Chapter 7
Advances in Land, Underground, and Ocean Disposal Techniques

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7.1 Introduction

The generation of ever-increasing quantities of hazardous waste necessitates the development and use of effective disposal strategies. Other than incineration, landfills, deep well injection systems, underground geologic repositories, and oceans represent several possible means of hazardous waste disposal options. In general, landfilling is a common disposal technique that relies on long-term containment of hazardous waste in a landfill. Deep well injection, on the other hand, involves the injection of liquid hazardous waste into subsurface porous, permeable, and saline water-bearing geologic zones. Emplacement of hazardous waste in underground geologic repositories is also an attractive option wherein the hazardous waste is isolated from the environment by means of a host rock. Finally, ocean disposal of hazardous waste involves the use of ocean incineration or ocean dumping techniques. This chapter provides an overview of the state of the art of the different hazardous waste disposal methods. The technical details are presented, and technological advancements in different aspects of the disposal methods are also discussed in this chapter.

7.2 Landfill Disposal

A landfill for hazardous waste, also known as an engineered or a secured landfill, is defined as a disposal facility where the waste is safely and securely placed in isolation from the environment and the public. Interest in the use of landfilling as a

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disposal method is mainly attributed to its low cost and unsophisticated design compared to other disposal methods (Batstone et al. 1989a; Blackman Jr. 2001; Williams 2005). Landfills must be properly designed, constructed, and operated in order to ensure safe disposal of the hazardous waste. The subsections below discuss all the significant aspects of hazardous waste landfills and their related advancements.

7.2.1 Landfill Restrictions

Most hazardous wastes can be disposed in properly designed landfills. However, there may be land disposal restrictions (LDR) for certain types and forms of hazardous wastes in order to avoid adverse environmental effects of landfills. The Title 40, Part 268 of the US Code of Federal Regulations outlines the prohibitions on land disposal, including landfilling and other methods such as well injection, unless the treatment standards have been met. Also, Title 40, Part 265 of the US Code of Federal Regulations draws special requirements pertaining to landfilling. As a summary, non-containerized hazardous waste containing free liquids and containerized liquid hazardous waste (except very small containers and small lab pack containers surrounded by absorbent material and placed in larger secondary containers) cannot be placed in landfills. When placed in containers, liquid hazardous waste can only be landfilled when it is solidified or mixed with appropriate absorbent (Wright et al. 1989). In addition, hazardous wastes on the F, K, P, and U lists, such as spent solvents, dioxin-containing wastes, corrosive wastes with pH <2, and wastes containing heavy metals or PCBs, need to meet the treatment standards before deemed suitable for disposal in landfills (Pichtel 2014). Such restrictions necessitate the use of appropriate physical, thermal, chemical, or biological treatment technologies in order to meet the treatment technology-based standards or the maximum concentration limits for specific wastes. The treatment may be based on a technology specified by the US EPA or any other technology (except dilution) as long as the maximum concentration limits are met (Pichtel 2014). As a general rule, the hazardous waste acceptance criteria for direct disposal into a landfill is as follows: calorific value: <3200 kcal/kg, nonbiodegradables: $<20\%$, flash point: >600 °C, pH: 4–11, reactive cyanide: <250 ppm, reactive sulfide: <500 ppm, water-soluble organics: $\leq 10\%$, and water-soluble inorganics: $\leq 20\%$ (Rao et al. 2017).

7.2.2 Landfill Site Selection

Selection of suitable hazardous waste landfill site must consider a range of engineering, environmental, regulatory, and economic factors. Detailed site and environmental assessments and cost studies need to be conducted before selecting a landfill site. The landfill site assessment involves extensive site survey to identify the possible pathways and environmental receptors of the releases from the landfill.

Also, it assesses the geological and hydrogeological conditions around the site in order to gather information about the soil, bedrocks, and groundwater and identify any potential foreseeable hazards such as landslides. Core samples are collected for detailed geologic and hydrogeologic evaluation. Borings are usually needed to fully characterize the soil and the subsurface geology. Determination of soil pH, cation-exchange capacity (CEC), and microbial activity is also important in order to assess the ability of soil to attenuate the constituents of hazardous waste. For instance, soils with high pH and CEC possess a superior ability to retain heavy metals (Wright et al. 1989). Also, soil microorganisms tend to decompose the organic matter present in the waste (Batstone et al. 1989b). For hydrogeologic evaluation, the depths of the water table and piezometric water levels in bedrocks/confined aquifers are determined. Also, the movement of groundwater is studied (O'Leary and Tchobanoglous 2002). Environmental assessment, on the other hand, involves a detailed analysis of the direct and indirect environmental effects that will result from the landfill and the activities involved during its construction, operation, and maintenance (Williams 2005). The data and results obtained from the site and environmental assessments are evaluated against the site selection criteria in order to decide on the suitability of the site for landfilling. In general, the landfill site selection is based on the following engineering, environmental, and economic criteria (Batstone et al. 1989a; Wentz 1989; O'Leary and Tchobanoglous 2002):

- The site selected for landfilling should have sufficient capacity to hold the quantity of hazardous waste that is generated over a long period of time.
- The site should be away from the populated areas to avoid risks to public health.
- It is preferable that the site is close to the point of hazardous waste generation to minimize transportation costs and risks.
- The site must be easily accessible through common means of transportation.
- The site should not be a high seismic activity impact zone.
- The surface/soil beneath the landfill should be impermeable or of low permeability.
- The soil at the site should have pH in the range 7–8 to allow for reduction of heavy metals and biodegradation of organic contaminants.
- Climate conditions at the site should not be severe.
- Areas such as protected lands, wetlands, floodplains, mudflats, sand dunes and those with high slopes, landslides, faults, soil erosion, subsidence, and underlying mines should not be used.
- The site should not have contact with surface or groundwater.
- Areas in contact with terrestrial and aquatic ecosystems should not be used.
- The site must meet the regulatory requirements.
- Public opinion must favor the site selection.
- The costs of site acquisition, development, operation, and maintenance must be reasonable.

It is important to note that the criteria mentioned above are not exhaustive. Several other important criteria such as social, environmental, and health costs and political and aesthetic factors related to the landfill must also be considered in selecting the most suitable site. As a result, site selection becomes a tedious and protracting decision wherein a large number of criteria, both quantitative and qualitative, must be satisfied simultaneously. This has attracted significant research interest in order to develop effective tools to assist in site selection for landfilling of hazardous waste.

The use of Delphi method to rank the site selection criteria has been studied (Zakaria et al. 2013). It has been shown that the Delphi method, which is based on surveying experts in the field, is both time and cost-effective. Results from the Delphi method have shown that the environmental criteria should be given priority when locating the landfill site followed by the social and economic criteria. Map overlayer technique has been proposed for locating suitable sites for hazardous waste landfilling (Yesilnacar and Cetin 2005). The technique involved the use of topographic, geologic, active fault, land use, earthquake zoning, erosion, climate, and transportation maps on a regional scale. Each map was evaluated to identify the potential landfill sites based on criteria related to geology, climate, temperature, precipitation, wind, topography, land use, erosion, seismicity, and transportation. The final site selection was based on over-layering and joint comparisons that were made possible through transparencies of the maps.

In a later study, the same technique was employed, however, with the use of a single geomorphological map combined with active fault and earthquake zoning maps (Yesilnacar and Cetin 2008). Advances have also been made in utilizing the spatial data from GIS for quick and reliable identification of proper landfill sites for the disposal of hazardous waste. For instance, combination of GIS and analysis hierarchical process (AHP) has been used to determine the most suitable location for landfilling of radioactive waste (Rezaeimahmoudi et al. 2014). The AHP model was based on pair comparison, and seven selection criteria were considered, namely, water resources, slope, population centers, roads, protected zones, faults, and geology. Suitable landfill sites were determined using the base maps that were created using GIS and incorporated with the expert opinion-based criteria weights from the AHP model. Similarly, in another study, GIS was used in conjunction with remote sensing to build a geospatial database (Abd-El Monsef and Smith 2019). Information for the database was retrieved from field surveys, satellite images, and literature. Using the weighted criteria (based on the Basel Convention), AHP was used to select the most suitable site for landfilling of hazardous waste.

As an alternative approach, integration of GIS and landfill susceptibility zonation methods was investigated for locating the candidate sites for hazardous waste landfills (Hafezi Moghaddas and Hajizadeh Namaghi 2011). A three-step methodology was adopted in this case. First, areas such as protected areas, urban and rural areas, fault plains, riversides, alluvial fans, main roads, dam's drainage basins, and groundwater resources were excluded. Second, landfill-suitability zonation maps were prepared. In these maps, suitable sites were first screened based on criteria such as geology, geomorphology, climate, land use, land cover, and topography. Then, a scored map was created by utilizing weighting and scoring for geology, geomorphology, land cover, slope, precipitation, and evaporation. In the last step, standard impact assessment study (Leopold matrix) combined with technical and economic considerations were used to select the final sites.

Other studies have focused on the integration of GIS and multi-criteria analysis methods for selecting suitable sites for hazardous waste landfilling. For example, in one study, GIS was used for initial screening in order to eliminate the undesirable sites (Sharifi et al. 2009). The initial screening was followed by multi-criteria decision analysis (MCDA) which was guided by a panel of experts to select the most suitable sites. Similar integration of GIS and multi-criteria analysis methods was used in other studies (Feo and Gisi 2014; Danesh et al. 2019; Stemn and Kumi-Boateng 2019). Also, integration of GIS with simple additive weighting (SAW) multi-criteria analysis method has been suggested (Khamehchiyan et al. 2011).

7.2.3 Landfill Design

The design of a secured landfill needs to satisfy several engineering and regulatory requirements in order to minimize the impacts on the environment, ecosystems, and public health. A well-designed secured landfill will completely isolate the hazardous waste and provide enough mitigation measures to deal with any releases/leaks from the disposed hazardous waste. The design of a landfill needs to incorporate several key elements (see Fig. 7.1). These key elements of design and engineering of a secured landfill are discussed hereafter.

Fig. 7.1 Landfill site preparation (Landfill design n.d.)

