CHAPTER 2

LITERATURE REVIEW

2.1 Landfill

Disposal of solid waste on-land is the most common method for municipal refuse in most developing countries. Definition of landfill by Mc Graw-Hill Dictionary of Engineering (1984) is: "Disposal of solid waste by burying in layers of earth in low ground". This practice has given various terms like "tips" and "controlled tipping" in U.K., "sanitary landfill" in U.S.A., "coups" in Scotland and "dumps" worldwide. Landfill is the cheapest means of disposal, only if suitable land is within economic range of the source of the waste.

As indicated in Fig.2.1, the waste can be disposed of either above the existing surface, below the surface, or both above and below. Above the ground disposal is particularly attractive for sites with shallow water tables. Above-ground landfills have the advantage that leachate can be drained by gravity, the facility is conspicuous and therefore is not easily forgotten and ignored, and construction of liner and drainage facilities call for a single composite liner with a leachate collection system (David, 1993).

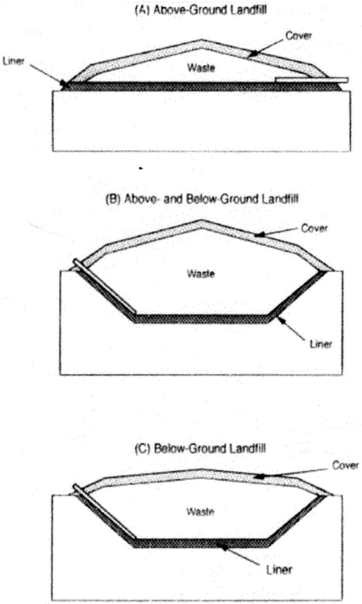


Fig. 2.1 Landfill Types: (a) aboveground (b) above and below ground (c) below ground (Source: Daniel, 1993)

On the basis of this classification, three hydro-geological categories have been suggested as an aid to landfill selection (Fig 2.2) (Mather, 1976a):

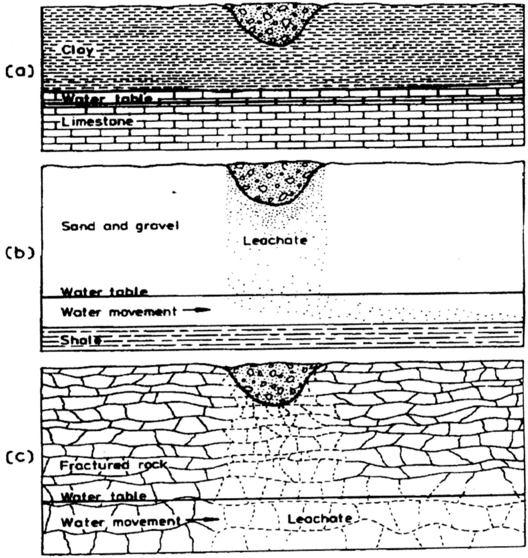
(1) those sites which provide a significant element of containment for wastes and leachates,

(2) those which allow slow leachate migration and significant attenuation, and,

(3) those which allow rapid leachate mitigation and insignificant attenuation.

2.1.1 Class 1 sites

Sites providing a significant element of containment are situated on relatively impermeable strata such as soft clays and marls or fine-grained compact rocks of low permeability such as slates, shales, and mud-stones (Fig. 2.2a). It necessary to inspect the site to confirm that the poorly permeable layer is continuous beneath the whole area of the landfill. The presence of layers of sand, gravel or similar permeable deposits may provide preferential routes for migration of leachate, although their existence may not be apparent from a surface inspection of the site.



(a) Class 1 or containment site. The leachate is contained within the landfill by the impermeable clay.
(b) Class 2 or attenuation site. The sand and the gravel allow slow leachate migration with significant attenuation. The underlying, impermeable, shale stratum protects deeper aquifers.
(c) Class 3 or rapid migration site. The fissured strata allow rapid leachate migration into the aquifer, with insignificant attenuation.

Fig. 2.2 Hydro-geological categories for landfill selection
(Source: Mather, 1976a)

If leachate is allowed to build up at the base of the landfill, several adverse effects may occur. The liquid may fill the site; eventually overflowing the sides polluting surface water or as the pressure increases, the base of the landfill may begin to leak (Mather, 1976b). The saturated waste will biodegrade anaerobically, leading to the generation of methane gas. The use of local, poorly permeable materials for intermediate cover may lead to perching of leachates at different levels in the landfill and thus the possibility of lateral seepage through the walls of the pit. It will also tend to prevent the free escape of landfill gases. This build-up of potentially explosive gas may interface with the reuse of the site after landfilling is complete. In view of these problems, careful management of a containment site is necessary. It is usually necessary to collect the leachate generated and to dispose of it in some way. Sewers may not be available at remote landfill sites, and pretreatment of the leachate may be necessary. However, this adds significantly to the costs of running the site. In addition, care is required to limit the amount of water entering the site, either as infiltrating rainfall or as liquid waste. In dry climates, the use of a poorly permeable top cover would effectively prevent rainfall infiltration and leachate generation, although now the dry wastes would decompose only very slowly, making reuse of the site more difficult.

2.1.2 Class 2 Sites

Sites allowing slow leachate migration and significant attenuation are those taking full advantages of the natural attenuation processes. Although these processes are complex and not completely understood, it is clear that a mechanism exists which will have an effect on most, if not, all-potential pollutants leached from landfill sites, including those derived from many hazardous wastes. An ideal site might be a dry pit in silt or fine sand through which flow is inter-granular and which is underlain by a significant unsaturated zone (Fig. 2.2b). The nearest water abstraction point being some distance away so as to allows dilution of any residual pollution, which reaches the water table. The presence of clay particles dispersed through the unsaturated zone is helpful in providing significant attenuation of metal ions. Sites on fissure strata may be acceptable if infiltration through the joints and fissures is sufficiently slow to allow time for the fissure water to diffuse into static interstitial water. (Mather, 1977).

2.1.3 Class 3 sites

Sites which allow migration of leachates at such a rate that there is insignificant attenuation, will be located on a variety of geological strata, including hard calcareous rocks in which ground water flow is mainly through fissures (Fig. 2.2c). Such sites are normally only suitable for the disposal of relatively inert wastes, although again each site should be treated on its merits. For example, a coastal pit in fractured rock where the aquifer is already contaminated by intrusion of saline water may be acceptable for water disposal (Mather, 1977).

2.1.4 Modern Sanitary Landfills

A modern sanitary landfill is designed to meet specified standards with respect to containment of leachate and gases (Figure 2.3). New design specifications are formulated for minimal impact to the environment, both short and long-term, with particular emphasis on groundwater protection. Landfill site selection is based on geology and soil type, together with groundwater considerations such as depth of the water table. Additionally fresh garbage is covered daily with a layer of soil. The bottom of the landfill is lined with low-permeability liners made out of high-density plastic or clay. In addition, provisions are made to collect and analyze leachate and gases that emanate from the degradation of MSW (Artiola, 1996).

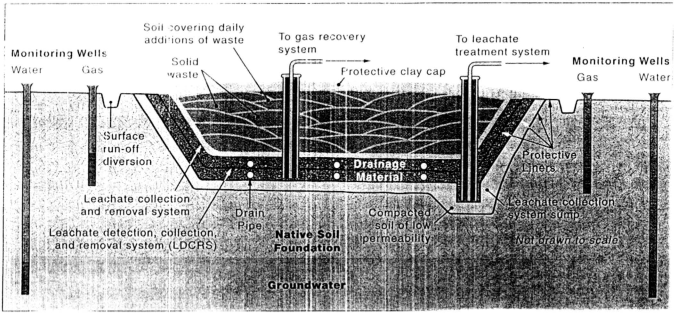


Fig. 2.3 Cross-section of a Modern Sanitary Landfill.
(Source: Artiola, 1996)

Construction of new landfills currently cost up to US $1 million per hectare. Additionally, they require costly permanent monitoring for potential pollutant release to the surrounding environment. Thus many communities are faced with difficult decisions with respect to disposal of MSW – despite the fact that modern landfill design and construction has improved efficiency with respect to pollution prevention (Artiola, 1996)

2.2 Waste Disposal

Municipal solid waste is waste produced by households or by commercial and production activities whose waste is similar to that of households. Most MSW is made up of packaging materials (i.e. glass, plastics, metals, corrugated paperboard, etc.), paper, food, used objects (cloths and utensils), and durable goods (electric appliance, etc.) (Tchobanoglous *et al.*, 1993). Some times MSW is commonly codisposed with household refuse with a wide variety of hazardous and industrial wastes from local factories and chemical companies. [US EPA, 1995].

