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ENVIRONMENTAL EFFECTS OF LANDFILLS

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ABSTRACT

In the past, landfilling involved burying municipal refuse directly or after on-site burning. Typically, little attention was given to proper siting and engineering to obviate the hazards of the generation of CH_4 and toxic leachates as wastes decomposed. Leachates were hopefully attenuated by natural processes (adsorption, precipitation, ion exchange, microbial decomposition or dilution in the unsaturated zone below landfills). Landfills slowly evolved by proper siting, design and management into efficiently operated bioreactors to produce purified CH_4 for use as a fuel, and leachates, which were treated biologically and chemically to minimize groundwater pollution. Microbial reactions in landfills are outlined. The amounts and composition of landfill gas and leachate as determined by the interaction of factors such as refuse composition, degree of compaction, temperature, moisture content, refuse age and depth are discussed. Typical inorganic and organic composition of landfill gases and leachates are presented.

The potential and real environmental effects on soils, plants, groundwater, aquatic organisms and humans of disposal of municipal refuse by landfilling are reviewed. Finally, the most recent trend in constructing refuse landfills to serve as final storage reservoirs which are deliberately kept dry to minimize gas and leachate production is discussed and illustrated. Present activities in waste recycling to conserve landfill space are outlined.

INTRODUCTION

It is estimated that approximately 200 million metric tons (t) of municipal refuse are produced annually in the United States. Approximately 20% or 40 million tons cannot be incinerated, including construction debris, bricks, stones, concrete, old appliances and other non-combustible materials. Of the approximately 160 million tons of refuse that will burn, only about 10%, or 16 million tons, are incinerated. If it was all incinerated, it would yield about 45 million tons of incinerator residue (fly ash and bottom ash) to be disposed (Wallgren, 1987). Thus, even though the proportions of refuse that are being incinerated or recycled are steadily increasing, most is still disposed in landfills.

REFUSE COMPOSITION

The approximate composition of the combustible fraction of refuse in the United States is listed in Table 1. The data were adapted from those of Campbell

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TABLE 1

Approximate composition of the combustible fraction of municipal refuse in the United States

Component	Composition (%)	Weight (%)
Paper	50.2	
Cardboard (corrugated)		26.7
Newspaper		18.0
Other, paperboard		13.3
Paper packaging		12.6
Office paper		12.2
Magazines, books		7.6
Tissue paper, towels		5.3
Other, nonpackaging		2.9
Paper plates, cups		1.4
Yard wastes	17.8	
Food wastes	16.8	
Plastics (packaging)	5.2	
Polyethylene (low density)		32
Polyethylene (high density)		31
Polystyrene		11
Polypropylene		10
Polyethylene terephthalate		7
Polyvinylchloride		5
Other		4
Wood	4.6	
Leather, rubber	3.4	
Textiles	2.0	

(1976) and Thayer (1989). It is obvious that the true quantities of combustible refuse will vary greatly with location and time. For instance, refuse from households with garbage disposal units will contain less food wastes. The quantity of yard wastes will vary with season and whether the location is urban or rural (Campbell, 1976). Also, the fraction of food wastes in refuse has been reported to decrease as the fraction of packaging wastes (paper, plastic, glass, metals) increases, owing presumably to reduced food spoilage (Alter, 1989). A

TABLE 2

Approximate composition of the noncombustible fraction of refuse in the United States

Component	Weight (%)
Metals	
Ferrous	25.08
Aluminum	3.63
Zinc-base	0.20
Copper-base	0.46
Glass, ceramics	34.65
Other	35.98

significant component under the category of "other" for paper and plastics would be disposable diapers, 18 billion of which are estimated to be disposed in the United States annually. Other components not listed in Table 1 would include sewage sludge, animal carcasses and many others.

The estimated composition of the noncombustible fraction of refuse was adapted from data of Law and Gordon (1979) and is given in Table 2. Again, these figures will vary with location, industrial diversity, levels of production and other factors. The category designated "other" would include ash from refuse and sewage sludge incinerators, dirt and numerous miscellaneous materials.

EVOLUTION OF THE LANDFILL

For several years, the philosophy of landfilling involved daily additions of refuse, each covered by soil at the time of dumping. Rainfall was allowed to enter the landfill and its location was hopefully chosen to promote cleansing the resultant leachate by passage through adsorptive clayey materials below. The landfill was therefore typically wet throughout, microbiologic decomposition of refuse was active and the CH_4 gas produced was sometimes collected for use as a fuel. More recently, elaborate efforts are being made to line as well as cover landfill areas with impervious and continuous synthetic materials to keep the moisture that is inherently in the refuse in and precipitation out so as to obviate leachate problems. Let us begin by discussing approaches that were used to site landfills that were to be kept moist and biologically active and how specific management practices (deliberate addition of moisture, refuse compaction, etc.) affected refuse decomposition and could be used to promote CH_4 production. Many of these siting strategies still apply today in determining suitable locations for "dry" landfills.

SITING LANDFILLS

Obviously, locating a suitable site for a refuse landfill should ideally meet certain general criteria such as proper zoning, central location, not immediately adjacent to residential areas and hopefully having some aesthetic or economic (park, recreational area) value when finally closed and landscaped. From a technical standpoint, the site must be accessible to refuse-carrying trucks throughout the year, have an adequate amount of earth cover which can be handled and compacted daily, and be of sufficient area to accommodate community wastes for a reasonable time and possibly to permit location of recycling facilities there. Most importantly, it must be topographically, geologically and hydrologically suitable and have appropriate soil to safeguard against potential surface and groundwater pollution by landfill leachate (the brown-colored liquid which exudes from refuse landfills) and uncontrolled gas (CH_4, H_2S) movement (O'Leary et al., 1986a).

In the past when pollutants in landfill leachates were to be naturally attenuated by passage through earth material, site selection based on the latter four geocriteria was essential. Emrich (1972) provides examples of the use of these criteria in siting refuse landfills. Landfills located on valley bottoms or flood plains will normally have moderate to deep soils, be near surface water and subjected periodically to flooding. Groundwater will be shallow. Grades will be low permitting easy access for handling equipment. Landfills located on valley walls will have shallow to moderate soil depth with shallow to moderate depth to groundwater. In such locations, steeper slopes may restrict some handling equipment. Surface water could usually be diverted by ditching. Landfills sited in ravines on valley walls may lack suitable cover material. Shallow groundwater may discharge at the ravine bottom during wet weather, moving up through the refuse yielding leachate discharge from the toe of the landfill. Landfills located on uplands or topographic divides would typically have moderate to deep soils and groundwater and surface water problems would be reduced. Bedrock near the surface in upland locations can create difficulties, however, when it must be ripped out.

Groundwater moves through bedrock, which is usually the parent material of soils. The response of bedrock to erosive processes will dictate the topography of an area. Bedrock should be evaluated geologically as regards type of rock, porosity and permeability, structure, weathering, depth to bedrock and the presence or absence of glacial materials and their texture. The most weather-resistant rocks will be very dense and comprise topographic highs having only a thin soil cover. They will show low porosity and permeability and such areas will be unsuitable for landfills. Rocks in valley bottoms are typically more readily weathered with moderate to deep soils, shallow to moderately deep groundwater and showing moderate to high porosity and permeability. Such locations are marginal for siting landfills. Rocks intermediate between the latter two types are found on valley walls and in upland areas and are best for locating landfills when the leachate is to be naturally renovated.

Thus, natural attenuation of pollutants in landfill leachate best occurs by passage through deeply weathered rocks or thick soils that are reasonably well drained. Highly permeable earth materials can simply result in mass movement of pollutants downward into groundwater. If the permeability of the earth material is too low, leachate may collect in the bottom of the refuse and eventually discharge laterally to the surface. Stief (1989) has tabulated the properties and requirements for underlying rock strata in regard to pollutant retardation capacity.

In siting landfills, hydrology must take into account both groundwater and surface water. As pointed out by Emrich (1972) regarding surface water, it is essential to determine its location and direction of drainage, flooding frequency, water quality and location of ephemeral streams. Concerning groundwater, one must determine water quality, its rate and direction of movement, its discharge points, its depth and the variation of depth with season and all nearby uses of the water.

Leachate will generally not result until the refuse in a landfill reaches field

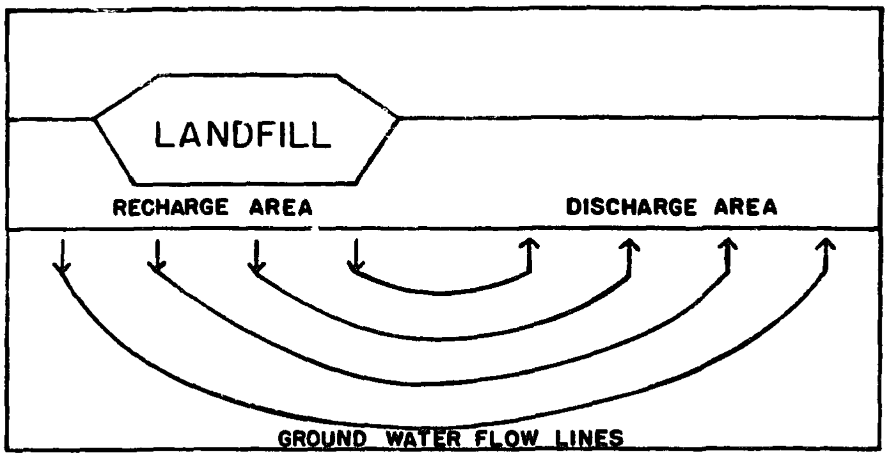

Fig. 1. Groundwater recharge and discharge areas.

capacity. In the Northeast, there is little recharge of landfills by surface water in January and February since the ground is frozen. Recharge is greatest in March to early May due to melting snow, spring rains and low evapo-transpiration. Recharge decreases from late May through August with fewer rains and the emergence of vegetation and higher temperatures thus increasing evapotranspiration. In September through December, recharge again increases as vegetation dies and temperatures decrease (Hughes et al., 1976). Since soil maps describe conditions to only a depth of 5 feet (1.5 m), soil borings to a depth of at least 20 feet (6 m) below the base of the proposed landfill are necessary to assess groundwater conditions. Some of the borings should extend below the water table. These can be converted to monitoring wells to enable groundwater collection and to follow groundwater table fluctuations. Measurement of water levels in the wells will indicate direction of horizontal groundwater (and potential pollutant) flow, i.e. from those having higher levels to those with lower levels. Possible vertical movement of groundwater can be detected by installing multi-level wells. Such vertical movement will characterize the site as a discharge or a recharge area (Fig. 1).

A discharge area has groundwater flowing vertically upward and will typically be saturated. A landfill sited in such a location will have groundwater flowing into it from below which must be removed by the leachate collection system or a groundwater level control system. Leachate from such a landfill will usually be retained in the vicinity of the landfill unless it reaches the surface and leads to runoff. In recharge areas, groundwater is moving down and away from the site and can potentially carry leachate contaminants from a landfill if located there. The area of pollutant migration may therefore be increased. Maps, obtainable from the United States Geological Survey, showing topography, surficial deposits, geologic formations, bedrock depth and type and depth to groundwater are useful. However, since they may cover large areas and therefore show limited detail, performing soil borings is usually recommended (O'Leary et al., 1986a).

Differing geologic conditions will dictate the design and positioning of refuse landfills. Four typical geologic conditions and the associated landfills are shown in Fig. 2 (O'Leary et al., 1986b). In Type A, the main concern is to

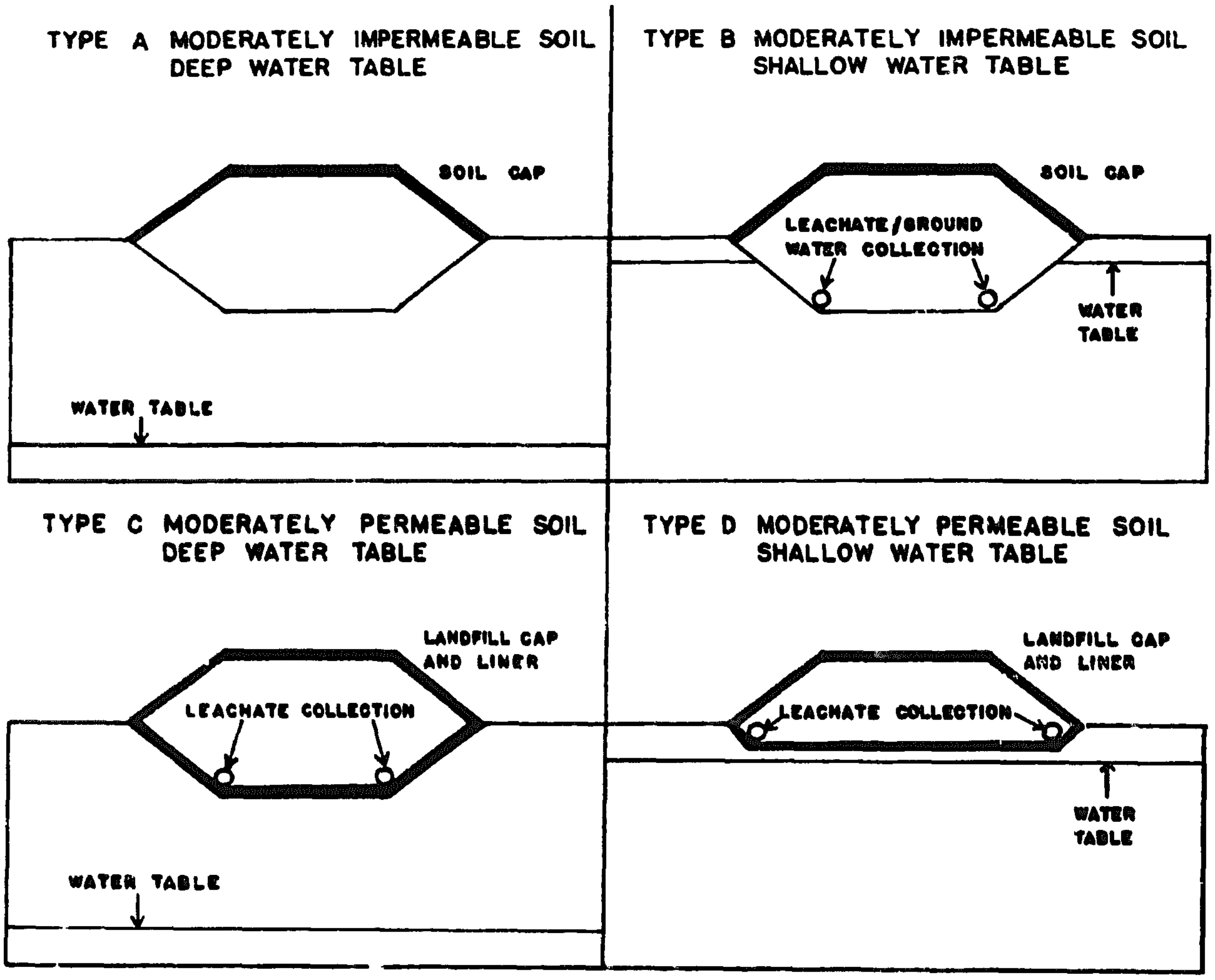

Fig. 2. Landfill designs for several geologic conditions.

prevent excess drainage of leachate. If the existing soil has a high clay content it may be used as the final landfill vegetative cover. The landfill base may have to be recompacted prior to refuse disposal. Installation of drainage pipes may also be necessary. For the conditions in Type B, the landfill may be built above ground as in Type D or a zone of saturation as shown may be constructed in which groundwater is actually induced to flow into the site through drainage pipes whereupon leachate is swept out for collection, treatment and final disposal.

In situation C, restriction of infiltration into the landfill by rainwater or water from other sources is the main consideration. Since the native soils are permeable this must be accomplished by importing clay and perhaps also using flexible membrane liners (FMLs) in the base and cover of the landfill as well as installing a leachate collection system. A more difficult siting problem is shown by Type D. The more permeable soil and shallow water table require that the landfill be constructed almost entirely above the existing ground surface and that more impermeable soil material be imported for the liner and cover.

