12   Waste disposal and landfill: Information needs

Waste can be loosely defined as any material that is considered to be of no further use to the owner and is, hence, discarded. However, most discarded waste can be reused or recycled, one of the principles of most waste management philosophies. What may be of no further use to one person and regarded as waste to be dumped, may be of use to the next person, and is the basis of the rag picking trade, the sifting through of refuse at landfills for recovery and resale, a very fundamental historical waste management practice still functioning in many countries, often conducted on a highly organised commercial basis.

Waste is generated universally and is a direct consequence of all human activities. Wastes are generally classified into solid, liquid and gaseous. Gaseous waste is normally vented to the atmosphere, either with or without treatment depending on composition and the specific regulations of the country involved. Liquid wastes are commonly discharged into sewers or rivers, which in many countries is subject to legislation governing treatment before discharge. In many parts of the world such legislation either does not exist or is not sufficiently implemented, and liquid wastes are discharged into water bodies or allowed to infiltrate into the ground. Indiscriminant disposal of liquid wastes pose a major pollution threat to both surface and groundwater. Potential groundwater contamination by liquid waste from households is covered in Chapter 10 and from industry in Chapter 11.

Solid wastes, the subject of this chapter, are mainly disposed of to landfill, because landfill is the simplest, cheapest and most cost-effective method of disposing of waste (Barrett and Lawler, 1995). In most low- to medium-income developing nations, almost 100 per cent of generated waste goes to landfill. Even in many developed countries, most solid waste is landfilled. For instance, within the European Union, although policies of reduction, reuse, and diversion from landfill are strongly promoted, more than half of the member states still send in excess of 75 per cent of their waste to landfill (e.g. Ireland 92 per cent), and in 1999 landfill was still by far the main waste disposal option for Western Europe (EEA, 2003). Furthermore, although the proportion of waste to landfill may in future decrease, the total volumes of municipal solid waste (MSW) being produced are still increasing significantly, in excess of 3 per cent per annum for many developed nations (Douglas, 1992). Landfill is therefore likely to remain a relevant source of groundwater contamination for the foreseeable future (Allen, 2001).

Solid waste composition, rate of generation and methods of treatment and disposal vary considerably throughout the world and largely determine the potential of waste to impair groundwater quality. The purpose of this chapter is to outline the risk that waste disposal presents to groundwater quality and the information that is required to assess this risk.

12.1   Types of solid waste

Wastes generated by the full extent of human activities range from relatively innocuous substances such as food and paper waste to toxic substances such as paint, batteries, asbestos, healthcare waste, sewage sludge derived from wastewater treatment and as an extreme example, high-level (radioactive) waste in the form of spent nuclear fuel rods. Numerous classifications of solid wastes have been proposed (e.g. Tchobanoglous *et al*., 1993; Ali *et al*.,

1999), and the following represents a simple classification of waste into broad categories according to its origin and risk to human and environmental health:

- household waste;
- municipal waste (MSW);
- commercial and non-hazardous industrial wastes;
- hazardous (toxic) industrial wastes;
- construction and demolition (C&D) waste;
- health care wastes – waste generated in health care facilities (e.g. hospitals, medical research facilities);
- human and animal wastes; and
- incinerator wastes.

Household waste represents waste generated in the home and collected by municipal waste collection services. Municipal solid waste (MSW) includes this plus shop and office waste, food waste from restaurants, etc., also collected by municipal waste collection systems, plus waste derived from street cleaning, and green (organic) waste generated in parks and gardens.

Storage of waste in a disposal facility serves to minimise the effects of waste on the environment. This is achieved by restricting any effluent derived from the waste to a single location, where emissions can be controlled. If control is lacking or inadequate, disposal facilities may become point sources of groundwater contamination. In many regions, centralised waste disposal has historically occurred by landfilling, wherein local quarries and gravel pits have been filled with waste because, in many cases, they simply constituted an appropriately sized hole in the ground. Such locations typically offered little protection against contamination of adjacent groundwater supplies. Legislation, designed to protect usable groundwater, has helped to reduce the incidence of this practice in many high to middle income countries (e.g. US EPA, 1974; CEC, 1980; NRA, 1992). Modern waste management practices involve disposal of waste in specially sited and engineered sites known as "sanitary landfills" (see Chapter 24).

Waste accepted in municipal waste landfills in developed countries would normally consist of municipal solid wastes, plus commercial and non-hazardous industrial wastes, and construction and demolition (C&D) waste. There is a tendency in many countries for C&D waste, usually regarded as inert, to be buried on the construction site where it is generated. However, since downward percolating rainwater may leach heavy metals from C&D waste, recent waste regulations in some developed countries requires all C&D waste to be disposed of in landfills.

Hazardous and non-hazardous wastes are differentiated in waste management legislation of many countries. A range of legal definitions exist for hazardous waste, but it can generally be thought of as waste or a combination of wastes with the capacity to impair human health or the environment due to its quantity, concentration, or physical, chemical or infectious characteristics when improperly used, treated, stored, transported or disposed. In many countries, hazardous (toxic) industrial wastes (both organic and inorganic), solid incinerator residues, bottom and fly ash are disposed in special hazardous waste landfills, and specialized disposal or incineration may also be practiced for healthcare wastes (see Box 12.1). In many

low- to medium-income parts of the world, where uncontrolled open dumps are common, all waste tends to be dumped together, regardless of its origins or its hazardous nature. A specific characteristic of leachate from hazardous industrial waste is that it may be toxic to the bacteria naturally present and thus delay biodegradation of organic substances in leachate.

Box 12.1. Health care and research facilities.

Health-care facilities can contaminate groundwater through wastes and wastewater containing infectious pathogens and pharmaceuticals. Health-care facilities may also release various diagnostics (e.g. radiochemicals, contrast media) and disinfectants depending on the kinds of medical examination and treatment being conducted and local practices for handling these substances. These include, but are not restricted to the following classes:

- cytostatic agents applied in cancer therapy;

- antibiotics;

- disinfectants for surface, instrument and skin disinfection;

- heavy metals such as platinum from excretion by patients treated with the cytostatic agents, mercury from preservatives, disinfectants, diuretic agents, amalgam separators;

- adsorbable organic halogenes (AOX) from solvents, disinfectants, cleaners and drugs containing chlorine, as well as iodised X-ray contrast media.

Research institutions may also employ solvents, other chemicals and radiochemicals, some of which are very hazardous to human health (e.g. mutagenic substances used in molecular biology). In addition, organisms used in production and/or research, especially pathogenic bacteria and viruses as well as genetically modified organisms are utilised.