Landfill Capacity

The capacity of the landfill is an important design consideration. Factors such as the expected volume/amount of hazardous waste to be disposed (taking into account the current and future waste generate rates), the waste density, the amount of cover material used, the volume occupied by the liner system, the number of lifts used, and waste settlement must be taken into account while estimating the landfill capacity. Settlement of the waste is expected due to physical rearrangement soon after disposal. Also, physical, chemical, and biological degradation along with overburden pressure result in waste settlement within the landfill (Williams 2005).

Disposal Method

Hazardous waste can be disposed in landfills using the trench, area, or canyon methods (O'Leary and Tchobanoglous 2002). The trench method is the most common and is based on disposal below the ground level, for example, in a natural or an excavated depression where the water table is low (Wright et al. 1989; Pazoki and Ghasemzadeh 2020). In particular, the hazardous waste is placed inside individual trenches/cells over the active landfill area. The use of individual cells helps in segregation of incompatible wastes. The dimensions of the cells can vary depending on the amount, size, and characteristics of the hazardous waste to be disposed. The sides of the cells are sloped with a ratio of 2:1–3:1 (O'Leary and Tchobanoglous 2002). Bulk hazardous wastes are placed in the cells to create a layer of 0.61–0.91 m thickness. The layer is compacted and then covered with a 0.3 m layer of covering material, such as soil, to prevent infiltration of water or escape of potential releases from the hazardous waste. The soil cover is placed at the end of each working day and is typically termed as daily cover. In addition, an intermediate cover is used for the cells which are filled or when the site is expected to be inactive for a prolonged period. The use of daily cover is essential to control water entry into the landfill. Containerized hazardous wastes are placed vertically in the cells at a reasonable distance from one another. The space between the containers is filled with soil or compatible bulk hazardous wastes. Daily and intermediate covers are also applied on top of the containerized hazardous wastes (Wright et al. 1989). Once all the cells within the active landfill area are filled, the complete layer of cells (knows as lift) may be stacked with another layer of cells to create a series of lifts.

The area method, on the other hand, is based on aboveground disposal. It is suitable for cases when excavation of cells is infeasible due to high groundwater conditions. In the area method, daily cover is applied using soil or geosynthetic blankets (O'Leary and Tchobanoglous 2002; Pazoki and Ghasemzadeh 2020). In the case of canyon method, special landforms, such as canyons, ravines, borrow pits, and quarries, with natural features of depression and steep sides, are utilized to create the landfill (O'Leary and Tchobanoglous 2002).

Leachate Control

Leachate is the liquid that forms at the bottom of the landfill due to the initial presence of water in the waste, physicochemical changes occurring within the landfill, and infiltration of water via precipitation and uncontrolled runoff (O'Leary and Tchobanoglous 2002). The characteristics of leachate depend on several factors such as properties of the disposed hazardous waste, moisture content, temperature, site hydrology, landfill depth, and landfill age (Singa et al. 2018a; Gautam et al. 2019).

Typically, the landfill leachate is dark (black or dense brown) in color and exhibits low biochemical oxygen demand (BOD), high chemical oxygen demand (COD), and high redox potential (Gautam et al. 2019). Several studies have attempted to analyze and characterize the leachate from hazardous waste landfills. High concentrations of boron, organic phosphates, 1,4-dioxane, phthalates, bisphenol A, phenols, ethers, and chlorine have been detected in leachate samples from hazardous waste landfills (Yasuhara et al. 1999; Yamamoto et al. 2001). In another study, 190 different chemical compounds, including heavy metals, were detected in leachates from hazardous waste disposal sites (Yasuhara et al. 1997). The concentrations of organic phosphates and phthalates were found to be 0.8–10,900 ng/L and 0.1–2800 ng/L, respectively. Other constitutes of leachates from hazardous waste landfills include inorganic compounds such as iron, calcium, and magnesium and organic compounds such as acetic acid, methylene chloride, butyric acid, 1,1-dichloroethane, and trichlorofluoromethane (Ghassemi et al. 1984). Besides these, micro-pollutants such as PAHs and phthalate acid esters may be present in the leachate (Singa et al. 2020). Due to the presence of harmful constituents, leachate from hazardous waste landfills has been reported to produce toxic effects in soil, surface and groundwater, and even in humans in proximity to the landfills (Xu et al. 2018). This necessitates implementation of appropriate leachate containment, collection, removal, and treatment systems to avoid adverse effects of leachate on the environment and human health and ensure safe design and operation of the landfills.

Liner systems are employed to contain the leachate within the landfill and prevent its migration to the surrounding environment. Title 40, Part 265 of the US Code of Federal Regulations stipulates the use of two or more liners for hazardous waste landfills. A typical double-liner system is placed at the bottom and side slopes of the landfill prior to hazardous waste emplacement. The leachate accumulates within the liner system and then, by gravity, moves to one or more central collection sumps through a series of perforated drainage pipes (collection laterals). The removal of leachate from the landfill should be effective to ensure that the leachate level on the liner does not exceed 0.3 m. This requires that the bottom of the landfill is sloped and that sufficient number of drainage pipes are provided. Once collected, the leachate can be transferred for treatment or ultimate disposal. In a typical double-liner system, the bottom liner may be composed of compacted clay, a flexible membrane (synthetic) liner, or any natural material of suitable thickness and hydraulic conductivity (permeability). The top liner, however, must be a flexible membrane liner (Wright et al. 1989).

Liner systems require proper design and installation with appropriate selection of the liner material. The choice of liner material depends on the characteristics of the hazardous waste and the leachate and the geological and hydrogeological conditions (Williams 2005). The desirable liner features are as follows (Batstone et al. 1989a; Hovater 1989; Williams 2005):

- High compatibility with the hazardous waste to be disposed
- High chemical and biological resistance
- High durability
- Low hydraulic conductivity ($\leq 1 \times 10^{-7}$ cm/s)
- Optimum thickness ($\geq$5 m)
- High stability against overburden pressure (low compressibility)
- High resistance to climate-induced stresses such as freeze-thaw cycles
- Absorptive or attenuative capability
- Non-decaying (long service life)
- Easy to install
- Low cost

Clay, geosynthetic clay, or flexible membrane liners can be employed. Clay liners are based on natural clay soil (composed of clay minerals) of suitably low permeability. Typical clay minerals include illite and kaolinite, vermiculite, smectite, and chlorite (Pichtel 2014). The permeability of a clay liner depends on clay mineralogy, particle size distribution, plasticity, strength, moisture content, and degree of compaction (Williams 2005). To meet the hydraulic conductivity as a liner ($\leq 1 \times 10^{-7}$ cm/s), clay soil should contain at least 20% fine particles and maximum 10% gravel-sized particles and exhibit a plasticity index greater than 10%. In addition, rocks with diameter larger than 2.5–5 cm should not be present (Pichtel 2014). To create the clay liner, naturally occurring clay soil is excavated and then sieved to remove large solids. Subsequently, the moisture content and degree of compaction of the clay soil are adjusted to control the permeability of the final clay liner. When the amount of clay minerals in the clay soil is low, bentonite clay is added to achieve a reasonably low hydraulic conductivity. These liners can be referred to as bentonite-enhanced soils (Williams 2005). It is important to note that the hydraulic conductivity of clay liners can increase under thermal cycles or wet-dry cycles induced by climate conditions. Experimental investigations under simulated landfill conditions have showed that, for soils with low plasticity index of 9.5%, the hydraulic conductivity of the resulting clay liners can increase by one order of magnitude or by 12 times after 30 thermal cycles or 2 wet-dry cycles, respectively (Aldaeef and Rayhani 2015). This necessitates the use of a cover, such as a geomembrane or soil layer, to protect the clay liner from exposure to atmosphere during construction.

Geosynthetic clay liners consist of a bentonite layer that is supported or encased by a geotextile fabric or a geomembrane. In the case of supported type liner, a layer of bentonite is placed on top of a geomembrane. Encased-type liner, on the other hand, is composed of two geotextile layers with a bentonite intermediate layer. The layers are held together mechanically via needle punching, stitching, or chemical adhesion. Geosynthetic clay liners offer low permeability, high mechanical strength, simple and rapid installation, and an ability to self-heal through swelling of bentonite (Kong et al. 2017). Also, the service life can be up to thousands of years given that bentonite loss, hydraulic conductivity loss, and desiccation are avoided (Hoor and Rowe 2013). Loss of hydraulic conductivity can be due to suppression of osmotic swelling in the bentonite layer caused by its interaction with the leachate constituents (Jo et al. 2005; Benson et al. 2010; Setz et al. 2017). Desiccation, on the other hand, is a consequence of thermal gradients caused by temperature increase during decomposition of the disposed waste. The thermal gradients cause the moisture to move away from the geosynthetic clay liner, thereby, resulting in desiccation and subsequent cracking of the bentonite layer (Southen and Rowe 2005; Azad et al. 2012; Hoor and Rowe 2013).

To avoid desiccation and loss of hydraulic conductivity, recent research efforts have directed efforts toward the use of polymer-treated bentonite in geosynthetic clay liners. When tested against low-concentration contaminants, it has been shown that polymers can enhance the hydraulic performance of geosynthetic clay liners (Elhajji et al. 2001). In particular, anionic polymer has been employed to decrease the hydraulic conductivity of calcium bentonite in laboratory experiments (Razakamanantsoa et al. 2012). Results from another study have showed that bentonite-polymer composite, prepared by polymerization of acrylic acid within bentonite slurry, exhibits higher swelling capability and lower hydraulic conductivity compared to natural sodium bentonite when tested against aggressive inorganic solutions (Scalia et al. 2014). Lower hydraulic conductivity of geosynthetic clay liners with polymer-treated bentonite has also been reported in another study (Tian et al. 2019). In addition, the hydraulic conductivity of bentonite-polymer composite geosynthetic clay liners has been found to be suitably low to suppress the migration of heavy metals (Li et al. 2020). Recently, it has been reported that a geosynthetic clay liner with polymer-treated bentonite is less susceptible to cracking due to desiccation compared to the one with unmodified bentonite. However, this behavior was only observed when the temperature on the geosynthetic clay liner was 40 °C. Above this temperature, the difference in the degree of desiccation was found to be negligible (Yu et al. 2020).