BIOREACTOR LANDFILLS

As outlined by Aragno (1989), a refuse landfill contains organic materials from countless sources as well as conditions of temperature, moisture, pH, etc.

which are conducive to microbial life. Landfills have been likened to bioreactors in terms of the enzymatically mediated reactions which occur and the resulting metabolic products such as CH_4 which is evolved and may be utilized. Since landfills receive refuse continually over several years, the progressive stages of aerobic, fermentative and methanogenic degradation of refuse are occuring simultaneously, but typically in different but neighboring locales. In this sense, landfills might more appropriately be considered ecosystems with different microbial populations which interact owing to their proximity, contact by the same common water, diffusing metabolic solutes, gases and heat.

Major microbial reactions in landfills

Although the general scheme of landfill reactions can be outlined as in Fig. 3, very little detailed information is available on the intermediate hydrolytic and fermentative reactions (Rees, 1980a). An understanding of the fate of organic compounds in landfills has only been inferred from chemical analysis of the gas and liquids collected. The major end products of refuse fermentation are volatile fatty acids, CO_2 and CH_4. Based on their concentrations in refuse, metabolism of carbohydrates is by far the dominant reaction followed by that of proteins and lipids.

The biodegradability of cellulose, the major carbohydrate found in domestic refuse, will depend on the extent of its prior processing and the degree of

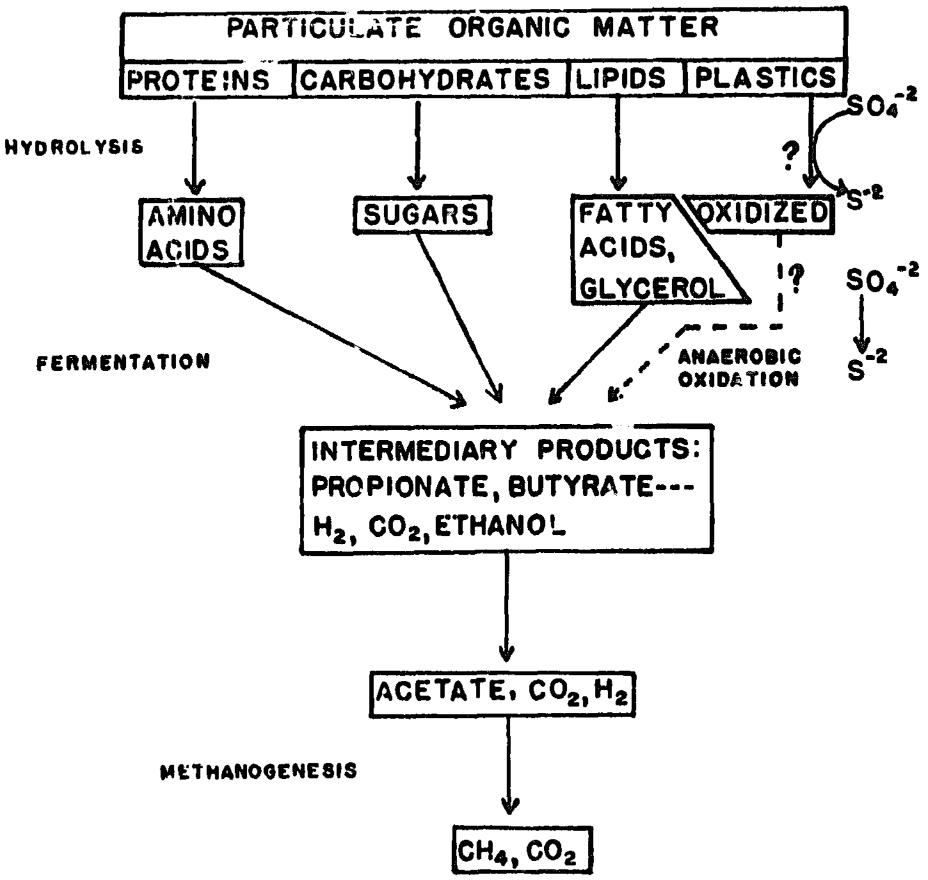

Fig. 3. Proposed degradation scheme for the organic fraction in a refuse landfill.

TABLE 3

Percentages of major organic and inorganic fractions in municipal refuse and domestic sewage sludge

Component	Percent dry weight
Domestic sewage sludge	
Hemicellulose	6.15
Cellulose	34.48
Lipids	14.01
Protein	18.98
Ash	34.88
Municipal refuse (21% moisture)	
Cellulose, sugar, starch	58.8
Lipids	5.7
Protein	2.6
Plastics	1.5
Ash, metals, glass	31.4

crystallinity and liquification. The ratio of cellulose/hemicellulose/lignin probably approximates that of newsprint, i.e. 70:15:15. The biodegradability of cellulose from paper and food wastes is probably similar to that found in sewage sludge in which the primary structure of cellulose has been destroyed during sewage treatment. Table 3 gives a comparison of the approximate composition of cellulose, protein and lipid in domestic sewage sludge and municipal refuse (Rees, 1980a). Landfilled garden wastes would probably biodegrade at about the rate of rumen contents in which the crystalline and liquified plant cellulose is still intact. In a landfill, lignin will undergo a slow transformation into humic substances, which are very stable. In this regard, the cellulose/lignin ratio in landfill refuse is indicative of the extent of cellulose disappearance (Jones et al., 1983). In refuse landfills, cellulose is hydrolyzed to glucose and cellobiose. These sugars are rapidly fermented to CO_2, H_2, ethanol and acetic, propionic, butyric, valeric and caproic acids.

The metabolism of proteins in landfills is probably similar to that during anaerobic sewage digestion with hydrolysis to peptides and amino acids. Amino acid deamination then yields short chain carboxylic acids, CO_2 and NH_3. The latter reaction is the sole source of the branched chain, isobutyric and isovaleric acids. These two organic acids and those resulting from carbohydrate metabolism are typically found in refuse landfill leachates. Ammonium ion is also reported to be present at appreciable levels in landfill leachates which, as pointed out by Rees (1980a), is surprising since N in landfills is normally considered to be a limited nutrient with refuse having a high C/N ratio. In this regard P, which is also considered deficient and rate-limiting owing to its presence as insoluble precipitates with heavy metals such as Fe and Mn, is typically found at appreciable concentrations in landfill leachates (Rees, 1980a). This may be due to the dissolution of these precipitates by organic acids and by reduction of metals such as Fe and Mn. Ferrous and manganous phosphates are far more water-soluble. Lipid degradation in landfills likely consists of hydrolysis of glycerides followed by removal of two-carbon units from long chain fatty acids (beta oxidation) to ultimately yield acetate.

Some xenobiotic (man-made synthetic) compounds may be microbially degraded in landfills with ease, while some may not. Others, such as DDT, may be only partially degraded. Tetrachloroethylene is dechlorinated anaerobically to vinyl chloride, which is more toxic. Certain synthetic compounds can only be decomposed microbially under laboratory conditions which are controlled and where genetically-altered microorganisms may evolve through mutations which can then decompose the compound (cultural enrichment). This phenomenon is not likely to occur in landfills. Some xenobiotics can only be degraded if non-xenobiotic compounds of similar structure are present. Then enzymes responsible for metabolism of the latter compounds may be induced to degrade the xenobiotic as well (co-metabolism) (Aragno, 1989).

The possible degradation of plastics in landfills has also been reviewed by Rees (1980a). Much of the information on this subject derives from studies of the effect of landfill leachates on plastic landfill liners. Crystalline polyethylene, polypropylene and polybutylene have not been found to be greatly affected by leachates within 1 year. More swelling was noted with some of the thermoplastics such as chlorinated polyethylene, chlorosulfonated polyethylene and polyvinylchloride. Susceptibility to attack is not only a function of the polymer, but may also be enhanced by leaching out of plasticizers used in polymer construction thus exposing the polymer backbone to further attack. Degradation of polyester-based polyurethanes occurs in landfill leachate by microbial hydrolysis. If enhanced polymer degradation is desired, it may be possible to introduce weak links into the polymer by copolymerization, the latter bonds being protected during use by the presence of plasticizers. After discard in a landfill, the plasticizers would presumably be leached out thus exposing these more vulnerable bonds to enzymatic or chemical degradation.

Styrene monomers are also susceptible to aerobic microbial decomposition. Microbial degradation of plastics may be inhibited by the presence of toxic monomers such as vinyl chloride and styrene, but the effectiveness of this inhibition may be reduced by adsorption of such monomers by paper in the refuse. It is possible that plastic may be degraded by anaerobic oxidation with SO_4^{2-} as the terminal electron acceptor (Rees, 1980a).

GAS PRODUCTION IN LANDFILLS

Refuse in landfills undergoes microbial decomposition with the production of gas. The composition of the gas is determined by the types of microorganisms which predominate during successive stages of refuse degradation. Fig. 4 illustrates gas composition during municipal refuse decomposition in a landfill (Farquhar and Rovers, 1973). This decomposition occurs in four phases. The

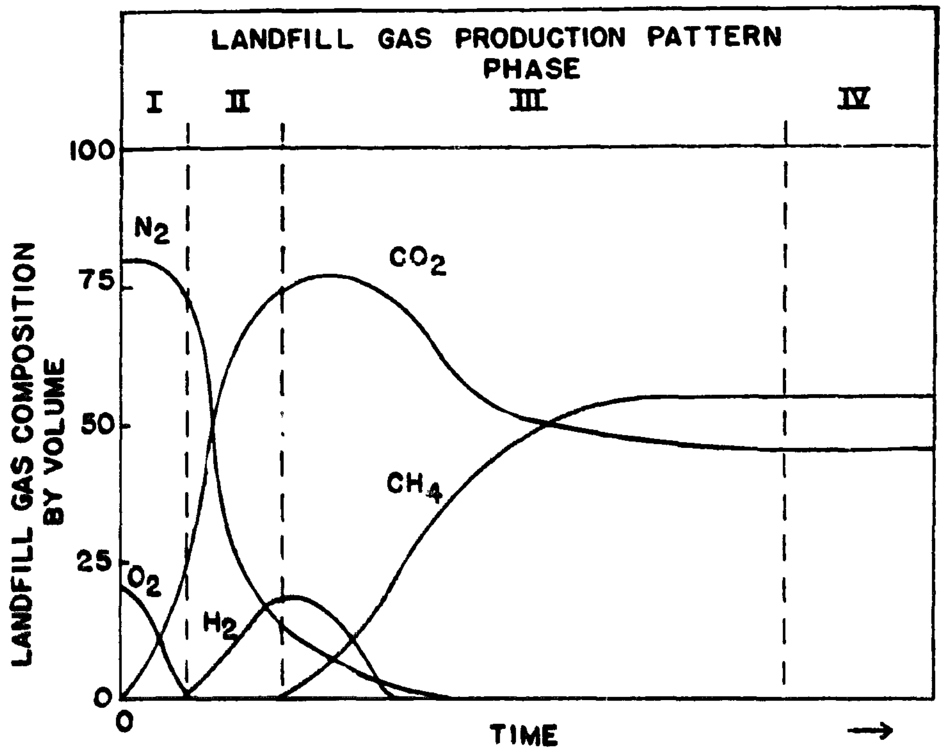

Fig. 4. Refuse landfill gas production pattern.

first phase (I), lasting a few weeks or less, involves aerobic decomposition with consumption of O_2 present in the refuse at the time of placement with production of roughly equal molar quantities of CO_2 as well as water as a by-product. Very little displacement of N_2 takes place during this phase. Phase II is anaerobic, but non-methanogenic. Oxygen is depleted during this phase and anaerobic and facultative microorganisms predominate. Carbon dioxide production peaks and production of volatile organic acids and H_2 begins during this phase. Methane is not produced during Phase II, possibly because insufficient concentrations of CO_2 exist in solution to act as a hydrogen acceptor.

Phase III is called the anaerobic methanogenic unsteady phase, during which the concentration of CH_4 increases to a relatively constant terminal value. Other compounds are also produced, some of which are released into the landfill atmosphere. During this phase, methanobacterium use up H_2 rapidly and CO_2 and N_2 are reduced to some terminal concentrations. From 8 to 16 months may be required to complete these three phases. During Phase IV, the composition of the gases and their rates of production remain steady, as determined by prevailing conditions. Sudden variations in gas production can occur, however, if nutrient depletion or accumulation of inhibitory substances takes place.

A knowledge of the enzymic activity associated with methanogenesis and gas production in landfills is important for a general understanding of the associated reactions. Rees and Granger (1982) measured the activities of protease, amylase and cellulase (which hydrolyze protein, starch and cellulose, respectively) with depth in a landfill operated as an efficient bioreactor. The data is illustrated in Fig. 5. The rates of protease and amylase activity show the same general pattern as that of gas production (Fig. 6), i.e. increasing dramatic-

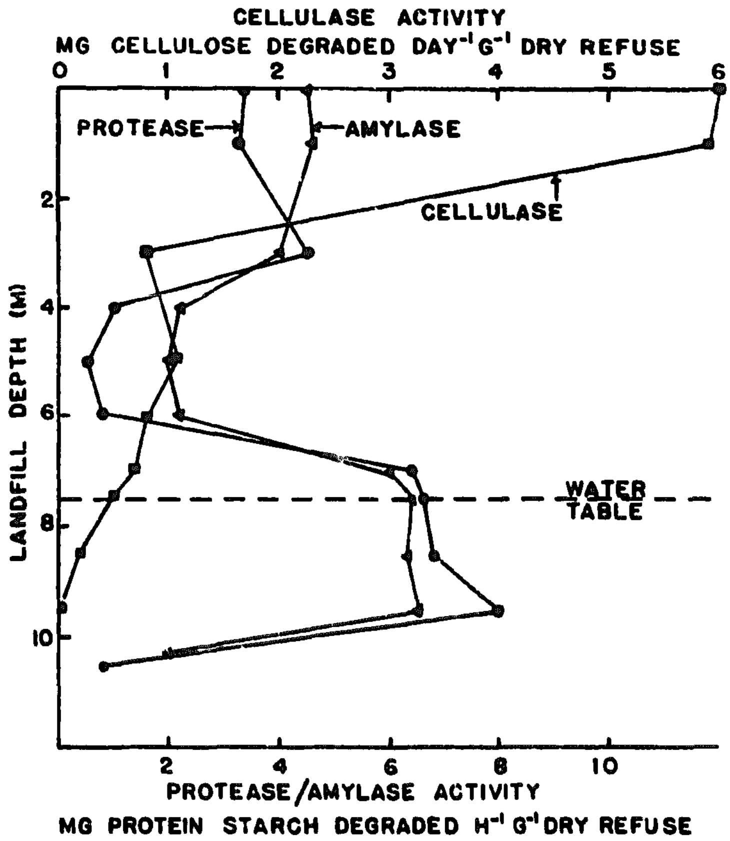

Fig. 5. The activities of amylase, protease and cellulase in a landfill as a function of depth (Rees and Granger, 1982).

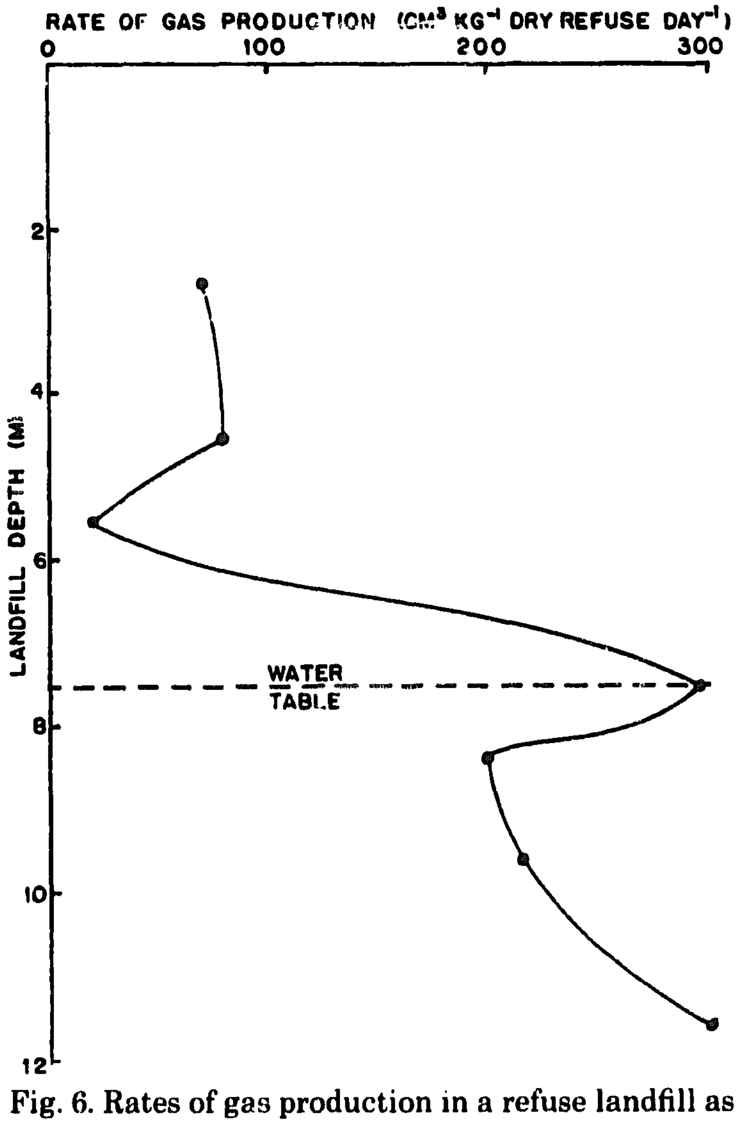

Fig. 6. Rates of gas production in a refuse landfill as a function of depth (Rees and Granger, 1982).