In 1965, U.S. Congress passed the first national solid waste legislation, and U.S. Congress enacted the Solid Waste Disposable Act of 1965 (PL89-272) in 1965. In October 1976, the Resources Conservation and Recovery Act (RCRA, pronounced rick-rah) was enacted (U.S. Congress, 1976a). This act restricts the disposal of hazardous waste into land, bodies of water or the atmosphere. The Toxic Substance Control Act of 1976 (TSCA) was enacted to regulate the introduction and use of new hazardous chemicals (U.S. Congress, 1976b). Prior to the adoption of the 1976 Resorurce Conservation and Recovery Act, municipal landfills commonly accepted (along with household refuse) a wide variety of hazardous and industrial wastes from local factories, chemical companies and other generators of toxic waste. In 1976, 90% of industrial wastes were codisposed with municipal wastes into landfills (US EPA, 1988a). These wastes included drums of toxic chemicals, liquid waste streams from industrial processes, and large quantities of sludge and other materials contaminated with industrial chemicals. As a result of such nondiscriminatory waste disposal practices, municipal solid waste landfills located in communities throughout the country are repositories of large amounts of toxic and hazardous waste. For example, U.S. municipal landfills contained an estimated 250,000 tonnes of toxic and persistently hazardous Polychlorinated Biphenyls (PCBs) (Liptak and Bela, 1991).

The codisposal of incinerator ash with municipal solid waste in landfills presents a potential environmental threat because the ash contains metals that can dissolve and be mobilized by the acidic leachate in landfills. Incinerator ash used to be routinely discarded into municipal landfills, and under certain conditions codisposal remains a viable waste management practice. Heavy metals may also be deposited in municipal landfills as construction and demolition wastes, discarded automobiles and other transportation equipment wastes.

Malaysia realized the problems of improper disposal of toxic and hazardous waste into landfills few years ago. A survey was conducted in 1985 to determine the types and quantities of hazardous wastes and it was found that approximately 200,000m^3 of wastes was being generated annually (Ibrahim, 1988). It also showed that the waste was created by 700 industrial facilities located mostly on the West Coast of Peninsular Malaysia. These plants can be divided into 31 types, each possibly generating different types and quantities of waste streams. The waste streams can be classified into 3 major groups:

(a) Liquid waste streams, such as acids, alkalis, effluents containing heavy metals, water based paints/ink/pigment wastes etc

(b) Oily wastes, solvent wastes, pesticides wastes, solvent based ink/paint/pigment etc and

(c) Dust ash, slag and sludge.

Currently, Malaysia has one modern disposal site to receive these wastes in Bukit Nanas. Toxic and hazardous wastes in Malaysia are defined by a listing of waste categories, in which 70 categories of waste are currently listed.

2.3 Composition of Municipal Waste

The materials discarded from homes and some commercial businesses have changed drastically over the years. Not surprisingly, the vast cultural and technological changes of the past century have transformed the contents of municipal waste. An ever-increasing rate of waste generation, caused by a well established "consume and throwaway" attitude, has become a staple of the modern lifestyle. Urban waste typically includes paper, plastics, metals, glass, food and food container (like Styrofoam, PVC bags, plastic and glass bottles, metal and Aluminum cans etc.) and other debris (Daniel, 1993).

There are approximately 230 municipal dumping sites in Peninsular Malaysia, an average of 1.8 dumping sites per Municipal council (Kementerian Perumahan dan Kerajaan Tempatan, 1990). The total amount of solid waste generated in major urban area in Peninsular Malaysia is given in Table 2.1. From the table we can see that in 1990 about 36% of the total waste generated was attributable to the Kuala Lumpur area. Also, between the period 1980 to 1990, there is a noticeable increase in the generation of solid waste in each urban area when the generation of solid waste in Kuala Lumpur increased about two fold (IEM/ICE, 1994).

Table 2.1 Solid Waste generated in Major Urban Areas in Peninsular Malaysia.

Urban Center	Solid Waste Generated (Tonnes/day)		
	1970	1980	1990
Kuala Lumpur	98.8	310.5	586.8
Johor Bahru	41.1	99.6	174.8
Ipoh	22.5	82.7	162.2
Georgtown	53.4	83.0	137.2
Klang	18.0	65.0	122.8
Kuala Terengganu	8.7	61.8	121.0
Kota Bharu	9.1	56.5	102.9
Kuantan	7.1	45.2	85.3
Seremban	13.4	45.1	85.2
Melaka	14.4	29.1	46.8

(Source: IEM/ICE, 1994)

The composition of municipal solid wastes reflects the wealth of the society, economic status and their way of life (Wan Abdul Rahim, 1992). Waste composition of the refuse from three different

Malaysian councils are compiled in Table 2.2 (Mohmad Nazri Omar, 1994). Due to higher socio-economic status, Kuala Lumpur generated the largest percentage of paper waste while Petaling Jaya has the largest percentage of plastic waste because of the location of many industries.

Table 2.2 Compositions (%) of the refuse from three different Malaysian councils.

Component	Kuala Lumpur	Petaling Jaya	Penang
Garbage	40.8	36.5	32.8
Paper and cardboard	30.0	27.0	25.5
Wood	3.2	7.0	14.4
Plastic	9.8	16.4	11.2
Textile	2.5	3.1	3.4
Metal	4.6	3.9	2.6
Glass and Ceramics	3.0	3.1	1.4
Rubber	0.2	0.8	-
Others	5.9	3.0	8.0

(Source: Mohmad Nazri Omar, 1994)

The US EPA estimates that approximately 50% of the MSW in the United States is paper products and yard waste, roughly 40% is metal, food waste, and plastics, and the rest is mostly wood, rubber, leather, textiles, and miscellaneous inorganic materials (USEPA 1996). The mixture also contains household hazardous waste from cleaning products, pesticides, paints and finishes, paint thinners, adhesives, and automotive maintenance products. Small businesses such as dry cleaners, machine shops, and automotive repair shops add degreasers, oils, and solvents to the waste mix. In USA between 55 and 65 percent of the municipal solid waste stream originates from residential premises and 35 to 45 percent is commercial waste. (US EPA, March 1996).

2.4 Decomposition of Municipal Waste

2.4.1 Understanding Biological and Chemical Processes in Landfills

The decomposition of waste in landfills is a complex and highly variable process. Since the principal products of waste decomposition, leachate and methane gas may be problematic for surrounding communities, an elementary understanding of the degradation process is advantageous to citizens who are participating in local municipal landfill activities (Stegen, 1985).

Once waste is deposited in a landfill, biological and chemical processes act to change its structure and characteristics. This process of waste decomposition or biodegradation is a natural process. Waste, that is in contact with oxygen and moisture will eventually be broken down by microorganisms into simpler components. A general reaction for this decomposition is:

$$\text{Organic waste} + \text{Oxygen} \longrightarrow CO_2 + H_2O + \text{biomass} + \text{heat} + \text{partially degraded materials.}$$

An example of the complete aerobic decomposition of a waste of general composition $CH_aO_bN_c$ is :

$$CH_aO_bN_c + \tfrac{1}{4}(4a-2b-3c)\,O_2 \longrightarrow CO_2 + \tfrac{1}{2}(a-3c)\,H_2O + cNH_3$$

Aerobic degradation produces carbon dioxide at levels as high as 90 percent and the temperature rises to as high as 70^{0}C. The elevated carbon dioxide results in the formation of carbonic acid in the refuse (as shown in the equation below), thus resulting in acidic pH levels in the leachate (McBean *et al.*, 1995).

$$CO_2 + H_2O \longrightarrow H_2CO_3 \text{ (carbonic acid)}$$

The time frame for waste decomposition varies greatly depending upon the nature of the waste (its physical and chemical composition) as well as the conditions present in the landfill. The process of waste degradation produces by-products that, if not properly controlled, can endanger the health of humans and the environment.