For health-care and research facilities, situation assessment should be based on an inventory of substances used or produced, and of processes, which have the potential to emit pathogens or chemicals. Such assessments should necessarily cover storage (containment), handling and disposal practices (e.g. disinfection of wastes) as well as patients' excretion rates and concentration in the effluent. Additionally, assessment should address how effectively these practices are being implemented and the ultimate destination of disposal (e.g. local dump, sewage mains), as this will determine the nature and magnitude of the risk to groundwater.

Human and animal wastes are usually not disposed of in landfills, although animal carcasses and waste from abattoirs may in some countries be disposed of in dumps and landfills. Human corpses are not generally regarded as waste, but they degrade in a similar way to other organic waste, and also produce leachate in significant quantities. The majority of corpses are buried in cemeteries (see Box 12.2), although a significant proportion are cremated (incinerated), the proportion varying from country to country depending on the proportions of different religious groups in the population and their funeral rites. The main health concern with human and animal wastes is the high concentrations of pathogenic organisms associated with this type of waste, and the potential it has to spread disease.

Box 12.2. Cemeteries.

According to the most recent global statistics, the death rate from all causes is 9 per 1000 of the population. In many regions burials are concentrated into relatively small areas, such as

municipal cemeteries, where each body introduces a heavy burden of organic, inorganic and biological parameters into the subsurface. Hydrogeological factors have historically not been taken into account when locating cemeteries and the potential impact of cemeteries on groundwater quality has not been considered.

Animal and human remains, although not considered a 'waste product', represent a risk to the quality of local groundwater because of the proliferation of microorganisms that occurs during the process of corpse decomposition (Pacheco *et al.*, 1991). The number of bacteria in a human body is greater than the number of human cells. Many of the bacterial cells are harmless saprophytes that benefit the host (e.g. by synthesising vitamins or by metabolising toxic waste products). However, some of the species will be pathogenic, or have the potential to be pathogenic. In addition, the human body is host to a variety of different viruses, fungi and protozoa that may cause disease if transmitted to a susceptible person. Most pathogens will remain viable for a period of time after the host dies; the length of time depending on the pathogen. In most cases long-term survival of the pathogen is unlikely, but notable exceptions have generated concerns amongst the general public and the scientific community during investigations of burial sites: The examination of graves containing the remains of smallpox, cholera, anthrax and plague victims, as well as victims of the 1918 influenza pandemic, have been subject to rigorous controls to prevent the potential dispersal of the pathogen from the burial site.

One of the main agents in decomposition (putrefaction) is *Clostridium perfringens*. These bacteria spread along blood vessels causing haemolysis, proteolysis and gas formation in blood and other tissues. The liquids produced through putrefaction contain a high density of microorganisms. Very few studies have been carried out on the microbiology of human putrefaction. The best known is by Corry (1978), who has published a catalogue of bacterial species that have been isolated from human cadavers; some of the species listed are pathogenic. These liquids can migrate down into the water table, particularly as coffins and caskets are not water tight and are liable to decay. Microbiological contaminants that may result from the decomposition of cadavers include *Staphylococcus spp.*, *Bacillus spp.*, *Enterobacteriacae spp.*, faecal streptococci, *Clostridium spp.*, *Helicobacter pylori*, enteroviruses, rotavirus, calicivirus, and F-specific RNA phage.

Spongberg and Becks (2000) list additional potential chemicals that can be released from cemeteries. These include arsenic and mercury (embalming and burial practices), formaldehyde (embalming, varnishes, sealers and preservatives) as well as lead, zinc and copper (coffins). Spongberg and Becks (2000) also discuss investigations in Ohio where increases in zinc, copper and lead in the soil at a large cemetery were observed. Significant increases in arsenic were thought to indicate contamination from embalming fluids or wood preservatives.

There are several historical accounts of pollution of water wells in the vicinity of cemeteries (e.g. Teale, 1881), but few recent studies of the microbiological impact of cemeteries on groundwater (West *et al.*, 1998). An analysis of groundwater quality beneath an active cemetery in the UK provided evidence that confirms the risk to groundwater, although no pathogens or viruses were isolated. The impacts on groundwater, of three cemeteries in Sao Paulo and Santos, Brazil have been monitored by Pacheco *et al.* (1991), by installing piezometers throughout each of the cemetery sites. One cemetery is situated on Tertiary sediments, 4-12 m above the water table, one is on weathered granite, 4-9 m above the water table, and the third is on Quaternary sandy marine sediments, 0.6-2.2 m above the water table. Contamination of the piezometers by faecal coliforms, faecal streptococci and sulphite reducing clostridia was found to be widespread throughout all of the cemeteries. Thus, clearly, assessing groundwater pollution potential therefore needs to include the potential for pathogens from cemeteries, particularly from large cemeteries.

The rate at which waste is generated corresponds roughly with levels of income. In high-income countries of Europe and North America between 500 and 750 kg of solid waste are produced per person per year (OECD, 1997). In contrast, urban populations in most low-income countries, for example in Nigeria and Côte d'Ivoire, generate between 100 and 200 kg of solid waste per person per year (Attahi, 1999; Onibokun and Kumuyi, 1999). Despite this lower rate, rapid urbanisation, particularly in low income developing countries has left little space for disposal of the increasing amounts of waste material being generated in urban settings (Sangodoyin, 1993). As a result, uncontrolled disposal (i.e. "fly tipping") is rife in many countries, and is a diffuse source of groundwater contamination.

12.2   Waste storage, treatment and disposal sites

The processes of storage, collection, transport, treatment and disposal of wastes all have the potential to pollute the environment and particularly groundwater due to uncontrolled migration of fluids (leachate) derived from the wastes. In addition to the potential for groundwater pollution at sites where wastes are produced and stored prior to collection, sites associated with the treatment and disposal of wastes, where leachate may be generated include:

- landfills (both controlled as sanitary landfill or uncontrolled as open dumps);
- scrap-yards;
- cemeteries;
- waste collection and processing facilities; and
- composting facilities.

For situation assessment, landfills are most readily identified with the pollution of groundwater by waste-derived liquids. However, any site where waste is concentrated, processed (e.g. recycled) and stored even for a short period of time, may be a potential point source of groundwater contamination. Such processing facilities are often not well regulated or licensed and frequently occur in urban or semi urban settings, where local water supply points may be impacted by these activities. An inventory of these locations, the types of waste handled and management processes for waste products will aid in the assessment of the polluting capability of such sites.