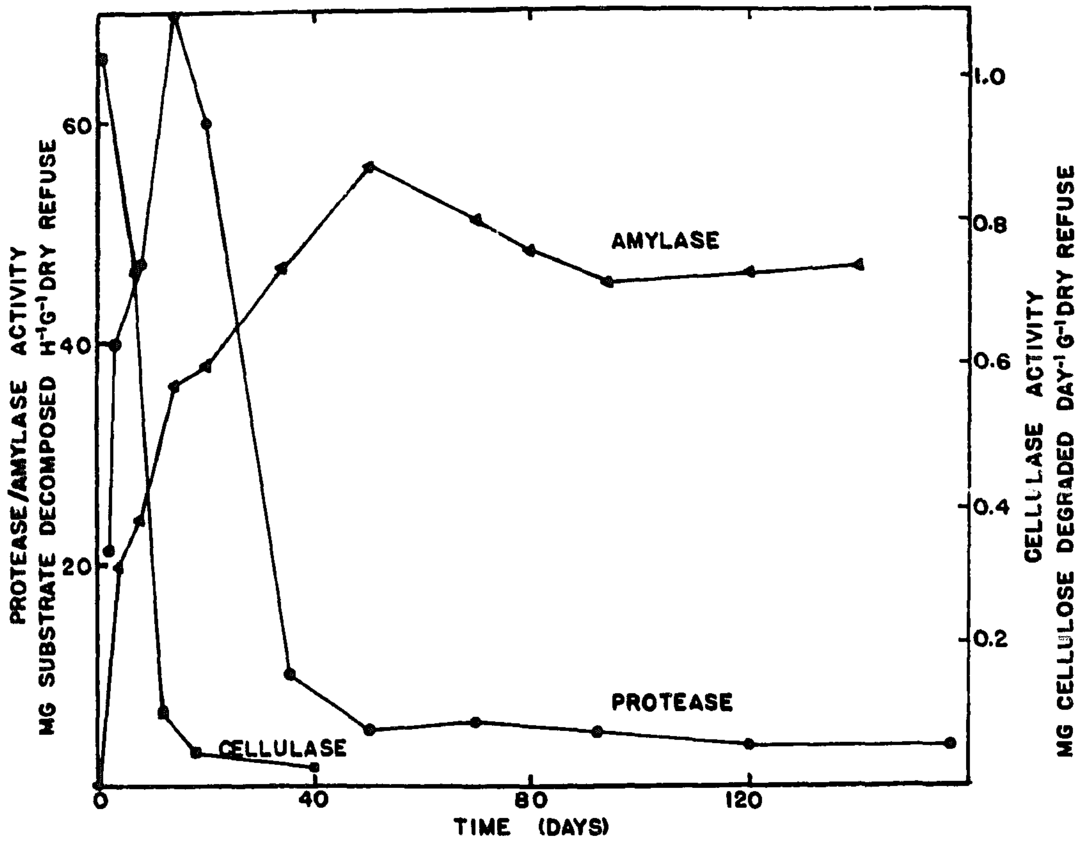

Fig. 7. The activities of amylase, protease and cellulase in pulverized, water-saturated refuse as a function of time.

ally at the water table. Cellulase activity was high only in the top 1 m of the landfill. The proportion of cellulose/lignin was about 4.5:1 above the water table to 0.5:1 below it. Fungi were apparently active in decomposition of cellulose since optimum cellulase activity was observed at pH 5.

The influence of water on enzymic activity in anaerobically degrading, water-saturated refuse was studied (Rees and Granger, 1982). The data obtained are shown graphically in Fig. 7. There was a three-fold increase in protease activity during the first 20 days and a 1000-fold increase in amylase activity during the first 50 days. Thereafter, these enzyme activities decreased. Conversely, cellulase activity declined rapidly from the start of the experiment. Since cellulosic materials are the major carbohydrate source in refuse, the mechanism of initial loss of cellulase activity was investigated and shown to be due to destruction of cellulase by proteolytic enzymes. The importance of this mechanism in actual refuse landfills is not clear, but Filip and Küster (1979) showed that proteolytic bacteria were the most prevalent group in the initial stages of refuse degradation in landfills.

Optimizing gas production and leachate quality in refuse landfills

As pointed out by Rees and Granger (1982), uncontrolled landfilling of domestic refuse is accompanied by its fermentation to CO_2 and CH_4 as well as a wide range of volatile carboxylic acids and NH_4^+. This can lead to pollution of ground and surface water by landfill leachates, and CH_4 can cause potential explosive or fire hazards. Only a percentage of the carbon compounds is converted to CH_4 and CO_2 and the concentration of carboxylic acids in the leachate may be very high. Therefore, several decades may be required for landfill stabilization; leachate quality is poor and gas is produced at a slow and ill-defined rate. Instead, operation and management of a refuse landfill as a controlled bioreactor can lead to efficient CH_4 production at more predictable rates enabling its collection and utilization, minimization of risks of gas migration out of the landfill, diminished odor problems, and improved leachate quality. They may also favor the safe degradation of toxic industrial wastes if co-disposed with domestic refuse. Since refuse temperature, moisture content and density are major factors determining its conversion to CH_4, management practices which optimize these parameters will be discussed.

Temperature and moisture control

Rees (1980b) summarized the major contributions to the thermal regime of an anaerobic domestic refuse landfill as heats of reaction and neutralization, solar radiation, aerobic metabolism, specific heat of water/refuse mixtures and heat losses to air and soil. If properly managed, even in temperate climates landfill temperatures can rise to 45°C and above under anaerobic conditions, resulting in the reduction of fatty acid concentrations in the leachate. The optimum mesophilic temperature for CH_4 production from domestic refuse in a conventional anaerobic digester is about 40°C. Few landfills achieve this temperature for sustained periods in temperate regions. They can, however, if managed as a bioreactive landfill. The rate of heat production in a landfill is determined by the rate of decomposition of organic matter. Thus the temperature attained by a landfill will be determined by the balance between the rates of heat production and addition (solar energy) and the rate of heat loss to the surrounding soil and atmosphere.

As stated by Rees (1980b), saturation of landfill refuse with water is essential if CH_4 production is to be optimized. Addition of a large volume of water to the surface of a fresh landfill, however, is likely to result in the production of a comparably large volume of leachate containing a high concentration of short chain fatty acids which will either inhibit methanogenesis directly or by lowering pH. Rees (1980b) suggests overcoming this by allowing water to rise slowly through the refuse from the bottom of the site. At an experimental refuse landfill site he found a rate of water addition of 1–2 m per year to be acceptable. Rates below 1 m $year^{-1}$ may be satisfactory for initiating fermentation and those appreciably above 2 m $year^{-1}$ might produce cooling effects inhibiting the fermentation. Initiating fermentation in a bioreactor-type landfill can be promoted by addition of large quantities of methanogenic microorganisms in the form of effluent and sludge from an anaerobic sewage digester since the population of such microorganisms in fresh refuse is typically low. This, plus the slow addition of water from the bottom of the landfill, will start the multi-phase CH_4 generation slowly thus obviating the accumulation of carboxylic acids and low pH.

TABLE 4

Leachate composition (mg dm^{-3}) of three landfills as related to the depth of refuse (as an insulator) above the reactive (methanogenic) zone

Land-fill code	Depth of refuse above (m)	pH	Acetate	Propionate	Butyrate	Higher fatty acids	Cl^-	SO_4^{2-}	Na^+	NH_4^+
A	> 4	7.9	90	0	0	0	2000	62	1420	1550
B	3	5.5	5345	4175	7390	3500	1500	1600	1520	840
C	< 2	5.5	8455	4140	4460	5930	910	890	800	900

Based on temperature measurements made throughout the year in a bioreactive-type landfill in England, Rees (1980b) concluded that it would be unlikely that temperatures of 40–45°C could be achieved unless an insulating layer of about 4 m of refuse is located above the reactive zone due to loss of heat to the atmosphere.

Table 4 shows the leachate quality of a managed bioreactive refuse landfill (A) with at least 4 m of insulating refuse above the reactive zone (to reduce atmospheric heat loss) as compared with two others (B and C) with less refuse above (Rees and Granger, 1982). The greater reactivity and efficiency of CH_4 fermentation in landfill A is reflected in the low concentrations of organic acids and SO_4^{2-} in the leachate. This, coupled with the high NH_4^+ concentration, contribute to a pH of 7.9, which favors methanogenesis. Ammonium concentration remained high, indicating that the source of N was sufficient to sustain the high reactivity of the methanogenic microorganisms. The temperature of the

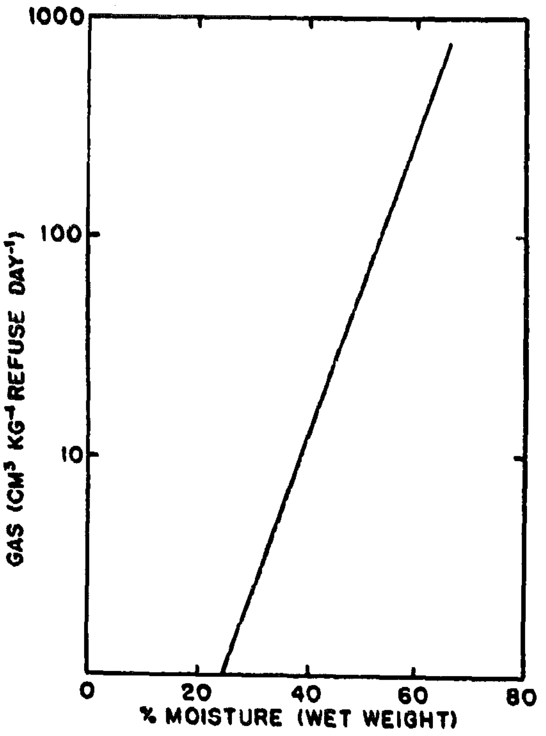

Fig. 8. The relation between the moisture content of domestic refuse and gas evolution during anaerobiosis.

refuse in landfill A at depths of 3–4 m remained in the range of 35–40°C throughout the year even though the air temperature in January reached about 0°C. Conversely, the lower temperatures in the reactive zones of landfills B and C inhibited methanogenesis with lower conversion of organic acids to CH_4 and CO_2. An interesting suggestion by Rees (1980b) is that it might be possible to take advantage of the high temperatures (70–80°C) attainable in landfills where aerobic metabolism of refuse is occurring to initiate landfill reactions in winter, in which case accumulation of high inhibitory concentrations of organic acids would be less likely to occur.

Figure 8 shows the effect of the water content of refuse on the rate of gas production during its fermentation. The graph was constructed by Rees (1980b) from several experimental literature values. This illustrates that the logarithm of the rate of gas production is directly proportional to the moisture content of the refuse. At 55% moisture in refuse, sufficient heat can be generated to raise landfill temperature above 30°C in a year.

Refuse density

Refuse compaction is important to enhanced CH_4 production. It brings the refuse into intimate contact with water, nutrients and microorganisms. High compaction pushes a system of limited moisture closer to an optimum field capacity (Buivid et al., 1981). Loosely-packed refuse may have some advantage if landfill leachate is recycled as it avoids channeling and therefore lack of surface contact. The mechanism of the effect of refuse density on methanogenesis in landfills is, however, not clearly understood. For instance, if dry refuse (25% moisture as received) is compressed before landfilling, gas production during fermentation is enhanced. Conversely, if refuse is placed in a landfill at different densities and then wetted, gas production is enhanced at lower densities. These results have been explained by assuming that water is being squeezed out of the dry refuse at higher densities and therefore being made available to microorganisms. With wetted refuse, it is argued that high initial densities impede water entry and faster reaction rates occur when water can enter the refuse, i.e. at lower densities. The leachate quality of fermenting baled refuse is also higher (lower organic carbon concentration) than that of fermenting unbaled pulverized refuse (Rees, 1980a). It is obvious that cellulosic wastes with impermeable coatings or tightly bound books or catalogues might present especially difficult problems of microbial accessibility (DeWalle et al., 1978). In this regard, during microbial digestion of refuse, bacteria adhere to the surface by van der Waals forces, electrostatically or by secretion of exopolymers which act as a glue. This attachment will prevent their wash-off by percolating liquids. Microorganisms often associate together physically in this manner forming mixed microbial communities (Aragno, 1989).

Interacting factors affecting landfill gas production

Farquhar and Rovers (1973) developed a schematic representation of factors that affect landfill gas production. This is shown in Fig. 9 and illustrates how

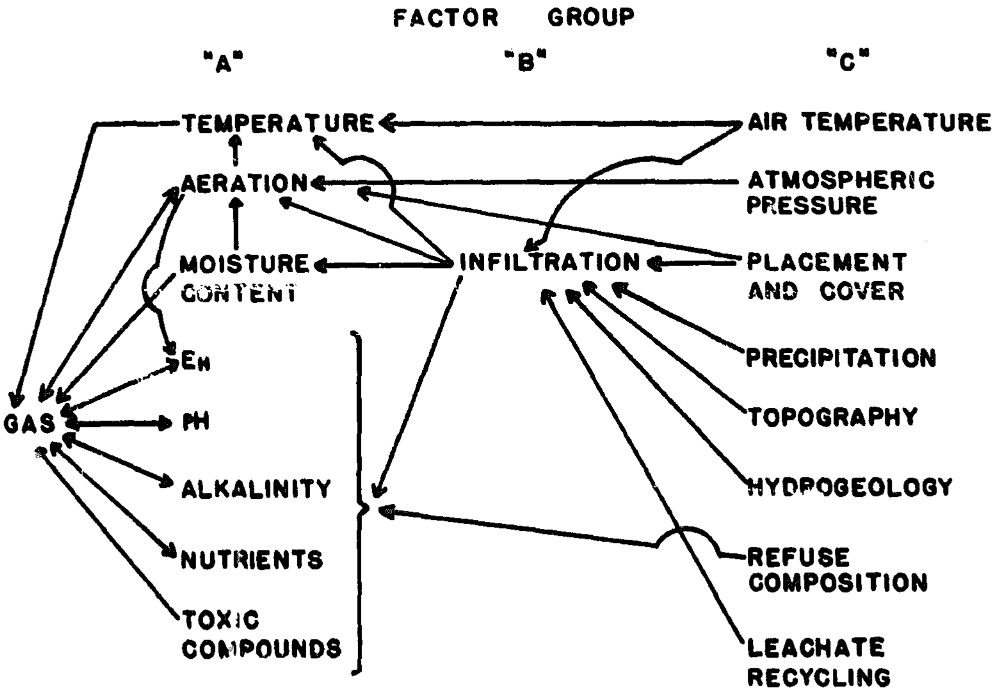

Fig. 9. Factors influencing gas production in refuse landfills.

the various factors can interact in ultimately affecting gas evolution from decomposing refuse. For instance, under the Group A factors, which pertain to the immediate microbial environment, the authors explain that a lowering in temperature would reduce CH_4 production. This would lead to an accumulation of organic acids thus reducing alkalinity and pH. These conditions would further reduce CH_4 production. Considering the Group B factor, infiltration, its magnitude and composition can affect most of the factors in Group A. The factors in Group C (placement and cover, topography, hydrogeology, refuse composition, leachate recycling), several of which may be influenced by procedures during landfill design and operation, also interact with those in the other groups. Thus air temperature is a partial determinant of refuse temperature and may therefore influence infiltration and affect evaporation. Exchanges between air and gases within the refuse will be affected by atmospheric pressure. The movement of gases and water at the surface of the landfill will be affected by refuse placement and the materials and procedures used to cover the refuse. Precipitation, topography and hydrogeology will affect the magnitude of landfill infiltration and leachate recycling will affect its composition. Obviously, refuse composition as regards factors such as age, cellulosic content, C/N ratio and the presence of microbial inhibitors will influence several of the factors in Group A.

