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Landfill Leachate Production and Gas Generation Numerical Model

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Landfill Leachate Production and Gas Generation Numerical Model

by

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ABSTRACT
LANDFILL LEACHATE PRODUCTION AND GAS
GENERATION NUMERICAL MODEL

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Old Dominion University

Advisor: Dr. A. Osman Akan

Numerous processes occur in landfills which lend themselves to modeling. Many of the processes are mutually interdependent. An unsteady numerical model is developed combining the major processes. The three-dimensional moisture transport equations and boundary conditions are solved using an implicit finite difference scheme. The boundaries are determined through a two-dimensional runoff model for the landfill surface and a one-dimensional leachate liner flow model at the bottom of the landfill. The runoff model accounts for evapotranspiration, runoff, infiltration, and leachate recirculation. Richard's equation is solved for saturated and unsaturated vertical flows and Darcy's Equation is solved for lateral flow between adjacent saturated landfill cells. Results of the moisture flow are used to solve contaminant production and transport equations. Contaminant production uses moisture flow and previous leaching history to generate source terms. The source terms and recirculated contaminants are used to implicitly solve contaminant transport equations which account for advection, diffusion, and dispersion of the contaminant. Landfill temperatures are predicted by solving an energy equation implicitly. Temperatures are combined with moisture content and gas production history to determine gas generation. The model is applied to three Wisconsin lysimeters and a Kentucky landfill to demonstrate the simulation of leachate and contaminant production and transport. Comparison to the HELP water balance model is also done for a Wisconsin lysimeter. The model is also applied to an existing landfill to demonstrate the gas generation portions of the model.

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NOMENCLATURE

a	stochastic values for wastes (2.5.1)
a	soil diffusion constant (3.5.3)
a_i	height of liner above the datum
a_s	maximum fraction of waste chemical oxygen demand
A	Fourier coefficient (2.2.2)
A	Arrhenius equation coefficient (2.5.2)
A	gas production parameter (2.5.3)
A	elevation of liner peak above datum (3.4.1)
AA_j	derivative terms for equation at node j
A_i	slope of saturation vapor pressure
A_x, A_y	cross sectional flow areas per width δ for runoff
b	stochastic values for wastes (2.5.1)
b	soil diffusion constant (3.5.3)
B	Fourier coefficient (2.2.2)
B	grid width (2.2.3)
BB_j	derivative terms for equation at node j
BOD_5	Biological Oxygen Demand (5 day)
c	stochastic values for wastes
C	contaminant concentration (2.3.2, 3.5.3)
C	remaining gas production substrate (2.5.3)
C	contaminant production rate coefficient (3.5.5)
C_p	specific heat
C_w	specific heat of water
C_{waste}	specific heat of waste
C_{rain}	contaminant concentration of rain
CC_j	derivative terms for equation at node j
CH_4	methane

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CO	carbon monoxide
CO_2	carbon dioxide
COD	chemical oxygen demand
°C	degrees Celsius
d	liner thickness (2.4, 3.4)
d	stochastic values for wastes (2.5.1)
D	diffusion coefficient
D_h	hydraulic dispersion coefficient
D_o	equivalent diffusion coefficient in free water
D_s	soil diffusion coefficient
D_z	vertical diffusion coefficient
DD_j	resultant terms of equation at node j
e	fraction of waste converted to methane gas
Ea	energy of activation
E_{0i}	potential evaporation on day i
f	cell decay rate
f_j	function at point j
$\tilde{f}_j$	finite difference residual
FC	field capacity
F_{mc}	gas production rate due to moisture content
g	acceleration of gravity
G	psychometric constant (2.2.2)
G	volume of gas produced (Appendix A)
G_p	gas previously produced
G_{ult}	ultimate expected gas production per unit mass
h	months in Fourier analysis (2.2.2)
h	head (2.2, 2.3, 3.2)
h	leachate depth above the liner (3.4)
h_c	leachate depth over the liner
H	water surface elevation
H_i	net solar radiation on day i (langleys)
H_j	specified head
H_2	hydrogen gas
H_2S	hydrogen sulfide
i	day index in Fourier analysis (2.2.2)

i,j,k	indices denoting spatial location
i_e	rainfall excess rate
I	specified infiltration flux()
I_e	source/loss terms from rainfall and infiltration
J	joules
K_1	gas production rate
K_2	adjusted gas production rate
k	Nikuradse's roughness (2.2.3)
k	gas order of reaction coefficient (2.5.3)
k_i	heat diffusion coefficient in respective direction
K	channel conduction parameter (2.2.3)
K	methane reaction rates (2.5.2, Appendix A)
K	maximum rate of substrate utilization (2.5.3)
$°K$	degrees Kelvin
K_d	saturated hydraulic conductivity of drainage layer
K_L	liner saturated hydraulic conductivity
K_r	relative permeability of the cell
K_s	saturated permeability of the landfill cell
K_s	waste concentration
L	albedo for solar radiation (2.2.2)
L	water source term (3.2)
L	length of liner (3.4.1)
L	volume of gas remaining to be produced (Appendix A)
L_n	lateral flow into node from adjacent node
m	parameter exponent for runoff (2.2.3)
m	porous soil parameter (2.2.4, 3.2, 4)
m	slope of gas production curve (3.6.2)
n	manning roughness coefficient (2.2.3, 3.3.1)
n	temporal step index
n	stochastic values for waste (2.5.1)
n	number of data points (4.2.1.1)
N	number of nodes in column
NR	number of rows
NC	number of columns
N_2	nitrogen

O_2	oxygen
p	pressure head
P	gas previously produced
P_c	total contaminant previously produced
Q	drainage flow rate along liner surface (2.4, 3.4.1)
Q	maximum amount of gas per mass (2.5.3, 3.6.1)
Q_{d1}	liner flow into element from upstream
Q_{d2}	liner flow out element downstream
Q_x, Q_y	runoff discharges per width
$\dot{q}$	heat flow rate
q_b	leakage rate through the barrier
q_t	total overland flow
q_x	overland flow in respective direction
q_y	overland flow in respective direction
R	rate of leachate percolating from waste (2.4, 3.4)
R	gas constant (3.6.2, 2.5.2)
R_x, R_y	hydraulic radius
R_i	solar radiation on day i
Rx	reaction source terms
s	fraction of waste chemical oxygen demand
s	deviation data (4.2.1.1)
S	concentration of substrate (2.5.3)
S	liner slope (3.4.1)
S_f	friction slope
S_{fx}, S_{fy}	friction slopes for runoff
S_w	degree of saturation of the cell
ΔS_{sw}	change in water stored in solid waste
S_{ox}, S_{oz}	bed slopes in the respective directions
t	time
Δt	time step
T	temperature
T_{atm}	atmospheric temperature
T_i	mean temperature in °K on day i
TOC	total organic carbon
ULT	total contaminant per mass of dry waste

v	fluid velocity
v_z	vertical advective velocity
V	average interstitial flow velocity (3.5.3)
V	total volume of leachate (3.5.5)
$\bar{V}$	average annual value
V_h	mean monthly value for month h in Fourier analysis
V_i	interpolated value on day i
W	weight
W_A	water from rainfall or snow
W_B	water leaving the bottom
W_{CM}	water cover material
W_E	water lost due to surface evaporation
W_{LG}	water lost in formation of landfill gas
W_{SW}	water in incoming solid waste
W_{TS}	water in incoming treatment plant sludge
W_{WV}	saturated water vapor lost with landfill gas
x,y	lateral distance (2.2, 3.2, 3.3)
x	distance along the liner (2.4, 3.4.1)
x,y,z	spatial coordinates (2.5.2)
X	concentration of microorganisms
y	depth of leachate on the liner
y_{model}	simulation data point
y_{actual}	experimental data point
Y	depth of standing of flowing water on surface
Δw	perpendicular width of liner node
$\Delta x,\ \Delta y$	distance between node points
Δx_i	lateral length for the element in x direction
Δy_k	lateral length for the element in y direction
z	vertical distance
Δz	vertical thickness of landfill node
α	porous soil parameter
α_y, α_x	open channel flow friction parameters
δ	lateral width of runoff cell
λ	Darcy-Weisbach friction coefficient (2.2.3 Table 2)
λ	porous media empirical constant (3.5.3)

ϕ	porosity of the waste layers
ϕ_e	effective porosity of the drainage layer
ρ	density
ρ_w	density of water
ρ_{waste}	density of waste
θ_c	solids retention time, days

1. INTRODUCTION

Landfills have served for many decades as the ultimate disposal sites for residential, commercial, and industrial (both innocuous and hazardous) wastes. Landfill technology has evolved from the open dump, in which the wastes were burned to reduce the volume, to highly engineered sites designed to minimize the impact on the environment. Improvements in landfill engineering have been primarily aimed at reducing leachate production, collecting and treating leachate, and limiting leachate discharge to the assimilative capacity of the surrounding soil (Farquhar 1989). This has been accomplished through leachate collection systems, liner and cover designs, and leachate monitoring systems. Environmental concerns and legislation regarding the operation of landfills have become very stringent, thus boosting the requirements. Instances of uncontrolled landfill leachate reaching groundwater sources and uncontrolled gas generation has caused great concern. This demonstrates the need for tools to predict the performance of landfills for future designs, planning, and completed landfill site closings.

The performance of a landfill can be measured from the leachate and gas generation of the landfill. Farquhar (1989) points out that regardless of whether leachate is collected and treated or discharged

to the soil, or whether gas is flared or used as an energy source, it is imperative to have estimates of the leachate and gas flows as the landfill is developed, closed, and for post-closure purposes. Hence, the development of computer models to make these type of predictions is both useful and necessary.

1.1 LITERATURE SURVEY

A literature survey has been completed to identify the major processes occurring within a landfill and the tools used to characterize these processes and designs. The different processes and designs include: precipitation, runoff, evapotranspiration, infiltration, leachate generation, leachate transport, leachate collection, waste biodegradation, gas generation, gas transport, landfill liner design, and contaminant transport. Mathematical and computer models have separately been developed to describe some of these processes and are summarized.

The development of models to predict the leachate generation and flow as well as gas generation is relatively new. Fenn et al. (1975), Dass et al. (1977), Perrier and Gibson (1982), Gee (1981), Lu et al. (1981), Kmet (1982), and Schroeder et al. (1983a, 1983b) reported models in the literature which were formulated to predict leachate flow discharging out of landfills based on a hydrologic water balance method (WBM). This method, first proposed by Fenn et al. (1975), is a manual procedure generally solved using monthly averaged values of the amount of water percolating through the solid waste. This

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percolation quantity is determined to be the total precipitation (P), minus the runoff (RO), the change in soil moisture content (MC), and the evapotranspiration (ET). In water balance methods, the process of moisture passage through the solid wastes and barriers is not considered (Ahmed *et al.* 1992; Farquhar 1989).

There have also been computer models developed using the Water Balance Method as a basis with various modifications. Gee (1981) used two variations of the Water Balance Method to predict leachate flow at an active landfill and compared the results to actual measurements made in the field. The predictions were approximately a factor of two higher than actual. Lu *et al.* (1981) conducted similar comparisons at five landfills using 25 different methods to estimate the various terms in a Water Balance Method (precipitation, runoff, infiltration, evapotranspiration, initial moisture content, soil storage, and percolation). Again, the average leachate flow estimates were in error by a factor of two, however, the poorest estimates were up to 100 times greater than the measured leachate flows (Farquhar 1989).

Kmet (1982) used a Water Balance Method with modifications to account for infiltration and runoff during winter conditions. He simulated leachate production in Ham's (1980) eight field lysimeters with excellent success. The Hydrologic Simulation of Solid Waste Disposal Sites (HSSWDS), a model developed by Perrier and Gibson (1982), and the Hydrologic Evaluation of Landfill Performance (HELP), reported by Schroeder *et al.* (1983a), are currently the most widely accepted Water Balance Method computer models, with the HELP model

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being considered by many the best of the available computer models. This is evident in the fact that the HELP model has become compulsory for Superfund Site evaluation (Farquhar 1989).

The HELP model developed by Schroeder *et al.* (1983a), is a quasi-two-dimensional deterministic model which computes the long term leachate flow in a quasi-steady-state flow condition. The HELP model is a tabulation of a moisture balance and was initially developed to perform evaluations on hazardous waste disposal landfills, however, its use has been extended to solid waste landfills. The hydrologic processes modeled include: precipitation, surface storage, runoff, infiltration, percolation, evapo-transpiration, soil moisture storage, and lateral drainage. The lateral drainage process is the only aspect which uses a quasi-two dimensional technique. The model requires climatologic, soil, and landfill design inputs that include: combinations of vegetation, cover soils, waste cells, lateral drainage layers, relatively impermeable barrier soils layers, and synthetic membrane covers and liners.

There have been numerous methods developed to describe the flow of water through unsaturated and saturated porous material among which are those reported by Hanks and Bowers (1962), Whisler and Watson (1968, 1969), Hanks *et al.* (1969), Freeze (1969), Smith and Woolhiser (1971), Giesel *et al.* (1973), and Demetracopoulos *et al.* (1986). These methods use variations of the Richards Equation (Richards 1931) and propose that the flow (and the corresponding moisture content) is considered to be a continuous function of time and space. The refuse material is treated in landfill modeling as homogeneous and the non-

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linear parameters, moisture content, permeability, and heads, are assumed homogeneous in each node. The determination of the non-linear terms at different nodes can be done using mathematical models reported by Russo (1992), Abriola and Pinder (1985a, 1985b), and Demetracopoulos *et al.* (1986).

Ahmed *et al.* (1992) developed a numerical model to compute the time variation of leachate flow in landfills using a two-dimensional moisture transport equation. Unsteady boundary conditions were developed for one-dimensional runoff, evapotranspiration, and infiltration. The model developed a leachate mound at the landfill bottom and allowed for lateral flow in the saturated zones and vertical flow through the landfill liner using Darcy's law. Since landfill surfaces usually behave in a two-dimensional nature, consideration of another runoff model to predict runoff is necessary. Two-dimensional kinematic flow models could be used to determine runoff on top of the landfill. Models have been reported by Constantinides and Stephenson (1981), Stephenson and Meadows (1986), Hromadka and Durbin (1986), and Guymon and Hromadka (1986) describing two-dimensional overland flow.

Farquhar (1989) pulled technical literature together to summarize trends and data for typical leachate composition as a function of age. Using a leachate prediction model (such as the HELP model), site geometry, and contaminant leaching curves, he presented a model to characterize leachate composition (quality). He also examined the impact of microbial processes on the leachate composition. Fungaroli and Steiner (1979a, 1979b), Ham (1980), Wigh

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and Brunner (1981), and McGinley and Kmet (1984) have experimentally investigated the factors which impact leachate quality. The factors include compacted density, waste composition, moisture addition, depth, and refuse age. McGinley and Kmet (1984) and Fungaroli and Steiner (1979a, 1979b) combined data from these investigations and produced leachate contamination curves for various constituents.

Farquhar (1989) used one of these curves in the discussion of his method to calculate leachate contaminant concentrations in the field. The transport of the contaminants through the landfill and soil is important. Source terms for the contaminants need to be determined for the transport models. Contaminant transport has been modeled by Burnett and Frind (1987a, 1987b), Mahmood and Sims (1986), Nair *et al.* (1990), and Cederberg *et al.* (1985). Cederberg *et al.* (1985) provided a model for groundwater mass transport and chemistry equilibrium known as TRANQL. The model uses multicomponents to solve the mass transport equations and the chemical equations for various species. Demetracopoulos *et al.* (1986) and Russo (1991) developed transport equations which account for diffusion and advection of the contaminants with the sources and losses. Bresler (1973) developed expressions for a diffusion coefficient for the transport equations which combined diffusion with dispersion.

Farquhar (1989) also has developed tables to estimate contaminant concentration ranges as a function of age for many of the different components. Currently leachate composition estimates for assessing the impact of leachate on surrounding soil, groundwater, and wastewater treatment facilities are made from this list of

concentration ranges. Integrating a water flow model and a contaminant transport model using the the contaminant production curves as a source term will provide more accurate predictions of landfill leachate generation.

Along with leachate production is gas production, specifically carbon dioxide and methane. DeWalle *et al.* (1978) conducted experimental gas generation studies using steel containers filled with solid wastes while maintaining them under different environmental conditions. He was able to show the effects of temperature, moisture content, waste size, dry density, and ideal pH conditions, on gas production rates. Many others (Merz 1964; Merz and Stone 1970; Rovers and Farquhar 1973; Ramaswamy 1970; Pfeffer 1973; Cooney and Wise 1975) conducted experimental tests for gas production with results which were similar to DeWalle's reports. Hartz (1980) and Hartz *et al.* (1982) studied and quantified the impact of different temperatures on landfill gas production rates for methane gas.

Various schemes to predict the methane production are presented in EMCON (1980) and Tchobanoglous *et al.* (1993) using triangular distributions and an estimated gas production per mass of solid waste (measured as dry or wet depending on the model). The schemes assume that gas production increases linearly until half of the potential gas is produced, at which point the rate slowly falls off linearly. The total life-time of the gas production and the total gas production is estimated. Depending on the model, the peak production rate will occur after a certain percentage of the total estimated life-time has expired (one-half, or one-third, for example.). A

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linear line is drawn up to the peak and then back down to the end of gas production point. Gas production is then estimated to occur along the two curves. Hartz (1980) studied data from real landfills which were producing the percentage of methane which is expected from landfills. He analyzed numerous types of mathematical models to describe the gas production behavior. He and Hartz *et al.* (1982) employed the effects of temperature on the various gas production rates.

Modeling of gas flow in a landfill was presented by Findikakis and Leckie (1979). They considered one-dimensional flow in the vertical direction and used a substrate limiting production model to determine the amount of methane, carbon dioxide, and nitrogen produced. Then they solved diffusion equations to determine the flows through the landfill. Peer *et al.* (1992) have developed an empirical model of methane emissions from landfills. They presented an empirical relationship linking methane production to potential capacity, time, and a constant which is a function of moisture content, nutrient availability, pH, and temperature.

Models to evaluate the design and effectiveness of landfill liners and collection systems under various conditions are reported by Wong (1977), Demetracopoulos *et al.* (1984), Peyton and Schroeder (1988), Korfiatis and Demetracopoulos (1986), Lentz (1981), McEnroe and Schroeder (1988), and McEnroe (1989a, 1989b). Using one-dimensional models, different liner and collection system factors were modeled. These factors included liner slopes, length between drainage pipes, saturated hydraulic conductivity of the lateral-

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drainage layer, saturated conductivity of the soil liner, fraction of the area under a synthetic liner which allows leakage, thickness, inflow volumes, and saturation depths above the liner.

Jayawickrama *et al.* (1988) reported on an experimental and theoretical evaluation of liquid leak rates through flaws in synthetic liners into a compacted soil base. Flaws include imperfect seaming, rips or punctures, or shear failure of the supporting base. They examined the following parameters and their effects on the leak flow rate: head of the liquid above the synthetic liner, hydraulic conductivity of the sub-base material, size and shape of the flaw, and the type and thickness of the synthetic membrane.

The determination of all these landfill factors (waste composition, rainfall, evapotranspiration, runoff, moisture content, liner design, gas generation, and gas transfer) has been separately done through the models presented in this section. Each factor has effects on the other factors and together they are extremely complicated with numerous simplifying assumptions and estimations. This all needs to be taken into account to develop a total model of a landfill in order to simulate the correlated effects.

1.2 OBJECTIVES OF THE STUDY

None of the models reported in the Literature Survey simulate all of the major processes taking place in a landfill and hence, they do not account for the interaction between each process. A large

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research effort studying the factors affecting the processes and the interactions between the processes is ongoing. The desire to develop a comprehensive numerical model incorporating the various processes in order to aid studies of the environment is the goal of this research. The current use of wide leachate contaminant concentration ranges to estimate leachate effects on the soil, groundwater, or treatment plants, reveals the need for more accurate predictions of leachate quantity and quality. Many of these procedures can be combined with contaminant production curves to predict the quality of the landfill leachate. There are numerous unknowns and assumptions which must be made to model these processes. However, as research better defines the processes, unknowns, and empirical relationships, they can be used to modify the current model. The incorporation of better estimations will improve the model predictions. The ability to make predictions of landfill behavior and use of this knowledge will improve designs, maintenance, and thus improve environmental quality.

This study develops a landfill computer model using the available relationships and mathematical models to describe the processes occurring in the landfill. The major processes modeled and interconnected are: precipitation, evapotranspiration, surface runoff, infiltration, leachate generation, leachate transport, leachate collection, landfill liner performance, leachate recirculation, contaminant production, contaminant transport, and gas generation. The model determines the moisture content spatially and temporally in the landfill. The water entering and passing through

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the landfill is modeled to determine the quantity of leachate collected in the landfill collection system. The moisture content is also used with leachate contaminant production curves and gas generation empirical relations to determine leachate contaminant concentrations and methane gas production for each element as a function of time. Transport of the leachate contaminants is modeled to provide an estimate of leachate quality collected at the landfill base. The mass of contaminants which leach out and the mass lost due to gas generation can be used to make predictions about the landfill support structure.

2. THEORETICAL ASPECTS

2.1 LANDFILL DESIGN OVERVIEW

A sanitary landfill is a complex engineering project requiring detailed planning, specifications, careful construction, and efficient operation. The landfill can be conceptualized as a biochemical reactor, with solid waste and water as the major inputs, and with gas and leachate as the principal outputs (Tchobanoglous *et al.* 1993). Traditionally, waste materials have been deposited in voids or on land with little or no agricultural or commercial value. Lack of financing and expertise has led to considerable environmental problems including water pollution, air pollution, and vermin (Crawford and Smith 1985).

Conceptually, solid wastes are disposed of in landfills by spreading them out in thin layers (approximately two feet), compacting the waste, and placing approximately six inches of cover material over the waste (Tchobanoglous *et al.* 1993). This waste pocket is called a load or a lift. Lifts are built on top of each other to fill up the landfill. When the landfill reaches the design height, a cover or cap is placed over the top of the landfill. This cap can be constructed of impervious and/or synthetic material. It is desired to have a 5% grade on the cap to promote runoff from precipitation

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and reduce infiltration into the landfill. The cover is not usually put in place until the landfill is filled (or a cell is completed in large landfills). Filling a cell takes time (order of years) and hence, the cell may not receive a final cover for many years. Also, various cell filling schemes are followed to ensure an even build up of the landfill. Thus, the landfill experiences changing conditions in the amount of water (uncapped cells receive much more water than a capped cell) it receives and hence, leachate amounts percolating through the landfill (Farquhar 1989).

Water enters a landfill from precipitation, recirculation, and as a component of the waste. When the water content exceeds the local capacity of water that the material (soil and waste) can hold, it percolates down through the waste. The percolation will pick up contaminants (dissolved or particulate) as it passes through the waste. This percolation is known as leachate, and it will collect on the bottom of the landfill. The percolation can also collect locally (ponds) in parts of the landfill. It is important that the daily cover material be a permeable material in order to prevent local pond formation (Crawford and Smith 1985). Leachate passing out the bottom of the landfill will permeate down into the local water table. At that point it will contaminate groundwater supplies and will flow with the groundwater. Eventually it will reach wells and become a health hazard, in which the severity depends on the particular contaminants involved. It is important to reduce the amount of leachate and collect it as it percolates down through the waste.

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VEGETATION

SLOPE = 3 TO 5%

TOPSOIL

SAND

LOW PERMEABILITY SOIL

SLOPE = 3 TO 5%

WASTE

SAND

DRAIN

SLOPE = 3 TO 5%

LOW PERMEABILITY SOIL
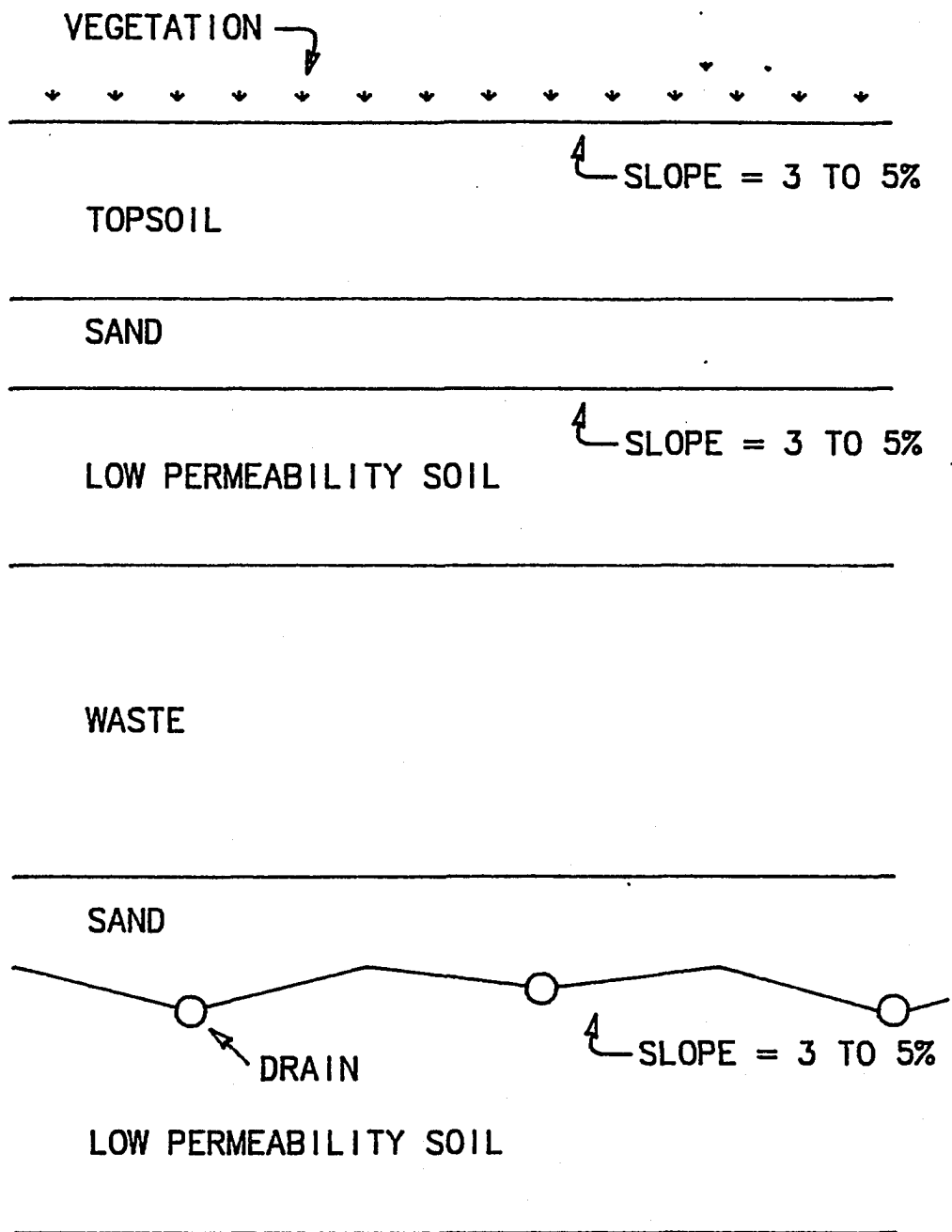
Figure 1. Typical landfill profile (not to scale)
(After Schroeder *et al.* 1983a)

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Sanitary landfills are initially constructed with a liner and leachate collection system on the bottom. Many landfills prior to 1991 regulations do not have a leachate collection system and many old landfills failed to use a liner. The liner is made of compacted low permeability material, such as clay, or a synthetic impermeable material, or both. A layer of very permeable material is laid on top of the liner to collect the leachate which has percolated down through the waste. The surface of the liner is sloped to force leachate to flow laterally toward the collection system drains (Figure 1).

The drains (usually PVC pipe), located in the low points of this very permeable layer, collect, and take the leachate to a central collection point. The leachate collection system removes the leachate to reduce the build-up of saturated leachate above the liner. Leachate build-up over the liner would tend to force its way vertically through the liner into the soil (water table) under the landfill (McEnroe and Schroeder 1988). Leachate in the collection system can be gravity drained or pumped to storage tanks depending on the particular design. There are older landfills which are not designed with a leachate collection system. Below their liner is a fail-safe leachate system, which collects leachate as it passes through the liner. Some of the landfills with leachate collection systems have a second collection system under the liner to collect leachate and give indications of leaky liners. The collection systems come in various designs.

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Solid wastes will decompose in the landfill giving off gases. The major constituents of the gases are carbon dioxide (CO_2) and methane (CH_4), which is explosive. Many landfills are designed with a gas collection system to remove these gases for energy or disposal (flaring). The gases tend to migrate vertically upward, however, when leachate comes in contact with carbon dioxide, the gas can dissolve into the liquid and be carried downward. If this carbon dioxide reaches the water table under the landfill it will cause the groundwater pH to decrease. Hence, environmental monitoring is required at sanitary landfills to ensure contaminants are not released to the surrounding environment (EMCON 1980). There are three categories of monitoring: (1) vadose zone monitoring for gases and liquids, (2) groundwater monitoring, and (3) air quality monitoring. An example of typical instrumentation for environmental monitoring is shown in Figure 2 (Tchobanoglous *et al.* 1993).

There are various types of landfill designs such as the excavated cell/trench method, the area method, and the canyon/depression method. The selection of the method will depend on the existing conditions such as surface water hydrology, topography, climatologic conditions, ultimate use of the completed landfill, available land area, and site access (Tchobanoglous *et al.* 1993). Regardless of the design, there are many common processes occurring in the design of a landfill and these processes lend themselves to being modeled.

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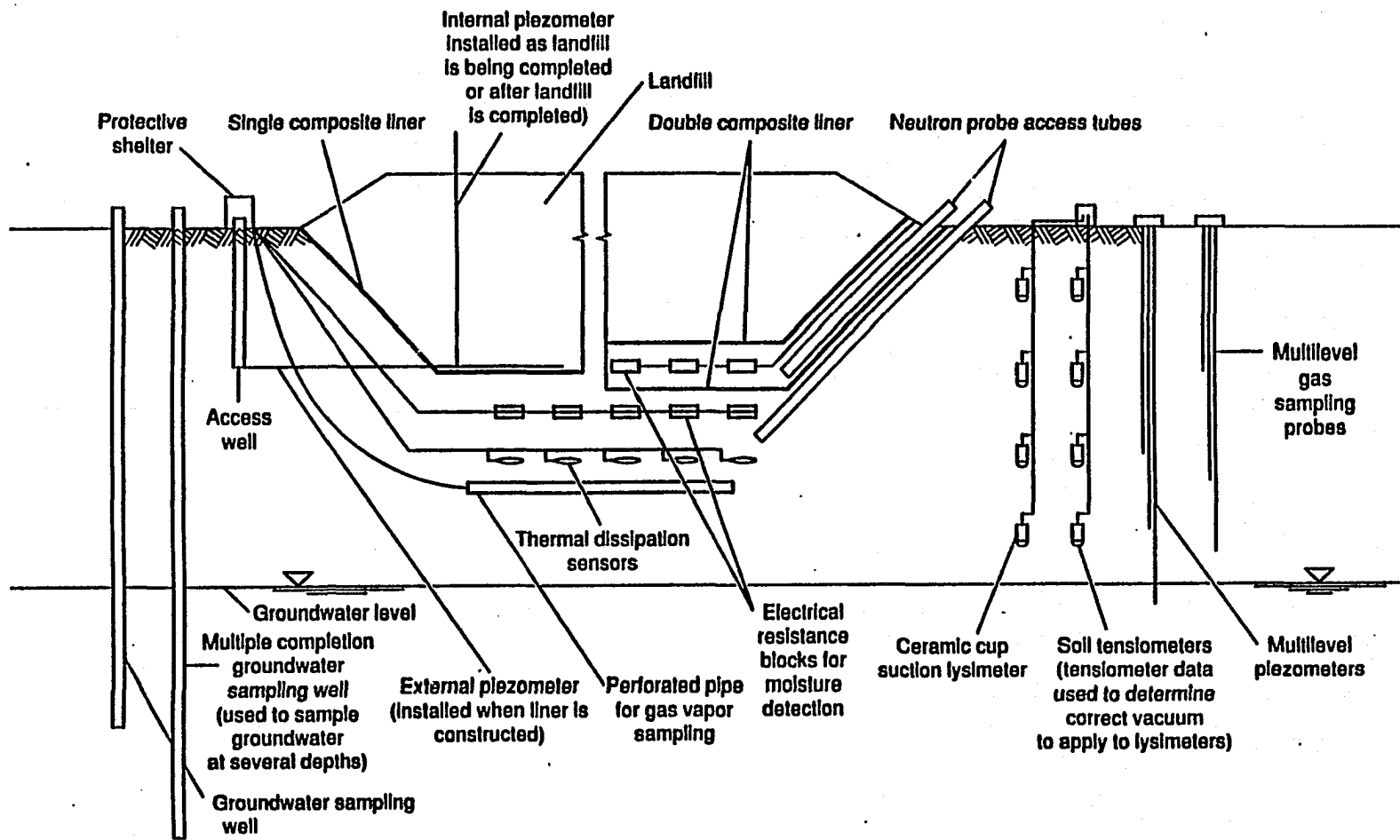
Figure 2.   Instrumentation of a landfill for the collection of environmental monitoring data (After Tchobanoglous *et al.* 1993).

2.2 LEACHATE

Landfill leachate is the liquid that has percolated through solid waste extracting dissolved or suspended materials from the waste. The dissolution and suspension of contaminants, which were stationary in the refuse, are mobilized producing contaminated leachate. Leachate normally is made up of the liquid which enters the landfill from external sources and the liquid produced from the decomposition of the wastes. These external sources may be composed of surface drainage, rainfall, groundwater, underground springs, and recirculated leachate previously removed from the landfill. As the liquid percolates through the solid wastes that are undergoing decomposition, both biological materials and chemical constituents are leached into the solution (Farquhar 1989; Tchobanoglous *et al.* 1993).

Typical data on the composition of leachate from new and mature landfills can be found in numerous references (Crawford and Smith 1985; EMCON 1980; Farquhar 1989; Owens and Khera 1990; Tchobanoglous *et al.* 1977; Tchobanoglous *et al.* 1993) and an example is given in Table 1. The chemical composition (or quality) will vary greatly depending on a number of factors including the quantity produced, the original nature of the waste, the various chemical and biochemical reactions which may be occurring, the age of the landfill, and the events going on inside the landfill (Schroeder *et al.* 1983a; Tchobanoglous *et al.* 1993).

The waste is broken down through anaerobic decomposition which begins after the oxygen is used up. Anaerobic decomposition is

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Table 1.

Typical Data on the Composition of Leachate from New and Mature Landfills

| Constituent | New landfill (less than 2 years) | | Mature landfills (10 years) |
	Range	Typical	Range
BOD_5	200-30000	10000	100-200
TOC	1500-20000	6000	80-160
COD	3000-60000	18000	100-500
Total suspended solids	200-2000	500	100-400
Organic nitrogen	10-800	200	80-120
Ammonia nitrogen	10-800	200	20-40
Nitrate	5-40	25	5-10
Total phosphorus	5-100	30	5-10
Ortho phosphorus	4-80	20	4-8
Alkalinity as $CaCO_3$	1000-10000	3000	200-1000
pH	4.5-7.5	6	6.6-7.5
Total hardness as $CaCO_3$	300-10000	3500	200-500
Calcium	200-3000	1000	100-400
Magnesium	50-1500	250	50-200
Potassium	200-1000	300	50-400
Sodium	200-2500	500	100-200
Chloride	200-3000	500	100-400
Sulfate	50-1000	300	20-50
Total iron	50-1200	60	20-200

Value, mg/l

(After Tchobanoglous *et al.* 1993)

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considered to occur in two phases. In the first phase, facultative bacteria, known as acid formers, alter the complex organics (cellulose, fats, proteins, and carbohydrates) to simple organic materials through hydrolysis, fermentation, or biological conversion. No methane is produced in this stage and the end products are usually organic fatty acids. The small amounts of energy released in this phase are used by the bacteria for growth. The second stage is done by methane forming bacteria. They take the products of the first stage and convert them into methane and carbon dioxide (EMCON 1980; McGinley and Kmet 1984). Archer and Kirsop (1990) break the stages down even more. They consider four metabolic stages involved in the decomposition of waste which produces methane. They are hydrolysis, acidogenesis, acetogenesis, and methanogenesis. The hydrolysis stage bacteria break complex organic carbon into monomeric compounds (simple chemical compounds which can be polymerized). Acidogenesis breaks them into organic acids and hydrogen (H_2). The acetogenesis bacteria breaks them into acetic acid or H_2 and CO_2. At that point, the methanogenesis breaks the acetic acid into methane CH_4 and CO_2 (Archer and Kirsop 1990).

2.2.1 Leachate Generation

Although the quantity of leachate produced is affected to some extent by decomposition reactions, it is largely governed by the amount of external water entering the landfill and the initial moisture content of the waste when loaded. Various techniques have

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been developed to estimate the free water or leachate production using a water balance (or water budget) on the landfill. These methods consider a mass balance among precipitation, surface runoff, evapotranspiration, and waste moisture storage (Crawford and Smith 1985). This balance is accomplished by summing all the water inputs to the landfill and subtracting all the water lost as water vapor or consumed in chemical reactions. The resultant water is held in the landfill material. If this water exceeds the field capacity of the material, leachate is generated. A water definition sketch of a landfill cell is given in Figures 3a and 3b. It breaks the flows into several water inputs (from above, from waste moisture, from cover material, and from sludge) and water losses (formation of gas, water vapor out, and water out below) (Tchobanoglous *et al.* 1993). The following discussions will focus on the various inputs and outputs of the leachate in the landfill.

2.2.2 Evapotranspiration

The major landfill water input is from precipitation. Most models (HSSWDS and HELP for example) separate the precipitation into evapotranspiration, runoff, and subsurface drainage (or infiltration). During a given rainfall, water is continually being intercepted by trees, plants, root surfaces, etc. Evaporation and transpiration are ongoing and continue after the rainfall has ended (Perrier and Gibson 1982). Evaporation is the changing of water from a liquid to a gas using energy from the sun for this change of state.

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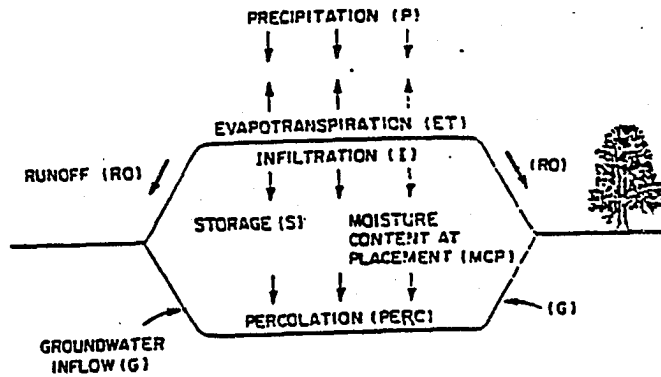
Figure 3a. Illustration of water movement in a landfill (After EMCON 1980).
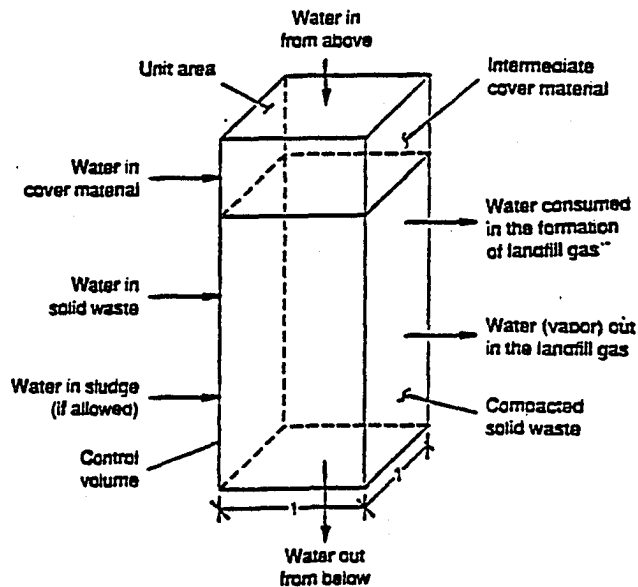
Figure 3b. Sketch of a landfill cell (After Tchobanoglous *et al.* 1993).

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It affects runoff, infiltration, percolation and water table movements. The rate of evaporation is affected by the reflective ability of the surface on which the water lies (known as its albedo), the color of the surface, the air movement over the surface, and the degree of solar radiation falling on the surface. Evaporation occurs almost continuously during daytime and to a lesser extent during the night. The process is most efficient with direct solar radiation and is diminished if this intensity is reduced due to clouds for example. The wind is necessary to remove the receiving air away from the surface. If there is no wind, the air directly above the surface becomes saturated and this stops the evaporation process. Relative humidity and temperature of the receiving air determines the amount of water that can be absorbed and also are important factors (Crawford and Smith 1985).

The next accountable loss of rainfall water is through transpiration. Transpiration is the passage of water from the ground through the roots of plants into the leaves and to the atmosphere. Although a small quantity of this water may be retained in the plant for growth, the bulk of the water taken up in the roots passes through the plant and is returned to the atmosphere. It is difficult to separate evaporation and transpiration when the ground is cultivated. Thus, it is normal practice to combine these two water losses and call it evapotranspiration. The amount of moisture returned to the atmosphere through evapotranspiration varies depending on the precipitation, temperature, humidity, and type of plants. Some plants can have deeper roots and draw water from deeper

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depths. There have been various formulae developed to estimate evaporation (Crawford and Smith 1985).

The water which enters the soil from precipitation is known as infiltration water. The amount of infiltration depends on the runoff characteristics, the gradient of the slope, and the permeability of the surface (Perrier and Gibson 1982). The modified Penman method developed by Ritchie (1972), and used by the HELP model (Schroeder *et al.* 1983b), has been used to compute potential evapotranspiration with the following expression:

$$E_{0i} = \frac{1.28 \ A_i H_i}{25.4 (A_i + G)}$$ (1)

where E_{0i} = potential evapotranspiration on day i (in)

   A_i = slope of saturation vapor pressure on day i

   H_i = net solar radiation on day i (langleys)

   G = psychometric parameter assumed constant at 0.68

This evapotranspiration model is a function of the energy available, the vegetation, the soil water transmissivity, and the soil's moisture content. The slope of saturation vapor pressure, A_i, is computed from the following equation:

$$A_i = \frac{5304}{T_i^2} \exp \left(21.255 - \frac{5304}{T_i} \right)$$ (2)

where T_i = the mean temperature in °K on day i. The net solar radiation, H_i, is computed from the following relation:

$$H_i = \frac{(1-L)R_i}{58.3}$$ (3)

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where L is the albedo for solar radiation (constant at 0.23) and R_i is the solar radiation on day i (langleys) (Schroeder *et al.* 1983b).

The daily mean temperature and solar radiation values are interpolated from monthly mean temperatures and solar radiation values. The interpolation fits a simple harmonic curve using an annual period and a Fourier analysis as follows:

$$V_i = \overline{V} + A \cos \left(\frac{2\pi(i-0.5)}{365} \right) + B \sin \left(\frac{2\pi(i-0.5)}{365} \right) \qquad (4)$$

where V_i = interpolated value of day i

$\overline{V}$ = average annual value

A = coefficient = $\frac{2}{12} \sum\limits_{h=1}^{12} V_h \cos \left(\frac{2\pi(h-0.5)}{12} \right)$

B = coefficient = $\frac{2}{12} \sum\limits_{h=1}^{12} V_h \sin \left(\frac{2\pi(h-0.5)}{12} \right)$

V_h = mean monthly value for month h

The water balance method used by Schroeder *et al.* (1983a) uses the above equations to calculate the daily potential evapotranspiration demand.

