CHAPTER II

LITERATURE REVIEW
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2.1 BACKGROUND

The researchers all around the worlds of various organization and institute have reported leaching of pollutants from unplanned landfills to groundwater. Brett Robinson et. al. (2005) reported leaching of copper, chromium and arsenic from treated vineyard posts in Marlborough, New Zealand. But very few studies have been done in Developing Countries like India, where 16.67 % of world population is living (India Census, 2001). In the report of central pollution control Board and Central Ground Water Board, New Delhi, India is planning for planned dumping site in Noida and Bangalore.

High concentration of fluoride in groundwater was reported in Delhi and surrounding states of Delhi (CGWB, 1995). Meenakshi et. al. (2004), have reported fluoride contamination in groundwater of the Jind district of Haryana (neighboring state of Delhi), and reported fluorisis problem in people living in district. A vast amount of literature is available on presence of heavy metals in wastes (Tisdell & Brestin 1995, Shivhare and Pandey 1996 etc). Acidity of the soil solution and solubility of metals are closely related (Tayler et al., 1987). According to Taylor, the acidity of soil increases due 3 to 5 fold with increase in metal concentration. This causes high leaching of hazardous metal (Blais et al. 1993). The solid waste containing metals at low pH have high pollution potential to contaminate groundwater (Olaniya et al., 1992). Similarly Kunwar et. al. (2005) has tried to study surface and groundwater interaction in alluvial aquifer of Gangetic plain. Allelign et. al. (2005) has reported groundwater pollution by using concentration time series recorded during an integral pumping test in vicinity of contaminate zone. Inversion of concentration breakthrough curves recorded during an integral pumping test could be used as an alternative or complementary method to quantify contaminant mass fluxes, and hence determine the state of groundwater contamination (Allelign et. al. 2005). This approach is particularly suited when only a
few sampling points are available to quantify the level of groundwater contamination and little is known about the contaminant source area.

The high concentration of arsenic was reported in groundwater around Jalangi in West Bengal, India (Mohammad Mahmudur Rahman et. al. 2005). While, Zareena Khan et al. (2005) in his study indicates that soil constituents like organic matter, clay, iron and aluminum oxides increase the adsorption capacity of soils thereby prevent the leaching of hazardous organics into the groundwater.

2.2 IMPACT OF LANDFILL ON GROUNDWATER QUALITY

The study on landfill indicate that landfill leachates may pollute adjacent surface and ground water has lead to a number of studies in order to understand the effect of solid waste disposal on the ground water quality. It is estimated that there is some role of complexion in determining the fate of metals in sanitary landfill. The leachates formation depends upon several factors whose importance had not yet been adequately investigated or reported in the available literature (Bloomfield and Pruden, 1975). Some Scholar like Pohland reported that enhance degradation of landfill wastes, degrade or immobile harmful compounds within the waste mass and store excesses leachates (Pohland, 1975). Sui ling Wang et.al (2005) reported occurrence of arsenic contamination in groundwater in Canada and their spatial distribution in aquifer system by analyzing the bioremediation of contaminant in aquifer system.

The hydro-chemical study shows that ground waters in the vicinity of the landfill are characterized by high contents of organic and inorganic chemicals: more than 4.5 mS cm\(^{-1}\) in electric conductivity, 1600 mg l\(^{-1}\) in chloride, 1000 mg l\(^{-1}\) in sulfate, 15–25 µg l\(^{-1}\) in cadmium and 60–100 µg l\(^{-1}\) in chromium (Amina Chofqi et. al. 2005). The main pollution source is linked to the infiltration of leachate that conveys a strong polluting load and to the direct contact of the leachate with waters of the aquifer in the landfill carriers. This groundwater pollution and its distribution in the unsaturated zone are favored by: low depth of water table, high soil permeability, absence of drainage
systems of the leachate, direct contact of groundwater with leachate at the bottom of landfill carriers, semi-arid climate (Amina Chofqi, 2005).

2.2.1 Groundwater Contamination
Olaniya and Saxena (1977) had conducted studies for groundwater contamination for the period of two years (1969-1971) at Jaipur in Rajasthan, India. The results indicated higher concentration of TDS, COD, chloride, Fe salts was found to the depth of 450 meters. Whereas, the contamination of some toxic heavy metals like Cr, Cu, Fe, Mn, Cd, Pb in ground water around industrial and sewage waste of Parwanoo district, Himachal Pradesh, India was reported by Singh et al. (1976).