For situation assessment, a critical criterion in estimating potential groundwater pollution from waste disposal is the siting of all of the above mentioned waste treatment and disposal facilities, particularly sanitary landfills and open dumps (discussed in Chapters 12.3.2 and 12.3.3). Most modern landfills in high to medium income countries require licenses to operate (see Chapter 24.2), and must be engineered to prevent groundwater pollution. This generally involves lining the site with an artificial lining system, but liners leak and degrade with time (Chapter 24.3). Even if the site is well engineered and managed, with an artificial lining system installed, and even if or the waste materials are inert, leachate, which may have the potential to pollute groundwater, will be produced. It is therefore essential to assess the capacity of the underlying geology to protect groundwater in the event of liner failure. The likelihood of disposed wastes polluting groundwater depends on the thickness of the

unsaturated zone and the attenuation capacity of the overburden (i.e. any loose unconsolidated material which overlies solid bedrock) underlying the site, and also on the total and effective precipitation at the site, since the quantity and concentration of leachate generated is a function of the access of water to the waste. Thus the potential for pollution of groundwater will be least at sites carefully selected to take advantage of the most favourable geological/ hydrogeological conditions.

'Historic landfills (dumps)' generally were not subject to the regulations governing modern landfills, and were usually sited for convenience, such as the presence of a pre-existing hole into which the waste could be deposited. The general assumption that an aftercare period of 30 years is adequate to allow for degradation of waste to an inert state, is now being questioned, with recent studies (Hjelmar *et al.*, 1995; Wall and Zeiss, 1995; Kruempelbeck and Ehlrig, 1999; Röhrs *et al.*, 2000; Fourie and Morris, 2003) suggesting that waste may remain active for many decades and even hundreds of years, particularly under moisture-deficient conditions. This includes not only landfills from regions where evaporation exceeds precipitation, but also all lined and capped landfills employing the concept of dry entombment of waste.

In the past, hazardous and non-hazardous wastes were not distinguished so that hazardous substances may be stored in all of these landfills. For situation assessment, it is important to locate all waste disposal sites in the drinking-water catchment, including both currently operating landfills, and historic dumps, now closed and covered over (see Chapter 24). Assessment of all landfills, but in particular historic landfills, should include age and type of waste, underlying geology, most importantly type and thickness of overburden and thickness of the unsaturated zone. The state of degradation of the waste can be ascertained by analysing the leachate and landfill gases generated, as degradation of waste follows a distinctive pattern manifested in well-known and documented compositional variations in liquid and gaseous emissions. All of this, together with the proximity of all of these sites to sources of drinking water, can determine the threat to public health posed by waste disposal.

12.3   Factors governing contamination of groundwater by disposal of waste

Waste deposited in landfills or in refuse dumps immediately becomes part of the prevailing hydrological system. Fluids derived from rainfall, snowmelt and groundwater, together with liquids generated by the waste itself through processes of hydrolysis and solubilisation, brought about by a whole series of complex biochemical reactions during degradation of organic wastes, percolate through the deposit and mobilise other components within the waste. The resulting leachate, subsequently migrates from the landfill or dump and has the potential to contaminate local groundwater either through direct infiltration on site or by infiltration of leachate-laden runoff offsite. The risk posed to groundwater-fed drinking water sources by waste disposal in landfills or dumps can be considered in terms of three controls:

- waste composition and loading;
- leachate production; and
- leachate migration - attenuation and dilution.

12.3.1   *Waste composition and loading*

The composition and volume of disposed wastes vary nationally and regionally in relation to the local human activities, and the quantity and type of products that communities consume (Table 12.1). Discarded waste in lower income areas is typically rich in food-related waste, i.e. organic (carbon-rich) substances (Table 12.1). Although such waste is not in itself toxic, decomposition of organic matter can alter the physico-chemical quality of groundwater and enhance the mobility of hazardous chemicals including metals and solvents (see Chapter 12.3.2). The proportion of manufactured (e.g. paper) and potentially hazardous (e.g. textiles, metals, plastics) wastes increases in relation to income and degree of industrialisation (Table 12.1), and waste disposal leachate from highly industrialized settings may contain a wide range of anthropogenic contaminants (see Box 12.3). The types of hazardous substances likely to occur in discarded waste may be assessed from the types of industry, small-scale enterprise and other human activity of a particular area.

A major concern in many countries is also of waste import, particularly of hazardous wastes. Export from industrialised countries to low-income countries circumvents strict waste disposal regulations implemented in the country generating these wastes. Often this is highly organized, as informal, though illegal, transactions between an exporter and importer using false documentation (e.g. Mackenzie, 1989). Such waste export/import practices are difficult to detect, but important for situation assessment as disposal of such wastes is likely to pose a risk of groundwater contamination. It is, therefore, often necessary to collect information on both formal and informal (i.e. illegal) waste composition and loading.

Table 12.1. Solid-waste generation and composition from selected regions in the world (OECD, 1993, 1997; Attahi, 1999; Onibokun and Kumuyi, 1999; Lusugga Kironde 1999).

Location	Rate (kg/pers./year)	Composition (%)						
		Paper	Food	Plastics	Glass	Metals	Textiles	Other
China	285	3	60	4	1	0	2	-
Denmark	520	30	37	7	6	3	17	-
France	560	30	25	10	12	6	17	-
Iran	324	8	74	5	3	1	2	-
Mexico	320	14	52	4	6	3	20	-
Poland	290	10	38	10	12	8	23	-
USA	730	38	23	9	7	8	16	-
Abidjan (Côte d'Ivoire)	211	4	63	5	1	1	1	25
Ibadan (Nigeria)	153	15	43	4	6	21	1	10
Dar es Salaam (Tanzania)	142	9	60	2	1	3	1	25

Box 12.3. Organic contaminants in groundwater affected by landfill leachate in Germany.

A study investigating groundwater from 250 different municipal waste sites in Western Germany (Kerndorff *et al*., 1992) identified a wide range of organic contaminants within 10-100 m downgradient of the deposit, some of which occurred in a large number of samples and attained concentrations well into the range of mg/L (Table 12.2). Benzene and its alkyl derivatives (four compounds) constitute the majority of the seven non-halogenated contaminants. The highest mean concentration was obtained for volatile halogenous substances, predominantly for dichloromethane ($\approx$ 38 mg/L), *cis*-1,2-dichloroethene ($\approx$ 22 mg/L), vinyl chloride ($\approx$ 1.7 mg/L), and trichloroethene ($\approx$ 1 mg/L). The high concentrations associated with these volatile organic compounds confirm the significance of this class of substances as major emissions from waste disposal sites.