The evapotranspiration demand is first exerted on the water available on the surface (snow or precipitation). If there is inadequate water on the surface to satisfy this demand, water is taken from the soil column when the temperature is above freezing. Factors such as winter cover, grass stand, wilting points, leaf area indices, and weighting factors, are used in determining the total evapotranspiration. The total evapotranspiration determined is distributed throughout the evaporative zone of the soil cover using a

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weighting factor. Discussions pertaining to methods used for the actual calculation of losses due to evaporation and transpiration are found in Schroeder *et al.* (1983b).

2.2.3 Surface Runoff

After precipitation gains and evapotranspiration losses are found, the next task is to determine infiltration. Rain falling on the surface will infiltrate or run off the soil (due to gravity). Infiltration depends on the porosity and permeability of the soil layer. Soils which contain sand and gravel can absorb substantial quantities of water during heavy rainfall producing little surface runoff, while clays produce runoff from the start of short periods of light rainfall. Infiltration rates increase if the soil is dry. However, as soil becomes saturated, the infiltration rate decreases, particularly if the soil consists of very fine particles or colloids, which swell slightly when coming into contact with moisture. Dense vegetation can sometimes increase infiltration as the roots prevent compaction of the soil and also dries the subsoil through the transpiration of the water into the atmosphere (Crawford and Smith 1985).

Surface ponding and runoff occurs when rainfall is in excess of evapotranspiration and infiltration. When the rain begins to fall, it fulfills the initial requirements of saturating the top layer. Natural depressions then begin to collect the water in small puddles. In addition, minute depths of water begin to build up on permeable

and impermeable surfaces within the waste disposal site. This stored water collects in small rivulets, conveying the water into small channels in surface runoff. The rate of the runoff is affected by the characteristics of the catchment area. The area, orientation, slope, shapes, vegetation, topography, and altitude of the catchment affect the runoff rates. The installation of artificial drainage channels convey water off the landfill to enhance runoff (Crawford and Smith 1985).

Many models have been proposed for surface runoff. The landfill surfaces or covers are usually two-dimensional and curvature in two dimensions should be considered. An overland flow model developed by Constantinides and Stephenson (1981), uses kinematic equations to solve the plane surfaces, cascades and overland flow channel situations. The kinematic equations are as follows:

$$\frac{\partial h}{\partial t} + \frac{\partial q_x}{\partial x} + \frac{\partial q_y}{\partial y} = i_e \qquad \text{(Continuity Equation)} \qquad (5)$$

$$q_x = \frac{1}{q_t} (\alpha_x h^m)^2 \qquad\qquad\qquad (6a)$$

$$q_y = \frac{1}{q_t} (\alpha_y h^m)^2 \qquad \text{(Kinematic Equations)} \qquad (6b)$$

where $q_t = (q_x^2 + q_y^2)^{0.5}$

   t = time

   x, y = space coordinates in the horizontal directions

   h = the vertical coordinate

   i_e = rainfall excess rate

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q_x, q_y = flow rates per unit width in respective directions

α_x, α_y, and m = parameters

The α_x, α_y, and m parameters depend on which flow friction equation is used. Values for the parameters for different equations are listed in Table 2. An explicit finite difference scheme is used to solve the equations which transmits disturbances only in the direction of expected flow. Inputs may be natural or design storms, surface roughness, and canalization factor (to account for canalization or flow concentration in rills, furrows, or streams).

Hromadka and Durbin (1986) developed a model combining the continuity and momentum equations for long waves in shallow water into a diffusion equation with an explicit numerical scheme to predict the two-dimensional overland flow. The continuity equation is:

$$\frac{\partial Q_x}{\partial x} + \frac{\partial Q_y}{\partial y} + \delta \frac{\partial H}{\partial t} = 0 \tag{7}$$

where Q_x, Q_y = discharges per width

δ = unit width in the x and y directions

H = water surface elevation

The two momentum equations in the x and y directions are:

$$\frac{\partial Q_x}{\partial t} + \frac{\partial}{\partial x}\left(\frac{Q_x^2}{A_x} \right) + \frac{\partial}{\partial y}\left(\frac{Q_x Q_y}{A_y} \right) + gA_x\left(S_{fx} + \frac{\partial H}{\partial t} \right) = 0 \tag{8}$$

$$\frac{\partial Q_y}{\partial t} + \frac{\partial}{\partial y}\left(\frac{Q_y^2}{A_y} \right) + \frac{\partial}{\partial x}\left(\frac{Q_y Q_x}{A_x} \right) + gA_y\left(S_{fy} + \frac{\partial H}{\partial t} \right) = 0 \tag{9}$$

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Table 2.

Kinematic Wave Equation Parameters

Adapted from Constantinides and Stephenson (1981)

Parameter	Darcy/Weisbach	Manning-Strickler
α_x	$8 \ (S_{ox}/\lambda)^{.5}$	$7 \ (g \ S_{ox}/\lambda)^{.5} \ /k^{1/6}$
α_z	$8 \ (S_{oz}/\lambda)^{.5}$	$7 \ (g \ S_{oz}/\lambda)^{.5} \ /k^{1/6}$
m	$3/2$	$5/3$

λ is the Darcy-Weisbach friction coefficient, g is the gravitational acceleration, S_{ox} and S_{oz} are the bed slopes in the respective x and z directions, and k is equivalent to the Nikuradse's roughness.

where A_x, A_y = flow areas per width

δ = width

S_{fx}, S_{fy} = friction slopes in the x and y directions

g = acceleration due to gravity

If it is assumed that the friction slope can be approximated by steady flow conditions, Manning's equation is used to estimate Q_x and Q_y as follows:

$$Q_x = \frac{1.486}{n} A_x R_x^{2/3} S_{fx}^{1/2} \qquad (10)$$

$$Q_y = \frac{1.486}{n} A_y R_y^{2/3} S_{fy}^{1/2} \qquad (11)$$

where R_x, R_y = the hydraulic radii in the respective directions

x, y = lateral directions

n = manning roughness factor

A channel flow conduction parameter, K, for each direction is defined as:

$$K = \left(\frac{1.486 \ A \ R^{2/3}}{n \ |S_f|^{1/2}} \right) \qquad (12)$$

Approximating the friction slope by the slope of the water surface, substituting the momentum equations into the continuity equation, and adding in a rainfall source term, i_e, will yield a single partial differential equation as follows:

$$\frac{\partial}{\partial x} \left(K_x \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y} \left(K_y \frac{\partial h}{\partial y} \right) + i_e = B \frac{\partial h}{\partial t} \qquad (13)$$

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where B is the grid width. This equation can be solved using an explicit finite differences scheme (Hromadka *et al.* 1985; Hromadka and Durbin 1986; Guymon and Hromadka 1986).

2.2.4 Modeling Leachate Flow

The water which remains after runoff and evapotranspiration will infiltrate down into the waste. There are various methods to model this flow.

2.2.4.1 Water Balance Methods

Tchobanoglous *et al.* (1993) discuss the modeling of leachate flow. The landfill is divided into columns and each column is divided into cells with mass balances performed on each cell. Water sources entering each cell depends on where the cell is located in the landfill. If the cell is on top, the inflow is that which has percolated though the cover material. If the cell is below the upper layer, water inflow is the water which has percolated through the wastes in the layer above it. One of the most critical aspects of a water balance is to determine the amount of rainfall that actually percolates through the landfill cover layer. The HELP model (Schroeder *et al.* 1983a) uses a water balance and is a good determination of the amount of rainfall which actually percolates through the landfill, provided no geomembrane is used.

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There are many classifications of water in the landfill. Solid waste initially has water in it from its inherent moisture content and from rainfall (when containers are not sealed properly). The moisture content of residual and commercial municipal solid waste is about 20 percent, however, it will vary depending on the variability of the added moisture content during wet and dry seasons. The same is true for water content in the cover material. Any water which enters a landfill, and not consumed or exits as water vapor, will appear as leachate or will be held in the landfill against the pull of gravity. This water stored in the landfill is known as the field capacity, field-moisture capacity, or moisture-holding capacity (Tchobanoglous et al. 1993). Field capacity is the ratio between the weight of the water held by the soil after drainage and the weight of the dry solid. It varies from 5% for sands to about 25% for loams (Crawford and Smith 1985).

Water percolating through the waste and leaving the bottom cells of the landfill is leachate. Collection systems are built on the landfill bottom and sides and are designed to collect this leachate. Leachate percolating through the solid waste is necessary for the anaerobic decomposition of the organic material in solid waste. The determination of water needed (losses) can be derived from formulas discussed in Section 2.3.1. The potential quantity of leachate is the amount of moisture within the landfill in excess of the landfill field capacity. Tchobanoglous et al. (1993) report an estimation of this field capacity using the following:

$$FC = 0.6 - 0.55 \left(\frac{W}{10,000 + W} \right) \tag{14}$$

where FC = the field capacity

W = overburden weight calculated at waste lift's mid-height

The fraction of water in the waste is based on the dry weight of the waste. A general water balance equation for an estimation of leachate produced is:

$$\Delta S_{SW} = W_{sw} + W_{TS} + W_{CM} + W_{A(R)} - W_{LG} - W_{WV} - W_E + W_{B(L)} \tag{15}$$

where ΔS_{SW} = change in the amount of water stored in solid waste

W_{sw} = water (moisture) in incoming solid waste

W_{TS} = water (moisture) in incoming treatment plant sludge

W_{CM} = water (moisture) in cover material

$W_{A(R)}$ = water from above (rainfall or snow)

W_{LG} = water lost in the formation of landfill gas

W_{WV} = water lost as saturated water vapor with landfill gas

W_E = water lost due to surface evaporation

$W_{B(L)}$ = water leaving the bottom

The water balance is accomplished by adding the incoming water to that water already present in the cell at that time step. The total water present is then compared to the field capacity of the landfill. If the water present is greater than the field capacity, leachate will form. The quantity of leachate formed is a direct function of the amount of external water entering the landfill (Tchobanoglous *et al.* 1993).

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2.2.4.2 Richards Equation

A more physically-based method to model leachate flow is to solve the Richards equation (Richards 1931) at each cell or element. Models describing this moisture movement have been reported by Hanks and Bowers (1962), Whisler and Watson (1968, 1969), Hanks *et al.* (1969), Freeze (1969, 1971), Smith and Woolhiser (1971), and Giesel *et al.* (1973). The general three-dimensional equation for the flow through a porous medium is:

$$\frac{\partial}{\partial x}\left(K_s K_r \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_s K_r \frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_s K_r \frac{\partial h}{\partial z}\right) = \phi\, \frac{\partial S_w}{\partial h}\, \frac{\partial h}{\partial t} \tag{16}$$

where z = the vertical distance

x,y = lateral distance

K_s = the saturated permeability of the landfill cell

K_r = the relative permeability of the cell

S_w = the degree of saturation of the cell

ϕ = porosity

t = time

h = head $(z + p/\gamma)$.

The solution of this equation requires linking relationships between h, K_r, and S_w. The head, h, consists of the elevation head, z, and pressure head, p/γ. Defining the pressure head as P, Russo (1988) reported the following empirical relationships:

$$K_r = \exp(-\alpha\, \text{abs}(P)) \tag{17}$$

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$$S_w = (e^{-0.5 \alpha P}(1 + 0.5 \alpha P))^{2/(m+2)} \qquad (18)$$

where α and m are porous medium parameters.

Also appropriate initial and boundary conditions are needed to obtain a unique set of solutions to Equation (16). The initial distribution of the piezometric head within the porous medium constitutes the initial condition. The boundary conditions are usually prescribed in terms of specified heads or fluxes.

2.3 LEACHATE CONTAMINANTS

As the leachate percolates through the refuse, it picks up contaminants through dissolution and suspension of particles in the liquid, thus producing contaminated leachate. An increase in moisture aids microbial activity and thus the decomposition of the refuse. This metabolic decomposition contributes by-products to the leachate such as volatile fatty acids and alcohols, increasing its organic strength (Farquhar 1989). A large number of the organic compounds existing in the refuse and leachate are soluble. The solubility of the contaminants allows their release to percolating liquids. Increased rates of leachate infiltration thus decomposes and removes contaminants more quickly. Some organic compounds enhance the leaching potential and change the acidity of the leachate. Altering the liquid's pH also has effects on the solubility and may increase leaching. Degrees and types of leachate contamination are limited to the refuse material composition. Waste

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composition and quality often varies between communities. The major constituents of municipal solid waste are paper, food, garden, metals, and glass wastes (Farquhar 1989; McGinley and Kmet 1984).

These constituents are major sources of organic compounds which will be leached as large oxygen demands and total organic carbon concentrations. The best indicator of the degree of contamination of a municipal landfill is a measurement of these organic materials (McGinley and Kmet 1984). The biochemical oxygen demand (BOD) is not a specific contaminant, but a measure of oxygen depletion due to bacteria and other microorganisms engaged in breaking down organic matter (Vesilind *et al.* 1994). Total organic carbon (TOC) is a determination of the amount of organic carbon in a sample regardless of the carbon's oxidation state. This is useful since the ultimate oxidation of organic carbon is to CO_2 and thus it is an oxygen demand (McGinley and Kmet 1984; Vesilind *et al.* 1994). Another important sample is the Chemical Oxygen Demand (COD). COD measures the oxygen equivalent of the organic matter content of a sample which is susceptible to oxidation by a strong chemical oxidant. With COD results of a sample, BOD, organic carbon, and organic matter can be empirically determined after a correlation has been established (Greenberg *et al.* 1992).

In addition to indications of degrees of contamination of the leachate, the organic tests are also an economic factor for landfill operation. Costs to treat leachate at wastewater treatment plants are often determined using the BOD of the leachate. A treatment facility's major concerns are the daily variations in leachate

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strength and the total amount of oxygen demanding material generated. A large change in the oxygen demand could lead to instances of shock loading. This is more significant at a small treatment plant. The total organic load is used in designing a treatment plant's size and operating costs. Hence, estimation of treatment facility requirements for a future landfill is important. Periods of greater leachate generation usually result in larger BOD loads which must be treated (McGinley and Kmet 1984).

The metabolic conditions (could be local) in the landfill will depend on which stage of decomposition is occurring as described in Section 2.2. If a leachate sample is obtained during the acid phase of decomposition, the pH value will be low and the concentrations of BOD_5, TOC, COD, nutrients and heavy metals will be high. If the sample is taken during methane fermentation, the pH will be fairly neutral (in the 6.5 to 7.5 range) and the previously mentioned constituents will be significantly lower (EMCON 1980). Landfills which recirculate leachate have different characteristics. They find that the recirculated leachate acts like a buffer and keeps the pH from dropping as much during the acid phase, while promoting decomposition (Tchobanoglous et al. 1993).

As the material in the landfill decomposes to more stable states, the material leached also changes with respect to biodegradability. In a mature landfill for example, the leachate typically contains humic and fulvic acids, which are not readily biodegradable. The BOD_5/COD ratio is an indication of the biodegradability of the leachate. Ratios in the range of 0.4 to 0.6

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indicate that the organic matter in the leachate is readily biodegradable. The ratios in mature landfills are often in the range of 0.05 to 0.2 (Tchobanoglous *et al.* 1993). Chain and DeWalle (1977) note that as the refuse ages, the readily decomposable materials, such as organic acids which show up in BOD, are degraded faster than the more recalcitrant compounds, such as fulvic and humic organics, which show up in COD. Hence, the ratio decreases with age (McGinley and Kmet 1984).

Most contaminants tend to reach their maximum concentrations in the leachate quickly followed by a gradual reduction in the concentration. This is especially true of biodegradable organics. The readily biodegradable food and garden wastes produce high concentrations of organic matter and total Kjeldahl nitrogen in the leachate. Other less biodegradable wastes contribute organics at lower concentrations, but for longer periods of time (Farquhar 1989). Other factors also contribute to concentration levels. An increase in contact time of the leachate will increase the concentration. This occurs due to a low flow rate and usually when the landfill is reaching field capacity. In both instances, high concentrations result, but the overall load is not great since there is not a large volume of flow. Concentrations will also increase as a function of depth due to more exposure to waste by the leachate (Fungaroli and Steiner 1979; McGinley and Kmet 1984). Wigh and Brunner (1981) report that the maximum concentration of a contaminant occurred near the onset of saturation of the waste.

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Each landfill has a finite amount of contaminant mass which can be removed through leaching. This leaching potential is a combination of contaminant concentrations and the volume of leachate generated. As the waste approaches field capacity, the production of leachate increases while decomposition also increases due to available moisture. Large quantities of the contaminant are leached out during this period as seen in Figure 4 (McGinley and Kmet 1984). As contaminants are leached out, the availability of contaminants decreases and the concentrations in the leachate drop. Even though leachate continues to be generated, the contaminant mass leached is reduced. This is seen as a leveling off of the cumulative contaminant mass release curve in Figure 4. It could be assumed that a state is reached where further leaching produces contaminant loads which constitutes a small environmental hazard. It is possible that this may also reflect an equilibrium state in which the addition of contaminants to leachate is being offset by the removal of contaminants as it passes through the waste (Ham 1980). Most experiments show this leachate contaminants depletion state to occur at moisture loadings of 5 l/kg (Farquhar 1989).

Comparisons of different contaminant tests show differences in ultimate loads and contaminant release patterns. Factors responsible for these differences include waste composition, variations in biological activity, moisture addition, and contaminant removal rates (McGinley and Kmet 1984). Also, since the placement of the waste in the landfill is a function of time, the age of the refuse varies spatially in the landfill. The leachate will thus have contaminant

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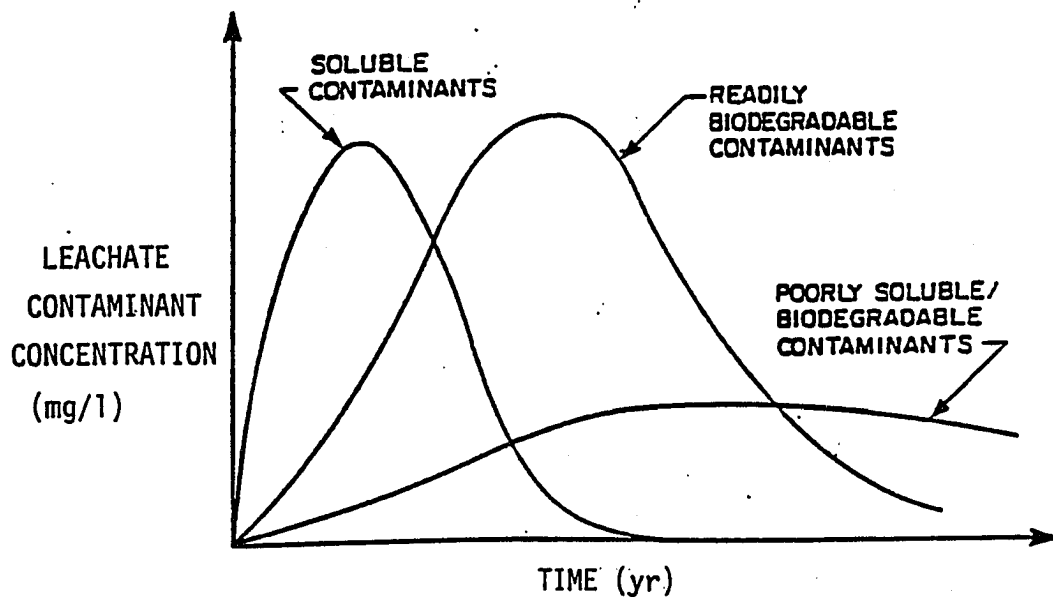
Figure 4. Idealized contaminant production curves
(After Farquhar 1989).

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concentrations and types which are produced from different ages and different sections of the landfill. This reliance on age and location continues after the landfill is closed. Other factors influencing contamination composition are the refuse density, placement sequence, depth, temperature, and time (Farquhar 1989). Waste quality or composition must be used to provide source terms to be combined with leachate generation rates, depths, and densities to construct a contaminant transport model to predict contaminant mass leaching from a landfill.

2.3.1 Modeling Leachate Contaminants

Numerous experiments have been done involving actual landfill sites and lysimeters to simulate landfill conditions. Lu *et al.* (1981) completed an extensive review of investigations reporting leachate production and contaminant concentrations. They created contamination production curves as a function of age. The plots showed a lot of scatter and represent upper limits for leachate contaminant concentrations at field installations. McGinley and Kmet (1984) have combined the data from realistic studies (Fungaroli and Steiner 1979a,b; Ham 1980; Wigh and Brunner 1981; and McGinley and Kmet 1984) which reflect field conditions and produced leachate contaminant curves as a function of moisture loading. Examples of these curves from Wigh and Brunner (1981) are shown in Figures 5 and 6 for the Chloride concentration and Chloride removal respectively.

41
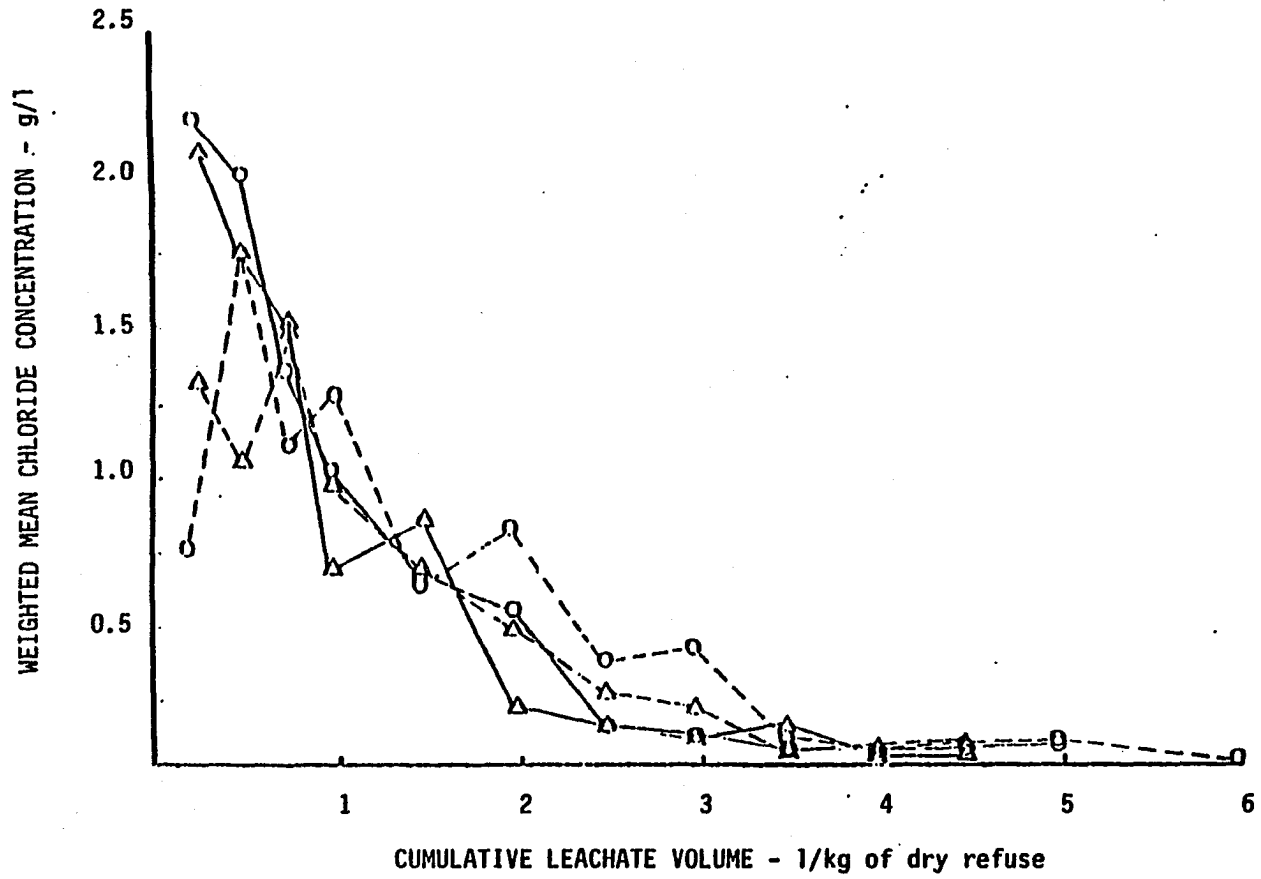
Figure 5. Chloride concentration history (After Wigh and Brunner 1981).
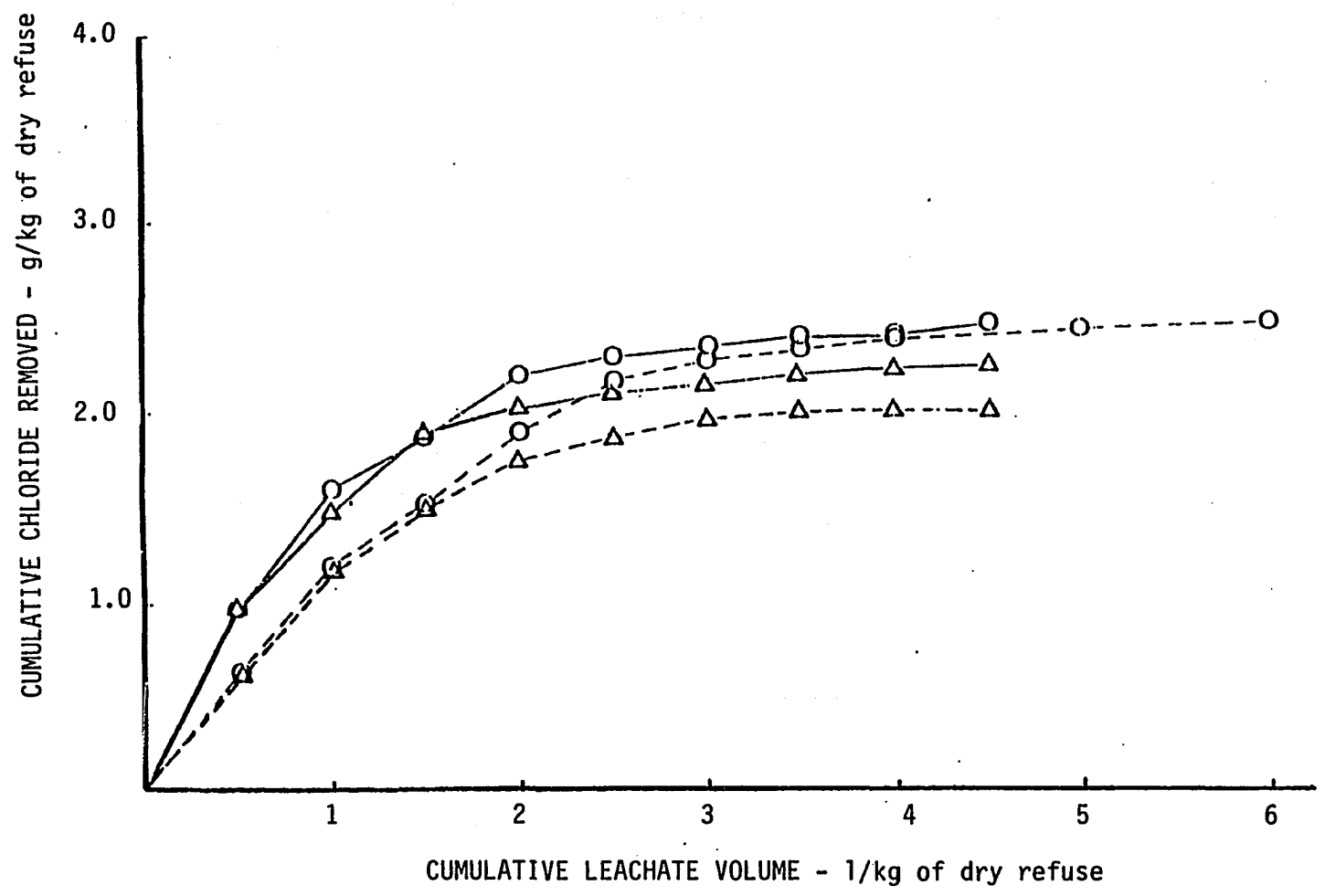
Figure 6. Chloride mass removal (After Wigh and Brunner 1981).

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Farquhar (1989) has developed a method to predict the leachate contamination concentrations. He has combined information on typical municipal solid waste compositions shown in Table 3. The wastes are divided into four different categories. Category A is the readily biodegradable food and garden waste which produce high concentrations of organic matter and total Kjeldahl nitrogen within the first few months of leaching. Less biodegradable organic matter is in category B, hence the concentrations from this source are not as great as those from category A. However, they degrade over a much longer time period (years). The metallic wastes of Category C (iron, aluminum, and zinc) begin leaching after a long period of time and will continue to leach for many years. The readily soluble components of Category D appear in the first few months and promote alkaline earth metals (calcium, magnesium, sodium, and potassium) and common anions (chloride, sulfate, phosphate, and carbonate).

The Farquhar (1989) method to predict leachate concentrations is a lumped model stacking cells on top of each other indicating different placement times (oldest on the bottom to youngest on top). Each cell will thus have a different age and leaching history. A Water Balance Method is used to determine the leachate percolated during a time period. The moisture loading will predict the amount of contaminant leached from the waste and a summation will determine the amount of contaminant in the leachate as it leaves the bottom cell. An example given by Farquhar (1989) shows that if 0.1 meter per month of leachate percolates through a 100 m^2 element loaded with 1.8×10^5 kg of dry waste in six months, the moisture loading will be:

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Table 3.   Typical Municipal Solid Waste Composition

	Component	Composition Range (Wet Weight)
A.	Food	5-20
	Garden Residue	15-20
B.	Paper	40-50
	Plastic	1-5
	Wood	2-5
	Other Organics	2-10
C.	Iron	5-10
	Other Metals	0-1
D.	Glass	5-10
	Other Inorganics	2-5
	Moisture Content	20-30

Data reported by Farquhar (1989); Rovers and Farquhar (1973); EMCON (1980); McGinley and Kmet (1984)

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$$\frac{6 \text{ mos.x } 100 \text{ m}^2 \text{ x } 0.1 \text{ m/mos.}}{1.8 \times 10^5 \text{ kg}} \text{ x } 1000 \frac{L}{m^3} = 0.33 \frac{L}{kg} \tag{19}$$

for the six month period. The chloride production curve shown in Figure 7 is consulted to determine chloride leached for the moisture loading. An example for a four element column is shown in Table 4. The average leachate chloride concentration out the bottom element is calculated to be 4000 mg/L. A weighted average for each chloride concentration from each column is done to predict the leachate concentration exiting a drain line. This scheme is repeated for all the contaminants which are desired and which have production curves available. All the transport processes are meant to be accounted for in the lumped assumption.

It is difficult to predict the results of leaching experiments using a time frame due to all the other variables involved. Most mass release curves have produced a leveling off of contaminant removal based on moisture loading as a master variable (McGinley and Kmet 1984). Combining a water flow model as described in Section 2.2.4.2 with the contaminant production curves will provide contaminant production as a function of the amount of leachate passing through the waste. It thus would present more accurate predictions of landfill contaminant generation and transport. The use of contaminant production curves can be coupled with a transport model instead of the lumped analysis. Geochemical transport models have been developed by Rubin and James (1973), Valocchi et al. (1981), and Charbeneau (1981), however, they were single specific chemical reaction models. Their assumption is that solutes modeled acted

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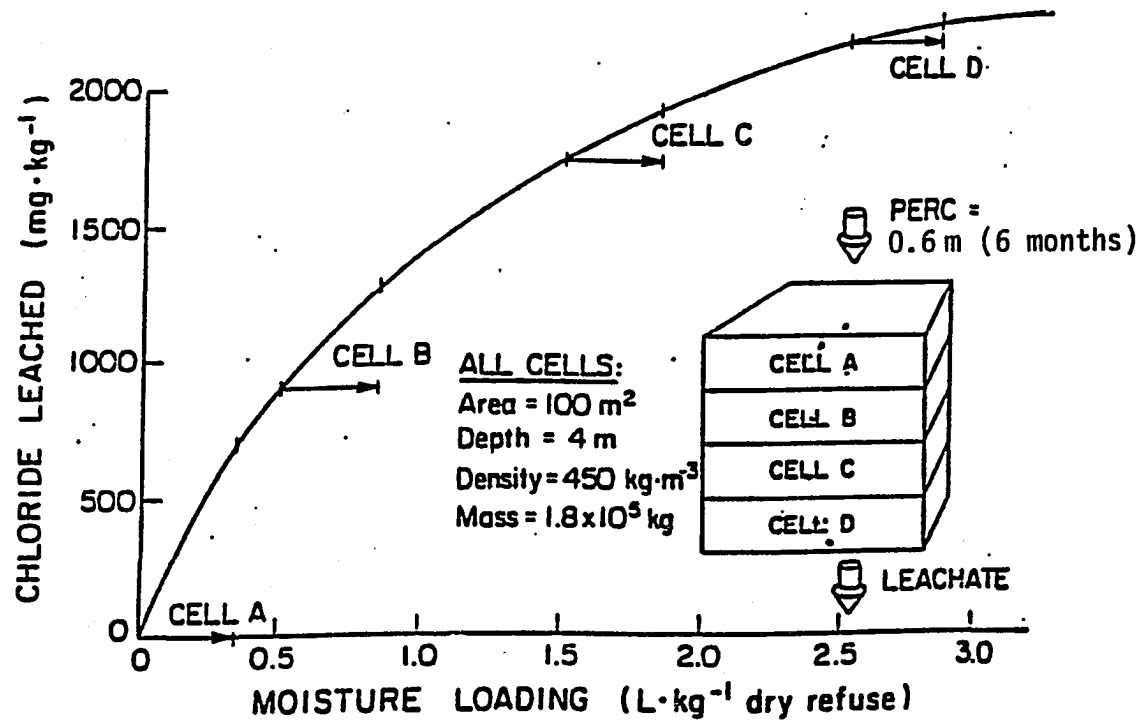
Figure 7. Leachate chloride example description (After Farquhar 1989).

TABLE 4. LEACHATE CHLORIDE CONCENTRATION EXAMPLE

Cell	Moisture Loading (liters/kg)		Amount of Chloride Leached (mg/kg)				Chloride Conc. (mg/L) leached in .33 L/kg
	Before (1)	After (2)	Before (3)	After (4)	Dif (5)	Sum (6)	Leached (7)
A	0.0	0.33	0.0	700	700	700	2120
B	0.5	0.83	910	1270	360	1060	3210
C	1.5	1.83	1740	1930	190	1250	3790
D	2.5	2.83	2160	70	70	1320	4000

Notes:

Column (3) and Column (4) are read from Figure 7.

Column (5) = the difference in chloride mass leached
(Column (4) - Column (5))

Column (6) = the summation of Column (5)

Column (7) = Column (6)/(0.33 L/kg)

(From Farquhar 1989)

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independently of the bulk solution. Modeling multicomponent solution transport has been investigated by Rubin (1983), Jennings *et al.* (1982), and Cederberg *et al.* (1985).

2.3.2 Contaminant Transport Models

The governing equation for a transport model in the vertical direction used by Demetracopoulos *et al.* (1986), Frind and Hokkanan (1987), and Russo (1991), is written as:

$$\frac{\partial(\phi S_w C)}{\partial t} = \frac{\partial}{\partial z}\left(D_z \frac{\partial C}{\partial z}\right) - \frac{\partial(v_z S_w C)}{\partial z} + Rx \qquad (20)$$

where C = contaminant concentration (mass per unit volume)

z = vertical length

S_w = moisture saturation of the element

D_z = vertical diffusion coefficient

ϕ = the porosity of the waste layers

v_z = vertical velocity (advection) of leachate

t = time

Rx = reaction source terms

The reaction source terms account for contaminants produced in the cell, contaminants entering in rain water, contaminants in recirculated leachate, contaminants in lateral seepage, and contaminant gains and losses through reactions. Each of these processes has an input to the mass balance equation.

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Two sets of boundary conditions and one set of initial conditions are required to solve Equation (20). The upper boundary has an influx of contaminants if they are present in the rainfall or if the landfill recirculates the leachate. The lower boundary has advective and diffusive losses of contaminants out the bottom cell. The equation can be discretized for each cell and the Thomas algorithm run with the contaminant concentration, C, being the implicit variable.

The diffusion coefficient must be determined to solve Equation (20) and is described by Bresler (1973). The means of contaminant transport are divided into two parts; molecular diffusion and convective transport. The convection is due to the average velocities, which are accounted for in Equation (20), and through hydraulic dispersion. Evaluations of mass transport have shown that convective dispersion had the greatest effect in carrying leachate contaminants in the direction of leachate infiltration. This was also true when the contaminants reached the ground water. Molecular diffusion due to leachate concentration gradients has a small effect on contaminant transport in the direction of the infiltration, however, it has a noticeable effect on the lateral transport. Adsorption due to chemical reactions acts to retard the migrations of contaminants from the landfill. This effect is most pronounced in media containing active materials such as clay (Fungaroli and Steiner 1979). Bresler (1973) has combined dispersion coefficients with coefficients for diffusion to generate one coefficient, D_z, for Equation (20).

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2.4 LANDFILL LINER AND LEACHATE COLLECTION SYSTEMS

Modern landfills have leachate collection systems on the bottom of the landfill in conjunction with the liner. These systems remove leachate as it percolates down from the waste to the top of the liner to reduce the head build-up and thus reduce the potential for leakage out of the landfill to the water table. Peyton and Schroeder (1990) evaluated various landfill liner designs to check compliance with the minimum technology guidance on liner systems.

Currently there is guidance for double-liner systems. Peyton and Schroeder (1990) describe the intention of the guidance for double-liner systems. On the top, it must have a primary leachate-collection and removal system. This primary system should have at least one foot (30 cm) of chemically resistant drainage layer on a minimum bottom slope greater than 2%, and a saturated hydraulic conductivity of at least $1x10^{-2}$ cm/s. The liner slopes to drain points which collect the leachate and convey it to the collection point. Under the primary system is a synthetic liner on top of a secondary leachate-collection and removal system (which has the same requirements as the primary system). Under it all is composite liner (synthetic liner plus low-permeability soil) or a thick, low permeability soil liner. The hydraulic conductivity of the soil liner should be less than $1x10^{-7}$ cm/s and the liner should be at least three feet thick. The collection system should ensure leachate depth does not reach one foot (Peyton and Schroeder 1990).

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The synthetic liners are essentially impermeable under ideal conditions, however, they can leak due to various reasons. There are six factors affecting the liner/leachate collection system performance; the rate of leachate drainage from the waste; the distance between drains; the saturated hydraulic conductivity of the drain layer; the slope; the thickness of the liner; and the hydraulic conductivity of the liner. The hydraulics of a liner system are controlled, to a great extent, by the hydraulic conductivity of the liner (previously the dominant design parameter), the hydraulic conductivity of the drainage layer, and the leachate input rates (McEnroe and Schroeder 1988).

Peyton and Schroeder (1990) evaluated various other landfill designs assuming a steady state linearized approximation to the Boussinesq free surface drainage equation. The use of sufficiently short time steps allowed the assumption of steady state. McEnroe and Schroeder (1988) presented a model for steady state leachate flow after the landfill is capped. It assumes unsaturated conditions under the liner, neglects spatial variation in the leakage rate, and uses the Dupuit approximation for lateral drainage. The model predicts the average and maximum saturated depth on the liner, the location of the maximum saturated depth, and the leakage rate. McEnroe (1989a) developed an unsteady model for estimating leakage through a compacted clay barrier for a single inflow event. He used the same equations as those used by Korfiatis and Demetracopoulos (1986), however, he questioned their use of the upper boundary conditions. He solved for unsteady flow in the saturated drainage

layer above the low permeability barrier by using a one-dimensional scheme with the flow traveling along the barrier surface. Inflows are the leachate percolating through the waste and losses are the leakage through the barrier.

Use of a transient liquid mass balance equation to simulate leachate seepage and drainage rates, and head build-up on landfill liner collection systems has been developed by Korfiatis and Demetracopoulos (1986). They solved the equations numerically. The following one-dimensional equation for saturated unsteady flow in which the leachate moves in a direction parallel to the sloping liner is given to describe the liquid balance shown in Figure 8:

$$\phi_e \frac{\partial h_c}{\partial t} \; + \; \frac{\partial Q}{\partial x} = R - q_b \tag{21}$$

where ϕ_e = effective porosity of the drainage layer

h_c = leachate depth above the liner

t = time

Q = drainage flow rate along the liner surface per unit width

x = flow direction parallel to the liner surface

R = rate of leachate percolating into the element on the liner

q_b = leakage rate through the barrier

The Dupuit assumption is used to define the discharge in the flow direction by the following:

$$Q = - K_d \; y \; \frac{\partial h_c}{\partial x} \tag{22}$$

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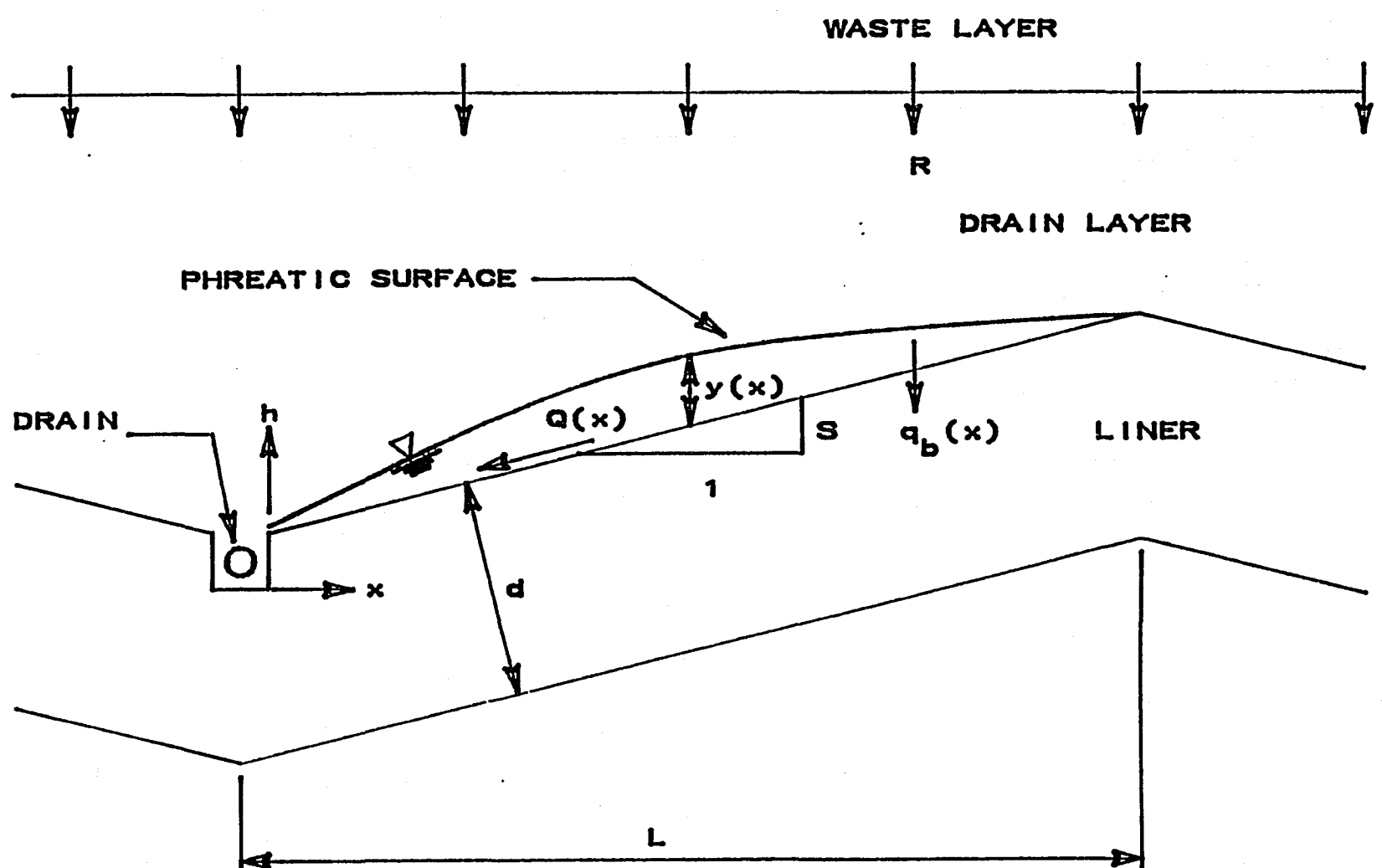
Figure 8. Typical design of a landfill liner and leachate collection system (After McEnroe 1989; Peyton and Schroeder 1990).

where y = the depth of the leachate on the liner

K_d = saturated hydraulic conductivity of the drainage layer

The $\frac{\partial h_c}{\partial x}$ term is the slope of the leachate flow surface (Korfiatis and Demetracopoulos 1986; and McEnroe 1989a).

