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Health Risks of Landfilling versus Combustion of Municipal Solid Waste: An Illinois Comparison

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ABSTRACT

Current policy envisions a hierarchy of steps for guiding the management of municipal solid waste (MSW); they are: source reduction, recycling, combustion and landfilling. The last two processes frequently spark public debate about health risks. Intensive efforts to eliminate these steps through recycling have demonstrably resulted in diversions of 50% or less; thus, the hierarchy still includes combustion and landfilling. Mitigation of their impacts on community health is the objective of added laws passed and regulations promulgated over the past decade. Paralleling these control efforts has been the development of multipathway assessment methodologies designed to provide at least a standard approach for comparing risks if not a reliable quantitative estimator of absolute risk. This paper updates previous risk-risk comparisons of landfilling vs. combustion of MSW by applying current methodologies to assess the technologies in the context of existing regulation. Risks of either technology fall within the regulatory precedents for acceptability during the operational phase (30 yrs) and the early closure phase (40 years), but the ultimate releases of leachate from the landfill generate potentially large risks over a time interval beyond this horizon. Alternative technologies are now available to improve substantially the risk management of both landfilling and combustion of MSW.

INTRODUCTION

Comparing the health risks from landfills with those of combustion provides a perspective of relative impacts that is usually omitted from site-specific discussions of waste management schemes. The analysis below compares order-of-magnitude risks associated with each technology assuming contemporary practices for new facilities. Upstream separation of the municipal solid waste (MSW) allows composting and recycling in either case of final disposal, and both of these processes should be regarded as intermediate steps between collection and final disposal. Preprocessing of refuse fuel at a combustion facility, however, affords opportunities for materials recovery and residue utilization that are not generally realized in current methods of landfilling.

In the early 1990s two papers[1,2] compared the human health risks of the two technologies using the methods and data then available. Other studies have compared the emissions from landfilling and combustion[3,4] with and without controls. The health risk comparisons assumed an environmental setting and a waste load to establish a common basis. They reported plausible limits of risks both with and without application of appropriate environmental controls. Actual data from sites in California and Massachusetts served as inputs for these studies. These papers support the conclusion that without controls, the landfill triggers much larger risks than combustion because of exposures through the groundwater pathway; however, the controls assumed for pollutant releases brought each technology into a range of acceptable risks. The time period of comparison was 70 years following the opening of each facility under the assumption that both the municipal waste combustor (MWC) and the landfill ceased to operate after the first 30 years of that period.

The present work updates these risk comparisons using current methods and data. Northern Illinois is the generic environmental setting for quantitative analyses of a landfill and a combustion facility, each of which accepts 2000 tons per day of refuse during an operating life of 30 years followed by closure. This update omits the uncontrolled cases that the earlier studies treated, and it simplifies the analyses to their bare essentials to enhance clarity. Many previous risk assessments demonstrate that cancer risks present a higher level of concern than non-cancer risks; hence, we limit the comparison to cancer risks. Wherever possible, regulatory requirements govern emissions and exposure parameters. For example, we use Maximum Available Control Technology (MACT) emission limits for combustor emissions, but there are no detailed limits on individual contaminants in landfill gas. In this case, we use pooled measurements for landfill gas since MACT standards will be not be promulgated until the year 2000.

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[1] Eschenroeder, A., S. Wolff, A. Taylor and D. Burmaster, *Health Risks of Alternative Methods of Municipal Solid Waste Disposal: A California Comparison,* Paper 90-182.3 presented at the 83rd Annual Meeting and Exhibition of the Air & Waste Management Association, Pittsburgh, PA June 24 - 29, 1990.

[2] Eschenroeder, A., S. Wolff, A Taylor and D. Burmaster, *Health Risks of Alternative Methods of Municipal Solid Waste Disposal: A Massachusetts Comparison,* Paper presented at the Society for Risk Analysis Annual Meeting, New Orleans, LA October 7 - 10, 1990.

[3] Jones, K., Comparing Air Emissions from Landfills and WTE plants, *Solid Waste Technologies* - March/April 1994, pp. 28 - 39.

[4] Licata, A. and D. Minott, Comparison of Air Emissions from Waste Management Facilities, *National Waste Processing Conference Proceedings, American Society of Mechanical Engineers*, March 1996.

Environmental models relate human exposures to releases of toxic substances, and consensus values of toxicity parameters serve as inputs to the dose response assessments. The air permit application and the health risk assessment for the Western Suburban Recycling and Energy Center (proposed to be sited in suburban Chicago, Illinois) provide emission rates and modeling (concentration and deposition) data for these pollutants[5]. The current edition of the standard regulatory reference for calculating air emissions AP-42[6] gives a suite of typical contaminant concentrations in landfill gas providing input for the formal hazard identification step. Similarly, a compilation of typically observed leachate contaminant levels[7] provides the database for the water pathway.

The next section ranks pollutant hazards by combining environmental release data with toxicity data to produce short lists of those pollutants that account for nearly all of the toxic threat. Exposure assessment and risk characterization form the subjects of another section. Concluding remarks summarize the results, discuss uncertainties and indicate refinements for future comparisons.