The fate of heavy metals in solid waste leachates leaking into the groundwater or fed to a wastewater treatment plant not only depends on its bulk concentration but also on the actual speciation of heavy metal. Hegde et al. (1992) had found that open wells and bore wells, located in the vicinity of sewage courses found with Ca, Cl and Na ions at Hubli city, Hyderabad. Similarly Shivkumar and Biksham (1995) in their work mentioned that the elements like Cu, As, Se, Fe and F are five to ten times higher in surface soil and subsurface soil and in ground water around Medak district, Hyderabad.

The industrial activities such as sugar mills, dying, bleaching and allied units release effluents without treatment, which may percolate down to the aquifer and foul it. The parameter analyzed in this water as pH 7.4- 8.0, EC 1940-21000 μmhos/cm, heavy metals Cu 1.04 –7.24, Cr 1.52-2.68, Cd ND-0.60, Zn 8.92-44.04 mg/L, in the order of Zn>Cr>Cu>Cd. Leachate will flow along the path of least resistance and in low hydraulic conductivity areas. The waste near flow channel was preferentially wetted compared to more distant waste, which suggests that channeling may decrease opportunity for leachates storage and enhancement of the biological activity.

Contamination of ground water is common in the areas surrounding city refuse dumping sites. This is more so where dumping is done in low-lying areas and the rate of percolation through the soil is high. The composition of leachates and the presence of heavy metals in the leachates from wastes had been reported by number of workers.
The rate and extent to which metals attenuate in the soil depends on the soluble complexes formed, the heavy metals speciation and pH of wastes (Bloomfield and Pruden, 1975; Glauser et al. 1988 and Blais et al. 1993). They had also reported high percentages of metals in the leachates of the size less than 500 molecular weight fraction. Das et al. (1989) studied the leaching behavior of granular solid wastes in Netherlands. Whereas, Knox and Jones (1997) in Southern Ontario (Canada) found, that leachates were capable of forming complex with Cd. They also reported that the formation of complex is associated with molecular weight of the compound. Their behavior suggested that complex might be formed between phenol’s hydroxyl groups and Cd.

Lin et al. (1996) reported that the soluble organic matter of leachates from municipal and industrial wastes influence the heavy metals movement. The researcher from Kuwait, analyzed leachates from Al-Sulaibiyah solid waste disposal site in Kuwait and found high concentration of Ni and V in the leachates samples, which indicated the presence of petroleum related wastes at the site. The other researcher examined the marine coastal sediments from Togo, to ascertain the geo-ecological impact of dumping of phosphorite tailing into the sea. They reported trace metal concentration range from 2-44 Cd, 22-184 Cu, 19-281 Ni, 22-176 Pb, 179-643 Sr, 38-329 V and 60-632 mg/Kg Zn.

2.3 LITERATURE REVIEW OF GROUNDWATER QUALITY

Groundwater contamination is the degradation of the natural quality of groundwater as a result of human activities. The adverse impacts of landfill leachates and adjacent surface water on ground water have prompted this study. Other scholar has tried research on the constituents of leachates (Kjeldah et al, 1993, Tejero et al, 1993, Flyhamman et al, 1995; Fatta et al, 1998) samples collected from different landfill distributed all along the world.

The study on fresh water resource and availability of water resource is emerging topic of future research. Further, water quality is one of the most burning topics of today
research; every day groundwater quality is deteriorating by anthropogenic activity. The research have reported change in water quality of water due to industrialization, urbanization etc. Chapelle and Wilson (1998) presented a method to determine a target level of source concentration treatment based on a uniform natural attenuation capacity (NAC) within the aquifer and demonstrated its use for meeting regulatory compliance concentration at location down gradient of the source. Some researcher used to plan an augmentation to an operating groundwater extraction network to accelerate the removal of dissolved TCE and PCE by extraction wells. Solution were sought for multiple goals that focused on mass or concentration values at specific times along property lines, within specific aquifer zones, and/or at selected wells.

There have been several prior reviews of the state of optimization in Groundwater remediation problems. These include Willis and Yeh (1987), Ahlfeld and Mulligan (2000), and Pinder et al. (2002) etc. The later contains a description of a number of issues and recent applications of optimization in groundwater remediation and monitoring.

