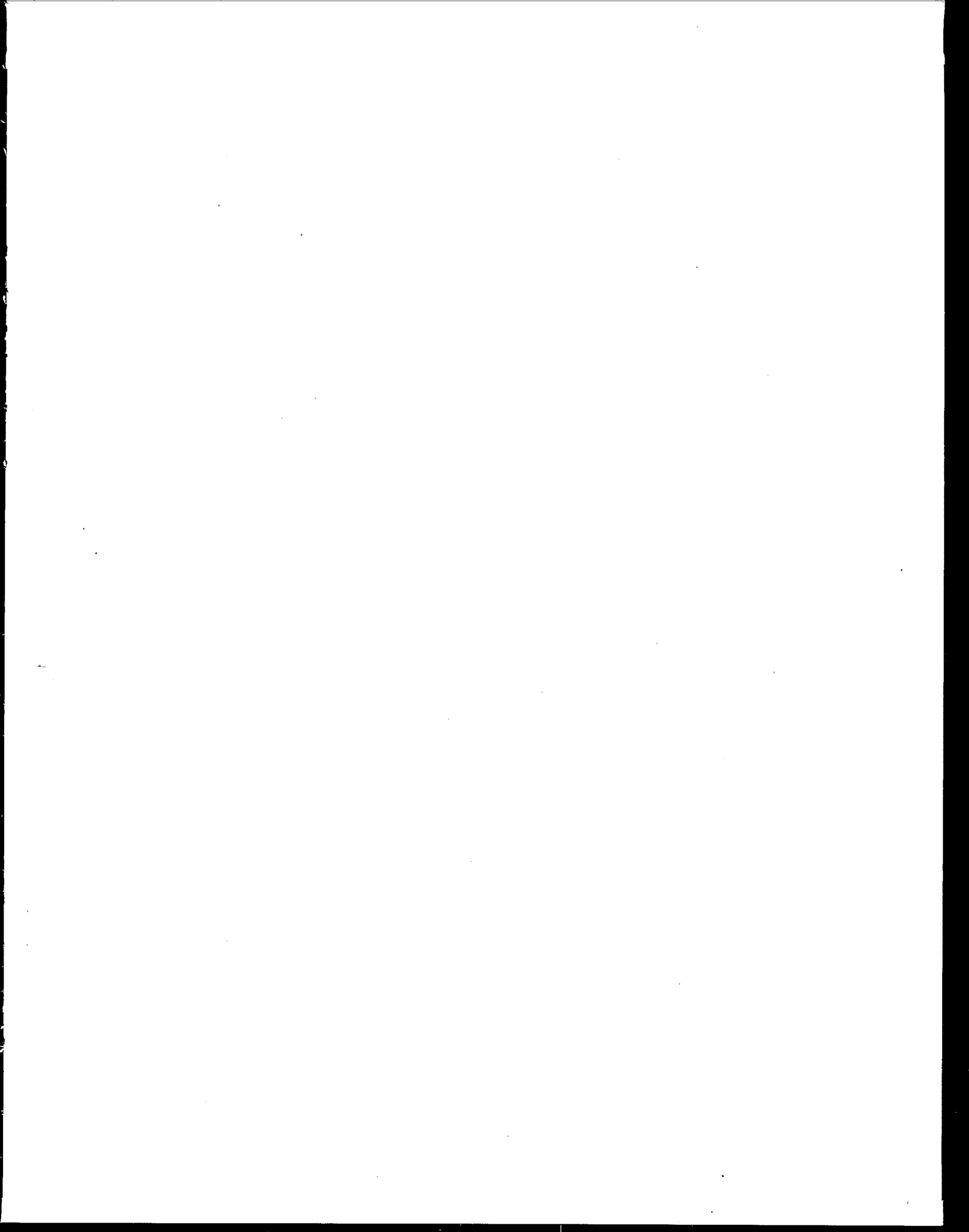
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