Leachate can also permeate through the liner in the vertical direction. The liner barrier is assumed to be saturated at all times in order to apply Darcy's law to determine the local leakage rate through the liner, q_b, which is written as:

$$q_b = K_L \left(\frac{y + d}{d} \right) \tag{23}$$

where K_L is the saturated hydraulic conductivity of the liner and d is the thickness of the liner (Korfiatis and Demetracopoulos 1986; and McEnroe 1989a).

The upstream boundary conditions are satisfied when $y\frac{\partial h_c}{\partial x} = 0$ (McEnroe 1989a). The downstream boundary condition postulates a free drainage condition. The gradient at the boundary is approximated by upstream differences between the last and next to last points of the grid (Korfiatis and Demetracopoulos 1986).

Flaws in the liner are present and Jayawickrama *et al.* (1988) studied leakage rates through flaws in the membrane. Peyton and Schroeder (1990) evaluated numerous designs for landfill liners. A leakage fraction was use to determine local leakage rates. It was defined as the fraction of the horizontal area of soil through which percolation is occurring under the leaking synthetic liner.

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2.5 GAS GENERATION.

Gases are produced in landfills through the organic biodegradation of the components of solid waste. A number of principal gases are present in large amounts while a number of trace gases are present in very small amounts. The principal gases are produced from the decomposition of the organic portion of municipal solid waste (Tchobanoglous *et al.* 1993). Initially, the bacterial decomposition is aerobic due to the air trapped in the wastes. The oxygen is quickly used up and the decomposition proceeds under anaerobic conditions.

The principle source of the aerobic and anaerobic organisms responsible for the decomposition is the soil material used as a daily and final cover. The various gases found in landfills include ammonia (NH_3), carbon dioxide (CO_2), carbon monoxide (CO), hydrogen (H_2), hydrogen sulfide (H_2S), methane (CH_4), nitrogen (N_2), and oxygen (O_2) (Tchobanoglous *et al.* 1993). However, carbon dioxide and methane are the principle gases produced from the anaerobic decomposition of the organic wastes, with carbon dioxide initially making up a large percentage of the gas produced as a result of aerobic decomposition (Tchobanoglous *et al.* 1977). When methane is present in the air in concentrations between 5 and 15 percent, it is explosive. Since only limited amounts of oxygen are present in a landfill when methane concentrations reach this level, there is little danger of an explosion. However, if methane gases migrate off-site and mix with air, obtaining this methane explosive mixture is possible (Tchobanoglous *et al.* 1993).

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There are other constituents found in a landfill known as trace gases. They will only briefly be mentioned here. Trace gases include argon, mercaptan sulfur, sulfides, disulfides, propane, isobutane, n-butane, isopentane, n-pentane, hexane, heptane, octane, nonage, toluene, and benzene (DeWalle *et al.* 1978; Schuyler 1973; Bowerman *et al.* 1977; Colona 1976). There has also been extensive landfill gas sampling by the California Integrated Waste Management Board (66 landfills) and in England. A total of 116 organic compounds were found in landfill gases and tables listing the findings can be found in Tchobanoglous *et al.* (1993).

Trace constituents have two basic sources. They are part of the incoming waste placed in the landfill or they are produced (biotic and abiotic reactions) in the landfill. Many trace compounds are in the liquid state and mixed in the waste, but they tend to volatilize and become volatile organic compounds (VOCs). The presence of these trace gases in the leachate will depend on their concentrations in the landfill gas in contact with the leachate. The occurrence of significant concentrations of VOCs in landfill gas is associated with older landfills that accepted industrial and commercial wastes containing VOCs. In the newer landfills which have banned the disposal of hazardous waste, the concentrations of VOCs in landfill gas have been found to be extremely low. Very little can be stated definitely about the rates of biochemical transformation of trace compounds (Tchobanoglous *et al.* 1993).

Carbon dioxide and methane are significant by-products of the decomposition of the wastes disposed in a landfill. There has

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developed a large interest in the generation of these gases, specifically methane. With the development of regional landfills to serve immense metropolitan areas, gas recovery is a significant energy source (in excess of 5 million cubic feet per day of equivalent pipeline standard gas (1000 BTU/scf)) being recovered from the large landfills (EMCON 1980; Archer and Kirsop 1990). Hence, the discussion of gases will be mainly concerned with the production of methane and carbon dioxide.

2.5.1. Theoretical Maximum Yield of Landfill Gases

EMCON (1980) reported on the prediction of the theoretical maximum yield of gases released during the anaerobic decomposition. This yield has been estimated in a number of ways and is governed by an empirical formulation. The generalized reaction to convert waste with an empirical formulation of $C_nH_aO_bN_c$ to methane, carbon dioxide, and bacterial cells ($C_5H_7O_2N$) is as follows:

$$C_nH_aO_bN_c + \left(2n + c - b - \frac{9sd}{20} - \frac{de}{4}\right)H_2O = \left(\frac{de}{8}\right)CH_4 + \left(n - c - \frac{sd}{5} - \frac{de}{8}\right)CO_2$$

$$+ \frac{sd}{20} C_5H_7O_2N + \left(c - \frac{sd}{20}\right)NH_4^+ + \left(c - \frac{sd}{20}\right)HCO_3^+ \qquad (24)$$
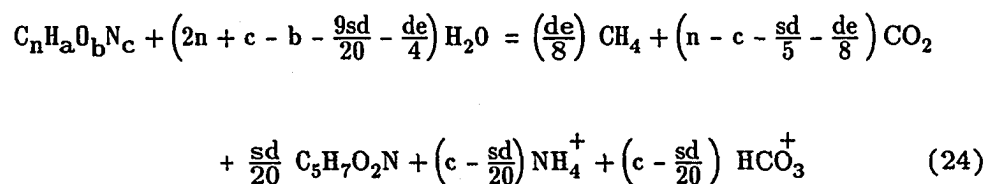
where a,b,c,n = initial stochastic values of the waste

d = 4n + a - 2b - 3c

s = the fraction of waste chemical oxygen demand (COD), synthesized or converted to cells

e = fraction of waste COD converted to methane gas for energy

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and s + e = 1. The value of s varies with the waste composition, the average solids retention time in the system, θ_c, and the cell decay rate, f, as follows:

$$s = a_s \left(\frac{(1 + 0.2 \, f \, \theta_c)}{(1 + f \, \vartheta_c)} \right) \tag{25}$$

where θ_c = solids retention time, days

a_s = maximum value of s, which occurs for θ_c = 0

f = cell decay rate, day^{-1} (per day)

The value 0.2 represents the refractory portion of the bacterial cells formed during cell decay. Values for a_s for various components of wastes; carbohydrates, protein, fatty acids, sludge, ethanol, methanol, and benzoic acid, are listed in EMCON (1980).

If the organic wastes are stabilized completely, Tchobanoglous *et al.* (1993) and Rich (1963) represent the overall conversion with a simpler equation:

$$C_a H_b O_c N_d + \left(\frac{4a - b - 2c + 3d}{4} \right) H_2 0 \rightarrow \left(\frac{4a + b - 2c - 3d}{8} \right) CH_4$$

$$+ \left(\frac{4a - b + 2c + 3d}{8} \right) CO_2 + dNH_3 \tag{26}$$
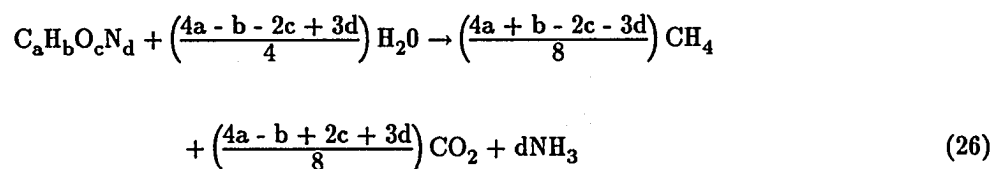
where a,b,c, and d are the stochastic values for the wastes.

EMCON (1980) reports two basic methods used to estimate the conversion of waste to gas; the basic stoichiometry of the conversion of organic matter to methane; and an approach utilizing an assumed conversion efficiency. The stoichiometric procedure assumes an efficiency of stoichiometry for the bioconversion of the organic

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matter to methane. For example, using the stoichiometry method with a composite refuse composition of $C_{99}H_{149}O_{59}N$, the potential ultimate yield of methane gas has been estimated to be 270 L CH_4/kg wet refuse. It is also estimated that the landfill gas would be 54% CH_4 and 46% CO_2. Other stoichiometric approaches include a method based on gross empirical formulas representing two major groups of organic wastes which constitute the major sources of landfill gas (paper and food) (EMCON 1980; Leckie 1974). This yields 230 L CH_4/kg. Another similar approach assumes that the refuse is cellulose on a dry weight basis, and that all of it decomposes to methane by:

$$C_6H_{10}O_5 + H_2O \quad \rightarrow \quad 3CH_4 + 3CO_2 \qquad (27)$$
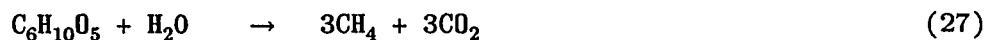
with an estimated yield of 415 L CH_4/kg (EMCON 1980; Wise et al. 1975). EMCON (1980) estimates the average yields for the stoichiometric equation methods to vary between 230-270 L CH_4/kg wet composite.

The second method of estimating the ultimate gas yield uses approximations of the overall biodegradability of typical composite refuse or of individual waste components. EMCON (1980) summarizes reports on numerous different methods used which have different variations on the assumptions of constituents with various gas yields reported by Leckie (1974), Schuyler (1973), Pfeffer (1974), Bowerman et al. (1977), and Blanchet et al. (1977). The biodegradability of materials methods resulted in estimated average yields of 47 to 120 L CH_4/kg wet composite. The total organic content method yields were 190-270 L CH_4/kg wet composite. Realistic theoretical estimates of potential total methane production range from 47 to 270 L CH_4/kg wet

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composite. EMCON (1980) reports that conditions prevailing in most sanitary landfills, will yield an actual total methane production in the range of 31 to 94 L CH_4/kg wet composite, and that this total production may be enhanced by managing factors to favor fermentation such as moisture and pH.

2.5.2 Factors Affecting Landfill Gas Production

Gas Production in a landfill exhibits a time dependency as it undergoes various evolutionary processes. These changes are characterized by five relatively distinct phases (Figure 9). The time associated with each phase of gas production is a function of the specific conditions within a landfill. Phase I (Aerobic or Initial Adjustment Phase), lasting several days to weeks, occurs while oxygen is present during waste placement. The biological decomposition occurs under aerobic conditions with carbon dioxide being the principle gas produced. The principal source of organisms is the soil material used for daily and final cover. Digested wastewater treatment plant sludge (place in some landfills) and recycled leachate are also other sources of organisms (EMCON 1980; Tchobanoglous *et al.* 1993).

When the oxygen is used up, Phase II (Transition Phase) begins with anaerobic conditions producing significant amounts of carbon dioxide and some hydrogen gas (EMCON 1980). As the landfill becomes anaerobic, nitrate and sulfate, which serve as electron acceptors in biological conversion reactions, are often reduced to nitrogen gas and hydrogen sulfide. This onset of anaerobic conditions can be

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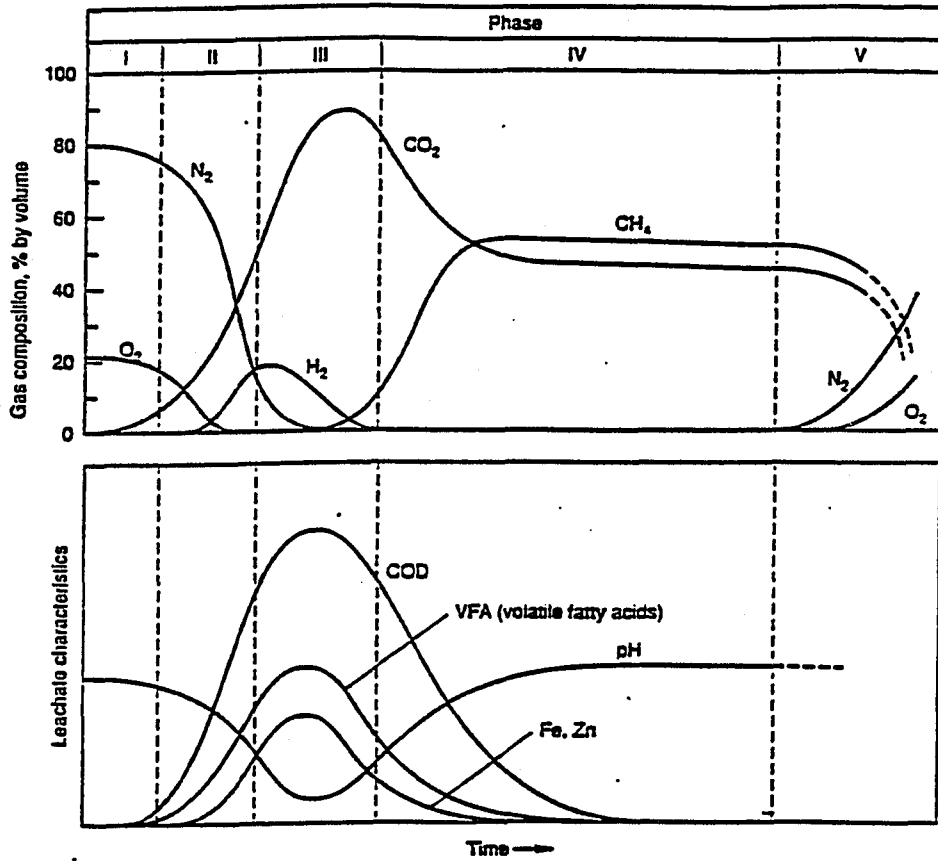
Figure 9. Generalized phases in the generation of landfill gases (After Tchobanoglous et al. 1993).

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monitored by measuring the oxidation/reduction potential of the waste. In Phase II, the pH of the leachate, if formed, starts to drop due to the presence of organic acids and the effect of elevated concentrations of CO_2 within the landfill (Tchobanoglous *et al.* 1993).

As the oxidation/reduction potential decreases, microorganisms begin a three step process to convert organic acids to methane and carbon dioxide in Phase III. The point of methane production is the start of Phase III (Acid Phase) and hydrogen is quickly depleted (EMCON 1980). Tchobanoglous *et al.* (1993) reported that the microbial activity started in Phase II accelerates with the production of significant amounts of organic acids and lesser amounts of hydrogen gas. There are three steps involved. Step one is the enzyme-mediated transformation (hydrolysis) of larger molecular mass compounds (lipids, polysaccharides, proteins, and nucleic acids) into compounds suitable for use by microorganisms as a source of energy and cell carbon. Acidogenesis, the second step, converts the compounds resulting from step one into smaller molecular mass intermediate compounds such as acetic (CH_3COOH), fulvic, and other organic acids. The microorganisms involved in this conversion are described as nonmethanogenic and consist of facultative and obligate anaerobic bacteria (acidogens and acid formers). Carbon dioxide is the principal gas generated during this phase and the pH of the leachate will drop to a value of 5 or lower due to the elevated landfill carbon dioxide concentrations and the presence of organic acids. The biochemical and chemical oxygen demands and the conductivity of the leachate will increase significantly during this

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phase due to the dissolution of the organic acids in the leachate. The low pH will also cause a number of inorganic constituents, principally heavy metals to be solubilized and many essential nutrients to be removed in the leachate. If the leachate is not recycled, the nutrients will be lost from the system. If leachate is not formed, the conversion products produced will remain in the landfill as sorbed constituents and in the water held by the waste.

Phase IV (Methane Fermentation Phase) is also anaerobic and is reached when gas production and composition approach a pseudo-steady-state condition. This gas production phase ranges from a few years to several decades depending on the landfill environment (EMCON 1980). A second group of microorganisms which begin developing toward the end of Phase III become more predominant. These microorganisms convert the acetic acid and hydrogen gas formed by the acid formers to methane and carbon dioxide. These microorganisms are strict anaerobes called methanogenic (methanogens or methane formers) (Tchobanoglous *et al.* 1993). Archer and Kirsop (1990) report small amounts of oxygen will be toxic to them. The methane and reduced acid formation proceed simultaneously. Since the acids and hydrogen gas produced by the acid formers have been converted to methane and carbon dioxide, the pH of the leachate will begin to rise to the range of 6.8 to 8.0. The concentration of BOD_5 and COD and the conductivity value of the leachate will be reduced. The higher pH values will cause fewer inorganic constituents to remain in solution and the concentration of heavy metals in the leachate will be reduced (Tchobanoglous *et al.* 1993).

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Tchobanoglous *et al.* (1993) discuss a Phase V (Maturation Phase). This occurs after the readily available biodegradable organic material has been converted to methane and carbon dioxide. As moisture migrates through the waste, portions of the biodegradable material previously unavailable will be converted. The rate of gas generation decreases significantly due to the removal of most of the nutrients and the slow biodegradability of the substrates remaining. The principal gases are still methane and carbon dioxide, however, small amounts of oxygen and nitrogen may be found depending on landfill closing measures. The leachate often contains humic and fulvic acids which are difficult to process down further biologically.

DeWalle *et al.* (1978), Schuyler (1973), Alpern (1974), California State Water Control Board (1964), and Bureau of Sanitation, Los Angeles (1975) have made predictions of different landfill effective life times (for gas generation) ranging from 10 years to a half-life of 100 years. The duration of the individual phases will vary depending on the distribution of the organic components in the landfill, the availability of nutrients, the moisture content of the waste, moisture routing, and degree of initial compaction (Tchobanoglous *et al.* 1993). The landfill factors affecting the rate of gas production (size, composition, age, moisture content, temperature conditions, and nutrient quality) are potentially manageable, except for temperature. The only factor currently managed is moisture content using leachate recirculation. A high moisture content (60 to 80%) promotes the maximum methane production rate (EMCON 1980).

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It is noted that the reaction to convert waste to gas requires water as seen in Equations (24), (26), and (27). Landfills lacking sufficient moisture content have been found in a mummified condition, with decades-old newsprint still in readable condition. Although the total amount of gas that will be produced from solid waste is derived from the reaction stoichiometry, local hydrologic conditions significantly affect the rate and period of time over which gas production takes place (Tchobanoglous *et al.* 1993).

DeWalle *et al.* (1978) have shown that the rate of gas production is increased by increasing moisture content. There currently is no relationship to determine the gas production rate as a function of moisture content. The rate of gas production is believed to increase linearly with moisture content when above 20%. Gas production drops off sharply when moisture content drops below 20%. This linear relationship has been shown in a few experiments conducted by DeWalle *et al.* (1978), Pfeffer (1973), Merz (1964), Merz and Stone (1970), Ramaswamy (1970), Rovers and Farquhar (1973), and Cooney and Wise (1975). They measured the gas production while increasing moisture content. A graphical illustration of their results is shown in Figure 10. It is noted that other parameters were constant during these tests.

It has been found that an increase in temperature also increases the gas production rate. DeWalle *et al.* (1978) produced Figure 10, which illustrates that the slopes of the various graphs increased with increasing temperatures. Like moisture content, this has been difficult to quantify. Two separate effects of temperature

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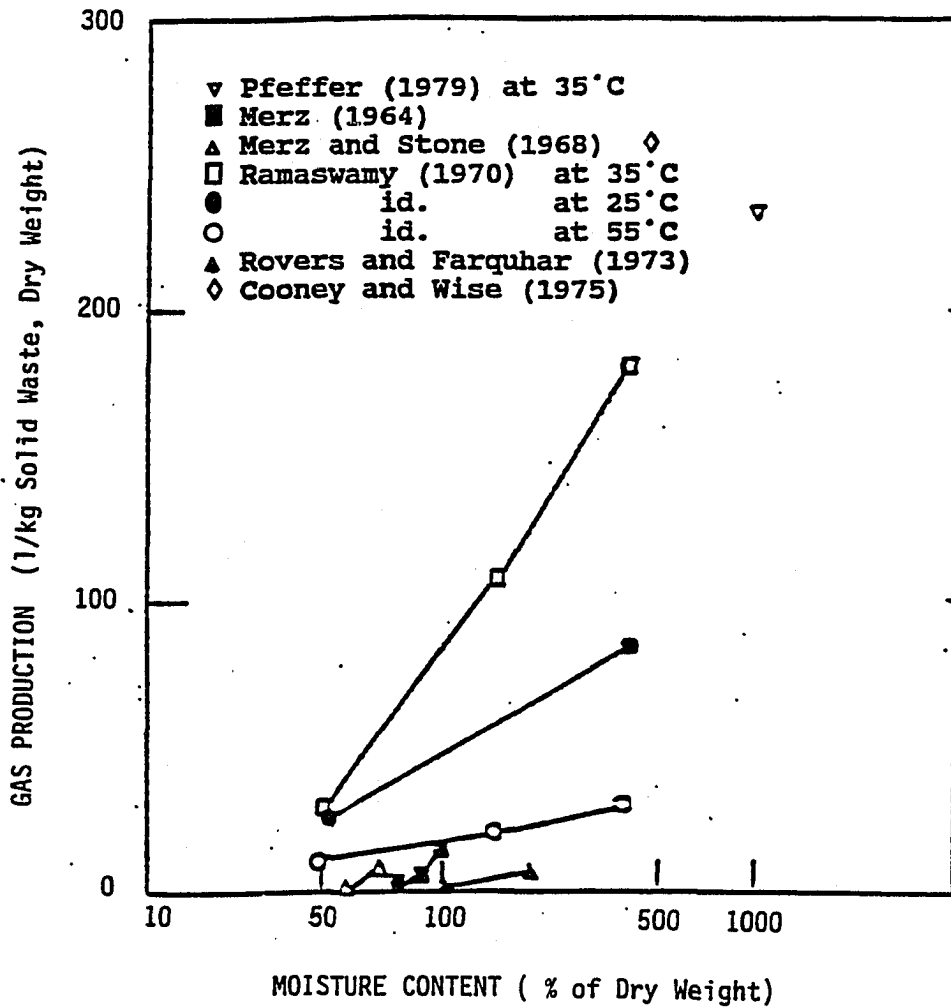
Figure 10. Gas Production as a function of moisture content (After DeWalle *et al.* 1978).

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on methane formation must be considered. The first effect is immediate changes in reaction rates as temperature changes. Long-term effects would consider adaption in the microbial population balance resulting from changes in temperature. The usual method of reporting temperature effects on reactions is the empirical Arrhenius relationship (Hartz *et al.* 1982):

$$K = A \, e^{E_a/RT} \tag{28}$$

which can also be written as:

$$\ln \frac{K_2}{K_1} = \frac{E_a \, (T_2 - T_1)}{R(T_2 \, T_1)} \tag{29}$$

where K_1, K_2 = methane evolution reaction rates

T_2, T_1 = temperatures (°K) for respective reaction rates

E_a = energy of activation (calories/mole)

R = the gas constant (1.987 calories/°K)

Hartz *et al.* (1982) studied samples representative of typical landfill solid wastes (five different sites in the United States) to quantify Ea under two conditions; a change in gas rate without a change in microbial population; and a change in population. Values for Ea were determined to be in the range of 18-24 kcal/mole CH_4 and an average of 23 kcal/mole CH_4. They also noted an optimum temperature to be 41°C with methane production ceasing between 48 and 55°C.

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2.5.2.1. Energy Transport

In order to use the relationship given in Equation (29), estimations of the temperature inside the landfill must be made. This will require a model of the heat transfer processes. The heat diffusion equation can be written for three directions (x,y,z) as (Incropera and Dewitt 1990):

$$\frac{\partial}{\partial x}\left(k_x\,\frac{\partial T}{\partial x}\right) + \frac{\partial}{\partial y}\left(k_y\,\frac{\partial T}{\partial y}\right) + \frac{\partial}{\partial z}\left(k_z\,\frac{\partial T}{\partial z}\right) + \overset{\bullet}{q} = \rho\,C_p\,\frac{\partial T}{\partial t} \tag{30}$$

where T = temperature

$\overset{\bullet}{q}$ = heat flow rate

ρ = density

C_p = specific heat

k_i = diffusion coefficient in the respective direction

t = time

x, y, z = space coordinates

The specific heat can be broken into specific heat for water and for waste. If one-dimensional heat flow is assumed and the diffusion coefficient, k_i, is assumed constant, the equation can be written:

$$\frac{\partial}{\partial z}\left(k_z\,\frac{\partial T}{\partial z}\right) + \overset{\bullet}{q} = \rho_{waste}\,C_{waste}\,\frac{\partial T}{\partial t} + \rho_{h_2o}\,C_{h_2o}\,\phi\,\frac{\partial S_w T}{\partial t} \tag{31}$$

where ϕ = porosity

S_w = degree of saturation

k_z = effective thermal conductivity in the vertical direction

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T = temperature

ρ_{waste} = density of waste

ρ_{h_2o} = density of water

C_{waste} = specific heat of waste

C_{h_2o} = specific heat of water (Akan 1984)

This equation accounts for the heating of the water and the waste inside the landfill. Boundary conditions on the top will be a specified temperature condition. Specified flux conditions on the bottom can be created by setting the two bottom node temperatures equal. Hence, an estimation can be determined for temperatures inside the landfill, and the effects of the temperatures on the gas production rate can be modeled.

Other factors affecting the methane gas production rate have been studied and their effects are summarized in Table 5. Optimum anaerobic conditions for rapid gas productions are rarely, if ever, observed in normally operated landfills. In addition to a limiting condition, mass transport is probably a rate-limiting factor since the contact opportunity between the organisms and the organic substances or inorganic nutrients is very limited.

2.5.3 Theoretical Kinetic Models for Gas Production

The importance of the physical and chemical variables in a landfill is known, however, it is currently not possible to accurately describe the conditions with respect to gas production

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Table 5. Factors Affecting Gas Production Rate

FACTOR	AFFECT	ASSUMPTIONS MADE
Refuse Composition	Rate and Maximum Yield	Maximum methane with high percentage of biodegradable material (food, garden wastes, paper, textiles, and wood). Increases production by by increasing organic content (add sewer sludge, manure, agriculture wastes etc) (EMCON 1980).
Toxic or Inhibitory Material	Upset Activity of methanogenic bacteria	Industrial wastes may include inhibitory materials (salts and toxic organic compounds). Salts stimulate production at low levels, but inhibit at higher concentrations. Increase methane production by limiting toxic or inhibitory material (ferrous and nonferrous metals less daily cover material) in the landfill (McCarty 1964).
Oxygen	Oxygen is toxic to anaerobic processes	High gas extraction rates may create a pressure gradient across the cover or perimeter, drawing in oxygen. Careful operation of gas recovery and sealing the landfill can increase methane production (EMCON 1980).
Moisture Content	High Moisture favors production rate	60-80% Moisture Content favors maximum methane production. Moisture Content of waste averages 25% when initially placed in the landfill. Increased moisture can have a direct effect on the production rate. The addition of water or sludge or recirculation of leachate can increase gas production rates (EMCON 1980; DeWalle et al. 1978).
pH	Optimum pH range: 7.0 - 7.2	Methane production will proceed in a pH range of 6.5 to 8.0. A drop below 6.0 may be toxic. The pH can be improved through the addition of chemicals and leachate recirculation. This will also improve mass transport (McCarty 1964).
Refuse Size	Size reduction enhances production	Reduced size increases the surface area for organisms to attack organic material (DeWalle et al. 1978).
Temperature	Warm temperature favors production	Production can be seasonal. Temperatures below 10°C show a drastic drop in production. Temperatures as high as 71°C have been reported. Temperature in the landfill is considered uncontrollable (DeWalle et al. 1978).

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inside a landfill. There currently is no explicit relationship to describe gas production behavior in a landfill due to different existing factors. Hence, the development of models needs to use the most simplified model consistent with fundamental principles, and empirically adjust the kinetic rate constants to account for variations in parameters. Three models use first-order kinetics to predict expected gas production. They are the Palos Verdes, the Scholl Canyon, and the Sheldon Arleta Kinetic Models (EMCON 1980; Bureau of Sanitation, Los Angeles 1975). A description of these three models can be found in Appendix A. It is noted that these models have not been verified by field data. Since the effect of each variable is not known, the variables are all lumped into a pseudo first-order expression resulting from applying limiting conditions to the Monod equation (Monod 1950). The equation is substrate limiting and is:

$$\frac{dS}{dt} = \frac{KXS}{K_s + S} \tag{32}$$

where K = maximum rate of substrate utilization per unit mass of MOs

X = concentration of microorganisms

K_s = waste concentration when the rate is one-half the maximum rate of substrate utilization

S = concentration of substrate surrounding the microorganisms

In the two extreme cases when S is very large (S >> K_s) and when S is very small (S << K_s), the equation can be approximated by:

$$\frac{dS}{dt} = KX, \quad \text{when } S >> K_s \tag{33}$$

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$$\frac{dS}{dt} = \frac{KXS}{K_s} \text{ , when } S \ll K_s \tag{34}$$

These give a zero order equation with respect to substrate concentration when it is very large and a first order equation with respect to substrate concentration when it is very small. When using this equation in a landfill, it must be assumed that organic waste is the limiting nutrient for the rate-determining methane bacteria.

Tchobanoglous *et al.* (1993) predict that under normal conditions, the rate of gas production (decomposition) reaches a peak within the first two years and then slowly tapers off, continuing up to 25 years or more. Variations in the rate of gas production from the anaerobic decomposition of the rapid (five years or less) and slowly (5 to 50 years) biodegradable organic material in municipal solid waste can be modeled as shown in Figure 11. Gas production is assumed to start at the end of the first full year of landfill operation. The yearly rates of decomposition for rapidly and slowly decomposable material are based on a triangular gas model in which the peak rate of gas production occurs between one and five years. The total rate of gas production from a landfill in which wastes were placed for a period of five years is obtained graphically by summing the gas produced from the rapidly and slowly biodegradable portions of the municipal solid waste deposited each year. The total amount of gas produced corresponds to the area under the rate curve. An example of the use of this model to determine the total amount of gas produced in a landfill is illustrated in Tchobanoglous *et al.* (1993).

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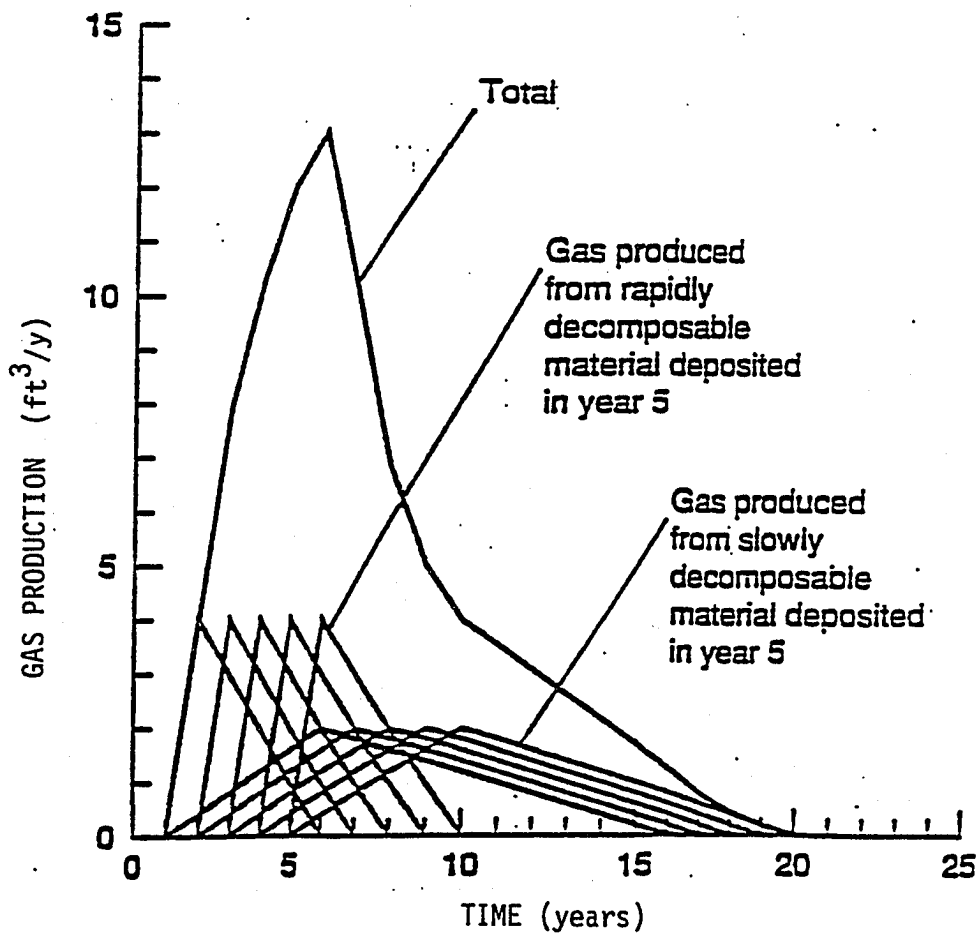
Figure 11. Variations in the rate of gas production from anaerobic decomposition of rapid and slowly biodegradable organic material (After Tchobanoglous *et al.* 1993).

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Hartz (1980) used real landfill gas production data to consider the use of different types of ordered reactions as models. A summary of the mathematical models he used are given in Table 6. The standard ordered reaction is of the form:

$$- \frac{dP}{dt} = A \ C^k \qquad (35)$$

where C = remaining substrate at time t

P = gas produced in time step

k = order of the reaction

A = fitting parameter

t = time

To conserve matter, an ultimate gas yield, Q, exists and is defined as the sum of the gas produced and remaining substrate. Substituting this into equation (35) gives the following gas production equation:

$$- \frac{dP}{dt} = A \ (Q - P)^k \qquad (36)$$

in which the $\frac{dP}{dt}$ term represents the rate of gas evolution and P is the total gas produced up to the point at which the production rate is determined (Hartz 1980).

In many landfills the available moisture content is insufficient to allow complete conversion of the biodegradable organic constituents in the waste. The optimum moisture content is 50 to 60%. Also, in many landfills, the moisture is not uniformly distributed. When the moisture content of a landfill is limited, the

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TABLE 6.

GAS PRODUCTION RATE MODELS

1. $\dfrac{dP}{dt} = A\ P^{-k}$

2. $\dfrac{dP}{dt} = A\ e^{-k_1 t}$

      plus $\dfrac{dP}{dt} = k_2$

3. $\dfrac{dP}{dt} = A\ (Q-P)^{3.0}$

4. $\dfrac{dP}{dt} = A\ e^{-kt}$

5. $\dfrac{dP}{dt} = \dfrac{K_m}{V_m(Q-P)} - \dfrac{1}{V_m}$

6. $\dfrac{dP}{dt} = A\ t^{-k}$

7. $\dfrac{dP}{dt} = A\ (Q-P)^{0.5}$

where $\dfrac{dP}{dt}$ = the gas production rate (ft^3/lb d)

    A = fitting parameters

    P = gas previously produced (ft^3/lb of waste)

    k = rate coefficients (k, k_1, k_2)

    t = time

    Q = maximum gas production per mass (ft^3/lb)

    V_m = maximum rate of reaction

After Hartz (1980)

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gas production curve is more flattened and extends for a greater period of time as illustrated in Figure 12. The production of landfill gas over extended periods of time is of great significance with respect to the management strategy to be adopted for postclosure maintenance (Tchobanoglous *et al.* 1993). There is still considerable uncertainty with regard to the rate of gas production, the ultimate amount of gas that can be produced, and the factors that influence the rate of production such as density, moisture content, temperature, solid waste composition, and exposure to air (seeping down from the cap) (DeWalle 1978).

2.5.4. Landfill Gas Transport

Usually, gases produced in soils are transported and released to the atmosphere through molecular diffusion (partial pressure for a gas exists). However, in an active landfill, the internal pressure is usually greater than atmospheric pressure and thus, landfill gas will be transported convectively (a total pressure gradient exists) as well as diffusively (Tchobanoglous *et al.* 1993). Convective flow is in the direction in which total pressure decreases and diffusive flow is in the direction in which its concentration (partial pressure) decreases. For some particular gases, convection and diffusion flows can be in opposing directions and tend to cancel each other out, however, most cases exhibit flow in the same direction. Diffusion can occur by several mechanisms to include; ordinary molecular

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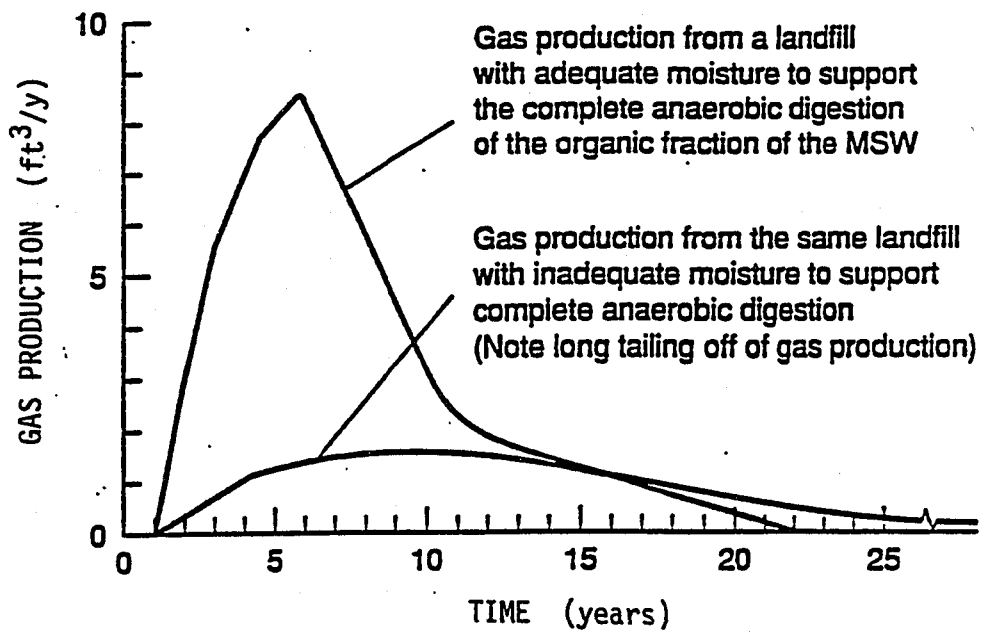
Figure 12. Effect of reduced moisture content on the production of landfill gas (After Tchobanoglous *et al.* 1993).

diffusion; Knudsen diffusion; and surface migration (significant only when diffusing gases are adsorbed onto the porous medium in a mobile layer) (EMCON 1980). Tchobanoglous *et al.* (1993) report that other factors include sorption of the gases into liquids or solids and the generation or consumption of a gas component through chemical reactions or biological activity. Diffusive flow is important in assessing the landfill gas hazard due to lateral migration, however, its effects are usually negligible where an induced exhaust system is used to create total pressure gradients throughout the landfill. In cases where gases are withdrawn from a landfill by applying a partial vacuum to a penetrating well, convective flow dominates (Tchobanoglous *et al.* 1993).

Modeling gas flow through porous media requires: (1) a set of equations describing mass transport for each gas which include a term for diffusive flow and convective flow; (2) an equation describing fluid flow (Darcy's Law); and (3) an equation of state for the gases. Some models (Mohsen 1975; Mohsen *et al.* 1977; Moore 1950) designed to describe gas flow through or from a landfill are available, but they do not contain an explicit term for gas production within the landfill. Findikakis and Leckie (1979) developed a model to describe gas migration within a landfill and incorporates an explicit term for the production of gases within the landfill volume. EMCON (1980) reports that all of the models require site-specific data such as porosity, permeability, gas composition, pressure gradients, or rate of gas production. None of the models can account for gas movement under a partial vacuum during pumping.

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The effects of landfill water movement are usually on a much larger time scale than gas migration. The rate of gas production increases quickly followed by a general leveling off and slow decrease in the production rate lasting many years. This means that the rate of production can be considered steady state over a period of months. Hence, the transport of the gas need not be modeled since the gas formed, quickly transports out of the landfill relative to the leachate transport (Young 1989).

Although most of the methane escapes to the atmosphere, both methane and carbon dioxide have been found at concentrations up to 40 percent at lateral distance up to 400 feet from the edges of unlined landfills. The extent of this lateral movement depends on the landfill cover and surrounding soil. Proper venting of methane should not pose a problem (except it is a greenhouse gas). Carbon dioxide poses a problem due to its density. Since it is 1.5 times as dense as air and 2.8 times as dense as methane, it tends to move toward the bottom of the landfill, thus increasing concentrations in the bottom of the landfill for many years. If a soil liner is used (a geomembrane liner can limit the transport of carbon dioxide), the carbon dioxide can move downward through diffusive transport through the liner, the underlying formation and into the groundwater. Carbon dioxide is readily soluble in water and react to form carbonic acid through the following reaction:

$$CO_2 + H_2O \rightarrow H_2CO_3 \tag{37}$$

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This reaction lowers the pH, which in turn can increase the hardness and mineral content of the groundwater through solubilization (Tchobanoglous *et al.* 1993).

In summary, there are numerous factors involved in the production of gases. Since the actual conditions and ongoing processes inside landfills are dependent on many variables, it is difficult to relate the factors to the production rates. Modeling the gas production uses linear models with assumptions for the total gas yield and the estimated life of the landfill. An empirical relationship will be put together to combine moisture content effects with temperature effects on the gas production rates.

3. DESCRIPTION OF THE MATHEMATICAL MODEL

3.1 GENERAL DESCRIPTION

The mathematical model presented in this study is a quasi-three-dimensional model designed to account for the governing processes occurring in a landfill conceptualized as a porous medium. The model discretizes a landfill into three spatial dimensions. Equations are solved to determine the various parameters at each node in the landfill per time step. Water flow through the landfill in the vertical direction is determined by solving the Richard's Equation. A two-dimensional runoff model on the surface is coupled with the subsurface flow model for the landfill. The water infiltration rate into the landfill at the surface is determined by the Richard's equation on the boundary. This infiltration rate is used by the runoff model as losses to determine the head on the top boundary of the landfill. This top boundary head is used by the Richard's Equation, hence they are coupled. The values of the water head and moisture content are solved at various points in the landfill, called nodes. After each node head is solved for each time step, a lateral flow determination (and leakage out of the side boundaries) is made for the saturated nodes. Lateral flow is neglected in unsaturated nodes. Lateral flows are treated as node

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source terms for the next time step. The landfill liner and leachate collection system is modeled to determine the boundary conditions at the bottom of the landfill.

The moisture content obtained by the Richards Equation is used to determine the contaminant production. The production is used as a source term for a contaminant transport portion of the model. A gas production rate is also determined in order to predict gas generation. The inputted time steps for the model are on the order of an hour for the flow and contaminant transport portions. The gas production portion uses a time step on the order of one day due to its approximately steady state behavior compared to water flow.

3.2 WATER FLOW MODEL

The governing equation for the three-dimensional saturated/ unsaturated flow of water through a porous medium described by Equation (16) repeated here as:

$$\frac{\partial}{\partial x}\left(K_s K_r \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_s K_r \frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_s K_r \frac{\partial h}{\partial z}\right) = \phi \frac{\partial S_w}{\partial h}\frac{\partial h}{\partial t} \tag{38}$$

where z = vertical distance

x = lateral distance

y = lateral distance

K_s = saturated permeability of the landfill cell

K_r = relative permeability of the cell

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S_w = degree of saturation of the cell

ϕ = effective porosity

t = time

h = head (z + P).