HAZARD AND DOSE-RESPONSE ANALYSES

Landfill gas generation and emissions

The hazard identification step of the risk assessment evaluates the degree of exposure potential as well as the toxicity of each substance. Toxic decomposition products of MSW find their way into both the landfill gas emitted into the air and the leachate discharged beneath the surface. The pollutants in the gas are volatile organic compounds (VOCs), and they are not taken up significantly by the soil or biota. Thus, for the gas the atmosphere is a dominant transport pathway, and inhalation is the principal gas exposure route. Collection systems capture most of the landfill gas and burn it in flares, engines or boilers, but actual practice suggests conservatively that about 25% of the gas still escapes as fugitive emissions[4,8]. Gas combustion destroys nearly all of the toxic gas

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[5] Holstein, E., and K. von Stackelberg, *Health Risk Assessment for the Proposed West Suburban Recycling and Energy Center,* Environmental Health Associates, P.A. report, August 1995.

[6] U.S. Environmental Protection Agency, *Compilation of Air Pollutant Emission Factors: Volume I: Stationary Point and Area Sources,* AP-42, Office of Air and Radiation, September 1997. Sec. 2.4

[7] U.S. Environmental Protection Agency, *Summary of Data on Municipal Solid Waste Landfill Leachate Characteristics - Criteria for MSW Landfills (40CFR Part 258),* EPA/530-SW-88-038, Office of Solid Waste, 1988.

[8] U.S. Environmental Protection Agency, *Air Emissions from Municipal Solid Waste Landfills - Background Information for Proposed Standards and Guidelines,* EPA-450/3-90-011a Office of Air Quality Planning and Standards, March 1991.

contaminants, but post combustion chemistry produces new ones; namely, dioxins[4,9]. In this context the term "dioxins" refers to the family of polychlorinated dibenzodioxins and polychlorinated dibenzofurans. The presence of dioxins suggests the need for an indirect pathway analysis involving the soil, the surface water and the food chain.

The design of the hypothetical landfill follows Resource Conservation and Recovery Act mandated subtitle D regulations. Its area is 810 hectares, and it occupies a square 900 m on a side. At a density of 650 kg/m^3, the refuse layer including cover is approximately 45 m thick at completion of landfilling.. Four power stations are deployed; one at the midpoint of each side of the square. Following U.S. practice in the majority of gas reclamation systems[10], this design employs reciprocating internal combustion engines . Coordinated operation of collection and engine operation maintains the 75% gas withdrawal averaged over the facility life; the remainder escapes as a fugitive emission. The liner is a composite of a flexible polymeric membrane and two feet of clay at a hydraulic conductivity of 10^{-7} cm/s with a leachate collection system that limits fluid head to 30 cm above the liner in compliance with 40CFR Part 258. Off site treatment of the leachate does not enter the risk assessment at this level of analysis, but it should be considered for site specific cases.

We focus first on the characterization of the landfill gas emissions. Two issues arise for the air pathways: (1) How much gas is generated each year from a landfill?, and (2) What is the level of toxic contamination of that gas? Both are addressed in U.S. EPA documentation[6]. The EPA's landfill gas model[11] provides values of annual gas generation in terms of the annual placement of refuse in the landfill. This same documentation tabulates extensively the trace contaminant levels in landfill gas. Our earlier papers[1,2] relied on the databases in our files for each of these issues. The authors of the EPA documentation emphasize that in both cases, the default parameters are averages over a very large and diverse sample of actual cases. This caution underscores the uncertainty in the data and models employed; however, these data adequately fulfill the needs of our generic analysis.

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[9] Caponi, F. E. Wheless and D. Frediani, *Dioxin and Furan Emissions from Landfill Gas-Fired Combustion Units*, Paper 98-RP105A.03 presented at the 91st Annual Meeting and Exhibition of the Air & Waste Management Association , San Diego, CA June 14 - 18, 1998.

[10] U.S. Department of Energy, Growth of the Landfill Gas Industry, Ch.10 in the *Renewable Energy Annual 1996*, available at www.eia.doe.gov/cneaf/solar.renewables/renewable.energy.annual on the internet, 1997.

Wherever possible in an actual site-specific analyses, it is preferable to use local measurements subjected to rigorous quality assurance and quality control procedures.

The gas generation history estimated by the EPA model covers both the operational and the post-closure phases of the landfill. The model simulates the placement and subsequent decay of refuse using a two parameter exponential equation. One parameter is the gas generation potential of the refuse, and the other is a decay rate constant. Figure 1 illustrates the results of the calculation using default values of the two parameters. The model sums over the staggered buildup and decay curves to produce the aggregated results used for emissions characterization. An average over a 70 year period satisfies the needs of the exposure assessment.

Table 1 identifies the air hazards and assesses the dose response relationships for landfill gas carcinogens based on the cancer unit risk factor. For each substance, this factor is the risk of getting cancer experienced over a lifetime of 70 years to a person inhaling air contaminated with 1 $\mu g/m^3$ of that substance. The product of the unit risk factor times the concentration of each substance in the gas forms a hazard rank value on each line of the table; these values appear in descending order of magnitude. Applying a dilution factor to this sum and factoring in the average gas flow, we implement a simple approach to exposure assessment, which is discussed below.

Gas engine emissions

A nominal value for the heat rate is our starting point for calculations of the gas engine exhaust emissions. The EPA handbook[12] on landfill gas development suggests that the typical performance is 12,000 btu/ kw-hr for the heat rate, which yields about 1.9 dscfm/kw when calculated from the F-value for the 50/50 split composition of CO_2 and CH_4 with a heating value of 500 btu/ft^3 using the formulas prescribed in the regulations (40CFR60.45). Twin stacks on each engine are 5m in height, 0.5m diameter, and the gas is at a temperature of 400°K. Any attempt to simulate the utilization of discrete engines is probably not justified in light of other uncertainties in the analysis; e.g., the 75%

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[11] U.S. Environmental Protection Agency, *User's Manual Landfill Air Emissions Estimation Model, Version 1.1,* Control Technology Center, available at www.epa.gov on the internet September, 1997.