Flexible membrane liners are based on synthetic materials with low permeability such as plastics or rubber. Commonly used flexible membrane liners for hazardous waste landfills are synthesized using polyvinyl chloride (PVC), high-density polyethylene (HDPE), low-density polyethylene (LDPE), chlorinated polyethylene (CPE), chlorosulfonated polyethylene (CSPE), and ethylene propylene diene monomer (EPDM) (Hovater 1989; Williams 2005; Pichtel 2014). Table 7.1 provides a comparison of these materials as landfill liners. Flexible membrane liners must be carefully selected in order to ensure compatibility with the hazardous waste. The chemical compatibility between wastes and membrane liners can be evaluated using the US EPA Method 9090 (Hovater 1989).

Careful installation of a flexible membrane liner is critical to its successful performance. It is important that the surface supporting the liner is smooth, even, and compacted. Also, proper seaming to join the individual liner sheets or rolls is important such that free edges are eliminated and tight seals and high seam strengths are

Table 7.1 Comparison of commonly used materials for flexible membrane liners in landfills. Information presented is summarized from Bell (2004) and Williams (2005)

	PVC	HDPE	LDPE	CPE	CSPE	EPDM
Chemical resistance	Low for organics; high for inorganics	Good	Good	Poor	Good	Poor for petroleum and halogenated solvents
Mechanical strength	High	Good (but susceptible to punctures)	Good (but susceptible to punctures)	Good	Low	High
Temperature tolerance	Poor	Good performance at low temperatures	Good performance at low temperatures	Good performance at low temperatures	Good performance at low temperatures	Good performance at low temperatures
Weather tolerance	Poor	–	–	Good	Good	Good
Ease of seaming	High	High	High	High	Low	Low

obtained. Typically, seaming is performed by overlapping the edge of the liner (5–15 cm) using thermal fusion, extrusion, chemical, or adhesive seaming techniques (Hovater 1989). The size and layout of the flexible membrane liner should be properly selected in order to minimize seaming of individual sheets or rolls. Also, as a recommendation, horizontal seams on slopes and transverse seams at the toe of slopes should be avoided. Seams on slopes should be parallel to the slope, and transverse seams should be 1–1.5 m away from the toe of slopes. Seam tests are of critical importance in order to assess the seam strength and integrity. These tests can be destructive, such as shear or peel tests using a field tensiometer, or nondestructive, such as dual seam, vacuum chamber, air lance, or ultrasonic methods (Cossu and Stegmann 2019). In addition, the liner should be anchored to the surface beneath (Hovater 1989). To support and protect flexible membrane liners, geotextiles (polypropylene or polyester fibers) and geonets (plastic drainage nettings) are employed (Hovater 1989; Williams 2005). As a secondary function, geotextiles also act as filtration media to remove solids from the leachate and avoid blockage of the drainage layers (Williams 2005). At the end of installation, hydraulic test should be conducted to identify any leakages from the liner. In addition, electrical leak location (ELL) survey can be conducted where high voltage across the liner (electrical insulator) is applied and the flow of current is used to detect the precise location of leaks. This method is well described in ASTM D6747 and D7002 standards (Cossu and Stegmann 2019). When installed properly, the design life of flexible membrane liner can range from many decades to many centuries (Rowe et al. 2019).

Besides the conventional liners discussed above, several research studies have proposed novel liner materials for hazardous waste landfills. Liner composed of a compacted mixture of bentonite and zeolite with optimized water content has been explored (Tuncan et al. 2003). Results indicated that the ideal landfill liner, with low hydraulic conductivity, was obtained when the bentonite to zeolite ratio was 0.10. Also, volcanic soil, with allophane as the main pedogenic mineral phase, has been investigated as landfill liner (Navia et al. 2005). The hydraulic conductivity was found to be suitable for use as landfill liner (in the range 5.16×10^{-9}–6.48×10^{-9} m/s). In addition, the volcanic soil liner possessed an ability to adsorb the pollutants in the leachate. Liners from crushed shales have been investigated (Mohamedzein et al. 2005). These liners had hydraulic conductivity in the order of 10^{-7} cm/s and showed good performance when subjected to calcium chloride solution. In another study, a mixture of sand and attapulgite (a natural clay) was employed as landfill liner (Al-Rawas et al. 2006). Scanning electron microscope (SEM) images revealed that attapulgite formed a coating between and around the sand grains which resulted in low porosity and, therefore, low hydraulic conductivity of the liner. It was reported that sand with 30% attapulgite and water content 2% above the optimum value satisfied the landfill liner requirements. In another study, marine clay soils were proposed as landfill liners (Chalermyanont et al. 2009). These liners exhibited hydraulic conductivities in the range 4.8×10^{-9}–1.1×10^{-8} cm/s and possessed ability to retain heavy metals such as chromium, lead, cadmium, zinc, and nickel. The use of marine clay as landfill liner was also investigated in a recent study, and hydraulic conductivity of $\leq 10^{-8}$ m/s was reported (Emmanuel et al. 2020b). In another recent study, olivine-treated marine clay was suggested as landfill liner material (Emmanuel et al. 2020a). Treatment of marine clay with 30% olivine was reported to produce the lowest hydraulic conductivity. In another recent effort, carboxymethyl cellulose (CMC) was used to decrease the hydraulic conductivity of bentonite liner (Fan et al. 2019). Results indicated that, for a given void ratio, CMC-treated bentonite possessed 20 times lower hydraulic conductivity compared to untreated bentonite. Also, the hydraulic conductivity of CMC-treated bentonite remained unchanged when exposed to real landfill leachate. A recent effort proposed a sustainable liner material composed of fly ash and bentonite (Garg et al. 2020). With 70% fly ash and 30% bentonite, the liner was able to meet the strength and hydraulic conductivity requirements. Numerical model studies suggested that the liner could be employed for 100 years when the applied thickness was 126–154 cm.

Leachate Treatment

Once collected, the leachate from hazardous waste landfills can be treated using a combination of biological, physical, and chemical techniques. The choice of treatment technique and the development of treatment train depends on the characteristics of the leachate, the constituents that need to be removed, and the required removal efficiency. Biological treatment processes are cost-effective and utilize microorganisms to degrade the nutrients and organic constituents present in the leachate. Both aerobic and anaerobic biological processes can be utilized for the treatment of leachate from hazardous waste landfills (Morgan 1990). However, care must be taken since the presence of toxins, heavy metals, and bio-refractory compounds and high concentration of sulfates and dissolved solids in the leachate tend to decrease the effectiveness of the biological treatment processes (O'Leary and Tchobanoglous 2002; Gautam et al. 2019). Aerobic processes for the treatment of leachate from hazardous waste landfills have been investigated. For example, lagoon with intermittent aeration has been utilized to treat phenol-containing oil shale ash heaps leachate (Orupõld et al. 2000). Results showed that the lagoon was able to achieve COD removal of 70% and phenol, methyl phenols, and dimethyl phenols removal of 95–99%. Also, activated sludge systems have been employed. For example, sequencing batch reactor (SBR) has been employed to treat leachate form an industrial waste landfill where 85–95% of total organic carbon (TOC) was removed (Irvine et al. 1984). SBRs are time-oriented systems that operate over repeated cycles (fill, react, settle, decant, and idle) and allow for flexible operation that can be easily controlled (Ozturk et al. 2019). Similarly, sequencing batch biofilm reactors (SBBRs) have been employed which utilize a packing to carry the activated sludge in SBR (Yuan 2014). These reactors have many advantages over conventional activated sludge processes such as larger surface area for bacterial growth, stable operation, and generation of smaller quantities of excess sludge (Chang et al. 2000). SBBRs with either membrane oxygenation system or bubble aeration have been used for leachate treatment (Dollerer and Wilderer 1996). Experimental studies with leachate from a hazardous waste landfill have shown that these reactors are able to reduce the dissolved organic carbon (DOC) by 60–68%. Also, combined anaerobic-aerobic SBR has been used to treat oil shale ash dump leachate (Kettunen et al. 1996). Besides removal of BOD and COD (97–99% and 73%, respectively), it was found that combined anaerobic-aerobic conditions can achieve phenol removal of up to 83–86%. Upflow anaerobic sludge blanket (UASB) reactors represent another option for the treatment of landfill leachate, specifically for the abatement of ammonium and nitrite (Mainardis et al. 2020). In a typical UASB, anaerobic sludge is suspended at the bottom of the reactor, and the flow of leachate is upward through the sludge blanket (Tiwari et al. 2020). It has been reported that UASB can remove 10.5–23.6% of influent COD and up to 78.7% of biphenyls in landfill leachate (Ismail et al. 2020).

Physical processes for leachate treatment include dissolved air flotation (DAF), carbon adsorption, air stripping, and membrane-based processes such as reverse osmosis (RO) (Renou et al. 2008). DAF is employed to remove suspended materials and microorganisms. In the context of leachate treatment, flotation has been utilized as a posttreatment after the biological treatment step for the removal of humic acids from synthetic landfill leachate (Zouboulis et al. 2003). Removal efficiencies as high as 99% were reported. Adsorption via activated carbon is a conventional water treatment technology that can efficiently remove the COD (organics). Air stripping, on the other hand, can effectively eliminate ammonium-nitrogen (NH_4^+-N) in wastewaters. Activated carbon adsorption and air stripping techniques have been applied to leachates from municipal solid waste (sanitary) landfills, and removal efficiencies of 91% (COD) and 99.5% (NH_4^+-N), respectively, have been reported (Renou et al. 2008). However, studies employing these techniques for the treatment of leachate from hazardous waste landfills are nonexistent. Nevertheless, both of these are well-established technologies for wastewater treatment and can be applied to leachates from hazardous waste landfills. The RO process utilizes a semipermeable membrane and high pressure to separate the contaminants (such as organics and inorganics) present in the leachate. It has been reported that RO can effectively treat industrial landfill leachate pre-treated using evaporation. Results showed that organics and ammonium reductions of 90% and 97%, respectively, can be achieved using RO (Di Palma et al. 2002).