Table 12.2. Organic contaminants in landfill leachate in Germany.

Parameter	Frequency of detection (%)^x	Concentration (µg/L)	
		Mean	*Maximum*
Tetrachloroethene	70.4	56.1	6,500
Trichloroethene	55.6	1,010	128,000
cis-1,2-Dichloroethene	30.1	22,100	411,000
Benzene	29.1	141	1,800
1,1,1-Trichlorethane	22.8	16.5	270
m/p-Xylene	22.8	39.9	447
Trichloromethane	22.0	76.2	2,800
1,2-Dichloroethane	18.8	107	210
Chloroethene (VC)	17.7	1,690	12,000
Toluene	16.5	73.2	911
Dichloromethane	14.9	38,100	499,000
Tetrachloromethane	14.4	1.2	23
4-Methylphenol (*p*-cresol)	13.7	42.0	283
Chlorobenzene	12.9	52.9	388
2-Methylphenol (*o*-cresol)	12.9	10.0	63
1,2-Dichlorobenzene	12.2	1.4	6.6
1,4-Dichlorobenzene	12.2	31.9	265
Naphthalene	12.1	2.2	13
Ethylbenzene	11.3	32.2	160
o-Xylene	9.5	13.8	69
2,4,6-Trichlorophenol	8.9	3.2	24
3,5-Dimethylphenol	8.1	16.2	61
Phenol	8.1	2.2	5.6
1,3-Dichlorobenzene	7.8	11.5	74
trans-1,2-Dichloroethene	7.5	57.1	135
Isopropylbenzene (cumol)	5.6	2.4	4.7
1,1-Dichloroethane	5.4	52.7	110
Acenaphthene	4.8	6.3	32
2,4-Dichlorophenol	4.8	3.5	17
3-Chlorophenol	4.8	12.7	23
p-Cymol[*p*-$CH_3C_6H_4CH(CH_3)_2$]	4.4	1.9	3.5
2-Ethyltoluene	4.4	0.6	1.0
2,4,5-Trichlorophenol	3.9	7.1	31
1,3,5-Trimethylbenzene	3.3	1.7	4.0
Phenanthrene	3.2	1.5	4.4
Tribromomethane	3.1	3.0	6.0

x: Number of samples: 90 to 277

Landfilled refuse is rich in microorganisms. Mature sites may be compared to large bioreactors in which the organic content of the waste is decomposed anaerobically. Most of the organisms that carry out these processes are harmless saprophytes, but a small percentage of the population may be opportunistically pathogenic microorganisms. Waste arising from households, medical practices and hospitals, veterinary practices, industrial sites and from environmental sources will contain pathogenic microorganisms. Whereas waste from industrial, medical and veterinary sources is more likely to be controlled or to be of known composition, domestic waste tends to be highly variable and of uncontrolled composition. An analysis of household waste in the UK showed that over 4 per cent of the waste comprised disposable nappies (diapers) of which about one-third may be soiled with faeces. Domestic waste also contains bloodstained materials, such as sanitary pads, tampons and discarded wound dressings and animal wastes, such as dog faeces and soiled cat litter. The potential for pathogens within this mixture of sources is extremely high. Pathogens may also be transported to landfill sites by vermin (rats) and other scavengers, in particular seagulls.

The fate of pathogens in landfill sites is not understood. Although it is generally assumed that most are rapidly inactivated by the conditions that prevail in the landfill environment, the potential of leachate and run-off from landfill sites to transport pathogens into local water resources should be addressed in situation assessment.

12.3.2   Leachate production

Most waste deposited in landfills is not inert. Degradation of many components of waste including food, paper and textiles consumes oxygen thereby changing the redox potential of the liquid present and potentially influencing mobility of other constituents. Plastics, glass and metal compounds tend to be less reactive and degrade more slowly. Under some conditions, metals may, however, become rapidly mobilised (see Chapter 4).

Percolating rainwater provides a medium in which waste, particularly organics, can undergo degradation into simpler substances through a range of biochemical reactions involving dissolution, hydrolysis, oxidation and reduction, processes controlled to a large extent within landfills and dumps by microorganisms, primarily bacteria. Table 12.1 indicates that the largest fraction of disposed waste is organic matter (e.g. food, paper), which has a well-documented degradation path. Mechanisms regulating mass transfer from wastes to leaching water, from which leachate originates, can be divided into three groups of processes :

- hydrolysis of solid waste and biological degradation;
- solubilisation of soluble salts contained in the waste; and
- suspension of particulate matter.

The first two groups of processes, which have the greatest influence on the composition of leachate produced, are associated with the stabilisation of waste.

Initially, organic matter, in the form of proteins, carbohydrates and fats, is decomposed under aerobic conditions (i.e. oxidised), through a series of hydrolysis reactions, to form carbon dioxide and water together with nitrates and sulphates via a number of intermediate products such as amino acids, fatty acids and glycerol. Such oxidation reactions are exothermic, so temperatures in the landfill become elevated. Carbon dioxide is released as a gas or is

dissolved in water to form carbonic acid (H_2CO_3) which subsequently dissociates to yield the bicarbonate anion (HCO_3^-) at near neutral pH.

Aerobic decomposition of organic matter depletes the waste deposit of oxygen (O_2) as buried waste in the landfill or refuse dump becomes compacted and circulation of air is inhibited. As oxygen becomes depleted, it is replaced as the oxidising agent by in succession, nitrate (NO_3^-), manganese (as MnO_2), iron (as $Fe(OH)_3$) and sulphate (SO_4^{2-}). In general, the aerobic stage is short, no substantial volumes of leachate are produced, and aerobic conditions are rapidly replaced by anaerobic conditions. The main stages of anaerobic digestion are (i) acetogenic (acid) fermentation, (ii) intermediate anaerobiosis, and (iii) methanogenic fermentation, all three of which can be operating simultaneously in different parts of the landfill.

