EP

United Nations
Environment
Programme

UNEP(DEC)/MED WG.264/Inf.8
9 May 2005

ENGLISH

MEDITERRANEAN ACTION PLAN

Meeting of the MED POL National Coordinators

Barcelona, Spain, 24 - 27 May 2005

GUIDELINES FOR SEWAGE SLUDGE TREATMENT, DISPOSAL AND USE

In cooperation with

WHO

UNEP
Athens, 2005

Table of Contents

1. EXECUTIVE SUMMARY

Sludge originating from wastewater treatment process is expected to increase due to the expanding employment of wastewater treatment technologies. This sludge containsboth valuable and non-dangerous compounds (including organic matter, nitrogen, phosphorus and potassium, and to a lesser extent, calcium, sulphur and magnesium), and pollutants which usually consist of heavy metals, organic pollutants and pathogens.

The characteristics of sludge depend on the original pollution load of the treated water, and also on the technical characteristics of the wastewater and sludge treatment carried out. However, the valuable or non-dangerous fraction in most cases constitutes 98-99% of the total solids. Considering this and the fact that the useful organic fraction usually accounts for 40-70% of the solids, it is understandable why the term "biosolids" is sometimes used instead of the term "sludge". It is a term aiming to emphasize the merits of the bulk quantity of the sludge, while at the same time reflects a certain degree of optimism with respect to the potential problems that may be caused by a negligible in quantity, but great in significance, portion of the sludge related to pollutants, such as metals, organic pollutants and pathogens, which originate from domestic use and services, runoff rainwater (in combined sewers) and connected industrial wastewaters.

A detailed presentation of the quantitative and qualitative characteristics of sewage sludge is given in Part 2.

Given that sludge disposal to surface waters and the sea is no longer an acceptable method of sludge elimination, landfilling under proper conditions to ensure prevention of air, soil and water pollution is a possible elimination practise not entailing excessive costs (although not insignificant due to the requirements for pollution prevention and ultimate restoration of the site). However, in the context of sustainability, elimination of potentially valuable resources, such as sludge, is in discrepancy with recent management concepts favouring reuse and recovery practices. Therefore, landfilling lies at the bottom of the prevention ladder, endorsed by several countries including the European Union and a strong preference for methods involving material and for energy recovery is given.

In view of this, landspreading of sludge for agricultural and other purposes (forestry, silviculture, land reclamation) is promoted within the EU as a first priority, followed by methods for energy recovery (i.e., incineration). This priority is reflected in the anticipated overall future decline of landfilling practises from about 40% to 20% and the corresponding increase of landspreading and incineration (from 45% to 55% and from 14% to 25% respectively).

Landspreading is expected to gain in significance and become the main method of sludge management. Furthermore , a wider scope of land application is promoted which, in addition to agricultural landspreading, includes sludge used in forestry, silviculture and land reclamation.

Incineration is also a method of increasing insignificance deserving attention especially for sludge originating from large cities or receding sludge from a wider area under favourable transportation conditions. An overview of the alternative methods with respective pros and cons, as well as requirements for sludge treatment is presented in Part 3.

As already mentioned, landspreading is expected to gain in significance and become the dominant method of sludge management. It should, however, be stressed that although almost 99% of the sludge contains compounds of agricultural value, the presence of pollutants (pathogens, heavy metals and micro-organics) even at the very small fraction of

1%, creates the need for a very careful approach addressing not only technical aspects but also social and economical. A detailed presentation of these issues is presented in Part 4.

The concerns are discussed in detail and the two main approaches and their respective governing principles are presented in connection to existing indicative legislation frameworks. More specifically, the examples of the US and the EU are analyzed both in terms of current practices and guidance/directives and future trends. It should be noted that the purpose is not to provide an extensive comparative presentation of legislations in various countries, but to focus on the main rationale behind two indicative examples, to assist in drawing useful conclusions which can form the basis for formulation of codes and legislations appropriate for the specific conditions of individual countries.

A detailed discussion concerning sludge incineration is provided in part 5. Although landspreading is a more attractive option, experience has shown that as long as reuse of sludge in agriculture can be easily discredited by the media and thus be temporarily ruled out, alternative solutions must be provided. Incineration is an interesting alternative in such cases. Furthermore, it may prove to be the most cost-effective method for large cities or suitably situated agglomerations of towns. The main factors for the increasing interest in sludge incineration are related to technological improvements which ensure safe, reliable and efficient operation, the ban of sludge disposal to sea, the difficulties associated with landfilling (not sustainable, lack of suitable sites, increasing costs) and public concern with respect to possible long-term impact associated with landspreading.

Irrespective of the method of sludge disposal or reuse, some form of prior sludge treatment is needed. In Parts 3, 4 and 5, where the alternative sludge management practises are discussed, the treatment requirements for each alternative scheme are briefly presented.

Part 6 is devoted to a more analytical description of the treatment processes/stages which can be suitably combined in order to provide the required overall treatment for each management scheme. The various sludge treatment processes are grouped in three categories, depending on their main function. The first category refers to methods which are mainly used for water removal and subsequent sludge volume reduction. The second category refers to methods which are responsible for organic matter destruction and partial pathogen removal (stabilization) and the last category refers to methods which are mainly used for practical elimination of pathogens (sanitization).

2. QUANTITATIVE AND QUALITATIVE CHARACTERISTICS OF SEWAGE SLUDGE

2.1 Definitions

Sludge is a by-product originating from wastewater and water treatment processes. There are three main categories of sludge:

a. sludge originating from the treatment of urban wastewater, consisting of domestic wastewater or of the mixture of domestic wastewater with industrial wastewater and/or run-off rainwater;
b. sludge originating from the treatment of industrial wastewater; and
c. sludge originating from drinking water treatment.

These guidelines deal only with sewage sludge originating from the treatment of urban wastewater, i.e., with the residue generated during the primary (physical and/or chemical), the secondary (biological) and the tertiary (additional to secondary - very often nutrient removal) treatment. The residues of the processes of pre-treatment, also called preliminary treatment, are not considered as sludge. These residues are mainly coarse solid particles, grit, sand and grease and are typically disposed of in landfills.

Regarding sludge, as previously defined several types are usually recognized in the context of wastewater treatment.

Primary sludge: primary sludge is produced following primary treatment. This step consists of physical or chemical treatments to remove matter in suspension (e.g., solids, grease and scum). The most common physical treatment is sedimentation, which involves removal of suspended solids from liquids by gravitational settling. Another physical treatment is flotation, during which the particles rise to the surface by means of air bubbles introduced in the wastewater, and are removed by skimming. Plain sedimentation is the most commonly adopted method due to its simplicity and cost-effectiveness. This method removes about 50% of the suspended solids and produces sludge with a solids concentration ranging between 1.5% and 5%, depending on the mode and frequency of sludge removal. Chemical treatments are coagulation and flocculation, which are used to separate suspended solids when their normal sedimentation rates are too slow to provide effective clarification by gravity. Typically, the chemicals used as coagulants include Fe and Al salts, as well as lime. These chemical processes can achieve 90% removal of suspended solids and produce larger quantities of sludge not only due to the enhanced solids removal, but also due to the production of additional chemical sludge by as much as 25% to 150%, depending on the chemical used.

Secondary sludge: secondary sludge results from the growth of micro-organisms, mostly bacteria, which decompose the organic material and use part of it for synthesis during biological treatment of sewage. Different types of biological systems can be used, usually in the form of either suspended growth or attached growth biomass. The sludge thus produced is called surplus or excess sludge and consists mostly of bacteria with a dry solids content of approximately 1% (suspended growth systems) to 4-5% (attached growth systems).