In the case of chemical treatment processes, well-established methods such as coagulation/flocculation, chemical precipitation, and advanced oxidation processes (AOPs) can be utilized for leachate treatment. Coagulation/flocculation is typically used as a pre- or post-processing step for the treatment of leachate (Bakraouy et al. 2017). This treatment process can be used to remove suspended materials and organic and inorganic matter by addition of a coagulant to the leachate (Teh et al. 2016). The process results in agglomeration of small particles and colloids which are eventually removed as large particles. Typical coagulants that can be employed include aluminum sulfate, ferrous sulfate, ferric chloride, and ferric chloro-sulfate (Renou et al. 2008). Research studies have utilized coagulation/flocculation for the removal of humic acid (85%) (Zouboulis et al. 2004), turbidity (97%) (Amokrane et al. 1997), and COD and color (67% and 96%, respectively) (Monje-Ramirez and Velásquez 2004) from sanitary landfill leachates. Research studies have not utilized coagulation/flocculation specifically for the treatment of leachate from hazardous waste landfills. However, coagulation/flocculation is a well-established technology for wastewater treatment and can be utilized for the treatment of leachates from hazardous waste landfills. Chemical precipitation can be employed to remove specific contaminants from the leachate. For example, it can be used to remove NH_4^+-N using magnesium chloride hexahydrate ($MgCl_2 \cdot 6H_2O$) and sodium phosphate dibasic dodecahydrate ($Na_2HPO_4 \cdot 12H_2O$) (Li et al. 1999). Also, chemical precipitation, performed using ettringite ($Ca_6Al_2(SO_4)_3(OH)_{12}.26H_2O$) and barite ($BaSO_4$) minerals, has been utilized to remove 99% of sulfates in leachate from hazardous industrial waste landfill (Barbosa Segundo et al. 2019). AOPs rely on the generation of rapidly reacting hydroxyl radicals to remove contaminants and toxins from the leachate. Different types of AOPs have been utilized for the treatment of leachates from hazardous waste landfills. One possible option is ozonation which utilizes ozone to alter the molecular structure and oxidize the organics to biodegradable compounds that can be easily removed via biological posttreatment (Gautam et al. 2019). Ozonation of industrial waste landfill leachate has been studied where 50% reduction in COD was achieved (Haapea et al. 2002). Results from a recent study highlighted that ozonation can be used to remove up to 34.5% of dissolved organic carbon (DOC) from leachate sample obtained from a hazardous industrial waste landfill (Segundo et al. 2021). In the same study, it was shown that ozonation combined with hydrogen peroxide (H_2O_2) addition can increase the DOC removal to 45.2%. Other options include catalytic ozonation and ozonation combined with UV radiation or persulfate. These methods have been employed for the treatment of sanitary landfill leachates and may also be applied to leachates from hazardous waste landfills (Gautam et al. 2019). Due to its low treatment efficiency, ozonation is typically employed for pre- or posttreatment of landfill leachates. Fenton process is another type of AOP that can be employed for leachate treatment. In this process, H_2O_2 is activated in the presence of a ferrous catalyst to generate hydroxyl radicals. When used for the treatment of hazardous waste landfill leachate, the Fenton process has been reported to exhibit 56.49% COD removal (Singa et al. 2018b). In another study by the same authors, Fenton process in the presence of UV light (photo-Fenton process) was employed to achieve 68% removal of COD (Singa et al. 2018a). Also, heterogeneous Fenton oxidation, electrochemical oxidation, electrocoagulation, and electro-flotation (applied extensively to sanitary landfill leachate) may be employed for the treatment of hazardous waste landfill leachate (Gautam et al. 2019; Usman et al. 2020).

Some studies were done on combined processes for treating leachates from hazardous waste landfills. Combined process such as Fenton treatment followed by activated sludge process or activated sludge process followed by Fenton treatment has been found to be effective for the treatment of semicoke (hazardous waste rich in phenols) landfill leachate from an oil shale thermal treatment plant (Kattel et al. 2016). Besides efficient removal of BOD and COD, both processes exhibited lower treatment costs compared to the ozonation process. Also, the combination of activated sludge process and microfiltration has been investigated for treating leachate from a hazardous waste landfill (Setiadi and Fairus 2003). Results showed that the COD, BOD, and ammonia-N removal efficiencies of the combined process were 31.3%, 66%, and 98%, respectively.

A recent study has employed a combination of forward osmosis (FO) and membrane distillation (MD) to treat high salinity hazardous waste landfill leachate. The FO process utilized a sodium chloride (NaCl) draw solution (of high osmotic pressure) to transfer the liquid molecules in the leachate across a semipermeable membrane. The MD process, on the other hand, was used to treat the draw solution from the FO process. In the MD process, a temperature gradient was employed to allow for permeation of vapors (generated from the heated draw solution) across a hydrophobic membrane. The combined process showed TOC, salt, and total nitrogen (TN) removal efficiencies higher than 98%, 96%, and 98%, respectively. In addition, NH_4^+-N and heavy metals (mercury, arsenic, antimony) were completely removed. The biological, physical, and chemical treatment methods and their combinations discussed above are summarized in Table 7.2.

Due to the complex composition of landfill leachate, complete remediation necessitates the use of a multistage treatment strategy that combines different physical, chemical, and biological methods. A six-step treatment strategy has been recently proposed for the treatment of a hazardous industrial waste landfill leachate (Barbosa Segundo et al. 2020). The proposed treatment process consisted of (1) catalytic oxidation using H_2O_2 to remove sulfides and sulfites, (2) chemical

Table 7.2 Summary of studies related to the treatment of leachate from hazardous waste landfills

Treatment method	Treatment details	Main leachate characteristics	Removal efficiency	Reference
Biological	Aerated lagooning, intermittent aeration, nutrients: KH_2PO_4 (34–136 mg/l) and NH_4Cl (146–590 mg/l), activated sludge: 0.2 g mixed liquor suspended solids (MLSS)	BOD_7: 1650 mg/L, COD: 3090 mg/L, pH: 12	COD: 70%, phenol, methylphenols, and dimethylphenols: 95–99%	Orupõld et al. (2000)
	Sequencing batch reactor (SBR) (aerated), MLSS: 2600 mg/L, pH: 7.9–8.6	TOC: 2300 mg/L	TOC: 85–95%	Irvine et al. (1984)
	Sequencing batch biofilm reactor (SBBR) (aerated)	TOC: 2500 mg/L, COD: 5295 mg/L, BOD_5: 2600, pH: 9.1	DOC: 60–68%	Dollerer and Wilderer (1996)
	Sequencing batch biofilm reactor (SBBR) (combined anaerobic-aerobic)	BOD_7: 810–2700 mg/L, COD: 2000–4600 mg/L, pH: 12–13, phenols: 130–230 mg/L	COD: 73%, BOD: 97–99%, phenol: 83–86%	Kettunen et al. (1996)
	Upflow anaerobic sludge blanket (UASB) reactor, working volume: 10.8 L, sludge volume: 6 L	COD: 3421 mg/L, pH: 7.75	COD: 10.5–23.6%	Ismail et al. (2020)
Physical	Flotation, air flow: 200 cm^3/min	Humic acid: 50–300 mg/L	Humic acid: 99%	Zouboulis et al. (2003)
	Reverse osmosis (RO), polyamide membrane, pressure: 60 bar	BOD_5: 5000 mg/L, COD: 19900 mg/L, TOC: 5244 mg/L, pH: 8	Organics: 90%, ammonium: 97%	Di Palma et al. (2002)

(continued)

Table 7.2 (continued)

Treatment method	Treatment details	Main leachate characteristics	Removal efficiency	Reference
Chemical	Chemical precipitation using $BaSO_4$, $[Ba^{2+}]:[SO_4^{2-}] = 2:1$, pH: 8.5	BOD_5: 1100 mg/L, COD: 7063 mg/L, TDC: 3670 mg/L, pH: 8.7	Sulfate: 99%	Barbosa Segundo et al. (2019)
	Ozonation, dosage: 0.5 mg O_3/mg of COD	COD: 500 mg/L, pH: 10	COD: 50%	Haapea et al. (2002)
	Ozonation, dosage: 40 mg O_3/min	BOD_5: 360 mg/L, COD: 2809 mg/L, TOC: 3670 mg/L, DOC: 966 mg/L, pH: 8.4	DOC: 34.5%	Segundo et al. (2021)
	Ozonation with H_2O_2 addition, dosage: 50 mg O_3/min	BOD_5: 360 mg/L, COD: 2809 mg/L, TOC: 3670 mg/L, DOC: 966 mg/L, pH: 8.4	DOC: 45.2%	Segundo et al. (2021)
	Fenton oxidation process, pH: 3, H_2O_2/Fe_2^+ molar ratio: 3, reaction time: 150 min	BOD_5: 960 mg/L, COD: 3715 mg/L, pH: 9.53	COD: 56.49%	Singa et al. (2018b)
	Photo-Fenton oxidation process, H_2O_2/Fe_2^+ molar ratio: 3, reaction time: 90 min, UV source: 16 W	BOD_5: 850 mg/L, COD: 4123 mg/L, pH: 8.42	COD: 68%	Singa et al. (2018a)
Hybrid	Activated sludge/Fenton treatment, COD/H_2O_2/Fe^{2+}: 1/1/0.2 (w/w/w)	BOD_7: 330 mg/L, COD: 851, TOC: 367 mg/L, DOC: 243 mg/L, pH: 9.3	COD: 78%, BOD: 96%, DOC: 78%, NH_4-N: >99%, total phenol: 94%	Kattel et al. (2016)
	Fenton treatment/activated sludge process, COD/H_2O_2/Fe^{2+}: 1/0.5/0.1 (w/w/w)	BOD_7: 330 mg/L, COD: 851, TOC: 367 mg/L, DOC: 243 mg/L, pH: 9.3	COD: 62%, BOD: 91%, TOC: 56%	Kattel et al. (2016)
	Activated sludge/microfiltration, sludge retention time: 32 days, transmembrane pressure: 0.3 bar	COD: 2036 mg/L, BOD_5: 350 mg/L	COD: 31.3%, BOD: 66%, ammonia-N: 98%	Setiadi and Fairus (2003)
	Forward osmosis (FO)/membrane distillation (MD), FO draw solution: 4.82 M NaCl, MD feed temperature: 62.5 °C	TOC: 726.9 mg/L, salinity: 100,000 mg/L, NH_4^+-N: 18.5 mg/L	TOC: 98%, salts: 96%, TN: 98%, NH_4^+-N and heavy metals (mercury, arsenic, antimony): 100%	Zhou et al. (2017)

BOD biochemical oxygen demand, *COD* chemical oxygen demand, *TOC* total organic carbon, *TDC* total dissolved carbon, *DOC* dissolved organic carbon, *TN* total nitrogen.

precipitation to remove the sulfates, (3) biological treatment to remove organics and nitrogen species, (4) coagulation to remove suspended solids and some organics, (5) photo-Fenton process to degrade organics and enhance biodegradability, and (6) biological treatment to remove the biodegradables. The complete treatment train was able to reduce the COD to less than 1000 mg/L, a limit commonly acceptable for discharge to municipal wastewater treatment plants.