Acetogenic fermentation brings about a decrease in leachate pH, high concentrations of volatile acids and considerable concentrations of inorganic ions (e.g. Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+). As the redox potential drops, sulphate is slowly reduced, generating sulphides, which may precipitate iron, manganese and heavy metals that are dissolved by the acid fermentation. Decrease in pH is due to production of volatile fatty acids (VFAs) and to high partial pressures of carbon dioxide (CO_2), whilst the increased concentrations of anions and cations results from leaching (lixiviation) of easily soluble organic material present in the waste mass. Breakdown of organic material reduces the redox potential to < 330mV, which allows the next stage of the process to become initiated. Leachate from this phase is characterised by high values of biochemical oxygen demand (BOD, commonly > 10,000 mg/L), high BOD_5/COD (chemical oxygen demand) ratios (commonly > 0.7), acidic pH values (typically 5-6) and ammonia (NH_3) due to hydrolysis, and fermentation in particular of proteins.

Intermediate anaerobiosis commences with a gradual increase in the methane (CH_4) concentration in the gas, coupled with a decrease in H_2, CO_2 and volatile fatty acids. Conversion of the volatile fatty acids leads to an increase in pH values and to alkalinity, with a consequent decrease in the solubility of calcium, iron manganese and the heavy metals, which are probably precipitated as sulphides. Ammonia is released but is not converted to nitrate in such an anaerobic environment.

Methanogenic fermentation, the final stage in the degradation of organic wastes, operates within the extremely limited pH range of 6-8. At this stage in the degradation process, the composition of leachate is characterised by almost neutral pH, and low concentrations of volatile acids and total dissolved solids (TDS), indicating that solubilisation of the majority of organic components is almost complete, although waste stabilisation will continue for several decades. The biogas being produced has a methane content of generally > 50 per cent, whilst ammonia continues to be released by the acetogenic process. Leachate produced at this stage is characterised by relatively low BOD values, and low ratios of BOD/COD.

Degradation processes convert nitrogen into a reduced form (ammonium), and bring about mobilisation of manganese and iron and also liberation of hydrogen sulphide gas. Production of methane indicates strongly reducing conditions with a redox potential in the order – 400 mV. Unlike carbon dioxide, methane is poorly soluble in water.

Due to the decomposition of organic matter, leachate derived from landfills or dumps comprises primarily dissolved organic carbon (Table 12.3), largely in the form of fulvic acids (Christensen *et al*., 1998). The solubility of metals in leachate is enhanced through

complexation by dissolved organic matter. The solubility of organic contaminants (e.g. solvents) in waste may also be slightly enhanced through the presence of high levels of organic carbon in leachate. Hydrophobic compounds may be mobilised through leachate, as they adsorb to organic carbon in solution. For example, benzene- and naphthalene-sulphonates comprise between 1 and 30 per cent of the dissolved organic carbon in landfill-leachates recently analysed in Switzerland (Riediker *et al.*, 2000).

Table 12.3. Key characteristics of landfill leachates from England, Germany and USA (all values in mg/L except pH) (Robinson *et al.*, 1982; Ehrig, 1982; Fetter, 1993).

Parameter	England	Germany	USA
PH	6.2-7.4	6.1-8.0	5.4-7.2
Total dissolved solids	Not analysed	Not analysed	2,180-25,900
Chemical oxygen demand (COD)	66-11,600	3,000-22,000	1,120-50,500
Biochemical oxygen demand (BOD)	<2-8,000	180-13,000	100-29,200
Total organic carbon (TOC)	21-4,400	Not analysed	427-5,890
Ammonia-nitrogen	5-730	741	26-557
Total phosphorous	<0.02-3.4	5.7	0.3-117
Chloride	70-2,780	2-119	180-2,650
Iron	0.1-380	15-925	2.1-1,400
Manganese	0.3-26.5	0.7-24	0.03-25.9
Calcium	165-1,150	80-1,300	200-2,100
Magnesium	12-480	250-600	120-780

12.3.3   *Leachate migration*

In unsealed landfills above an aquifer, waters percolating through landfills and refuse dumps often accumulate or 'mound' within or below the landfill (Figure 12.1). This is due to production of leachate by degradation processes operating within the waste, in addition to the rainwater percolating down through the waste. The increased hydraulic head developed promotes downward and outward flow of leachate from the landfill or dump. Downward flow from the landfill threatens underlying groundwater resources whereas outward flow can result in leachate springs yielding water of a poor, often dangerous quality at the periphery of the waste deposit. Observation of leachate springs or poor water quality in adjacent wells/boreholes are indicators that leachate is being produced and is moving. Leachate springs represent a significant risk to public health, so their detection in situation assessment is critical in order to prevent access to such springs.

One method used to reduce the generation of leachate and, hence, hydraulic heads generating flow from a closed landfill is to place a capping of low permeability material (e.g. clay or high density polyethethylene - HDPE) over the waste deposit in order to reduce infiltration of rainwater. These should be recorded in situation assessment because if a landfill is capped to impede rainwater ingress, reducing leachate volumes, a more concentrated leachate will be generated. Also, microbiological and biochemical reactions will be inhibited thereby prolonging the degradation process and the activity of the waste possibly for decades or even

centuries. Groundwater pollution potential from older capped landfills may therefore be higher than from younger, open landfills.
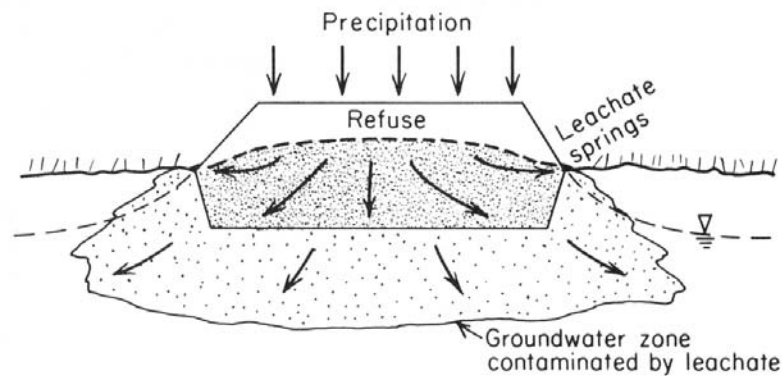

Figure 12.1. Conceptual diagram of leachate migration from a landfill (from Freeze and Cherry, 1979).

Leachate migration is also affected by the manner in which waste is deposited. Compaction of waste prior to deposition reduces its permeability, whereas regular application of a topsoil cover between the loading of waste to landfills induces layering. These characteristics inevitably give rise to preferential flow paths through landfills. Johnson *et al.* (1998) found, for instance, that residence times for rainwater entering a landfill varied from a period of a few days to several years. This is reflected in the frequently temporal nature of leachate "springs", which can appear in wet seasons but subsequently disappear in dry seasons to leave patches of discoloured soil (Jefferis, 1993). Inspections of potential leachate production should, therefore, focus on periods towards the end of wet seasons or following excessive rainfall events. Further, situation assessment needs to account for uncertainties in both the prediction and monitoring of leachate migration from landfills and dumps, in consequence of the complex hydrogeology of waste deposits. This is further addressed in Chapter 24 in relation to problems of planning and management.