Naturally occurring anaerobic microorganisms biologically break down some waste components at the lower levels, emitting methane, other gases and chemical by-products in the process. Precipitation percolates through the landfill, the flowing water dissolves some materials and picks up a variety of chemicals. Moving toward the bottom or sides of the landfill, this liquid mixture is called "leachate." The leachate and landfill gas produced by municipal landfills may pose serious threats to human health and the environment (Thornloe, 1992).

Leachate and gas production is both influenced by the contents of the landfill, the local climate and the manner in which the facility is operated. With a proper collection systems we can recover the gases produced and estimate their fuel value for energy production. Gases produced during the anaerobic decomposition of wastes contains methane (CH_4) and carbon dioxide (CO_2) as the major components with minor quantities of hydrogen (H_2), hydrogen sulfide (H_2S), nitrogen (N_2) and oxygen (O_2) (Fig. 2.4), (Barry, 1991). For the improvement of existing landfills it is necessary to install gas vent facility as a counter measure for gas production and to achieve a faster stabilization period for landfills (Kementerian Perumahan dan Kerajaan Tempatan, 1990).

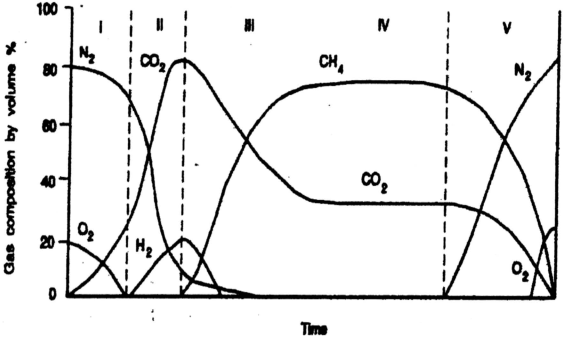


Fig. 2.4 Phases in landfill gas generation
(Source: Barry, 1992)

Generally, the biodegradation is characterized by a five-stage process. The particular chemicals found in the leachate or air emissions originating from a specific landfill area are defined at each of the five stages (California Air Pollution Control Officers Association, 1990) and are discussed in detail below.

2.4.2 Stage I: Initial Adjustment

This period begins when the waste placed in a landfill cell absorbs moisture and settles. Oxygen is present, and bacteria, fungus, and aerobic insects such as mites, millipedes, isopods and nematodes initially decompose the waste. As these organisms consume degradable paper and food products, chemical reactions occur within the landfill. When the landfill cell's oxygen supply is exhausted, anaerobic condition sets in. Aerobic organisms die off and facultative anaerobes and anaerobic bacteria become the principal decomposers. Unabsorbed moisture in the landfill cell becomes leachate. This transition from aerobic to anaerobic conditions occurs between 6 and 18 months after waste is initially placed in a cell (Stegen, 1985).

2.4.3 Stage II: Transition

In Stage II, anaerobic bacteria converts the substances generated by the aerobic bacteria in stage I to acetic, lactic, and formic acids and alcohol such as methanol and ethanol. The many acids produced in this stage move through the waste and appear in the leachate. The resulting highly acidic leachate indicates that waste is in this "transition" stage, which lasts for several months (Gaudy and Gaudy, 1988).

2.4.4 Stage III: Acid formation

Anaerobic decomposition continues and leachate originating from the area undergoing Stage III contains high concentrations of organic acids. As pH values continue to lower, the increasingly acidic solution causes waste metals such as chromium, iron, and manganese to dissolve in the leachate.

Nutrients such as nitrogen and phosphorus become available to the anaerobic bacteria, allowing the microbial population to expand and further break down the waste. During this phase, a diverse mix of anaerobic bacterial species exist in the landfill, with some bacteria consuming the acids and maintaining a stable environment by preventing the pH from dropping excessively low. Fermenting bacteria convert organic acids, alcohol, hydrogen and carbon dioxide to acetate (Stanforth *et al.*,1979).

2.4.5 Stage IV: Methane Fermentation:

Methane-producing bacteria transforms the acetate, methanol, and other chemicals generated in Stages II and III to methane, carbon dioxide, and water. The methanogenic bacteria produces significant amounts of methane and carbon dioxide, with each gas constituting about 50 percent of the landfill gas composition. As these gases are produced, acidic conditions are somewhat neutralized in the landfill. This change in pH causes metals in the waste to change the chemical form, becoming less mobile and therefore less likely to dissolve in leachate. As a result, the metal concentrations in the leachate decline. This stage of methane production may last from 15 to 60 years (Thornloe, 1992). Worldwide, landfills are a significant methane source, contributing approximately 20 percent of total global anthropogenic methane emissions (California Air Pollution Control Officers Association, 1990).

2.4.6 Stage V: Final Maturation

Less scientific data exists concerning the final biodegradation stage of landfills, and descriptions of Stage V rely more heavily on speculation than do description of the previous stages. Scientists assume that after the decomposable materials have degraded in Stages I through IV, bacterial populations will decline, gas and leachate production will end, and the landfill will stabilize. However, landfills built with modern designs have not existed long enough to determine what characterizes a mature landfill and when such a stage occurs. In addition, it is questionable whether the end of biological activity in a landfill signifies an end to its environmental and human health threats. Though modern landfills are built with the intent to protect the public and the surrounding environment from toxic or undesirable by-products of decomposing waste, the current methods of "containment" may inadvertently prolong the process of waste decomposition and lengthen the time period during which leachate and landfill gas can threaten communities (Woodwell, 1984).

After the 30-year post-closure period, however, cracks in the decomposing top cover can allow oxygen and water to enter the landfill again, potentially re-stimulating biological activity in the non-decomposed waste and creating leachate and landfill gas. Since the landfill management is not required to monitor the landfill or to maintain the landfill's top cover and bottom liner after the 30-year period, leachate and methane gas can again become potential hazards to the surrounding community. Leachate can covertly contaminate groundwater, remaining undetected until a dangerous situation develops. According to one landfill researcher, "Landfills represent a significant source of contamination for groundwater *ad infinitum*" (Senior and Subari, 1990).

2.5 Leachate

2.5.1 Introduction

The liquid produced during the decomposition process as well as the water that has seeped through the ground cover and worked its way out of the refuse is known as leachate. Flowing through the waste, leachate transports a wide variety of chemicals to the extremities of a landfill.

When this leachate moves into the surrounding soil, groundwater or surface water, it can cause considerable pollution problems.

2.5.2 Leachate Production:

Leachate forms when liquids originating from rain, melted snow, or waste itself percolates through landfill cells and moves to the bottom or sides of a landfill. The mechanism by which contaminants are leached out is summarized in Fig. 2.5 Flowing through the waste, leachate transports a wide variety of chemicals to the extremities of a landfill (Crawford and Smith, 1985). The quantity of leachate produced by a landfill is highly correlated with the amount of precipitation around the landfill (USEPA, 1988a). In areas with high precipitation rates, the production of leachate can be greater than in drier areas since much of the precipitation percolating through a landfill becomes leachate. The amount of liquid in landfilled waste also affects the amount of leachate the landfill generates. Under certain conditions, groundwater is another source of liquid that contributes to leachate volume. If a landfill is built in a wetland, adjacent to a river, or in other areas where groundwater is not far beneath the landfill's base, groundwater can rise up from below a landfill and provide additional liquid to mix with landfilled waste and produce leachate.

There is also a correlation between landfill size and amount of leachate produced. Large landfills generate greater volumes of leachate than small landfills. According to US EPA, (1988b) estimates, a 100-acre landfill in the northeastern United States can produce 57 million gallons of leachate every year. The largest landfill in the world -- Fresh Kills, in Staten Island, NY-- is over 2965 acres and produces 4 million gallons of leachate per day, or 1.5 billion gallons per year (Gerba *et al.*, 1997).