Contemporary increasing demand of fresh water for drinking water supply or in industry and agriculture imposes a key role of groundwater as one of main water sources. However, nowadays-intensive fertilizer applications in agriculture as well as atmospheric deposition constantly lead to groundwater quality deterioration. The diffuse sources of pollution are difficult to describe because, loads are usually governed by spatially and temporally non-homogeneous processes of physical and biochemical phenomena (Wu and McGechan, 1998, Refsgaard et al., 1999 and Vinten Dunn, 2001). A sound understanding of soil processes is necessary to enable better management of human activities that will result in minimizing groundwater and surface water contamination (Hack-ten Broeke and De Groot, 1998, Hendriks et al., 1999 and Sung et al., 2002). In general, the unsaturated zone models are useful tools in predicting short- or long-term effects of applied measures and can be used to optimize agricultural practice while minimizing the impact on the environment (Feddes et al. 1988, Wu and
The unsaturated zone models have a varied degree of abstraction concerning the simulation of complex physical and biochemical soil processes (Feddes et al., 1988, Willigen, 1991 and Diekkruger et al., 1995) and for that reason it is not easy to choose a suitable model to estimate the impact of landfill and agriculture on groundwater quality. Therefore, it becomes important before using these models to examine their scientific background and applicability (Wu and McGechan, 1998, Vanclooster et al., 2000, Fernandez et al., 2002, Simunek et al., 2003 and Eitzinger et al., 2004). Recent comparative investigation of soil water flow and nitrogen transport models has been reported also in lately completed EC funded project (Walther et al., 2002). In a project two well-known Dutch model, i.e. SWAP version 2.0.7d (Van Dam et al., 1997) and ANIMO version 3.5 (Kroes and Roelsma, 1998) were tested for experimental plots in Bulgaria. SWAP (Soil–Water–Atmosphere–Plant) model was used to simulate transport of water in the soil while ANIMO (Agricultural–Nutrient–Model) has been used to model nitrogen movement and transformation (SWAP is not intended to consider nutrients dynamics). The performed literature survey evidenced that although SWAP and ANIMO were assessed and implemented many times in the Netherlands (Rijtema and Kroes, 1991, Van Dam et al., 1997, Kroes and Roelsma, 1998, Hendriks et al., 1991 and Hack-ten Broeke, 2001; etc.) or out side (Trabada-Crende and Vinten, 1998, Wu and McGechan, 1998, Sarwar et al., 2001, Vinten and Dunn, 2001, Gusev and <Nasonova, 2003 and Eitzinger et al., 2004; etc.), actually there is no information about application of these models in countries of Balkan Peninsula region and especially in Bulgaria. Hence, any attempt to examine SWAP and ANIMO, which are originally intended to Dutch circumstances, into moderate continental climate or for different soil types and agriculture practice hopefully will be interesting for both the scientific community and agricultural practice. Furthermore, this work intends to extend previous efforts for applications of different models for assessment of the impact of landfill on groundwater quality.
The major sources of nitrogen in the Nottingham area are mains leakage and contaminated land with approximately 38% each of a total load of 21 kg N ha\(^{-1}\) year\(^{-1}\) (Fernando et. al, 2005).

**2.4 SOLID WASTE DISPOSAL**

2.4.1 Landfill

The leachate from Al-Sulaibiyah landfill site in Kuwait was found with higher concentration of Ni and V, this indicated the disposal of predominately petroleum related waste at the site. Muttamara and Leong (1997) have studied impact assessment of solid waste disposal site of the On-Nooch site in Bangkok, Thailand. They have reported higher concentration of heavy metals than the allowable level in the wastewater discharged the leachates treatment plant (Cr 1.03, Mn 107, Hg 0.025 mg/L). Researcher around the world reported importance of lining system in the landfill for contamination of ground water and surface water.

The level of Methane and \(\text{CO}_2\) in ambient air of the study area were found to be 3.48-65.71 and 886-1758 mg/m\(^3\) respectively which is much higher then the normal concentration of \(\text{CH}_4\) and \(\text{CO}_2\) in the air (\(\text{CH}_4: 2.41 \text{ mg/m}^3\), \(\text{CO}_2: 585 \text{ mg/m}^3\)). The study is performed a field trial concerning the establishment of plants cover on deposit of wastes from the Ammeberg zinc mines in Sweden, reported contamination of groundwater from zinc mine. The maximum concentration of Zn was found in root and concentration in shoot was about 10% of the soil. Zn concentration decreased with increasing biomass above ground.

2.4.2 Waste Water

Industries utilize large quantities of water for their processes and generate wastewater as effluent containing toxic elements, hazardous substances and various chemicals. The intrinsic value of waste is not known fully until now. There is need for environmentally safe disposal and minimization of wastes generation. The most of industries avoid the proper treatment of their waste to save money in cost of other living being
2.4.3 Characterization of Wastewater

Shivkumar et al. (1996) found abundance of elements such as Cu, As, Se, F, Fe were 5 to 10 times more than the permissible limits at Patencheru industrial area, Medak district in Andhra Pradesh, India. The average concentration of variables in wastewater were found as pH 7.72, TDS 164873, F 8.503, Cr 97.09, Mn 1122.64, Fe 356.36, Cu 6272, Zinc 1259, As 934.09, Se 216, Sr 154, Cd 52, Pb 36 mg/L.