Mixed sludge: the primary and secondary sludge described above can be mixed together generating a type of sludge referred to as mixed sludge.

Tertiary sludge: tertiary sludge is generated when carrying out tertiary treatment, an additional process to secondary treatment designed to remove remaining unwanted nutrients (mainly nitrogen and phosphorus) through high performance bacterial or chemical processes. Tertiary sludge is usually associated with the removal of phosphorus, which may be performed using chemical processes or biological treatment. Chemical processes consist of chemical precipitation using additives followed by sedimentation. Physical/chemical removal of phosphorus increases the quantity of sludge produced by an activated sludge plant by about 30%. Biological treatment employs specific micro-organisms, which are able to store phosphorus that accumulates within the bacteria enabling its removal with the rest of the sludge. Tertiary sludge can also be associated with sand filtration following biological treatment, aiming to produce an effluent of very high quality. This effluent is often characterized as “reclaimed water” suitable for reuse purposes.

Stabilized (digested) sludge: this term applies to the primary, secondary or mixed sludge after receiving a typical treatment within a wastewater treatment plant, in order to stabilize its organic matter and reduce the generation of odours, reduce its pathogen load and reduce its mass due to destruction of organic matter. Several treatments can be applied to sludge to achieve this goal, in most cases in the form of aerobic or anaerobic mesophilic digestion.

Dewatered - stabilized sludge: stabilization in the form of digestion does not result *per se* in reduction of the water content of the sludge and its total volume. For the reduction of the water content and subsequently the volume of the sludge, dewatering, often in combination with thickening, is usually employed. The methods used are typically applied to digested sludge and range from drying beds to mechanical dewatering devices, such as filter presses, belt presses and centrifuges. The solids content of the dewatered sludge can range from 18% to 35%, depending on the type of sludge and the dewatering method adopted.

2.2 Quantitative characteristics of sewage sludge

Due to expanding application of sewage treatment technologies, the volume of the produced sewage sludge is increasing. Direct evidence of this correlation can be seen in Europe, mainly due to the enforcement of European Environmental Legislation and more precisely of the 91/271/EC Directive concerning urban wastewater treatment. On the basis of the January 1999 Commission Report concerning the implementation of the 91/271/EC Directive, it was anticipated that a 50% increase in the production of sludge (in terms of tonnes of dry matter) would occur over the period of implementation of the Directive 1992 - 2005. More specifically, the sludge produced from the then 15 Member States, would increase from 5.5 million tonnes of dry matter in 1992 to 8.3 million tonnes of dry matter in 2005 (Figure 2.1). As the year 2000 was a landmark in the sense that treatment should have been provided for all majoragglomerations (above 15,000 p.e.), it is reasonable to estimate that, despite observed delays, currently a quantity in the order of 8 million tonnes is produced in the 15 Member States. This corresponds on average to about 55 g dry matter/inhabitant/day. However, this figure is misleading since it does not reflect the diversity of national wastewater treatment systems, including the connection rate of each country. An estimate of the sludge produced per inhabitant served by a wastewater treatment plant, is therefore higher; close to a figure of 70-80 g dry matter/day.

It has often been reported that the sludge produced per population equivalent served is about 40-60 g dry matter/p.e./day, depending mainly on the type of treatment, e.g., extended aeration, digestion of sludge, etc. An explanation of this deviation can be offered by considering the meaning of the population equivalent which incorporates wastewaters from non-domestic activities (commercial, industrial etc.) within a city. As a result, for any given agglomeration the population equivalent figure is greater that the inhabitant figure, often by as much as 50%; thus the two figures (70-80 g dry matter/inh./day and 40-60 g dry matter/p.e./day) are not contradictory.

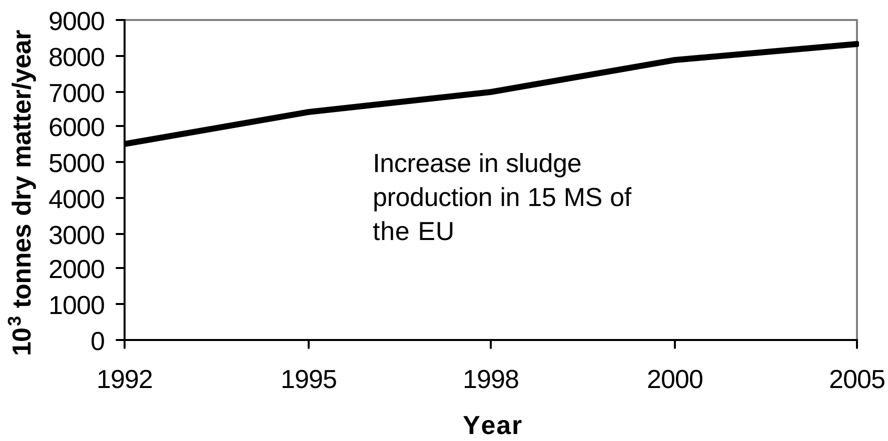

Figure 2.1. Production of sludge in EU-15 (Source: EC Report, 2001).

These quantities are based on dry matter and do not represent the actual quantities of sludge produced including moisture. For dewatered sludge the figures based on dry matter should be multiplied from between threefold to fivefold, to obtain the amount of sludge produced, whilst in the case of liquid non-dewatered sludge a twenty-fold increase can be adopted.

2.3 Composition of sewage sludge

Sludge originating from the wastewater treatment process, contains both valuable and non-dangerous compounds (including organic matter, nitrogen, phosphorus and potassium, and to a lesser extent, calcium, sulphur and magnesium) and pollutants which usually consist of heavy metals, organic pollutants and pathogens.

The characteristics of sludge depend on the original pollution load of the treated water, and also on the technical characteristics of the wastewater and sludge treatment carried out. However, the valuable or non-dangerous fraction in most cases constitutes 98-99% of the total solids. Considering this and the fact that the useful organic fraction usually accounts for 40-70% of the solids, it is understandable why the term "biosolids" is sometimes used instead of the term sludge. It is a term which aims to emphasize the merits of the bulk quantity of the sludge, while at the same time reflects a certain degree of optimism with respect to the potential problems that may be caused by a negligible in quantity, but great in significance, portion of the sludge related to pollutants, such as metals, organic pollutants and pathogens, which originate from domestic use and services, runoff rain water (in combined sewers) and connected industrial wastewaters.

2.3.1 Valuable or non-dangerous fraction

This fraction includes organic matter, nitrogen, phosphorus and other compounds such as potassium, sulphur, magnesium, sodium etc.

Organic content: sludge organic matter is mostly in the form of hydrocarbons, amino-acids, proteins or lipids. Its content in urban sewage sludge is high (usually more than 50% of the dry matter), but varies according to the treatment and conditioning techniques applied. Content level may be reduced due to dilution after incorporation of lime or salts for instance. Table 2.1 compares the content of organic matter of urban sewage sludge against other urban wastes and animal manure. This organic matter can be of value either as a useful soil additive or as a source of energy.