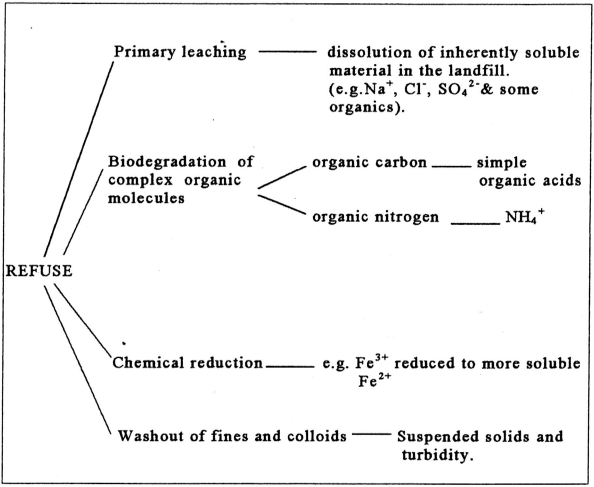


Fig. 2.5 The main mechanisms by which materials leach from the landfill. (Source: Crawford and Smith, 1985)

Primary leaching is the dissolution of soluble salts or soluble organic material that exists in the original fill. The dissolved organic materials such as humic acid will produce a brownish color in the leachate. Much of the original organic material in the fill will have low solubility, but the biodegradation of this material tends to produce more soluble end products such as simple organic acids and alcohol. Although these products may undergo further biochemical reactions to release gaseous end products, some of the soluble organic material leaches out of the landfill. The nitrogen present in the original organic material tends to be converted into ammonium ions, (NH_4^+), which are readily soluble and may give significant quantities of ammonia in the leachate. A landfill quickly becomes anaerobic, thus becoming a chemically reducing environment. This induces oxidized ions such as those in ferric salts to be reduced to ferrous. Ferrous salts are more soluble and thus iron leaches from the landfill. The sulfate in the landfill may be biochemically reduced to sulfides, although this may lead to evolution of small quantities of hydrogen sulfide but is also equally likely to remain in the landfill as highly insoluble metal sulfides. Suspended solids and turbidity may be present in the leachate due to the washout of fine solid material from the landfill.

2.5.3 Leachate composition:

The characteristics of the leachate depend on the nature of the waste, the stage of waste stabilization and numerous other variables. Chian and DeWalle (1976, 1977a, 1977b) and Chian (1977), summarized the chemical quantity of leachate from 30 sanitary landfills. The age of landfill and the degree of stabilization have a significant effect on the waste composition of leachate. The typical compositions of three different landfill leachate at ages 1, 5, and 16 years are provided in Table 2.3. Other factors that contribute to the variation in quality of leachate are solid waste characteristics, composition of waste, size of fill and degree of compaction, the moisture content and degree of rainwater infiltration, temperature, sampling and analytical methods. Keenan *et al.* (1983) and Pfeffer (1986) have reported the average composition of leachate collected each year over a period of four years and is given in Table 2.4. Lu *et al* (1984), analyzed the data of laboratory field test cells at actual landfill sites and developed a relationship between landfill age and various constituents in leachate like BOD, COD, TOC, total alkalinity, calcium, potassium, sodium, sulfate and chloride. Containing hundreds of different chemicals, the chemical composition of municipal landfill leachate varies greatly within individual landfills over time and space, as well as, among different landfills. The rates of biological and chemical activity taking place in a specific landfill or landfill cell can also affect leachate quality by altering the way that waste dissolves in or migrates with leachate (Charnley & Gail, 1988). Based on the results of many studies, Tchobanoglous *et al* (1993) developed leachate characteristic data for 2 and 10 years old landfills (Table 2.5). A summery of published data from investigations in North America, Europe and United Kingdom is shown in the Table 2.6, along with the range of values with respect to metal ion content in leachate samples. The behavior of metal ion content within leachate is not fully understood, variations in infiltration rates and climate may lead to the wider ranges of concentrations.

Though pathogenic microorganisms do not usually appear in landfill leachate at concentrations high enough to cause public health concern, other substances found in leachate and leachate/groundwater mixtures may contain toxic chemicals in excess of federal or state standards for drinking water quality adversely affect human health (Barlaz *et al,* 1990). Incomplete scientific knowledge of landfill leachate, however, makes it difficult to determine its overall toxicity. Although standards have been developed for some of the chemicals present in leachate that have been proven to have toxic effects, other leachate chemicals have neither been studied or regulated. In addition, the mix of different chemicals contained in leachate may result in chemical combinations that are more harmful than the individual leachate chemical.

Table 2.3 Typical Composition of landfill leachate from 1, 5, 15 years old landfills.

Parameter	Age of landfill		
	1 year	5 year	15 year
BOD	7,500-28,000	4,000	80
COD	10,000-40,000	8,000	400
PH	5.2-6.4	6.3	-
TDS	10,000-14,000	6,794	1,200
TSS	100-700	-	-
Specific Conductance	600-9,000	-	-
Alkalinity ($CaCO_3$)	800-4,000	5,810	2,250
Hardness ($CaCO_3$)	3,500-5,000	2,200	540
Total P	25-35	12	8
Ortho P	23-33	-	-
NH_4-N	56-482	-	-
Nitrate	0.2-0.8	0.5	1.6
Calcium	900-1,700	308	109
Chloride	600-800	1,330	70
Sodium	450-500	810	34
Potassium	295-310	610	39
Sulfate	400-650	2	2
Manganese	75-25	0.06	0.06
Magnesium	160-250	450	90
Iron	210-325	6.3	0.6
Zinc	10-30	0.4	0.1
Copper	-	<0.5	<0.5
Cadmium	-	<0.05	<0.05
Lead	-	0.5	1.0

Note: All values are in mg/L except specific conductance measured as microhms per centimeter and pH as pH units.

(Source: Chian and DeWalle, 1976, and 1977a).

Table 2.4 Landfill leachate characteristics over 4-year period.

	Year 1	Year 2	Year 3	Year 4
BOD_5	4,460	13,000	11,359	10,907
COD	11,210	20,032	21,836	18,533
TSS	1,994	549	1,730	1,044
Dissolved Solids	11,190	14,154	13,181	13,029
PH	7.1	6.6	7.3	6.9
Alkalinity ($CaCO_3$)	5,685	5,620	4,830	5,404
Hardness ($CaCO_3$)	5,116	4,986	3,135	4,652
Calcium	651	894	725	818
Magnesium	652	454	250	453
Phosphate	2.8	2.6	3.0	2.7
Ammonia-N	1,966	724	883	1,001
Kjeldahl-N	1,660	760	611	984
Sulfate	114	683	428	462
Chloride	4,816	4,395	3,101	4,240
Sodium	1,177	1,386	1,457	1,354
Potassium	969	950	968	961
Cadmium	0.04	0.09	0.10	0.09
Chromium	0.16	0.43	0.22	0.28
Copper	0.44	0.39	0.32	0.39
Iron	245	378	176	312
Nickel	0.53	1.98	1.27	1.55
Lead	0.52	0.81	0.45	0.67
Zinc	8.70	31	11	21
Mercury	0.007	0.005	0.011	0.007

Note: All values are in mg/L except pH as pH units.

(Source: Keenan, 1983 and Pfeffer, 1986)

Table 2.5 Typical data on the composition of leachate from new and mature landfills.

	Value, mg/L*		
	New landfill (less than 2 years)		Mature landfill (greater than 10 years)
Constituents	Range**	Typical***	Range**
BOD_5 (5-day biochemical oxygen demand)	2,000 - 3,0000	10,000	100-200
TOC (total organic carbon)	1,500-20,000	6,000	80-160
COD (chemical oxygen demand)	3,000-60,000	18,000	100-500
Total suspended solids	200-2,000	500	100-400
Organic nitrogen	10-800	200	80-120
Amonia nitrogen	10-800	200	20-40
Nitrate	5-40	25	5-10
Total phosphorus	5-100	30	5-10
Ortho phosphrus	4-80	20	4-8
Alkalinity as $CaCO_3$	1,000-10,000	3,000	200-1,000
pH	4.5-7.5	6	6.6-7.5
Total hardness as $CaCO_3$	300-10,000	3,500	200-500
Calcium	200-3,000	1,000	100-400
Magnesium	50-1,500	250	50-200
Potassium	200-1,000	300	50-400
Sodium	200-2,500	500	100-200
Chloride	200-3,000	500	100-400
Sulphate	50-1,000	300	20-50
Total Iron	50-1200	60	20-200

*except pH, which has no units.

**Representative range of values. Higher maximum values have beer reported in the literature for some of the constituents.