Typical landfill gas composition is shown in Table 5 (Ham, 1979). Compounds that have been found as hydrocarbons include benzene, heptane and nonane as well as other trace compounds such as acetaldehyde, acetone, ethylmercaptan and argon (Gandolla et al., 1982). This composition can, however, vary widely depending on many factors, including composition, density, temperature and moisture content of the refuse, landfill age and leachate recycling through the

TABLE 5

Typical landfill gas composition

Constituent	Percent
CH_4	47.4
CO_2	47
N_2	3.7
O_2	0.8
H_2	0.1
H_2S	0.01
CO	0.1
Paraffin hydrocarbons	0.1
Aromatic hydrocarbons	0.2
Trace compounds	0.5

landfill. Other compounds, including toluene, xylenes, propyl benzenes, vinyl chloride, tetrachloroethylene, methanethiol and methanol, have been reported in gas from landfills where both domestic and industrial wastes were co-disposed (O'Leary and Tansel, 1986a).

Management of landfill gas

Engineering systems to recover CH_4 as a fuel source from closed landfills or those being actively filled have been constructed and are operated mostly by private firms in the United States (Berenyi and Gould, 1986). The economics of such gas production has been studied and the requirements determining feasibility have been summarized by Bogardus (1987). These include:

(i) at least one million tonnes of refuse in place,
(ii) an average depth of about 12 m,
(iii) an active fill area of about 16 ha,
(iv) a receipt rate during the sites operational life of about 365 t per day, and
(v) nearby willing users of the gas.

Whereas most landfills in the past have tended to be relatively small, newer ones will expectedly be much larger and therefore more amenable to economically feasible gas production (Croke and Zimmerman, 1986).

A typical landfill gas abstraction process design is shown in Fig. 10 (Richards and Hornsby, 1988). Direct use of landfill gas in boilers is comparatively simple with only the need to filter out particulates and remove moisture (which can range from 5% to saturated) by chilling. In contrast, pipeline quality gas requires removal, in addition, of CO_2, non-methane hydrocarbons and corrosive H_2S, mercaptans, halocarbons (Bogardus, 1987) and the last traces of water (Hekimian et al., 1976). Landfill gas has an energy value ranging from 450 to 550 BTUs* per cubic foot (15 885–19 415 BTUs m^{-3}) as compared with natural gas having about 1000 (35 300 BTUs m^{-3}) (Hekimian et al., 1976).

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*1 BTU = 1.06 kJ.

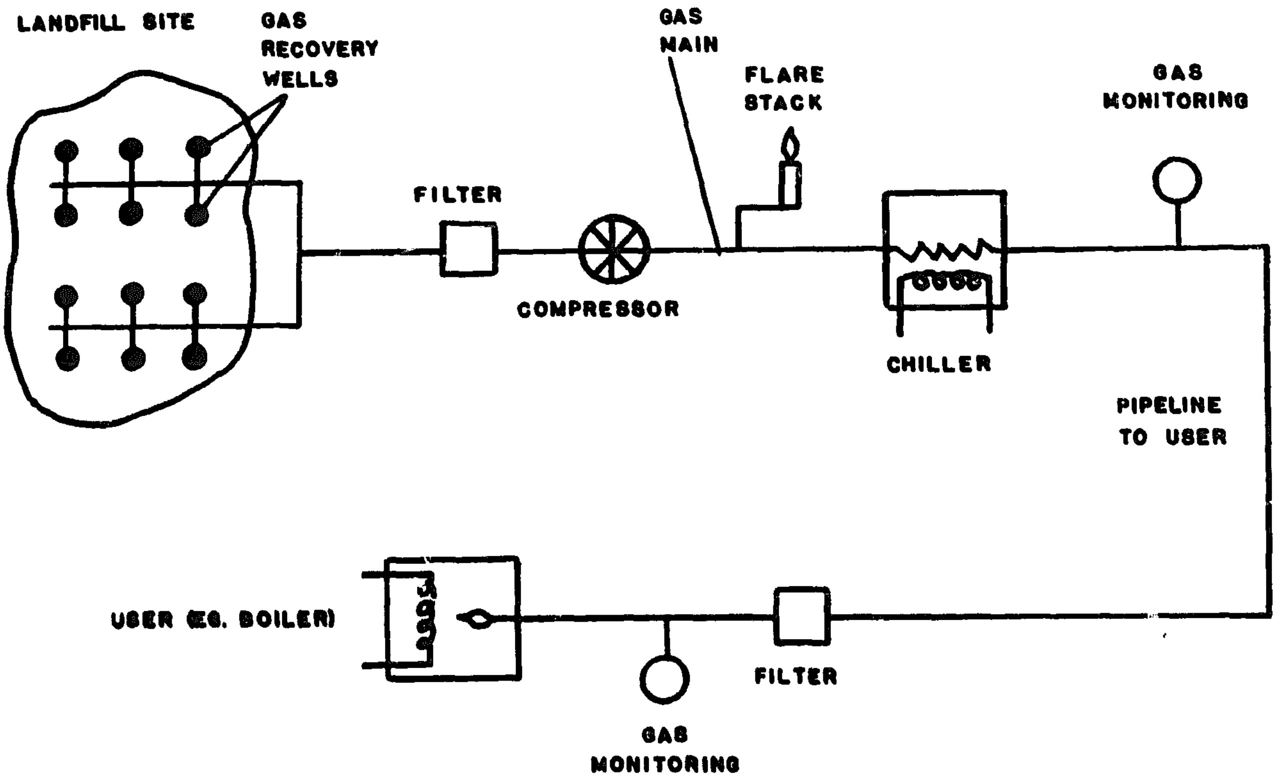

Fig. 10. Typical landfill gas abstraction layout.

Bogardus (1987) has outlined some of the assets and liabilities of landfill gas. The rate of landfill gas production depends on many factors, including refuse composition, cellulosic content, and age, moisture content, depth, pH and temperature. Steady generation typically occurs within 1–2 years after the original landfilling begins and can continue for 10–30 years after landfilling has been completed. Actual rates depend on porosity and depth of cover materials, landfill configuration, collection efficiency, and gas withdrawal rate limitations imposed to prevent excessive air from entering the refuse (Crutcher et al., 1982). A commonly used range for potential recovery rates is 80–280 cubic feet per ton of refuse (2.5–$8.8\,m^3\,t^{-1}$) per year. Predicting future gas production from a landfill is most accurate if field measurements are made with test wells at the intact landfill per se rather than from laboratory studies of gas obtainable from refuse withdrawn from it (Ham, 1979; Siegal, 1987). Calculated values are based on assumptions which are usually not valid under actual landfill conditions. These assumptions include:

(i) that complete degradation occurs, but it is known that plastics and lignin show little if any deterioration in a reasonable time,
(ii) that degradation is entirely anaerobic,
(iii) that refuse degrades under conditions such that a correct balance of substrates and nutrients is always available, and
(iv) that no portion of the degraded matter is utilized for microbial cell synthesis (Ham, 1979).

The mechanics of gas movement through refuse and soil are very complex (Mohsen et al., 1980; O'Leary and Tansel, 1986a). Gas migration is either vertical or lateral. Since CH_4 is lighter than air, it tends to rise, the rate being strongly affected by barometric pressure. If the top of the landfill is sealed, or if wet surface soil or frozen ground exists. the gas will tend to diffuse laterally. It will follow the path of least resistance, migrating more easily through sand or gravel than silt or clay with possible hazards to nearby buildings. Maximum lateral migration distances of > 300 m have been observed for CH_4. Methane concentrations in air of between 5 and 15% by volume are explosive if ignited (Bogardus, 1987). Since the O_2 content of landfill gas may vary widely, the explosive limit of the gas may be attained at even lower CH_4 concentrations. If landfill gas is not being actively collected for use, a passive solution to the horizontal migration problem may be to install shallow gas venting trenches or pipes through which it can escape to the atmosphere or be burned off through the use of flares.

LANDFILL LEACHATES

Leachates production

Water containing dissolved constituents is squeezed out of refuse by its compression and compaction in landfills. Water is also produced during refuse decomposition. These processes, along with precipitation percolating through

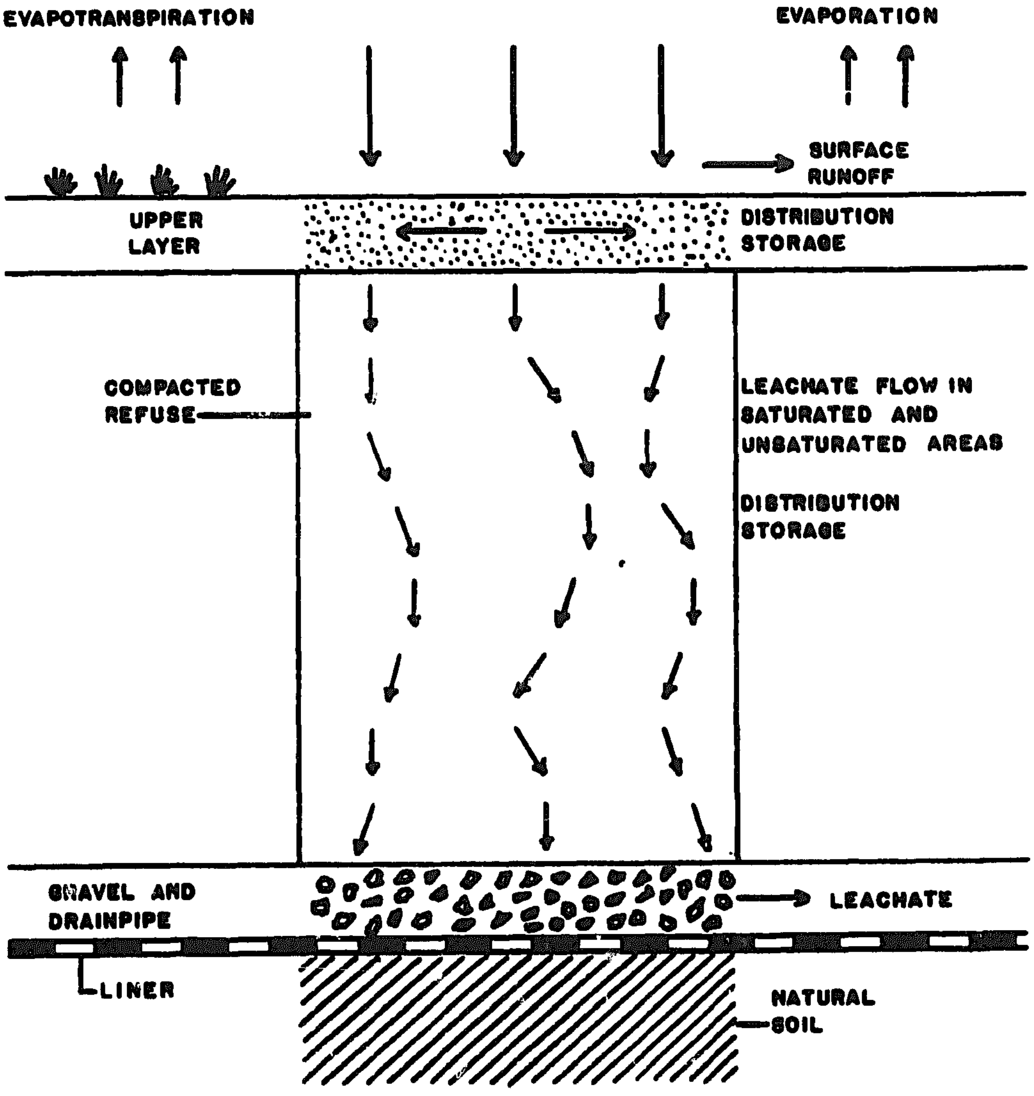

Fig. 11. Outline of the water budget of a municipal refuse landfill.

landfills, eventually causes the refuse to reach field capacity (about 125 l m^{-3} of refuse) and therefore leads to liquid wastes or leachates which can contaminate underground water sources below (Zanoni, 1972; Shuster, 1976). Leachate egress will occur at the lowest point or at a permeable horizon on the side of the site. Discoloration of water results, Fe precipitates coat land surfaces and vegetation, dissolved oxygen diminishes, nutrients are enriched and toxic metal contamination of streams and groundwater may result. Contaminated aquifers are not readily cleansed naturally and economical methods for their decontamination do not exist. It is important therefore to review the nature and fate of contaminants in leachates, factors affecting their magnitude and methods for their attenuation.

The water budget of a refuse landfill is shown in Fig. 11 (Stegman, 1983). Water input results from precipitation or melting snow entering the landfill as well as water produced as a result of respiratory processes in the refuse. Water loss is the sum of surface evaporation and evapotranspiration, lateral diffusion in the landfill cover and production of leachate. The volume of leachate produced is influenced by climate. In comparatively warm climates, leachate production usually follows precipitation quite closely. In colder climates, there may be a lag since much precipitation falls as snow.

Leachate composition

As water percolates downward through landfills, organic and inorganic constituents are dissolved. Leachate composition depends on the nature, particle size, degree of compaction and age of the refuse as well as the point of leachate sampling. Refuse composition may be highly variable. Fewer yard wastes will be discarded in winter months. The greater the refuse is compacted, the greater will be the volume of leachate since compaction reduces the filtration rate. Méndez et al. (1989) reported differences in leachate composition when sampled halfway down the depth of the landfill as compared with that sampled from the collection tank. The depth of refuse at the top of bioreactive landfills acting as an insulation against cold winter temperatures also affects leachate composition (see Table 4). Season can also affect the composition of leachates. The chemical oxygen demand (COD) of landfill leachates may be highest in spring (Méndez et al., 1989).

Concentration ranges of a number of constituents in leachates from 15 landfills in the United Kingdom are given in Table 6 (Robinson et al., 1982). The age of the refuse in landfills greatly affects leachate composition. This is illustrated in Tables 7 and 8 (Lema et al., 1988). The concentration of many constituents, including pollutants, in landfill leachates decreases with refuse age. The range of the pH of the leachates (5.7–8.3) coincides with that required for microbial activity. Whereas organic acids would tend to lower pH, the accumulation of NH_3 would have the opposite effect (Méndez et al., 1989). Phosphate levels in leachates may be low, but submergence would tend to solubilize P that is present as insoluble ferric and manganic phosphate by

TABLE 6

Composition of leachates from refuse landfills

Constituent	Concentration range (mg l^{-1})
pH	6.2–7.4
COD[a]	66–11600
BOD[b]	< 2–8000
TOC[c]	21–4400
Ammonia N	5–730
Nitrate N	< 0.2–4.9
Organic N	ND[d]–155
$H_2PO_4^-$	< 0.02–3.4
Cl^-	70–2777
SO_4^{2-}	55–456
Na	43–2500
Mg	12–480
K	20–650
Ca	165–1150
Cr	< 0.05–0.14
Mn	0.32–26.5
Fe	0.09–380
Ni	< 0.05–0.16
Cu	< 0.01–0.15
Zn	< 0.05–0.95
Cd	< 0.005–0.01
Pb	< 0.05–0.22

[a] Chemical oxygen demand. [b] Biological oxygen demand. [c] Total organic carbon. [d] Not detectable.