Despite the complexity of leachate migration through landfills, fundamental aspects of subsurface contaminant transport, reviewed in Chapter 2.4, can practically be applied to the movement of leachate-derived contaminants from a landfill or refuse dump. These include the thickness of the unsaturated zone, the permeability and moisture content of the earth materials within the unsaturated zone, and the hydraulic conductivity and local hydraulic gradient of geological units in the saturated zone. Poorly conductive units underlying the landfill or refuse dump, e.g. clay-rich material or the presence of an installed artificial liner inhibit leachate migration. On the other hand, discontinuities such as fissures and joints in the subsurface or faults or holes in a liner, dramatically increase leachate flow. For situation assessments, access to hydrogeological information (see Chapter 8) as well as information on design and condition of potentially installed lining system (see also Chapter 24) from both beneath and downstream of landfills, is vital.

Equally important as understanding the magnitude and direction of leachate flow is recognition of the significant biochemical changes that occur, as strongly reducing leachate (redox potential < -100 mV), mixes with shallow underlying groundwater, which is mildly to

strongly oxidising (redox potential > +100 mV). These changes, illustrated in Figure 12.2 represent a reversal of the reducing reactions which take place in the landfill, and give rise to a series of redox zones in the leachate plume adjacent to the landfill in the reverse order to the sequence described in Chapter 12.3.2. The leachate plume thus becomes less reducing and organic carbon in the leachate is rapidly oxidized to CO_2 through contact with oxygenated groundwater.
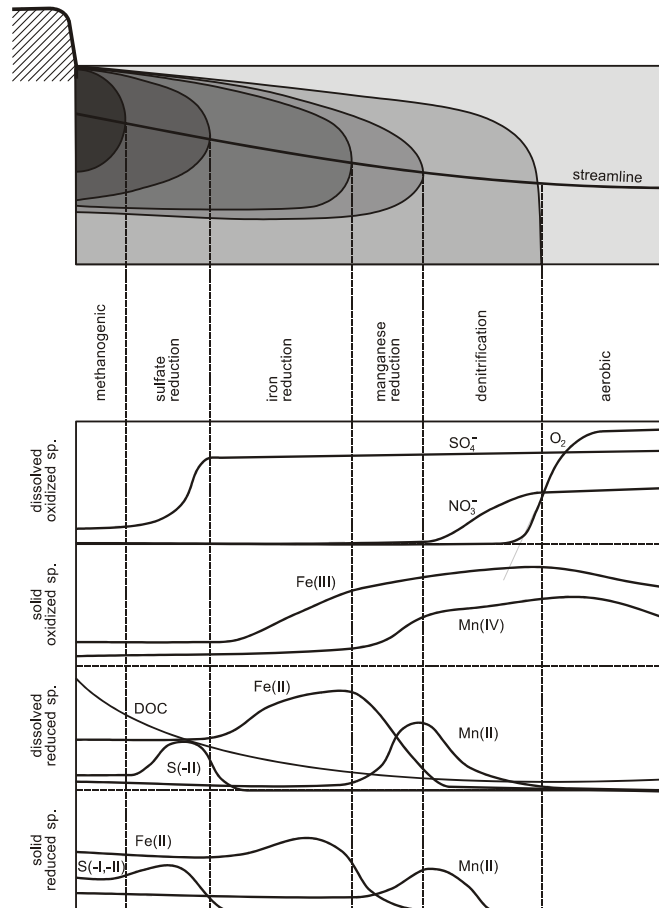

Figure 12.2. Schematic redox zonation in an originally aerobic aquifer downgradient from a landfill, and the distribution of redox-sensitive species along a streamline in the plume (from Fetter, 2001).

The leachate plume undergoes continuous transition in the direction of groundwater flow (Figures 12.2 and 12.3) until conditions are reached where it is no longer anaerobic, and attains redox levels identical to background levels in the aquifer. In this transition zone, chemically reduced species such methane and ammonia disappear, and aqueous nitrogen and sulphur are converted into their oxidised forms of nitrate and sulphate respectively. Iron is oxidised and precipitates as hydrous iron oxide, whereas in contrast, manganese, which is soluble over a wider range of electrochemical (i.e. redox) conditions, remains in solution longer (i.e. travels further with the leachate plume). Consequently, analysis for these compounds and comparison with background levels elsewhere in the aquifer can indicate the presence and extent of the plume. Significantly, a number of detailed studies of leachate plumes indicate that they rarely extend more than a few hundred metres from the landfill, before all but a handful of the most persistent contaminants are completely attenuated (e.g.

Christensen *et al.*, 1994: Robinson *et al.*, 1999). To determine the vertical extent of the plume, often, multiple depth sampling boreholes are required as indicated in Figure 12.3.
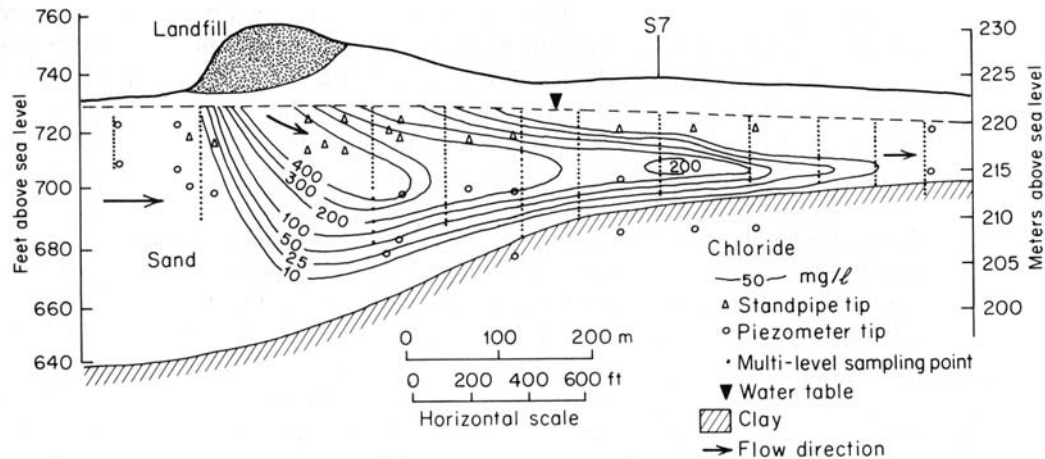

Figure 12.3. Mixing of landfill leachate with shallow groundwater in a sandy aquifer underlain by clay, as indicated by chloride concentrations (from Freeze and Cherry, 1979).