Landfill Gas Control

Unlike municipal waste landfills, gas release from landfills dedicated for hazardous waste disposal is uncommon. This is because most hazardous waste is received in stabilized or solidified form without biodegradable constituents (Pichtel 2014). However, in the case of organic hazardous waste disposal, landfill gas can be generated through anaerobic biodegradation of the organic matter. Typically, landfill gas contains methane and carbon dioxide (greenhouse gasses) as major components along with small amounts of volatile organic compounds (Williams 2005). With methane as one of the principal constituents, landfill gas can cause asphyxiation or form explosive or flammable mixtures with air. Also, the gas has an ability to travel vertically or laterally through the soils due to pressure and concentration gradients (Wright et al. 1989). Given these characteristics, monitoring and control of landfill gas become important in order to avoid harmful effects on site workers and the surroundings.

Landfill gas can be monitored via surface or subsurface monitoring techniques. Surface monitoring is based on the use of portable and wearable gas detectors with single or multi-gas sensors. Subsurface monitoring, on the other hand, utilizes probes for monitoring gas within the landfill and in the surroundings. The probes also allow for surface transfer and collection of gas (through a sampling valve) for laboratory analysis (Williams 2005). Some recent studies have proposed novel systems for monitoring of landfill gas. For instance, the use of infrared cameras/infrared thermography for the detection of landfill gas leaks has been suggested (Lewis et al. 2003; Tanda et al. 2017). Also, unmanned aerial vehicle (UAV) with embedded gas detectors has been utilized for landfill gas monitoring (Kim et al. 2021). Control of landfill gas is achieved via passive or active control systems. Passive control reduces the lateral migration of landfill gas by using atmospheric venting systems that are installed through the final landfill cover. This type of control is only recommended when the gas generation rate is low and toxic components are not present. Active control systems, on the other hand, rely on extraction of landfill gas by creating a negative pressure, for example, using a blower. Both vertical and horizontal extraction wells may be utilized for this purpose (O’Leary and Tchobanoglous 2002). The extracted gas is either collected or flared.

Final Cover

A final cover (cap) must be applied upon completion of the landfill or an individual cell. The final cover is an integral component of the landfill that serves a multitude of purposes listed below (Batstone et al. 1989a; O'Leary and Tchobanoglous 2002; Williams 2005):

- Contain, protect, and isolate the disposed hazardous waste.
- Prevent wind dispersion of the disposed hazardous waste.
- Reduce the infiltration of rainwater and surface water.
- Minimize the uncontrolled release of landfill gas.
- Minimize the ingress of air.
- Provide a surface for vegetation of the site.
- Suppress the proliferation of disease vectors and other organisms.

The final cover should meet the following criteria (O'Leary and Tchobanoglous 2002):

- Exhibit a low permeability.
- Maintain integrity, and possess an ability to withstand conditions such as erosion, abrasion, extreme climate, earthquakes, subsidence, and settlement within the landfill.
- Promote surface runoff.
- Allow for drainage of any percolated water.
- Have a low-maintenance requirement.

The final cover for secured landfills is constructed from a series of layers. A dense and compacted clay layer is first placed directly on top of the disposed hazardous waste. This layer functions to prevent the infiltration of water into the landfill. Next, a geomembrane cap, with chemical and physical properties similar to synthetic liners, is placed on top of the clay layer. The geomembrane cap also prevents the infiltration of water into the landfill. Unlike synthetic liners, the geomembrane cap is not exposed to landfill leachate, and, therefore, its chemical compatibility is not a serious concern. However, the geomembrane cap may experience strains due to waste settlement within the landfill. Nevertheless, its repair is easy due to proximity to the surface. Above the clay-geomembrane layer, a surface water collection and removal system is provided. This system is composed of granular soils, geonets, or geocomposites with drainage pipes and serves to direct the infiltrated water away from the bottom layers. Finally, a vegetative soil layer is added to complete the final cover for the landfill. The vegetative layer prevents wind and water erosion, enhances evapotranspiration, and improves the aesthetic features of the landfill. Typically, the minimum depth of each individual layer in the final cover is as follows: clay layer: 0.61 m, geomembrane cap: 0.02 m, drainage layer: 0.3 m, and vegetative layer: 0.6 m (Pichtel 2014). The final cover is typically sloped (3–5%) to promote runoff, minimize infiltration, and accommodate for waste settlement within the landfill (Wright et al. 1989). The suitability of the final cover (as well as the liner system for leachate control) and its susceptibility to percolation can be evaluated using the “Hydrologic Evaluation of Landfill Performance” (HELP) model. This model is a computerized water budget program that performs water balance on the landfill system using a quasi-two directional flow. It considers the flow in vertical direction, due to infiltration and evapotranspiration including saturated and unsaturated vertical flow, and in the lateral direction, due to lateral drainage and surface runoff, and takes into account the weather and the soil layer data (Piskin and Demirer 2007; Chabuk et al. 2018).

Advances have been made in order to utilize alternative, low-cost, and sustainable materials for the final cover in landfills. Evapotranspirative cover has been designed and employed in hazardous waste landfill (Zornberg et al. 2003). Unlike typical landfill cover that acts as a barrier, an evapotranspirative cover acts as a sponge to store the moisture during precipitation and then release the moisture back to the atmosphere as evapotranspiration. This type of cover is technically superior and is less vulnerable to desiccation and cracking compared to the clay layers, requires low maintenance, and can be easily constructed from a broad range of soils. In order to mitigate the effects of settlement and the resulting cracks in the final cover, using a self-recovering sustainable liner has been suggested (Kwon and Cho 2011). In this type of final cover, impermeable precipitates are formed from chemicals (such as diatomite and slaked lime with sodium carbonate catalyst) contained within the cover. In the case of crack formation or water infiltration, the final cover undergoes a self-recovery process in which the precipitates fill up the pores to maintain the hydraulic conductivity. In recent studies, the use of waste materials in the final cover has been explored. For example, the use of steel slag in landfill final cover has been investigated (Herrmann et al. 2010; Andreas et al. 2014). Mixtures of electric arc furnace slag and cementitious ladle slag were used within the final cover. The performance of the final cover in terms of infiltration and stability was found to be promising. Also, it was estimated that 60–70,000 thousand tons of construction materials required annually for landfill cover can be replaced by steel slags. Overall, the use of steel slag in the final cover allows for its economic recycling, reduces its quantity to be disposed, and decreases the material requirements for the construction of the final cover. Likewise, some recent studies have shown the possibility of using final covers containing mixtures such as clay/biochar (a carbon-rich solid obtained from pyrolysis of biomass) (Wong et al. 2016, 2017) and clay/fly ash (Shaikh et al. 2021).

Environmental Monitoring

Environmental monitoring is an important aspect of hazardous waste landfills. Routine monitoring of the vadose zone, groundwater, and air quality is critical to identifying any contaminant release from landfills and taking corrective actions to avoid harmful effects on the environment and public health. Several well-established techniques are available for environmental monitoring at landfills. These techniques either involve collection of samples for laboratory analysis (sampling techniques) or rely on some chemical and physical change to monitor the environment (non-sampling techniques) (O'Leary and Tchobanoglous 2002).

The vadose zone represents the unsaturated soil zone beneath the hazardous waste landfill. Monitoring of the vadose zone helps identify any release of leachate or gas from the landfill and provides early warnings of groundwater contamination. Liquid in the vadose zone can be monitored by sample collection using lysimeters (Wright et al. 1989; Singh et al. 2018). Typically, suction lysimeters are installed in the vadose zone. These are cylindrical devices consisting of a porous cup attached to a nonporous tubing. Vacuum is applied to collect a sample of the soil solution into the lysimeter through the porous cup. The collected sample is withdrawn into a sampling flask on the surface for laboratory or field analysis. Gas monitoring, on the other hand, may be performed via soil gas probes that obtain the gas samples from the vadose zone for analysis (O'Leary and Tchobanoglous 2002). Some research studies have focused on advancing the non-sampling methods for monitoring of the vadose zone. For instance, electrical leak detection method has been suggested (White and Barker 1997). In this vadose zone monitoring method, permanent grid of electrodes is installed beneath the landfill, and increase in electrical potential is used to detect leakages from holes in the liner system. In addition, time-domain reflectometry (TDR) has been suggested for continuous real-time monitoring of the vadose zone (Dahan et al. 2003; Aharoni et al. 2017).

Groundwater monitoring allows for detection of changes in the water quality caused by landfill leachate or gas. Monitoring wells with inert/corrosion-resistant casings are used for this purpose. Typically, four groundwater monitoring wells are installed (one up-gradient and three down-gradient). The required number of monitoring wells and their location must be decided, taking into account factors such as the nature of the aquifer, leachate characteristics, and groundwater depth, flow rate, and flow direction (Wright et al. 1989). Samples of groundwater can be collected using piezometers, and the groundwater quality can be monitored through hydrochemical analysis of the samples. Besides monitoring wells, recent studies have proposed alternative groundwater monitoring techniques. Electrical resistivity imaging (ERI) has been used for groundwater monitoring in landfills (Park et al. 2016). This technique relies on the fact that the electrical resistivity of landfill leachate is lower than that of clean groundwater. As a result, changes in the electrical resistivity of the groundwater can be related to contamination due to landfill leachate. This technique is attractive since it provides fast and reliable groundwater monitoring without the need for well drilling. Also, groundwater contamination at landfill sites and the spatial variation of contaminants have been studied using the very-low-frequency-electromagnetic (VLF-EM) survey (Monteiro Santos et al. 2006; Al-Tarazi et al. 2008). This survey utilizes radio signals with frequency ranges between 5 and 30 kHz to obtain and identify subsurface domains of low resistivities in which landfill leachate may have contaminated the groundwater.