Crockett et al. (1997), studied wastewater released by McMurdo station, which is the largest research station in Antartica. The wastewater from the station No.1- the macerated wastewater, mixed with brine and than discharged to Mc Murdo Sound. The organic contents were low but in heavy metals, especially Cu was found considerably high.

2.5 LITERATURE REVIEW ON SIMULATION OF SOLUTE TRANSPORT

The study about contaminant mass transport from point source like landfill is most burning topic of the current research. As above literature review indicates that so many researchers reported contamination of groundwater by leaching of pollutant from landfill but very few have been tried to understand the mechanism of movement of contaminant from point source like landfill. Since Bhalswa, Okhala and Indraprastha Landfill geology is mainly alluvium make it vulnerable for contamination by leaching of pollutant. Multi-component reactive transport models and variable density flow models are important tools for improving the understanding of governing processes in groundwater systems. Representatives of multi-component reactive transport models are e.g. MIN3P, GIMRT/CRUNCH, PHREEQC-II (Parkhurst et al., 1980) and, PHAST, HydroBioGeoChem and some MODFLOW/MT3DMS based models such as RTR3D and PHT3D.

Numerous other models were developed for transport of a single solute species in aquifers with variable density, e.g. SUTRA, METROPOL, FEMWATER (Lin et al. 1996). Pinder and Cooper (1970) provided a numerical solution for calculating the Transient position of saline waterfront in a confined aquifer. Lee and Chang (1974)
carried out finite element analysis of the same. Two major approaches are generally used to model such coupled processes, namely the one-step or global implicit approach, and the sequential iteration or split-operator technique. The simulation of the differential downward movement of a lithium and a bromide plume at Cape Cod required multi-component transport simulations that considered the variable density of the plume(s) and lithium sorption. Furthermore, the interactions between reactive processes and density variations (i) for a controlled seawater intrusion experiment, where seawater was forced inland by pumping, thereby undergoing reactions such as Na/Ca exchange, calcite dissolution–precipitation, sulphates reduction and FeS precipitation, and (ii) for a landfill leachates plume, where the density influenced the distribution of redox-sensitive species and buffering reactions by Fe and Mn hydroxides.

2.6 LITERATURE ON MSW LANDFILL

Any waste produced from a domestic source, represents over two-thirds of the municipal solid waste (MSW) stream (OCED, 2001). Internationally, almost 70% of MSW is disposed of to landfill (OECD, 2001 and Zacarias-Farah and Geyer-Allely, 2003). MSW contains hazardous substances in the form of paints, vehicle maintenance products, mercury-containing waste, pharmaceuticals, batteries and many other diffuse products, which are discussed in the review paper by Slack et.al. (2004). Unlike the waste streams originating from industrial sources, hazardous substances in household waste are not strictly controlled under hazardous waste regulations such as the US Resource Conservation and Recovery Act 1976 (RCRA) and the European Hazardous Waste Directive 91/689/EEC (US Code, 1976 and European Council, 1991). As such, household hazardous waste (HHW) is disposed of to landfill along with general household waste. The amounts and significance of this disposal are poorly understood. Generally, it is assumed that amounts are small and therefore risks of disposal are negligible. Nevertheless, disposal information is lacking or, at best, unreliable and ambiguous. Changes to legislation requiring the separate disposal of MSW, industrial and other wastes, raises the importance of the hazardous element contained in MSW. Previous studies have found that, even without landfill co-disposal, leachates from
MSW are very similar in composition to those from mixed or hazardous landfills (Schrab et.al. 1993 and Kjeldsen et.al. 2002).

Emissions from landfill take a number of forms: gaseous emissions of volatile organic compounds (VOCs), airborne particulate matter and leachates. The contamination of groundwater by landfill leachates has been recognized by a number of researchers (Christensen et.al. 2001 and Kjeldsen et.al. 2002). Leakage potential may be mitigated by a number of factors, many enshrined in legislation, including landfill capping. In Europe, the recent Landfill Directive has further enforced the treatment of emissions from landfill sites (European Council, 1999). The European Council Decision setting out the criteria and procedures for waste acceptance at landfills (European Council 2002) utilizes the leaching limit values for the three-landfill types listed in the Landfill Directive (European Council, 1999). Only MSW and separately collected non-hazardous fractions of household wastes, according to Chapter 20 of the EWC, can be considered non-hazardous without testing.