Table 2.1

Content of organic matter in sludge from different treatment processes and origin

	Organic matter content % DM
Urban Sludge	
Aerobic digestion	60-70
Anaerobic digestion	40-50
Thermal treatment	<40
Lime treatment	<40
Composting	50-85
Urban compost	40-60
Green wastes composting	30-60
Animal manure	45-85

(Source: EC Report, 2001)

Known benefits of organic matter application to soil are improvement of the physical properties of the soil (such as structure), improvement of the retention capacity of minerals and water, improvement of the soil bearing strength, and the reduction of the potential for surface runoff and water erosion. Furthermore, degradation of organic matter can increase the soil content in compounds of agricultural value (such as N, S, Mg etc.), which are more slowly released than in the case of mineral fertilizers and, therefore, available for a longer period to crop. Finally, organic matter is an energy source for micro-organisms living in the soil.

As previously mentioned, sludge organic matter consists mainly of fairly readily degradable matter with small amounts of lignin or cellulose. Due to the relatively fast mineralization of this organic matter, a peak in the nitrate and pollutant level in the soil may be generated. Sludge treatment in the form of digestion, composting etc., which reduces the rapidly degradable material and/or adds stable organic matter to the sludge, may be beneficial, as, in this case, organic matter mineralizes more slowly and nutrients are gradually released, reducing the potential risk of nitrogen leaching to the groundwater.

Contrary to soil additive, the organic matter of the sludge has potential as a source of energy. Typically, 1 kg of sludge organic matter has a calorific value of about 5500 kcal/kg. Therefore, for sludge containing 50-70% organic matter, the calorific value of the sludge is 2250 – 3850 kcal/kg of dry matter. Given that water requires approximately 640 kcal/kg water to heating up and evaporation at 100% efficiency, it follows that under ideal theoretical conditions, a dewatered sludge with 15% dry matter content and 70% organic matter, or alternatively 25% dry matter content and 50% organic matter, can be subjected to a self burning process in an incinerator. However, in practise due to the required excess air and the fluctuations of such parameters as the composition and dry content of the sewage sludge, as well as due to losses during the heating and water evaporation process, the theoretical energy demand stated for water evaporation must be increased by about 40%. As a consequence the minimum solid content of the dewatered sludge for self burning should be around 28% and 35% for sludge with organic contents of 70% and 50%, respectively.

Nitrogen: Nitrogen is mostly found in sludge in organic form, and to a lesser extent as ammonium or nitrates. Other mineral forms of nitrogen are only found as traces. Inorganic nitrogen as NO_3 is the most water-soluble form of N and, therefore, is of the most concern for groundwater contamination because of its high mobility in most soil types. Inorganic nitrogen in the form of ammonium is volatized to ammonia and thus may not be available to plants.

Organic nitrogen, as a slowly released form of nitrogen, is available for plant growth. Nitrogen content of chemical fertilizers is more directly available to plants and is consequently released to groundwater or the atmosphere as ammonia, especially during high temperatures. Figure 2.2 presents three curves, representing patterns of nitrogen uptake by plants, sludge nitrogen and fertilizer nitrogen availability. The first two curves practically coincide, thus suggesting the suitability of sludge application in order to maximize nitrogen uptake by plants.

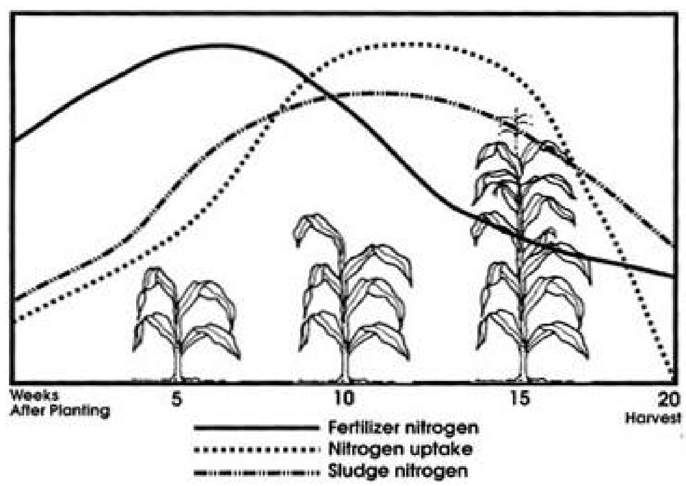

Figure 2.2. Nitrogen availability curves and nitrogen uptake patterns by plants (Muse *et al*, 1991).

The nitrogen content of the sludge is influenced by the treatment the sludge is subjected to and by sludge storage periods. In freshly produced dewatered sludge a N content of 2-5% of DM is typical.

Nitrogen availability to plants depends on the type of sludge. It varies between 4 and 60%, but great variations have been reported within one type of sludge, depending on treatment, with the lower figures for composted sludge and the highest for aerobically digested sludge (Table 2.2).

Phosphorus: phosphorus in sludge is mostly present in mineral form representing between 30 and 98% of the total phosphorus, depending on the type of sludge. Phosphorus content in sludge is related to the type of treatment adopted and varies between 1 and 6% of dry matter. In most cases, the content is around 1 -2% and the higher percentages are associated with sludge produced from treatment systems involving chemical or biological removal of phosphorus.

Table 2.2

Nitrogen availability depending on the type of sludge

Sludge type	Availability ?, %
Aerobic digested sludge	24-61
Anaerobic digested sludge	4-48
Digested composted sludge	7
Composted raw sludge	4
Thermally dried sludge	7-34

(Source: ADEME, 1996)

Composted sludge has a lower phosphorus content than non-composted sludge, due to the low phosphorus content of the co-products used during the process. Contrary to nitrogen, phosphorus content in sewage sludge is not significantly reduced during storage.

Other compounds of agricultural value: other compounds present in sludge, such as potassium, sulphur, magnesium, sodium and oligoelements (e.g., boron, cobalt, selenium and iodine) may be of interest in crop production, each of them being useful for plant development and growth. However, they appear in sludge in various forms (e.g., magnesium sulphate or magnesium oxide), and their effectiveness depends on their availability. In light of this, the agricultural value of these compounds is not extensively documented in literature.

2.3.2 Pollutants

The three main categories of pollutants that affect sludge quality are pathogens, heavy metals and poorly biodegradable organic compounds, also called persistent organic pollutants (POPs).

Pathogens, which originate from human and animal metabolism can, be eliminated by sludge treatment but removal in the treatment plant of the two other categories of pollutants does not seem technically or economically feasible. Instead of an “end of pipe” approach, preventive action should be taken at source. Inflows of heavy metals and organic compounds in the sewer must be reduced, taking into consideration: discharges from domestic uses and services, discharges from run-off rainwater into combined sewer system and discharges of connected industrial wastewater. Table 2.3 lists typical sources of pollutants associated with urban wastewater.

The contents of some pollutants used in households, services or industrial processes can be changed in order to avoid discharges of pollutants into the sewer. Other discharges into the sewer can be stopped by collection and separate treatment of the polluted waste. For example, thermometers or dentists’ amalgam fillings can be mercury free. Residues of paints, solvents, laboratory chemicals etc., should not end into the sink but be collected, recycled or treated separately.

Table 2.3

Sources of pollutants in urban wastewater

Pollutant sources	Domestic use and services	Run-off rainwater (combined system)	Connected industrial wastewater
Pathogens	Human metabolism	Animals faeces (pets)	Limited (meat industry)
Heavy metals	Paints (Pb), Amalgam fillings (Hg), Thermometers (Hg), Pipe Corrosion (Pb, Cu)	Rain (Pb, Cd, Zn), Tyres (Cu, Cd), Roof corrosion (Zn, Cu), Oil (Pb)	Various
POPs	Paints, solvents, Wood treatment, Medicines, Detergents, Cosmetics	Oil, Pesticides (gardens), Tar, Road de-icing, Rain (pesticides, combustion)	Various

2.3.2.1 Pathogens

The main source of sewage sludge contamination by pathogens is human faeces. The sanitary level of the population is directly related to the pathogen load of sludge, whereas fauna (rodents) and flora that may develop in sewers and animal droppings through runoff, also contribute to wastewater contamination. This load may be increased when food industry (dairy products production or slaughterhouses) is connected to sewers.