The values for K_r and S_w are expressed by Russo (1988) in the following empirical relationships:

$$K_r = \exp(-\alpha \cdot abs(P)) \tag{39}$$

$$S_w = (e^{-0.5 \, \alpha \, (P)} (1+0.5\alpha P))^{2/(m+2)} \tag{40}$$

where α and m are porous medium parameters.

The moisture flow in a landfill is dominantly vertical. Therefore, for simplicity the Richard's equation is written in one-dimensional form as:

$$\frac{\partial}{\partial z} \left(K_s \, K_r \, \frac{\partial h}{\partial z} \right) = \phi \, \frac{\partial S_w}{\partial h} \, \frac{\partial h}{\partial t} + L \tag{41}$$

where L is a source term. In the unsaturated zone, the lateral flow is ignored. For saturated cells, however, the lateral flow from the adjacent cells is treated as a source term, that is:

$$L = - \frac{\partial}{\partial x} \left(K_s K_r \frac{\partial h}{\partial x} \right) - \frac{\partial}{\partial y} \left(K_s K_r \frac{\partial h}{\partial y} \right) \tag{42}$$

in which $K_r = 1.0$ for saturated flow.

Equation (41) is a nonlinear partial differential equation for which an analytical solution is not available. Therefore, a finite

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difference method is employed in the mathematical model to solve this equation.

3.2.1 Finite Difference Equations for Moisture Flow

To apply the finite difference method a landfill is divided into a finite number of vertical cells. Each vertical cell stretches from the surface to the base of the landfill. The cell is then divided into a number of equidistant discrete elements as shown in Figure 13. The thickness of each element is Δz. The thickness of the landfill cover, the liner, and other intermediate layers are specified separately. These elements are numbered starting from the top of the cell and increasing in the the downward direction. The number of elements in each cell depends on the height of the cell and the specified thickness, Δz, of each element. Values for the permeability (K_s), relative permeability (K_r), porosity (ϕ), and head (h), are homogeneous throughout each element, but they can change from one element to another. These properties are simulated to act on the center of the element, called the node.

The physical domain is divided into a computational domain as shown in Figure 14. The vertical spatial steps in the computational domain will be designated by "j" and the temporal steps will be denoted by "n". Forward difference operators are applied to discretize Equation (41) in the vertical direction and in time. The discretized equation is as follows:

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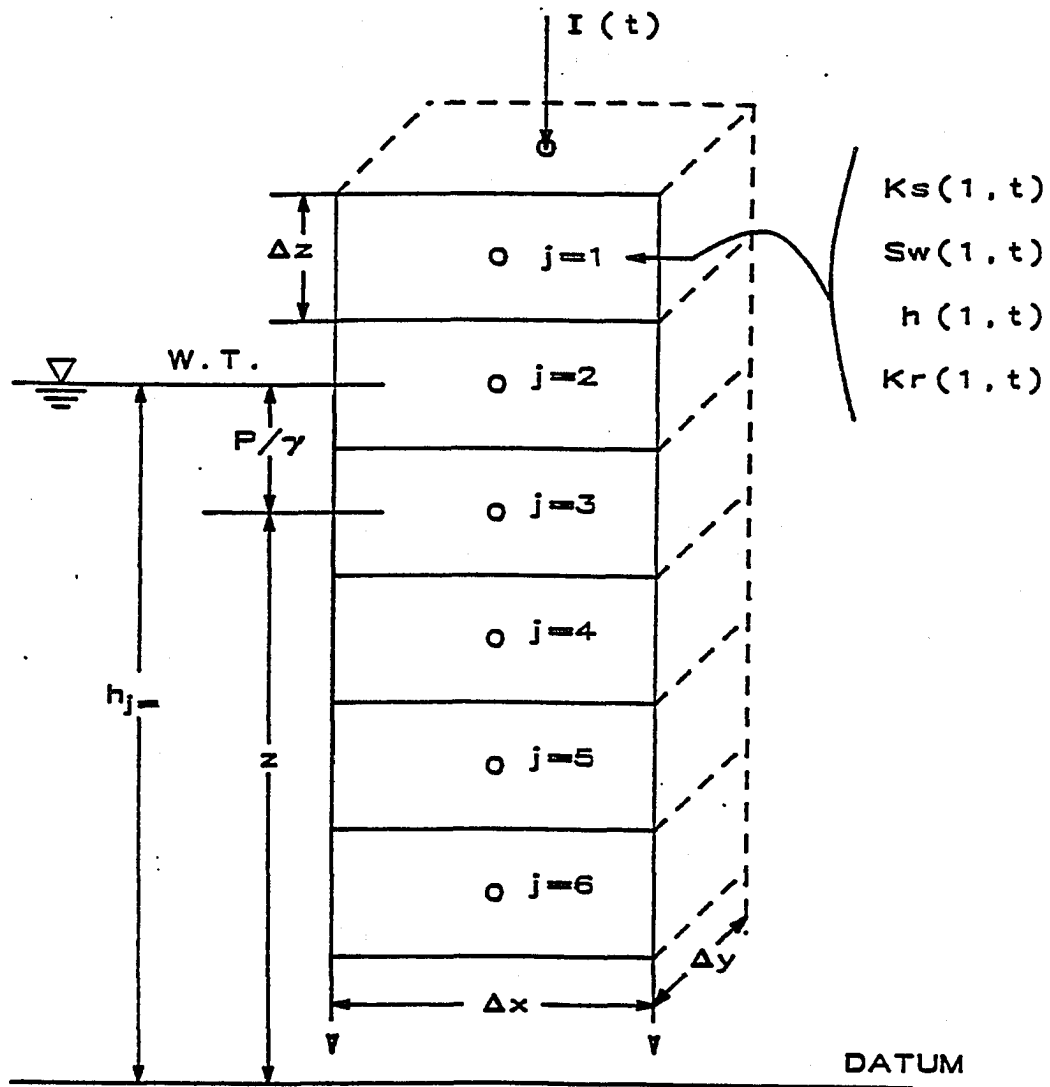
Figure 13. Spatial division of the vertical element

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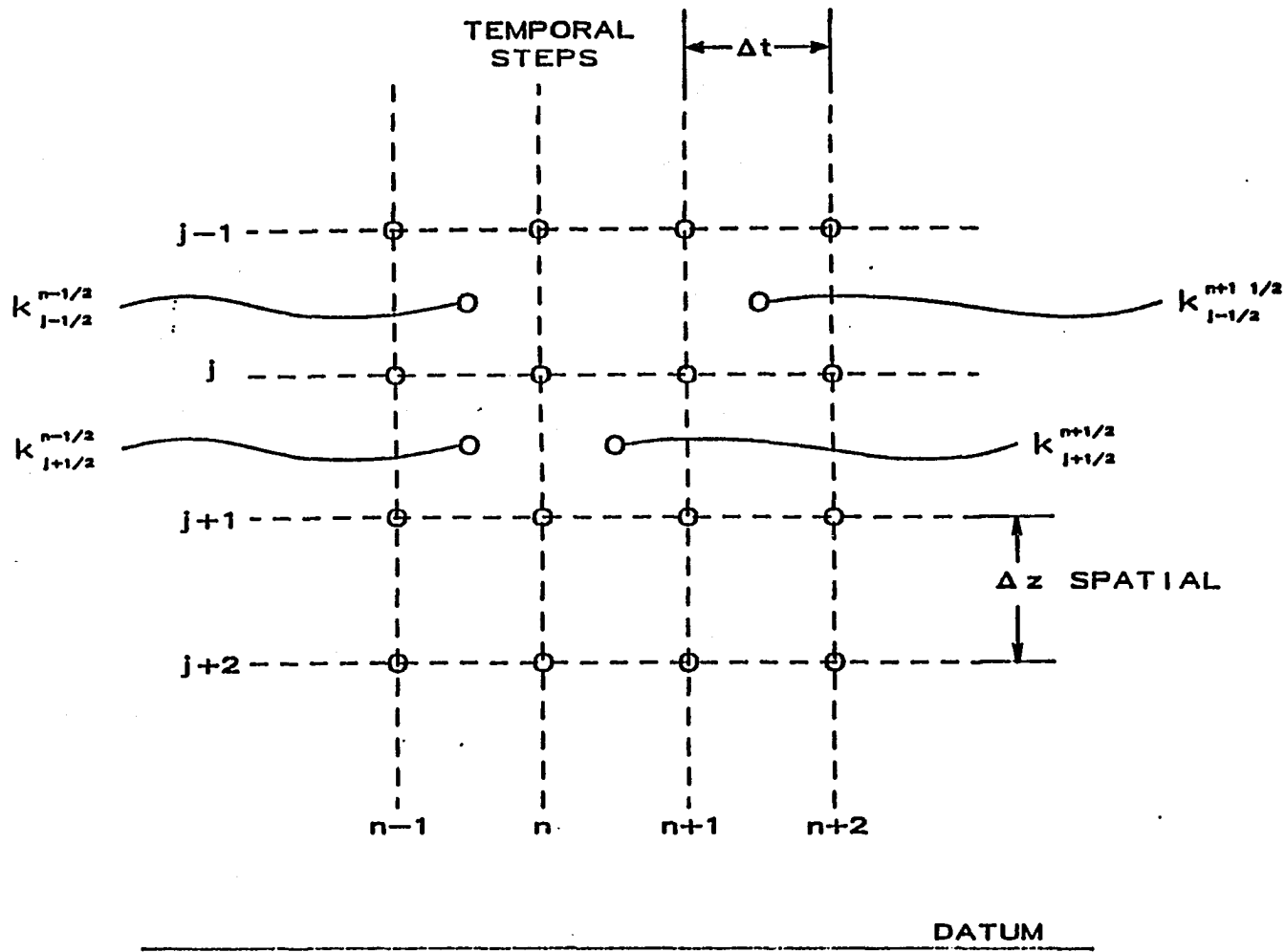
Figure 14.   Division of vertical cell into spatial and temporal steps.

$$K_{s_{(j+1/2)}} K_{r_{(j+1/2)}}^{n+1} \left(\frac{h_{j+1} - h_j}{\Delta z^2} \right)^{n+1} - K_{s_{(j-1/2)}} K_{r_{(j-1/2)}}^{n+1} \left(\frac{h_j - h_{j-1}}{\Delta z^2} \right)^{n+1} + L$$

$$= \phi \frac{\partial S_w}{\partial p} \left(\frac{h_j^{n+1} - h_j^n}{\Delta t} \right) \tag{43}$$

Variables are the same as described in Equation (39). However, the chain rule has been applied to replace $\frac{\partial S_w}{\partial h}$ by $\frac{\partial S_w}{\partial p} \cdot \frac{\partial P}{\partial h}$. Since P is a pressure head, $\frac{\partial P}{\partial h} = 1$. The (j+1/2) and (j-1/2) operators indicate the average value of the two elements for the parameter indicated. For example, the $K_{s_{(j+1/2)}}$ term represents the average K_S for the "j" and the "j+1" element. The "n+1" operator represents the values at the next time step, while the "n" operator represents the current time step. The averaged values are calculated as follows:

$$K_{s_{(j+1/2)}} = \frac{K_{s_{(j+1)}} + K_{s_{(j)}}}{2} \tag{44a}$$

$$K_{s_{(j-1/2)}} = \frac{K_{s_{(j-1)}} + K_{s_{(j)}}}{2} \tag{44b}$$

$$K_{r_{(j+1/2)}} = \frac{K_{r_{(j+1)}} + K_{r_{(j)}}}{2} \tag{44c}$$

$$K_{r_{(j-1/2)}} = \frac{K_{r_{(j-1)}} + K_{r_{(j)}}}{2} \tag{44d}$$

3.2.2 Initial and Boundary Conditions

Initial and boundary conditions must be given to obtain a unique solution to Equation (41). The initial distribution of the piezometric head within the landfill constitutes the initial

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condition. Two boundary conditions, one at the top and one at the bottom, are needed for each vertical element of the landfill.

The mathematical model allows specified flux and specified head conditions at either boundary. For the top boundary, the specified flux equation is written as:

$$I + K_s K_r^{n+1} \left(\frac{h_2 - h_1}{\Delta z}\right)^{n+1} = \phi \Delta z \frac{\partial S_w}{\partial h} \left(\frac{h_1^{n+1} - h_1^{n}}{\Delta t}\right) \tag{45}$$

where I is the specified infiltration flux and/or recirculated leachate rate. In the absence of leachate circulating during dry weather conditions, $I = 0$. During a rain event, $I =$ rate of rainfall until the surface layer of the landfill becomes saturated. From then on a specified head boundary condition is employed, that:

$$h_1 = Y \tag{46}$$

where Y is the depth of standing or flowing water over the landfill as determined from the runoff component of the model.

Mathematically, specified flux and specified head equations can be written for the bottom boundary in a similar way. However, the leachate flux across the bottom boundary is solved by the model. If no landfill liner is specified, the flux out the bottom of the model is set equal to the flux into the bottom element. If a liner is used, a specified head condition applies to the bottom boundary. The specified head for the bottom boundary is determined by the depth of saturated flow over the liner.

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3.2.3 Method of Solution for Moisture Flow Equation

The solution method described in this section is for a single vertical column. The same procedure is applied to all the vertical columns of the landfill separately. The unknown quantities in the moisture flow equations written in finite difference form are h_{j-1}^{n+1}, h_j^{n+1}, and h_{j+1}^{n+1}. All the other terms are either known or can be evaluated in terms of the unknowns using the linking equations such as Equations (39) and (40). The source term, L, is evaluated using the previous time step results and thus it is known. However, because the evaluation of L requires a three dimensional formulation, it will be discussed in a separate section below.

For a vertical element of the landfill made of N nodes, the top boundary equation, the bottom boundary equation, and Equation (41) written for N-2 interior nodes will form a set of N nonlinear simultaneous equations containing N unknowns: h_j^{n+1}, j=1 to N. Each of these nonlinear equations can be expressed symbolically as:

$$f_j \left(h_{j-1}, \ h_j, \ h_{j+1} \right) = 0 \qquad (47)$$

A generalized Newton iteration method is adopted to solve the nonlinear system of equations. Computation for the iterative procedure begins by assigning a set of trial values to the N unknowns. Substitutions of the trial values into the finite difference equations yields the residuals, $\bar{f}_j$. A new set of values for the unknowns h_j^{n+1} are estimated for the next iteration to make the residuals approach to zero. This is accomplished by calculating the

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corrections Δh_j^{n+1} such that the total differentials of the functions f_j are equal to the negative of the calculated residuals $\tilde{f}_j$, i.e.:

$$AA_j \cdot \Delta h_{j-1} + BB_j \cdot \Delta h_j + CC_j \cdot \Delta h_{j+1} = DD_j \qquad (48)$$

where $AA_j = \dfrac{\partial f_j}{\partial h_{j-1}}$

$BB_j = \dfrac{\partial f_j}{\partial h_j}$

$CC_j = \dfrac{\partial f_j}{\partial h_{j-1}}$

$DD_j = \Delta f_j = -\tilde{f}_j$

evaluated using the guessed values of h_j.

Equation (41) written for N nodes will yield N linear simultaneous equations containing N unknowns Δh_j, $j = 1,2,\ldots,N$. This system of linear equations is solved by use of a matrix conversion technique for the corrections Δh_j. These corrections are used to improve the guessed values of h_j, and the same procedure is repeated until all the corrections become tolerably small. It is worthwhile to note that the coefficient matrix of the matrix equation formed by the linear equation in each iteration cycle has a tridiagonal structure, and the system of equations can be solved very efficiently using the Thomas algorithm.

The coefficients AA_j, BB_j, CC_j, and DD_j are evaluated for interior nodes, that is for $j = 2$ to N-1, as follows:

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$$AA_j = \left(K_s K_r\right)_{(j-1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2} \tag{49a}$$

$$BB_j = -\frac{\partial S_w}{\partial P} - \left(K_s K_r\right)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi (\Delta z)^2} - \left(K_s K_r\right)_{(j-1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2} \tag{49b}$$

$$CC_j = \left(K_s K_r\right)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2} \tag{49c}$$

$$DD_j = -\tilde{f}_j = \frac{\partial S_w}{\partial P}\left(h_j^{n+1} - h_j^n\right) - \left(K_s K_r\right)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2}\left(h_{j+1} - h_j\right)^{n+1}$$

$$- \left(K_s K_r\right)_{(j-1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2}\left(h_{j-1} - h_j\right)^{n+1} - \frac{I \, \Delta t}{\phi \, \Delta z} \tag{49d}$$

Note that K_r and $\dfrac{\partial S_w}{\partial h}$ are evaluated using Equations (39) and (40). Also, the most recent values of h_j^{n+1} are used in Equation (49d).

For the top $(j = 1)$ and the bottom $(j = N)$ boundaries, the specified head boundary equations are written as:

$$AA_j = 0 \tag{50a}$$

$$BB_j = 1 \tag{50b}$$

$$CC_j = 0 \tag{50c}$$

$$DD_j = H_j \tag{50d}$$

where H_j is the specified head.

For specified flux conditions at the top boundary $(j = 1)$,

$$AA_j = 0 \tag{51a}$$

$$BB_j = -\frac{\partial S_w}{\partial p} - \left(K_s K_r\right)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2} \tag{51b}$$

$$CC_j = \left(K_s K_r\right)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi \, (\Delta z)^2} \tag{51c}$$

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$$DD_j = \frac{\partial S_w}{\partial p} \left(h_j^{n+1} - h_j^{m}\right) - (K_s K_r)_{(j+1/2)}^{n+1} \frac{\Delta t}{\phi(\Delta z)^2} \left(h_{j+1} - h_j\right)^{n+1} - I\frac{\Delta t}{\phi \Delta z} \quad (51d)$$

The bottom flux will be determined from the model and is the drainage to the leachate collection system. If a collection system is not installed, the drainage would be stored in the bottom of the landfill or leak through the liner to the soil under the landfill.

3.2.4 Lateral Flows

The possibility of lateral flows is established when a head difference exists between adjacent nodes in the landfill. The model simulates lateral flows when these conditions are reached for saturated nodes, which makes the model quasi-three dimensional. The landfill's vertical cells are separated into rows (i=1 to NR) and columns (k=1 to NC). After determining the heads for each vertical cell, the lateral flows between adjacent nodes are calculated for each node using Equation (42). The flowrate into the cell is determined by multiplying the velocity by the cross-sectional area between the adjacent cells. This flow is then divided by the surface area of the cell to get a source term in the units of length per time. This flow becomes a source term, "L", for Equation (41) for the next time step. As shown in Figure 15, each node will receive (or lose) flow from the four adjacent nodes. If the node is on a lateral boundary, a boundary condition for lateral flow will be specified. In finite difference form these lateral flow sources are established for positive flow to be inward as follows:

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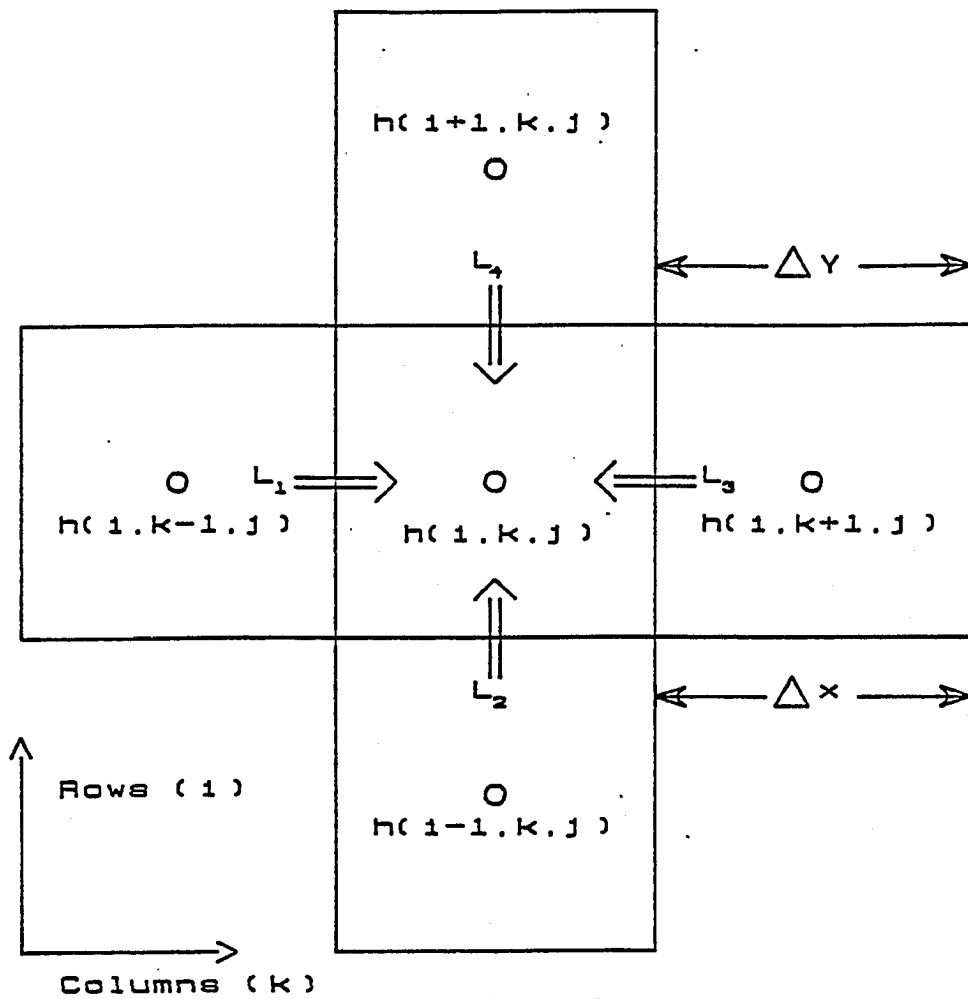
Figure 15. Lateral flow between adjacent cells.

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$$L_1 = \left(\frac{K_{s_{(i,k-1,j)}} + K_{s_{(i,k,j)}}}{2} \right) \left(\frac{K_{r_{(i,k-1,j)}} + K_{r_{(i,k,j)}}}{2} \right) \left(\frac{h_{(i,k-1,j)} - h_{(i,k,j)}}{\frac{\Delta y_{k-1}}{2} + \frac{\Delta y_k}{2}} \right) \frac{\Delta x \Delta z}{\Delta x \Delta y} \qquad (52a)$$

$$L_2 = \left(\frac{K_{s_{(i-1,k,j)}} + K_{s_{(i,k,j)}}}{2} \right) \left(\frac{K_{r_{(i-1,k,j)}} + K_{r_{(i,k,j)}}}{2} \right) \left(\frac{h_{(i-1,k,j)} - h_{(i,k,j)}}{\frac{\Delta x_{i-1}}{2} + \frac{\Delta x_i}{2}} \right) \frac{\Delta y \Delta z}{\Delta x \Delta y} \qquad (52b)$$

$$L_3 = \left(\frac{K_{s_{(i,k+1,j)}} + K_{s_{(i,k,j)}}}{2} \right) \left(\frac{K_{r_{(i,k+1,j)}} + K_{r_{(i,k,j)}}}{2} \right) \left(\frac{h_{(i,k+1,j)} - h_{(i,k,j)}}{\frac{\Delta y_{k+1}}{2} + \frac{\Delta y_k}{2}} \right) \frac{\Delta x \Delta z}{\Delta x \Delta y} \qquad (52c)$$

$$L_4 = \left(\frac{K_{s_{(i+1,k,j)}} + K_{s_{(i,k,j)}}}{2} \right) \left(\frac{K_{r_{(i+1,k,j)}} + K_{r_{(i,k,j)}}}{2} \right) \left(\frac{h_{(i+1,k,j)} - h_{(i,k,j)}}{\frac{\Delta x_{i+1}}{2} + \frac{\Delta x_i}{2}} \right) \frac{\Delta y \Delta z}{\Delta x \Delta y} \qquad (52d)$$

where $h_{(i,k,j)}$ = head for node i,k,j

Δy_k = lateral length for the element in the y direction ("k")

Δx_i = lateral length for the element in the x direction ("i")

$\Delta x,y,z$ = lengths of the center cell in respective directions

L_n = lateral flow into the node from the respective adjacent

node

K_s (i,k,j) = permeability of node i,k,j

K_r (i,k,j) = relative permeability of node i,k,j

and all values are at the current time step. The total lateral flow

into a node is given by:

$$L = L_1 + L_2 + L_3 + L_4 \qquad (53)$$

Boundary conditions for lateral leakage out of the landfill are also

simulated for saturated zones. Unsaturated nodes will produce zero

leakage

($L = 0$) and leakage from saturated zones will be governed by:

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$$L = - K_S \, K_r \left(\frac{h - z}{\frac{\Delta y}{2}} \right) \tag{54}$$

where L = lateral leakage from the saturated node across the boundary

K_S = saturated permeability of the node

K_r = relative permeability of the node

h = head of the node

z = vertical height of the node above a datum

Δy = lateral length of the node

All the lateral terms are summed up and input into the mass balance governing equation as a source term with the infiltration or bottom seepage terms on the vertical boundaries. At each time step the moisture content for each node is calculated along with lateral and vertical seepage from the landfill.

3.3 RUNOFF MODEL

The infiltration of water into the landfill is one of the most critical steps in order to directly determine the leachate and indirectly determine the gas generation. A model to describe the precipitation, evaporation, and transpiration is used in order to characterize the sources and losses of water. The mathematical models used in the HELP model (Schroeder *et al.* 1983b) for evapotranspiration perform these tasks.   Evaporation and transpiration will be determined and the resulting loss terms built into the upper boundary layer mass balance.

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In addition to losses due to evapotranspiration, surface runoff must be taken into account. Since landfill surfaces are usually of a two-dimensional form, a two-dimensional overland flow model is employed to determine the amount of infiltration.

3.3.1 Governing Equations for Runoff Model

The formulation of the model is similar to the previously reported by Hromadka and Durbin (1986) and Guymon and Hromadka (1986) and discussed in Section 2.2.3. The governing two-dimensional equation is:

$$- \frac{\partial Q_x}{\partial x} - \frac{\partial Q_y}{\partial y} + I_e = \delta \frac{\partial h}{\partial t} \tag{55}$$

where Q_x = flow per width in the x direction

Q_y = flow per width in the y direction

δ = width

h = the water surface elevation

I_e = source and loss terms for rainfall and infiltration

Assuming the friction slopes can be approximated by the slope of the water surface in the x and y directions, it can be written:

$$Q_x = \frac{A_x R_x^{2/3}}{n \left| \frac{\partial h}{\partial x} \right|^{1/2}} \frac{\partial h}{\partial x} \tag{56}$$

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$$Qy = \frac{A_y \, R_y^{2/3}}{n \, \left|\frac{\partial h}{\partial y}\right|^{1/2}} \frac{\partial h}{\partial y} \qquad (57)$$

where A_x = cross-sectional area between nodes (height $\cdot$ Δy)

A_y = cross-sectional area between nodes (height $\cdot$ Δx)

R_x, R_y = hydraulic radii in the x and y directions respectively

n = manning roughness coefficient.

3.3.2 Solution Method for Runoff Model

An explicit finite difference scheme is used to solve the runoff equations. Figure 16 shows the local grid system used to describe the discretized equations. If the heads at a time step are known, the values for flows into each node can be determined using:

$$Q_{x1} = \frac{A_x \, R_x^{2/3}}{n \left(\frac{|h_{i-1,k} - h_{i,k}|}{\Delta x}\right)^{1/2}} \left(\frac{h_{i-1,k} - h_{i,k}}{\Delta x}\right) \qquad (58)$$

A_x is the cross-sectional area between nodes given by water height times Δy. Also, Δx is the distance from one node point, i-1, to the adjacent node point, i. Likewise the incoming flow rates across the other three surfaces are given by:

$$Q_{x2} = \frac{A_x \, R_x^{2/3}}{n \left(\frac{|h_{i+1,k} - h_{i,k}|}{\Delta x}\right)^{1/2}} \left(\frac{h_{i+1,k} - h_{i,k}}{\Delta x}\right) \qquad (59)$$

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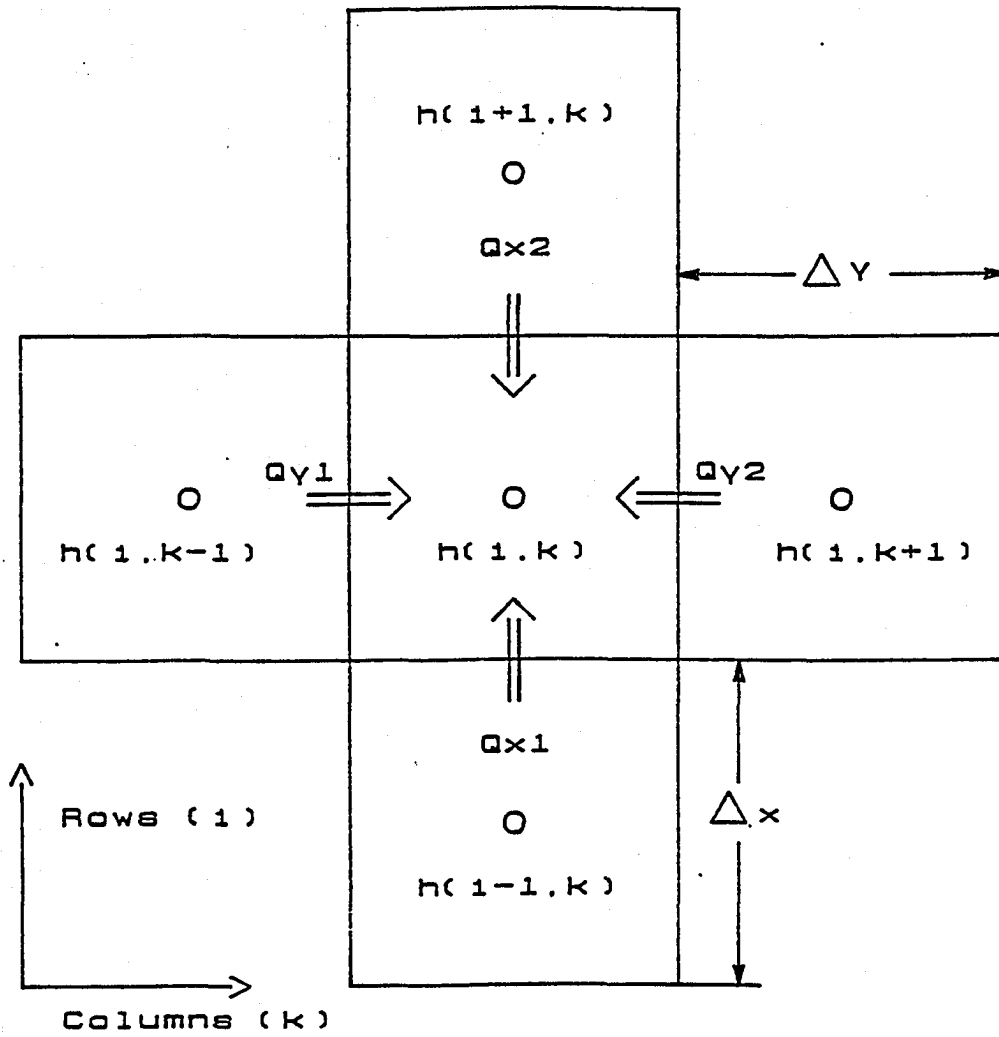
Figure 16. Lateral flow between adjacent cells in surface runoff.

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$$Q_{y1} = \frac{A_y \, R_y^{2/3}}{n \left(\frac{|h_{i,k-1} - h_{i,k}|}{\Delta y}\right)^{1/2}} \left(\frac{h_{i,k-1} - h_{i,k}}{\Delta y}\right) \qquad (60)$$

$$Q_{y2} = \frac{A_y \, R_y^{2/3}}{n \left(\frac{|h_{i,k+1} - h_{i,k}|}{\Delta y}\right)^{1/2}} \left(\frac{h_{i,k+1} - h_{i,k}}{\Delta y}\right) \qquad (61)$$

It is noted that flows into the node are considered positive and that the denominator is an absolute value. Adding the flows from each direction will determine the difference in flow, $\frac{\partial Q_i}{\partial x}$, for each direction. Hence, the flow terms from Equation (55) are written as follows:

$$\frac{\partial Q_x}{\partial x} = Q_{x1} + Q_{x2} \qquad (62a)$$

$$\frac{\partial Q_y}{\partial y} = Q_{y1} + Q_{y2} \qquad (62b)$$

Inputs into each node for rain and losses due to infiltration down into the landfill are treated as sources and losses. The sources and losses are entered into the horizontal surface nodes as source terms, I_e, and combined with the flow terms from Equations (55) to yield:

$$Q_{x1} + Q_{x2} + Q_{y1} + Q_{y2} + I_e = \frac{h_{i,k}^{n+1} - h_{i,k}^{n}}{\Delta t} \qquad (63)$$

The terms on the left hand side of the equation are evaluated at time

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n. It should be noted that the infiltration value for the time step n comes from the solution of the subsurface moisture flow equation. Using the current head, $h_{i,k}^n$, the model solves for the new node head, $h_{i,k}^{n+1}$, and the model advances in time explicitly (Hromadka and Durbin 1986).

The output of the explicit scheme is a hydrograph on the surface which is used to determine the water heads on the top boundary of the landfill cells. This head condition is used to determine the specified flux through the boundary node and into the model as infiltration for each time step. Since the runoff routine is for storm events, the model has an option to bypass the routine for long time scenarios. In this case, the infiltration is input as average values for various time periods, depending on the data available.

3.4 LEACHATE COLLECTION/LINER SEEPAGE

Flow out of the bottom boundary of the water flow model enters into the drainage layer of the leachate collection system (if one is installed). The leachate liner and collection system is modeled using a one-dimensional, saturated, unsteady flow model in which the flow moves in a direction parallel to the sloping liner. The system inflows are the leachate which has passed out the bottom of the waste column. Leachate on the liner flows down to the collection drain lines at the liner low points. Leachate also can permeate through the liner in the vertical direction. Landfills which do not have a

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drainage system will build up a leachate column and the only losses will be leakage through the liner.

3.4.1 Liner Model

The equation used to describe the one-dimensional flow through the drainage layer in Figure 17 as described by Korfiatis and Demetracopoulos (1986) and McEnroe (1989a) is:

$$\phi_e \frac{\partial h_c}{\partial t} + \frac{\partial Q}{\partial x} = R - q_b \tag{64}$$

where Q = lateral drainage flow rate along the liner

x = distance parallel to the liner measured upstream

R = recharge rate of leachate from the waste

q_b = leakage rate through the barrier liner

ϕ_e = the effective porosity of the drainage layer

h_c = leachate head or depth of water surface above a datum

t = time

The Dupuit assumption is made to define lateral drainage as:

$$q_b = - K_d \, y \, \frac{\partial h_c}{\partial x} \tag{65}$$

where y = depth of the leachate on the liner

K_d = saturated hydraulic conductivity of the drainage layer

$\frac{\partial h_c}{\partial x}$ = slope leachate flow surface (phreatic surface)

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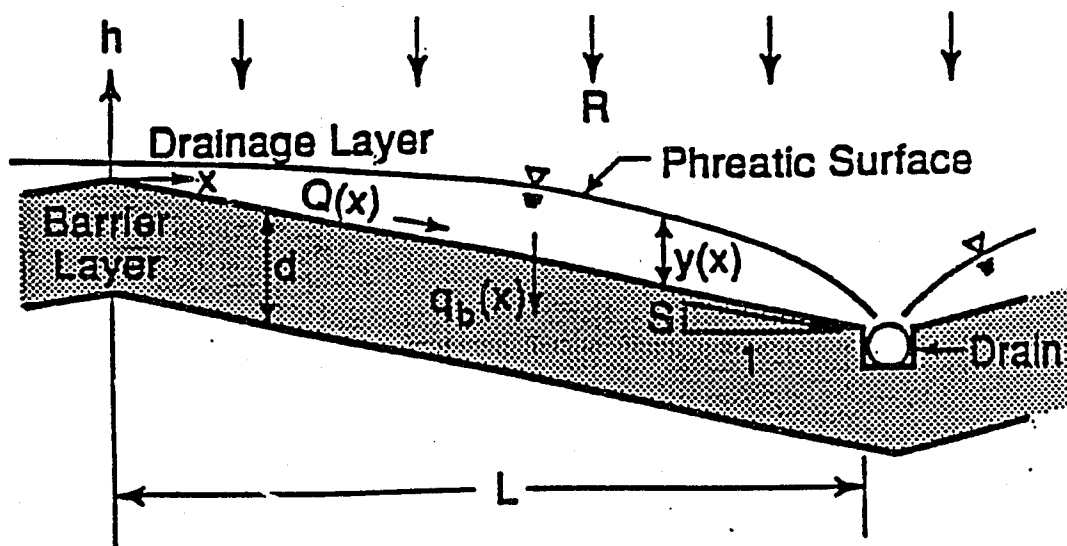
Figure 17. Definition of leachate collection system sketch (Adapted from McEnroe 1989).

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If the datum is taken from the top of the drain line, the leachate head, h, at any point along the liner would be the liner elevation plus the leachate depth, y. The distance along the liner, x, is measured from the upstream peak in order for the scheme to march in the direction of leachate flow. This elevation, A, of the liner peak above the datum (drain) could be computed by multiplying the liner length, L, by the slope, S. Hence, the elevation head could be computed by:

$$h = y + (A - S x) \tag{66}$$

Rearranging Equation (66), leachate depth can be written as:

$$y = h - (A - S x) \tag{67}$$

Assuming liner barrier is saturated, leakage through the liner is modeled using Darcy's law. The local leakage term in the equation, q_b, is written as:

$$q_b = K_L \left(\frac{y + d}{d} \right) \tag{68}$$

where K_L = saturated hydraulic conductivity (vertical) of the liner

   d = thickness of the liner

The possibility of a synthetic liner is present, and Jayawickrama *et al.* (1988) studied leakage rates through flaws in the membrane. Peyton and Schroeder (1990) evaluated numerous designs for landfill liners. In their equation to determine local leakage rates, a leakage fraction was used. The leakage fraction is the fraction of the

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horizontal area of soil through which percolation is occurring under the leaking synthetic liner. Instead of using synthetic liner leaks as a separate term, the hydraulic conductivity, K_L, used in Darcy's law is adjusted to account for the synthetic liner.

The permeable layer above the liner is discretized into several nodes along the liner and the flow terms Q, q_b, and R are multiplied by the node dimensions to give volumetric flows. Flows into each node are designated as positive and the mass balance in Equation (64) can be written:

$$\Delta x \Delta w \cdot \phi_e \frac{\Delta h}{\Delta t} = \Delta w \cdot (Q_1 - Q_2) + \Delta w \Delta x \cdot R - \Delta w \Delta x \cdot q_b \qquad (69)$$

where Q_1 = flow into the element from upstream per unit width

Q_2 = flow out of the element downstream per unit width

R = recharge rate of leachate from waste into node

Δw = the perpendicular width of the liner

Δx = length of the liner element

Dividing everything by Δw and substituting for the Q and Q_b terms yields the discretized equation:

$$\Delta x \cdot \phi_e \left(\frac{h_i^{n+1} - h_i^{n}}{\Delta t} \right) = K_d \, y_{(i-1/2)} \left(\frac{h_{i-1} - h_i}{\Delta x} \right)^{n+1} - K_d \, y_{(i+1/2)} \left(\frac{h_i - h_{i+1}}{\Delta x} \right)^{n+1}$$
$$+ \Delta x \cdot R - \Delta x \cdot K_L \left(\frac{y_i + d}{d} \right) \qquad (70)$$

Since y is a function of x, an average value of y will be taken at each boundary of the element ($y_{(i-1/2)}$ upstream and $y_{(i+1/2)}$ downstream).

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Multiplying both sides by $\dfrac{\Delta t}{\Delta x \, \phi_e}$ and averaging the y terms gives the function, f_i, as follows:

$$f_i = - h_i^{n+1} + h_i^{n} + \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(y_{i-1} + y_i\right)\left(h_{i-1}^{n+1} - h_i^{n+1}\right) - \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(y_i + y_{i+1}\right)\left(h_i^{n+1} - h_{i+1}^{n+1}\right)$$

$$+ \frac{R \Delta t}{\phi_e} - K_L \frac{\Delta t}{\phi_e}\left(\frac{y + d}{d}\right) \tag{71}$$

The y terms are functions of h and can be represented as the respective heads minus the height of the liner. Letting a_i be denoted as the height of the liner above the datum at node "i", it can be written:

$$y_i = h_i - a_i \tag{72}$$

and it can be shown that the Generalized Newton Method coefficients for the liner become:

$$AA_i = \frac{\partial f_i}{\partial h_{i-1}} = \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(2 h_{i-1} - a_{(i-1/2)}\right)^{n+1} \tag{73a}$$

$$BB_i = \frac{\partial f_i}{\partial h_i} = -1 + \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(-4h_i + a_{(i-1/2)} + a_{(i+1/2)}\right)^{n+1} - K_L \frac{\Delta t}{\phi_e d} \tag{73b}$$

$$CC_i = \frac{\partial f_i}{\partial h_{i+1}} = \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(2h_{i+1} - a_{(i+1/2)}\right)^{n+1} \tag{73c}$$

$$DD_i = - f_i = h_i^{n+1} - h_i^{n} - \frac{R \Delta t}{\phi_e} + K_L \frac{\Delta t}{\phi_e}\left(d + h_i - a_i\right)$$

$$+ \frac{K_d \Delta t}{2\phi_e(\Delta x)^2}\left(2h_i^2 - h_{i-1}^2 - h_{i+1}^2 - h_i\, a_{(i-1/2)} + h_{i-1}a_{(i-1/2)} - h_i\, a_{(i+1/2)} + h_{i+1}a_{(i+1/2)}\right) \tag{73d}$$

where $a_{(i+1/2)} = \dfrac{a_i + a_{i+1}}{2}$

$\qquad a_{(i-1/2)} = \dfrac{a_i + a_{i-1}}{2}$

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The upstream boundary conditions are satisfied when $y\frac{\partial h}{\partial x} = 0$. The mass balance equation shows this by setting the incoming flow, Q_{d1}, equal to zero for the top upstream node. Hence, the coefficients become:

$$AA_i = 0 \tag{74a}$$

$$BB_i = -1 - \frac{K_d \Delta t}{2\phi_e(\Delta x)^2} \left(2h_i - a_{(i+1/2)}\right)^{n+1} - K_L \frac{\Delta t}{\phi_e \cdot d} \tag{74b}$$

$$CC_i = - \frac{K_d \Delta t}{2\phi_e(\Delta x)^2} \left(-2h_{i+1} + a_{(i+1/2)}\right)^{n+1} \tag{74c}$$

$$DD_i = h_i^{n+1} - h_i^n + \frac{K_d \Delta t}{2\phi_e(\Delta x)^2} \left(h_i^2 - h_{i+1}^2 - h_i \, a_{(i+1/2)} + h_{i+1} \, a_{(i+1/2)}\right)$$

$$- \frac{R \Delta t}{\phi_e} + K_L \frac{\Delta t}{\phi_e}\left(d + h_i - a_i\right) \tag{74d}$$

The downstream boundary condition postulates a free drainage condition. The gradient at the boundary is approximated by upstream differences between the last and next to last points of the grid (Korfiatis and Demetracopoulos 1986). The two Q flow terms cancel each other out and the equation at the downstream node becomes:

$$\Delta x \, \phi_e \left(\frac{y_i^{n+1} - y_i^n}{\Delta t}\right) = \Delta x \cdot R - \Delta x \cdot K_L\left(\frac{y + d}{d}\right) \tag{75}$$

The coefficients for the downstream node are:

$$AA_i = 0 \tag{76a}$$

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$$BB_i = -1 - K_L \frac{\Delta t}{\phi_e \cdot d} \tag{76b}$$

$$CC_i = 0 \tag{76c}$$

$$DD_i = h_i^{n+1} - h_i^n - \frac{R \Delta t}{\phi_e} + K_L \frac{\Delta t}{\phi_e} \left(d + h_i - a_i \right) \tag{76d}$$

3.4.2 Liner Geometry

The landfill liner and collection system can consist of several drains lines. The liner surfaces sloped downward to the low point drain lines. Each drain line system consists of one or two liner slopes on either side of the drain line. Each liner slope is designated as a liner field. For example, a collection system with two drain lines could have a total of four liner fields, one on each side of a drain line.