***Typical values for new landfills will vary with the metabolic state of the landfill.

(Source: Tchobanoglous *et al.*, 1993)

Table 2.6 Metal Ions in Landfill Leachate.

Country	Site/Source	Na	Mg	K	Ca	Cr	Cd	Fe	Cu	Pb	Zn
U.K. 1	Compton Bassett	1320	159	858	94	0.06	0.01	9.8	0.17	0.13	0.62
U.K. 2	Bryn Posteq	904	85	195	477	0.1	0.1	242	0.13	0.1	4.9
U.K. 2	Harewood Whin	1150	820	900	2900	0.17	0.01	560	0.05	0.45	0.32
U.K. 2	Chapel Farm 1	1360	420	900	1040	0.05	0.02	340	0.04	0.2	0.09
U.K. 2	Chapel Farm 2	1210	310	840	1030	0.06	0.04	126	0.03	0.13	0.22
U.K: 3	Seamer Carr	2400	420	2050	4100	1	0.003	2050	0.05	0.61	130
U.K. 4	Brofiskin					1	0.2	28	0.04	0.1	0.2
U.K. 4	Maendy					0.2	0.2	300	0.2	0.1	10
U.K. 4	Eastfield Quarry					30		3000	5	3	50
U.K. 4	Rainham					0.3	0.05	1000	0.2	1	10
Switzer -land 5	Uttigen - Berne						2.78	9.0	0.25	0.12	0.71
U.S. 6	EPA	0 - 7700	16.5 - 15600	2.8 - 3770	5 - 4080			0.2 - 5500	0 - 9.9	0 - 5	0 - 1000
U.S. 6	Steiner	100 - 3800						200 - 1700			1 - 135
U.S. 6	Genatelli	450 - 500	160 - 250	295 - 310	900 - 1700		0.4	210 - 235	0.5	1.6	10 - 30
U.S. 6	Ham	106 - 357	22 - 62	107 - 242	111 - 245			20 - 60	0.04 - 0.1		0.04 - 0.5
U.S. 7	Illinois	1360	650	1140	3750	18	0.1	2200	0.48	1.25	104
U.S. 7	Wisconsin	1580	1140	2300	3900	0.5	0.375	1046	0.75	1.65	13
U.S. 7	Sonoma	240	515	74	1030	0.11	0.02	301	1.0	1.1	22
U.S. 7	Cincinnati	750	530	860	2325	0.2	0.36	735	0.71	3.12	52
U.S. 7	Atlanta	75	52.5	620	520	0.03	17	810	0.48	1.16	74
U.S. 7	College Park	37.5	51.5	35	215	0.015	0.01	560	0.15	0.21	6.5
U.S. 7	Winnetha	370	76	450	254	0.04	0.003	1.5	0.10	0.48	0.16
U.S. 8	Snohomish Co.							569	0.07		26.9
U.S. 8	King County							73	0.03		1.12
U.S.	Range of Values	37 - 7700	16.5 - 15600	2.8 - 3770	5 - 4080	0.015 - 18	0.01 - 17	0.2 - 5500	0.04 - 9.9	0.21 - 5	0.04 - 1000
U.K.	Range of Values	904 - 2400	85 - 820	195 - 2050	94 - 4100	0.05 - 30	0.01 - 0.2	9.8 - 3000	0.03 - 5	0.1 - 3	0.2 - 130
U.K.	Completion criteria for leachate.	1500	500	120	1000	0.5	0.05	2	1	0.5	1

1. (Robinson and Gronow. 1992)
2. (Robinson, Barr and Last. 1992)
3. (Barber and Maris. 1984)
4. (DOE. 1978)
5. (Baccini et al. 1987)
6. (Yong et al. 1992)
7. (Chan and DeWalle. 1975)
8. (Mott et al. 1987)

2.5.4 Leachate Migration:

Landfills built in wetlands, near rivers, and in other places where groundwater is close to the land surface pose specific groundwater contamination problems. In such areas, migrating leachate can quickly reach groundwater and can easily be transported to other locations. In north-central Florida, groundwater supplies were contaminated with high concentration of volatile organic compounds in leachate from two out of three landfills located between one and four meters above an aquifer. (Hallbourg *et al.*, 1992). When leachate reaches an aquifier, it mixes with groundwater held in the soil spaces and this mixture moves along the groundwater's flow path as a plume of contaminated groundwater. The landfill acts as a continuous pollution source as leachate migrates towards and infiltrates the aquifer, causing the edge of the plume to move further and further away from the landfill. A census on landfills in the United States conducted by the EPA during 1985 and 1986 analyzed 9,284 municipal solid waste landfills. Of these landfills, 586 had recorded at least one violation for contamination of groundwater and 660 had received at least one surface water violation (USEPA, 1988b).

As of February 1996, 330 municipal solid waste landfills are presently on the EPA's Superfund National Priorities List (SNPL), constituting 27% of the total NPL. As of February 1993, 228 landfills, which accepted municipal waste, were on the SNPL (USEPA, 1993). These sites present real or potential environmental and/or human health risks significant enough to require federal cleanup action.

Groundwater and leachate mixture moves slowly and steadily underground until it eventually empties into a surface water body. In many areas, groundwater provides much of the base flow of streams and lakes; if the groundwater contains contaminants, the surface water also becomes contaminated. In 1988, an EPA report to Congress identified 163 municipal solid waste landfills that violated various environmental standards. Groundwater was contaminated at 90% of the sites, and surface water quality was impacted at 45% (USEPA, 1988a). Drinking water contamination occurred from 33 of the municipal landfills (20%), and three additional sites bordered on drinking water contamination. The New York State Department of Environmental Conservation estimated that half of the state's 429 municipal landfills contaminated groundwater supplies (Concern Inc., 1988).

The natural processes of attenuation and decomposition can diminish the strength of leachate. As leachate moves with groundwater, some of the chemicals bind with soil particles and others are naturally degraded by microorganisms living in soil. Once leachate reaches the point of surface discharge, its concentration may also be diluted with lake or stream flow. These processes can make it difficult to detect or prove that migrating landfill leachate is the source of contamination in water supplies.

In fast flowing surface waters or large lakes, leachate may become so diluted that scientific instruments cannot detect its presence in the water column. Leachate contamination may also escape detection when the body of water into which leachate empties is already polluted. In this case, the additional Volatile Organic Compounds (VOCs), Soluble Organic Compounds (SOCs), metals, and other pollutants contributed by landfill leachate may be indistinguishable from the pollutants discharged into the water from other activities. In addition, metals deposited in a stream or lake tend to accumulate in the bottom sediments, rendering them undetectable by tests of the water column. They may be consumed by bottom-dwelling aquatic organisms and subsequently move up through the food chain as organisms consume one another. Humans, at the top of the food chain, can be adversely affected if they consume contaminated fish or other aquatic organisms.

According to a 1988 EPA analysis of SNPL municipal landfills, synthetic and volatile organic chemicals and metals contaminate most of these sites, and almost 75% of the landfill sites contaminate groundwater with these hazardous substances (USEPA, 1988a). Industrial waste, sludge and household hazardous waste deposited with municipal waste in the landfills were likely sources for the contaminants (USEPA, 1988b).

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Data from the 1980s indicate that thousands of municipal landfills are located less than one mile from surface water bodies, with the majority located within a one-quarter mile radius (USEPA, 1988b). Leachate traveling with groundwater can reach these surface waters and therefore contribute a wide range of contaminants to the surface water system. Direct discharge of contaminated leachate into surface waters may result in the loss of recreation, agriculture, and drinking water and create environmental degradation (US EPA, 1988c).

Leaking landfills raise particular concerns for people whose source of drinking water is wells located near landfills. These wells may draw up groundwater that is highly contaminated with leachate. The influence that an active well exerts over groundwater flow depends on factors such as the well's pumping rate, the geology of the region, and the presence of other wells in the area also affecting groundwater flow. By drawing in water from all directions, a pumping well alters the natural flow of groundwater, making it difficult to predict the direction of groundwater flow. The existence of multiple wells enhances the difficulty of predicting flow patterns; each well influences flow direction, and once any of the wells cease to operate, groundwater flow changes accordingly. Working against natural gradients, a well can induce groundwater that would normally flow towards a river to change direction and move towards the well.