Ambient air quality at hazardous waste landfills and in the vicinity can be monitored by collecting gas samples for laboratory or field analysis. Air samples can be collected using grab or active samplers. Grab samplers collect the gas in a collection chamber at regular intervals. Active samplers, on the other hand, allow for continuous collection and analysis of the air stream (O'Leary and Tchobanoglous 2002).

Post-Closure Care and Remediation

Post-closure care of a hazardous waste landfill should consider the following (Wright et al. 1989; O'Leary and Tchobanoglous 2002):

- Periodic inspection and maintenance of final cover in order to maintain its integrity.
- Continuous monitoring of vadose zone, groundwater, and ambient air.
- Periodic inspection and maintenance of environmental monitoring facilities.
- Analysis of samples from environmental monitoring facilities.
- Continuous operation and maintenance of leachate collection and removal system.
- Continuous operation and maintenance of gas control system.

Remedial action plan is required in order to take appropriate corrective actions in case any contaminant (leachate or gas) release is identified during post-closure environmental monitoring. The following points should be considered while developing the remedial action plan:

- Emergency procedures, such as site closure, evacuation, and liaison with emergency responders during flammable/toxic atmospheres.
- Methods and procedures for limiting the spread of contaminants in case of groundwater pollution.
- Procedures to rebuild or repair leachate control systems.
- Methods for treatment of contaminated groundwater.

7.2.4 Miscellaneous Landfill Considerations

Other requirements pertinent to the design and operation of a hazardous waste landfill are summarized below:

- The infrastructure around the landfill should be carefully planned. Access roads to the landfill site and to the disposal area should be provided.
- Equipment requirements should be evaluated, and appropriate equipment for excavation, soil compaction, and loading/unloading should be made available.
- A system should be in place to inspect the incoming hazardous waste and record and track the amount disposed.
- Equipment for loading, unloading, and transferring wastes should be provided.
- Unauthorized access to the landfill should be prevented by using fences and security measures.
- Wherever appropriate, safety and warning signs should be provided.
- The landfill operators should be adequately trained.

- Adequate PPE should be provided to the site operators.
- Safety equipment, such as first aid kits, should be available.
- Site offices and storage rooms for equipment should be provided.
- Appropriate welfare facilities should be provided.

7.3 Deep Well Injection

Deep well injection technique is applicable for the disposal of liquid hazardous wastes. It involves the injection of waste into subsurface (underground) porous, permeable, and saline water-bearing geologic zones that are confined vertically by impermeable strata (Warner 1989; Shammas and Wang 2010). Typically, the injection well consists of a series of concentric pipes that extend several thousands of feet from the surface level. The outermost pipe (surface casing) extends below the base of underground sources of drinking water (USDW). It is entirely cemented to the surface to avoid contamination of USDW. A long casing, extending into the injection zone, is provided within the surface casing. This casing is filled with cement up to the surface to prevent the flow of injected waste back to the surface. Liquid hazardous waste is injected into the well via injection tubing that is provided inside the long casing. The annular region between the inner casing and the injection tube is filled with a pressurized inert fluid (such as kerosene or diesel) and is sealed at the bottom using a removable packer to prevent the liquid backflow into the annulus (Shammas and Wang 2010; Pichtel 2014). At the surface, a wellhead caps the injection well which is provided with valves and gauges for injection control and monitoring (Batstone et al. 1989c). Detailed requirements related to injection wells (Class I wells) are outlined in the underground injection control program (UIC) program established by the US EPA, and the related regulations can be found in Title 40, Parts 144–148 of the US Code of Federal Regulations. As examples, deep well injection has been employed for the disposal of liquid radioactive waste (Rybalchenko et al. 2005), mercury-contaminated sludge (Yod-In-Lom and Doyle 2002), mercury sulfide and residual ash (Brkic et al. 2003), and acidic waste (de Graaff 1998).

7.3.1 Site Selection

Selection of a suitable site for deep well injection should consider the technical factors summarized below (Batstone et al. 1989c; Warner 1989; Shammas and Wang 2010):

- The injection zone should be saline water-bearing, sufficiently thick, and permeable enough to accept the wastes at safe injection pressures.

- The injection zone should not contain mineral resources of economic significance.
- The confining strata above and below the injection zone should be impermeable and sufficiently thick in order to confine the disposed waste.
- Geologic features such as faults, folds, and joints must be avoided to prevent escape of the injected waste.
- Fluid movement conditions in the injection zone should not allow for movement of waste in vertical and lateral directions.
- The injection and confining zones should not be penetrated with abandoned or unplugged wells.
- Sites with high seismic risk must be avoided.

As mentioned above, the confinement zones should ideally be impermeable. However, in practice, the confining zones are typically of low permeability that act to retard the movement of the injected hazardous waste. During its movement, the hazardous waste undergoes several geochemical processes that include ion exchange, osmosis, filtration, adsorption, and transformation. The velocity of the hazardous waste leaking through the confining zone is given by the following equation (Shammas et al. 2009):

$$v = \frac{Q}{A\varphi} = \frac{PI}{\varphi} \tag{7.1}$$

where v is the velocity (ft/day), Q is the leakage rate (ft^3/day), A is the leakage area (ft^2), φ is the porosity, P is the permeability (ft^3/day/ft^2), and I is the hydraulic gradient (ft/ft). The vertical permeability of the confining zone can be determined by analyzing the core samples obtained during drilling. When combined with the confining zone thickness and the pressure difference across the confining zone, the vertical permeability can be used to estimate the velocity of hazardous waste traveling through the confining zone. Besides permeability, the ion exchange capacity is often measured for the core samples from confining zone. The ion exchange capacity provides an estimate of the degree of subsurface treatment and attenuation which is of particular importance in the case of toxic wastes (Shammas et al. 2009).

The injection zone at the selected site should be able to receive the expected volume of hazardous waste to be injected. The hazardous waste and the injection zone should be characterized to avoid situations where undesirable changes in the injected waste and the injection zone may occur. These changes can be in the hazardous waste due to the injection zone conditions, chemical reactions between the waste and the injection zone formation or the injection zone fluids, or changes in the injection zone due physical/chemical interactions with the waste. Also, the dynamics of fluids in the injection zone is an important consideration since it affects the direction and rate of movement of the injected hazardous waste (Shammas et al. 2009).

7.3.2 *Waste Characteristics*

The characteristics of the liquid hazardous waste are important in assessing the suitability for disposal by deep well injection. The following factors should be taken into consideration (Warner 1989; Shammas and Wang 2010):

- The waste must be compatible with the materials used in the injection well system, with the confining and injection zones, and with the natural formation water. To ensure compatibility, pre-treatment of waste may be required prior to disposal.
- Wastes with high turbidity can cause plugging of the injection zone.
- Corrosive wastes should be neutralized as they can undergo undesirable reactions with the injection system components, the formation, and the formation water.
- High iron concentrations can also cause plugging/fouling due to changes in solubility caused by changes in the valence state.
- Organic carbon in the waste can result in fouling by aggravating the growth of microorganisms.

7.3.3 *Deep Well Injection Design*

Injection wells are typically drilled using the rotary method (Shammas and Wang 2010). An important design aspect is the bottomhole completion method that depends on the type of subsurface formation. Open-hole completion can be used for competent formations such as limestones, dolomites, and consolidated sandstones due to their ability to stand unsupported. In the case of incompetent formations, such as unconsolidated sands and gravels, gravel-packed completion is used. Also, perforated casing can be utilized for competent and incompetent formations where casing and cement are extended into the injection zone and perforations are provided to allow for waste injection (Warner 1989; Shammas and Wang 2010). The casing provides the necessary support to prevent collapse of the formation into the wellbore. Besides the bottomhole completion method, corrosion control and mechanical integrity are important design considerations. Corrosion control measures may include cathodic protection, use of corrosion-resistant materials in the well, and neutralization of corrosive wastes. Internal and external mechanical integrity tests are conducted to check for leakages in the casing, tubing, or packer and outside the casing, respectively. Internal mechanical integrity tests are conducted using the standard annulus pressure test (SAPT), the standard annulus monitoring test (SAMT), and the radioactive tracer survey (RTS). External mechanical integrity tests, on the other hand, include the use of temperature log, noise log, oxygen activation log, cementing records, or RTS (Gaurina-Medjimurec 2015). The procedures and the required equipment for conducting these tests can be found in the guidance documents prepared by the US EPA Region 5 UIC Branch (U.S. EPA 2008). The type of wellhead is another important design consideration. In the case where high backpressure is expected, for example, due to chemical reactions in the injection zone, the wellhead must be designed to bleed off the back flows to avoid excessive buildup of pressure and prevent the potential for blowout (Shammas et al. 2009).

Besides the injection well, the complete deep well injection disposal system requires the use of auxiliary upstream equipment. The waste is typically collected in a sump tank where an oil layer (in an open tank) or an inert gas (in a closed tank) is used to prevent air contact. For wastes containing oil, an oil separator is used downstream of the sump tank. The removal of oil is required to avoid plugging of the formation. A clarifier is then employed where particulates are allowed to settle under gravity. The residual particulates in the waste are removed using filtration. Metal screens coated with diatomaceous earth or cartridge filters can be used for this purpose. Filtration step is typically employed in the case the waste is injected into formations of low porosity. After filtration, the waste may be treated with a bactericide if the susceptibility to plugging due to high microorganism levels. The treated waste is collected in a holding tank from where it is finally injected into the injection zone using an injection pump. For highly porous formations, the liquid head may be sufficient for injection, and the injection pump may not be required (Shammas and Wang 2010).