TABLE 7

Composition of landfill leachates as related to the age of the refuse

Constituent	Age		
	Young	Medium	Old
pH	5.7–8.0	6.4–8.0	6.6–8.3
BOD (g l^{-1})	7.5–17.0	0.37–1.1	0.07–0.26
COD (g l^{-1})	10.0–48.0	1.2–22.0	0.67–1.9
$N(NH_3)$ (g l^{-1})	0.04–1.0	0.03–3.0	0.01–0.9
Zn (mg l^{-1})	0.53–34.2	0.18–0.22	0.19–0.37
Mn (mg l^{-1})	4.8–38.5	3.24	0.05–3.24
Ni (mg l^{-1})	0.3–6.11	0.01	0.09–0.47
Cr (mg l^{-1})	0.13–0.56	0.12	0.04–0.17
Cu (mg l^{-1})	0.08–0.30	0.02–0.11	0.03–0.12
Pb (mg l^{-1})	0.05–0.92	0.04–0.08	0.01–0.14
Cd (mg l^{-1})	0.02–0.10	0.02	0.01

TABLE 8

Volatile fatty acids (g l^{-1}) in landfill leachates as related to the age of the refuse

Age[a]	COD	Acetic acid	Propionic acid	Butyric acid	Other acids
Y	28–48	1.16–8.5	1.01–5.25	1.77–6.13	0.38–5.75
MA	3.75–22.3	0.56–3.5	0.19–0.91	0.19–1.81	0.08–2.5
O	5.03	0.34	0.29	0.41	0.08

[a] Y, young; MA, medium age; O, old.

reduction of the metals to the more water-soluble ferrous and manganous forms (Ponnamperuma, 1955). The decrease in the concentration of organic matter in leachates with refuse age is believed due to fermentation of the hydrolyzable organics with ultimate production of more microbially refractory compounds. Free volatile fatty acids typically appear in highest concentrations in leachates of newly deposited refuse. They are readily fermented, but initially enhance solution of heavy metals. Volatile amines, ethanol and other alcohols, hydrocarbons, esters and terpenes may also be present during the acidification stage of refuse degradation (Harmsen 1983). The stability of other compound classes with age has been reported to be: aromatic hydroxyl compounds > hydrolyzable amino acids > carbohydrates (Chian, 1977). Fulvic acids decrease in concentration in leachates with age and humic acids increase, the latter being very stable to further microbial degradation (Chian and DeWalle, 1977; Artiola-Fortuny and Fuller, 1982). These compounds are high molecular weight polymers containing carboxylic, phenolic and keto groups. These stable compounds may typically have molecular weights > 1000 (Harmsen, 1983). Conversely, Chian (1977) reported fulvic acid-type compounds with molecular weights ranging from 500 to 10 000 as the most stable in landfill leachates. Active landfills which contain old as well as recently deposited refuse may yield leachate that contains all of the above compound classes simultaneously. Assuming that precipitation drains through a landfill evenly, the larger the landfill, the smaller will be the concentration of volatile fatty acids in the overall leachates since the newly deposited refuse will represent an ever smaller proportion of the old (Lema et al., 1988).

Heavy metals and toxic organics in leachates

The heavy metal concentrations in leachates of refuse in the acidification stage will typically be much higher than those of refuse in the fermentation stage owing to metal solubilization and complexation by volatile fatty acids (Harmsen, 1983). Aromatic hydroxyl compounds (Lema et al., 1988) as well as humic and fulvic acids (Schnitzer, 1969; Knox and Jones, 1979; Shuman and Cromer, 1979; Filip et al., 1985; Weis et al., 1989) can also complex metals such as Cu, Cd, Pb, Fe, Ni, Mn, Co, Ca and Zn. Reduction of ferric iron by anaerobic microbial activity has been reported to solubilize heavy metals such as Cd, Ni, Pb and Zn when the latter metals were co-precipitated with iron oxide (Francis and Dodge, 1990). The importance of this reaction in solubilizing such metals in landfills is unknown. The production of S^{2-} through reduction of SO_4^{2-} in landfills may have the opposite effect by reaction with such heavy metals to yield highly insoluble sulfides (Hounslow, 1980). The existence of recycling programs may greatly reduce the presence of toxic substances (Hg, Pb, Ni, and Cd from batteries, for instance) in refuse and therefore in leachate. Whether or not incinerator ash, sewage sludge or industrial wastes are intentionally or inadvertently co-disposed with refuse will also be determinants.

Landfill leachates have also been reported to contain a wide range of toxic organic compounds, including aliphatic and aromatic hydrocarbons, halogenated organics and other classes (Khare and Dondero, 1977; Sawhney and Kozloski, 1984; Schultz and Kjeldsen, 1986; Barker, 1987; Chichester-Constable et al., 1987; Först et al., 1989). Carcinogenic polycyclic aromatic hydrocarbons have also been reported in landfill leachates, which may account for their mutagenicity (Kamiya et al., 1989). The water-solubility of chlorinated hydrocarbons such as DDT and PCBs has been reported to be enhanced owing to complexation with fulvic and humic materials (Carter and Suffet, 1982; Hassett and Milicic, 1985; Chiou et al., 1986).

NATURAL ATTENUATION OF CONSTITUENTS IN LEACHATES

Natural attenuation is one method used to purify landfill leachates. It assumes that passage of leachate through the unsaturated zone below the landfill will remove undesirable constituents from it (James, 1977) and what is not removed will be sufficiently diluted by groundwater to an acceptable level. Many compound classes have been identified in leachate plumes below landfills such as aliphatic and aromatic acids, phenols and terpenes deriving from decomposing plant material as well as chlorinated and non-chlorinated hydrocarbons, nitrogen compounds, alkylphenol polyethoxylates and alkyl phosphates of industrial origin (Reinhard et al., 1984). Although natural attenuation landfills are no longer permitted in some areas such as Wisconsin, it is important to consider the reactions involved since leachate does leak into such unsaturated zones, whether intended or not, and can pollute groundwater. A number of mechanisms of attenuation by soil constituents is involved including filtration, adsorption, microbial action, ion exchange, precipitation and dilution. These mechanisms have been outlined by Bagchi (1987).

Filtration and adsorption

Filtration is the physical trapping of the suspended or settleable solid particulates in landfill leachates by the random pore structure of the soil. The solids in leachates may result from processes such as chemical precipitation or microbial growth. Finer soil materials and lower leachate hydraulic gradients will improve filtration. Adsorption of molecules onto the surface of clay particles can effect a considerable attenuation of pollutants in leachates. Adsorption is pH-dependent and the adsorption capacity of a particular clay for compounds in a specific leachate can only be determined by experimentally establishing the adsorption isotherm of that system. Thus leachate quality as effected by natural attenuation is usually site-specific as regards landfill location. Organic compounds may also be trapped by small molecule diffusion into clay lattices as they expand upon wetting (Barshad, 1952).

Microbial action

Microbial action takes place in the unsaturated and saturated zone below landfills. The reactions they may accomplish or contribute to in these zones include decomposition of carbonaceous wastes to produce CO_2 and organic acids, exhaustion of available O_2 to yield an anaerobic environment and oxidation or reduction of inorganic compounds. Complexation of metal ions, methylation of metal and metalloids and transformation of CN^- to mineral nitrogen compounds and finally to N_2 gas by denitrification also occur. Production of large and small organic molecular species which may adsorb leachate constituents and production of colloidal organic debris which can infiltrate pore spaces and reduce soil permeability (Bagchi, 1987) are also microbially mediated. If the buffering capacity of these zones is sufficient to neutralize the organic acids in such leachates, microbial population growth and activity will be promoted (Blakey and Towler, 1988).

Ion exchange

Ion exchange occurs mostly with clays, their cation exchange capacity being due to isomorphous substitution, broken bonds and edges and replacement of hydrogen ions with other exchangeable ions such as Na or Ca. Of the four basic groups of clay minerals, their cation exchange capacity (CEC) decreases in the order smectite > chlorite > illite > kaolinite. The ease of replacement of one ion for another depends on ion size, valence and concentration but, in landfill leachate, concentration (mass action) is the predominating factor influencing ion exchange. The CEC of clay minerals depends on pH only above about 6.0 where ionization of H from exposed OH groups at crystal edges occurs. In studies with landfill leachates passed through clays, K, NH_4^+, Mg and Fe were moderately attenuated, while Pb, Cd, Hg and Zn were strongly attenuated. Attenuation capability decreased in the order: montmorillonite > illite > kaolinite (Griffin et al., 1976). Although less well understood, anion exchange sites exist in clays which may hold organic or inorganic anions. Anion exchange capacity of clays increases as soil pH decreases.

Precipitation

As the concentrations of dissolved chemical species approach their solubility limit, precipitation becomes a dominant attenuation mechanism. Soil pH determines acid–base reactions as well as influencing equilibrium reactions and the relative concentrations of OH^-, CO_3^{2-}, S^{2-} and other ions in such systems which typically result in heavy metal precipitation. As pointed out by Bagchi (1987), since landfill leachates are typically anaerobic, the reducing reactions use up hydrogen in acidic soils and increase CO_2 pressure in calcareous soils. Thus the pH of a landfill leachate-saturated soil converges to a near-neutral value regardless of the initial soil pH. Thus the shift in pH of leachate in groundwater from that near the landfill to a near neutral value at some distance therefrom can serve as a marker of its movement. LaBauve et al. (1988) reported soil pH to be the most important characteristic influencing retention of Cd, Ni, Pb and Zn in batch equilibrium studies, although CEC, Fe oxides and percentages of clay and organic matter were also important. The presence of a synthetic landfill leachate (LaBauve et al., 1988) or known ligands (Slavek and Pickering, 1981) which form stable anionic complexes with such metals enhances their mobility and inhibits their precipitation.

Dilution

Diffusion and dispersion are the mechanisms of dilution of leachate. Diffusion will be determined by the difference in chemical composition between the leachate and groundwater as the former equilibrates with the latter. In this regard, it must be remembered that the background concentration of specific

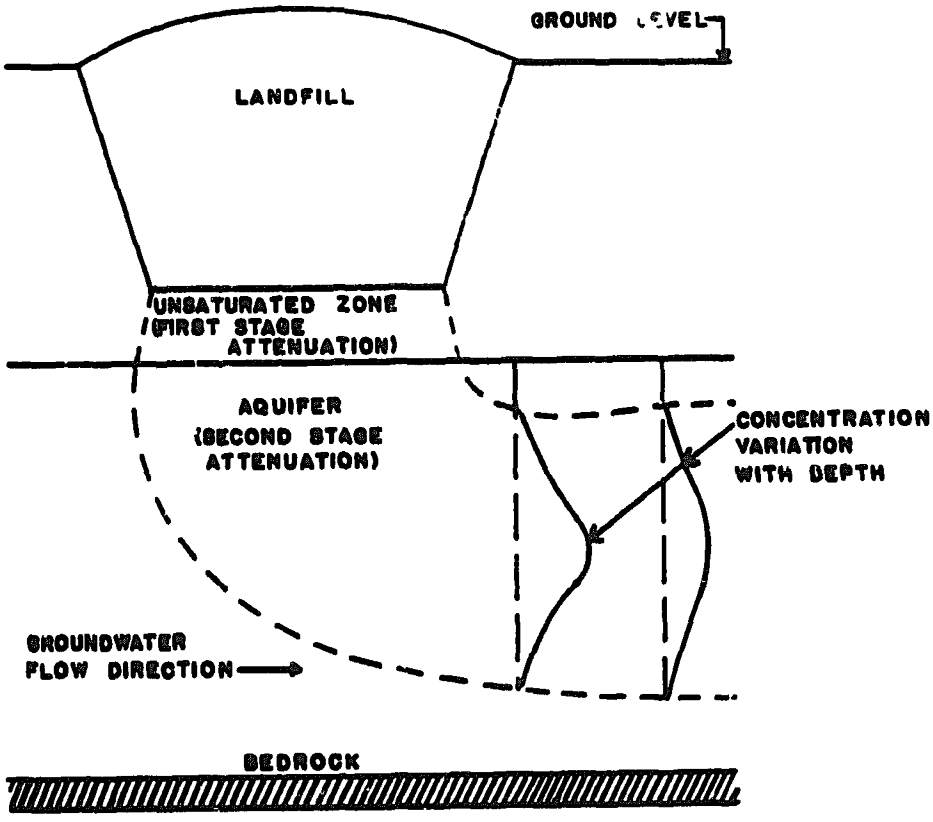

Fig. 12. Stages of attenuation of landfill leachate from a natural attenuation-type landfill, dispersion of leachate in soil and variation in concentration of leachate constituents within the leachate plume.

elements in groundwater may be high in some locations because the minerals in those areas are naturally high in those elements. Dispersion can occur in longitudinal (with the direction of flow) or transverse directions. Longitudinal dispersion results from different macroscopic velocities as portions of the leachate move through larger or less tortuous paths. Transverse dispersion (normal to the direction of flow) is caused by repeated splitting and deflection of flow by solid particles in the aquifer. Theories of dispersion may be more applicable to sand and gravel deposits than to clayey soils. A possible outline of a landfill plume is depicted in Fig. 12 (Bagchi, 1987), showing stages of natural attenuation, dispersion of leachate in soil and variation in concentration of leachate constituents within the plume.

Bagchi (1987) points out that an ideal soil stratigraphy beneath a landfill for effective natural attenuation of leachate would consist of

(i) an unsaturated zone composed mainly of silty clay with a relatively high CEC and a permeability of 1×10^{-4} to 1×10^{-5} $\mathrm{cm^{-1} s^{-1}}$,

(ii) a sandy groundwater table immediately below this with a higher permeability ($1 \times 10^{-3} \mathrm{cm\,s^{-1}}$), and

(iii) a thick enough aquifer so that plume development is not influenced by bedrock..

This ideal soil stratigraphy may only occasionally exist below landfills, however, with many other variations possible. The velocity of movement of leachate must be reasonably slow to allow attenuation reactions to occur as completely as possible.

Much remains to be learned about natural attenuation processes. The distribution of trace organics in such leachate plumes, for instance, may be very complex (Reinhard et al., 1984). It must also be realized that attenuation reactions that serve to cleanse leachates are reversible. For instance, adsorbed pollutants may later be desorbed by other preferentially adsorbed or solubilizing leachate constituents. Also, bacteria that absorb pollutants, eventually die and their subsequent mineralization may result in pollutant release if the latter has not been metabolized. Cations such as NH_4^+ in leachates may replace toxic metal ions on exchange sites, therefore releasing them (Baedecker and Back, 1979). Precipitated metals may be resolubilized by changes in pH or complexation. It is probable that natural attenuation landfills always result in some degradation of groundwater quality (Bagchi, 1987). The principles involved in predicting the fate and transport of organic compounds in groundwater have been reviewed (Davis and Olsen, 1990; Olsen and Davis, 1990).

LEACHATE TREATMENT

The high organic matter content, odor and volume of landfill leachates threaten pollution of groundwater in many locations and thus urgently require treatment. The selection and application of suitable leachate treatment methods are complex because of their great variability in composition and concentration. This subject has been reviewed by Lema et al. (1988).

Treatment as sewage

Piping landfill leachate to sewage wastewater treatment plants is one possible method. This would seem applicable since leachate usually contains an excess of N and sewage an excess of P. Therefore, these elements, essential for microbial degradation of organic matter, would not be deficient. However, leachates can vary greatly in their content of organic and inorganic constituents as influenced by refuse composition, age and compaction as well as season of the year. Leachates may contain CODs up to 200 times that of urban sewage and therefore can comprise only a small percentage of sewage plant input to permit satisfactory treatment. Other problems such as toxicity to microorganisms effecting sewage digestion, diminished sludge settling, precipitation of Fe oxides and corrosion may occur. Under these conditions, the final plant effluents may be frothy and contain high concentrations of NH_3.

Landfill recirculation

A pretreatment method involves recirculation of leachate through the landfill, which enhances biodegradation of organic matter and shortens the time for final refuse stabilization by helping to maintain an optimum moisture content in the refuse. This technique results in partial cleansing of the leachate and reduction in its volume due to evaporation and considerably enhances CH_4 gas production. Recirculation may not be possible if the prior water balance of the landfill results in leachate accumulation. The pH of the landfill should be maintained at about 7 and channelling should be minimized by uniformly distributing such recycled leachate. A variation of the leachate recirculation method involves the use of lagoons for anaerobic treatment of leachate, a volume of which is recycled through the landfill. Several auxiliary ponds may also be located neaarby to adjust for seasonal fluctuations in leachate production. This method effects far greater reduction in BOD and coliforms than recirculation alone.