In addition to posing health threats to communities whose drinking water is supplied by wells, leachate-contaminated groundwater can also adversely affect industrial and agricultural activities that depend on well water. For certain industries, contaminated water may affect product quality, decrease equipment lifetime, or require pretreatment of the water supply, all of which cause added financial expenditures. The use of contaminated water for irrigation can decrease soil productivity, contaminate crops, and move possibly toxic pollutants up the food chain as animals and humans consume crops grown in an area irrigated with contaminated water. Though contamination from landfills principally affects human health, leachate that migrates to surface water bodies also impacts the integrity of the ecosystem. Catastrophic events such as fish kills are unlikely, but the continuous discharge of contaminated leachate to a surface water body can cause chronic long-term effects as the pollutants accumulate in sediments and aquatic organisms (USEPA, 1988a). Since the effects are subtle, damage may be difficult to assess and discover (USEPA, 1988c).

2.6 The Soil Environment

On the surface of the earth there is a thin skin of material, called soil. Soil consists of mineral material (sand, slit and clay), organic material from dead organisms, living organisms, water and air. The kind of soil present in an area is primarily dependent on the climate and on the kind of parent mineral material present. Regardless of the kind of soil present, there is a layer of top soil that contains most of the organic material and a subsoil that contains materials deposited by the movement of water from the surface to deeper layers of the soil.

Soil pollution is often thought of as resulting from chemical contamination such as through the use of excessive amount of pesticides and fertilizers that can result in surface water or ground water contamination (Bollag and Liu, 1990). However, there are other forms of soil pollution or degradation, including erosion, compaction, and salinity. Soils have often been neglected when they are used for on site disposal of waste chemicals and unwanted materials. Soil is capable, to some degree of adsorbing and neutralizing many pollutants to harmless levels through chemical and biochemical processes, if sufficient time is allowed.

2.6.1 Soil Physical Properties

Soil physical properties are soil texture, soil structure, and bulk density and pore space. Soils contain solids, liquids and gases. The open volume, or pore space, that a soil possesses depends on the arrangement of soil solids. It is necessary to understand the composition and arrangement of solids to apprehend the effect of solids on movement of liquids and gases.

Other physical properties of soils are soil color and temperature. The general texture of a soil depends on the proportion of particles of various sizes. Soils are composed of solid materials ranging in size from stones to fine clays. The larger materials, known as coarse or mineral fragments include stone. These fragments chemically and physically weathered over long periods of time forms the smaller particles of sand, silt and clay. Soil particles are defined on the basis of their diameter, although rarely, these particles do exist as spherical objects. For instance, clay minerals are three dimensional, layered structures that commonly have a plate-like appearance. Soil particle sizes often differ with different classification schemes used by various groups. The U.S. Department of Agriculture system soil classification is given in Table 2.7.

According to the proportion of sand, silt and clay, soils are broadly classified into several discretionary textural groups (Fig.2.6). There are 12 soil texture classes defined by the relative proportion of sand, silt and clay that comprise a soil sample. They are sandy clay, silt clay, clay loam, sandy clay loam, silty clay loam, sandy loam, slit loam and loamy sand (Soil Survey Staff, 1975 and 1992).

Table 2.7 The classes used to describe the distribution of soil particles based on size.

Soil Class	USDA system* (mm diameter)	International system (mm diameter)
Gravel	Above 2.0	Above 2.0
Very coarse sand	_	1.0-2.0
Coarse	0.2-2.0	0.5-1.0
Medium sand	_	0.1-0.5
Fine sand	0.02-0.2	0.05-0.1
Silt	0.002-0.02	0.002-0.05
Clay	Less than 0.002	Less than 0.002

* This system of the United States Department of Agricultural is also used in several other countries (Source: Soil Survey Staff, 1975 and 1992).

There are generally two methods used to determine soil texture: (1) the field method done by hand, and (2) the mechanical analysis method. The mechanical analysis method to determine the sand, silt, and clay content requires the removal of coarse fragments by sieving the soil through a 2-mm sieve. Chemical treatments are also necessary to remove cementing agents such as organic matter and carbonates. The percentage of sand, silt, and clay should always total 100%; and once found, the soil texture may be determined using the textural triangle, Figure 2.6 (Wild, 1993).

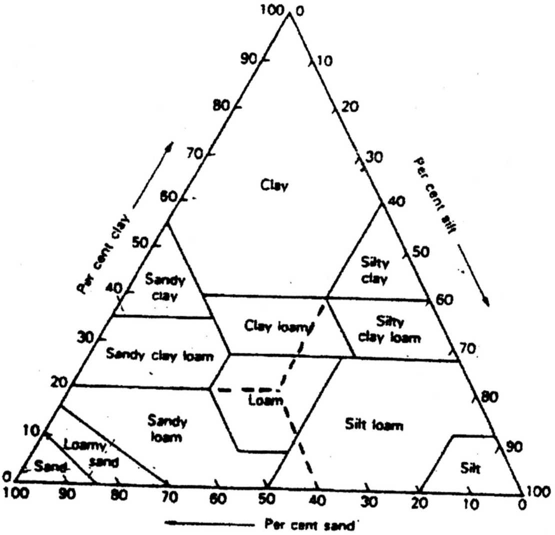

Fig. 2.6 Textural triangle indicating the range in percent sand, silt and clay for each soil textural class
(Source: Wild, 1993)

Only two of the soil particle percentages are actually necessary to determine the soil textural class since the point at which the two meet does not change when the third particle percentage is used. The soil textural class of the particular soil remains virtually unchanged over time equivalent to the human life span. Soil texture is often considered a basic property of the soil. Soil descriptions used in soil surveys record the soil textural class of each horizon. A disturbance of an area such as water or wind erosion, however, could alter the textural class of the soil surface of both the soil being eroded and the soil receiving the erosional deposition. Also over long time periods, weathering and the translocation of soil materials may eventually alter soil texture.

Soil structure is classified based on type, size and grade. Common soil structure types include granular, platy, subangular and angular blocky, prismatic and columnar shapes. Soil particles that are held together by chemical and physical forces form stable aggregates. Natural aggregates are called peds. Collectively, the type of soil aggregates or peds defines soil structure. Soil structure influences the amount of water that enters a soil called infiltration and amount of gas diffusion at the soil surface. Soil structure also plays an important role in the movement of liquid and gaseous substances through soils. Porosity is a function of soil structure (Hillel, 1980).

Structural size classes vary with the type of structure considered, and range from very fine to very coarse. By observing the soil structure in place in a soil pit and by determining how well developed the structure, structural grade is determined. Structural grades include weak, moderate, and strong classifications. Particle and bulk density measurements are useful for estimating the type of soil minerals present and the degree of soil compaction, respectively. Particle density is the mass of a particle per volume (kg/m^3 or g/cm^3). Pore space and the weight of water are not included in particle density measurements. Common soil minerals (quartz, feldspars, micas, and clay minerals) have particle densities between 2.60 and 2.75 g/cm^3. A value often used to represent the average soil particle density is 2.65 g/cm^3. Bulk density is a measure of mass per volume of a soil. Undisturbed soils are usually used for bulk density measurements so that a true representation of the amount of solid present in a particular soil volume can be calculated. With disturbed soil, natural soil pore spaces are destroyed. Bulk density is calculated on an oven-dried weight basis and does not take into consideration the amount of water present in the soil at the time of sampling. Mineral soils, unless developed in volcanic ash, generally have bulk densities greater than 1.0 g/cm^{3}with a common bulk density of a loam soil being 1.3 g/cm^3. Soils with high densities (e.g. 2.0 g/cm^3) will likely have slow water infiltrationand permeability, which can result in ponding or surface runoff. Organic soils typically have bulk densities of less than 1.0 g/cm^3. Mineral soils derived from volcanic ash often have bulk densities in the range of 0.3-0.85 g/cm^3. Soil minerals are classified as primary and secondary minerals due to their origin. Primary minerals are those that formed during the cooling of molten rock and are predominantly silicate materials. Igneous rocks are composed entirely of primary minerals; metamorphic and sedimentary rocks can contain abundant amounts of either primary or secondary minerals.