[12] U.S. Environmental Protection Agency, *Turning a Liability into an Asset: A Landfill Gas-to-Energy project Development Handbook*, EPA 430-B-96-004,Office of Air and Radiation, September 1996.

collection efficiency. The engines emit 2% of the mass flow of each toxic organic compound entering in the fuel in accordance with regulatory requirements, and 100 pg TEQ/Nm^3 characterizes a comparative standard[8] for bounding the dioxin concentration in the exhaust. Ref. 8 indicates that this is currently the most stringent of all international emission standards. In the absence of data, the dioxin speciation for the engine exhaust is assumed to follow the same pattern as that of the resource recovery emissions. The rationale underlying this default assumption is to assure a common basis of comparison. The detailed development of the speciation appears below under the discussion of resource recovery emissions. The gas engine PCDD/F speciation pattern is identical to that of the resource recovery for the present purposes.

Leachate discharges

As in the case of gas emissions, both quantity and quality considerations characterize the water discharges. The water balance on the landfill primarily establishes the quantity of leachate generated; however, the flow through the bottom liner system, in the final analysis, controls the releases to the environment. Two possibilities present themselves for quantifying such releases: (1) selection of the *de minimis* polymeric liner leakage rate acceptable under regulatory assumptions[13] or (2) calculation from Darcy's Law of the steady state flow through a liner meeting regulatory hydraulic conductivity requirements[14]. In our choice of the latter alternative, we use the Lee and Jones-Lee[14] prediction of a 25 year migration time through the 61 cm thick clay liner; this establishes a source flux for leachate discharge down to the groundwater. EPA's polymeric liner leakage study[13] states, for example, that *"the permeation rate for TCE through HDPE liners is approximately four orders of magnitude greater than that of water"*. Moreover, EPA's solid waste disposal facility criteria document[15] states that *"...even the best liner and leachate collection system will ultimately fail due to natural deterioration, and recent improvements in municipal solid waste landfill containment suggest that releases may be delayed by many decades at some landfills"* In

[13] U.S. Environmental Protection Agency, *Technical Considerations for De Minimis Pollutant Transport through Polymeric Liners* EPA/600/2-88/042 Office of Research and Development, August 1988.
[14] Lee, G. and A. Jones-Lee, *Assessing the potential of Minimum Subtitle D Lined Landfills to Pollute: Alternative Landfilling Approaches,* Paper 98-WA71.04(A46) presented at the 91st Annual Meeting and Exhibition of the Air & Waste Management Association , San Diego, CA June 14 - 18, 1998.

another document[16], the agency states that "*Once the unit is closed, the bottom layer of the landfill will deteriorate over time, and, consequently, will not prevent leachate transport out of the unit.*" Thus, a combination of factors suggests, at best, only a temporary detention of leachate by the polymeric liner; therefore, the hydraulic conductivity, the head and the thickness of the clay liner establish the rate limiting resistance to leachate discharge. The EPA criteria further state on page 33359 of ref. 15 that the agency's standard for conducting risk assessments is to select the "*...highest lifetime health risk that would be experienced over a 300-year simulation period.*" Our assumption that the clay liner remains intact over the entire 300-year period is, perhaps, unduly optimistic. This discharge rate further implies that, below the membrane liner, the organic pollutants essentially travel at the same speed as the leachate without any retardation. The low organic content of the clay and the soil at the bottom of the vadose zone suggests that retardation is negligible. The continued long term production of the organic pollutants is apparently sustained[14] by the "dry tomb" operation under the present regulatory doctrine.

Water quality considerations balance the concentration against the toxicity of the contaminants; another example of combining the hazard identification with the dose response assessment. Table 2 evaluates the potential health hazards of household water use from sources contaminated by the leachate plume. The multipathway factor accounts for the possible exposures beyond ingestion; namely, inhalation of vapors indoors, dermal absorption from water in the shower and inhalation of volatilized compounds in the shower. The unit risk factor is the basic ingestion toxicity. In a manner similar to that with the landfill gas, the summation of the hazard rank values gives the lifetime risk of using undiluted leachate for a domestic water supply. In the exposure assessment the time delay in arriving at a well intake and the plume dilution reduce the risk below this upper limit value. The Integrated Risk Information System[17] is the source of the unit risk factors in Table 2. These data represent the consensus results of the dose-response assessment for the purposes of comparative analyses.

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[15] U.S. Environmental Protection Agency, 40 CFR Parts 257 and 258 Solid Waste Disposal Facility Criteria; proposed rule, *Federal Register* v. 153, n. 168, pp. 33314 - 33422,August 30, 1988.
[16] U.S. Environmental Protection Agency, *Criteria for Municipal Solid Waste Landfills*, Washington, DC, July 1988
[17] U.S. Environmental Protection Agency, *Integrated Risk Information System,* available on the internet at www.epa.gov, data from 1998 downloads.