The well design should also incorporate the necessary elements required for the protection of aquifers that are of domestic, industrial, or agricultural value. The aquifers may be contaminated by the injected hazardous waste or the displaced formation fluids. The necessary steps required for aquifer protection depend on the migration pathway taken by the contaminants. Defects in the casing can provide pathways for the injected hazardous waste to escape into the nearby aquifers. To avoid this, the casing material should be compatible with the hazardous waste. In addition, periodic casing integrity tests should be carried out. Techniques such as downhole camera and high-resolution Vertilog can be employed for casing inspection and identification of defects. Also, the use of separate tubing for injection can minimize casing defects by isolating the casing from the injected fluids. Vertical migration of contaminants through the annular region between the casing and the wellbore can also result in aquifer contamination. To eliminate this, the casing is cemented to the wellbore. Also, leakage through the confining zone (due to the presence of fractures) can contaminate the aquifer. This can be avoided by ensuring that the injection zones are deep, carefully selecting the deep well injection site, and thoroughly studying the geology of the confining zone (Shammas et al. 2009).

7.3.4 Monitoring Requirements

Typically, the volume, flow rate, chemistry, and biology of the hazardous waste; injection and annulus pressure; corrosion rate; and leakages need to be monitored during deep well injection operation. The volume and chemistry of the injected hazardous waste provide an estimate of the distance traveled by the waste within the injection zone. Biological monitoring and analysis of hazardous waste, on the other hand, is required to ensure that microorganisms are not being introduced into the well. The injection and annulus pressures are monitored continuously to avoid excessive pressures that may result in hydraulic fracture of the injection and confining zones and cause damage to the well facilities. In the case of corrosion monitoring of the well tubing and casing, corrosion coupons are typically installed in the well. These weight-loss specimens are made of the same material as the tubing and the casing. The weight of corrosion coupons is measured periodically to estimate the corrosion rate. To monitor and detect leakages in the casing-tubing annulus and the tubing, conductivity probes are employed. These probes can detect changes in the fluid chemistry caused by leakage of the injected waste. Alternatively, the inert fluid in the annulus region can be cycled continuously, and the return flow can be analyzed to detect the presence of hazardous waste that leaked into the casing-tubing annulus (Batstone et al. 1989c; Warner 1989).

7.3.5 Modeling of Deep Well Injection

Most of the recent advances are related to disposal by deep well injection and are related to the development of mathematical models that describe different aspects of deep well injection systems. Stochastic modeling of flow and transport in confining layers of deep well injection systems has been presented (Rhee et al. 1993). In this model, the confining layers were assumed to be binary random structures of pure sand and pure shale and were defined using Monte Carlo methods.

Modflow finite difference model was used to simulate three-dimensional flows in the confining layer. Results indicated that rapid transport may occur through the confining layer if the average shale fraction was less than 0.65. In addition, two-dimensional finite element model was used to model diffusion and advection-dispersion in the confining layer. With shale fraction greater than 0.65, the simulation results showed that the model waste (dilute acetonitrile solution) did not extend beyond the confining layers over a period of 10,000 years.

Numerical model for studying the movement of injected waste within the hydrogeologic system has been presented (Jin et al. 1996). The model is available in both cylindrical and Cartesian coordinates and was developed by creating a convection cell around the injection well with a buoyant injection that formed a lens within the injection zone. In another study, a well injectivity decline (WID) simulator was developed for modeling the well performance during deep well injection (Saripalli et al. 2000). The simulator was employed to study the well performance by considering factors such as the waste quality and suspended materials, formation characteristics, completion type, injection rate, injection pressure, initial well or formation damage, and gravels surrounding the wellbore. Simulation results showed that well plugging and, consequently, poor injection performance is caused by high concentration of suspended solids in the waste, low injection rate, low injection pressure, formation heterogeneity, and low formation porosity and permeability.

Recently, a mathematical model with an analytical solution was presented to describe the contaminant plume movement at injection disposal site of liquid radioactive waste (Malkovsky et al. 2019). The model considered both topography-driven (regional) and buoyancy-induced components of the groundwater flow. The results from analytical solution were found to be in good agreement with the numerical solution.

Besides the models related to the well, the cost of deep well injection has been mathematically studied (Mogharabi and Ravindran 1992). In this study, a model for selecting the best disposal system design and operating policies was proposed based on linear goal programming techniques. The proposed method resolved the conflicting objectives such as cost, environmental regulations, equipment utilization, and waste quality requirements before injection and produced 40% savings by utilizing the design predicted by the model.

7.4 Underground Geologic Repositories

Emplacement of hazardous waste in deep underground geologic repositories or mines is considered to be one of the best disposal techniques. Historically, geologic repositories were employed for disposal and isolation of radioactive waste, such as reprocessing effluents and spent fuel-rod assemblies, in salt host rocks (Testa 1994). With advancements in the field, geologic repositories of alternative host rocks have been explored, and the use of geologic repositories has been extended to other types of hazardous and toxic wastes. Disposal in geologic repositories is an attractive option given the ever-increasing regulatory requirements and prohibitions placed on landfilling and deep well injection techniques. In general, disposal by emplacement of hazardous waste in geologic repositories offers the following advantages (Testa 1994; Kaliampakos et al. 2006):

- Complete isolation and protection of hazardous waste.
- Very low probability of hazardous waste leakage.
- Very low probability of leakage to the surface environment.
- Protection of hazardous waste from severe weather conditions and effects of earthquakes.
- Limited or no generation of wastewater/leachate.
- Easy segregation of hazardous waste which makes future inspections easier.
- Limited need for long-term and aftercare monitoring due to high level of protection provided by the geologic medium.
- Lower operating cost compared to landfills due to lower monitoring requirements.
- Low land and construction costs in the case abandoned underground mines are utilized.
- No concerns related to aesthetic and visual impacts of disposal.

7.4.1 Site Selection

As with landfilling and deep well injection, selection of an appropriate site is a critical part of disposal in underground geologic repositories. The type and characteristics of the host rock and the stability and hydrogeological characteristics of the site are among the key criteria that influence the decision on selecting the most suitable geologic repository for the disposal of hazardous waste (Testa 1994; Kaliampakos et al. 2006).

The host rock, into which the repository is excavated, is the main geologic barrier that isolates the hazardous waste from the biosphere. In general, the host rock should exhibit the following characteristics (Testa 1994; Pusch 2006a):

- Low permeability to ensure high isolation capacity.
- High thermal conductivity to maintain low temperatures.
- High strength and stability to endure the effects of geologic activities such uplift and seismic events.
- Absence of unfavorable geologic features such as discontinuities, faults, folds, and joints or other features allowing infiltration of groundwater.
- High degree of homogeneity (both vertical and lateral).
- Large lateral area to allow for excavation of the repository and provide adequate protection to the waste.

Potential host rocks for geologic repositories include crystalline, argillaceous, and salt rocks; basalts; volcanic tuffs; and anhydrites (Testa 1994; Pusch et al. 2018). Crystalline rocks are composed of tightly packed grains of minerals and are formed from solidification of magma of molten or partially molten rocks (igneous rocks) or from sedimentary rocks under high pressures and temperatures (metamorphic rocks) (Pusch et al. 2018; Ewing and Park 2021). Crystalline rocks such as granite and gneiss are composed of quartz (10–40 wt%), feldspars (10–75 wt%), and heavy minerals (5–20 wt%) (Pusch et al. 2018). In general, crystalline rocks possess excellent stability for underground repository construction but exhibit high permeability (Pusch 2006a).

Also, granite offers high chemical stability, low water content, and good sorptive capacity. However, the use of granite in repositories is challenging due to frequent presence or generation of faults and high excavation costs (Testa 1994). Argillaceous rocks, on the other hand, exhibit very low permeability but poor stability when compared to crystalline rocks (Pusch 2006a). Shales are well-known examples of argillaceous rocks. Besides their low porosity and permeability, shales possess a high sorptive capacity and an ability to seal fractures due to plastic flow. However, due to the inherent presence of water, shales may release water into the repository under thermal loads.

Salt rocks (salt domes) have been widely utilized for the disposal of radioactive wastes. These rocks are formed from salt deposits under high pressures and temperatures and are essentially homogeneous, free of discontinuities, impermeable, geologically and chemically stable, and typically possess a large lateral area.

However, the presence of brine in rock salts can corrode the waste containers. This is specially of concern in the case of radioactive wastes where the thermal field can cause migration of brine toward the waste canisters. Also, the solubility of rock salts in groundwater or other water sources, waste sinkage due to creep, and compatibility of salt with the hazardous waste may limit the utilization of rock salts as the preferable choice of host rock in geologic repositories (Testa 1994; Pusch 2006a).

Basalts possess intermediate thermal conductivity, high thermal load capacity, good sorptive capacity, high strength, and low permeability. However, basalts typically consist of zones of secondary permeability which may enhance their ability to allow infiltration of water. Volcanic tuffs (welded or zeolitic) can also serve as geologic barriers in geologic repositories. Welded tuffs, formed from volcanic ash, possess low porosity and permeability and high thermal load capacity and have strength and thermal conductivity values comparable to those for basalts. Zeolitic tuffs contain zeolites and exhibit high sorption capacity due to their open structure. The open structure, however, imparts them with high porosity and permeability and moderate strength. Anhydrite deposits, composed of anhydrous calcium sulfate, are homogenous and impermeable and possess high thermal conductivity and chemical stability, making them suitable for geologic repositories. However, interactions with water can convert anhydrites to gypsum which can induce changes in porosity and permeability (Testa 1994).

The stability of the repository structure is an important consideration during site selection. High structural stability is critical to ensure safe and long-term disposal of hazardous waste. The suitability of the site is also dictated by its hydrogeologic characteristics. The presence of artificial penetrations and their depths and locations at the proposed site should be identified. Visual inspections and geophysical methods can be employed for this purpose. In addition, laboratory analysis should be carried out to study the mineral content and its composition, heterogeneity of the host rock, and the presence of fluids and their characteristics. In general, the environment in the geologic repository should be dry with little or no groundwater. Also, the geo-mechanical properties of the host rock such as density, porosity, permeability, water content, plasticity, strength, compressibility, and swelling potential should be determined via laboratory analyses. Any structural discontinuities, faults, and tectonic activities should also be identified using, for example, field mapping (Testa 1994).