The model divides each liner field into lateral liner strips which run perpendicular to the drain lines (Figure 18). Since the leachate flow from each vertical waste cell may vary, the width of the strip, Δw, is the same as the waste cell width above it. The corresponding dimension of the waste cell would be the Δy distance discussed in Section 3.2.1. It is possible to have several waste columns positioned along the liner strip. The total length of the liner strip is the sum of the lengths (Δx) of the waste cells above the liner. This will line up the waste cells above the liner strips.

Each liner strip is broken into several nodes. Using the dimensions of the waste cells, the dimensions of the liner strip, and the assigned Δx of the liner strip, the model determines the
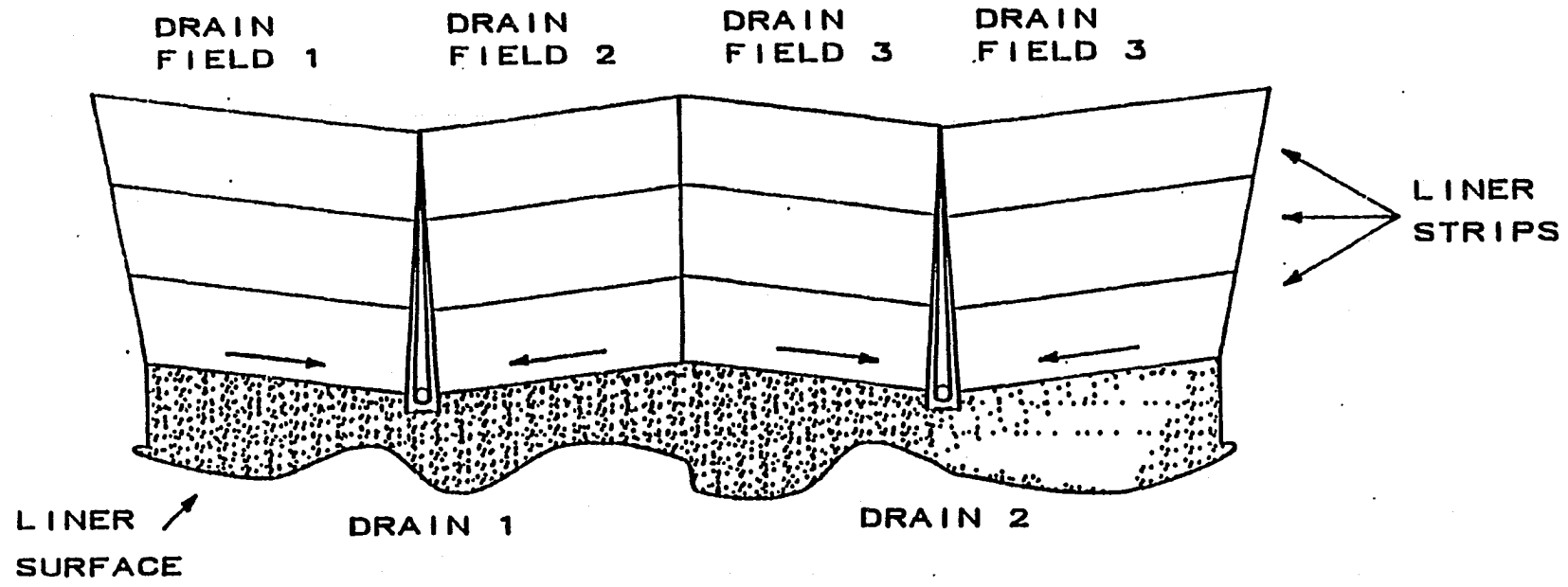
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Figure 18. The division of each liner field into lateral strips which run perpendicular to the drain lines. Each drain line can serve two drain fields.

landfill waste cell which drains to each node on the liner. Hence, the source term, R, for each liner node is the leachate percolating out the bottom of the landfill above it (Figure 19). The one-dimensional solution is solved for each liner strip at each each time step. Total flow from each drain line is determined by summing up the flow from each liner strip leading into the respective drain.

The initial conditions will be written as $h(x,0) = h_o(x)$, the depth of the head over the liner at t=0. This initial saturation level will usually be zero assuming the drainage layer is fully drained. The output from the liner will be the leachate flows as well as the leachate head above the liner surface. If desired, the liner model can be bypassed from the main program.

3.5 LEACHATE CONTAMINANT MODEL

The method to predict leachate contaminant composition in the model uses contaminant production curves to generate source terms for the contaminants. The sources are coupled with a contaminant transport model. Estimations of the waste types and loading data for each element in the column are made. The moisture loading (or percolation) for each element in the column is determined by the water model for each time step. The moisture loading is the amount of leachate (liters) per mass (kilograms) of dry refuse in the element. Using the moisture loading and the leachate contamination curves for various contaminants, the model generates a source term for each contaminant.

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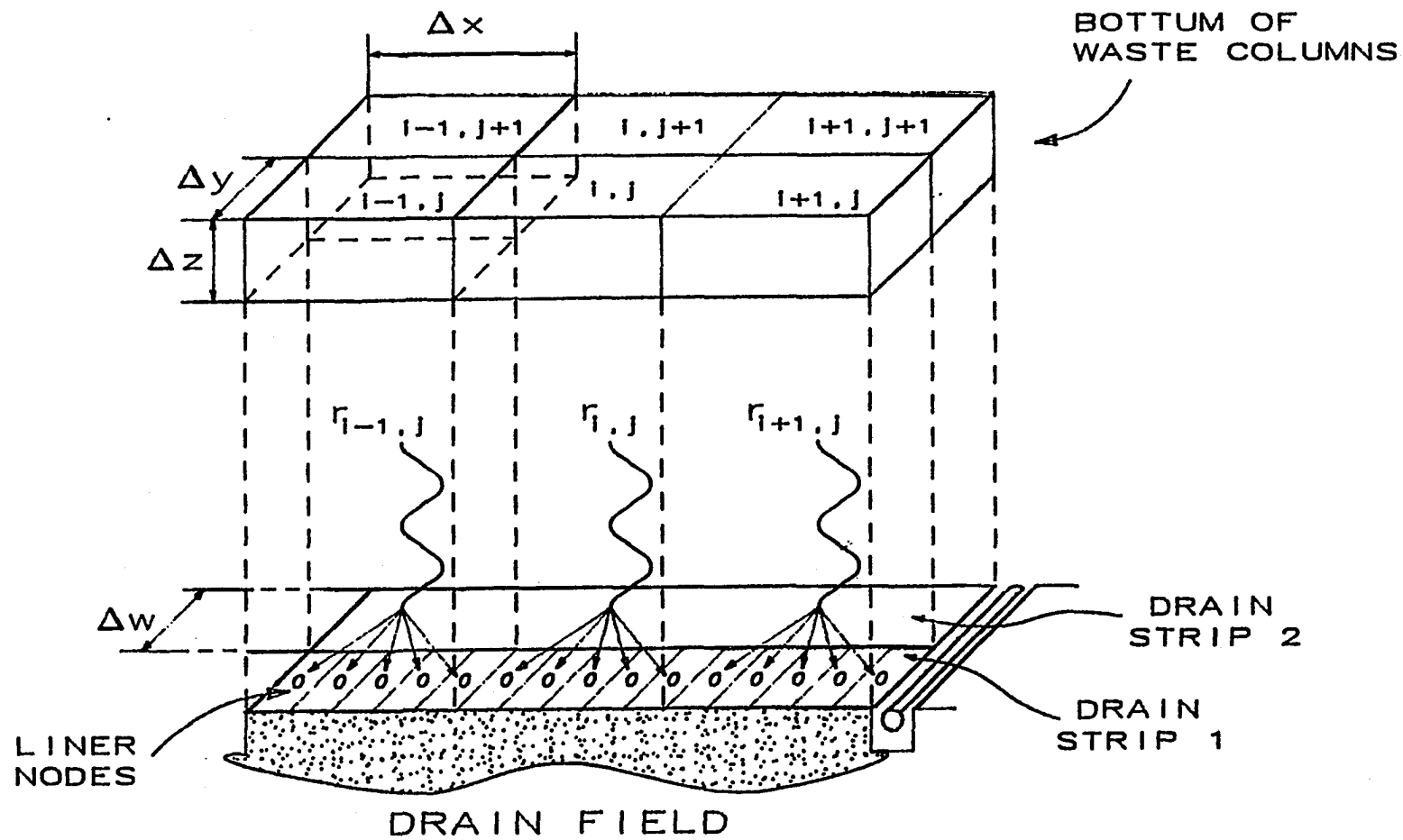
Figure 19.   Leachate from waste cells percolating down into different nodes
in the liner strip.

The contaminant transport is assumed to occur dominantly in the vertical direction. Therefore, the model is based on one-dimensional transport equations. However, it is recognized that lateral transport can occur at least due to diffusion and dispersion. The lateral transport is accounted for by inclusion of a source term in the vertical transport equation. The evaluation of the source term requires a three-dimensional formulation and it is discussed in a separate section. This approach is fully compatible with the formation used in the water model. The governing equation for contaminant transport in the vertical direction is:

$$\frac{\partial(\phi S_w C)}{\partial t} = \frac{\partial}{\partial t}\left(D \frac{\partial C}{\partial z}\right) - \frac{\partial(v_z S_w C)}{\partial z} + Rx \tag{77}$$

where C = contaminant concentration, mass per volume

z = vertical length

S_w = moisture saturation of the element

D = vertical diffusion coefficient

ϕ = the porosity of the waste layers

v_z = vertical velocity (advection) of leachate

t = time

Rx = reaction source terms $\left\{\frac{M}{L^3 T}\right\}$

The reaction source terms account for contaminant production, contaminants in rain water infiltration, contaminants in recirculated leachate, contaminants in lateral seepage, and contaminant reactions. Development of each source term will be shown later.

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Equation (77) is discretized similar to the water flow model with forward difference operators and v_z is replaced by v to give the following:

$$(S_{w_j} C_j)^{n+1} - (S_{w_j} C_j)^n = \frac{\Delta t}{\phi} \left\{ D_{(j+1/2)} \left(\frac{C_{j+1} - C_j}{\Delta z^2} \right)^{n+1} + D_{(j-1/2)} \left(\frac{C_{j-1} - C_j}{\Delta z^2} \right)^{n+1} \right\}$$

$$- \frac{\Delta t}{\phi} \left\{ \frac{v_{(j+1/2)} S_{w_{(j+1/2)}} C_{(j+1/2)} - v_{(j-1/2)} S_{w_{(j-1/2)}} C_{(j-1/2)}}{\Delta z} \right\} + \frac{Rx \, \Delta t}{\phi} \quad (78)$$

Similarly to the water model, the (j+1/2) operator is the average of the property at the j+1 element and the j element.

3.5.1 Numerical Solution of the Contaminant Transport Model

Similar to the water flow model, the Generalized Newton Method is used and the function of the discretized governing equation is as follows:

$$f_j = (S_{w_j} C_j)^n - (S_{w_j} C_j)^{n+1} + \frac{\Delta t}{\phi \Delta z^2} \left\{ D_{(j+1/2)} \left(C_{j+1} - C_j \right)^{n+1} + D_{(j-1/2)} \left(C_{j-1} - C_j \right)^{n+1} \right\}$$

$$- \frac{\Delta t}{\phi \Delta z} \left\{ v_{(j+1/2)} S_{w_{(j+1/2)}} C_{(j+1/2)} - v_{(j-1/2)} S_{w_{(j-1/2)}} C_{(j-1/2)} \right\} + \frac{Rx \, \Delta t}{\phi} \quad (79)$$

Setting DD_j equal to the negative of the function, f_j, and taking the derivatives of the function yields the following coefficients for the Thomas Algorithm:

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$$AA_j = \frac{\partial f_j}{\partial C_{j-1}} = \frac{\Delta t}{\phi \Delta z^2} D_{(j-1/2)} + \frac{\Delta t}{2\phi \Delta z} V_{(j-1/2)} S_{w_{(j-1/2)}} \qquad (80a)$$

$$BB_j = \frac{\partial f_j}{\partial C_j} = - S_{w_j}^{n+1} - \frac{\Delta t}{\phi \Delta z^2}\left(D_{(j+1/2)} + D_{(j-1/2)}\right)$$

$$- \frac{\Delta t}{2\phi \Delta z}\left(V_{(j+1/2)} S_{w_{(j+1/2)}} - V_{(j-1/2)} S_{w_{(j-1/2)}}\right) \qquad (80b)$$

$$CC_j = \frac{\partial f_j}{\partial C_{j+1}} = \frac{\Delta t}{\phi \Delta z^2} D_{(j+1/2)} - \frac{\Delta t}{2\phi \Delta z} V_{(j+1/2)} S_{w_{(j+1/2)}} \qquad (80c)$$

$$DD_j = - f_j = (S_{w_j} C_j)^{n+1} - (S_{w_j} C_j)^n - \frac{\Delta t}{\phi \Delta z^2}\left\{ D_{(j+1/2)}\left(C_{j+1} - C_j\right)^{n+1} + D_{(j-1/2)}\left(C_{j-1} - C_j\right)^{n+1}\right\}$$

$$+ \frac{\Delta t}{\phi \Delta z}\left\{ V_{(j+1/2)} S_{w_{(j+1/2)}} C_{(j+1/2)} - V_{(j-1/2)} S_{w_{(j-1/2)}} C_{(j-1/2)}\right\} - \frac{Rx \, \Delta t}{\phi} \qquad (80d)$$

The source term Rx will be discussed in Section 3.5.5.

3.5.2 Boundary Conditions for Contaminant Transport

Two sets of boundary conditions and one set of initial conditions are required to solve the equations for the contaminant concentrations. The upper boundary has an influx of contaminants due to contaminants in the rainfall. Landfills which recirculate the leachate will also exhibit a contaminant influx into the top boundary. However, there is no diffusion into the top boundary. Losses out the bottom of the top cell include diffusion and advection. Hence, the contaminant mass balance equation for the upper boundary is:

$$\frac{\phi}{\Delta t}\left\{(S_{w_j} C_j)^{n+1} - (S_{w_j} C_j)^n\right\} = D_{(j+1/2)}\left(\frac{C_{j+1} - C_j}{\Delta z^2}\right)^{n+1} +$$

$$\left(\frac{V_{(j-1/2)} S_{w_j} C_{rain} - V_{(j+1/2)} S_{w_{(j+1/2)}} C_{(j+1/2)}}{\Delta z}\right) + Rx \qquad (81)$$

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where C_{rain} is the contaminant concentration of the rain infiltration and $v_{(j-1/2)}$ is the rate of infiltration. The function is differentiated with respect to "C" to yield the upper boundary coefficients for j=1:

$$AA_j = 0 \tag{82a}$$

$$BB_j = - S_{w_j}^{n+1} - \frac{\Delta t}{\phi \Delta z^2} D_{(j+1/2)} - \frac{\Delta t}{2\phi \Delta z} v_{(j+1/2)} S_{w_{(j+1/2)}} \tag{82b}$$

$$CC_j = \frac{\Delta t}{\phi \Delta z^2} D_{(j+1/2)} - \frac{\Delta t}{2\phi \Delta z} v_{(j+1/2)} S_{w_{(j+1/2)}} \tag{82c}$$

$$DD_j = (S_{w_j} C_j)^{n+1} - (S_{w_j}C_j)^n - \frac{\Delta t}{\phi \Delta z^2} D_{(j+1/2)} \left(C_{j+1} - C_j \right)^{n+1}$$

$$+ \frac{\Delta t}{\phi \Delta z} \left(v_{(j+1/2)} S_{w_{(j+1/2)}} C_{(j+1/2)} - v_{(j-1/2)} S_{w_{(j-1/2)}} C_{rain} \right) - \frac{Rx \cdot \Delta t}{\phi} \tag{82d}$$

The boundary condition for the bottom element assumes that the contaminant concentration is the same as the element above it. The thickness of the bottom element is one half the Δz thickness of the other elements in the column. This is done to put the node point (center of the element) on the boundary. The mass balance equation for the bottom element is written as follows for j=N:

$$\frac{\phi \Delta z}{2 \Delta t} \left\{ (S_{w_j} C_j)^{n+1} - (S_{w_j}C_j)^n \right\} = D_{(j-1/2)} \left(\frac{C_{j-1} - C_j}{\Delta z^2} \right)^{n+1}$$

$$- \left(v_{(j+1/2)} S_{w_j} C_j - v_{(j-1/2)} S_{w_{(j-1/2)}} C_{(j-1/2)} \right) + \frac{Rx \Delta z}{2} \tag{83}$$

Setting C_N equal to C_{N-1}, the Thomas algorithm coefficients for the bottom element, that is for j=N, become:

$$AA_j = \frac{\Delta t}{\phi \Delta z} v_{(j-1/2)} S_{w_{(j-1/2)}} \tag{84a}$$

$$BB_j = -S_{w_j}^{n+1} - \frac{2\Delta t}{\phi \Delta z} v_{(j+1/2)}S_{w_j} + \frac{\Delta t}{\phi \Delta z} v_{(j-1/2)} S_{w_{(j-1/2)}} \tag{84b}$$

$$CC_j = 0 \tag{84c}$$

$$DD_j = (S_{w_j} C_j)^{n+1} - (S_{w_j} C_j)^n$$

$$+ \frac{2\Delta t}{\phi \, \Delta z} \left(v_{(j+1/2)} S_{w_j}C_j - v_{(j-1/2)} S_{w_{(j-1/2)}}C_{(j-1/2)} \right) - \frac{Rx \, \Delta t}{\phi} \tag{84d}$$

3.5.3 Contaminant Dispersion-Diffusion Coefficient

The dispersion-diffusion coefficient, D, used in the equations was presented by Bresler (1973) describing the factors involved in determining the term. As pointed out in Section 2.3.2, the means of contaminant transport are usually divided into two parts; molecular diffusion and convective transport. The coefficient for diffusion varies depending on the contaminant, the porous medium and the moisture content. In the mathematical model the coefficient of diffusion is defined as:

$$D_S = D_0 a e^{b\phi S_w} \tag{85}$$

where D_S = diffusion coefficient in the landfill

ϕ = porosity of the waste

S_w = degree of moisture saturation

a,b = empirical constants characterizing the waste material

D_0 = equivalent diffusion coefficient in a free water system

It should be noted that $D_S < D_0$ (Bresler 1973).

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The convective transport coefficient, D_h, due to hydraulic dispersion is a function of velocity as defined by the following relationship:

$$D_h = \lambda \ |V| \qquad (86)$$

where V is the average interstitial flow velocity in centimeters per second and λ is an empirical constant depending on the porous medium (Bresler 1973). Combining the diffusion coefficient with the dispersion coefficient gives a combined diffusion-dispersion coefficient, D_z:

$$D_z = D_h + D_s \qquad (87)$$

The average interstitial velocity is found by averaging the interstitial velocities across the upper and lower surfaces of each element.

3.5.4 Lateral Contaminant Transport

Lateral transport is modeled to occur due to convection and diffusion similar to the vertical transport. The interstitial velocities used for convection are developed in the moisture flow portion of the model. These velocities are used to develop the lateral dispersion coefficients, as discussed in Section 3.5.3., and for convection of the contaminant in the lateral direction. A lateral diffusion coefficient was also developed as in Section 3.5.3. These coefficients were used and Equation (77) was solved for

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transport in the lateral direction. Sign convention was set up in which flows into each element were assumed to be positive. It is possible for each element to have up to four lateral inputs (or losses) depending on the geometry and lateral boundaries.

At each time step, the model determines the flows throughout the landfill. Using the flows, the vertical contaminant transport routines are solved. The contaminant concentrations and flows for each new time step are used to solve the lateral contaminant transport. Solution results of the lateral transport equations consist of a mass flow of the contaminants into (or out of depending on the signs) each element for that time step. Each gain (or loss) of contaminant mass is converted to a contaminant source term (Rx) for the vertical contaminant transport equation for use in the next time step.

3.5.5. Contaminant Source Terms.

The contaminant source term, Rx, in the vertical transport equations accounts for sources and losses of contaminant mass. Possible sources and losses include: contaminant production, contaminants in the rain water, contaminants in leachate which is recirculated, lateral seepage, and contaminants due to reactions. The handling of these different sources is discussed herein.

Contaminant production from landfill waste is the major input to the transport equation. Experiments have reported results of

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contaminants leached per mass of waste as a function of the amount of leachate (volume) which has leached through the waste (McGinley and Kmet 1984; Fungaroli and Steiner 1979a,b; Wigh and Brunner 1981). An example of these type of results for chlorides leached is shown in Figure 20. It is seen that the maximum yield of chlorides is approximately 2.5 grams per kilogram of dry waste. This yield occurs after approximately 3 liters of leachate have passed through the kilogram of waste. The model monitors the amount of leachate which has passed through each kilogram of waste in the element. Using the leachate volumes and inputted contaminant production functions, the model determines the contaminant production in each element for each time step. An equation has been developed to predict a source input to the transport, given the amount previously produced and the amount of water leaching through per time step. The equation can be written:

$$P_c = ULT(1-10^{-CV}) \qquad (88)$$

where P_c = total amount of contaminant previously produced in the element

ULT = total contaminant which will be produced per mass of dry waste

C = production rate coefficient

V = total volume of leachate which has passed through waste

The value for P_c is the amount of contaminants produced up to the current time step. The source term will be P_{contam} at the current
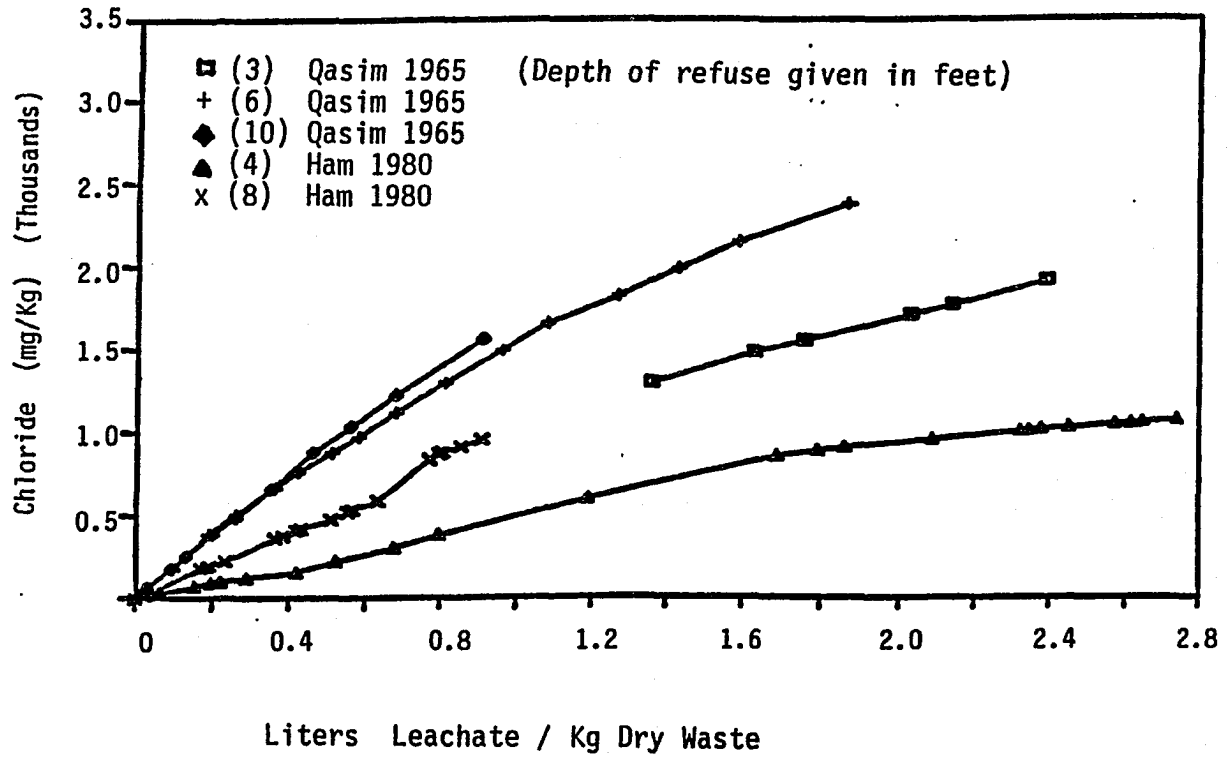
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Figure 20. Mass release of chloride from Municipal Solid Waste (After McGinley and Kmet 1984).

time step minus the P_{contam} at the previous time step. It is seen that initially, when a small amount of leachate has passed through the waste, the term 10^{-CV} is nearly 1 and the amount produced is very small or zero. As the volume of leachate increases, the term approaches zero, and the total production approaches the ultimate load allowed. The rate coefficient will determine how much leachate will pull all the contaminants out of the waste. The production is input as a mass source for the element in the next time step.

Contaminant input due to rainfall is a source for the upper boundary of the vertical cell. The concentration of the contaminant in rainwater is input into the model. Note, this gives the model some versatility to be used as a predictor of the results of a large spill of a contaminant onto the ground. Using the rate of infiltration and the concentration, a mass source for the upper boundary is determined. This mass inflow is put in as a specified flux boundary condition in Equation (81). The term C_{rain} specifies the concentration of the contaminant in the rain water. If the landfill recirculates it's leachate, the concentration of the contaminant in the leachate is determined at the landfill bottom. This concentration multiplied by the leachate flowrate gives the mass input into the top boundary. This is put into the upper boundary element as a source term.

Lateral transport inputs contaminant mass into adjacent elements. The lateral contaminant mass transport for each element is determined as described in Section 3.5.4. At the completion of time step, lateral transport due to concentration gradients and flow in

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the lateral direction is computed. A mass input to each element is determined and is accounted for in the next time step as a source term.

Reactions can occur which will either be a loss or a gain for a contaminant. These would depend on the presence of other contaminants in the leachate and waste. Inputting the reaction rates and accounting for the concentrations of other contaminants in the element will produce a gain or loss of contaminant mass. This is also input as a source term for each element for the next time step.

3.6 GAS PRODUCTION MODEL

Major factors for gas production in a landfill are moisture content and temperature of the cells. The flow portion of the model solves for the moisture content at each node. Gas generation produces heat used as a source for a heat diffusion model to predict temperatures. The gas production model uses the moisture content in each element, coupled with temperature and gas amounts previously produced, to estimate gas production. The production rate equation modeled is Equation (36) from Section 2.5.3. The equation uses reaction rates, gas previously produced, and the amount of gas left to produce to determine a gas production rate. A factor is introduced to adjust the production rate for variations in moisture content.

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3.6.1 Gas Production Factors

Similar to contaminant production, the production of gas is limited by the amount of waste available. For a given mass of waste, there is an ultimate amount of gas which can be produced. The rate of the production to reach the ultimate amount depends upon the temperature and moisture content available. There are also some conditions where no gas is produced when moisture content and temperature parameters are outside an optimum range. Section 2.5.2. discussed the effects of moisture content on the gas production rate. DeWalle *et al.* (1978) showed that the gas production varied from 2.1 ml/Kg·d at 35% moisture content up to 15.4 ml/Kg·d at 100% moisture content. They also reported that gas production drops off sharply below 30% and ceases below 20% moisture content.

Using the moisture content determined by the model for each element, an empirical formula to adjust gas production as a function of moisture content was developed. A plot of gas production as a function of moisture content is presented in Figure 21. The linear slope of the line is found to be $\frac{0.205 \ ml/Kg \cdot d}{\% \ moisture \ content}$. Temperature variation data (Hartz 1980; Hartz *et. al.* 1982) was obtained at 45% moisture content. A factor, F_{mc}, was developed to adjust the gas production rate of a node at a moisture content other than 45%. The factor is defined by:

$$F_{mc} = 1 + \left(\frac{S_w - 45\%}{4.15 \ ml/Kg \cdot d} \right) m \qquad (89)$$

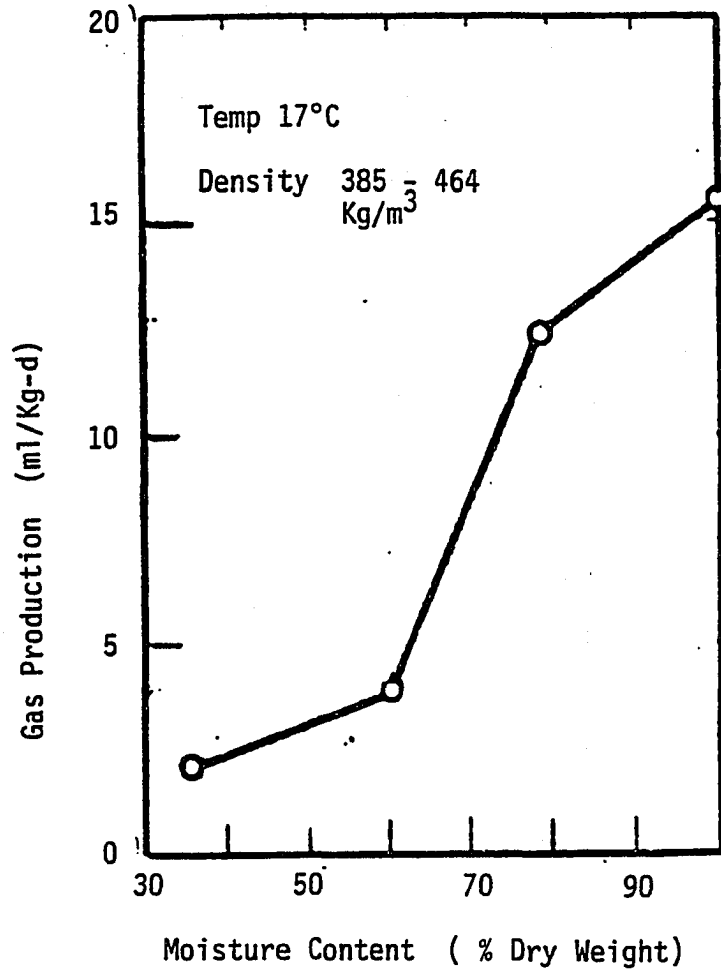where S_w = moisture content of the cell

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Figure 21. Gas production as a function of moisture content
(After DeWalle *et at.* 1978)

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m = slope of gas production as a function of moisture content

4.15 = gas production at 45% moisture content

Substituting the slope found in Figure 21 gives:

$$F_{mc} = -1.223 + 4.9398 \, S_w \qquad (90)$$

This gives a dimensionless factor to account for gas production rate variations due to changes in moisture content.

Hartz (1980) listed seven different mathematical models to predict the methane gas production. Two of them, namely Equations (35) and (36), were used in the gas portion of this model and coupled to the moisture content factor, F_{mc}. Incorporating the moisture content factor into Equation (35) gives:

$$\frac{dP}{dt} = - F_{mc} \, A \, P^{-k} \qquad (91)$$

where F_{mc} = moisture content factor

A = production rate parameter

P = gas previously produced

k = reaction rate coefficient based on temperature

$\frac{dP}{dt}$ = the gas production rate at time t

Alternatively, using Equation (36):

$$\frac{dP}{dt} = - F_{mc} \, A \, (Q - P)^k \qquad (92)$$

where Q = ultimate gas production per mass of waste.

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The reaction rate coefficient is adjusted for temperature variations in the landfill cell. The model used Equation (29) in Section 2.5.2. to adjust the reaction rate coefficient as follows:

$$\ln k_2 = \ln k_1 + \frac{E_a(T_2 - T_1)}{R(T_2\ T_1)} \tag{93}$$

where E_a = the activation energy

T_2 = temperature at which the new rate is being determined

T_1 = reference temperature

k_2 = gas production rate for T_2

k_1 = gas production rate at the reference temperature

R = gas constant

Temperatures of the different elements need to be determined to adjust the rate coefficient in Equation (93).

3.6.2 Temperature Prediction in the Landfill

The heat diffusion equation is modeled to predict temperatures in the landfill. Equation (30) in Section 2.5.2.1. gave the three dimensional heat diffusion equation. One-Dimensional heat flow is assumed for the model and k_i is also assumed constant. The specific heat is broken into two parts; specific heat of water and specific heat of waste. The one-dimensional heat formulation was discussed in Section 2.5.2.1. as Equation (31) and is repeated here as:

$$\frac{\partial}{\partial z}\left(k_z \frac{\partial T}{\partial z}\right) + \dot{q} = \rho_{waste}\, C_{waste}\, \frac{\partial T}{\partial t} + \rho_w\, C_w\, \phi\, \frac{\partial S_w T}{\partial t} \tag{94}$$

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where ϕ = porosity of the waste

S_w = degree of moisture content

k_z = thermal diffusion coefficient in the vertical direction

ρ_{waste} = density of the waste

ρ_w = density of the water

C_{waste} = specific heat of the waste

C_w = specific heat of the water (Akan 1984)

This equation accounts for the heating of the water and the waste. The discretized form of Equation (93) is written:

$$K_z \left(\frac{T_{j+1} - T_j}{\Delta z^2} \right)^{n+1} + K_z \left(\frac{T_{j-1} - T_j}{\Delta z^2} \right)^{n+1} + \dot{q} = \frac{\rho_{waste} C_{waste}}{\Delta t} \left(T_j^{n+1} - T_j^{n} \right)$$

$$+ \frac{\phi \, \rho_w \, C_w}{\Delta t} \left(S_{w_j} T_{j-1}^{n+1} - S_{w_j} T_j^{n} \right) \tag{95}$$

The units of the terms are $\frac{J}{m^3 \, hr}$ and the $\dot{q}$ is the heat source. The heat source is the energy generation by the waste when gas is produced. To solve this equation, methods similar to those employed for water flow (Section 3.2.1) and contaminant transport (Section 3.5.1) are used. A function, f_j, is generated and the four coefficients for each element are as follows:

$$AA_j = \frac{\partial f_j}{\partial T_{j-1}} = \frac{\Delta t \, K_z}{\Delta z^2} \tag{96a}$$

$$BB_j = \frac{\partial f_j}{\partial T_j} = - \, \rho_{waste} \, C_{waste} - \rho_w \, C_w \, \phi \, S_w - \frac{2 \Delta t \, K_z}{\Delta z^2} \tag{96b}$$

$$CC_j = \frac{\partial f_j}{\partial T_{j+1}} = \frac{\Delta t \, K_z}{\Delta z^2} \tag{96c}$$

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$$DD_j = -f_j = \rho_{waste} C_{waste} \left(T_j^{n+1} - T_j^n \right) + \phi \, \rho_w C_w \left(S_{w_j} T_j^{n+1} - S_{w_j} T_j^n \right)$$

$$- \frac{\Delta t \, K_z}{\Delta z^2} \left(T_{j+1} - T_j^{n+1} \right) - \frac{\Delta t \, K_z}{\Delta z^2} \left(T_{j-1}^{n+1} - T_j^n \right) - \dot{q} \, \Delta t \quad (96d)$$

To solve the heat equation, the top boundary exhibits a specified temperature condition. Since the top will be cover material, there will be no heat generation in the upper boundary element. The specified temperature coefficients are for j=1:

$$AA_j = 0 \quad (97a)$$

$$BB_j = 1 \quad (97b)$$

$$CC_j = 0 \quad (97c)$$

$$DD_j = T_j - T_{atm} \quad (97d)$$

where T_{atm} is the average monthly air temperature inputted for the evapotranspiration routine.

The bottom boundary condition is treated the same way the contaminant transport boundary was handled. The temperature in the bottom element was set equal to the element above it. The flow of energy was also adjusted for the bottom element thickness being one-half the thickness of the other elements in order for the node to be on the boundary. These assumptions and conditions yield the following coefficients for the bottom element, j=N:

$$AA_j = 0 \quad (98a)$$

$$BB_j = - \rho_{waste} C_{waste} - \rho_w C_w \phi S_{w_j} \quad (98b)$$

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$$CC_j = 0 \tag{98c}$$

$$DD_j = \rho_{\text{waste}} C_{\text{waste}} \left(T_j^{n+1} - T_j^n\right) + \phi\rho_w C_w\left(Sw_j T_j^{n+1} - Sw_j T_j^n\right) - \dot{q}\,\Delta t \tag{98d}$$

The parameters needed to solve the system of equations are read via an input file for the model.

3.6.3 Cell Gas Production

The gas production rate discussed in Section 3.6.1 for each element is determined daily. Since the rate of gas production is very stable and time scales are on the order of months (Young 1989), it was not necessary to update the rate as often as water and contaminants. The model inputs include the ultimate amount of gas per mass of waste, reaction rate, reference temperature, and corresponding fitting parameter. The model tracks the amount of gas which has been previously produced using the following equation to determine gas produced, ΔG, in the current time step:

$$\Delta G = \frac{dP}{dt}\,\Delta t \tag{99}$$

where $\frac{dP}{dt}$ = gas production rate

Δt = time period for the gas production (day)

The gas produced in the time step is added to the gas previously produced, P, for use in Equation (91) or (92). As the amount of gas previously produced approaches the ultimate gas production capability the amount of gas drops off regardless of the gas production rates. This will model the initial jump in gas production followed by a gradual decrease in the rate for many years (Young 1989).

3.7 MODEL SETUP SUMMARY

The model is set up to read two input files. The first is the model input data file. All the parameters to run the model are specified in the input data file. The model initially asks the user the name of the input data file. This allows the user to prepare several input files for use by the model. The second file is the precipitation file. This is prepared to specify the rainfall events or the daily or monthly rainfall if the model is to be run for long periods of time. The specified time step for the model is on the order of hours. The model has the ability to reduce the time step in the case of gradients which are steeper than the specified time step. Using the precipitation and runoff routine, the model develops the flow portion for the current time step. The flow portion gives flow rates, permeabilities, and saturation levels throughout the landfill. These are then used to determine contaminant production and transport as well as gas production. A flow chart outlining the model operation steps is given in Figure 22.
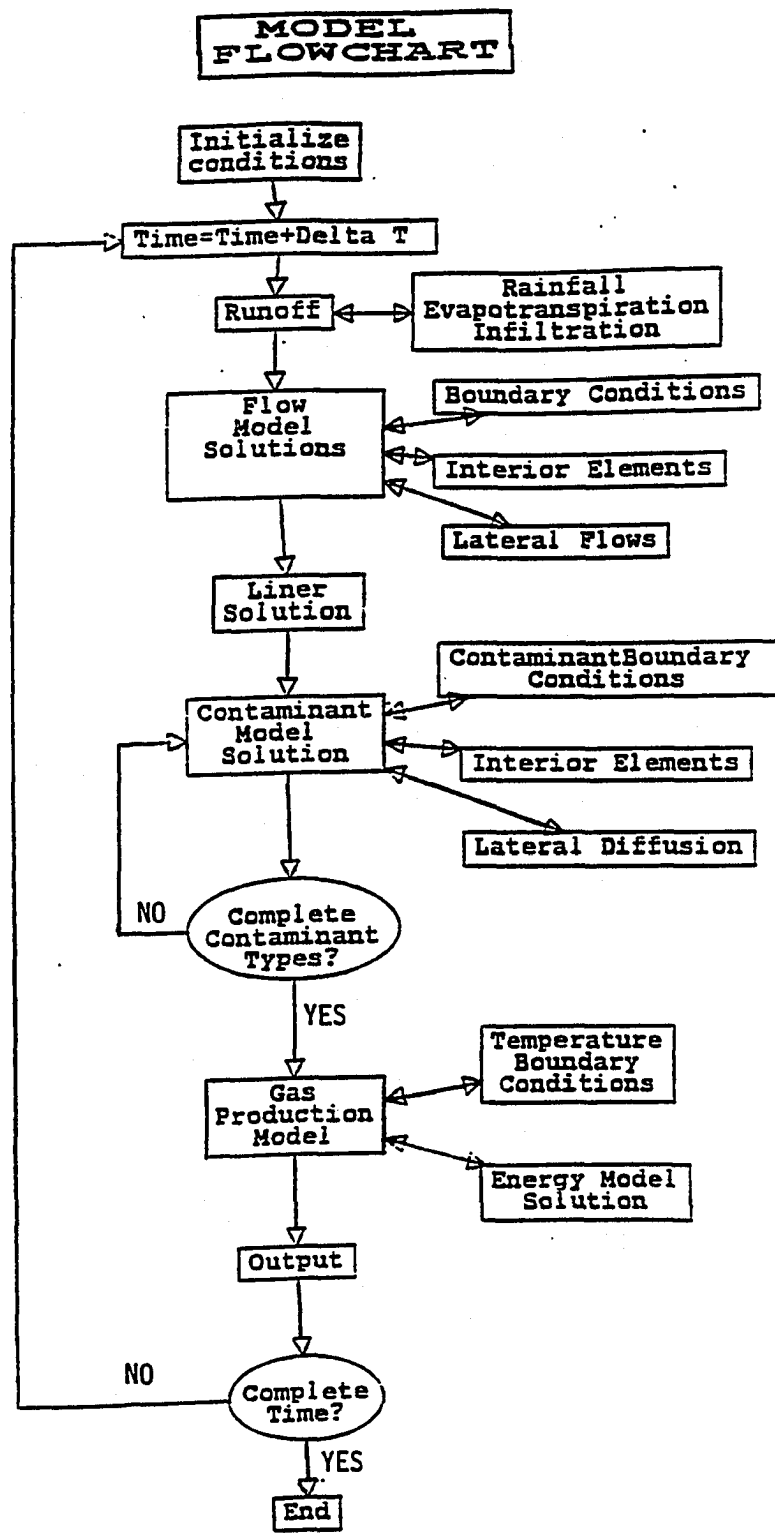
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Figure 22. Model Flowchart.

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4. MODEL VERIFICATION

4.1 PRELIMINARY MODEL TESTS

Prior to comparing the model to various landfills for calibration, several tests were run on the model to ensure it was operating correctly numerically. The model contains numerous routines which have been previously described. Each routine was tested including the variations when present.

The first test for every routine was the no-flow or zero production situations. This was done to ensure the model didn't artificially produce flow sources, contaminant sources, or gas sources, depending on what the routine was modeling. Flow routines were tested to ensure no water sources emerged due to numerical approximations. The different flow routines tested were the vertical water flow, the lateral seepage, the surface runoff, the liner and collection system, the infiltration, and the recirculation. Contaminant routines were checked to ensure artificial sources of contaminants didn't occur. These routines included contaminant transport in the vertical and lateral directions. Likewise, the gas generation was checked to ensure that if there were no sources, then gas was not produced. This check included the heat transport routine

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used to predict the local cell temperature which influences gas production. The no-flow tests were all completed satisfactorily.

The second set of tests were to ensure the model maintained mass conservation. Mass conservation runs were done for the water flow, runoff, liner flow, and contaminant transport routines. The tests checked the water flow, runoff, and liner flows for conservation of water while the contaminant transport tested the conservation of the contaminant. The water flow routine tests verified that the water inputs to the model were equal to the losses due to vertical flow out the landfill bottom, lateral flow across the lateral boundaries, losses due to evapotranspiration, and the change in mass storage in the landfill.

The runoff routine was checked to ensure that the water input through precipitation was accounted for in infiltration, lateral runoff, or changed to storage (ponding) above the landfill cells. The liner tests showed that the input water percolating down onto the liner was either lost to the liner drain, permeated through the liner, or was used as storage on the liner as the phreatic surface level increased. Testing of the contaminant routine ensured that the mass of contaminants produced was equal to the mass which leached out or was stored in the different cells. These checks showed the large concentration increases in the cells as contaminants from the cells above leached down.