Pathogens found in sewage sludge are of five main types: bacteria, viruses, fungi and yeast, parasitic worms, and protozoa. Their accumulation in sludge occurs either by direct settling (mainly eggs, cysts and protozoa that have sufficient density) or by adsorption on suspended matter such as activated sludge flocs (bacteria and viruses).

Three main types of risks are connected with collection and processing of sludge, namely occupational health risks, risk concerning the product safety and environmental risks. Pathogens can present a public threat if they are transferred to food crops grown on land where sewage sludge has been applied.

Bacteria: bacteria found in sludge are numerous including *Salmonella spp.* and *E. coli*. Table 2.4 presents a selection of bacterial pathogens typically found in sewage sludge and the diseases or symptoms related to their presence (Epstein, 2002).

Table 2.4

Selection of bacterial pathogens of concern in sewage sludge

Bacterial pathogen	Disease / Symptoms
Salmonella	Salmonellosis, Gastroenteritis
Salmonella typhi	Typhoid fever
Mycobacterium tuberculosis	Tuberculosis
Shigella sp.	Shigellosis, Bacterial dysentery, Gastroenteritis
Campylobacter jejuni	Gastroenteritis
E. coli (pathogenic strains)	Gastroenteritis
Yersinia sp.	Versiniosis
Vibrio cholerae	Cholera

(Source: Epstein, 2002)

Salmonella is the most important because of the risk to grazing animals - *Salmonella spp*. is naturally present in the environment. *Escherichia coli* is naturally present in the human and animal digestive tract. About 140 serological groups have been listed, of which only a few are pathogenic (for instance *E. coli* O157) when their proportion increases. They are useful indicators of faecal pollution of water. Observed levels of *E. coli* in the environment are important. *Shigella spp, Pseudomonas, Yersinia, Clostridium, Listeria, Mycobacterium, Streptococcus* and *Campylobacter* are types of pathogenic bacteria also found in sludge.

Typical concentrations of bacterial pathogens in sludge in terms of number per g DM (Dry Matter) are presented in Table 2.5.

Table 2.5

Concentration of bacterial pathogens in primary and secondary sludge

Pathogen	Organisms	Primary sludge (per g DM)	Secondary sludge (per g DM)
Bacteria	*Total coliforms*	10^8 - 10^9	$7x10^8$
	Feacal Coliforms	10^7 – 10^8	$8x10^6$
	Enterococci	10^6 – 10^7	$2x10^2$
	Salmonella spp	10^2 – 10^3	$9x10^2$
	Clostridium spp	10^6	-
	Mycobacterium	10^6	-

Viruses: many types of viruses may be found in sludge such as *Enteroviruses (Poliovirus, Echovirus, Coxsackievirus A* and *B*), *Adenovirus, Reovirus, Astrovirus, Calcivirus* and *Parvovirus* (Table 2.6).

Enteroviruses occur widely in sewage sludge in concentrations 10^2-10^4 per g dry matter. Hepatitis A virus, which is a human specific virus, may also be present. There is no record of the human immunodeficiency virus (HIV) having been isolated from faeces, and epidemiological evidence shows that sewage and water have not been implicated in the transmission of HIV.

Table 2.6

Selection of viruses of concern in sewage sludge

Viruses	
Adenovirus	Respiratory disease, Gastroenteritis
Poliovirus	Poliomyelitis, Meningitis, Fever
Coxsackie virus A	Herpangina, Respiratory disease, Meningitis, Fever
Coxsackievirus B	Myocarditis, Congenital heart anomalies, Respiratory disease, Meningitis, Pleurodynia, Rash, Fever
Echovirus	Meningitis, Respiratory disease, Rash, Diarrhoea, Fever
Reovirus	Not clearly established
Astrovirus	Gastroenteritis
Calcivirus	Gastroenteritis
Parvovirus	Enteric infection
Norwalk agents	Gastroenteritis
Hepatitis A virus	Infectious hepatitis
Rotavirus	Gastroenteritis

Parasites: parasites are organized living bodies, which need a host to grow or reproduce during one or many steps of their life cycle. Different types of parasites exist, such as helminths, mushrooms or protozoa. Some of them may develop a cyst or egg stage in order to resist environmental stress. Helminths are worms and include Cestodes and Nematodes. Protozoa are unicellular organisms, most of them living in aqueous environments. Different types of parasitic worms and protozoa may be found in sludge, as shown in Table 2.7.

Parasites are found in sludge in concentrations of 10^2-10^3 per g dry matter.

Epidemiological importance of sludge related pathogens: pathogens may survive for a remarkable period of time in sludge, the soil environment (usually within the top 2-3 cm of the soil layer) and plants (Table 2.8).

Possible modes of transmission of sludge pathogens and their epidemiological importance are summarized in Table 2.9. The direct or indirect transmission of zoonotic agents to farm animals is generally regarded as the most relevant fact in connection to agricultural utilization of untreated or insufficient treated sludge.

Direct transmission to humans by handling contaminated products in the household, although a relatively rare event, must be regarded as a risk. In addition, accidental contact of immunocompromised persons to contaminated sludge or sludge products may result in an infection. The occupational risks in processing and handling of sludge and related products must also be taken into account. The indirect transmission to humans is of special importance. The introduction of pathogens into the food chain via contaminated fertilizer, leading to contaminated animal feed and thus to infection of farm animals and/or excretion of pathogens is of basic epidemiological significance. The risk of transmission of pathogens in human food by living vectors such as insects, rodents and birds from processing, handling and agricultural utilization of slurry has also to be taken into account. Introduction of pathogens into the environment leads to carriers in the natural fauna and moreover, to an introduction of transmissible undesired properties of bacteria like antibiotic resistant plasmids into micro-flora and biocenosis.

Table 2.7

Parasites of concern in sewage sludge

Protozoa	
Entamoeba histolytica	Amoebic dysentery, Amoebiasis, Acute enteritis
Giardia lamblia	Giardiasis, Diarrhoea
Balantidium coli	Balantidiasis, Diarrhoea, Dysentery
Cryptosporidium	Gastroenteritis
Toxoplasma gondii	Toxoplasmosis
Helminths – Nematodes	
Ascaris suum	Fever, Respiratory effects
Ascaris lumbricoides	Ascariasis, Digestive and nutritional disturbances, Abdominal pain, Vomiting
Ancylostoma duodenale	Hook worm disease, Ancylostomatitis
Necator americanus	Hook worm disease
Enterobius vermicularis	Enterobiasis, Intestinal inflammation, Mucosal necrosis
Strongyloides stercocalis (threadworm)	Strongyloidiasis, Abdominal pain, Diarrhoea
Toxocara canis (dog roundworm)	Fever, Abdominal pain, Neurological symptoms
Trichuris trichuria (whip worm)	Trichuriasis, Abdominal pain, Diarrhoea, Anaemia
Helminths – Cestodes	
Taenia saginata (beef tapeworm)	Taeniasis
Taenia solium (pork tapeworm)	Taeniasis
Hymenolepis (dwarf tapeworm)	Taeniasis