Resource recovery emissions

Combustion contaminants considered here include: inorganic substances (beryllium, cadmium, chromium VI, nickel being the carcinogens among them), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Table 3 lists the emission rates and cancer slope factors of the carcinogen emissions from the resource recovery facility. Although long lists of additional compounds may be compiled, the emissions chosen essentially drive the risk. As we know from many municipal waste combustor (MWC) risk assessments, the PCDD/Fs dominate the cancer probability mainly through indirect (non inhalation) exposure pathways[18]. For this reason they receive an especially detailed treatment in our analysis. The WESREC data[5] serve as our starting point for the total emissions of these substances under the assumption of conforming with the limit set by MACT for large MWCs under 40 CFR Parts 51, 52, and 60 in response to sec. 129 of the Clean Air Act Amendments of 1990. The rationale for this standard of 30 ng/dscm for new sources appeared in the Federal Register[19] in 1991. EPA's dioxin reassessment document[20] provides a homologue mass distribution based on data from ten MWCs. The combinatorial probability of occurrence serves as a basis for the distribution of congeners with chlorine substituted in the 2,3,7,8 positions versus the others. The international convention for toxic equivalency factors[19] provides weighting factors, and the draft guidelines for health risk assessment for hazardous waste combustors[21] provides the 2,3,7,8 TCDD cancer slope factor Properties of Aroclor 1254, a commercial mixture, serve as a prototypical PCBs, and properties of benzo-a-pyrene establish the surrogate PAH except for its toxicity. Following Holstein and von Stackelberg,[5] 2% of the bulk PAH is assigned the cancer slope factor of benzo-a-pyrene. The PCBs and the PAHs do not receive the same level of detail as the PCDD/Fs because they play less significant roles as determinants of risk. Just as the leachate treatment plant was omitted from the landfill analysis, the off site landfills

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[18] Levin, A., D. Fratt, A. Leonard, R. Bruins, and L. Fradkin, Comparative Analysis of Municipal Waste Combustors, *Journal of the Air and Waste Management Association,* v. 41 n. 1 pp. 20 - 31, 1991.

[19] U.S. Environmental Protection Agency, Standards of Performance for New Stationary Sources and Final Emissions Guidelines; Final Rules, *Federal Register* v. 58, n. 28, Monday February 11, 1991 pp. 5488 - 5527.

[20] U.S. Environmental Protection Agency, *Estimating Exposure to Dioxin-Like Compounds Volume III: Site-Specific Assessment Procedures* EPA/600/6-88/005Cc Office of Research and Development, June 1994.

[21] U.S. Environmental Protection Agency, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Volume III* EPA530-D-98-001A Office of Solid Waste and Emergency Response, July 1998.

for ash and overflow refuse are omitted from the resource recovery assessment at this level of analysis.

EXPOSURE ASSESSMENT AND RISK CHARACTERIZATION

Landfill gas fugitive and engine pollutants

The design features of the landfill define the geometrical relationships of size, shape and source distribution. To recap, the fill area is a square 900 m on a side, and the gas engines are clustered at four power stations deployed at the midpoints of each side. The model inputs an area source for the surface fugitive emissions and four aggregated point sources; one for each power station. The EPA Industrial Source Complex model provides long term averages of concentration and deposition at an array of receptor points. The closest receptors are on a square 1200 m on a side that is concentric with the landfill. This establishes the 150 m point of compliance (POC) distance envisioned in the Subtitle D regulations on all sides of the facility. The relationship between concentration and deposition at each receptor considers both dry and wet transport to the surface in the same way as for the resource recovery case in order to assure a fair comparison. A pool of five years of meteorological data drives the air quality modeling.

The unit concentration at the maximally exposed "individual" becomes the starting point for the exposure assessment. Combined with the emission rate, this result gives the dilution rate for the contaminants in the landfill gas both from the area source and from the engines. Using the unit concentration for the engine emissions, we simply scaled the multipathway risks attributable to the resource recovery facility down to the size of the landfill power plant.

The calculated data for ambient conditions from p 3-42 of ref. 21 serve as the bases for assigning the vapor to particle mass ratio for each dioxin homologue. In our modeling results, the vapor component of dioxin deposits at a rate of 1 cm/sec as a conservative value among the higher values in the observed deposition rates[22] listed in Table 6-3 on page 6-28 of the cited reference. Vapor deposition is not currently treated by the recommended regulatory models, but methodologies are

under development. The volatile contaminants have negligibly small indirect exposure contributions; therefore, they are neglected for all but the inhalation pathways. In contrast, dioxin exposures due to emissions from the engines involve many indirect pathways. Included in this analysis are soil ingestion, soil dermal contact, produce ingestion, water ingestion, fish ingestion and mother's milk ingestion. Representations of indirect dioxin exposures in the landfill engine spreadsheets are identical to those developed for resource recovery exposure and are described under that subsection below.

Risk characterization combines the exposure assessment results with those of the dose-response assessment. For the pathways in this scenario associated with landfill gas fugitive and engine emissions the incremental lifetime cancer risk is 5 in a million.

Leachate contaminants

Any domestic water source affected by the leachate becomes a transfer medium for dissolved contaminants so that drinking water ingestion could become the primary exposure route that immediately comes to mind. For this portion of the exposure assessment the focus is on groundwater. Again, the downgradient POC receptor distance of 150 m controls the plume dispersion thereby determining the dilution ratio over time. The sensitivity of this pathway to site specific factors suggests the treatment of several subcases. The ranges of parameters span aquifer thicknesses of 10, 30 and 100 m, and groundwater velocities of 0.3, 1 and 3 m/yr. These are consistent with the expectation of sandstone water bearing rocks in northern Illinois[23] with appropriate ranges of peizometric slopes and hydraulic conductivities[24]. Using longitudinal dispersivities of 10% of travel distances[25] and transverse dispersivities of 30% of longitudinal dispersivity[26], calculations show that the plume fills the aquifer vertically at the trailing edge of the source, and that horizontal dispersion has negligible effect on the receptor set at the POC at the

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[22] U.S. Environmental Protection Agency, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions*, NCEA-C-0238 (an update to EPA/600/6-90/003) National Center for Environmental Assessment, February, 1998.