Along with the conservation checks, the model was also checked for maintaining symmetry under symmetrical input. This was

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accomplished by similar inputs to identical vertical cells in the model and comparing the results in the different cells. Various schemes were used for inputting water into different cells. The symmetry would not have been maintained if numerical instabilities were present. The first scheme input a water infiltration into the center cell only. The heads developed in the other cells were checked to see a lateral spreading of the water. The heads were also checked to ensure they were identical in the geometrically similar cells. For example, the four cells adjacent to the center cell had identical heads. The four corner cells were expected to have identical heads, but with values less than the center cell or the cells adjacent to the center cell. The second symmetry test was inputting identical flows only into the four corner cells. Again, checks were done to ensure that the geometrically similar cells developed identical heads. A third test inputted identical flows into a row of cells in the landfill and checked results.

All these tests were run until steady state conditions were reached in the landfill. Steady state was reached when the water storage conditions were no longer changing and the flow into the landfill was equal to the flow out the bottom plus the lateral flow out the sides. At that point, the heads of each cell became steady with time. Tests were also done by turning off the lateral seepage out the side of the landfill. In these cases, the steady state heads were higher since all the water flowing into the landfill had to flow out the bottom. These symmetry and steady state tests were done for the different water routines. Similar tests were run for the

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contaminant routine monitoring the contaminant concentrations in the cells instead of the heads. Steady state conditions for the contaminant model did not coincide with the water flow model since the production of the contaminant in the cells is constantly changing. The contaminants model was run until depletion from the landfill.

The preliminary tests were also run which changed the inputs with time. Storm events were simulated to check the runoff routine. Initially, all the precipitation would infiltrate into the top boundary cell. When the cell reached saturation, excess precipitation caused ponding to occur in the runoff routine. The model shifted to a constant head boundary condition using the depth of the water in the runoff routine as the head on the top boundary cell. When the precipitation ended, the constant head condition remained until all the water in the runoff routine cells were lost to infiltration or runoff. At that point, the model switched back to the specified flux condition and the storage in the upper cell would drain into the landfill until field capacity was reached or be lost to evapotranspiration. Tests were run in which the precipitation remained off and it was checked that the landfill would discharge excess water until its field capacity was reached at which point it would also go to a steady state situation. When all the preliminary tests were made, the model was ready to be calibrated using real data.

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4.2 TEST COMPARISONS WITH REAL LANDFILL DATA

Test cases using real landfills have been reported in literature. Ideally, one landfill would be modeled using all the aspects of this model. However, everything reported in literature is concerned with certain aspects of a landfill. For example, tests run for the verification of the HELP model (Peyton and Schroeder 1988) deal only with leachate flow. Hence, various cases have been modeled to check different aspects of the model. The cases modeled include three University of Wisconsin lysimeters (Ham 1980), the Boone County Landfill project (Wigh 1984), and a privately owned landfill in Virginia. A comparison is made with the HELP model for leachate production with one of the University of Wisconsin lysimeters.

4.2.1 University of Wisconsin Lysimeters

The University of Wisconsin built and monitored eight test cells or lysimeters from 1970 to 1977 to evaluate different landfill conditions. The cells were large enough to permit full-scale landfill equipment to be used on the representative waste. Each cell was 30 x 60 ft (9.1 x 18.3 m) in surface area. The first six cells were approximately 4 ft. (1.2 m) deep and loaded with 100 tons (91 metric tons) of waste each. The last two cells were 8 ft. (2.4 m) deep with 215 tons (196 metric tons) loaded in each. A plan view of the test cell facility is shown in Figure 23. Pairs of cells were surrounded by concrete walls and separated by plywood. The bottom liner of each cell sloped 3% to a drain and consisted of graded
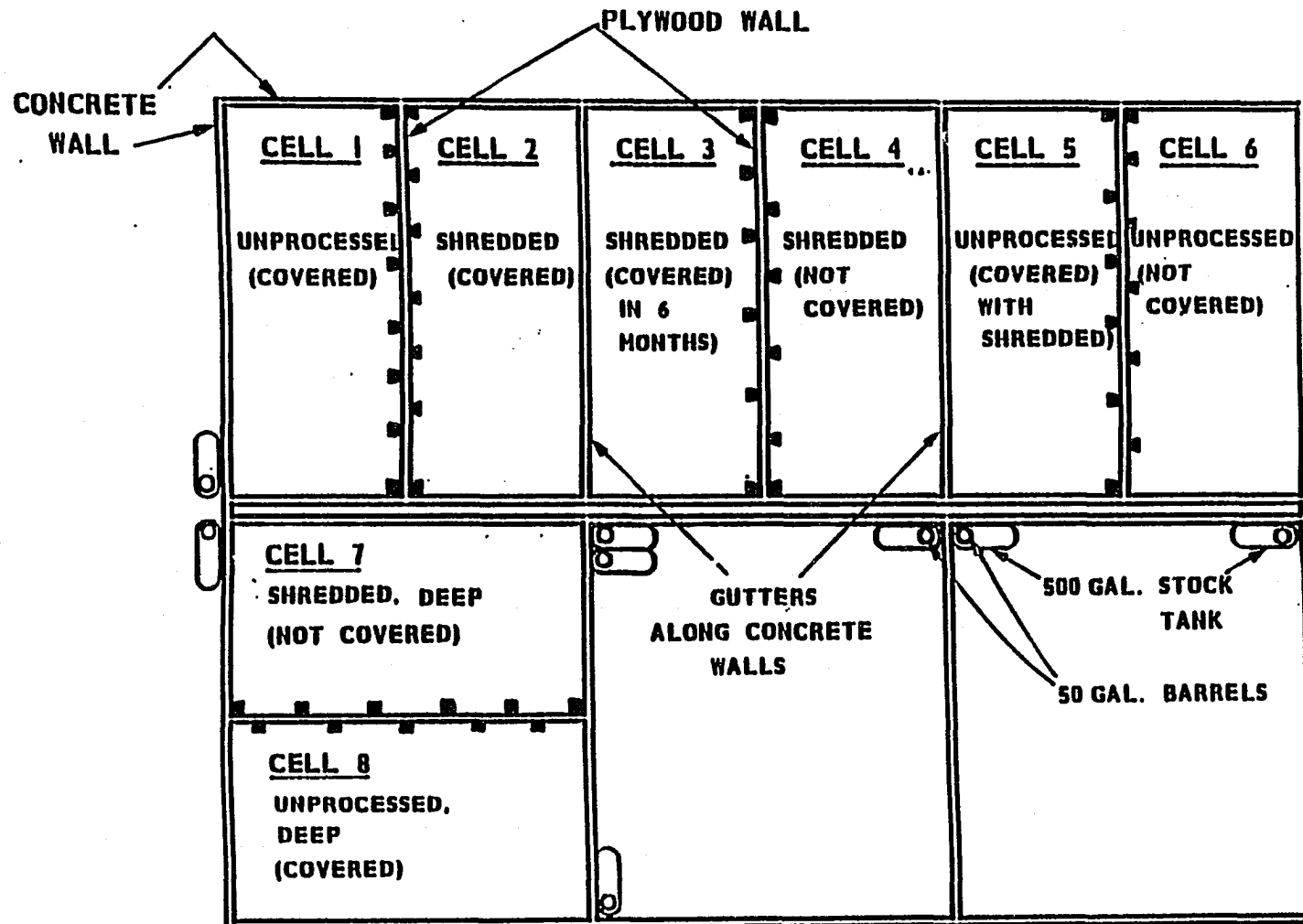
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Figure 23. Plan view of the University of Wisconsin test facility (After Ham 1980).

compacted sand, covered with four inches of crushed stone and one inch of bituminous layer. On top of the bituminous layer was a 6 mil polyethylene sheet, overlaid with four inches of crushed, course granite. The crushed granite layer acted to carry leachate along the 3% slope to the drains. Testing was also conducted to ensure the granite would not affect the leachate quality through chemical reactions (Ham 1980). Two of the four feet deep cells, Cells 1 and 2, and one eight feet deep cell, Cell 8, were modeled. The other cells had special tests being conducted (covering after a certain time, no cover in place, placing different percentages of waste, screening to prevent insects, etc.) and were not modeled.

Cells 1 and 2 were constructed and placed in service in September, 1970, and Cell 8 was constructed in August, 1972. The refuse was received from city collection trucks which were carrying residential and light commercial waste (i.e., an occasional small neighborhood store). The trucks were chosen at random and routed to the cells for disposal. Cells 1, 2, and 8 were each covered immediately with 6 inches of compacted sandy-silt soil. The compacting was done by an experienced operator using the normal time, compaction effort, and layer thickness. Waste placed in Cells 1 and 8 was unprocessed refuse while waste in Cell 2 was shredded prior to being placed (Ham 1980).

Monthly data collection included precipitation, runoff, leachate volumes, and various leachate composition tests. Leachate quality tests consisted of chemical oxygen demand (COD), specific conductance, pH, calcium hardness, total hardness, alkalinity,

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chloride, iron, ammonia, nitrogen, organic nitrogen, total ammonium nitrogen, nitrate, and total and soluble phosphate. The rainfall data is given in the tabular and graphical form. A graph of rainfall data is shown in Figure 24 and the tabular form is given in Appendix B. There are two scales to represent the two different starting dates for Cells 1 and 2 and Cell 8. The runoff and leachate collected volumes for each cell are also given by Ham (1980) and are shown in Appendix B.

4.2.1.1 Model Simulation of University of Wisconsin Water Flow

Input files for the three lysimeters (designated Cell 1, 2, and 8) and a precipitation file for the rainfall during their periods were developed. The time step used was 0.5 hours and each simulation was run for the duration of the data. The simulations of Cells 1 and 2 were run for 68 months (mid-September, 1970 through mid-May, 1976) and the simulation of Cell 8 was run for 58 months (mid-August, 1972, through mid-June, 1977). The height of each node, Δz, in the model was 0.2 meters for Cells 1 and 2 and 0.4 meters for Cell 8. The thickness of the top cover was simulated to be 0.15 meters. There was no lateral leachate seepage out of the sides of the cells due to the concrete walls. The lateral distances, Δx and Δy, of each node were 9.1 meters and 18.3 meters, respectively. The dry density of the waste was estimated to be 445 kg/m^3 (Ham 1980).

Various soil parameters for the cover and refuse had to be determined. A saturated permeability of 7.2 x 10^{-3} m/hr and a
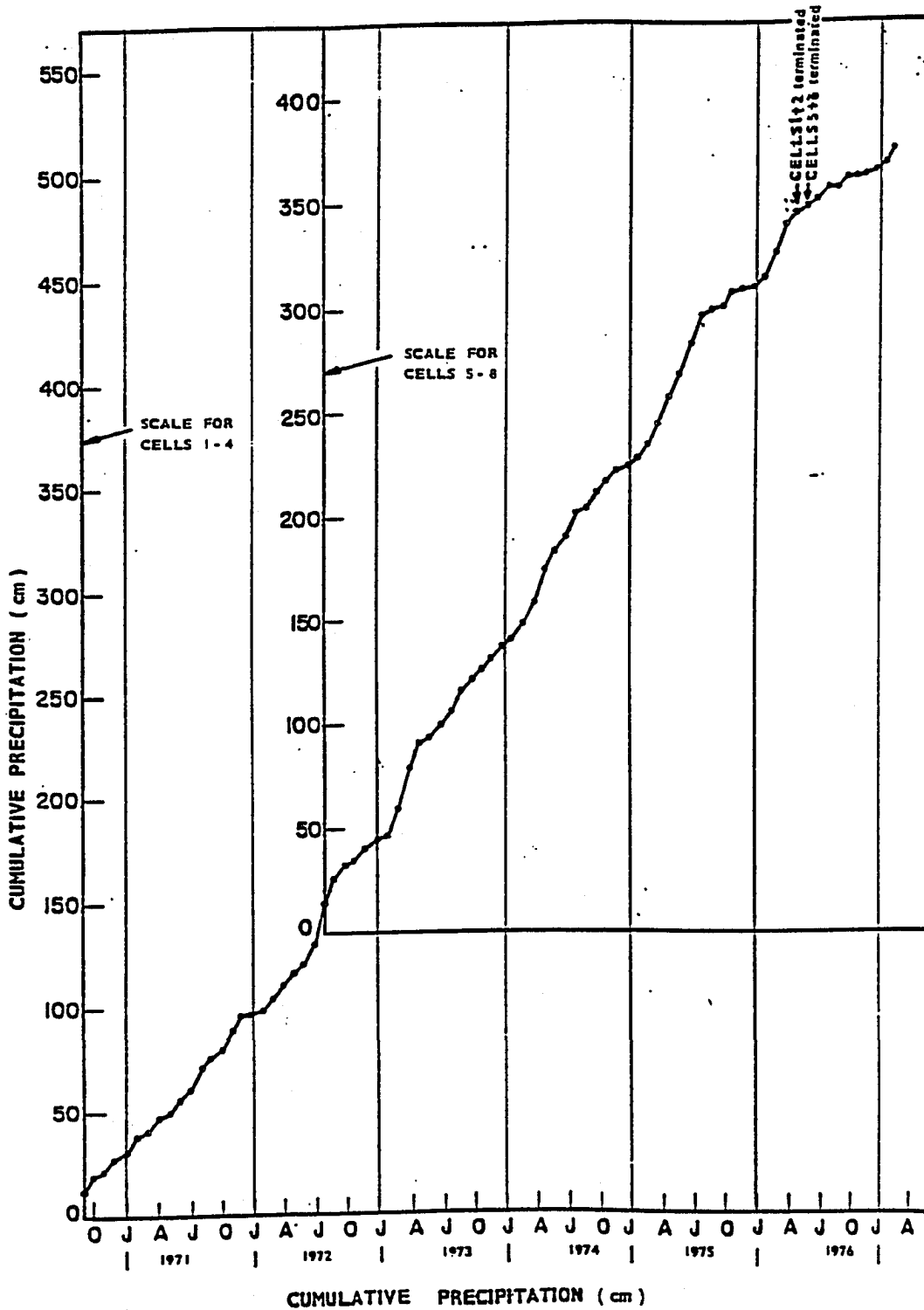
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Figure 24. Rainfall data for University of Wisconsin test cells (After Ham 1980).

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porosity of 0.52 were used for the waste. The saturated permeability and porosity used for the cover was 1.4 x 10^{-2}m/hr and 0.46 respectively (Schroeder *et al.* 1984). Estimations of the coefficients for the soil relative permeability, K_r and saturation, S_w, as used in Section 3.2, were made. Values used for porous medium parameters α and X_m were 5.3 and 2.0 respectively (Russo 1988). The maximum saturation was 1.0 and the minimum saturation used was 0.05. An initial estimation for the moisture content of the waste was 0.20. Using Equations (39) and (40) with the assumed values of α and X_m, the corresponding initial value of K_r was 1.47 x 10^{-5} and that of $\frac{\partial S_w}{\partial p}$ was 0.16. This information was used to select the initial head of each element or node. A separate program was written to help in selecting the initial head, h, to support the moisture content of the waste being loaded. All the assumptions for each simulation were read by the model via an input file. An example of an input file is given in Appendix C.

The model simulation was run for the three different cells. The cumulative model leachate output as a function of time is plotted in Figures 25 through 27 corresponding to Cells 1, 2, and 8 respectively. Also plotted on each graph is the actual cumulative leachate collected from the respective cell during each test. A visual inspection shows that the model values track fairly well with the actual leachate values. One notable point on the graphs is the jump in actual leachate collected between months 32 and 35 for Cells 1 and 2 and between months 8 and 11 for Cell 8. This pronounced jump is much larger than the other noted yearly increases which correspond

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HAM LYSIMETERS CELL 1
CUMULATIVE LEACHATE
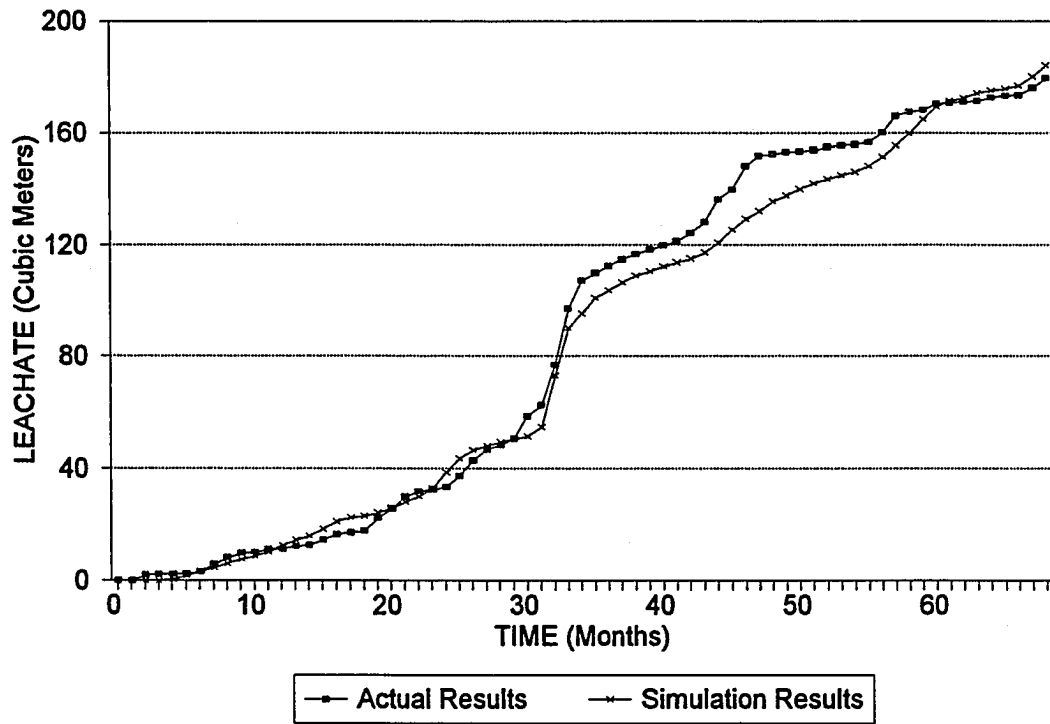

Figure 25. Cumulative leachate in Cell 1.

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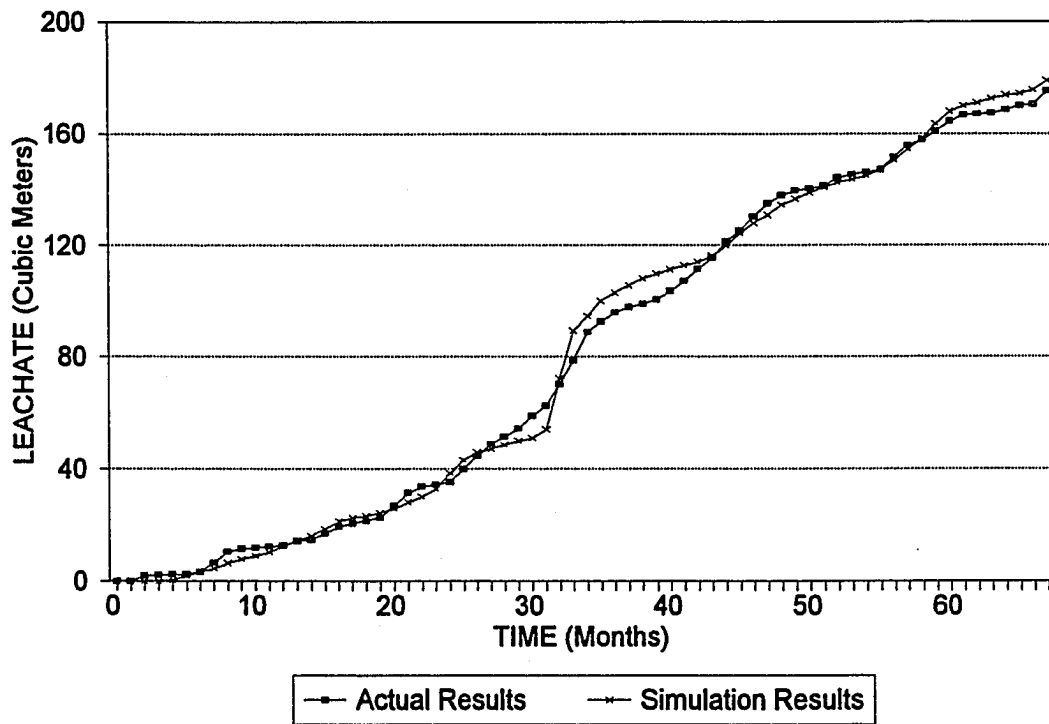# HAM LYSIMETERS CELL 2
CUMULATIVE LEACHATE

Figure 26.   Cumulative leachate in Cell 2.

143
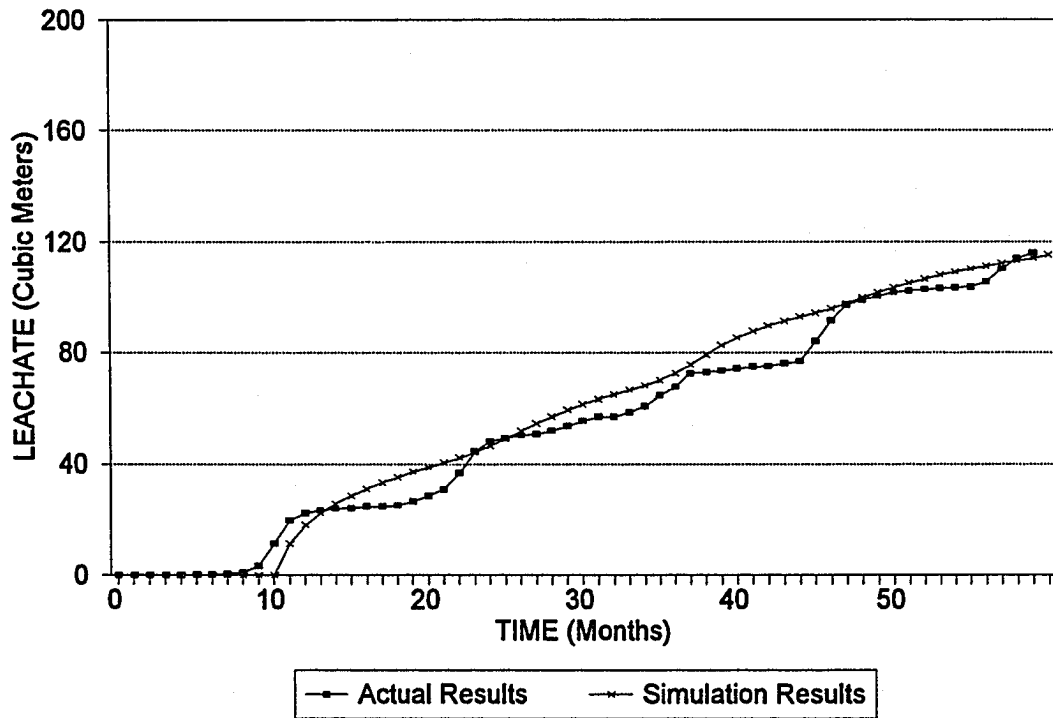
HAM LYSIMETER CELL 8
CUMULATIVE LEACHATE

Figure 27. Cumulative leachate in Cell 8.

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to the rainy season. These readings were recorded for March through May of 1973. There was an exceptionally heavy rainfall during this time period and it destroyed the runoff system. Hence, the runoff was not collected, but channeled back into the cell and taken off the bottom as leachate (Ham 1980). The amount of leachate collected was greater during these three months due to all the runoff being converted to leachate. In these cases, the extremely large amount of water flow through the cells possibly caused channeling through the waste layer. Similarly, a gutter was broken in Cell 2 in February, 1976 (Month 65) and the runoff was routed through the cell into the leachate system. A routine was built into the model for this three month time situation of channeling. This was accomplished by reducing the evapotranspiration during this three month time frame.

It was important to model the effects due to possible channeling due to the reliance on leachate flow by the contaminant production and transport. The production of the contaminant in the cell at a time step is dependent on the amount of leachate which has previously passed through the cell. If the extra leachate passing through the waste is not modeled, the source terms for the contaminant production will not account for the actual leachate passing through the cell. The actual cell received the leachate and leached the contaminants accordingly. Hence, the simulation cell must show the large leachate flow through it for the production.

It was desired to develop a correlation coefficient to measure variations between the actual data and the model simulation results. This variation or comparison is defined in terms of how much the

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predicted simulation values deviate from the actual experimental data. A standard deviation as discussed in Eide *et al.* (1986), is generally determined by squaring each deviation, summing the resulting squares, dividing the sum by the number of points, and taking the square root of the total. Standard deviations are usually done on data points and comparing them to a mean value to determine if the data is dispersed or bunched together. The standard deviation to compare simulation data to actual data is defined as:

$$s = \left(\frac{\sum (y_{model} - y_{actual})^2}{n} \right)^{0.5} \tag{100}$$

where s = the deviation

y_{model} = the simulation data point

y_{actual} = the experimental data point

n = the number of data points

It has been found that for small samples (n < 30) the denominator would be n-1 (Eide *et al.* 1986). All of the samples involved have more than 30 data sample points. The deviations of the simulation leachate curves in Figures 25, 26, and 27, compared to the three Ham lysimeters, are 7.66, 4.02, and 6.63, respectively.

4.2.1.2 Model Simulation of University of Wisconsin Contaminants

An examination of the various contaminants measured by Ham (1980) and other landfill contaminant studies allows for the modeling of several different types of contaminants. Bresler (1973) gives the

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coefficients for dispersion and diffusion, as discussed in Section
3.5.3, for chlorides. Since chlorides are a conservative substance
and the transport coefficients are already available, the model will
simulate chloride transport for the three lysimeters. As discussed
in Section 2, an important contaminant for estimating treatment costs
is Biochemical Oxygen Demand (BOD) and Total Organic Carbon (TOC).
The model can currently handle three contaminants. Data was
available for the Chemical Oxygen Demand (COD), hence a simulation
run was also done for COD in Cell 8.

Bresler (1973) suggests using values of 0.002 and 10.0 for the
coefficients a and b, respectively, a value of 0.04 cm^2/hr for D_o, and
setting lambda at 0.39 cm as defined in Equations (85) and (86), for
chlorides. Coefficient values for COD were not given, however, since
contaminant transport in the vertical direction is mostly due to
convective velocities, dispersion and diffusion variations due to
coefficient differences are negligible. The same coefficients were
used for COD simulations.

Coefficients to describe the production of each contaminant at
each node must be selected. The total amount of a contaminant
released to the leachate per dry mass of waste is defined as the
ultimate load coefficient. After reviewing the experimental data
(McGinley and Kmet 1984; Farquhar 1989; Fungaroli and Steiner
1979a,b; Wigh and Brunner 1981), the ultimate load selected for
chlorides depends on the composition of the waste. Certain wastes
will leach more chlorides than other types. Most of the curves from
the experimental data indicate the ultimate chloride production to be

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approximately 1500 mg/Kg of dry waste. This was not always the case, especially with the Ham Cell 1, and this value depends on the waste composition.

The source production depends on the amount of contaminants previously produced and the rate at which they will be produced. From Equation (88) it is seen that the rate is affected by the amount of water which has leached through the element for the time step. Various rate coefficients were tested and compared to the curves developed by Wigh and Brunner (1981) to determine the best rate coefficient. A rate of 0.45 was selected and Figure 28 shows the theoretical curve compared to the actual curve.

The initial concentration of the contaminant must be specified to satisfy the initial conditions. The water in the waste has an initial concentration of chlorides prior to leaching. The initial leachate contaminant concentrations reported in different experiments varied from 400 mg/l to 1000 mg/l. Most of them were approximately 900 mg/l. The initial concentration used for simulations was 900 mg/l. When an initial concentration is specified the ultimate load must be adjusted to account for the initial mass. Using the initial concentration, the estimated moisture content, and the porosity of the waste, the amount of chloride mass per mass of dry waste to fulfill the initial concentration was estimated. The ultimate load was then adjusted since the waste already gave up this initial mass of chlorides. The estimated chlorides in the rain water was selected to be zero for the cells. There was no recirculation of leachate and that model function was also bypassed.

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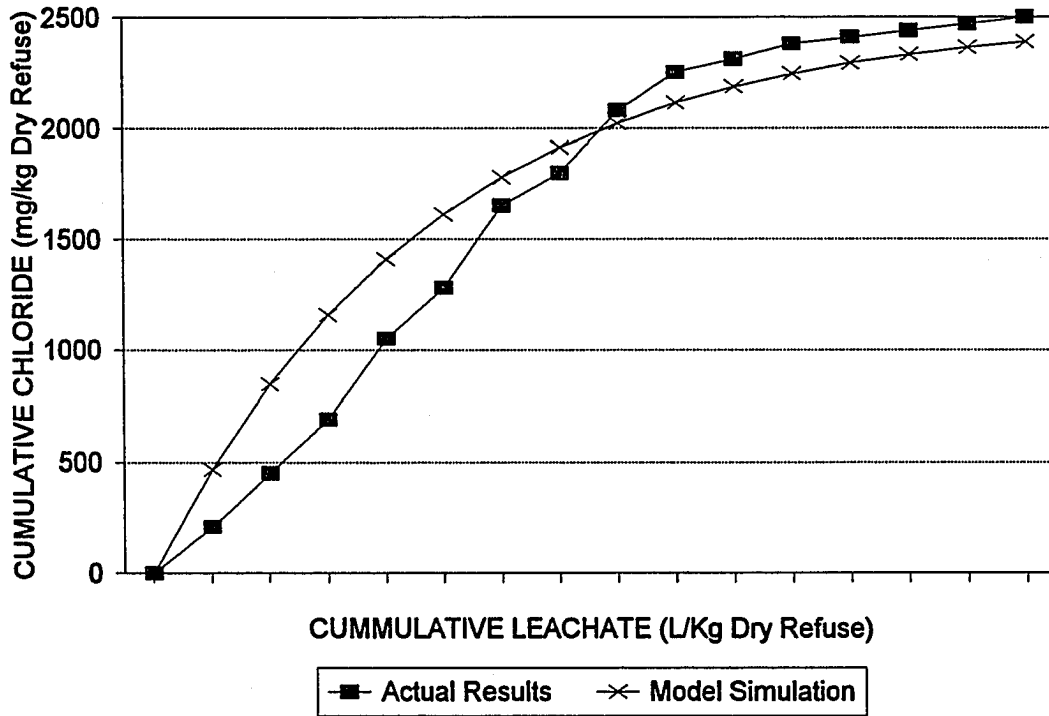# CHLORIDES vs VOLUME
C = 0.45

Figure 28. Theoretical rate coefficient for chloride production

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The model simulation for chlorides was run for the three different cells using the above input values. The cumulative mass of chlorides being leached out of the cells was determined by multiplying the leachate volumetric flow for each time step by the chloride concentration in the bottom element of the landfill. Each time step's cumulative mass was added to the total chloride mass. The cumulative chloride mass leached from the actual cells was determined by multiplying the leachate concentration of chlorides by the amount of leachate produced for the month.

The model simulation and actual data for the cumulative mass of chlorides leached as a function of time are plotted in Figures 29, 30, and 31 for Cells 1, 2, and 8. It is noted that Cell 1 chloride levels are small compared to the other cells. McGinley and Kmet (1984) provide discussion on Cell 1. They point out that Cell 1 showed relatively rapid stabilization and never exhibited severely contaminated leachate when compared to the other cells. The input values of the ultimate chlorides and initial chlorides for Cell 1 were reduced to simulate the actual results. The initial chloride concentration used for Cell 1 leachate was 300 mg/l and the ultimate chloride output adjusted to 800 mg/kg of dry waste.

A simulation of COD leached from Cell 8 was also run. The values for COD are substantially greater than those for chlorides. The initial values of COD were set at 20,000 mg/l and the ultimate COD leached was 40,000 mg/kg of dry waste. The results of the simulation of COD in Cell 8 is shown in Figure 32.

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HAM LYSIMETERS CELL 1
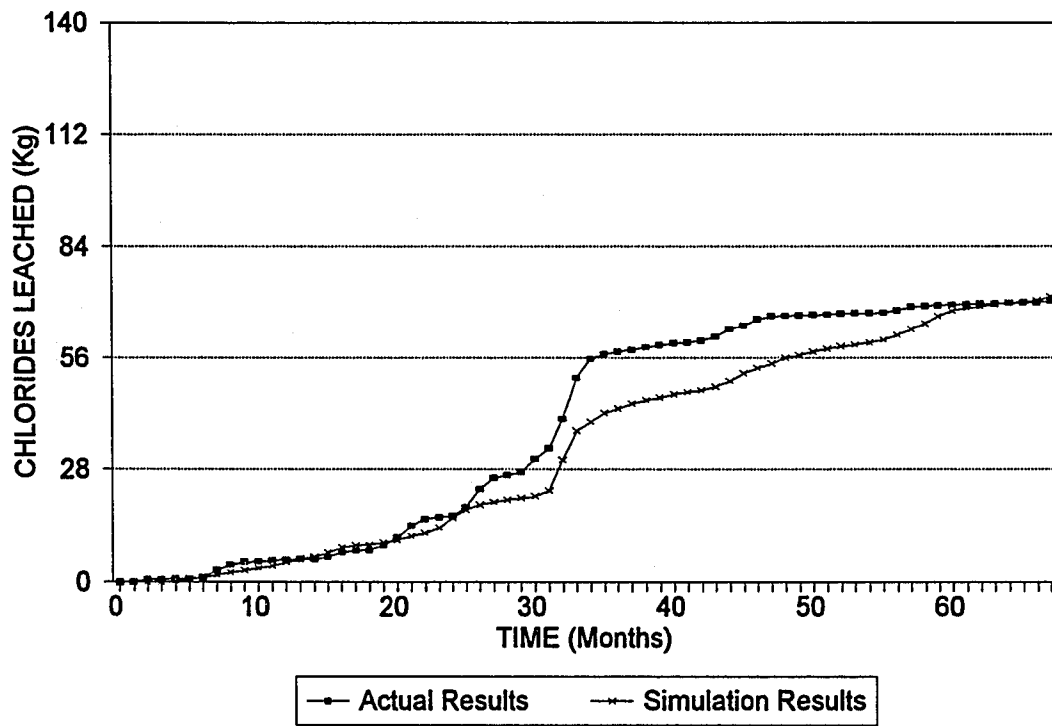## CUMULATIVE CHLORIDES LEACHED

Figure 29. Cumulative chlorides in Cell 1.

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HAM LYSIMETERS CELL 2
CUMULATIVE CHLORIDES LEACHED

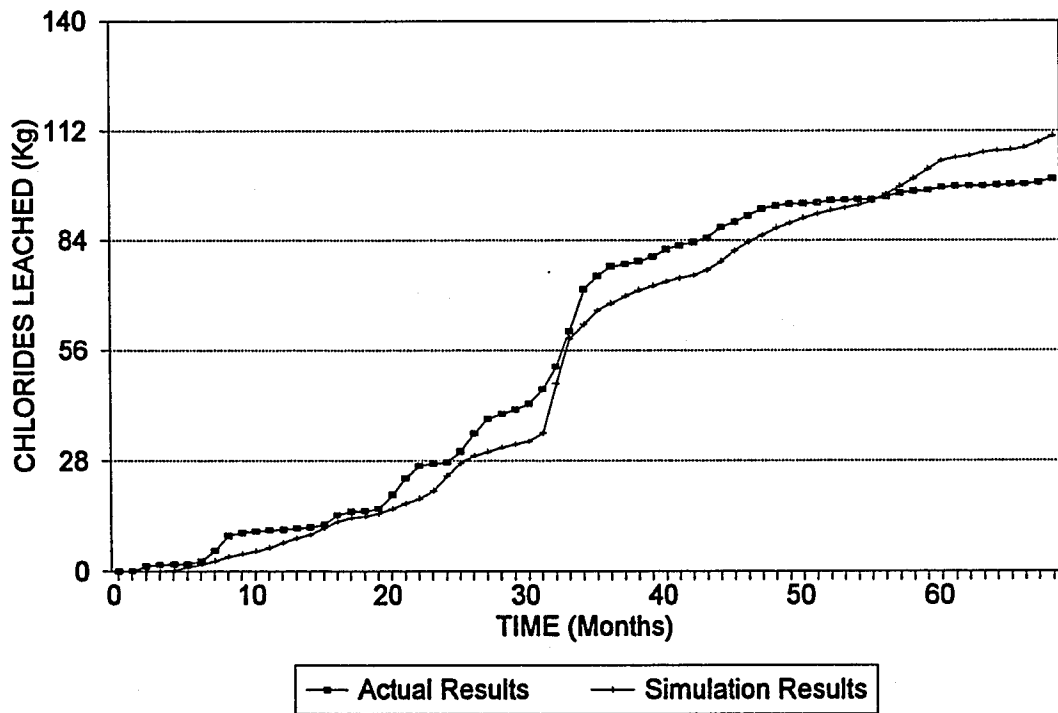

Figure 30. Cumulative chlorides in Cell 2.

152

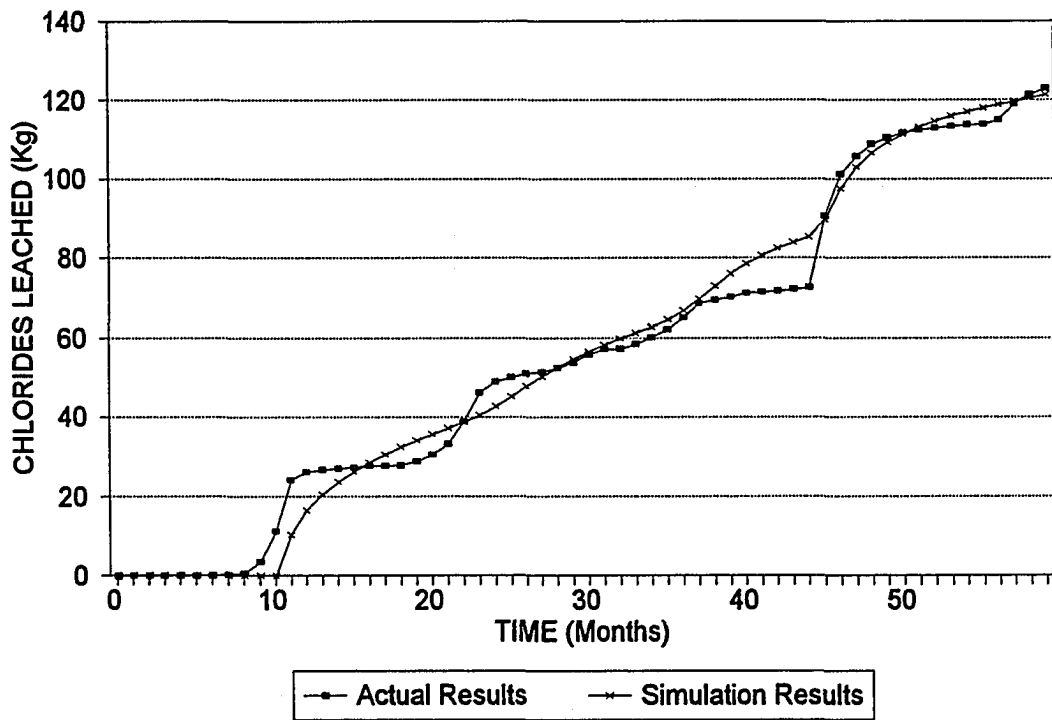# HAM LYSIMETER CELL 8
CUMULATIVE CHLORIDES LEACHED

Figure 31. Cumulative chlorides in Cell 8.

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HAM LYSIMETER CELL 8
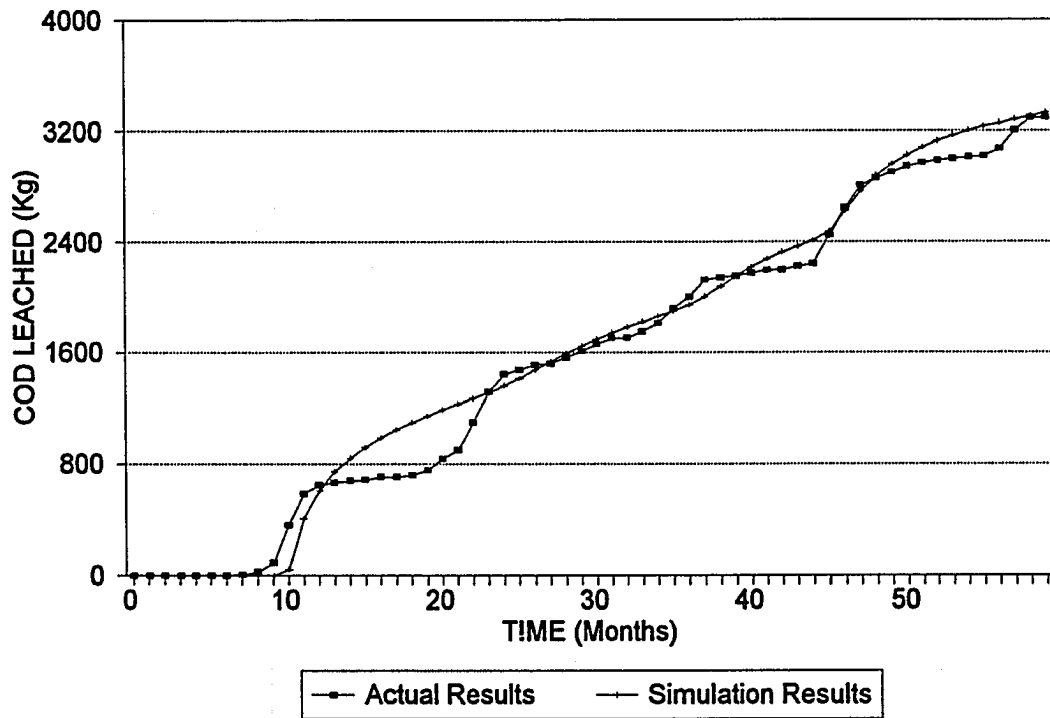## CUMULATIVE COD LEACHED

Figure 32. Cumulative COD in Cell 8.

Visual inspection of the three cells indicates the simulations track the leached chloride and COD trends with a good degree of accuracy. The deviations of the model simulations of chlorides for Cells 1, 2, and 8 are 7.55, 6.127, and 4.88, respectively. The deviation for COD in Cell 8 is 148. The large difference in the deviations between the two contaminants for the same cell is due to the levels of contaminants. The amount of chlorides leached in Cell 8 is 125 kg. The amount of COD leached for the same cell is 3295 kg. It would be hard to compare the two deviations unless they were normalized. If this was done by dividing each by the total amount of contaminant leached, new values of chloride deviation would be .039 and the value of the COD deviation would be .045.

4.2.1.3 Comparison with the HELP Model

The water flow portion of the simulation of the University of Wisconsin lysimeters can be compared to the output from a simulation using the Hydrologic Evaluation of Landfill Performance (HELP) model developed by Schroeder *et al.* (1983a). A verification of the HELP model was done by Peyton and Schroeder (1988) using the University of Wisconsin lysimeters 2 and 4. The first two years were treated as equilibrium periods and are not used in the comparisons. Cells 2 and 4 were selected since one was covered and the other was left uncovered. The difference between the University of Wisconsin cell designs was depth, cover, and shredded refuse. Peyton and Schroeder (1988), felt that depth and shredded refuse would not affect the

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water balance. Hence, they modeled Cell 2, since it was covered, and Cell 4, since it remained uncovered. A simulation of Cell 2 has already been reported in Section 4.2.1.1 and a comparison will be made between actual data reported by Ham (1980), the HELP verification reported by Peyton and Schroeder (1988), and this model simulation of Cell 2 leachate.