Table 2.8

Survival of pathogens in soil and plants

Pathogens	Survival in soil	Survival in plants
Bacteria: *Salmonella,* Coliforms	< 70 days (often < 20 d)	< 100 days (often < 20 d)
Enteroviruses	< 100 days (often < 20 d)	< 60 days (often < 15 d)
Helminths: *Ascaris, Taenia saginata*	Several months	< 60 days (often < 30 d)
Protozoa: *Entamoeba histolytica*	< 20 days (often < 10 d)	< 10 days (often < 2 d)

Table 2.9

Epidemiological importance of processed wastes and residuals as well as resulting products

A. Direct transmission to farm animals - Contamination of meadows - Introduction of pathogens by storage and processing close to susceptible animals - Aerogenic transmission by spreading the materials into farmland
B. Direct transmission to humans - Handling of contaminated products in the household - Occupational exposure to contaminated products - Accidental transmission to immunocompromised persons
C. Indirect transmission to farm animals - Via feed from contaminated sites - Via living vectors
D. Indirect transmission to humans - Via introduction of zoonotic agents into the food chain - Via food contaminated by living vectors
E. Introduction into the environment - Generation of carriers in the fauna - Introduction into the micro-flora

2.3.2.2 Heavy metals

Numerous heavy metals are present in sludge. Heavy metals may affect plant health and growth, soil properties and micro-organisms, livestock and human health, and accumulate in the environment. With respect to sludge, special attention has been given to the following seven heavy metals usually found in sludge: lead (Pb), zinc (Zn), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni).

Lead*:* lead is largely used in the industry for pipes, battery, and ammunition production. It is also incorporated in paintings. Its use in petrol is being reduced. There are two main origins of lead in sludge: water from road runoff and alteration of old pipes. Industrial effluents may also contain lead.

It seems that only 5 to 10% of lead ingested via drinking water or foodstuffs is assimilated, up to 90% of which is stored in the skeleton and then slowly transferred into the blood. Principal excretion route is urine. Its half-life in blood is of about 20 - 30 days, whilst in bones about 10 to 20 years. Lead causes anaemia and renal disturbance. Under exposure at high levels (1,200 µg/l in blood), paralysis of upper members and encephalopathy have been observed. Children exposed present slower brain development. Long-lasting absorption of lead in blood concentrations of 400 µg/l results in chronic intoxication. As a consequence, children may suffer from psychomotor and intellectual disturbances, and adults from hypofertility. The tolerable weekly exposure has been set at 25 µg/kg of body weight in 1993 (FAO and WHO, expert committee, 1993).

Zinc: zinc is used in surface treatment and is mostly used in alloys. It is also found in battery, as protective layer in the building industry, in textile, pharmaceutical and insecticide industry. Zinc in sludge originates mostly from pipe alteration, and to a secondary extent, from industrial effluents.

Zinc is essential in the animal kingdom for many physiological processes: growth and cellular differentiation, reproductive functions and embryo development, the integrity of the skin and healing, the immune system, the development and functioning of the nervous system and the sensory system. Zinc is fixed in the bones, liver and kidneys.

Maximum tolerable dietary levels for animals have been assessed to be about 500 mg/kg DM (Smith, 1996). Meat and cereal products are the food categories that contribute most to the human intake of zinc, providing respectively 41% and 21% of total contribution. The average contribution from food only covers 60 to 70% of the nutritional level in zinc. The recommended nutritional amount of zinc is 15 mg per day for humans, or about 1.5 mg/kg of body weight per week. An increase in zinc levels in food, whilst still within an acceptable tolerance could prove beneficial to human health.

<u>Cadmium</u>: cadmium is a soft, ductile metal which is usually obtained as a by-product from the smelting of lead and zinc ores. The principal use of cadmium is as a constituent in alloys and in the electroplating industry. Other uses of cadmium include paints and pottery pigments, corrosion resistant, coating of nails, screws, etc., process engraving, cadmium-nickel batteries, and as fungicides. Cadmium is also naturally present in soils and mineral fertilizers. Cadmium in sludge has a mainly an industrial origin, but can also originate from household effluents (cadmium is present in cosmetic products and gardening pesticides). It may also result from the runoff of rainwater, after atmospheric deposition of the metal.

Cadmium accumulates in the organism as its biological half-life is about 30 years. It is particularly toxic to animals and has been found to cause growth deficiencies and provoke cancers in some animal species.

Long-term human exposure to cadmium leads to renal dysfunction, and epidemiological studies carried out on exposed workers population showed a wide variety of effects, such as irritation of upper respiratory tract, metallic taste in the mouth, cough and chest pain. Cadmium and cadmium compounds have been classified as carcinogenic.

Estimates of typical cadmium intake are, on average, 20 µg/person/day, coming mainly from vegetables and cereals. This intake is about one third of the tolerable weekly intake of 7 µg/kg body weight, suggested by the joint FAO/WHO Expert Committee in February 1993.

<u>Nickel</u>*:* nickel is used for the production of stainless steel and in alloys for coins and production of various instruments. It is also used for metal surface treatment and battery production. Nickel in sludge originates from household effluents (cosmetic products and pigments) but also from industrial effluents due to activities mentioned above.

Nickel intake has been estimated between 150 and 800 µg/day with 60 µg from drinking water. Individual daily requirements are about 35 µg. Acute toxicity only occurs in adults following absorption of around 250 mg of the metal ingested in the form of soluble salts. Nickel is not a metal that accumulates to any significant extent throughout the food chain.

<u>Copper</u>*:* major sources of copper are industry (copper industry, non-ferrous metals industry, incineration). Copper in sludge and wastewater results mainly from household effluents (domestic products, pipes corrosion) but can also be of industrial origin (surface treatments, chemical and electronic industry).

Copper is involved in many physiological functions, including haematopoiesis, elastin and collagen synthesis, and in oxydo-reduction reactions. Copper is also a co-enzyme in many metalo-proteins. It is an essential element of low toxicity. Copper is not considered a human carcinogenic. Instead, pathological symptoms are more related to copper deficiency. The main sources of copper in human food are meat products (27% of total contribution), cereals (28%), fruit and vegetables (21%) and dairy products (13%). Adult copper requirements vary between 1.5 to 3 mg per day, or approximately 150-300 µg/kg body weight per week.

<u>Chromium</u>: large amounts of chromium are found in the terrestrial crust. The most important part of the extracted chromium is used in alloys, i.e., to produce stainless steel. It is also used for its heat resistance and wood protection properties and, in the chemical industry, as a tanning agent and pigment. Chromium may be found in several forms, mainly trivalent (referred to as CrIII), or hexavalent (referred to as CrVI).

According to the level of industrialization of a region, the origin of chromium found in sludge can be attributed to:

- 35 to 50% from industry (surface treatment, tannery, chemical oxidation);
- 9 to 50% from runoff (dust, pesticide, fertilizers); and
- 14 to 28% from household effluent.

Cadmium is essential to human and animal nutrition as it is required for the metabolism of sugar. The two different oxidation states do not present the same level of toxicity, the hexavalent form being more toxic. Hexavalent chromium, contrary to the trivalent form, easily crosses membranes and binds to cellular proteins. The toxic effect of the hexavalent form is to a large degree due to the strong oxidizing effect of this ion (EPA, 1995). It has been shown that chromium could have gastro-intestinal effects, as well as impacts on the nasal wall and mucous membranes. Once absorbed, very little chromium is assimilated (about 0.43% assimilation) and through mechanisms that are little understood. Chromium VI has been classified as carcinogenic to humans.