Failure of any of the engineered control measures can result in the release of a cocktail of chemicals, as reported by Schwarzbauer et.al. (2002). For older landfill, the implementation of measures to prevent release to the environment is less well defined with the result that aquifer contamination was far more common as were elevated levels of localized VOCs (Reinhard et.al., 1984). Leachates treatment and discharge also possesses risks to the environment through ineffective treatment and/or discharge to particularly sensitive receiving waters (Silva et.al. 2004).

While leachates contamination of the groundwater environment is less likely from modern landfills as a consequence of engineered barriers and leachates collection, the risk still exists. Knowledge of leachates composition is necessary for the implementation of site remediation following barrier breakdown and for installation of practicable treatment processes. Although major components of landfill leachates, especially ammonical nitrogen, can be predicted with some certainty using models to predict the possible typical leachates resulting from the deposition of generic waste
groups, the trace composition of leachates is inherently variable due to the heterogeneity of specific waste composition and other factors relating to the landfill (Reinhart, 1993, Robinson, 1995 and Blight et al. 199). Leachates composition is also an indication of the types of waste disposed and the processes occurring within the landfill. The presence in leachates of heavy metals and hazardous organic contaminants, such as halogenated aliphatic compounds, aromatic hydrocarbons, phenolic compounds and pesticides are direct indicators of the disposal of hazardous wastes in MSW (Christensen et al., 2001, Kjeldsen et al., 2002 and Isidori et al., 2003). However, care must be taken with MSW leachates analyses that reveal the presence of harmful substances due to the co-disposal of industrial liquid wastes and manufacturing wastes with MSW. The co-disposal of hazardous and non-hazardous wastes is a practice soon to be banned in EU Member.

As concern about chemicals in household products increases (Blundell, 2003), the potential consequences to the environment from the disposal of HHW are also moving to the fore. It is therefore important to ascertain the level of risk inherent in the disposal of HHW to landfill, as permitted by current legislation.

A review of leachates analyses together with an assessment of landfill biogeochemistry permits extrapolation from leachates contaminant back to HHW disposal, with leachates composition potentially acting as an indicator for landfill discarded HHW. The contamination risks associated with the disposal of HHW at each stage of the disposal-to-landfill-to-emissions pathway have not been examined in detail before.

2.7 LANDFILL LEACHATE COMPOSITION

Leachates contain inorganic and organic elements. Xenobiotic organic compounds (XOCs) and heavy metals are generally classified as the hazardous substances occurring in leachates. Hazardous XOCs and heavy metals can be toxic, corrosive, flammable, reactive, carcinogenic, teratogenic, mutagenic and eco-toxic, among other hazards, and can also be bio-accumulative and/or persistent. MSW landfill leachates analyses permit identification of the commonly found XOCs and heavy metals derived from waste with
a domestic origin. Extrapolation back to the original source is often possible, allowing the risks of discarding certain wastes to landfill to be assessed. Environmental and human health risks arise from exposure to hazardous substances in groundwater, surface water, gaseous emissions and dust evolution as a result of leakage through the barriers of new-engineered landfills, dispersal from older landfills without barriers or as a result of ineffective leachates or gas treatment prior to discharge.

However, the types of waste discarded do not solely determine leachates composition. Conditions existing within the waste body also contribute to the type of landfill emissions. Chemical and biological transformations of the waste and interactions of plant-derived matter and XOCs/heavy metals, influenced by the various redox phases undergone through the life of a landfill, affect emissions at any point in time (Robinson and Gronow, 1993). To fully assess the risk of landfill disposal of HHW, the life span of the landfill requires continual monitoring and leachates analyses. If the phases are well understood, as indicated by the extensive literature concerning conditions within the landfill, then it is possible to map the degradation of particular waste streams and the possible emissions that result.

XOCs derive directly from the waste discarded to landfills. However, conditions within the landfill determine the final composition of leachates. Bio-decomposition of putrescible waste increases general levels of dissolved organic carbon (DOC) (Reinhard et al, 1984). With due higher levels of DOC, the greater the elution through sorption to organic matter of hydrophobic XOC pollutants such as phthalate esters (Bauer and Herman, 1997, Bauer et al, 1998 and Oman and Rosqvist, 1999). DOC concentrations also affect the mobility of metals (Van der Sloot, 1998 and Meima et al, 1999).

Ammonical nitrogen is produced in much higher concentrations than XOCs with levels consistent across the different landfill phases (Robinson and Gronow, 1993 and Christensen et al, 2001). While, it is not classified as hazardous, ammonical nitrogen has the potential to act as one of the dominant environmental pollutants produced from landfills containing putrescible wastes and hence poses problems for the management
of all landfills. It has frequently been described as the most frequent pollutant of groundwater, emanating from landfills at concentrations of greater magnitude than other emissions (Christensen et al. 2001, Barlez et al., 2002 and Baker and Curry, 2004).