The simulation output reported by Peyton and Schroeder (1988) was in cumulative leachate in inches as shown in Figure 33. It is noted that the HELP simulation begins after two years in order for the cell to stabilize. Hence Cell 2 was loaded in September, 1970, and the HELP model simulation begins in September, 1972. The HELP data was converted to cumulative volume (cubic meters) by multiply the cumulative drainage (inches) by the area of the cell (30 ft x 60 ft) and converting it to cubic meters. The HELP output is plotted with the model simulated output previously shown in Figure 26. The comparison of the actual, HELP, and this model is seen in Figure 34. The first 24 months only show the actual data and the model simulation. At month 24, the HELP model begins tracking. The model simulation of cumulative leachate compares very favorably to the actual data as well as the HELP model. The deviation of the Cell 2 simulation was previously reported as 4.02. If the first 24 months are removed from the calculation, the deviation is 4.76. The deviation between the HELP model and the actual data is 13.7, which is approximately three times greater than the Cell 2 simulation.
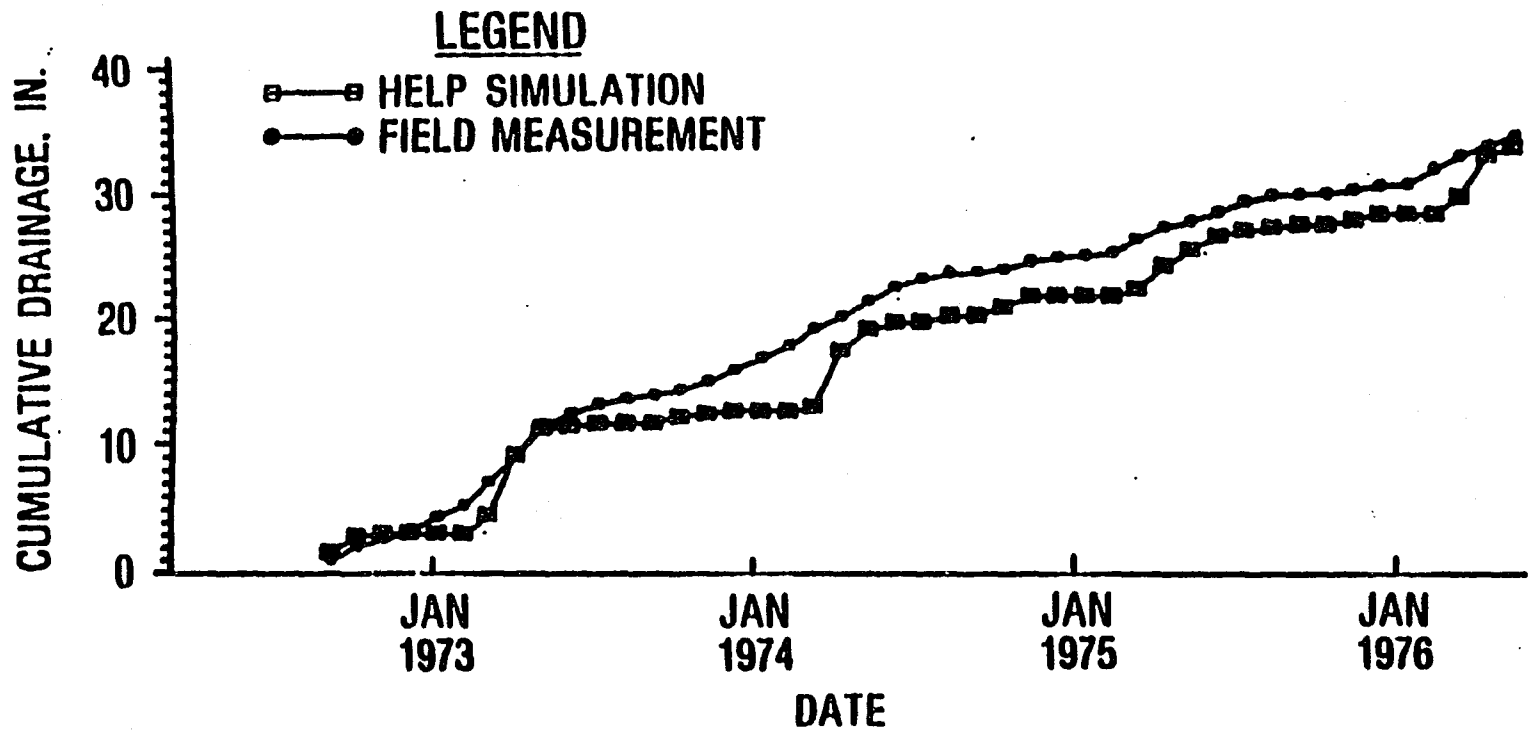
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Figure 33. HELP model simulation of cumulative leachate in Cell 2 (After Peyton and Schroeder 1988).

HELP COMPARISON CELL 2
CUMULATIVE LEACHATE
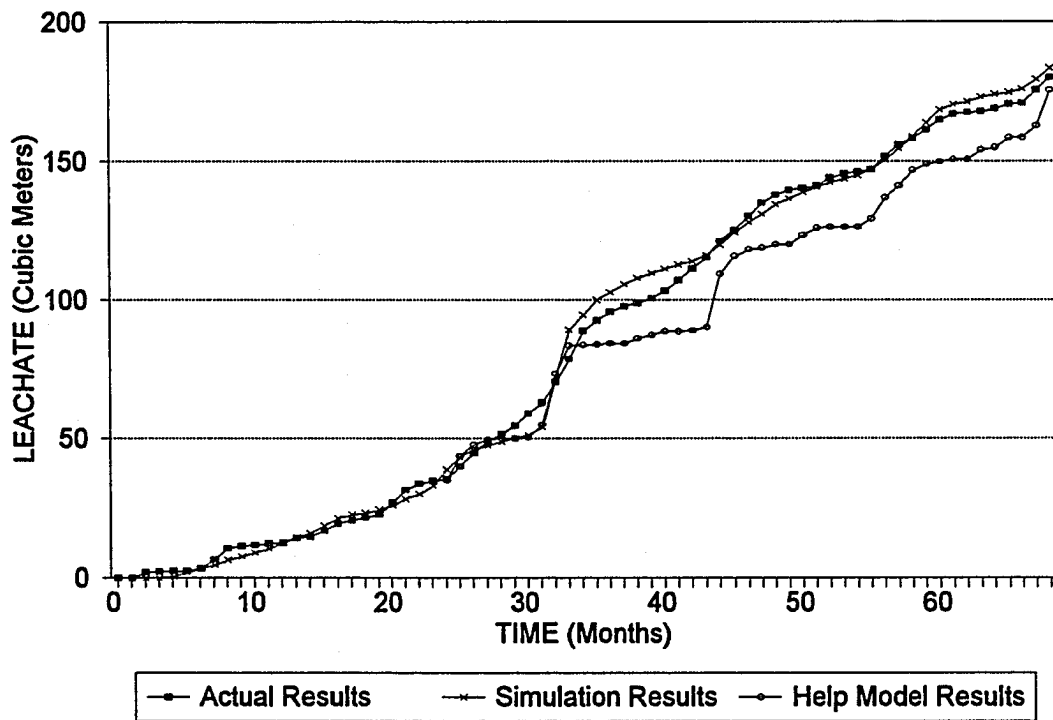
Figure 34. Comparison of the model to the HELP model using Cell 2 data. The HELP simulation begins at month 24.

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4.2.2 Boone County Landfill

Landfill research was also conducted at the Boone County Landfill in Kentucky. Five sanitary landfill test cells were constructed containing municipal solid waste in 1971 and 1972. Four of the cells were small test cells. Test Cell 1 was constructed similar to normal landfill cells and it was operated from June, 1971 until February, 1979. The cell, shown in Figure 35, was 45.4 m long by 9.2 m wide. There were vertical side walls and ramps on both ends sloping to the drain system. The composite liner system was constructed of clay and synthetic liners. The collection drain system had dual drains (one over the other) surrounded by silica gravel (one over the other) as shown in Figure 35. The gravel was limestone free to prevent undesirable chemical reactions (Wigh 1984).

Residential solid waste with a total volume of 665 cubic meters was spread in .6 m lifts and compacted to a mean dry density of 429 kg/cubic meter with a moisture content of 27 percent. Approximately .6 m of soil was placed over the refuse and compacted to a dry density of 1597 kg/cubic meter. The permeability of the cover was tested and found to be 3.15×10^{-5} cm/sec. Rainfall and leachate data were given as yearly amounts with 72.5% of the rainfall lost to runoff and evapotranspiration (Wigh 1984). The model was used to simulate the Boone County Landfill using the calibrated values determined from the University of Wisconsin Lysimeters.

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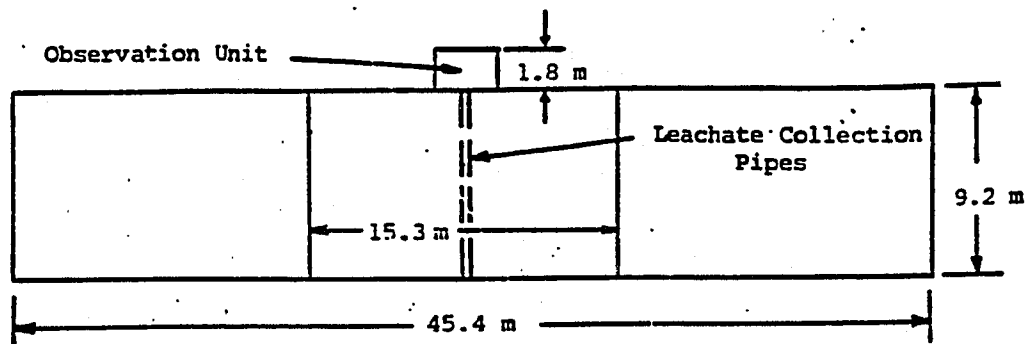
PLAN
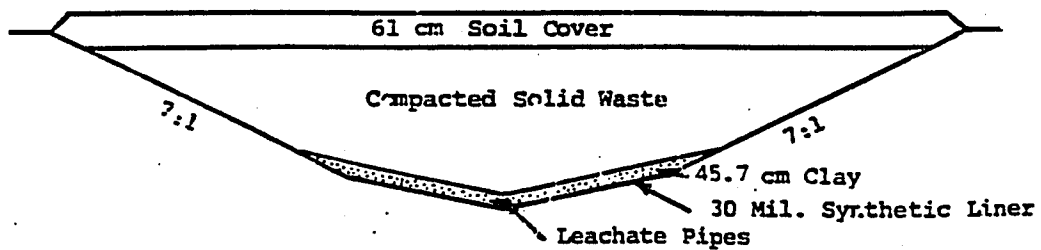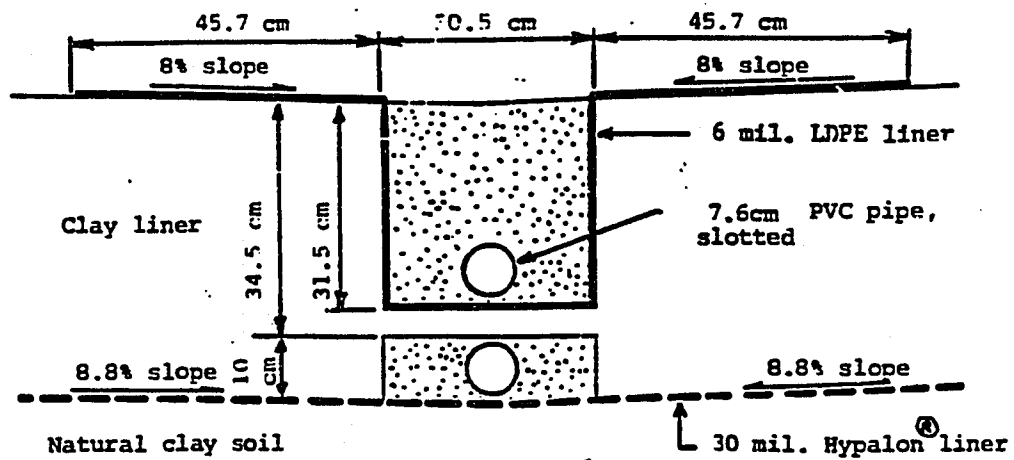
SECTION

DRAIN PIPES

Figure 35. Boone County Landfill Cell 1 design and leachate collection system (After Wigh 1984).

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4.2.2.1 Model Simulation of Boone County Water Flow

A simulation was run for the Boone County Landfill. Input files were developed for Cell 1 using information reported by Wigh (1984). The time step used was 0.5 hours and the simulation was run for 92.5 months (7.7 years). The input file broke the landfill cell into two vertical cells and combined the data. Several cases were run using different node heights, Δz, with the output being approximately equal. The thickness of the cover was set at 0.6 meters. It was assumed that the cell construction did not lend to lateral flow, hence the lateral seepage was simulated to be zero. The dry density of the waste was inputted to be 429 kg/cubic meter. Estimation of the refuse's saturated permeability and the porosity was set at 7.2 x 10^{-3} m/hr and 0.52, respectively (Schroeder *et al.* 1984). Surface cover permeability was set at 1.65x10^{-2}m/hr and surface porosity was set at 0.58. Similar to Section 4.2.1.1, estimations of the soil relative permeability and saturation coefficients were 5.3 and 2.0 for α and X_m. Corresponding values of K_r and $\frac{\partial S}{\partial P}$ were 1.47 x 10^{-5} and 0.16, respectively. The selected value for chlorides in the rain water was set at zero. There was no recirculation and that function in the model was bypassed. A precipitation file was developed using the yearly rainfalls as reported by Wigh (1984).

The model simulation results obtained are plotted with the measured results for Cell 1 in Figure 36. The model simulation shows that leachate is not produced until approximately the eighth month. When the cell storage is filled, the model goes to a quasi-steady

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BOONE COUNTY LANDFILL
CUMULATIVE LEACHATE
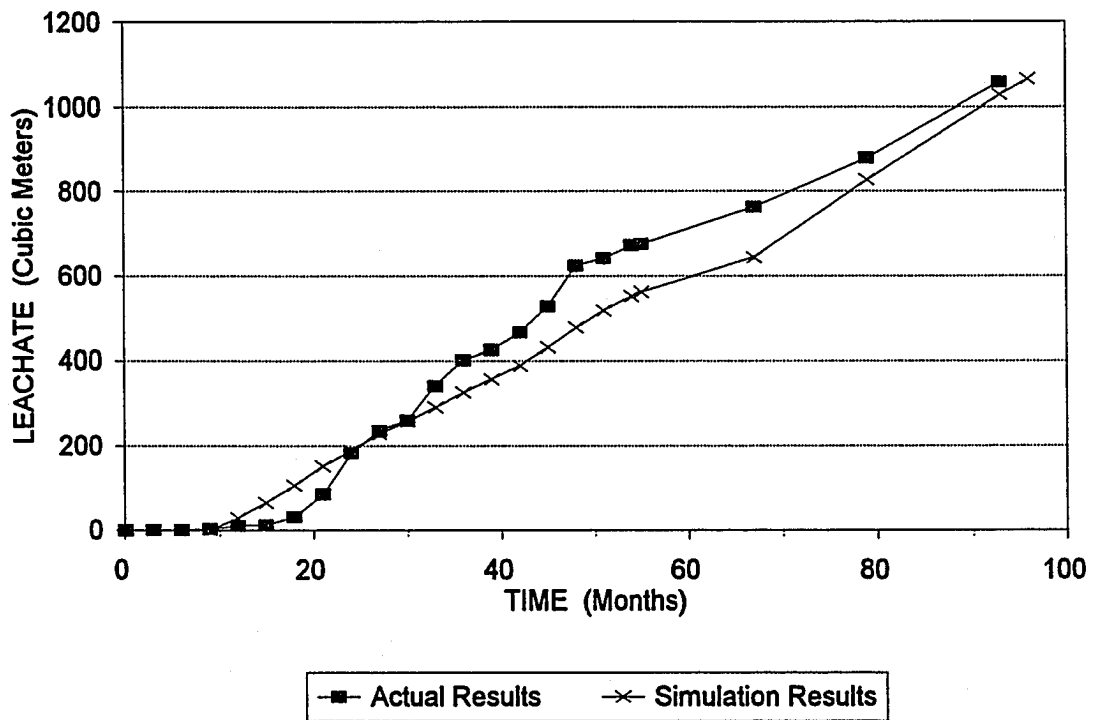
Figure 36. Cumulative leachate for the Boone County Landfill.

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state condition. The actual measured data points are available at quarterly intervals until the 55th month and then at yearly intervals. The model simulation points, although available monthly, are plotted at the same intervals as the actual times to avoid clutter. Visual observations indicate the model values track fairly well with the actual leachate values. A deviation was not done since the actual output data was not given at a steady interval.

4.2.2.2 Model Simulation of Boone County Contaminants

Chloride contaminants leached out of the cell were measured by Wigh (1984) at the Boone County Landfill and have been simulated. Diffusion and dispersion coefficients are the same as those used in Section 4.2.1.2. Values of 0.002 and 10 for the coefficients a and b, respectively and a value of 0.04 cm^2/hr for D_o were selected. A value for λ of 0.39 cm was also used (Bresler 1973). Production coefficients for chlorides were selected similar to Section 4.2.1.2. The ultimate chloride load used was 1200 mg/kg of dry waste. A rate coefficient of 0.45 and an initial concentration of 900 mg/l were selected.

The model simulation for chlorides was run for Cell 1 of Boone County Landfill. The simulated cumulative mass of chlorides being leached out of the cells as a function of time was plotted with the actual values in Figure 37. It is evident from the slope of the curve that chlorides are being depleted with time. The model is not showing as great a rate of chloride leaching from month 20 to month

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BOONE COUNTY LANDFILL
CUMULATIVE CHLORIDES LEACHED
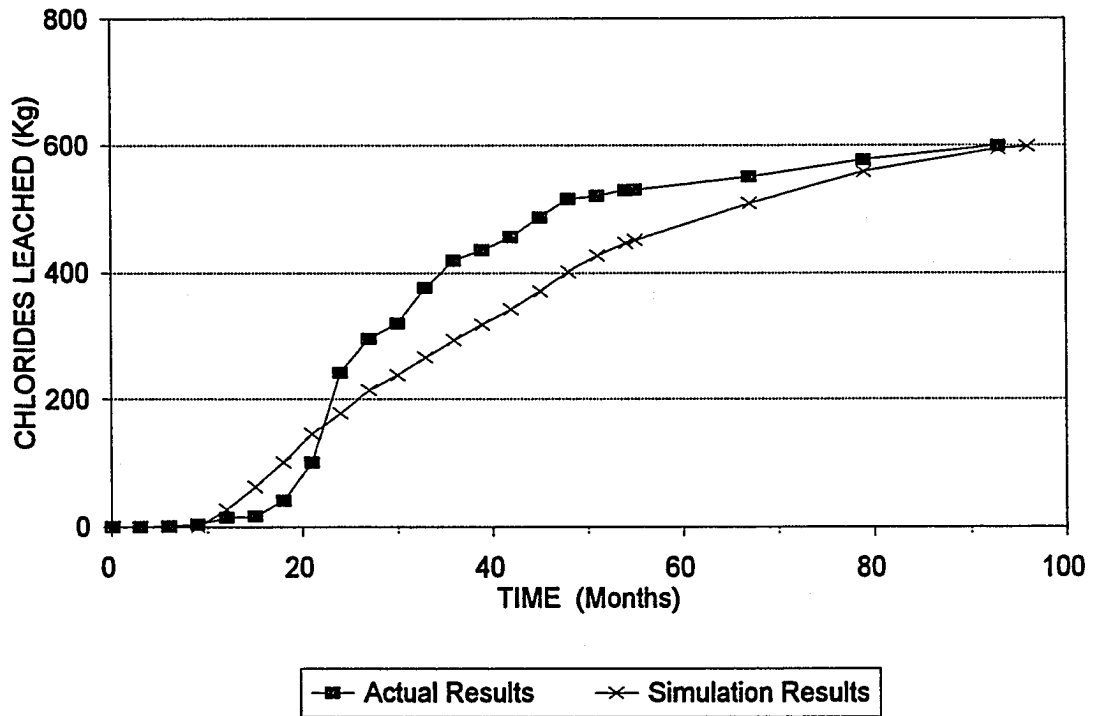
Figure 37.   Boone County Landfill cumulative chlorides leached.

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30, however it does catch up and is showing the overall depletion of chlorides from the cell. This is more evident than the previous tests since the test ran for a greater time (95 months) period. It is important that the model simulation is showing a depletion of chlorides from the cell.

4.2.3 Private Landfill Gas Production

A private landfill in Virginia has been in operation since 1976 and a site view is shown in Figure 38. The figure shows the area which has been filled or is currently working. A full drawing of the landfill shows future plans for expansion to the southwest. The northeast portion of the landfill was filled and a final cap was put in place in 1985. Cells 1,2,3, and 4 were working and a final cap was put in place in 1990. Currently Cells 4, 5, 12, and 13 have completed fill and have a temporary cap in place awaiting final cap. Cells constructed starting with cell 5 and 12 and beyond have a double or composite liner. Cells constructed up to cells 4 and 13 all have a single liner on the bottom. This is shown on the site drawing by the SL and DL designation schemes. New regulations require a double or composite liner on the landfill bottom. Currently cells 6 and 11 are the working cells.

Gas wells have been installed to remove and process the gases. The wells are throughout the landfill up to cells 5 and 12. Gas wells have not yet been placed in cells 6 and 11 since they are still working. The gas removed from the landfill is used in three

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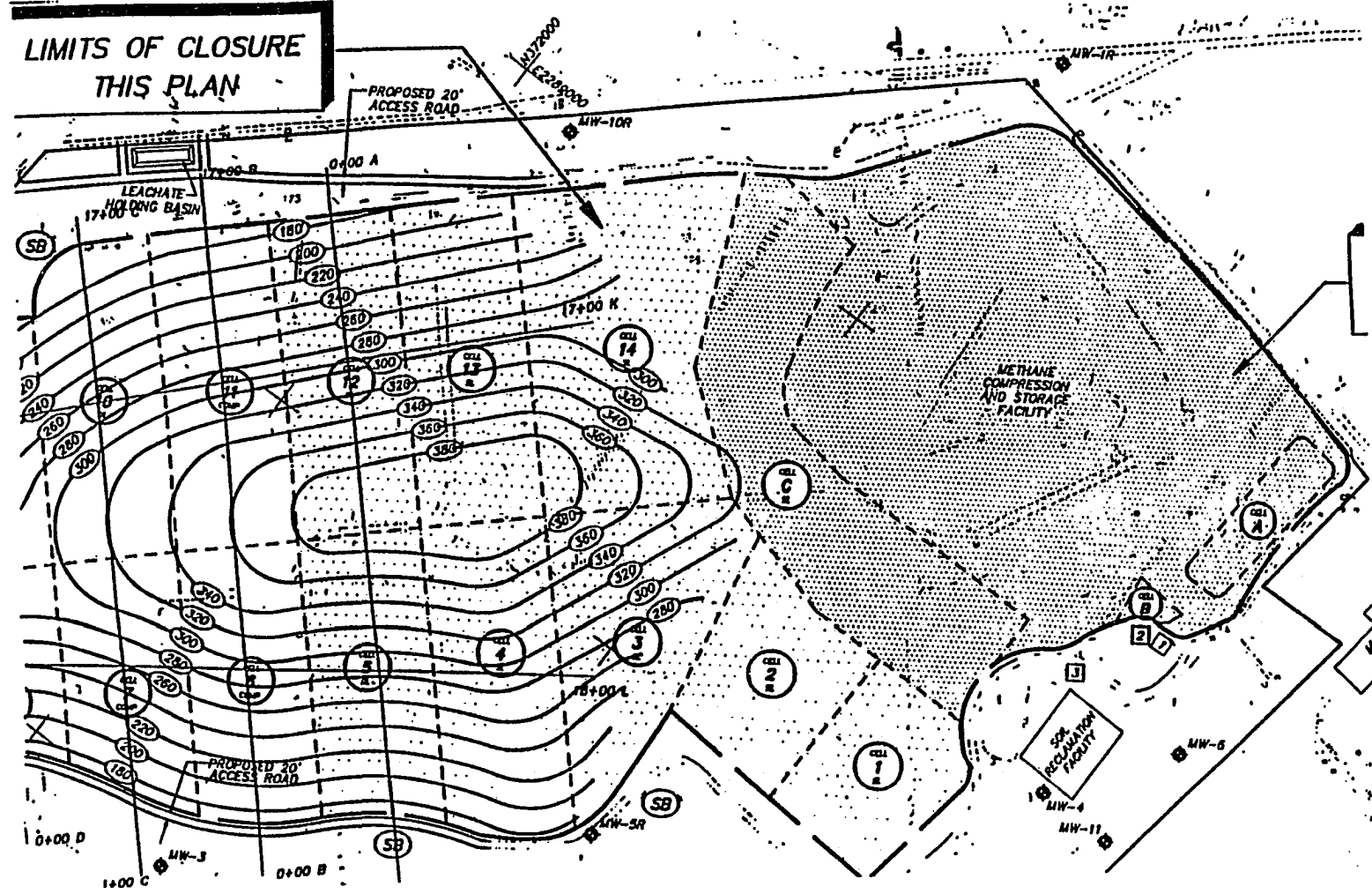
Figure 38. Site plan of a private landfill producing methane gas.

operations. They produce 6 megawatts of power using 14 generators which burn the methane gas. The power supplies the site and is also sold to the power company. They also sell the methane to a themro process company which uses the energy for the treatment of contaminated soils. The third use is the on-site asphalt plant. The gas production rates from the landfill is given in Table 7.

The landfill has obtained a permit which allows them to conduct recirculation operations using the leachate collected. This reduces leachate treatment costs and promotes gas generation. Leachate from the landfill is recirculated using trucks on the top of the cells with the double liners.

4.2.3.1 Model Simulation of Landfill Gas Generation

The private landfill was modeled for gas generation. Rain fall data was not given, hence, average yearly rainfall for the area was used and the evapotranspiration routine was set up to match the leachate rates with a leachate collection rate determined over six months (January to June, 1993). The rates given in Table 7 are in cubic feet per time. The model was built to give gas volumes in cubic feet. Two different mathematical gas generation models discussed in Section 3.6.1 were incorporated into this model. Runs were made for both equations and a comparison was made.

The landfill was divided up into elements for the model and the areas and heights of each element was determined using the drawings.

Table 7.

METHANE GAS PRODUCTION FOR PRIVATE LANDFILL

Unit	Operation	Days	Gas Rate
Electric Power Plant (6 MW)	15 hours/day	6 Days/wk	3000 CFM
TPS	12 hours/day	6 Days/wk	3000 CFM
Asphalt Plant	10 hours/day	3 Days/wk	600 CFM

Total Methane Production: 30.2 million cubic feet/week

129 million cubic feet/month

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The cell histories were used to determine the time that different cells have been producing gas. The current overall gas generation rate will have inputs from different parts of the landfill which have been producing gases for different periods of time, and thus, will have different rates. Ideally the gas production rate of each cell would be known, however, this was not the case. The model was run for different times depending on the landfill, and the gas production from the various cells was added for the total production and compared to that in Table 7.

Input data for the energy routine was determined and read in as model input. These included average air temperatures for the different months of the year. Specific heats, C_p, of water of 4182 $\frac{J}{Kg \cdot K}$ and a specific heat of waste of 1050 $\frac{J}{Kg \cdot K}$. The specific heat of water was determined from a list of properties. Using specific heats of the materials in municipal solid waste and determining a weighted average yielded a specific heat of 1166 $\frac{J}{Kg \cdot K}$. Rich (1963) reported an average waste specific heat of 1050 $\frac{J}{Kg \cdot K}$. The thermal conductivity, k, was determined using a weighted average of the thermal conductivities of the waste properties. A value of 17,5 $\frac{kJ}{m \cdot K \; hr}$ was determined. The value used for the density of water was 987 Kg/m^3.

Gas production using Equation (92) was first modeled. The factors reported by Hartz (1980) were used for the model parameters. Values of 0.319 ft^3/lb and 1.035 were used for Q and A respectively. These produced very high production rates in the initial time steps and the total gas produced quickly approached the maximum gas yield.

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Even if the moisture content factor, F_{mc} taken out of the equation, the initial gas rate exceeded the gas rates reported in literature. A gas production limiter was built into the model, and results were obtained. Using this method the model predicted a gas production rate of 1.01 million cubic feet of gas per month. This does not compare favorably with the 129 million cubic feet per month as reported in Table 7.

Gas production using Equation (91) was modeled next. Hartz (1980) reports using parameter values of 2.63 for k and 1.87×10^{-2} for A. An initial value for P had to be assumed since a small number taken to the negative 2.63 power will be very large. An initial value of .1195 ft^3/lb was used. This value yielded an initial production rate of .005 ft^3/lb day which is reported as being the maximum production rate (Hartz 1980). A plot of the cumulative methane produced from a cell is shown in Figure 39. The gas production rates of the same cell are shown in Figure 40. Adding up all the landfill cells yields a model predicted methane rate of 143 million cubic feet of gas per month. This is 11 percent higher than the value reported from Table 7.

Hartz (1980) used analyzed data from five landfills. A set of parameters fit very well to the landfill data. The set gave values of 1.5 and 1.75×10^{-4} for k and A respectively. A run was made using these parameters. The results yielded predicted gas production rates of 68.7 million cubic feet of methane per month. This is low by approximately 48 percent. The best results have come form the model Hartz (1980) recommends with the parameters he suggests.

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CUMULATIVE LANDFILL GAS
CELL 1-6-3
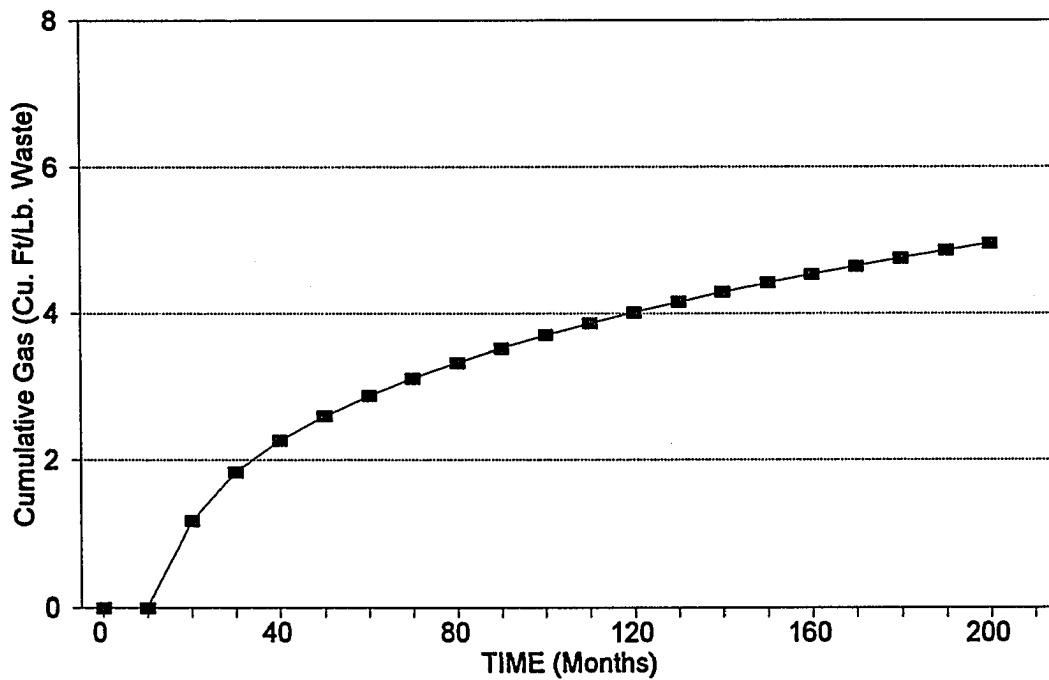
Figure 39. Cumulative gas produced from a cell in the landfill.

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GAS PRODUCTION RATES

CELL 1-6-3
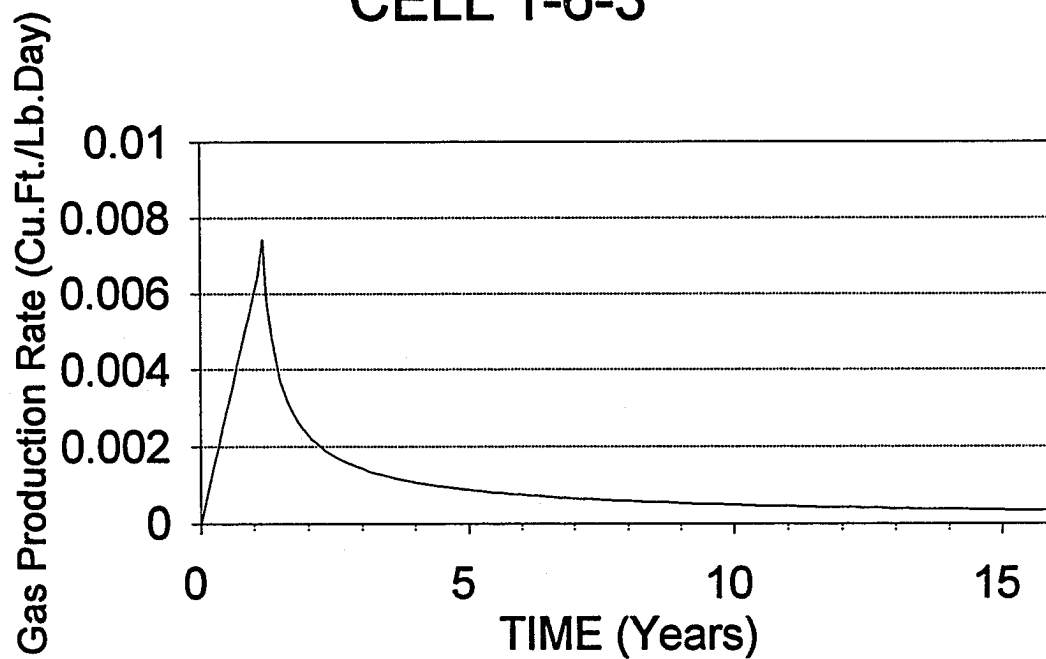
Figure 40. Gas production rates from a cell in the landfill.

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A plot of landfill temperature after 12 months of simulation time is shown in Figure 41. The surface temperature is at zero depth and the temperature rises as depth increases. These temperatures are what the model used in determining the adjusted coefficient for that time step.

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LANDFILL MODEL TEMPERATURE
After Twelve Months Simulation
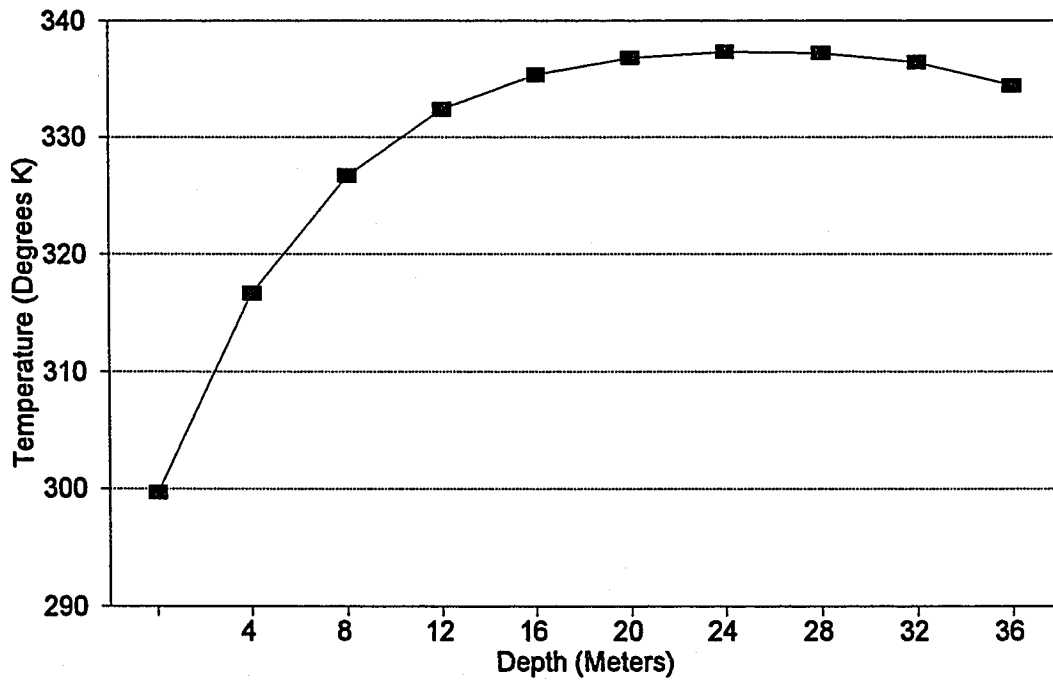
Figure 41.   Temperature in the landfill as predicted by the model after a simulated 12 months.

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5. SUMMARY AND CONCLUSIONS

A numerical quasi-three-dimensional unsteady model has been developed which couples many of the landfill processes together in order to predict the landfill leachate, contaminants, gas production, and temperatures. The model separates a landfill into three spatial dimensions and assigns nodes in each direction. Flow equations are solved to determine the heads and saturation at each node in the landfill per time step. These quantities are used to determine vertical and lateral flows through each node at each time step. Flow rates are used to determine contaminant production and transport at each node. The moisture content and temperature are used to determine methane gas generation in each node.

Many of the factors affecting the water flow in a landfill are modeled and used by the flow routine. The water infiltrating the surface is determined using precipitation, evapotranspiration, and a two-dimensional overland flow model coupled to the landfill flow model. Values for the saturation and permeability of the waste inside the landfill are determined from the heads at each node through empirical relationships. Vertical flows in the landfill are determined by Richard's equation while lateral flows between saturated cells in the landfill are determined by Darcy's equation.

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These equations are solved with the Generalized Newton Method and the Thomas Algorithm. Visual comparison of flow simulation data and root mean square deviation analysis showed strong correlation to actual data.

Flows out the landfill bottom can be modeled as it enters the drainage layer of the leachate collection system. The leachate liner and collection system was modeled using a one-dimensional, saturated, unsteady flow model in which the flow moves in a direction parallel to the sloping liner. Seepage through the liner into the ground under the landfill can be modeled using the leachate head on top of the liner, as determined by the model, and the permeability of the liner.

Contaminant production was modeled using leachate flow through the landfill. Chloride and COD production curves were coupled to contaminant transport equations and leachate flow equations. The contaminant transport equations were equivalent to the water flow equations and solved using the same techniques. Visual and root mean square analysis indicated good agreement with real data for chlorides and COD. The recirculation of leachate and contaminants in the infiltrating water were also included in the two transport routine, but not tested.

Gas production in a landfill was modeled using the moisture content, temperature, and previous gas production history in each node. The determination of temperature required an energy transport model. This was equivalent to the flow and contaminant transport

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routines and solved in the same manner. The heat of reaction for methane was used as a source term for the energy equation and the air temperature was used as a boundary condition. Gas transport was not modeled since the time reference of gas is larger than that of contaminants or water transport. Gas production is assumed to be steady state compared to the water and contaminant flow routines.

The model was initially checked using noflow, mass balance, and symmetry tests. It was then verified using real landfill data which included four University of Wisconsin lysimeters, the Boone County Landfill project, and a privately operated landfill in Virginia. A comparison was made with the HELP model for leachate production with test Cell 2 of the University of Wisconsin lysimeters. The deviation of the HELP simulation from the actual data was 13.7. The deviation for this model was 4.76. Equally good simulation was accomplished with the Boone County Landfill.

Contaminants were simulated and compared to the University of Wisconsin and Boone County landfill test results. Simulation results were very dependent upon the ultimate amount of contaminant per unit mass of waste and with the initial contaminant concentrations. The ultimate amount of contaminants to be leached from a mass of waste and the initial concentrations depends on the waste. The anticipated ultimate load depends on the waste composition. The initial concentrations of chlorides in the leachate were approximately 900 mg/l, with Cell 1 below that value at 300 mg/l. Initial concentrations for COD were approximately 20,000 mg/l.

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Data from a private landfill was used to verify the gas production routine. The input data was very sparse and only the gas portion was modeled. The model prediction was 11 percent greater than the actual gas production. Other types of mathematical models were tested. Favorable results were found with only one model.

It is possible to couple several of the landfill processes together to predict the behavior of some important landfill processes. An extremely critical factor in the successful prediction of the processes will be the input data available. With many changes ongoing in landfill regulations, more data will become available and required for landfill work. This will result in more modeling for planning and permitting.

5.1 FUTURE RESEARCH

The ability to conduct follow on research using this model is very strong. The model has many routines which can be expanded to look at other engineering areas as well as solving landfill problems. The water flow model can be used for ground water flow in an unsaturated/saturated zone. The runoff model coupled to the water flow model could be used to predict infiltration of water into the water table. The contaminant transport routine could be expanded to include a wide range of contaminants. One possibility is to model the slow seepage of contaminants to predict effects of spills after numerous years. Possible contaminants could vary from oil spills to

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radioactive seepage. The liner routine could be used to evaluate liner designs using different permeabilities and leachate flow rates. The overland flow model could be used for numerous overland flow simulations.

Other recommendations for follow on research include:

1. Developing the contaminant diffusion and dispersion coefficients for other contaminants or materials.

2. Finding a landfill with good simultaneous data for leachate, contaminants, gas generation, precipitation, and cover data to use this model to simulate all of the factors together.

3. Developing a routine which determines the overall mass loss from the landfill through gas production and contaminant losses to leachate in order to determine the long term structural stresses on the landfill.

4. Continue working with the recirculation portion of the model in order to make predictions on the amount of recirculation a facility will be able to conduct and the amount they will have to treat.

5. Developing a large spill capability. With the contaminant built in as a source in the infiltration, production need not be used and the transport of a contaminant can be modeled.

6. Developing a capability to look at seepage into the ground with no liner present, or an infinite depth. This could be useful with the seepage of radioactive liquids at a drum disposal site.

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Also, with the storage for highly radioactive spent fuels running out, plans for the disposal of the fuels will need to be made.

REFERENCES

Abriola, L. M., and Pinder, G. F. (1985a). "A Multiphase Approach to the Modeling of Porous Media Contamination by Organic Compounds 1. Numerical Simulation", Water Resources Research, Vol. 21, No. 1, pp. 11-18.

Abriola, L.M., and Pinder, G.F. (1985b). "A Multiphase Approach to the Modeling of Porous Media Contamination by Organic Compounds 2. Numerical Simulation", Water Resources Research, Vol. 21, No. 1, pp. 19-26.

Ahmed, S., Khanbilvardi, R.M., Fillos, J., and Gleason, P.J. (1992). "Two-Dimensional Leachate Estimation Through Landfills", Journal of Hydraulic Engineering, Vol. 118, No.2, p. 306.

Akan, A. O. (1984). "Mathematical Simulation of Snowmelt and Runoff from Snow Covers", Frontiers in Hydrology, Water Resources Publications, Littleton, Colorado, pp. 79-92.

Alpern, R. M. (1974). "Gas Generation in a Sanitary Landfill," thesis presented to California State University, at Long Beach, California., in 1974, in partial fulfillment of the requirements for the degree of Master of Science.

Archer, D. B. and Kirsop, B. H. (1990). "The Microbiology and Control of Anaerobic Digestion", Anaerobic Digestion: a Waste Treatment Technology, Critical Reports on Applied Chemistry, Vol. 31, Elsevier Applied Science, London, 1990.

Blanchet, M. J., and staff of the Pacific Gas and Electric Company. (1977). "Treatment and Utilization of Landfill Gas", Mountain View Project Feasibility Study, EPA-530/SW-583.

Bowerman, F. R., Rohatgi, N. K., Chen, K. Y., and Lockwood, R. A. (1977). A Case Study of the Los Angeles County Palos Verdes Landfill Gas Development Project, Ecological Research Series, EPA-600/3-047.

Bresler, E. (1973). "Simultaneous Transport of Solutes and Water under Transient Unsaturated Flow Conditions", Water Resources Research, Vol. 9, No. 4, pp. 975-987.

181

Bureau of Sanitation, City of Los Angeles. (1975). "Estimation of the Quantity and Quality of Landfill Gas from the Sheldon-Arleta Sanitary Landfill," Internal Report.

Burnett, R. D., and Frind, E. O. (1987a). "Simulation of Contaminant Transport in Three Dimensions 1. The Alternating Direction Galerkin Technique", Water Resources Research, Vol. 23, No. 4, pp. 683-694.