Deficiency symptoms may be observed when Cr is present at very low concentration in the diet. Studies of human nutrition have shown that a daily diet is often deficient in chromium. Measurements carried out on different diets in North America and Europe showed daily intakes varying from 20 to 30 µg per day and lying slightly below what should be contained in a normal daily diet (50-200 µg).

<u>Mercury</u>*:* mercury can be found in different chemical forms, which determine its toxicity and bioavailability. In its inorganic form, mercury is present in the air as dust or in water. It has a natural presence in the environment but also originates partly from industrial activities: mining, founding, coal combustion and incineration. Mercury can be easily found in gaseous form. In organic form, mercury is mainly present in alimentation as it results from a biological process and, therefore, concentrates in the food chain. Mercury in sludge comes from pharmaceutical products, broken thermometers, runoff water and industrial discharges.

Metal mercury impacts on human health have mainly been observed on the nervous system. Symptoms are trembling (initially affecting hands) and emotional fragility. Neuromuscular affections have also been observed. Other forms of non-organic mercury may also induce renal dysfunction. Methyl-mercury has effects on the nervous system, inducing delayed development. Methyl-mercury has also been classified as possibly carcinogenic according to studies carried out on animals, but data available on humans does not allow for definite conclusions. Other forms of mercury have not yet been classified as to their carcinogenicity.

Mercury levels in cereals, meat products, fruits and vegetables range from 6 to 20 µg/kg. Dairy products and the soil strata contain only low amounts of mercury. Fish is the primary source of mercury in food. In terms of human consumption, the WHO and FAO recommend a maximum daily intake of 43 µg per day for the total amount of mercury absorbed by an human adult and 29 µg per day in the case of methyl-mercury.

2.3.2.3 Organic pollutants

There is a large number of POPs that occur in sewage, which can persist through treatment processes such as anaerobic digestion and then build up in soils to which sewage sludge is applied. On the whole, persistent compounds are quite hydrophobic and they bind to soil organic matter. However, there is a large range of both hydrophobicity and the volatility of the compounds involved.

Many persistent organic pollutants like PCBs, dioxins and pesticides (DDT) are known endocrine disrupters and they are, because of their physicochemical properties (low water solubility), accumulated in sewage sludge. Reuse of sludge may lead to a recirculation of these persistent compounds to human food items and to animal feed.

Most organic pollutants are not taken up by plants. However, a risk of contamination of the food chain exists when spreading sludge directly onto crops, especially on plants which are to be consumed raw or semi-cooked.

Soil and sludge ingestion on land used for grazing is the main route for animal contamination. Accumulation of bioaccumulative compounds such as PCDD/Fs, PCBs or PAHs, may occur in meat and milk. However, it is presently not possible to assess the quantities and fates of organic compounds ingested by animals.

It appears that the consumption of animal products is the major source of human exposure to sludge-borne organic pollutants, due to the ingestion of soil by livestock. As in the case of heavy metals, it is assumed that the specific contribution of sludge-borne organic pollutants to the human diet is very low, when considering the reduced proportion of the utilized agricultural area onto which sludge spreading takes place.

It should be noted that, at present, no universally accepted and validated analytical method exists for analyzing most organic compounds. There is also a lack of data concerning levels of organic pollutants in European sewage sludge as no regular survey has been performed in the past. However, concern has been expressed by several countries (Denmark, Germany, Sweden, Poland) as well as by the Commission of the European Union (EU, 1999) regarding the following groups of organic pollutants: PAH (Polycyclic aromatic hydrocarbons), PCB (Polychlorinated biphenyls), PCDD/F (Polychloro-dibenzo-dioxins/furans), AOX (Sum of organohalogenous compounds), LAS (Linear alkylbenzenesulphonates), NPE (Nonylphenol and Nonylphenolethoxylates) and DEHP (Di-2-ethylexyl-phthalate).

<u>PAH (Polycyclic aromatic hydrocarbons)</u>: PAHs are composed of 2 to 7 aromatic rings associated in a compact way. They are mostly found in liquid form. PAHs are numerous. Among others, the following compounds may be mentioned: naphtalene, polyphenyls acenaphtene, phenanthrene, fluorene, fluoranthene, pyrene, benzo(a)pyrene.

Naphtalene is used in the colouring industry, as a component in wood treatment products, and in mothballs. The polyphenyls are used as refrigerating fluid or as fungicide in the paper industry. PAHs are also generated as by-products of incomplete combustion in certain industries in which carbon and hydrogen are pyrolyzed: iron and steel industry, rubber industry, etc. They are produced in a mixed form, and their relative proportion in the mixture could enable the tracing of their origin.

PAHs can be acutely toxic, but generally at very high doses, making acute systemic toxicity observable in some animal tests but not likely to occur in humans, except in industrial context.

Napthalene is not highly toxic. On the contrary, bi and polyphenyls have an affect on the nervous system as they are lipophilic. They also have an impact on the liver. Some of them have been classified as possibly carcinogenic, like Benzo-a-pyrene which is assumed to be the most toxic of the PAHs. Some other PAHs have a carcinogenic effect after chemical activation through enzymes in the body.

There are three sources of PAH in sludge:

- PAHs are contained in exhaust gas and in the runoff of raining water on roads;
- PAHs are generated in the fumes of industrial thermal units and may reach the soil through rainwater; and
- PAHs are also found in industrial effluents.

PAHs can concentrate strongly in sludge and are little degraded by biological processes of water treatments.

According to available data in literature, sludge can contain between 0.018 and 10 mg/kg DM of PAHs in EU Member States.

Generally, PAH uptake by crops is low and does not represent a risk to the humanfood chain, even when sludge is applied to lipid rich root crops (especially carrots) which is a worst case condition of PAH exposure.

In accordance to the aforementioned, it may be assumed that there are very few transfers of PAHs to the environment media and the food chain. Therefore, the level of exposure to sludge-borne PAHs is likely to be low.

<u>PCB (Polychlorinated biphenyls)</u>: PCB is a group of substances obtained by chlorination of biphenyls. There are about 200 different kinds of PCB, so-called congeners, differentiated by their level of chlorination. PCBs are not naturally present in the environment and used to be incorporated in inks or as dielectric or heat-exchange fluid, and may have a lot of other industrial uses: lubrication, wood protection, paints, etc. They are, however, ubiquitous in the environment but their production was stopped in the 70's and their level in the environment has gradually fallen over the last years. PCB's primary transport route is atmospheric transport.

Higher chlorinated PCB mixtures have been shown to be carcinogenic in laboratory animal experiments. Recent research also indicates that exposure to PCBs may cause reproductive and neurodevelopmental changes in exposed laboratory animals and in some people with environmental exposure to PCBs. They also may have teratogenic action, as well as impacts on the liver and thyroid.

PCBs come from the industry and from oils. They also come from everyday products such as paper and alimentation. PCB content in sludge varies between 0 and 250 mg/kg DM in European Union Member States.

Uptake of PCB by plants under field conditions is fairly well documented and appears to be very limited.

The amount of PCB that animals might ingest due to consumption of plants that have taken up or adsorbed PCB vaporized from the soil, cannot be calculated as the data are insufficient. The level of contamination of PCB in animal tissues increases with the level of chlorination of the PCBs. The maximal concentration of PCBs in milk fat was four to five times larger than the dietary content. The contribution from sludge to meat and dairy products seems to be below 1% of the total PCB content in these foodstuffs.