[23] Heath, R., *Groundwater Regions in the United States*, U.S. Geological Survey Professional Paper 2242, 1984.

[24] Freeze, A. and J. Cherry, *Groundwater*, Prentice-Hall, Englewood Cliffs, NJ, 1979.

[25] Pickens, J. and G. Grisak, Scale-Dependent Dispersion in a Stratified Granular Aquifer, *Water Resources Research*, v. 17, n. 4 pp. 1191-1211, August 1981.

plume centerline. The longitudinal dispersion of the leachate front is symmetric; hence, for 300-year exposure calculations, a simple rectangular prism approximation suffices. Mass balances using the locally averaged rainfall from two stations over 5 years (873 mm) with the assumption of 15% infiltration determine the overall dilution of the leachate reaching the POC receptor. Figure 2 summarizes the risk calculations using the product of dilution factor and the hazard rank sum derived in Table 2. The results range from about 1000 to 6000 in a million cancer risk; values much larger than those associated with the gas and engine emissions.

Resource recovery exposures and risks

Wet and dry particulate deposition as well as vapor deposition feed the indirect exposure pathways. As in the case of the landfill, maximum exposure determines the choice of deposition and air concentration values. Following the conservatism of ref. 5, our calculation considers these two maximum values to coincide in space, when in fact they do not. Inhalation exposures assume 100% bioavailability in the dose calculations, which depend on body weight and inhalation rates. Lifetime incremental inhalation risks total out to 1 in a million.

A box model describes the soil pathway with a series of loss processes balancing the deposition inputs[22]. Fate and transport calculations indicate that the extreme persistence of the three families or organic carcinogens considered leaves biodegradation as the default process for removing pollutants from the soil box. A box depth of 20cm describes tilled soil and 1cm describes untilled soil. The former determines root transfer to plants and the latter, personal exposures like direct ingestion or dermal absorption. Data discussions in the multipathway methodology document[22] suggest a conservative choice of a 10 year half life for dioxins, and the sludge assessment document[27] provide values of about 0.2 year for BaP and 6 years for PCBs. Inorganic pollutants fill the box for the 30 year facility operating life, and the concentration remains at this level with no removal processes for the remaining 40 years of the exposure period. Soil ingestion and dermal absorption are the direct pathways considered here. Each of these pathways generates an incremental lifetime cancer risk of 0.3 in a million from the resource recovery facility.

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[26] Bredehoeft, J. and G. Pinder, Mass Transport in Flowing Groundwater *Water Resources Research,* v. 9, n. 1, pp 194-210, February 1973.

Produce. Including fruit and vegetables, is divided into exposed and protected categories for particulate deposition and vapor infusion calculations. The principal risk-driving chemicals display negligible transport via root uptake. Child and adult consumption rates[21] with the produce concentrations determine exposures, and multiplication by ingestion slope factors determine risks. Produce pathways cause an incremental lifetime cancer risk of 4.6 in a million. Air to leaf vapor transfer is a significant contributor.

For water and fish consumption, the waterbody of choice is a 3.2 hectare pond year placed at the point of maximum concentration and deposition which experiences 23 water changes per year. A level II fugacity model[28] partitions the pollutants between the 0.9 m water column and the 0.1 m sediment layer. Erosion inputs to the water are unimportant because of the low slopes of the watershed area. Sediment transport into the pond is limited by long detention times in upstream wetlands, and the dam that forms the pond prevents pond sediment from exiting; hence biodegradation governs pollutant decay in the sediment layer. Sediment-based bioaccumulation factors[22] relate fish tissue concentration to those in the environment. As in the case of produce consumption, differentiation between child and adult doses carry through the calculation. Incremental lifetime cancer risks are 1.4 in a million for water ingestion from the pond and 3.7 for fish ingestion from the pond.

Mother's milk exposure calculations use the method of Smith[29] as applied in ref. 5. In this approach, the average daily adult dose rate of PCBs and dioxins drive a compartment model set up between the mother's body and that of the infant. Half lives of the pollutant in the mother's body and the fraction of maternal body weight represented by fat are the main parameters of the mass balance. The pollutants partition with the fat in the lactation process, so that the amount of fat in the milk combined with the nursing rate and duration establishes the lifetime dose increment to the

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[27] U.S. Environmental Protection Agency, *Land Application and Distribution and Marketing of Sewage Sludge - Technical Support Document,* PB89-136576, Office of Water Regulations and Standards, 1986.
[28] Mackay, D., Finding Fugacity Feasible, *Environmental Science and Technology,* v. 13, pp. 1218-1223, 1979.
[29] Smith, A., Infant Exposure Assessment for Breast Milk Dioxins and Furans derived from Waste Incinerator Emissions, *Risk Analysis,* v. 7, pp 347-353, 1987.

infant. This pathway accounts for 3 in a million incremental lifetime cancer risk, bringing the grand total of resource recovery risks to 14 in a million for the maximally exposed individual.