Aerobic and anaerobic digestion

Although the majority of publications dealing with leachate treatment methods involve laboratory simulations, the results obtained provide guidelines for predicting their success in the field. Venkataramani et al. (1984) have reviewed pertinent liteature on the use of aerobic and anaerobic digestion methods for the treatment of landfill leachates. The aerobic biodegradation of landfill leachates proceeds with carbohydrates initially utilized, followed by fatty acids, amino acids and finally humic materials (with molecular weights > 50000) in that order. Nitrification of NH_3 is also accomplished. The unmetabolized refractory organic matter remaining may typically be fulvic acid-type substances with molecular weights between 500 and 10000. At appropriate organic loading levels and sufficient retention times, COD and BOD stabilization efficiencies can be almost quantitative (97–99%) along with effective removal of heavy metals. However, nutrients such as N and P have to be added to attain these results if organic loadings are too high and retention times too short.

Anaerobic digestion is more economical for the biological stabilization of leachates since these systems do not have the high energy requirements associated with aeration in aerobic digestion. Anaerobic digestion also yields CH_4, which can be sold as a heat or power source. The disadvantages of anaerobic treatment are the long start-up period, poor quality of effluent, and greater sensitivity to variable organic loads and toxic substances. If longer retention times are used and optimum temperature is maintained, BOD removal can be considerably improved. Heavy metal removal can be good, but generally not as efficient as in aerobic processes.

The concentrations of organic and inorganic constituents in leachates generally decrease with the age of the landfill and its age can be correlated to the ratios of COD/TOC, BOD/TOC and VS/FS (volatile solids/fixed solids). These ratios tend to decrease as the age of the landfill increases. For instance, COD/TOC may vary from about 3.3 for the relatively new landfill to 1.16 for an old one. Measurement of these ratios is therefore useful in determining the most efficacious leachate treatment methods. Older landfills typically produce leachates in which the easily oxidized organics are largely absent. The more refractory compounds remaining and not amenable to decomposition by aerobic or anaerobic digestion may be more easily removed by physical/chemical processes.

Physical/chemical methods

Whereas physical/chemical treatment methods are not effective for organic removal from leachates from freshly decomposing refuse, good results are obtained with those from old landfills or biologically stabilized leachates Table 9. Venkataramani et al. (1984) summarize the effectiveness of several such methods as related to the nature of the leachate being treated. Lema et al. (1988) have summarized the practical considerations in the use of physical/chemical leachate treatment methods. Although lime is most commonly employed for chemical precipitation, alumina, $FeCl_3$, $FeSO_4$ and polymers have also been used. Elimination of color, suspended solids, NH_4^+ and heavy metals is good, but reduction in COD is at best 40% and considerable amounts of sediment are produced due to the chemicals added. Chemical oxidation can be effected with $Ca(OCl)_2$, O_3, H_2O_2, Cl_2 or $KMnO_4$, any of which is more effective than precipitation methods for color removal, but the reduction in COD is only about 50%. Also, inorganic halogen-containing oxidants may produce toxic organohalogen compounds.

Reverse osmosis involving the passage of leachate under pressure through cellulose acetate or other membranes efficiently filters out inorganic ions and organic matter. The leachate must be pretreated by liming to pH 12 and then

TABLE 9

Relationships between the nature of landfill leachates and the expected efficacy of various physical or chemical methods of organic removal

Nature of leachate				Efficacy of treatment processes						
COD/TOC	BOD/COD	Age of landfill (years)	COD (mg l^{-1})	Biological treatment	Lime precipitation	Oxidation		Reverse osmosis	Active carbon	Ion exchange resin
						$Ca(ClO)_2$	O_3			
	> 0.5	New (< 5)	> 10 000	Good	Poor	Poor	Poor	Fair	Poor	Poor
2–2.8	0.1–0.5	Medium (5–10)	500–10 000	Fair	Fair	Fair	Fair	Good	Fair	Fair
< 2.0	< 0.1	Old (> 10)	< 500	Poor	Poor	Fair	Fair	Good	Good	Fair

adjusted to pH 3–6 with H_2SO_4 to remove, respectively, colloidal matter and $CaSO_4$ to prevent membrane clogging.

Activated carbon in columns or as a powdered batch additive achieves satisfactory removal of organic matter, especially with old, stabilized leachates. Disadvantages are the frequent need to regenerate the columns and an equivalently high consumption of powdered charcoal. Although ion exchange resins as a polishing step may remove considerable quantities of organic anions and are particularly good for inorganic ions when operated in the mixed bed mode they have been found to be uneconomical when the total dissolved solids (TDS) in the solution exceeds 200 mg l^{-1} (Venkataramani et al., 1984).

Many other treatment methods for landfill leachates have been used and investigated. Raising the pH of leachate to about 11 with lime and bubbling in air (stripping) can accomplish removal of NH_3 (Venkataramani et al., 1984). Humic (Yamazaki et al., 1983) and fulvic (Sawai et al., 1989) acids in leachates have been converted by gamma irradiation to more biodegradable forms. Genetically engineered microorganisms (Loper, 1989) developed to degrade specific toxicants may facilitate leachate treatment. The decontamination of landfill leachate by spraying it on plants as a form of irrigation has been studied (Menser et al., 1979a), but heavy metal uptake by such crops is undesirable.

EFFECTS OF MUNICIPAL REFUSE LANDFILLS ON THE ENVIRONMENT

A number of studies has dealt with effects of municipal refuse landfills and their management practices on the environment. It is pertinent to review this area of research dealing with the several segments of the environment which have been investigated.

Soils and plants

Effects of refuse or leachates on soils

One method being studied for disposal of refuse or landfill leachates is their incorporation or addition to the soil as a source of nutrient elements for plant growth. In theory their purification will be effected by bacterial activity, filtration, evaporation, ion exchange adsorption and other processes in soils. Thus sandy soils will favor the oxidation of organic matter and clay soils the removal of heavy metals. Disposal of collected landfill leachates (Winant et al., 1981; Chan, 1982) or refuse compost (Giusquiani et al., 1988) by incorporation in soils predictably alters soil properties as determined by the composition of the respective waste. Nutrient element and organic matter enrichment, increases in the concentrations of heavy metals and sometimes drastic changes in pH have been reported. Giusquiani et al. (1988) reported significant increases in the solubility of P when refuse compost was incubated with soils. This may be attributed to the formation of phosphohumic complexes and coating of sesquioxide particles by humus, both of which might reduce the P-fixing capacity of the soil. They also reported increases in exchangeable K, and available Mn and Zn concentration. Gordon et al. (1988) reported a significant decrease in microbial biomass in a forest soil to which landfill leachate was applied. This may have been due to waterlogging of the soil or to toxicity to the bacteria since the leachate contained several organic solvents including 2 ppm of toluene.

Effects on plants of refuse application to soils

Several papers have been published dealing with the effects of amendments of composted or simply pulverized refuse to soils on plant growth, nutrient or toxic element uptake or nutrient leaching. The available N and P in sewage sludge partially corrects for the typically wide C/N ratio in refuse so the two are commonly mixed before composting. Finely shredding the refuse gives best results, but this process is expensive (Phung et al., 1977). Jokela et al. (1990) reported up to a 70% increase in tree growth (stem wood biomass) of slash pine treated 16 years earlier with municipal garbage composted with sewage sludge. Increased concentrations of N, P, K, Ca, Mg, Cu, Al, Fe and B were found in tree tissues. Boron uptake by ryegrass grown on refuse compost-amended soil has also been reported (Nogales et al., 1987). Heavy metals such as Cd, Pb, Cu and Zn may be absorbed by crops grown on refuse-amended soils depending on rate of refuse application, resulting soil pH and the nature of the crops (King et al., 1977). Low soil pH enhances heavy metal uptake. Leafy vegetables typically absorb higher concentrations of heavy metals (Chu and Wong, 1987; Berthet et al., 1989).

Effects of leachate application on plants

A number of studies has involved the application of landfill leachates to soils and their effects on plant growth and element uptake. Gordon et al. (1989a) investigated the effect of landfill leachate application on red maple and sugar maple seedlings in a greenhouse pot study. Whereas no increase in the height of red maple was observed, stem diameter was significantly increased with irrigation with untreated leachate (diluted to 75% with water) or pretreated (lime, activated carbon) leachate. Iron foliar concentrations increased in red maple, which received the untreated leachate both as a foliar and soil application. Depending on the treatment, Cu and Ca foliar levels decreased, while Mn concentrations remained high. Leachate application did not alter foliar levels of P, K, Mg, B or Zn. Sugar maple seedlings which received saturation drainage cycles with undiluted or untreated leachate showed symptoms of vegetative stress within 24 h and 100% mortality after five such cycles. The authors stress the importance of proper control of such leachate irrigation owing to the rapid response of forest vegetation to altered chemical environments.

Shrive et al. (1990) studied physiologic and spectral responses of sugar maple sapling leaves in a forest spray-irrigated with landfill leachate. Photosynthetic rates were reduced 34–53% and water use efficiency dropped an average of 70%

as a result of the treatment. Lower leaves showed distorted spectral patterns due to the accumulation of leachate precipitates on the adaxial leaf surface. Similarly, Gordon et al. (1989b) studied the effect of leachate spraying in a hardwood forest. Sugar maple and American birch variously showed increases in N, P, Mg, K, Fe and B concentrations in foliar leaves as a result of the application. A significant increase in diffusive resistance and decrease in transpiration rate was found in the foliage of the treated trees.

In studies with forage plants, Nordstedt et al. (1975) reported no adverse effects on pasture grasses when landfill leachate was applied. In most instances, increases in Ca, Mg and P in the soil occurred. Menser et al. (1979a) spray-irrigated landfill leachate onto forage grasses including reed canarygrass, tall fescue, orchardgrass, bromegrass and bermudagrass. The elements Na, Fe, Mn, Cl and S increased in all except orchardgrass. Liming the crops limited concentration of Mn by the grasses. Since the leachate was low in heavy metals such as Cd, Ni, Cr, Pb and Co, accumulation of these by the grasses was well below limits considered toxic for plants or foraging animals. Menser et al. (1979b) also showed some accumulation of heavy metals by smartweed and ragweed when irrigated with landfill leachate, but not to toxic levels. Smartweed concentrated high concentrations of Mn, however.

In a greenhouse study, Wong and Leung (1989) reported higher yields of Chinese white cabbage and Chinese flowering cabbage when they were irrigated with landfill leachates diluted to 5, 10, 20 and 40% with water. Yield was reduced, however, for acacia and inhibition of root growth was found for all three plant species. The plants showed increased uptake of N, Na, Fe and Mn and the soil showed higher levels of ammonia N, nitrate N, total N, exchangeable Na, P and electrical conductivity as a result of the leachate treatment. Menser (1981) obtained variable results in a pot study in which soybean plants were irrigated with leachate diluted up to 20:1 with water. Growth inhibition, lowered plant concentrations of N, P and K and higher concentrations of Mn were found as compared with control plants, which received Hoaglands nutrient solution. The author concluded that the use of landfill leachate for the hydroponic culture of plants would require proper regulation to avoid imbalances in micronutrient (Mn, Fe, Zn) concentrations. Menser and Winant (1980) grew bushbeans, lettuce and radish in the field and in the greenhouse in sand culture with treatments including irrigation with increasing dilutions of landfill leachate and no leachate irrigation (controls). Normal levels of essential elements and non-toxic concentrations of toxic metals were found in plants. The only sign of phytotoxicity observed was stunting and chlorosis in the primary leaves of bushbeams, which may have been caused by excessive Mn concentrations.

Although some successes have been reported as regards increased growth rate of trees and other plants, it is difficult to recommend the application of landfill leachate to agricultural soils as a means of its disposal and purification. The composition of leachate is highly variable with time and location. This, coupled with numerous soil types and possible plants, the latter varying greatly in their sensitivity to phytotoxic constituents and ability to concentrate toxic elements, makes proper management of this practice extremely complicated and expensive. Furthermore, most studies on this subject have been relatively short term and the long term effects of application of leachate, which must be continuously disposed, is unknown. As stated by Lema et al. (1988), "It seems that spraying leachate on land is a noxious practice which courts the risk of polluting groundwater and runoff and may be toxic to plants."

Establishing cover crops over refuse landfills

A problem facing landfill contractors is to successfully revegetate the cover material over refuse landfills to reduce erosion, improve appearance and perhaps restore the area for other uses. As noted by Duell et al. (1986), the establishment of vegetation on closed refuse landfills is known to be generally precarious. Such revegetated sites typically contain irregular areas with no vegetation or dying vegetation. The cause is often landfill gases such as CH_4 and CO_2 emerging to the surface producing fowl smelling and darkened anaerobic soil. In some instances, a red-rust line of Fe oxide delineates the aerobic soil from anaerobic soil below. These gases can comprise 50% of the soil's gaseous composition. The amount of gas generated by refuse landfills will vary with the percentage of putrescible or volatile material in the refuse and the age of the landfill (Gilman et al., 1985). Such anaerobic soils have been reported to show higher pH values owing to denitrification and accumulation of NH_4^+ and elevated levels of Ca, Fe, Mn, Cu and Zn. Changes in Mg, K, P, and B were inconsistent. Elevated temperatures sometimes occurring in these soils can also inhibit plant growth (Flower et al., 1977).

A number of steps have been taken to minimize those factors inhibiting plant growth. Venting of landfill gas to the atmosphere (or for purification and sale) to obviate its diffusion into the plant root zone is probably the most successful approach to preventing phytotoxicity (Spreull and Cullum, 1987). Construction of barriers, including layers of clay, plastic or placement of soil deep below plant roots to prevent gas migration, can also be advantageous. Supplemental fertilization and irrigation is often necessary since the normal uninterrupted soil profile which brings water and nutrients from subsoil by capillarity has been replaced by fill (Flower et al., 1981). Application of municipal wastewater sludge to the soil cover, which provides N, organic matter and increased water holding capacity, has also been found advantageous for revegetating and reclaiming such areas (Wilson et al., 1985). Finally, selection of plant species which may more easily adapt to landfill cover conditions has been studied. For instance, blackgum (*Nyssa sylvatica*) is quite resistant to undesirable landfill cover conditions, whereas rhododendron (*Rhododendron* "Roseum Elegans") is very susceptible (Duell et al., 1986). Some plants do well the first year they are planted on new soil cover over landfills, but show diminished vegetative quality in subsequent seasons (Shimell, 1983). Short rotation trees may be successful, especially if the soil cover is of high quality and the refuse is old

(Ettala, 1988; Ettala et al., 1988). If grasses fail to grow on a landfill cover, trees will fail also unless a gas barrier is installed (Duell et al., 1986).

GROUNDWATER

The contamination of groundwater by leachate plumes has received more study than any of the other possible adverse environmental effects of landfills. As early as 1932, the contamination of groundwater by dumps was recognized (Garland and Mosher, 1975) and numerous studies of these episodes have been published (Walker, 1969; Emrich, 1972; Murray et al., 1981; Robinson and Lucas, 1984; Baxter, 1985; Russell and Higer, 1988; Borden and Yanoschak, 1990). Such polluted groundwater will typically contain inorganic ions or organic constituents contained in the original refuse as well as organics produced by microbial decomposition. Total dissolved solids, Cl^-, SO_4^{2-}, and hardness are often significantly elevated above background concentrations in uncontaminated water (Coe, 1970). Iron and Mn, which can be high, may impart a brownish color to laundered goods and a bad taste to beverages (Garland and Mosher, 1975). High Fe and Mn levels can result from reducing conditions, which solubilize iron by reduction of ferric and manganic compounds in the refuse and later by the reduction of Fe and Mn oxide coatings on sand particles in the aquifer (Nicholson et al., 1983). Contaminated groundwater may even become unsuitable for industrial use if it results in scaling or corrosion.

Movement of contaminants in groundwater is complex and can extend distances of up to a kilometer or more. Plume movement can be inadvertently hastened by increasing the velocity of groundwater flow due to intensive operation of well pumps downstream (Garland and Mosher, 1975). Contamination of such wells can sometimes be alleviated by installing strategically positioned new wells which are counter pumped so as to steer the plume away from the affected area (DeWalle and Chian, 1981). Predicting the movement of landfill plumes requires study not only of the general regional but also the local geohydrology of the area immediately surrounding the landfill (MacFarlane et al., 1983). Local geohydrology can, in some instances, control the plume geometry and pollutant movement. Local factors such as differences in aquifer permeability, interference from impeding strata or corrugation in the underlying bedrock can be the determinants (Sawhney and Raabe, 1986).