Besides the aforementioned factors, it is important that the proposed site is located away from ore deposits and oil and gas fields and areas of high population density. In the case the proposed site is an abandoned mine, the remaining exploitable ore and its current and future economic significance should be considered before converting the mine into geologic repository for hazardous waste disposal.

Selection of sites for geologic repositories has been explored in some recent research studies. To avoid local and regional opposition, the importance of public participation in site selection processes of nuclear waste repositories has been highlighted (Krütli et al. 2010). In this study, a functional dynamic view of public participation was proposed that combined the decision-making process with specific types and extents of public participation. Different levels of public participation were considered (information, consultation, collaboration, and empowerment) and were combined with the decision-making process in a temporal and phased framework.

In a very recent study, a new approach to site selection was proposed that utilized GIS technology (Perković et al. 2020). The proposed approach utilized site exclusion and comparison criteria. Exclusion criteria included flooding safety, seismotectonics and seismology, lithological and geomorphological characteristics, hydrogeology, population density, protection of natural and cultural heritage, mining and mineral exploitation, and protected areas. Comparison criteria, on the other hand, included technical aspects, installation safety, and location acceptance. Using the selection criteria and merge layers and symmetrical difference layers in GIS technology, a map of potential sites was finally created.

7.4.2 Repository Design

The geologic repository should be designed with sufficient capacity to accommodate the expected volume of hazardous waste to be disposed. The design should also consider the repository life span which depends on the characteristics of the hazardous waste to be disposed. For certain wastes, the hazard will decrease with time (e.g., radioactive wastes), while other wastes may require a much longer repository life span (e.g., wastes with indefinite toxicity). In general, the geologic repository should be designed to isolate the disposed waste over a long period of time (Testa 1994). The design and dimensions of drifts (horizontal or nearly horizontal openings or tunnels within the repository) and rooms are also important considerations. The geometry and dimensions of drifts and rooms should provide adequate clearance to accommodate and move the waste and the equipment. Typically, horseshoe- or rectangular-shaped drifts and rooms are utilized (Pusch 2006a).

Three types of geologic repositories can be used for hazardous waste disposal, namely, existing mines, salt caverns, and new mines. Existing mines that have been abandoned after extraction of the exploitable ores can be utilized for hazardous waste disposal. These mines are usually mined using the conventional room-and-pillar method (Testa 1994). Abandoned mines require careful examination before hazardous waste disposal since stabilization may be required depending on the presence of unstable rocks and internal stresses due to tectonics and glaciation (Pusch 2006a). Several inactive underground mines have been reused as waste repositories in Europe. A comprehensive list of these mines can be found elsewhere (Kaliampakos et al. 2006).

Salt caverns, on the other hand, are cavities in salt formations that are developed by drilling and cementing concentric casings into the salt formation. An uncased hole is also drilled to expose the salt formation to leaching. Water introduced via the annulus dissolves the salt which, in the form of brine, returns to the surface via the outing casing. Waste is disposed into the caverns via solution mining which can be achieved using one of the several possible methods. For example, brine-balanced method can be used for the disposal of liquid and slurry wastes. Once injected into the cavern, the brine is displaced and forced up the casing for collection. It is important that the specific gravity of the liquid or slurry waste is higher or lower than the brine so that the waste remains at the bottom or top of the cavern, respectively. If the specific gravity of the waste is close to that of the brine, cross-contamination of brine limits the applicability of the brine-balanced method. To avoid cross-contamination of brine, the gas-balanced method can be utilized where the brine is displaced using an inert gas at high pressure. The cavern is sealed at the minimum design pressure after which the gaseous, liquid, or slurry waste is injected until the design pressure is reached. However, in this case, the size of the cavern should be limited to maintain structural integrity.

In situ disposal of solidified waste is another option where the waste is mixed with a cement of polymer slurry prior to injection into the cavern. The size of the cavern is also limited in this method to ensure the structural stability. In addition, string-of-pearls method can be utilized. In this method, a series of stacked caverns are constructed. Brine is removed from the deeper cavern and filled with the waste. The top portion of the cavern is then sealed with a cement plug after which the waste can be directed to the upper cavern. Again, to maintain structural integrity, the size of the cavern is limited in this method (Testa 1994). Besides existing mines and salt caverns, new mines can be constructed given that the selected site meets the selection criteria discussed earlier.

Depending on the nature of waste to be disposed, geologic (mine) repositories may require engineered (man-made) barrier systems which supplement the natural barrier provided by the host rock for effective isolation and containment of the waste. For instance, disposal of radioactive waste typically requires the use of a multi-barrier system. Waste containers/canisters, placed inside the repository, can be considered as part of the engineered barrier system. However, to ensure effective isolation of the waste from the biosphere, more sophisticated engineered barrier systems are required. These barrier systems around the waste are typically constructed using clay and cement/concrete (Pusch 2006b). In the case of clay barriers, bentonite and bentonite/sand mixtures are typically used due to their inherently low permeability in saturated state and self-healing ability (Sellin and Leupin 2013).

Recent advances related to disposal in geologic repositories are related to the design, use, and performance evaluation of the engineering barrier systems in radioactive waste repositories. The physicochemical properties of bentonite barriers are susceptible to changes due to factors such as humidity, temperature variations, and fluid interactions. A model to describe long-term diffusion reaction in bentonite barrier for radioactive waste confinement is available (Montes-H et al. 2005). This model utilized thermokinetic hydrochemical code (KIRMAT: kinetic reactions and mass transport) to simulate the chemical transformations due to geochemical and cation exchange reactions, the diffusion of chemical species into the barrier, and the changes in the swelling capacity. Results indicated that the bentonite barrier was significantly affected after 10,000 years due to contact with geologic fluid. Also, the results highlighted that the swelling capacity declined significantly within the geological barrier-engineered barrier interface. In another study, KIRMAT was used to study the changes in the bentonite barrier due to the geologic fluid and Fe^{+2} ions from the radioactive waste canisters (Marty et al. 2010). The results again highlighted significant changes in the bentonite barrier due to interactions with the natural barrier. Also, feedback effect of corrosion products resulted in slow diffusion of Fe^{+2} through the bentonite barrier and reduced the corrosion rate from 5 to 0.2 μm/year. The presence of bacterial communities in bentonite barrier layers has been established (Lopez-Fernandez et al. 2015), and the effects sulfide-producing bacteria on the barrier performance has been studied (Pedersen et al. 2017). It was found that bentonite clays immobilized the corrosive sulfide from sulfide-producing bacteria and the rate of sulfide diffusion depended on the bentonite density. The immobilization of sulfide reduced the transport of sulfide toward the metal canisters, thereby, reducing the corrosion susceptibility. However, the sulfide also reduced ferric iron in contact with the bentonite clay to ferrous iron. This was found to cause destabilizing effects of ferrous iron on the dioctahedral clay smectites. Also, additives for bentonite barriers have been proposed. It has been shown that addition of activated carbon in bentonite barrier helps in sorbing technetium radionuclide (Makarov et al. 2021). The performance of cement/concrete barriers has also been explored using modeling studies. In one study, the geochemical interactions between concrete barrier and mudrock were investigated using the reactive transport code Hytec, and it was shown that sulfate can strongly alter concrete engineered barrier based on pure Portland-based cement (Trotignon et al. 2007). In another study, the reactive chemical transport model of HYDROGEOCHEM 5.0 was utilized to show that hydrogen ion, sulfate, and chloride can significantly degrade the concrete barrier in radioactive waste repositories (Lin et al. 2016).

7.5 Ocean Disposal

Although technically feasible, offshore disposal of hazardous waste in oceans is highly constrained by regulatory requirements. Theoretically, ocean disposal can be in the form of ocean incineration or ocean dumping. Compared to land-based incineration facilities, ocean incineration on ships is of lesser concern to the public due to its operation far from the coast. Also, gas cleaning facilities are typically not required since hydrogen chloride can be effectively diluted in the seawater (Batstone et al. 1989c). Ocean incineration is regulated under the "Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter" of 1972, also known as the London Convention. As per this global convention, which has 87 Contracting Parties as of January 2021 (U.S. EPA 2021), ocean incineration of sewage sludge and industrial waste is completely prohibited.

Ocean dumping, on the other hand, is based on the principle of dilution and dispersion of waste that is dumped directly into the ocean. In this disposal technique, it is assumed that, once discharged into the ocean, the waste is immediately diluted to very low concentrations such that its environmental impacts become negligible. To avoid environmental impacts, the use of ocean dumping should be limited to the disposal of wastes that can be naturally degraded, neutralized, or transformed by the chemical and biological process occurring within the ocean. Ocean dumping can be carried out in shallow or deep sea. Shallow-sea dumping has low transportation cost and localizes the adverse impacts of the disposal. Deep-sea dumping, on the other hand, allows for higher degree of dilution and dispersion (Visvanathan 1996). However, similar to ocean incineration, ocean dumping of industrial waste and sewage sludge has been prohibited under the London Convention. In the USA, under the Marine Protection, Research, and Sanctuary Act of 1972 (MPRSA, which implements the requirements of the London Convention) and its 1988 amendment, the Ocean Dumping Ban Act (ODBA), it is unlawful to carry out ocean dumping for wastes such as high-level radioactive wastes, sewage sludge, medical wastes, industrial wastes, known carcinogens, mutagens, or teratogens and certain heavy metals (U.S. EPA 2020). Due to the stringent requirements and prohibitions placed on offshore hazardous waste disposal, recent advances in this area have not been witnessed.

7.6 Summary

Landfills, injection wells, and underground geologic repositories can be used for the safe disposal of hazardous waste. However, selection of suitable site and proper design is of critical importance to ensure the effectiveness of these disposal methods and guarantee long-term isolation of the hazardous waste. Recent research studies have made significant efforts in providing the necessary guidelines and methodologies for the selection of suitable site and improving the design of landfills, injection wells, and underground geologic repositories. In the case of ocean disposal, its use for hazardous waste disposal is highly constrained due to imposed prohibitions.

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