Migration of reactive constituents in leachate, such as microorganisms, organic solvents and metals, is inhibited through biochemical reactions in the plume (e.g. precipitation, volatilisation), and by the interaction of these constituents with the geological materials forming the aquifer matrix (e.g. adsorption, cation exchange), as discussed in Chapters 3 and 4. These processes reduce contaminant concentrations in local groundwater by removing contaminants from solution.

Concentrations of unreactive (i.e. conservative) species in leachate can, however, only be reduced through dispersion and dilution. The extent to which dilution can reduce the concentrations of waste-derived contaminants in the leachate plume adjacent to the landfill or dump, depends upon the magnitude of both groundwater and leachate flows, together with the relative concentrations of contaminants in both the leachate and in the natural groundwaters of the aquifer upstream of the landfill (see Chapter 12.4).

As leachate migrates from a waste deposit in the direction of groundwater flow, the plume disperses (i.e. spreads due to differing contaminant flow paths and flow velocities), and also diffusion through the aquifer. Concentrations of both reactive and conservative contaminants decrease with distance along the groundwater flow path (Figure 12.3). It should, however, be recognised that exceptions to this general trend occur when a contaminant is transformed into a more toxic compound, as occurs in the dehalogenation of perchloroethene (PCE) to trichloroethene (TCE). It should be noted that the concentration of a pollutant at any point removed from its source may vary throughout the year due to seasonal influences on recharge and release of the contaminant, or reaction times governed by variations in factors such as temperature.

12.4   Assessing groundwater contamination associated with waste sites

The checklist below provides guidance on how to approach assessing the likelihood of groundwater contamination through wastes and landfills found in a given drinking-water catchment. Much information for estimating pollution potential can be gleaned from amounts and types of wastes deposited, site management and site location in relation to aquifer vulnerability. As discussed in Chapters 2 and 8, this approach is not always easy, as the hydraulic conductivity and hydraulic gradient are crucially dependent on whether the aquifer has an intergranular or fissure permeability. Flow velocity can be several orders of magnitude higher in the latter. Also different contaminants may migrate at different velocities.

A number of countries use drinking-water protection zone concepts (Chapter 17) to delineate boundaries within which activities such as waste disposal are banned. Their delineation faces the same problems of understanding the hydrology of the setting. However, where protection zones exist, they are valuable for situation assessment which would begin with checking implementation (i.e. whether waste disposal is indeed being kept outside of the protection zone). Also, reviewing the information basis for their delineation will help to understand both the hydrogeological setting as well as the quality of the information base available for determining aquifer vulnerability.

Where hydrogeological understanding is poor and means to improve it are limited, a default approach to assessing pollution potential from wastes is to investigate distances between waste disposal and drinking-water abstraction, and to assess the potential hazard on the basis of the current general body of knowledge on landfill leachate plume attenuation and migration. A number of studies monitoring unlined landfills in operation before the advent of containment landfills have been ongoing over the past 20 years (e.g. Christensen *et al.*, 1994; Blight, 1995; Robinson *et al.*, 1999; Williams, 1999; Williams *et al.*, 1999; Ball and Novella, 2003; Butler *et al.*, 2003). These show that leachate plumes do not usually exceed a length of 1000 m, even in fast-flowing aquifers over periods in excess of 50 years after the initial wastes were deposited, and even within geological media with supposedly poor attenuation potential, such as sandy overburden. The processes of degradation and attenuation operating within the plume result in the front of the plume becoming stationary as degradation processes keep pace with. The migration of plume, and most pollutants, even complex organic compounds, degrade rapidly within the plume and are attenuated within a few hundred metres (Christensen *et al.*, 1995; Hancock *et al.*, 1995).

In the process of developing a GIS model for landfill site selection (Allen *et al.*, 2001), a survey of buffering distances used in various site selection criteria indicated that for individual dwellings with their own water wells in rural areas, a distance of 500 m was widely used, and except in extreme cases this would constitute a safe distance from a landfill for a water abstraction point. This distance could be reduced considerably on the upstream side of the landfill, if the direction of groundwater flow is known. Similarly, studies of leachate plumes (Christensen *et al.*, 1994) indicate that they do not tend to exceed the width of the landfill, so the plume does not fan out from the landfill in the direction of groundwater flow. Where hydrogeological information is available, such as the type of aquifer, groundwater flow direction, and flow velocity, considerably smaller buffer distances of the order of 100-200 m would be adequate on the upstream and lateral sides of the landfill. On the whole, when assessing whether a landfill is safely distant from a water abstraction point, a distance of

500 m would in most cases be adequate, whilst a distance of 1000 m would be extremely conservative.

In contrast to many other human activities which cause diffuse groundwater pollution potential, landfill concentrates this to point sources. This facilitates assessing their pollution potential through screening and monitoring programmes which do not necessitate sophisticated chemical analyses of the wide range of potentially occurring pollutants, but rather select a few persistent substances, such as NH_3 and Cl, to detect and characterise leachate plume migration. Such an approach reflects the major influence that landfill leachate can exert on the abundance and concentration of individual substances present in groundwater. This is primarily valid for organic contaminants, which are almost exclusively anthropogenic, their presence in groundwater often indicating the influence of a waste site, but it is also valid for naturally occurring inorganic groundwater constituents, the content of which is increased by landfill leachate. Leachate migration can therefore be assessed by analysing the concentrations of common inorganic parameters in groundwater down-gradient from a landfill in relation to their concentrations in groundwater sampled sufficiently up-gradient, i.e. where it is not influenced by the landfill-derived contamination.