CONCLUDING REMARKS

Comparative health risks for facilities under current regulations

In health risk assessments the absolute numerical values are extremely uncertain. In the analyses presented above, many of the pitfalls are avoided by focusing on comparative values rather than absolute values. Every effort is made in a comparative risk assessment to enforce a consistent set of ground rules as to facility size, duration of operation, physical environment and transport of pollutants. A summary of the comparative risks of two technologies for the final disposal of municipal solid waste is very simple:

Incremental Lifetime Cancer Risks for Maximally Exposed Individual

	Risks from air emissions	Risks from water discharges
Subtitle D Landfill	5 in a million	1000 in a million
MACT Combustor	15 in a million	0

Considering only those exposures originating with atmospheric emissions, both technologies fall into a range of acceptable risks as defined by the decisions of most regulatory agencies. But the risks over the long term from groundwater exposures to landfill leachate dominate the other pathways by orders of magnitude. This conclusion emerges over a wide range of plausible hydrogeological parameters. Indeed, the Lee and Jones-Lee paper[14] marshals forth an impressive array of factual evidence raising doubts about the present regulatory strategy. The heirarchy of waste management processes embraced by most environmental advocates and by the U. S. Environmental Protection Agency is: 1. Source reduction, 2. Recycling , 3. Combustion

Beyond the striking comparative results, several improvements and expansions would improve the approach to the analysis presented above. An investigation of uncertainty and variability will shed light on the validity of these conclusions. The approach could range from a simple sensitivity study to a formal Monte Carlo analysis. Also, improvements in the physical and chemical descriptions of

the phenomenology will clarify certain questions: How much wake downwash effect does the elevated landfill profile exert on the gas engine plumes? Rough estimates for the POC suggest increases in impact by perhaps threefold for enhanced plume dispersion. What is the speciation of the PAHs and the PCBs? -- How do they affect toxicity profiles? What is the fracture failure rate of clay liners? -- What is the upset frequency for MWCs and what are the off design emissions? - What additional risks occur as a result of metal emissions observed in landfill gas combustion products?[30] It is difficult to imagine refinements in the calculations that will change the large groundwater risks.

Beyond current regulations - Pollution prevention

The landfill alternatives advocated by Lee and Jones-Lee[14] could potentially reduce the log term groundwater risks markedly. One aspect of this approach abandons the "dry tomb" philosophy and substitutes a wet cell technology. One objective of introducing additional water is the acceleration of anaerobic activity of methogenic bacteria. The idea is that faster decomposition and gas generation would detoxify the refuse residue on a time scale comparable with that for the decay of the flexible membrane liner. A groundwater and leachate pump and treat operation maintains the water circulation through the cell. This operation is conducted in a double composite liner system. An engineered high conductivity aquifer between the liners provides a hydraulically isolated reservoir for rapid detection and collection of leachate leakage. Lee also recommends shredding of the waste prior to landfilling to enhance decomposition so that the gas production cycle may run as little as 5 years. It is suggested that leachate recycle be sustained until gas generation ceases. At that time flushing with fresh water purges remaining leachate to be pumped and treated as part of the closure protocol. The authors of ref. 14 are careful to point out that this intense pollution prevention program comes at a cost, but it is likely that there is justification on the basis of balancing externalities. That is, the avoided long term exposures of humans and ecosystems to polluted groundwater linked with the enhanced reclamation of the land once used for refuse burial seem like worthwhile returns on the investment.

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[30] Carpenter, J. and L. Gammie, *Metals Emissions from Landfill Gas Combustion,* Paper 97-TA49.05 presented at the 90th Annual Meeting and Exhibition of the Air & Waste Management Association , Toronto, Ontario, Canada, June 8 - 13, 1997.

Advanced combustion technology presents encouraging possibilities for substantial risk reduction through pollution prevention for MWCs. The chief difference between these improvements and those described above for landfills is that they actually save money up front at the point of investment. Energy Answers Corporation has pioneered the "shred-and-burn" combustion technique in its design and operation of the SEMASS resource recovery plant in southeastern Massachusetts. Pickers remove large objects that foul grates and destroy furnace linings, and hammermill shredders reduce the waste to sizes 15 cm and smaller. This allows magnetic separation of the large majority of ferrous metals prior to firing. An air suspension firing system prevents slagging because the combustion zone is lifted up from the grates. Heat release rates per unit area run three times that of mass burn units so that footprint areas are dramatically reduced. Sloping grates are unnecessary thereby reducing boiler house height by five or six meters. Enhanced combustion efficiency reduces the need for excess air so that ducting, fans, air pollution control equipment and breeching are reduced in size. All of these attributes combine to lower debt service, direct operating and maintenance costs while at the same time reducing emissions of hazardous air pollutants. SEMASS pollutant emissions of hazardous air pollutants are considerably below the MACT standards. Over a four year average of stack test data dioxin emissions are 6% of the MACT limit; cadmium, 7%; mercury 7%; and lead, 17%. These degrees of control are attained with a spray dry absorber followed by a fabric filter. The risks delineated above for MACT are around the de minimis level, but the shred-and-burn system is well below these levels.