Biological or chemical transformations in the solid phase or leachates can lead to the formation of toxic substances from relatively innocuous organic compounds. 1,4-dioxane a controlled substance could result from such a transformation, whilst carbon tetrachloride, another toxic compound, is a principal constituent of PVC (Health and Safety Commission, 2002).

Results from various studies reveal different degrees of degradation for a variety of XOCs under the various redox conditions found within and adjacent to the landfill, as reviewed by Kjeldsen et al. (2002). As anaerobic conditions persist in capped landfills, any leachates plume progresses from strongly reducing methanogenic conditions in the landfill through the progressively weaker reducing redox zones marked by sulphate, iron and nitrate reduction to aerobic conditions in the aquifer (Reinhard et al, 1984 and Read et al., 2001).

The heavy metal content of leachates shows a reduction from acid phase to methanogenic phase due in part to increased sorption to DOC and metal precipitation with sulphate and carbonates (Robinson et al, 1993, Christensen et al, 2001 and Kjeldsen et al. 2002). Work by Suna Erases and Onay (2003) found that 90% of all heavy metals present in a landfill simulation were attenuated within 10 days of methanogenic onset through heavy metal precipitation. The application of heavy metal balances to landfills has revealed that less than 0.02% of heavy metals disposed of to landfill are leached within the first 30 years of the lifespan of the landfill due to metal immobilization by organic/inorganic sorption and precipitation (Belvi et al, 1989 and Flyhammer, 1995). For instance, mercury in landfills is predominately found as an insoluble sulphide precipitate and is therefore resistant to leaching. However, it is reported that the anaerobic conditions of modern landfills may encourage the
biomethylated transformation of mercury into more soluble and volatile methyl mercury (Jones et.al, 1978 and Compeau et.al, 1985), with the result that (Lindberg et.al. 2001) argue that landfills may be a dominant source of atmospheric mercury. However, the Environment Agency for England and Wales reported that volatile mercury has not been detected above toxicity thresholds in gaseous emissions, determining loss from landfills predominantly through adsorption to particulate matter released as dust (Parker et.al, 2002). Transformations and degradation of the waste types disposed to landfills can cause landfill-discarded waste to become more-or-less hazardous. Landfill processes should therefore be considered when assessing leachates composition, described in the following section.

2.7.1 Organic compounds

More than 200 organic compounds have been identified in municipal landfill leachate (Yasuhara et.al., 1997, Paxeus, 2000 and Schwarzbauer et.al., 2002), with upwards of 35 compounds having the potential to cause harm to the environment and human health (Paxeus, 2000). More than 1000 chemicals have been identified in groundwater contaminated by landfills generally (Christensen et.al. 2001 and Kjeldsen et.al. 2002). The majority of these compounds are derived from decomposing vegetation and degradation products of natural materials (Reinhard et.al., 1984 and Schwarzbauer et al, 2002); with cellulose and hemi-cellulose alone comprising up to 60% of the total dry weight of MSW (Barlaz et al 1989). Such compounds, aliphatic and aromatic acids, phenols and terpenes, have a tendency to degrade as any leachates plume migrates from a site (Reinhard et.al., 1984 and Leenheer et.al., 2003). Ammonical nitrogen and dissolved organic matter, as total organic carbon (TOC), biochemical oxygen demand (BOD) and chemical oxygen demand (COD), encompass the primary organic degradation products (Robinson et.al, 1993, Christensen et.al., 2001 and Kjeldsen et.al.2002).

Landfill types: (1) co-disposal landfills cited by (Kjeldsen et.al.,2002); (2) MSW landfills although co-disposal likely in all (Robinson et.al, 1993); (3) co-disposal landfills (Reinhard et.al., 1984) (4) simulation using household waste (Oman and Rosquivst, 1999) (5) co-disposal landfill (Christensen et.al., 2001); (6) no details
supplied (Schwarzbauer et al., 2002); (7) co-disposal landfill (Yasuhara et al., 1997); (8) co-disposal landfill (Zou et al., 2003); (9) mixed landfills receiving different wastes (Jonsson et al., 2003); (10) simulation using household waste (Bauer and Herrman, 1997); (11) MSW landfill (Leenher et al., 2003); (12) co-disposal landfill (Ask Reitzel et al., 2002); (13) co-disposal landfill (Baun et al., 2003); (14) co-disposal landfill (Lyngkilde and Christensen, 1992); (15) co-disposal landfill (Paxeus, 2000).