In order to rank the impact on groundwater of the leachate migrating from a landfill, Kerndorff *et al*. (1992) use a contamination factor (cf), representing the ratio of the measured concentration in the groundwater 10-100 metres down gradient of the landfill to the concentration in the uncontaminated groundwater up-gradient of the site. If the site is not leaking, or if the substance measured is not involved in the leakage event, the ratio should be 1.0. However, if the substance is leaking from the site, the ratio will increase to a value greater than 1.0. Thus, the larger the leakage event, the larger the resultant contamination factor. This approach identifies specific inorganic substances (those with the highest mean contamination factors) likely to be associated with landfill leakage events and therefore suitable for the indication of groundwater contaminations caused by landfills. In the above example, they proved to be the following: arsenic with a cf_{mean} of 122, ammonium with 65.5, cadmium with 26.9, nitrite with 25.7, boron with 21.6, chromium with cf_{mean} of 15.8, and nickel with 14.8. However, in using this approach it must be remembered that substances with high contamination factors are not necessarily those with the highest hazard potential nor those with the highest loads. They merely indicate the potential occurrence of groundwater contamination from a landfill with high loads of substances which may be hazardous due to their toxicity and/or persistency if they move through the aquifer towards a water supply.

12.5   Checklist

NOTE ▶   *The following checklist addresses some key issues that need to be considered when characterising waste disposal and landfill activities in the drinking-water catchment area. It is neither complete nor designed as template for direct use but needs to be specially adapted for local conditions. The analysis of the potential of groundwater pollution from human activity requires combination of the checklist below with information about socio-economic conditions (Chapter 7), aquifer pollution vulnerability (Chapter 8), and other specific polluting activities in the catchment area (Chapters 9-11 and 13).*

ⓘ   **Is waste disposed in the drinking-water catchment area?**

✓ Compile an inventory of sanitary landfills and legal or illegal uncontrolled waste disposal sites or dumps

✓ Compile an inventory of sites potentially producing special types of waste, such as health care facilities, cemeteries, scrap yards, slaughterhouses, industries (consider checklist for Chapter 11)

✓ Compile an inventory of sites storing, processing or treating wastes

✓ Compile historic data from the areas and facilities of interest

✓ For each inventory, identify relevant procedures, processes, responsibilities (who is in charge?) and substances/products in use

✓ Evaluate whether disposal sites were selected according to aquifer vulnerability and physical conditions in the catchment area (e.g. water table, soil, hydrogeology): consider checklist for Chapter 8

✓ …

ⓘ   **What kind and which amounts of waste are disposed is the drinking-water catchment area?**

✓ Estimate the amount of wastes produced and deposited in the drinking-water catchment

✓ For given deposits, assess the type and content of wastes (e.g. domestic, industrial, hospital) deposited (as a first step from the income of the community and local industry)

✓ Assess the likelihood of disposal of hazardous substances (e.g. from industry or hospitals)

✓ Estimate the amount and type of waste collected and deposited on controlled sites (sanitary landfills), on unregulated dumps or that is randomly scattered

✓ Check for indication of illegal wastes imported from other countries and their nature

✓ …

ⓘ   **What is the condition of the disposal sites?**

✓ Evaluate siting, design, construction and technical condition of individual waste disposal sites in relation to aquifer vulnerability and physical conditions in the catchment area (e.g. water table, soil, hydrogeology): consider checklist for Chapter 8

✓ Check whether containment structures are in place and intact (e.g. lining)

- ✓ For sites with hazardous wastes, assess particularly the adequacy of protective structures in place, e.g. defence wells, drainage, containment (see Chapter 24)
- ✓ Assess the type of wastes and wastewater generated by these facilities and whether specific structures exist for separate collection of hazardous wastes or wastewater
- ✓ Identify key structural and technical strengths and weaknesses of individual disposal sites in relation to their groundwater pollution potential (see also Chapter 24)
- ✓ …

ⓘ **Are good management practices in place?**

Note: See Chapter 24 for the information background for these items.
- ✓ Check whether waste management concepts are in place, e.g. for waste reduction and waste separation
- ✓ Assess whether implementation of such waste management concepts is satisfactory
- ✓ Check whether regulations are well known by administration and other staff
- ✓ Check whether broader environmental management concepts pertinent to waste disposal are understood
- ✓ Identify key strengths and weaknesses of the management practices implemented
- ✓ Assess whether containment structures for hazardous agents are intact and monitored at adequate intervals
- ✓ Check whether regular information is distributed, and whether training with respect to handling of wastes is adequate
- ✓ Check whether principles of good practice are followed by health care and research units working with highly infectious material and/or hazardous substances
- ✓ …

ⓘ **Are hazardous events likely to increase groundwater pollution potential?**

- ✓ Evaluate whether and how stormwater events would enhance transport of pollutants to the aquifer
- ✓ Evaluate which spills and accidents are likely to cause groundwater pollution
- ✓ …

ⓘ **Is drinking-water abstracted in proximity to waste disposal sites??**

- ✓ Determine the direction and magnitude of the local hydraulic gradient, and whether drinking water wells are up-gradient or down-gradient of the waste depository
- ✓ Assess distance between (formal and informal) waste disposal sites and drinking-water wells (see Chapter 8)
- ✓ Check adequacy of wellhead protection measures, wellhead construction and maintenance as well as sanitary seals used (see Chapter 18) to prevent ingress of contaminants from waste disposal sites
- ✓ …

ⓘ **Are groundwater quality data available to indicate pollution from waste disposal activities?**

- ✓ Find out if leachate and/or groundwater monitoring programmes are in place around waste disposal sites

    ✓ Check whether seasonal leachate patterns are expected in relation to precipitation
    ✓ Compile data from local or regional waste disposal surveys, research projects or previous monitoring programs
    ✓ Check need and options for implementation of new or expanded monitoring programs likely to detect contamination from waste disposal facilities
    ✓ …

(i) **What regulatory framework exists for waste disposal?**

    ✓ Compile information on national, regional, local, or catchment area specific legislation, regulations, recommendations, or common codes of good practices on siting, construction, operation, maintenance of sites
    ✓ Check whether a regulatory framework exist for waste avoidance, waste separation, and particularly for waste disposal, and whether enforcement appear sufficient to protect groundwater
    ✓ Check whether the regulatory framework adequately addresses environmental and specifically groundwater protection
    ✓ Identify gaps and weaknesses known which may encourage specific pollution problems
    ✓ If wastes are imported, check whether this is due to stricter regulations in the country of origin, and whether the imports are legal
    ✓ …

(i) **Documentation and visualisation of information on waste disposal practices!**

    ✓ Compile summarising report and consolidate information from checklist points above
    ✓ Compile summary of types and amounts of substances expected from the specific waste disposal sites and sites potentially producing special types of waste
    ✓ Map formal and informal waste disposal sites and sites potentially producing special types of waste, preferably including suspected "hot spots" of contamination (use GIS if possible)
    ✓ …

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