TABLES

Table 1

LANDFILL GAS INHALATION CARCINOGEN HAZARD RANKING BASED ON THE U.S. EPA DEFAULT CONCENTRATIONS[6]

Compound	Average Conc. µg/m3	Unit Risk Factor m3/µg	Hazard Rank Value
acrylonitrile	1.40E+04	6.80E-05	9.50E-01
1,1,2,2 - tetrachloroethane	7.75E+03	5.80E-05	4.50E-01
Bromodichloromethane	2.13E+04	1.80E-05	3.84E-01
tetrachloroethylene	2.57E+04	5.50E-06	1.42E-01
benzene	1.14E+04	8.30E-06	9.49E-02
vinyl chloride	1.91E+04	2.60E-06	4.96E-02
1,2 - dichloroethane	1.69E+03	2.60E-05	4.39E-02
1,1 - dichloroethylene	8.06E+02	5.00E-05	4.03E-02
trichloroethylene	1.54E+04	1.60E-06	2.47E-02
methylene chloride	5.05E+04	4.70E-07	2.37E-02
1,4 - Dichlorobenzene	1.28E+03	5.70E-06	7.32E-03
chloroform	1.49E+02	2.30E-05	3.43E-03
carbon tetrachloride	2.56E+01	1.50E-05	3.84E-04
		Sum of hazard rank values =	2.21E+00

Table 2

WATER PATHWAY EVALUATION OF LEACHATE CONTAMINANTS AT THE SUBTITLE D LANDFILL

Substance	Average Leachate Conc[7] mg/l	Unit Risk Factor[17] l/mg	Multi-pathway Factor	Hazard Rank Value
vinyl chloride	36.1	5.40E-05	1.04	2.0E-03
carbon tetrachloride	202	3.70E-06	2.84	2.1E-03
benzene	5.6	8.30E-07	1.64	7.6E-06
arsenic	41.8	5.50E-05	1.00	2.3E-03
1,2-dichloroethane	1841	2.60E-06	1.12	5.4E-03
1,1,2-trichloroethane	210	1.60E-06	1.12	3.8E-04
1,1,2,2-tetrachloroethane	210	5.80E-06	1.03	1.3E-03
	Total of Hazard Rank Values=			1.3E-02

Table 3

EMISSIONS AND SLOPE FACTOR DATA FOR
THE RESOURCE RECOVERY FACILITY

Contaminant	Annual Average Emission Rate[5] (g/sec)	Cancer Slope Factor[17,21] Inhalation $(mg/kg\text{-}d)^{-1}$	Oral $(mg/kg\text{-}d)^{-1}$
Arsenic	8.62E-04	1.5E+01	1.5E+00
Beryllium	8.62E-05	8.4E+00	8.4E+00
Cadmium	8.62E-04	6.3E+00	
Chromium VI	3.10E-04	4.2E+01	
Nickel	8.62E-03	8.4E-01	
PAHs	4.32E-03	7.3E+00	7.3E+00
PCBs*	8.62E-06	4.0E-01	4.0E-01
TEQ as 2,3,7,8-TCDD	1.24E-09	1.5E+05	1.5E+05
Total PCDD/Fs	4.30E-08		