Burnett, R. D., and Frind, E. O. (1987b). "Simulation of Contaminant Transport in Three Dimensions 2. Dimensionality Effects", Water Resources Research, Vol. 23, No. 4, pp. 695-705.

California State Water Control Board. (1964). "In situ Investigation of Movement of Gases Produced from Decomposing Refuse," Engineering Science, Inc., Publication No. 31.

Cederberg, G. A., Street, R. L., and Leckie, J. O. (1985). "A Groundwater Mass Transport and Equilibrium Chemistry Model for Multicomponent Systems", Water Resources Research, Vol. 21, No. 8, pp. 1095-1104.

Charbeneau, R. J. (1981). "Groundwater Contaminant Transport with Adsorption and Ion Exchange Chemistry: Method of Characteristics for the case Without Dispersion", Water Resources Research, Vol. 17, No. 3, pp. 705-712.

Chian, E. S. K., and DeWalle, F. B. (1977). Evaluation of Leachate Treatment, Volume I, Characterization of Leachate, EPA-600/2-77-186a.

Colona, R. A. (1976). "Methane Gas Recovery in Mountain View Moves into Second Phase," Solid Waste Management, Vol. 19, No.5, p.90.

Constantinides, C. A., and Stephenson, D. (1981). "Two-Dimensional Kinematic Overland Flow Modeling", Proceedings of the Second International Conference on Urban Storm Drainage, Volume I, Hydraulics and Hydrology.

Cooney, C. L., and Wise, D. L. (1975). "Thermophylic Anaerobic Digestion of Solid Waste for Fuel Gas Production," Biotechnology and Bioengineering, Vol. 17, p. 1119.

Crawford, J. F., and Smith, P. G. (1985). Landfill Technology, Butterworths, London.

Dass, P., Tamke, G. R., and Stoffel, C. M. (1977). "Leachate Production at Sanitary Landfill Sites", Journal of the Environmental Engineering Division, ASCE, Vol. 103, No. EE6, pp.981-988.

Demetracopoulos, A. C., Korfiatis, G. P., Bourodimos, E. L. (1984). "Modeling for Design of Landfill Bottom Liners", Journal of the Environmental Engineering Division, ASCE, Vol. 110, No. 6, pp.1084-1098.

Demetracopoulos, A. C., Sehayek, L., and Erdogan, H. (1986). "Modeling Leachate Production from Municipal Landfills", Journal of the Environmental Engineering Division, ASCE, Vol. 112, No. 5, pp. 849-866.

DeWalle, F. B., Chian, E. S. K., and Hammerburg, E. (1978). "Gas Production from Solid Waste in Landfills", Journal of the Environmental Engineering Division, ASCE, Vol. 104, No. EE3, pp. 415-432.

Eide, A. R., Jenison, R. D., Mashaw, L. H., and Northup, L. L. (1986). Engineering Fundamentals and Problem Solving, McGraw-Hill Book Company, New York.

EMCON ASSOCIATES. (1980). Methane Generation and Recovery from Landfills, Ann Arbor Science Publishers.

Fair, G. M., and Moore, E. W. (1932). "Heat and Energy Relations in the Digestion of Sewage Solids", Sewage Works Journal, pp. 430-439.

Farquhar, G. J. (1989). "Leachate: Production and Characterization", Canadian Journal of Civil Engineering, Vol. 16, pp. 317-325.

Fenn, D. G., Hanley, K. J., and DeGeare, T. V. (1975). "Use of the Water Balance Method for Predicting Leachate Generation at Solid Waste Disposal Sites", EPA-530/Sw-169, U.S. Environmental Protection Agency, Cincinnati, Oh.

Findikakis, A. N. and Leckie, J. O. (1979). "Numerical Simulation of Gas Flow in Sanitary Landfills", Journal of the Environmental Engineering Division, ASCE, Vol. 105, No. EE5, pp. 927-945.

Freeze, R. A. (1969). "The Mechanism of Natural Ground-water Recharge and Discharge 1. One-dimensional, Vertical, Unsteady, Unsaturated Flow above a Recharging or Discharging Ground-water Flow System", Water Resources Research, Vol. 5, No. 1, pp. 153-171.

Freeze, R. A. (1971). "Three-dimensional, Transient, Saturated-Unsaturated Flow in a Groundwater Basin", Water Resources Research, Vol. 7, No. 2, pp. 347-366.

Frind, E. O., and Hokkanan, G. E. (1987). "Simulation of the Borden Plume Using the Alternating Direction Galerkin Technique", Water Resources Research, Vol. 23, No. 5, pp. 918-930.

Fungaroli, A. A. and Steiner, R. L. (1979a). "Investigation of Sanitary Landfill Behavior, Volume I, Final Report", EPA-600/2-79-053a, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Fungaroli, A. A. and Steiner, R. L. (1979b). "Investigation of Sanitary Landfill Behavior, Volume II, Supplement to the Final Report", EPA-600/2-79-053b, U.S. Environmental Protection Agency, Cincinnati, Ohio.

183

Gee, J. R. (1981). "Prediction of Leachate Accumulation in Sanitary Landfills", Proceedings of the Fourth Annual Madison Conference of Applied Research and Practice on Municipal and Industrial Waste, Department of Engineering and Applied Science, University of Wisconsin Extension, Madison, Wisconsin, pp. 170-190.

Giesel, W., Renger, M., and Strebel, O. (1973). "Numerical Treatment of the Unsaturated Water Flow Equation: Comparison of Experimental and Computed Results", Water Resources Research, Vol.9, No. 1, pp. 174-177.

Greenberg, A. E., Clesceri, L. S., and Eaton, A. D. (1992). Standard Methods for the Examination of Water and Wastewater, 18th Edition, American Public Health Association.

Guymon, G. L. and Hromadka, T. V. (1986). "Two-Dimensional Diffusion-Probabilistic Model of a Slow Dam Break", Water Resources Bulletin, Vol. 22, No. 2, pp. 257-265.

Ham, R. K. (1980). "Decomposition of Residential and Light Commercial Solid Waste in Test Lysimeters", EPA-SW-190c, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Hanks, R. J., and Bowers, S. A. (1962). "Numerical Solution of the Moisture Flow Equation for Infiltration into Layered Soils", Soil Science Society Proceedings, pp.530-534.

Hanks, R. J., Klute, A., and Bresler, E. (1969). "A Numeric Method for Estimating Infiltration, Redistribution, Drainage, and Evaporation of Water from Soil", Water Resources Research, Vol. 5, No. 5, pp. 1064-1069.

Hartz, K. E. (1980). "Studies of Methanogenesis in Samples from Landfills", thesis presented to the University of Wisconsin, Madison, in partial fulfillment of the requirements for the degree of Doctor of Philosophy.

Hartz, K. E., Klink, R. E., and Ham, R. K. (1982). "Temperature Effects: Methane Generation from Landfill Samples", Journal of the Environmental Engineering Division, ASCE, Vol. 108, No. EE4, pp. 629-638.

Hromadka, T. V., Berenbrock, C. E., Freckleton, J. R., and Guymon, G. L. (1985). "A Two-Dimensional Dam-Break Flood Plain Model", Advanced Water Resources, Vol. 8, pp. 7-14.

Hromadka, T. V. and Durbin, T. J. (1986). "Two-Dimensional Dam-Break Flood-Flow Analysis for Orange County Reservoir", Water Resources Bulletin, Vol. 22, No. 2, pp. 249-255.

Incropera, F. P. and DeWitt, D. P. (1990). Fundamentals of Heat and Mass Transfer, Third Edition, John Wiley & Sons, New York.

Jayawickrama, P. W., Brown, K. W., Thomas, J. C., and Lytton, R. L. (1988). "Leakage Rates Through Flaws in Membrane Liners", Journal of the Environmental Engineering Division, ASCE, Vol. 114, No. 6, pp. 1401-1420.

Jennings, A. A., Kirkner, D. J., Theis, T. L. (1982). "Multicomponent Equilibrium Chemistry in Groundwater Quality Models", Water Resources Research, Vol. 18, No. 4, pp. 1089-1096.

Kmet, P. (1982). "Environmental Protection Agency's 1975 Water Balance Method-its uses and limitations", Guidance Report, Bureau of Solid Waste Management, Wisconsin Department of Natural Resources, Madison, Wisconsin.

Korfiatis, A. M., and Demetracopoulos, A. M. (1986). "Flow Characteristics of Landfill Leachate Collection Systems and Liners", Journal of the Environmental Engineering Division, ASCE, Vol. 112, No. 3, pp. 538-550.

Leckie, J. O. (1974). "Estimation of Potential Methane Production from Solid Waste Landfills", unpublished report for EMCON Associates, September, 1974.

Lentz, J. J. (1981). "Apportionment of net recharge in landfill covering layer into separated components of vertical leakage and horizontal seepage", Water Resources Research, Vol. 17, No. 4, pp. 1231-1234.

Lu, J. C. S., Morrison, R. D., and Stearns, R. J. (1981). "Leachate Production and Management from Municipal Landfills: Summary and Assessment",Proceedings on Seventh Annual Research Symposium, Land Disposal: Municipal Solid Waste, EPA-600/9-81-002a, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Mahmood, R. J., and Sims, R. C. (1986). "Mobility of Organics in Land Treatment Systems", Journal of the Environmental Engineering Division, ASCE, Vol. 112, No. 2, pp. 236-245.

McCarty, P. L. (1964). "Anaerobic Waste Treatment Fundamentals", Public Works, Vol. 95, No. 9-12.

McEnroe, B. M. (1989a). "Drainage of Landfill Covers and Bottom Liners: Unsteady Case", Journal of the Environmental Engineering Division, ASCE, Vol. 115, No. 6, pp. 1103-1113.

McEnroe, B. M. (1989b). "Drainage of Landfill Covers and Bottom Liners", Journal of the Environmental Engineering Division, ASCE, Vol. 115, No. 6, pp. 1114-1122.

McEnroe, B. M., and Schroeder, P. R. (1988). "Leachate Collection in Landfills: Steady Case", Journal of the Environmental Engineering Division, ASCE, Vol. 114, No. 5, pp. 1052-1062.

185

McGinley, P. M., and Kmet, P. (1984). "Formation, Characteristics, Treatment and Disposal of Leachate from Municipal Solid Waste Landfills", Special Report, Bureau of Solid Waste Management, Wisconsin Department of Natural Resources, Madison, Wisconsin.

Merz, R. C. (1964). "Investigation to Determine the Quantity and Quality of Gases Produced During Refuse Decomposition," University of Southern California to State Water Control Board, Sacramento, California, 1964.

Merz, R. C. and Stone, R. (1970). "Special Studies of a Sanitary Landfill," United States Public Health Service, Bureau of Solid Waste Management Report EPA-SW 8R6-70.

Mohsen, F. N. M. (1975). "Gas Migration from Sanitary Landfills and Associated Problems", Ph.D. Thesis, University of Waterloo, Waterloo, Ontario, Canada.

Mohsen, F. N. M., Farquhar, G. J., and Kouwen, N. (1977). "Modeling Methane Migration in Soil", Journal of the Environmental Engineering Division, ASCE, submitted for publication.

Monod, J. (1950). "Technique de Culture Continue: Theories et Applications", Annals Institute Pasteur, 79, pp. 390-410.

Moore, C. A. (1950). "Theoretical Approach to Gas Movement Through Soils", Progress Report on EPA Contract No. 68-03-0326.

Nair, S., Longwell, D., and Seigneur, C. (1990). "Simulation of Chemical Transport in Unsaturated Soil", Journal of the Environmental Engineering Division, ASCE, Vol. 116, No. 2, pp. 214.

Owens, I. S., and Khera, R. P. (1990). Geotechnology of Waste Management, Butterworths, London.

Peer, R. L., Epperson, D. L., Campbell, D. L., von Brook, P. (1992). "Development of an Empirical Model of Methane Emissions from Landfills", EPA-600/R-92-037, Research Triangle Park, North Carolina.

Perrier, E. R., and Gibson, A. C. (1982). "Hydraulic Simulation on Solid Waste Disposal Sites," USEPA Document SW-868.

Peyton, R. L., and Schroeder, P. R. (1988). "Field Verification of HELP Model for Landfills", Journal of the Environmental Engineering Division, ASCE, Vol. 114. No.2, pp.247-269.

Peyton, R. L., and Schroeder, P. R. (1990). "Evaluation of Landfill-Liner Designs", Journal of the Environmental Engineering Division, ASCE, Vol. 116. No.3, pp.421-437.

Pfeffer, J. T. (1973). "Reclamation of Energy from Organic Refuse," Grant No. EPA-R-800776.

186

Pfeffer, J. T. (1974). Reclamation of Energy from Organic Wastes, Environmental Protection Technology Series, EPA-670/2-74-016.

Ramaswamy, J. N. (1970). "Nutritional Effects on Acid and Gas Production in Sanitary Landfills", thesis presented to West Virginia University, at Morgantown, West Virginia, in 1970, in partial fulfillment of the requirements for the degree of Doctor of Philosophy.

Rich, L. G. (1963). Unit Processes of Sanitary Engineering, Wiley, New York.

Richards, L. A. (1931). "Capillary Conduction of Liquids through Porous Mediums," Physics, Vol. 1, No. 5, pp. 318-332.

Ritchie, J. T. (1972). "A Model for Predicting Evaporation from Row Crop with Incomplete Cover", Water Resources Research, Vol. 8, No. 5, pp. 1204-1213.

Rovers, F. A., and Farquhar, G. J. (1973). "Infiltration and Landfill Behavior," Journal of the Environmental Engineering Division, ASCE, Vol. 99, No. EE5, Proc. Paper 10047, pp. 671-690.

Rubin, J. (1983). "Transport of Reacting Solutes in Porous Media: Relation between Mathematical Nature of Problem Formulation and Chemical Nature of Reactions", Water Resources Research, Vol. 19, No. 5, pp. 1231-1252.

Rubin, J. and James, R. V. (1973). "Dispersion-affected Transport of Reacting Solutes in Saturated Porous Media: Galerkin Method Applied to Equilibrium-Controlled Exchange in Unidirectional Steady Water Flow", Water Resources Research, Vol. 9, No. 5, pp. 1332-1356.

Russo, D. (1988). "Determining Soil Hydraulic Properties by Parameter Estimation: On the Selection of a Model for the Hydraulic Properties", Water Resources Research, Vol.24, No.3, pp. 453-459.

Russo, D. (1991). "Stochastic Analysis of Simulated Vadose Zone Solute Transport in a Vertical Cross Section of Heterogeneous Soil During Nonsteady Water Flow", Water Resources Research, Vol. 27, No. 3, pp. 267-283.

Russo, D. (1992). "Upscaling of Hydraulic Conductivity in Partially Saturated Heterogeneous Porous Formation", Water Resources Research, Vol. 28, No. 2, pp. 397-409.

Schroeder, P. R., Gibson, A. C., and Smolen, M. D. (1984). "The Hydrologic Evaluation of Landfill Performance (HELP) Model. Volume II, Documentation for Version 1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Schroeder, P. R., Morgan, J. M., Walski, T. M., and Gibson, A. C. (1983a). "The Hydrologic Evaluation of Landfill Performance (HELP) Model, Volume I. User's Guide for Version 1, EPA/530-SW-84-009.

Schroeder, P. R., Morgan, J. M., Walski, T. M., and Gibson, A. C. (1983b). "The Hydrologic Evaluation of Landfill Performance (HELP) Model, Volume II. Documentation for Version 1, EPA/530-SW-84-010.

Schuyler, R. (1973). "Energy Recovery at the Landfill", Seminar of the Governmental Refuse Collection and Disposal Association, Santa Cruz, California.

Smith, R. E., and Woolhiser, D. A. (1971). "Overland Flow on an Infiltrating Surface", Water Resources Research, Vol. 7, No. 4, pp. 899-913.

Stephenson, D. and Meadows, M. E. (1986). Kinematic Hydrology and Modeling, Elsevier, Amsterdam.

Tchobanoglous, G., Theisen, H., and Eliassen, R. (1977). Solid Wastes: Engineering Principles and Management Issues, McGraw-Hill Book Company, New York.

Tchobanoglous, G., Theisen, H., Vigil, S. (1993). Integrated Solid Waste Management: Engineering Principles and Management Issues, McGraw-Hill, Inc. New York, 1993.

Valocchi, A. J., Street, R. L., and Roberts, P. V. (1981). "Transport of Ion Exchanging Solutes in Groundwater: Chromatographic Theory and Field Simulation", Water Resources Research, Vol. 17, No. 5, pp. 1517-1527.

Vesilind, P. A., Peirce, J. J., and Weiner, R. F. (1994).Environmental Engineering, Third Edition, Butterworth-Heinemann, Boston, 1994.

Whisler, F. D., and Watson, K. K. (1968). "One-Dimensional Gravity Drainage of Uniform Columns of Porous Materials", Journal of Hydrology, Vol. 6, pp. 277-296.

Whisler, F. D., and Watson, K. K. (1969). "Analysis of Infiltration into Draining Porous Media", Journal of the Irrigation and Drainage Division, ASCE, Vol. 95, No. IR4, pp. 481-491.

Wigh, R. J. (1979). "Boone County Field Site Interim Report, Test Cells 2A, 2B, 2C, and 2D", EPA-600/2-79-058, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Wigh, R. J. (1984). "Landfill Research at the Boone County Field Site", EPA-600/2-84-050, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Wigh, R. J. and Brunner, D. R. (1981). "Summary of Landfill Research, Boone County Field Site", Proceedings on Seventh Annual Research Symposium, EPA-600/9-81-002a, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Wise, D. L., Sadek, S. E., Kispert, R. G., Anderson, L. O., and Walker, D. H. (1975). "Fuel Gas Production from Solid Waste", Biotechnology and Bioengineering Symposium, No. 5, pp. 285-301, John Wiley & Sons, Inc.

Wong, J. (1977). "The design of a system for collecting leachate from a lined landfill site," Water Resources Research, Vol. 13, No. 2, pp. 404-409.

Young, A. (1989). "Mathematical Modeling of Landfill Gas Extraction", Journal of Environmental Engineering, ASCE, Vol. 115, No. 6, pp. 1073-1087.

APPENDIX A

EMCON (1980) describes the Palos Verdes Kinetic Model as a two-stage, first order mathematical model which assumes the first stage gas production rate is proportional to the volume of the gas already produced. Thus the gas production rate in the first stage is increasing exponentially with time and is shown as:

$$\frac{dG}{dt} = K_1 G \tag{A1}$$

where t is time, G is the volume of gas produced prior to time t, and K_1 is the first stage gas production rate constant. The first stage model assumes an ultimate gas production amount. When half the ultimate gas production is reached in the first stage, the model transitions to the second stage kinetics which are represented by:

$$\frac{dL}{dt} = -K_2 L \tag{A2}$$

where L is the volume of gas remaining to be produced after time t, and K_2 is the second stage gas production rate constant. It is seen that the rate of gas production is an inverse exponential function. The model breaks wastes into three main categories: readily decomposable (food and grass); moderately decomposable wastes (paper, wood, and textiles); and refractory wastes (plastic and rubber). Values for K_1 and K_2 for each main group and methods to develop them

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can be found in EMCON (1980). Graphs of the main categories and a composite of the rate of gas production vs. time and cumulative total gas production vs. time are shown in Figure A1.

The Sheldon Arleta Kinetic Model is also a two stage, first-order model similar to the Palos Verdes model. However, it is based on an article by Fair and Moore (1932) related to anaerobic digestion of sewage sludge and on a master's thesis by Alpern (1974). This assumes that the peak gas production rate occurs at 35% of the time required to complete 99% of the gasification. The model also classifies waste into two categories; readily decomposable and relatively slowly decomposing materials and it takes placement time into account separating each category into years. Hence, for a twelve year span, the two groups would yield 24 separate groups to be summed up for total gas production. Constants and methods for the use of the Sheldon Arleta model are given in EMCON (1980).

EMCON (1980) further reports that these two models do not take moisture content, mass transport, unfavorable pH, or oxygen intrusion into account as a possible rate limiting factor. It is also believed that maximum gas production rates in a landfill occur at times substantially shorter than those predicted by the models. In sewage sludge digesters under optimal conditions, the microbial mass may expand geometrically until substrate concentration drops below some critical level. This is not true with a landfill where many factors may limit biomass growth before the amount of remaining organic matter becomes limiting. It is believed that the maximum production rate in a landfill would occur quickly (less than 40% of the maximum
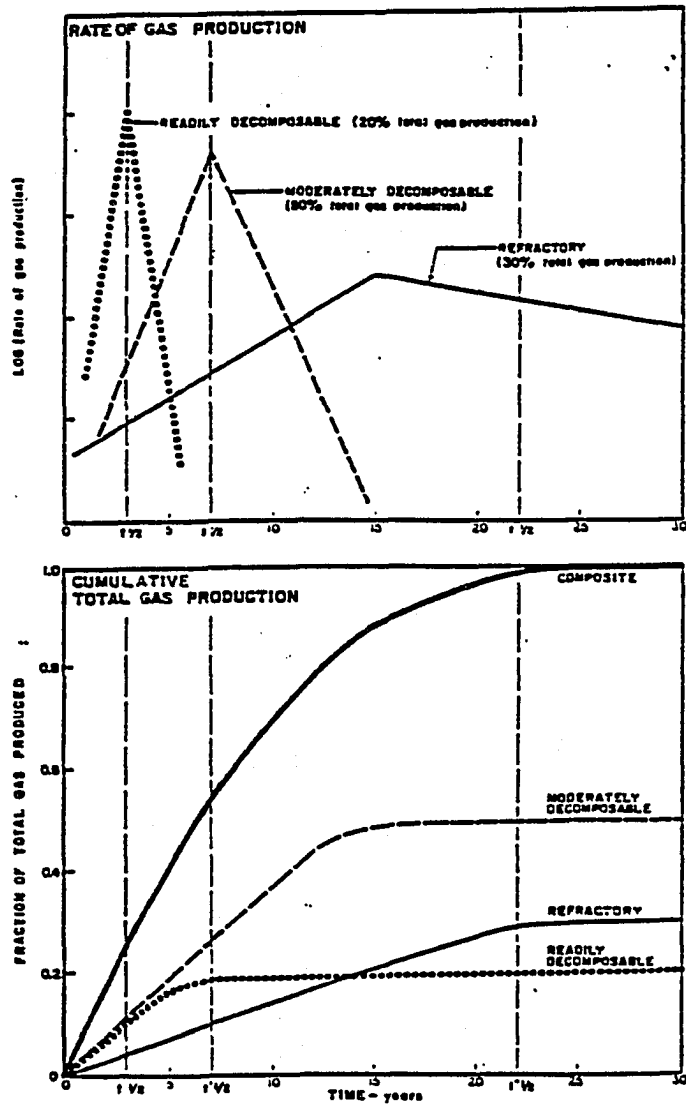
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Figure A1. Rate of gas production and cumulative total gas
production (Palos Verdes Kinetic Model)
(After EMCON 1980).

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yield has been attained), followed by a relatively slow decrease in the production rate. This is in contrast to the Sheldon Arleta model which predicts maximum production rate to occur after 30 years.

A third model, the Scholl Canyon Kinetic Model, is a single-stage, first order kinetic model which assumes that gas production reaches a maximum after a negligible lag time time during which anaerobic conditions are established and microbial biomass is built up and stabilized. The rate then decreases as the organic fraction of the landfill refuse diminishes. Hence, the landfill is able to support the decreasing biomass. Growth is limited by substrate and is analogous to the reduction of biochemical oxygen demand (BOD). This substrate-limited microbial growth is described by:

$$\frac{dL}{dt} = - kL \qquad \left(\frac{\text{volume of methane}}{\text{mass of refuse-time}}\right) \qquad \text{(A3)}$$

where t is time, L is the volume of methane remaining to be produced after time, t, per mass of refuse, and K is the gas production rate constant. The methane production rate is $-\frac{dL}{dt}$, or kL, with typical units L/kg/yr. The refuse mass can be broken down into sub-masses which are placed in the landfill each year. Each sub-mass can be designated by a subscript "i" and the methane production rate equation can be written:

$$kL = \sum_{i=1}^{n} r_i k_i L_{o_i} e^{-k_i t_i} \qquad \text{(A4)}$$

where n = number of sub-masses considered

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r_i = fraction of total refuse mass contained in sub-mass i

t_i = time from placement of sub-mass i to point in time at which composite production rate is desired

k_i = gas production constant for sub-mass i

L = volume of methane remaining to be produced after time t

k = gas production rate constant

L_{o_i} = total volume of methane to be produced for the sub-mass

Through experimentation, the composite methane production rate can be estimated. When combined with placement times, t_i, and fractions of total refuse, r_i, the rate constant, k, can be determined through trial and error for different estimations of the total volumes, L_o, of methane. Figure A2 shows the range of methane generation rates for different assumed values of L_o (EMCON 1980). Similar to the previous models, there are other factors which can be rate limiting in a landfill. However, these simple models are currently being used for the methane production rates in landfills until the measurements and technology develop a better tool.

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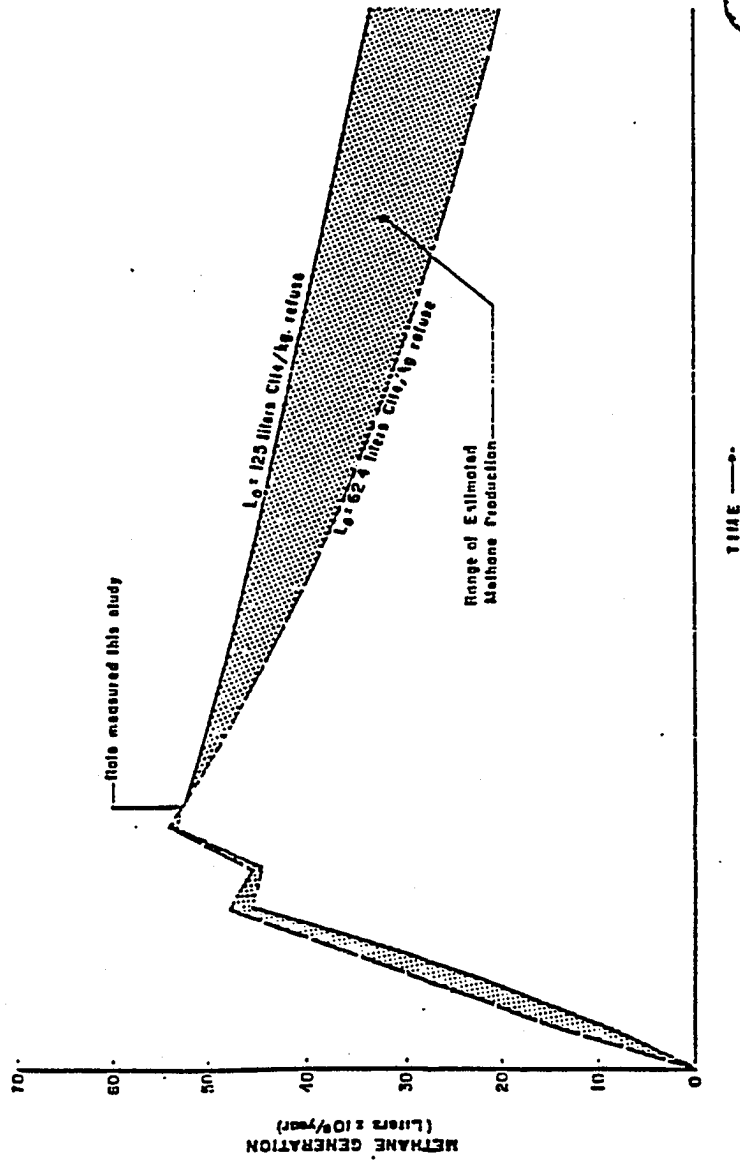
Figure A2. Estimated methane production (Scholl Canyon Kinematic Model) (After EMCON 1980).

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APPENDIX B

INPUT DATA FILES FORMAT

The values for various input data are read by the code through two input data files which are accessed during a code initialization subroutine. They are set up in a rigid order which must be adhered to for the proper operation of the code. The data variable and spacing are critical.

The code is run by typing the code name "BADF" at the DOS prompt. It immediately asks the user to "ENTER THE NAME OF THE INPUT FILE", which is specified by the user. The input file name should be no more than 8 characters plus the three file identifier characters. This will generate an output file with the same name, but uses an OUT file identifier. For example, if the file DATA1.INP is entered, the corresponding output file will be found in the current directory under DATA1.OUT. The code also builds the following other output files: LCONTAM.OUT, LEACH.OUT, LGAS.OUT, LINER.OUT, LPLOT.OUT, and LTEMP.OUT. These files are for corresponding data output which deals with the name of the file. The "L" was added to some of the names in order to keep the output files together during a file listing.

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INPUT VARIABLES FORMAT

This section describes the input required to run the simulation. The data is entered in the INPUT file in a rigid order.

Line 1 - Title of Simulation (A80)

HEADER1 = descriptive title of the simulation (A80).

Line 2 - Description of the simulation (A80)

HEADER2 = description of landfill simulation (A80).

Line 3 - Line 4 header name (A80)

HEADER3 = names of Line 4 variables for user convenience (A80).

Line 4 - Landfill simulation control parameters (1F10.0, 5I10, 1F10.0)

thr = simulation time step, hrs (F10.0).

nstart = starting time index (I10).

nstop = stopping time index for the simulation. Note, if the
     simulation time step, thr, is 0.5 hours and nstart is 1 and
     nstop is 100, the simulation will run for 50 hours (I10).

nt = a multiplying factor for the time step, usually set at 1 (I10).

nwrite = time step interval to write to the output files (I10).

jstop = index for the bottom of the landfill (I10).

cm = factor for manning equation depending on system of units used
(1 for S.I. and 1.49 for English) (F10.0).

Line 5 - Line 6 header name describing landfill physical sizes (A80)

HEADER4 = names of Line 6 variables (A80).

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Line 6 - Landfill simulation control parameters (1I10, 3F10.0, 1I10)

nr = number of rows in the landfill (I10).

nc = number of columns in the landfill (I10).

dz = height of each vertical element, meters (F10.0).

lambda = contaminant dispersion coefficient, meters (F10.0).

recirc = percentage of leachate to be recirculated (F10.0).

nosys = number of contaminant systems to be simulated (I10).

Line 7 - Line 8 header name describing soil and dispersion data (A80)

HEADER5 = names of Line 8 variables, (A80).

Line 8 - Landfill simulation control parameters (8F10.0)

caa = dispersion/diffusion empirical constant (a) of Section 3.5.3 characterizing the waste material, (F10.0).

cbb = dispersion/diffusion empirical constant (b) of Section 3.5.3 characterizing the waste material, (F10.0)

cdo = equivalent diffusion coefficient in free water, m^2/hr (F10.0).

satm = minimum degree of moisture saturation of waste (F10.0).

kkrm = minimum permeability of the waste (F10.0).

dsdpm = minimum gradient of soil saturation for satm (F10.0).

ttop = thickness of the top cover soil (F10.0).

Line 9 - Description of data for individual vertical cells (A80)

HEADER6 = description of landfill simulation (A80).

Line 10 - Line 11 header describing waste characteristics (A80)

HEADER7 = names of Line 11 variables, (A80).

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Line 11 - Landfill simulation control parameters (2I5, 5F10.0, 1E10.0

1F10.0)

Note: The model determines the number of vertical cells by multiplying the number of rows, nr, by the number of columns, nc. Line 11 will have to be repeated nr x nc times with soil entries for each vertical cell (Example: If there are 8 rows and 12 columns, line 11 will be written 96 times).

nrow = index to identify row number for data (I5).

ncol = index to identify column number for data (I5).

alf(nrow,ncol) = soil parameter, α, in Section 3.2 (F10.0).

xm(nrow,ncol) = soil parameter, m, in Section 3.2 (F10.0).

$S_{w_{max}}$(nrow,ncol) = maximum degree of moisture saturation (F10.0).

$S_{w_{min}}$(nrow,ncol) = minimum degree of moisture saturation (F10.0).

por(nrow,ncol) = waste porosity (F10.0).

Ks(nrow,ncol) = saturated permeability of the waste, m/hr (E10.0).

hhi(nrow,ncol) = initial pressure (negative) head for each element, meters (F10.0). Note: The initial pressure head is determined from the initial moisture saturation of the waste. The hhi term is then subtracted from the elevation of each element to determine the initial total head of each element.

Line 12 - Line 13 header describing vertical cell dimensions (A80)

HEADER8=names of Line 13 variables, (A80).

Line 13 - Landfill simulation control parameters (2I5, 6F10.)

Note: Same note in line 11 applies.

nrow = index to identify row number for data (I5).

ncol = index to identify column number for data (I5).

Htt(nrow,ncol) = elevation of the top landfill cell, meters (F10.0).

Htb(nrow,ncol) = elevation of the bottom of the cell, meters (F10.0).

stime(nrow,ncol) = starting time for the vertical cell, hrs (F10.0).

dwaste(nrow,ncol) = density of the waste, kg/cubic meter (F10.0).

dx(nrow,ncol) = respective x dimension of the cell, meters (F10.0).

dy(nrow,ncol) = respective y dimension of the cell, meters (F10.0).

Line 14 - Line 15 header describing runoff dimensions (A80)

HEADER9 = names of Line 15 variables, (A80).

Line 15 - Landfill simulation control parameters (2I5, 3F10.0)

Note: Same note in line 11 applies.

nrow = index to identify row number for data (I5).

ncol = index to identify column number for data (I5).

elev(nrow,ncol) = elevation of runoff cell surface, meters (F10.0).

rm(nrow,ncol) = manning roughness coefficient of surface (F10.0).

hwt(nrow,ncol) = elevation of the water table for constant head at the bottom simulations, meters (F10.0).

Line 16 - Line 17 header describing runoff dimensions (A80)

HEADER10 = names of Line 17 variables, (A80).

Line 17 - Simulation control parameters (1F10.0, 1I10, 3F10.0)

flagl = liner flag (1=bypass, 0=simulate) (F10).

long = flag for the runoff routine (1=bypass, 0=simulate) (I10).

runtim = time step reduction factor the model can make for stability (F10.0).

fix = Maximum moisture saturation allowed (F10.0).

field = field capacity of the waste. When the element moisture saturation reaches this capacity, percolation out of the element begins (F10.0).

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Line 18 - Line 19 header describing runoff dimensions (A80)

HEADER11 = names of Line 17 variables (A80).

Line 19 - Landfill simulation control parameters (1F10.0, 1E10.0), 2F10.0)

psurf = porosity of the landfill cover (F10.0).

ksurf = saturated permeability of the landfill cover, m/hr (E10.0).

rowf = flag for runoff prevention along the rows (1.0 is flow and 0.0 is no-flow (F10.0).

colf = flag for runoff prevention along the columns (1.0 is flow and 0.0 is no-flow) (F10.0).

Line 20 - Line 21 header describing landfill lateral seepage (A80)

HEADER12 = names of Line 21 variables (A80).

Line 21 - Landfill simulation control parameters (4F10.0)

Note: Seepage flags are allowing flow across the landfill boundaries. If the value is 1.0, seepage is allowed. If the value is 0.0, no seepage is allowed.

clhs = flag allowing seepage out of the left column boundary (F10.0).

crhs = flag allowing seepage out the right column boundary (F10.0).

rlb = flag allowing seepage out of the lower row boundary (F10.0).

rub = flag allowing seepage out of the upper row boundary (F10.0).

Line 22 - Line 23 header describing contaminants (A80)

HEADER13 = identifies contaminant bypass and contaminant group (A80).

Line 23 - Landfill simulation control parameters (4F10.0)

mcby = contaminant bypass flag (1 is bypass, 0 is simulate) (I10).

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Line 24 - Line 25 header describing contaminants (A80)

HEADER14 = identifies contaminant bypass and contaminant group (A80).

Line 25 - Landfill simulation control parameters (4F10.0)

Note: Line 25 will be repeated for each contaminant system, nosys, in Line 6.   Each line will describe a different contaminant system.

czrain = concentration of the contaminant in rain (mg/1) (F10.0).

ultload = total amount of the contaminant which will be produced per mass of dry waste from Section 3.5.5 (mg/kg) (F10.0).

pbc = production rate exponent in Section 3.5.5 (F10.0).

pic = initial concentration of contaminant, mg/1 (F10.0).

Line 26 - Line 27 header describing contaminant system selected (A80)

HEADER15   =   identifies   contaminant   system   (1=zinc,   2=chlorides,

3=other).

Line 27 - Landfill simulation control parameters (1I10)

Note: Line 27 will be repeated nosys times to identify all the contaminant systems to be simulated.   The systems are identified to the code in the computer.   For example, if 1 system is to be simulated and the nsys selected is 2, the simulation runs the routines for chlorides using the parameters identified earlier.

nsys = system identifying number for simulation (I10).

Line 28 - Line 29 header describing liner bypass (A80)

HEADER16 = identifies liner bypass.

Line 29 - Landfill simulation control parameters (2I10)

nlf = number of fields in the liner (I10).

linby = liner bypass (1 is bypass, 0 is simulate) (I10).

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Line 30 - Line 31 header describing liner physical characteristics (A80)

HEADER17 = names of Line 31 variables for user convenience.

Line 31 - Landfill simulation control parameters (5I5, 2F10.0)

Note: Line 31 will be repeated for each liner field identified in Line 29, nlf.

nl = index for liner field (I5).

lrows = number of rows in the field (I5).

lcol = number of waste columns inputting to the in the field (I5).

lheadr = drain number for the field (I5).

nx = number of liner elements along the liner (I5). Note, there may be 5 vertical columns from the landfill which percolate into 30 liner elements along the liner.

sl = slope of the liner (F10.0).

xdist = length of each element (nx) (F10.0).

Line 32 - Line 33 header describing liner soil characteristics (A80)

HEADER18 = names of Line 32 variables for user convenience.

Line 33 - Landfill simulation control parameters (5F10.0)

DKs = saturated permeability of the sand above the liner (F10.0).

BKs = saturated permeability of the liner materials (F10.0).

porlin = porosity of the sand above the liner (F10.0).

sff = liner imperfections factor (F10.0).

tliner = thickness of the liner (F10.0).

Line 34 - Line 35 header describing evapotranspiration (A80)

HEADER19 = names of Line 35 variables for user convenience.

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Line 35 - Landfill simulation control parameters (2F10.0, 1I10)

Ravg = Average radiation flux, langleys (F10.0).

ttavg = average yearly temperature for the area, °C (F10.0).

neby = Evapotranspiration bypass (0 is bypass, 1 is simulate)(I10).

Line 36 - Line 37 header describing monthly conditions (A80)

HEADER20 = names of Line 37 variables for user convenience.

Line 37 - Landfill simulation control parameters (1I10, 2F10.0)

Note: Monthly radiation flux and temperatures entered for each month.

month = index denoting the month (I10).

solm(m) = solar radiation flux for the month, langleys (F10.0).

tmp(m) = monthly average temperature, °C (F10.0).

Line 38 - Line 39 header describing gas production data (A80)

HEADER21 = gas production bypass description.

Line 39 - Landfill simulation control parameters (1I10, 2F10.0)

igby = gas bypass flag (1 is bypass, 2 is simulation) (I10).

Line 40 - Line 41 header describing energy transport data (A80)

HEADER22 = names of Line 41 variables for user convenience.

Line 41 - Landfill simulation control parameters (1I10, 2F10.0)

cwst = specific heat of the waste, $\frac{J}{kg\,°K}$ (F10.0).

ch2o = specific heat of water, $\frac{J}{kg\,°K}$ ((F10.0).

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wkc = conductivity of the waste, $\frac{J}{m^\circ K\, day}$ (F10.0).

tti = initial temperature of the waste, $^\circ$C (F10.0).

heat = heat of reaction from creation of methane $\frac{cal}{mole}$ (F10.0).

juds = Julian date for the start of the simulation to correspond to atmospheric conditions (I10).

Line 42 - Line 43 header describing gas production data (A80)

HEADER23 = names of Line 43 variables for user convenience.

Line 41 - Landfill simulation control parameters (4F10.0, 1I10)

kgas = reference gas production reaction rate factor for reference temperature, t1, as discussed in Section 3.6.1 (F10.0).

ultgas = ultimate gas production per mass of waste (F10.0).

Ea = Activation energy for methane production, $\frac{kcal}{mole}$, Section 3.6.1 (F10.0).

t1 = reference temperature for the gas reaction coefficient (F10.0).

agas = empirical gas constant for gas production (F10.0).

tgi = initial amount of gas previously produced at start (F10.0).

igas = time of gas production (I10).

If errors are detected in reading the input file, it will stop the simulation and inform the user on which line the input error was found.

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PRECIPITATION

This section describes the precipitation input for the model. This
is entered in a different data file known as cell.inp.

Line 1 - Number of data lines to read (1I10)

infts = number of precipitation data input files to read (I10).

Line 2 - Precipitation simulation values (I12, 1F16.0, 1E16.0)

Note: Line 2 is read infts times to get all the precipitation data.

mo = month of the rain (I12).

tistrt(infts) = time rain rate began, hr (F16.0).

inf(infts) = rainfall rate m/hr (E16.0).

```
AFLOW INPUT FILE FOR LEACHATE LANDFILL INPUT
TYPE OF LANDFILL   HAM LYSIMETER # 2  (ONE COLUMN)
   Tmin  *  nstart *  nstop *     nt  *  nwrite *  jstop *  Cm   *
     .5         1     98885         1      1440        15    1.486
Num row *  Num col*    dz  *  lambda *  recirc *  nosys *       *
      1          1       0.2    0.0039       0.0        1
Dspers a* Dspers b* Dspers i* sat min *  Kr min * dsdp min* top thck*
   0.002      10.0  0.000000      0.20   .0000147     0.16      0.15
                Data for Columns
row* col*    alf  *    xm  *    Smx  *  ·Smn  *    por  *    Ks  *    hh  *
  1   1       5.3       2.0       1.0      0.05      0.52   7.2E-03       .8
row* col*    Htt  *    Htb  *   time  * density *   dx  *    dy  *
  1   1       4.3       3.1       0.0      445.     9.144   18.288
row* col*  elevat * roughns *   hwt  *
  1   1       4.52      0.04       3.1
linr flg*   long  * runtim * sssmax * fld cap *
    1.0        1      1.00     0.95      0.97
surf por* surf Ks * row flg * col flg *     *         *         *         *
    .46   1.40E-2      0.0       0.0
seep lhs* seep rhs* seep low* seep up *            Seepage out sides
    0.0       0.0       0.0       0.0
      Contaminant Bypass     ( 1 is bypass)          CONTAMINANTS
          0
Rain Con* Ult load* Disp exp* Int conc*      (COD,Chlorides)
  0.00000      100.       .35      300.
      0.0     1500.       .45      800.
Systems *    (1=COD,  2=Chlorides )  n=1,nsys
      2
num linrs  liner bypass (1 is bypass)                 LINER DATA
      2            1
nlf * rows cols drns nx *  slope  * distance*
  1   1    2   1    4       0.03      9.144
  2   1    2   1    4       0.03      9.144
Sand Ks * Liner Ks* porosity*    SF   * tliner *
    1.81   .0000181     0.25      0.0      2.0
Avg Rad * Avg Temp* Ev bypas*                    EVAPOTRANSPIRATION
   330.0       15.          0
Month   *  Rad flx*  Temp  *
      1       330.      273.
      2       330.      276.
      3       350.      280.
      4       330.      285.
      5       330.      286.
      6       330.      295.
      7       330.      300.
      8       330.      295.
      9       330.      290.
     10       330.      285.
     11       330.      280.
     12       330.      273.
     gas bypass  ( 1 is bypass)         GAS PRODUCTION AND TEMPERATURE
       1
Cp waste*  Cp H2O *  Kc wast* Init Tmp* Heat Rxn* Jul dat *     Energy Transpo
  1116.0     4182.    420000.     300.    31000.       258
Ref  K1 * Max gas * Activ En* Ref Temp* Ref Agas* Init gas*  ngas  *     Gas
    3.0      .319       20.      293.     1.035       0.0       1
```

Figure A3. Sample of an input data file.

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