The occurrence of plasticizers in leachate has been widely reported (Bauer and Herman, 1997, Bauer et al., 1998, Mersiowsky et al., 2001, Yamamoto et al., 2001, Mersiowsky, 2002, Jonsson et al. 2003 and Marttinen et al., 2003). Principles among this group of compounds are phthalates, of which di-(2-ethylhexyl) phthalate (DEHP), diethyl phthalate (DEP), diisononyl phthalate (DINP) and dibutyl phthalate (DBP) have been extensively used in the manufacture of consumer goods (Mersiowsky, 2002 and Schwarzbauer et al., 2002).

The occurrence of pharmaceutical products in landfill leachate has also been reported, with propyphenazone, ibuprofen and clofibic acid identified in leachate leaking through the faulty bottom seal of a domestic landfill in Germany (Schwarzbauer et al., 2002). Holm et al., (1995) describe the rapid methanogenic degradation of a group of pharmaceuticals in groundwater contaminated by landfill leachate including propyphenazone, sulphonamides and barbiturates. Phenazone, an analgesic similar to propyphenazone, was identified in soil and groundwater below a MSW landfill by Ahel and Jelicic (2001). Eckel et al. (1993), further identified pentobarbital in groundwater from a landfill. Work by Schecker et al. (1998), investigated the elimination of a cytostatic drug falling under the auspices of the European Waste Catalogue (European Commission, 2000), from a sanitary landfill. Approaching 50% of the ifosfamide added to a methanogenic landfill was eliminated after 120 days. More general assessments of groundwater contamination not necessarily linked to landfill seepage have identified a greater variety of products (Jones et al., 2001) such as the analgesics diclofenac and ketoprofen (Herbere et al., 1997 and Sacher et al., 2001); antibiotics sulfamethoxazole, dehydroerythromycin and sulfamethazine (Hartig et al., 1999 and Hirsch et al., 1999)
and the antiepileptic drug carbamazepine (Seiler et al., 1999 and Schwarzbauer et al., 2002) and the β-blocker sotalol (Sacher et al., 2001).

Pesticides and herbicides are frequently recorded in MSW landfill leachate. N,N-Diethyltoluamide (DEET), bentazon, MCPA and particularly mecoprop (2-(4-chloro-2-methylphenoxy) propionic acid) are common and persistent in anaerobic landfill conditions (Schultz and Kjeldsen, 1986, Gintautas et al., 1992, Lyngkilde and Christensen, 1992, Kjeldsen, 1993, Oman and Hynning, 1993, Christensen et al., 2001 and Kjeldsen et al., 2002). Christensen et al., (2001) and Kjeldsen et al., (2002) cite the discovery of at least a further 40 different pesticides in landfill leachate, including the Red Listed aquatic pollutants atrazine and simazine (Alloway and Ayres, 1997). Detergent surfactants are reported from leachates analyses, particularly sulphonates and alkylphenol polyethoxylates (Riediker et al., 2000 and Kjeldsen et al., 2002). Such surfactants occur in laundry detergents and personal care products, although the latter surfactant, particularly as nonylphenol ethoxylates, is not commonly used due to concerns over environmental and health problems, especially endocrine disruption (Jobling and Sumpter, 1991). Nevertheless, nonylphenol has been reported to be present in municipal landfill leachates in some countries (Paxeus, 2000, Behnisch et al., 2001 and Coors et al., 2003). Synthetic musks, used to fragrance detergents and personal care products, are reported to possess toxic capabilities and, although not yet monitored in leachate, have been recorded in sewage receiving waters in a number of countries, but first found in Japan Yamagishi et al., 1981, Yamagishi et al., 1983, Ford, 1991 and Daughton and Ternes, 1999).

Although concentrations of XOCs are only a small fraction of the total carbon content of leachates, the levels observed in a multitude of studies indicate that concern for aquifer contamination is justified (Albaiges et al., 1986, Lyngkilde and Christensen, 1992, Kjeldsen, 1993, Kjeldsen et al., 1998). With water quality levels in many countries set as low as 0.1 µg/l for certain XOCs, the concentrations achieved in leachates may therefore be perceived as a potential threat to public health if incorrectly managed.
2.7.2 Inorganic components

The study of inorganic element of leachates is most important because of its toxic nature (Yasuhara et al., 1997). Heavy metals found in the inorganic fraction are of particular interest due to their hazardous nature (European Commission, 2000). Commonly occurring heavy metals in landfill leachates include zinc, copper, cadmium, lead, nickel, chromium and mercury (Reinhart, 1993). Heavy metals can form metal colloids or complexes, particularly with organic matter, removing the metal from direct detection as free metals (Gounaries et al., 1993, Christensen and Christensen, 1999 and Jensen and Christensen, 1999). As such, the heavy metal content of leachates can be significantly higher than free metal detection studies allow, and calculations based solely on the water solubility constants of the pollutants will underestimate their concentrations (Jonsson et al. 2003).