Distribution of sewage sludge for soil improvement may increase PCB concentrations in meat and dairy products, but is unlikely to affect concentrations in fish and drinking water. Distribution of sludge would lead to an increase of about 0.1% in milk products and by about 0.5% in meat products. Therefore, sludge-borne PCB seems to contribute very little to the total human exposure to PCB.

<u>PCDD/F (Polychloro-dibenzo-dioxins/furans)</u>: dioxins and furans are not very different in their structure. They are constituted of two chlorinated Benzene rings linked by a dioxin (two oxygens) or furan (one oxygen) cycle. As for PCBs, there are different levels of chlorination. Therefore, about 200 congeners exist. In the industry, PCDD/Fs are not used as such, but are by-products of combustion reaction. They appear during the manufacture of insecticides, herbicides, antiseptics, disinfectants and wood preservatives. They are naturally produced in very small amounts following forest fires, for instance.

PCDD/Fs are usually generated during combustion of products containing organic matter and chlorine. Therefore, one significant potential source of dioxins and furans is the incineration of waste. They are destroyed at high temperature, but they may reform during the cooling phase at about 400 – 500°C.

It is considered that dioxins and furans have the same toxicity. The position of the WHO is that the tolerable daily intake is 10 pg per kg body weight per day. Even as trace, it generates chloracne and impacts the skin pigmentation. It has also impacts on the liver, is carcinogenic and teratogenic. The half-life of the dioxins and furans in the human body is about 6 years and they are lipophilic. They are therefore cumulative.

Three origins have been identified:

- as by-products of the industry, they can appear in industrial effluents;
- they are present in the environment in a diffuse form (for instance after deposition on soil and plants). They can enter the sewage system after run-off from street and roofs; and
- PCDD/Fs are present in the commercial preparations of insecticide products.

PCDD/Fs can concentrate in sludge. Sludge loss may happen through biological degradation or volatilization, but dioxins could also be generated during the wastewater treatment process because of biological activity.

The dioxins and furans found in the upper parts of plants seem to come from the air. It is explained by atmospheric deposition to foliage and adsorption of contaminants from the gaseous phase, which are derived principally from other sources.

The influence of sludge amendment on the PCDD/F concentration in aboveground plant tissues can be ignored in the pathway analysis of human exposure.

Distribution of sludge is expected to increase animal exposure to PCDD/Fs via soil ingestion. The PCDD/Fs pass into the bloodstream via the gastrointestinal system at varying absorption rates, according to the congener involved. When the compounds are poorly chlorinated, the absorption rate is higher. These compounds are not metabolized and are stored in the adipose tissues or may be eliminated into the milk. Drinking water and inhalation have been shown to be negligible intake sources.

The exposure routes to dioxins are numerous. It can be inhaled from air, but can also be ingested after deposition of dioxins and furans and their concentration in plant or animal fat and milk.

The total contribution of sludge-borne PCDD/Fs seems to be small. However, due to their physicochemical properties, dioxins are also persistent in the human body and concentrate along the food chain. Therefore, even low exposure levels are of importance.

The application of sludge to crops does not significantly influence human exposure to PCDD/Fs, due to the inefficiency of transfers from soil to plant tissues. In contrast, transfers of dioxins and furans from sludge-amended pasture to livestock via ingestion of soil and sludge adhering to vegetal are critical with regard to human exposure, and these transfer processes are the principal pathways influencing the human diet.

Subsurface injection of sludge rather than surface spraying wo uld clearly reduce the potential entry of PCDD/Fs into the human food chain.

<u>DEHP (Di-2-ethylexyl-phthalate)</u>: DEHP belongs to the esters of phthalates, which are all esters of the phthalic acid. It accounts for over half of the total use of phthalates and is also the most well studied of these compounds. DEHP may be used as a plasticizer, with application in the construction and packaging industries (i.e., in the production of PVC), as well as in the production of components of medical devices.

Studies have shown testicular atrophy and neoplastic effects on the liver in rats and mice. They can also induce a teratogenic effect, even if they do not themselves have a teratogenic effect. Actually, only few studies are available for toxicity on humans. They concluded that there is a need to reduce exposure arising from the use of plastic devices.

DEHP-like compounds originate from effluents of the plastic industry and from compounds in plastic matter, which can be transferred in wastewater. DEHP and phthalate-content in sludge is between 20 and 660 mg/kg DM in EU countries.

Uptake of DEHP by plants is not very well documented. When considering the available data, uptake of DEHP by plants appears to be low because of its rapid biodegradation in soil. DEHP seems to be ingested primarily with concentrated feed rather than with grass and hay or soil. Because DEHP is contained in concentrated food, animal kept indoors also ingest the substance. Exposure to phthalates can also occur via drinking water.

It is assumed that the main exposure route for terrestrial organisms to DEHP is the water phase. Based on the very limited data available, it would seem that uptake into plants is small and that DEHP is not accumulated in animal tissues. If this is the case, distribution of sludge ought to contribute very little to human exposure to DEHP.

<u>NPE (Nonylphenol and Nonylphenolethoxylates)</u>: NPE are surface-active agents used in detergents and washing powders. Nonylphenol is formed in wastewater treatment plants when incoming nonylphenolethoxylates are converted during digestion of the sludge.

Available data is insufficient, but tensioactives are not highly toxic as such. However, their metabolites after degradation are often more toxic and harder to degrade. Several studies show that NP and other alkylphenols have estrogenic effects both in vitro and in vivo.

The main origin of those compounds in sludge is the daily and industrial use of detergents. Their metabolites can appear in the sludge during its biological evolution. They concentrate in sludge but undergo rapid biodegradation under aerobic conditions.

NP does not seem to be taken up by plants. Several studies concluded that internal concentrations in spring barley grains were independent of whether or not the soil was contaminated with NP (EPA, 1995), or that no increase in the NP content of grains was observed following sludge application. However, not enough data is available.

Little information has been published concerning NP/NPE uptake by livestock. The amount of NP originating from sewage sludge and other sources and consumed with fodder by meat and dairy stock seems very small. The main source of NP/NPE for livestock seems to be drinking water, which most likely is not polluted with NP/NPE originating from sludge. Judging from the little data available, NP does not accumulate in animal tissue and is eliminated rapidly.

Concerning NP/NPE, it seems that the main exposure route is the consumption of drinking water, which is not likely to be affected by the agricultural use of sewage sludge. Keeping in mind that nonylphenol is rapidly degraded in soil and that uptake by plants is assumed to be low, the contribution of the agricultural use of sewage sludge may be considered as very low.

<u>LAS (Linear alkylbenzenesulphonates)</u>: LAS and NPE are surface-active agents used in detergents and washing powders. LAS is easily degraded under aerobic conditions. Actually available data is insufficient, but tensioactives are not highly toxic, as such. However, their metabolites after degradation are often more toxic and harder to degrade. There is currently a lack of knowledge concerning the toxicological effects and definition of these metabolites. LAS can cause allergies and have effects on skin pigmentation. Some metabolites, such as aliphatic amines, can have effects on the liver, kidneys and heart, but are not carcinogenic.

The main origin of these compounds in sludge is the daily and industrial use of detergents. Their metabolites can appear in sludge during its biological evolution. They concentrate in sludge but undergo rapid biodegradation under aerobic conditions. Due to these chemical properties, LAS levels are much higher in anaerobic digested sludge than in aerobic digested sludge.

The substance may be expected to be taken up and transported in plants. An experiment on potential accumulation of LAS in potatoes, cabbage, leeks and carrots showed, however, that the concentration of LAS in plant tissues was below the analytical limits of detection.