Secondary minerals are formed in the soil from soluble products derived from the weathering of rocks. Clay minerals are one of the most important soil secondary minerals due to their large surface area and reactivity with ionic and dissolved organic compounds, and will be discussed in more detail in the next section. Carbonates and sulfates accumulate in B-horizons and are the dominant secondary minerals present in soils in arid and semiarid regions.

Carbon is present in all soils in the form of organic matter. Soil organic matter contents vary from less than 1% in course-textured soils and soils of arid regions to nearly 100% in some poorly drained organic soils; typical farmland topsoil may contain 2-10% organic matter. Organic matter influences soil physically, chemically and biologically. Soil structure is often improved with the addition of organic materials such as manure, sludge, composts, and crop residues that are returned to the soil.

2.6.2 Soil Chemical Properties

Soil has been likened to an extremely elaborate chemical laboratory in which a large number of reactions, involving almost every known element, take place. Certain reactions are comparatively simple and well understood, but the vast majority is extremely complex and not completely understood by soil scientists.

Organic matter plays an important role in the chemistry of soils. Soil properties associated with soil organic matter include soil structure, macro- and micronutrient supply, cation exchange capacity (CEC), and pH buffering. Additionally, organic matter is also a source of carbon and energy for microorganisms. Soil organic matter is comprised of decomposed plant and animal residues. It is a highly complex mixture of carbon compounds that also contain N, S, and P. Organic matter is made up of humid substances and biochemical compounds. Soil organic matter can absorb trace element pollutants e.g. Pb, Cd, Cu, which will reduce the chance of contamination of surface- and groundwater. The advantage is the absorption of pesticides and other organic chemicals. This reduces the possibility of pesticide carryover effects, prevents contamination of the environment, and enhances both biological and nonbiological degradation of certain pesticides and organic chemicals. In addition, organic matter is known for its capacity to absorb inorganic and organic gases (Sposito, 1989).

Although there are many benefits of soil organic matter, there are also detrimental effects that occur under certain situations. Soil with high organic matter content may require higher rates of pesticide application for effective pest control. Water contamination is then of concern if these pesticides leached or are transported by wind or water. Soil also have a finite capacity to absorb trace elements and should not be used for long term applications of organic wastes which contain high level of trace elements (Bohn, 1979).

Ion exchange is one of the most significant functions of soils. When isomorphic substitutions occur, cations of lower charge substitute for cations of high charge and a permanent charge develops. Charge sites can also develop as a function of pH. Surface charges can form on clay minerals, metal oxides and organic matter. These charged sites are the result of ionization or protonation of uncharged sites. Ionization results in a negatively charged site and protonation results in a positively charged site. Both these reactions are dependent on pH. Under acidic soil conditions, some clay minerals, and metal oxides will have charged, anionic exchange sites. Inorganic and organic ions having charges that are opposite of the exchange site gets attracted to the soil surface.

2.7 Soil Attenuation

Soil attenuation is the ability of a soil to reduce the potency or concentration of polluted water passing through it. Above the water table, in the "unsaturated zone," both air and water fill the pores between soil particles. As leachate moves slowly through this zone toward the saturated aquifer, small soil particles can attenuate certain leachate chemicals. Some chemicals, such as positively charged lead, zinc, cadmium and mercury metals, are easily attenuated. As leachate containing these metals flows through soil, the metals stick or adsorb to the soil and are removed from the leachate. Other leachate pollutants, such as VOCs and acids, are not easily attenuated, and they move unimpeded through soil.

The chemical makeup of a soil and the characteristics of its binding sites affect its attenuation capability. The binding sites of a soil are the areas that adsorb chemicals; once the binding sites of the soil particles become full, they can hold no more chemicals. As they fill to capacity, the binding sites become selective. Only preferentially bound chemicals, or those that form tight bonds with soil, will be attenuated, and chemicals that bind loosely to soil will be replaced. For example, as leachate flows through soil, lead particles in the leachate can easily replace manganese that is adsorbed on soil because lead is more preferentially bound to soil than manganese. In this situation, lead would be removed from the leachate and stick to the soil, but manganese would leave the soil and re-enter the leachate, posing a continued threat to groundwater.

Different soils have different abilities to attenuate and exchange chemicals, called the "ion exchange capacity" of a given soil. In addition, once the soil binding sites are full, even chemicals that can normally bind to the soil will not be attenuated but will move through the soil towards the groundwater.

The mechanisms involved in the soil attenuation are complex and falls into three categories that are summarized below-

1. Physical mechanism.
2. Chemical mechanism.
3. Microbiological mechanism.

2.7.1 Physical Mechanism

Physical mechanisms are the physical forces in processes that bring about attenuation. The important physical mechanisms are (a) Filtration, (b) Diffusion and Dispersion, (c) Dilution, and (d) Physical Sorption.

(a) Filtration

Filtration is a mechanism by which leachate constituents are physically trapped. The random pore structure of the soil system serves to physically entrap suspended and settleable solids in a leachate, just as gravity filtration through sand and other media will remove solids in water treatment. Filtration efficiency depends on the pore size and hydraulic gradient will improve filtration. It is difficult to estimate the percentage of attenuation through filtration for any parameter, however, this will be an operative mechanism whenever chemical precipitation, biological growth, and other processes produce undissolved solid particles (Farquhar and Rogers, 1976; Fuller & Korte, 1976; Fuller, 1977).

(b) Diffusion and Dispersion

Diffusion and Dispersion are two mechanisms by which leachate is diluted by the aquifer due to molecular diffusion caused by the concentration gradient of contaminants. Constituent moves from a region of high concentration to a region of low concentration (Mang *et al.*, 1978). Hydrodynamic dispersion is the result of variation in pore velocities within the soil. Dispersion occurs in both longitudinal and transverse directions (Perkins and Johnston, 1963).

(c) Dilution

Dilution of the leachate by ground water reduces the concentration of its constituents. The ratio of dilution is proportional to the solution flux of both leachate and groundwater. Also, the finer the soil texture, slower the rate of dilution. Chloride, nitrate, hardness and sulfate found in municipal landfill leachate are attenuated only by dilution (Bagchi, 1994).

(d) Physical Sorption

Physical sorption is a function of Van der Waale forces, and hydrodynamic and electrokinetic properties of soil particles. Only a small portion of the reaction of trace contaminants in soil/water solutions can be defined as physical adsorption. Bacteria and virus removal however, is an important physical adsorption mechanism (Gilbert, 1976).

2.7.2 Chemical Mechanism

Chemical mechanisms involve (a) Chemical precipitation, (b) Adsorption, (c) Complexation, and (d) Ion exchange. In the reducing zone, insoluble heavy metal sulfides and soluble iron sulfide are formed.

(a)Chemical precipitation

Chemical precipitation involves a phase change in which dissolved chemicals are crystallized and deposited from a solution because their total concentration exceeds their solubility limits. The solubility limits depend on factors such as ionic species and their concentration, temperature, pH, redox potential (EH), and concentration of dissolved substances (Gebhard, 1978). In the area between reducing to oxidizing zones ferric and inorganic hydroxides precipitate out. Other compounds may be co-precipitated, particularly with ferric hydroxide due to fixation of trace substances on the surface of the ferric hydroxides.

(b) Adsorption

Adsorption is the process by which molecules adhere to the surface of individual clay particles. Because of the difficulty in distinguishing adsorption from exchange reactions experimentally it is sometimes referred to as adsorption-exchange reactions (Philips and Nathwani, 1976;Minnesota Pollution Control Agency, 1978;Roberts and Sangrey. 1977). However, there is a basic difference between adsorption and exchange reactions. Adsorption will cause a decrease in the total dissolved solid (TDS) in the leachate, whereas exchange reactions will not. Therefore adsorption may be considered to really attenuate leachate, whereas exchange reactions will simply change the type of ions present in the exfiltrate (the liquid generated after leachate percolation through the unsaturated zone).

Much of the available information pertains to adsorption for specific ions, although some studies have included analyses of adsorption isotherms for the leachate-soil systems (Griffin and Shimp, 1976;Griffin, 1977).

Although some generalization of leachate quality for municipal waste can be made for a region, in most cases, leachate quality is significantly site specific. Thus, in the absence of experimental results using the on-site soil and leachate obtained from the waste to be disposed of at a particular site, site-specific quantitative analysis cannot be performed. The opposite of adsorption-desorption-occurs in many systems (Gebhard, 1978).