A number of physical and chemical methods has been employed to determine the presence, magnitude, age, characteristics and movement of landfill plumes. Drilling wells and withdrawing samples at increasing depths for analysis provides approximations of these parameters. Conductance measurements as an indicator of electrolyte levels in water as well as analysis for specific constituents are useful. Chloride is often determined as it is generally considered an indicator ion for tracing leachate movement since it is largely unaffected by subterranean fixation reactions (Sudicky et al., 1983). However, Cl^- is not a desirable parameter for monitoring landfill leachate movement near saline surface waters (Rule, 1979). Similarly, Murray et al. (1981) used a

LiBr spike to follow a landfill plume since these ions are relatively insensitive to subsurface removal and their low background levels permit detection under these conditions of high dilution. They also determined the ratio of inorganic N to total dissolved P as an indicator of the extent of leachate contamination since this ratio approximates 20 for leachates and about six or somewhat higher for domestic wastewater. The authors caution, however, that these ratios could be altered in some instances by possible loss of P or NH_4^+ by adsorption or ion exchange processes on clay minerals. Temperature measurement has also been used to locate a plume beneath a landfill since it may be about 5°C above background (MacFarlane et al., 1983). Determining the age of leachates has also been studied by analysis of the tritium concentration in the groundwater (Egboka et al., 1983). The test is based on the very low background levels of tritium in a groundwater prior to 1953, after which major atmospheric tests of nuclear weapons began.

Preventing groundwater pollution is crucial. The time necessary for an aquifer to cleanse itself may be decades and artifically removing contaminants from a groundwater reservoir is extremely expensive if not impossible (Garland and Mosher, 1975). Attempts to estimate the risk to humans associated with organic constituents from landfill leachates contaminating drinking water have been made, but predicting outcomes such as increased occurrence of cancer may be in error by more or less than an order of magnitude (Brown and Donnelly, 1988).

FISH AND AQUATIC ORGANISMS

Landfill leachates are highly toxic to fish and aquatic organisms. The few studies that have been done have compared leachate toxicities among different organisms, attempted to identify the high impact toxicants present and the effectiveness of leachate treatment procedures in reducing toxicity. *Daphnia pulex* gave results comparable to rainbow trout (*Salmo gairdneri*) and sockeye salmon (*Oncorhynchus nerka*) when used to measure LC_{50} values of landfill leachates. Regression analyses showed that Zn and tannin were important determinants of toxicity in the leachate (Atwater et al., 1983). Exposure of rainbow trout to a landfill leachate diluted 200-fold with water showed an adaptation to the stress of this exposure as indicated by a return of their serum cortisol levels to the normal resting range. However, when exposed to a 20-fold dilution, their cortisol remained elevated for the full term of the study, indicating an inability to adapt to this stress. Histologic examination tended to corroborate these results in that structural interrenal changes did not appear in the former group (200-fold dilution) until 7 days of exposure, but appeared on Day 2 with the latter (20-fold dilution group (McBride et al., 1979).

Landfill leachate was found to be highly toxic to aerobic luminescent bacteria (*Photobacterium phosphorium*), moderately toxic to zooplankton (*Daphnia magna*) and slightly toxic to fathead minnows (*Pimephales promelas*) (Plotkin and Ram, 1984). The authors point out that the considerable variation between organisms demonstrates the importance of conducting toxicity tests using several organisms from different trophic levels to ascertain the potential impact of a pollutant discharge on an aquatic ecosystem. Wong (1989) showed that the use of alum [$Al_2(SO_4)_3$] to reduce total solids in a landfill leachate significantly reduced its toxicity to a freshwater fish, tilapia (*Sarotherodon mossambicus*). Using rainbow trout (*Salmo gairdneri*), Cameron and Koch (1980) found that 94% of the observed toxicity of a landfill leachate could be explained by variations in its content of unionized NH_3, tannin, copper and hydrogen ion concentration. Aged leachate or that produced during periods of high precipitation showed reduced toxicity. Leachate recycled through the landfill or that which was treated with peat and combined physical/chemical treatment also reduced toxicity to the trout.

HUMANS

Pathogens

Ultimately, humans can be adversely affected by disposal of wastes in landfills. There are several possible modes of human exposure to pathogens, toxicants and gases during refuse collection, landfilling and thereafter. Several studies have been published on the identification of pathogens in refuse to which garbage collectors and handlers may be exposed. Peterson (1974) reported the isolation of poliovirus from soiled disposable diapers and considered them a potential human health hazard if disposed with solid waste without prior disinfection. Pathogens potentially may derive from medical and veterinary wastes and also sewage sludge. In a comprehensive survey of pathogenic bacteria in household refuse as well as wastes from small and large hospitals, Jager et al. (1989) reported essentially no differences in counts between the two. Küster and Schmitten (1981) reported the presence of antimicrobial substances in fresh garbage, but the biocidal effect diminished during the rotting process. As pointed out by Keene (1989), the conditions necessary to transmit infectious material (cause human disease) include the presence of a pathogen of sufficient virulence, sufficient numbers of the pathogen, a port of entry into the body and a susceptible host. There is concern, however, about inadvertent skin punctures by improperly sterilized and discarded medical syringes. Although one study indicated that garbage-exposed workers may be at risk of leptospirosis (Clark and Linnemann, 1986), there is no hard evidence, however, that significant numbers of garbage handlers have contracted infectious diseases as a result of their occupation (Pahren, 1987).

In studies of the survival of pathogens in refuse that is composted, it was found that most organisms succumb within 1 h at temperatures between 55 and 60°C. This temperature range is attainable during composting, but the material must be regularly mixed. Pathogens may also be reduced by antibiotics produced by actinomycetes and fungi present as well as competition for nutrients in such mixed cultures. Elevated levels of airborne pathogens have also been reported in composting facilities where milling and screening operations are conducted (Pahren, 1987). In a Swedish study, compost-exposed workers had higher levels of endotoxin antibodies and it was postulated that symptoms among workers may have been related to endotoxin exposure.

Donnelly et al. (1982) found high survival rates of fecal coliforms and streptococci in household or hospital refuse that had been placed in lysimeters 9 years earlier. They also showed survival of a number of pathogenic indicator organisms in refuse in a landfill after the same time period. Others have also reported long-term survival of pathogens in landfills (Pahren, 1987).

Rahkonen et al. (1987) studied the presence of pathogens in air to which workers at landfills may be exposed in Finland. Exposures were highest at the largest landfills and in warm windy weather. It was recommended that dust be minimized by wetting the refuse, that the cab of the landfill compactor contain pressurized air and that any in-flowing air be filtered to protect the driver. In an epidemiologic study by Hertzman et al. (1987) of workers or the residents near a landfill in Ontario, Canada, multiple criteria were used to identify possible health-related problems, but those with the highest credibility included respiratory, skin, narcotic and mood disorders. Although the validity of the results was compromised by several factors, the authors concluded that the adverse effects seen were more likely due to chemical exposure than perception of risk.

Landfilled refuse must be covered with soil and compacted daily to discourage build-up of populations of potentially disease-carrying flies and rodents (Anderson, 1964) as well as fires (Anonymous, 1981a). Rodents can be controlled using poisoned baits (Anonymous, 1987). Milling refuse before landfilling controls flies and rodents by diluting and dispersing food particles and complicating burrowing, but it is expensive (Ham, 1971).

Gases and odors

The production of CH_4 and toxic and odorous gases in refuse landfills and their movement horizontally and vertically are potentially serious problems for residents adjacent to landfills and moreso for possible occupants or workers in buildings constructed directly above such landfills. The problem is exacerbated by the deposition of ever more organic wastes (as compared with the ash remaining in landfills when open burning was permitted), the construction of ever larger landfills, more efficient capping, which promotes lateral diffusion of gases, and the increasing demand for building space (Parker, 1987). Horizontal migration of gases has been reported up to 400 m from a landfill (Emberton and Parker, 1987). Since low percentages (5–15%) of CH_4 in air may be explosive and have been known to enter buildings through cracks in below-grade walls or through service ducts such as sewers or conduits and cause explosions, fires and injuries, a number of steps can be taken to minimize the risk. These have included installation of vertical perforated plastic pipes to the bottom of the landfill equipped with pumps to remove such gases, which are flared off above the surface. These are ideally equipped with shutdown alarms. Such systems have been used very successfully to control volatile emissions from landfill sites (Walsh et al., 1988). Well pipes are also installed at the periphery of the landfill to permit monitoring for possible horizontally diffusing gases (Stearns and Petoyan, 1984). One can also install a ventilation trench filled with coarse gravel around the periphery of the deposited waste (Parker, 1987). Individual buildings constructed over landfills should be built on a concrete slab under which is a ventilated gravel or a heavy plastic liner. Surface cracks in the slab must be plugged (Emberton and Parker, 1987).

A number of toxic, odorous and corrosive compounds have been identified in gases emanating from refuse landfills. Toxic organics have included vinyl chloride (Emberton and Parker, 1987; Walsh et al., 1988), PCBs (Murphy et al., 1985), and a wide range of aromatic and aliphatic compounds and their halogenated derivatives as well as esters, ethers and miscellaneous other organics (Young and Parker, 1984; Harkov et al., 1985; Morris et al., 1988; Walsh et al., 1988). Gases such as H_2S, NH_3, CO, SO_2, NO, NO_2, NO_x and O_3 have also been reported (Al-Omar et al., 1987). Lodenius and Brannschweiler (1986) studied the possible emission of toxic metals (Hg, Zn, Cd, Cu) from landfills, but results were negative. Although certain of these compounds might not be expected to derive from domestic wastes, it is assumed that many hazardous wastes are deliberately or inadvertently discarded in refuse landfills also. As pointed out by Bass et al. (1990), many household wastes (cleaning agents, paints, pesticides, solvents, detergents) discarded as domestic refuse are hazardous. Organosulfur compounds (Emberton and Parker, 1987), H_2S (Johnson, 1986) and organic acids (Colenutt, 1979) are considered main contributors to disagreeable odors near landfills. Odors may be magnified during periods of low atmospheric pressure when volatilization of gases from poorly capped landfills would be favored (Young and Parker, 1984). Hydrogen sulfide production in landfills may be enhanced when gypsum ($CaSO_4 \cdot 2H_2O$) board from demolished buildings is discarded (Young and Parker, 1984; Johnson, 1986). Chloroorganic compounds which produce HCl upon combustion can seriously corrode engine parts when gas containing these are used to produce heat and electricity in landfill gas utilization plants (Dernbach, 1985).

Settling

Settling (subsidence) of deposited refuse with time is another deterrent to building over landfills (Balmer and Cowart, 1971). Ninety percent of the settling will occur within the first year, but continued settling may occur for 25 years or more (O'Leary and Tansel, 1986b). The composition of the waste and its degree of compaction will affect the rate of settling. Readily decomposable waste will obviously settle faster than construction debris. Settling will occur more rapidly if precipitation is high. It will be slower if the landfill cover is thicker (O'Leary and Tansel, 1986b). If construction is essential, it is best to build using pilings that are resistant to acidic deterioration (Emberton and Parker, 1987), but, if possible, building over putrescible landfill wastes should be avoided (Parker, 1987).

MODERN LANDFILL DESIGN

Final storage versus bioreactor landfills

It is evident that CH_4 production from landfills can be enhanced if factors such as the moisture content of the refuse is optimized by deliberate addition of water from above or below as well as controlling other determinants such as temperature, refuse composition, pH, and compaction. However, owing mainly to environmental concerns about contamination of groundwater by landfill leachate or hazards from migrating gases, the latest approach to landfill design focuses on keeping moisture from entering the refuse bed either from a rising water table below or from precipitation or surrounding runoff above. Elaborate leachate collection systems below and covers above the refuse as well as proper refuse sorting and compaction are therefore specified for newly constructed landfill sites. In this sense these "dry" landfills are considered final storage reservoirs for refuse rather than bioreactors.

Managing leachates

During the past few years, Federal and State regulatory agencies have upgraded their specifications for both refuse and hazardous waste landfills, recommending double linear systems to obviate or minimize leachate migration and allow its collection, monitoring and subsequent treatment (Schevon and Damas, 1986). A typical cross-section of this landfill design (Landreth, 1988) is illustrated in Fig. 13. The top and bottom liners are flexible membrane liners (FML). The recommended thickness is 30 mils. Several polymeric FML materials are available (Anonymous, 1986), but high-density polyethylene has become a choice material in the waste containment industry (Cadwallader, 1985) based on properties such as tensile strength, toughness, durability, chemical and stress crack resistance, ease of repair and cost (Anonymous, 1984; Smith and Parker, 1984). Some liner materials may possess many of the required physical properties, but show unpredictable peculiarities after installation. For instance, rats have an affinity for certain vinyl liner materials and may gnaw through it.

The intact primary (top) liner is designed to prevent leachate from migrating into the secondary liner (below) during the operating period of the facility. Use of these FML liners appears to be the only practical method of accomplishing this (Landreth, 1988). The secondary liner is actually a two-component system. Below this FML is a 36-inch (91.4-cm) layer of compacted, low-permeability soil to minimize migration of any leachate constituents out of the landfill if a breach in the bottom FML were to occur. It should have a hydraulic conductiv-

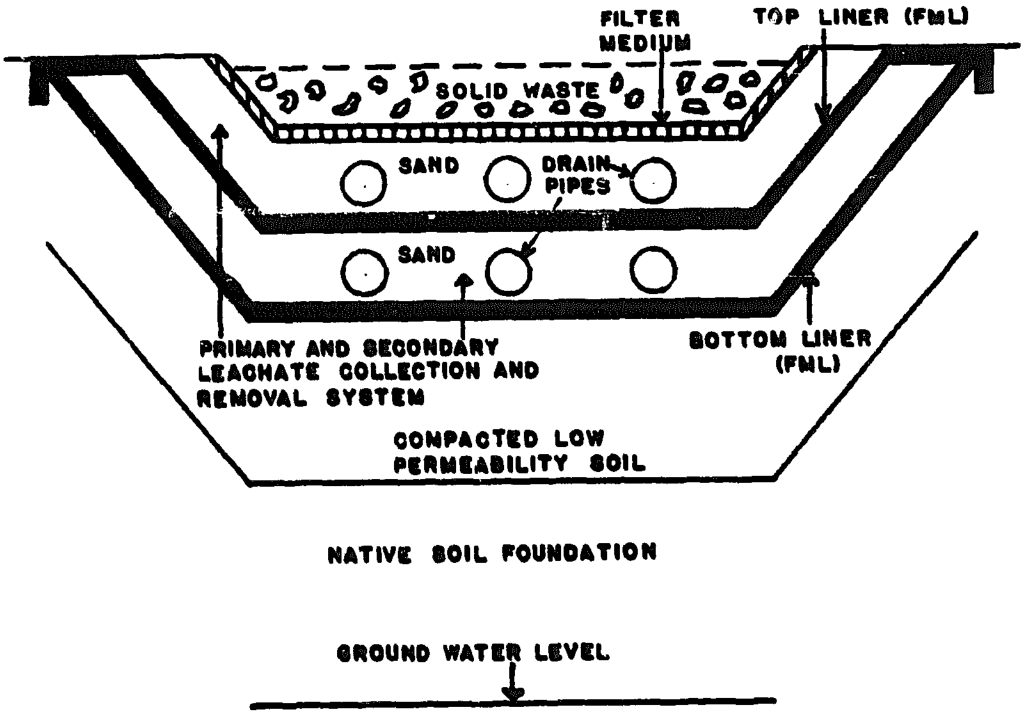

Fig. 13. Cross-section of a dry storage-type landfill with an FML/composite double liner system.

ity of 1×10^{-7} cm s^{-1}. The primary and secondary drainage systems each include a minimum of a 12-inch (30-cm) thick granular drainage layer such as clean sand with a hydraulic conductivity of 1×10^{-2} cm s^{-1} which protects the FML. This sand should meet filter gradation requirements with not more than 5% passing the No. 100 U.S. sieve size and 100% passing a 6.3 mm (0.25 in U.S.) sieve size.