* Cancer slope factor for PCBs ingested in food = 2 $(mg/kg\text{-}d)^{-1}$

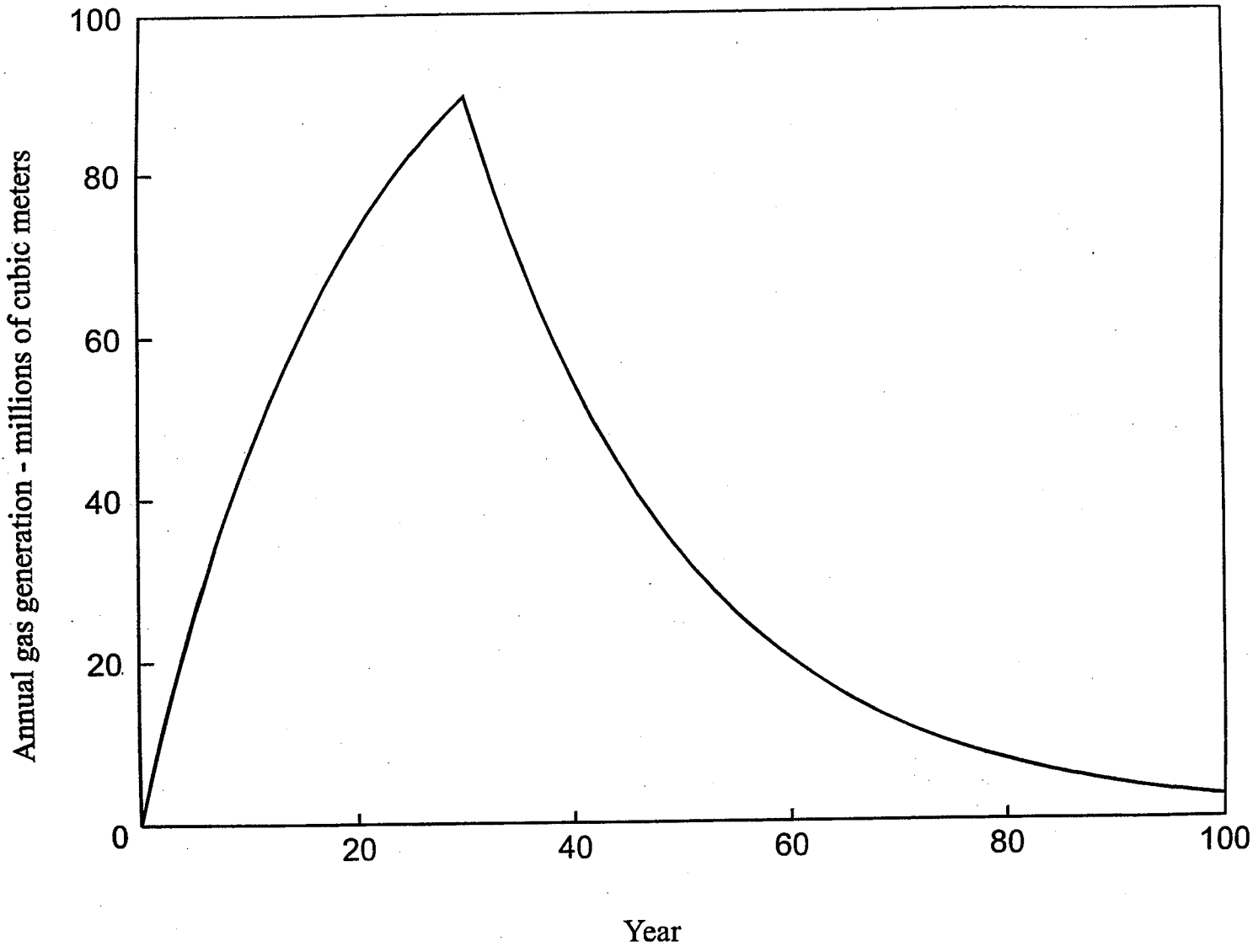

Fig. 1 - ANNUAL TOTAL GAS GENERATION from a 2000 ton per day landfill

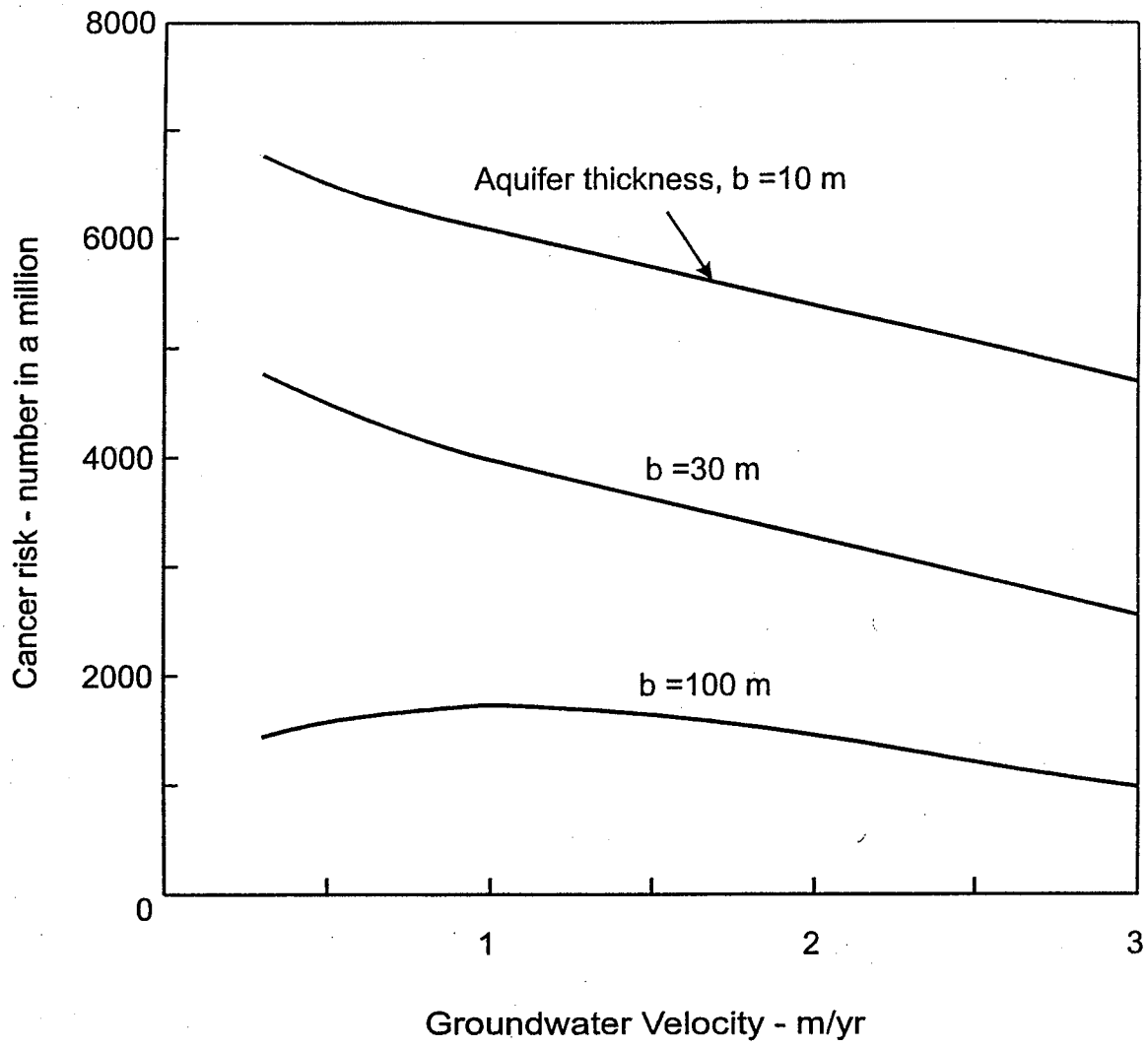

Fig. 2 - GROUNDWATER CANCER RISKS FROM LANDFILL LEACHATE