(c) Complexation

Complexation or chelation is the formation of inorganic-organic complexes. The extent of leachate attenuation by complexation in the soil environment is not known, and is difficult to predict.

(d) Ion exchange

Clay minerals have the property to exchange ions of one type with ions of another type without disturbing the mineral structure (Lu *et al.*, 1985). The total capacity of soils to exchange cations is called the *cation exchange capacity* (CEC). The CEC of any particular soil is effected by the kind and quantity of clay mineral, organic content, and by the pH of the soil.

2.7.3 Biochemical Mechanism

Various biochemical processes can occur as the leachate passes through soil or rock. The nature of the biochemical reactions can differ widely, depending on the oxidizing or reducing nature of the subsurface water.

Although the organic components of domestic waste leachate may be attenuated, some organics such as phenols, oils, solvents, and pesticides can be very resistant to biodegradation and may only be attenuated by physical processes such as adsorption.

Relatively little is known about the biochemical rates, pathways, intermediate steps and end products of the biodegradation of organic contaminants in subsurface waters. In the zone of the greatest organic pollution, the dissolved oxygen may be used up totally by the biochemical reactions. The microorganisms take-up further oxygen by reducing nitrate and sulfate. In such a reducing zone, high values of ferrous iron and ammonia may be found. In a reducing zone liberation of carbon dioxide and methane as an end product of organic biodegradation may occur, but reaction rates are likely to be slow. A reducing zone is likely to occur in the area where high BOD leachate is issuing from the landfill into the subsurface water. If the water table is at the base of the landfill or the landfill dips below the water table, the reducing zone occurs in the aquifer (Paul and Clark, 1989).

2.8 Groundwater and Surface Water Pollution

Though new landfills may be built with liners that act as temporary barriers to leachate migration, the vast majority of landfills built before

1993 do not have such liners. In such unlined landfills, or in landfills with leaking liners, gravity causes leachate to move through the landfill, out to the bottom and sides, and through the underlying soil until it reaches the groundwater zone or aquifer. There, the leachate mixes with and travels along the groundwater in the underground path.

An aquifer is a zone of underground soil or rock that is saturated with groundwater. Within the aquifer, groundwater fills the spaces or pores between the soil and rock. Particles flow slowly down a gradient that is determined by underground geology, and eventually resurfaces into a lake, stream, wetland or coastal water. The geographic location and depth of an aquifer varies according to climatic and site-specific characteristics. Soil type and geologic structure are indicators of aquifer presence in an area. Aquifers are often found in subsurface layers of sand, where large pores allow easy groundwater movement. Clay soil layers, which have much smaller pores, do not allow groundwater to flow as easily and are less likely to act as aquifers. The upper boundary of the saturated zone is known as the water table, and this water table rises and falls periodically depending on precipitation levels and seasonal effects. In rainy periods, increased volumes of water move through the soil, saturate a larger area, and raise the water table closer to the surface. In drier periods, less water infiltrates the pore spaces of an aquifer and the water table may drop.

Landfill leachate is also a source of liquid that can significantly affect water table levels by increasing the saturated area. A completed landfill site in Northumbria, Yorkshire Region had been used for the disposal of cyanide containing substances. The resultant leachate caused contamination of ground water to the level above those specified for drinking water (NRA, 1991).

Specific leachate contaminants are found at various depths in an aquifer depending on their molecular structure. For example, trichloroethylene and other industrial solvents, which are heavier than water may sink to the bottom of the aquifer and collect in a slowly dissolving pool. Lighter organic contaminants that do not dissolve in water, such as gasoline and other fuels, may float on top of and travel along with groundwater.

Over the past 40 years efforts have been made to determine the impact of sanitary landfill and other waste impoundment upon ground water quality. LeGrand (1969) proposed a *point count* system based on monographs. These points are used to determine the pollution impact of sanitary landfill upon the ground water. Hagerty (1973) presented the *site ranking* system proposed by Booz Allen Applied Research, Inc. (1973) for hazardous waste sites.

Complex models and applications were also provided by Lu *et al.* (1984). Walton (1989a, 1989b) provided few analytical microcomputer programs for quick and easy simulation and graphing of two dimensional groundwater flows. USEPA (1989) provided information on modeling subsurface contaminant transport and fate, and management considerations. Bear *et al.* (1992) compiled a review paper on the fundamentals of groundwater modeling and discussed analytical models, uncertainty, and model misuses. The International Ground Water Modeling Center (IGWMC) operates a clearinghouse service for information and software pertinent to groundwater modeling. (Van der Heijde and Elnawawy, 1993).

2.9 Heavy metals in landfill soil and landfill leachate

When deposited into the ground, metal particles frequently bind to the soil and do not easily dissolve or migrate with water. In the acidic conditions of a landfill, however, metals such as cadmium, copper, iron, manganese, and lead can dissolve and migrate with leachate. Other chemical reactions in a landfill can also change the state of metals, allowing them to attach to other particles and travel with leachate. Most metals are not carcinogenic when consumed, but they produce other serious toxic effects. Pollution gives rise to anomalously high concentrations of the metals relative to the normal background levels; therefore, presence of the metal is insufficient evidence of pollution, the relative concentration is important. (Alloway and Ayres, 1994).

The 1988 USEPA study consolidated leachate data from 70 municipal landfills (USEPA, 1988b). Several metals that cause a variety of health effects exceeded drinking water standards, including antimony, beryllium, cadmium, lead, mercury, nickel and thallium. Table 2.8 presents the health effects of some of the metals found in the landfill leachate.

Table 2.8 – Toxic effects of some metal ions on human health.

Metal	Toxic effect
Cadmium	Carcinogenic
Chromium	Carcinogenic, Tubular ,Necrosis of kidney
Copper	Carcinogenic, Disorder of copper metabolism – hepatic cirrhosis
Iron	Gastrointestinal irritation
Magnesium	Muscle weakness
Potassium	Diarrhoea, nephrotoxicity
Zinc	Irritability, nausea

(Source: USEPA, 1988b)

Numerous other landfill studies have found metals in migrating leachate. For example, a 1988 study on four municipal landfills in southern Ontario, Canada that examined leachate for inorganics, and discovered that the concentrations of cadmium exceeded Ontario's drinking water

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Numerous other landfill studies have found metals in migrating leachate. For example, a 1988 study on four municipal landfills in southern Ontario, Canada that examined leachate for inorganics, and discovered that the concentrations of cadmium exceeded Ontario's drinking water standard of 0.005 mg/l.Cadmium is acutely toxic to animals, indicating the potential health risks posed by leachate contamination in groundwater supplies. In another example leachate from four municipal solid waste landfills in southern Ontario, Canada were analyzed for chemical composition and speciation. Samples contained elevated levels of alkali and alkaline earth metals and chloride ions. All leachates contained cadmium concentrations exceeding the acceptable levels for potable water (Boltan and Evans, 1991).

The behavior of the metal ion content within leachate is not fully understood, as the landfill leachate acts as a dynamic environment in which the number of processes affecting the concentration and mobility of metals in the leachate operate. Attenuation of metal ion content is often related to a predominate process (Bagchi, 1987). These processes are summarized in Table 2.9. The dominant process at a particular site within the landfill controls the relevant proportion of the metal ions in the leachate. Whichever process operates depends on a number of factors including pH, temperature and the presence of other metal ions.

Table 2.9 Attenuation and mobility of metal ions in a landfill environment.

Metal	**Major attenuation Mechanism**	**Mobility in clayey environment**
Cadmium	Precipitation, Absorption	Moderate
Calcium	Precipitation, Ion Exchange	High
Chromium	Precipitation, Ion Exchange, Absorption	Low or High
Copper	Absorption, Ion Exchange, Precipitation	Low
Iron	Precipitation, Ion Exchange, Absorption	Moderate
Lead	Absorption, Ion Exchange, Precipitation	Low
Magnesium	Ion Exchange, Precipitation	Moderate
Potassium	Absorption, Ion Exchange	Moderate
Sodium	Ion Exchange, Dilution	High
Zinc	Absorption, Ion Exchange, Precipitation	Low

(Source: Bagchi, 1987)