A 6-inch thick (15.2-cm) graded granular layer or synthetic fabric filter is located immediately below the refuse and above the primary FML to prevent clogging. This granular drainage material, which can be sand, should be washed to remove fines before use. Refuse is typically placed in new landfills in discrete cells or units (Vasuki, 1986). The primary leachate collection system should cover the bottom and sidewalls of these cells and therefore permit leachate to be carried away by gravity or by means of a sump pump to a treatment facility. The primary and secondary leachate collection and removal system should be constructed so as to prevent leachate depth above the primary from exceeding 1 foot (30.5 cm). In summary, the primary and secondary leachate collection systems must be able to withstand chemical attack from refuse leachates and the stresses and disturbances of overlying refuse, waste cover materials and landfill operating equipment.

Through the use of these drainage systems, many areas that were formerly considered marginal as landfill sites may now qualify since the leachate may be collected and treated before reaching either surface or groundwater. Even quarries that may have fractured rock floors may qualify as landfill sites, particularly if they are in a groundwater discharge zone (Harris and Naylor, 1973; Goodings and Schram, 1985).

It should be noted that whereas FMLs may be adequately resistant to chemical attack by typical refuse leachate (Haxo et al., 1985), specific organic chemicals or oily wastes that may be inadvertently or deliberately discarded in municipal refuse landfills can severely affect the integrity of these liner materials (Haxo et al., 1986). The impermeability of clay liners, including those containing bentonite, which is due to their swelling properties (Alther, 1983; Jepsen, 1984; Hoeks et al., 1987; Grantham and Robinson, 1988), may also be seriously compromised by specific organic liquids, acids and bases (Morrison, 1981; Anderson, 1982; Hoeks et al., 1987; Johnson et al., 1989).

REFUSE HANDLING

A profile of the types of wastes that a landfill is to receive should precede landfill siting. In particular, sources which may be unknowingly discarding hazardous waste with refuse should be identified and informed (O'Leary et al. 1986a). The nature of refuse components to be landfilled will be modified by recycling programs in the community. Carefully spreading and compacting refuse as it is placed in landfills is most important. Compacting and baling the refuse before placement in landfills considerably extends landfill capacity. Some communities have reported distinct advantages in shredding or grinding refuse prior to landfilling. Such subdivided refuse may be more readily compacted, requires less soil for covering (by avoiding large voids), is far less unsightly and does not attract the numbers of scavenger birds that intact refuse does. The cost in time, manpower and replacement parts for milling equipment increases, however (Franz, 1971; Ham et al., 1971, 1972a, b; Anonymous, 1981b; Salimando, 1987).

LANDFILL COVERS

As well as covering refuse with soil daily as it is placed in landfills to control blowing litter, odors, fire and vectors such as birds, rats and flies, landfill cells must be ultimately sealed when full. The functions of this final cover include those of the daily soil cover but, in addition, to prevent or minimize downward percolation of rainwater and therefore leachate and to promote site reclamation and esthetics. In order to meet these requirements, the cover must be designed to resist water and wind erosion, cold-weather distress, disruption by animals or plants and be stable as regards slumping, cracking and slope failure (Hatheway and McAneny, 1987).

Minimal cover components for a refuse landfill are shown in Fig. 14. The cover is a multilayer system which must be as impermeable as the liner system. It should require minimum maintenance and enhance drainage from its surface while minimizing erosion. It must accommodate settling to reduce the potential for compromising the function and continuity of the cover (Landreth, 1988). As shown in the figure, the cover consists of separate vegetation, drainage and barrier layers above the refuse. The thickness of the layers are recommended minimum values. The vegetation minimizes erosion and promotes removal of excess moisture through evapo-transpiration. The final slope of the top should

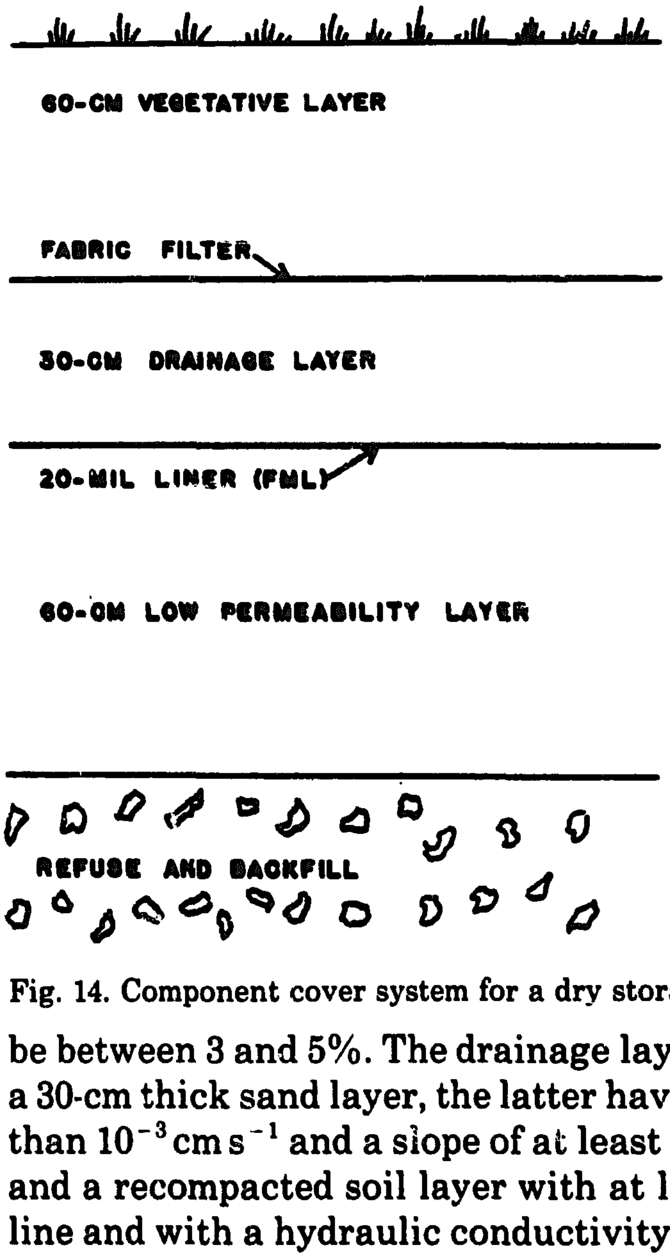

Fig. 14. Component cover system for a dry storage-type refuse landfill.

be between 3 and 5%. The drainage layer may include a geotextile fabric above a 30-cm thick sand layer, the latter having a hydraulic conductivity not greater than 10^{-3} cm s^{-1} and a slope of at least 2%. The barrier layer consists of a FML and a recompacted soil layer with at least a 2% slope, located below the frost line and with a hydraulic conductivity not exceeding 10^{-7} cm s^{-1}. Some landfill designs may include barriers against burrowing animals and deep-rooted vegetation (Johnson and Urie, 1985) as well as gas collection and vent systems (Landreth, 1988). A properly designed and sloped cover will therefore convert rainfall and melting snow into runoff at a rate fast enough to prevent significant infiltration, yet slow enough to prevent erosion. The lateral drainage layer can almost totally obviate infiltration into the refuse below unless the FML has been breached and fractures have occurred in the compacted barrier below (Andersland and Al-Moussawi, 1987; Hatheway and McAneny, 1987; Booth and Price, 1989).

GAS PRODUCTION IN MODERN LANDFILLS

Municipal refuse as collected may typically contain about 25% moisture. Of course, as indicated earlier, compaction of refuse with minimum moisture enhances methanogenesis by optimizing surface contact between microorganisms, moisture and nutrients. Additional water may derive from precipitation during landfilling. Water is also produced by reactions in landfills such as aerobic decomposition of carbohydrates. Methane production has been found to occur in refuse containing only 10% moisture (Hartz and Ham, 1983). Although the rate of methanogenesis under these conditions may be considerably below optimum, gas is being abstracted from several such modern landfills for commercial use. Since gas production from landfills in which water is not deliberately added (in fact an appreciable amount is removed with the gas) is relatively new, the long-term economics of the process is unknown.

CONSERVING LANDFILL SPACE

Source reduction

Whereas proper incineration of refuse can reduce the weight of material by over 90%, other strategies such as source reduction, composting, recycling and production of biodegradable plastics are receiving renewed emphasis and far greater environmental acceptability. The advantages of source reduction can vary. Replacing a food container of one composition by that of another may have little advantage if neither will decompose in a reasonable time in a landfill. Eliminating food packaging entirely typically leads to more spoiled food in the solid waste stream. Conversely, manufacturing so as to increase the useful life of consumer products is beneficial. For example, nylon stockings can be fabricated so as to be far less prone to developing runs or holes, but manufacturers will not produce them fearing reduced sales.

Composting

Composting is receiving renewed attention. Some cities are composting yard wastes (leaves, limbs, lawn clippings). Food wastes from households, food processing plants and breweries are also being composted and sold. In some instances, sewage sludge, if low in industrial contaminants, is included in the mixture.

Recycling

Recycling programs involving source separation of wastes by the consumer are being implemented nationwide. However, material recovery facilities are also being built where community wastes are separated into various fractions. As described by Vasuki and Canzano (1988), this can involve subdividing the solid waste using a hammermill and shredder. Paper and plastics are separated out with air classifiers. Different types of plastics may be sorted by further air classification or flotation methods. Ferrous metals are removed with magnets, glass is separated by screening and flotation processes and nonferrous metals (mainly Al) are removed on a specially designed ramp with permanent magnets of alternating polarity (which induces eddy currents in nonferrous metals) and a high intensity eddy current separator. In 1989, 9 million automobiles, 2 million tons of scrap Cu and 3 million tons of scrap Al were recycled in the United States.

Glass may be recycled into glass containers, reflective paint, abrasive materials, filtering products for water purification, road asphalt and insulation. Rubber from scrap tires has been chipped and used directly or in an asphalt base for road construction (Anonymous, 1989).

Twenty-eight percent of the paper was recycled in the United States in 1989. Paper constitutes the largest portion of refuse, but lends itself to recycling into many products. These include newsprint, insulation, interior wallboard and exterior siding for buildings, corrugated paper, excelsior, cardboard, padded shipping bags, toilet tissue, paper towels, writing paper, envelopes, paper plates and cups. Recycled paper products such as cardboard and insulation which do not have to be white do not require de-inking. This is an advantage since there are extremely few de-inking facilities in the United States presently. Waste paper may also be fermented to alcohol (Green et al., 1988). In some states, newsprint is being shredded and used for farm animal bedding. Cows have been reported to eat such paper bedding (Grieve et al., 1982), however, and the fate of printing inks, if ingested, as regards possible milk and meat contamination is unknown. Twenty-six million tons of used paper was recycled last year.

Much activity is now underway in the recycling of plastics. Plastic bottles are commonly hand-sorted into different types, but an infrared sorting system can also be used based on the specific infrared absorption bands of the particular resin (Brown, 1989). Polyethylene terephthalate plastic (PET) soda bottles are recycled into backing and fibers for carpets, hard hats, fiberfill for sleeping bags and ski jackets and geotextile fabrics for sanitary landfill liners. High density polyethylene (HDPE) milk jugs and detergent containers can be reprocessed into trash containers, traffic cones and many other products. Polystyrene, as used in fast food containers, can be converted into building insulation and packing material. Waste polyvinyl chloride (PVC) leather and polyurethane have been recycled into molded arm and head rests in automobiles (Miyama, 1988).

Plastics can also be mixed (co-mingled) during recycling. Thus low density polyethylene, HDPE and PET are mixed during recycling to produce speed bumps, park benches and plastic lumber, which replaces pressure-treated lumber (Brown, 1989). However, such plastic mixtures must contain a substantial proportion of virgin plastic material to make an acceptable product. It may be possible to blend many such plastics prior to recycling in the future by addition of "compatibilizers" (chemicals that form bonds between the molecular chains of different polymers).

Plastics are estimated to contain about 28% of the Cd and 2% of the Pb found in refuse. Therefore, recycling of plastics into food containers or packaging requires careful presorting to exclude such materials. The aesthetics of such processes must also be considered, e.g. the concern that recycled plastic that contacts food may have formerly been incorporated in disposable diapers.

Ash from refuse incinerators may also be recycled. Since the ash remaining after incineration of refuse (or sewage sludge) is often considered hazardous and is disposed by landfilling, there is concern that heavy metals in it may become contaminants in landfill leachates. Researchers are therefore incorporating such ashes in cement, ceramics or glass to immobilize such metals before landfilling or to develop products useful in construction or for decorative purposes (Carbone et al., 1989). The disadvantage of landfilling such products is that the addition of other materials to the ash increases its bulk and uses up additional landfill space.

Biodegradable plastic products such as plastic film used as a mulching material in vegetable production incorporate light-activated chemicals that cause the polymer chains to break apart (Gilead, 1985). At the end of the growing season, the mulch material remaining on the surface can purportedly be plowed under for final decomposition by soil microorganisms. Research is also underway to produce plastic film incorporating starch (Studt, 1990). When microbes in soil or water utilize the starch, the plastic is mechanically weakened and fragments. An internal catalyst reacts with the polymer causing further fragmentation, promoting greater surface area for microbial attack. However, in landfills where light is absent and moisture is limiting, biodegradation will be minimal.

Limitations to recycling

The total percentages of many recyclable materials that are reprocessed are still relatively small. As with all manufacturing activities, economics will be the deciding factor in the future of recycling. Aluminum recycling is probably most successful since there is a continuing demand for it and the cost of producing aluminum from recycled material is only a small fraction of that of starting with the mining of bauxite. Ferrous metals can be efficiently removed (70–90%) from solid waste, but its sale is difficult owing to the current economic state of the United States steel industry. More mini-mills capable of melting such scrap are needed and the export market is very cyclic. The percentages of some metals that were recycled in the United States in 1989 were Al, 32; Cu, 43; Pb, 55; and Zn, 19. Sixty million tons of used Fe was also recycled during 1989. The recovery of high-purity glass is technically feasible, but the cost of production far exceeds the current market value. The sale of glass is often dependent on the nearness of a glass factory or the cost of transportation becomes prohibitive. The continued use of glass containers also competes with the manufacture of plastic and aluminum containers. Care must also be taken in proper sorting of glass for recycling. For example, container glass must be melted separately with no borosilicate, plate glass, noncontainer glass or porcelain present. A single piece of porcelain in molten container glass can cause an explosion and workers have been killed in such incidents.

If paper is to be recycled and mills are going to retool to handle these secondary fibers, more users must be willing to buy the resultant products.

Demand presently undergoes sharp unpredictable cycles affected by the value of the dollar versus foreign currencies and profitability to shippers. Shippers can charge more if virgin paper products are being shipped than those made from recycled paper.

The quality of composted material can be affected by the presence of heavy metals or organics such as PCBs. The sale of composted wastes must usually rely on local demand and is often given away. Interstate regulation of compost complicates obtaining permits for such sales. One practical use is as a replacement for topsoil over a refuse landfill (Vasuki and Canzano, 1988). Finally, recycling processes must control their own air and water pollution as well as by-product disposal problems. More stringent environmental laws necessarily push up the costs of recycling also.

FUTURE OUTLOOK

As summarized by Baccini and Lichtensteiger (1989), the future problems associated with various methods of handling solid wastes cannot be accurately predicted. We do not know enough about the specific reactions which occur in refuse landfills whether operated in the bioreactor or final storage mode. Synthetic liners used as barriers to the entry of water or egress of leachate may last for decades, but not forever. We need to learn more about:

(i) The chemical and physical (crystalline or noncrystalline) composition of the refuse,
(ii) The geotechnical and environmental properties of landfill sites, and
(iii) The methods to be used to monitor the landfill.

Ideally, the waste placed in landfills should be (i) inorganic, (ii) in an oxidized form, and (iii) solid and poorly soluble in water. Incineration of refuse may be a first step to achieve these latter properties. Finally, it may be possible to modify consumer products to facilitate their recycling or at least to enable their safe and facile incineration or rapid decomposition when landfilled. In this regard, it has been suggested that, in the future, closed landfills might be reopened and "mined" to remove materials that could then be recycled or incinerated. The emptied landfill would then be restructured to serve as a modern "dry storage" landfill for disposing strictly the above inorganic, oxidized or water-insoluble type wastes. High cost might be the impediment here, however.

The solution to the nations solid waste problem in the future will have to include recycling, incineration and landfilling. The solid waste stream will always include materials that cannot or will not be recycled as well as those that are noncombustible. These and ash, which is the ultimate by-product of refuse incineration, will still have to be disposed by landfilling.

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