Generally heavy metals, demonstrating high levels of sorption and precipitation, do not constitute a groundwater pollution threat due to poor migration into the leachates plume and low initial concentrations leached from the solid waste (table 2.1). Heavy metals can, however, reach problematic levels despite depressed concentrations relative to other substances evolved into the leachates, Ehrig (1983), Kjeldsen et al. (2002) and Robinson (1995) found that heavy metal levels, particularly mercury and cadmium, in domestic waste landfill leachates are barely detectable and pose little threat to groundwater. Zinc, however, is usually recorded at concentrations orders of magnitude greater than other heavy metals (Christensen et al. 2001). Heavy metal contamination of aquifers is almost exclusively as soluble high molecular weight organic complexes. Such contamination of groundwater can be of environmental and potable drinking water concern, as the drinking water standards provided in table demonstrate (Reinhard et al., 1984). This concern is heightened by the identification of concentrations of heavy metals exceeding legislative permits in leachates sampled in the USA and Eastern Europe, including drinking water limits shown in table (Lu et al., 1985 and Mikac et al., 1998). Even with the maximum permissible levels for heavy metals as high as 5 mg/l (US Drinking Water Standard for zinc), weak leachates concentrations can often approach this limit (Kjeldsen et al., 2002). Nevertheless, particulate matter contaminated
with heavy metals has been cited as one of the primary sources of heavy metal emissions from landfills (Parker et al., 2002).

Other contaminants of landfill leachates can be referred to as inorganic macro-components (Christensen et al., 2001). As occurs for dissolved organic matter and the less concentrated subgroup xenobiotic organic compounds, inorganic macro-components occur at much higher concentrations compared to heavy metals. Iron and manganese fall into this group, as they cannot be considered to be heavy metals, along with calcium, sodium, potassium, ammonia/ammonium and others. Ammonical nitrogen (ammonia and/or ammonium) is recorded at high levels in most landfill leachates studies with Robinson (1995) and Kjeldsen et al. (2002), describing, it as the dominant pollutant.

Table 2.1 Heavy metal concentration ranges detected in landfill leachates

<table>
<thead>
<tr>
<th>Metal</th>
<th>Use</th>
<th>Conc. range (mg/l)</th>
<th>US drinking water standards (mg/l)</th>
<th>UK drinking water standards (mg/l)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cadmium</td>
<td>Batteries, appliances</td>
<td>0.0001–0.4</td>
<td>0.005</td>
<td>0.005</td>
<td>Christensen et. al., 2001</td>
</tr>
<tr>
<td>Nickel</td>
<td>Batteries, appliances</td>
<td>0.0036–13</td>
<td>N/A</td>
<td>0.05</td>
<td>Christensen et. al., 2001 and Kruempelbeck and Ehrig, 1999</td>
</tr>
<tr>
<td>Zinc</td>
<td>Batteries, packaging</td>
<td>0.003–1000</td>
<td>5</td>
<td>5</td>
<td>Christensen et. al., 2001</td>
</tr>
<tr>
<td>Copper</td>
<td>Electrical appliances</td>
<td>0.002–10</td>
<td>1.3</td>
<td>3</td>
<td>Christensen et. al., 2001 and Jensen and Christensen, 1999</td>
</tr>
<tr>
<td>Lead</td>
<td>Batteries, appliances</td>
<td>0.001–5</td>
<td>0</td>
<td>0.05</td>
<td>Christensen et. al., 2001</td>
</tr>
<tr>
<td>Chromium</td>
<td>Electrical appliances</td>
<td>0–1.62</td>
<td>0.1</td>
<td>0.05</td>
<td>Jensen and Christensen, 1999 and Robinson, 1995</td>
</tr>
<tr>
<td>Mercury</td>
<td>Batteries, appliances</td>
<td>0.00005–0.16</td>
<td>0.002</td>
<td>0.001</td>
<td>Christensen et. al., 2001</td>
</tr>
<tr>
<td>Arsenic</td>
<td>Appliances</td>
<td>0.01–1</td>
<td>0</td>
<td>0.05</td>
<td>Christensen et. al., 2001</td>
</tr>
<tr>
<td>Cobalt</td>
<td>Appliances</td>
<td>0.005–1.5</td>
<td>N/A</td>
<td>N/A</td>
<td>Christensen et. al., 2001</td>
</tr>
</tbody>
</table>