Data is lacking concerning LAS transfer to livestock.

Very little data is available on LAS transfer ability. LAS are not persistent in soil and are quickly degraded. Plant uptake is assumed to be low. The safety margins appear to be more than adequate to protect terrestrial plants and animals from harm by using LAS during irrigation with secondary sewage sludge elements, or upon soil fertilization with sewage sludge.

Table 2.10 summarizes the properties, occurrence, fate and transfer of the principal organic contaminants in sewage sludge and soil.

3. ALTERNATIVE SEWAGE SLUDGE MANAGEMENT PRACTISES

3.1 Overview of methods and trends

Although manure has been used as a fertilizer and soil conditioner for centuries, use of sludge for the same purposes is a fairly recent development, to a large extent, being the result of a wide scale application of sewage treatment facilities. Safe sludge disposal to sanitary landfills has been extensively used as an elimination method. However, recently in the context of sustainable development, it has been recognized that sludge can be a valuable material which can be reused.

Within the European Union, sludge elimination through landfilling is discouraged and sea disposal has not been allowed since 1999 (EC Directive 91/271). According to EU Directive 1999/31 on waste landfilling, disposal of organic material through landfilling should be minimized and sludge disposal should be reduced by 45%. Inevitably, alternative outlets and/or sophisticated treatment are required to meet European objectives. The challenge of the next 10-20 years will be to identify the sustainable balance between sludge production, recycling and disposal, and the protection of human and environmental health in an affordable, cost-effective and sustainable manner. According to the European Union management policy, reuse and recycling is urged. In the US, sludge reuse is also favoured and is currently applied to approximately 35% of the total amount of dry solids produced (EPA, 1995), in the form of reuse in agriculture (79%, from which 12% is available in bags), forest land (3%), reclamation sites (9%) or public contact sites (12%). Recently, the term “biosolids” has been widely adopted to replace “sewage sludge”, in order to make it more attractive to the public and potential customers.

Alternative methods for sludge utilization and disposal are shown in Figure 3.1. Stabilized (aerobically or anaerobically) and dewatered sludge is, in most cases, the typical by-product of wastewater treatment facilities. This sludge can be used in agriculture, though with some restrictions, and in combination with appropriate operational procedures. Incineration, often with energy recovery, is an alternative option although expensive and not very popular due to concerns related to the possibility of air pollution. Disposal of the produced ash is usually performed by landfilling of the ash, although melting the ash and production of slag which can be subsequently used as a building material, is a promising alternative. Pathogen elimination by subjecting sludge to additional treatment such as composting, lime treatment, drying, pasteurization or other suitable methods produces a sanitized sludge that can be widely used in an unrestricted faction. Drying of sludge offers an additional alternative use as fuel in cement industries or power plants. Partially dried sludge (50% solids content) can also be used as covering material in landfills. As already mentioned, sludge disposal into the sea has not been allowed since 1999 and for the Mediterranean region, the Protocol of the Barcelona convention regarding the prevention and elimination of pollution of the Mediterranean Sea by dumping of ships was adopted in 1995, although the protocol is still not legally binding.

Table 2.10

Summary of properties, occurrence, fate and transfer of the principal organic contaminant groups found in sewage sludge and sludge treated soils (Smith, 1996)

Compound Group	Physico-chemical properties	Concentration in sludge	Degradation	Leaching potential	Plant uptake	Transfer to animals
Polycyclic aromatic hydrocarbons (PAH)	Water soluble/volatile to lipophilic	1-10 mg kg^{-1}	Weeks⇒10 years Strongly adsorbed by soil O.M.	None	Very poor Foliar absorption	Possible but rapidly metabolized Not accumulated
Phthalate acid esters	Generally lipophilic, hydrophobic and non-volatile	High 1-100 mg kg^{-1}	Rapid Half-life <50 days	None	Root retention Not translocated	Very limited
Linear alkylbenzene-sulphonates (LAS)	Lipophilic	Very high 50-15000 mg kg^{-1}	Very rapid in aerobic environment	None	None	None
Alkylphenols Polychlorinated biphenyls (PCBs)	Lipophilic Complex, > 200 congeners low water solubility, highly lipophilic and semi-volatile	100-3000 mg kg^{-1} 1-20 mg kg^{-1}	Rapid < 10 days Very persistent Half-life several years Strongly adsorbed by soil O.M.	None None	Minimal Root retention Foliar absorption Minimal root uptake and translocation	Minimal Possible into milk/tissues via soil ingestion Long half-life
Polychloro-dibenzo-dioxins/furans (PCDD/Fs)	Complex, 75 PCDD congeners, 135 PCDF congeners, Low water solubility, highly lipophilic and semi-volatile	Very low <few µg kg^{-1}	Very persistent Half life several years Strongly adsorbed by soil O.M.	None	Root retention Foliar absorption Minimal root Uptake and translocation	Possible into milk/tissues Via soil ingestion Long half-life
Organochlorines pesticides	Varied, lipophilic to hydrophilic, some volatile	<Few mg kg^{-1}	Slow> 1 year Loss by volatilization	None	Root retention Translocation not important Foliar absorption	Via soil ingestion persistent in tissues
Monocyclic aromatics	Water soluble and volatile	<1-10 mg kg^{-1}	Rapid	Moderate	Limited due to low persistence Rapidly metabolized	Rapidly metabolized
Chlorobenzenes	Water soluble/volatile to lipophilic	<0.1-50 mg kg^{-1}	Lower mol wt types lost by volatilisation Higher mol wt types persistent	High to low	Possible via roots and foliage Maybe metabolized	Important for persistent compounds
Short-chained halogenated aliphatics	Water soluble and volatile	0-5 mg kg^{-1}	Lower mol wt types lost by volatilisation Higher mol wt types persistent	Moderate	Foliar absorption Possible translocation	Low
Aromatic and alkyl amines	Water soluble and low volatility	0-1 mg kg^{-1}	Slow	High	Possible	Low
Phenol	Varied, lipophilic high water solubility and volatile	0-5 mg kg^{-1}	Rapid	Moderate to low	Possible via roots and foliage	None

O.M. organic matter, mol wt: molecular weight

Due to expanding application of sewage treatment technologies, the volume of the produced sewage sludge increases. A direct evidence of this correlation can be seen in Europe, and more particularly due to the enforcement of the European Environmental Legislation and more precisely of the 91/271/EC Directive concerning urban wastewater treatment. On the basis of the January 1999 Commission Report concerning the implementation of the 91/271/EC Directive, it was anticipated that a 50% increase in the production of sludge (in terms of tonnes of dry matter) would occur over the period of implementation of the Directive 1992 - 2005.

More specifically, the sludge produced from the then 15 Member States would increase from 5.5 million tonnes of dry matter in 1992 to 8.3 million tonnes of dry matter in 2005 (Figures 3.2 - 3.3).

The elimination of surface water disposal and decline in landfilling is anticipated from about 40% to 20%. The remaining 80% is to be managed either by landspreading (about 55%) or by incineration, which is expected to gain in significance (from about 14% in 1992 to about 25% in 2005). Examination of selected Member States reveals significant differences and problems with convergence. Thus, in Greece landfilling is predicted to remain almost the sole method of disposal. In Denmark an almost steady state condition has been achieved both in terms of quantities and methods, with landspreading representing the method adopted for 55-60% of the sludge and incineration for about 25%.

In Germany and the UK landspreading is expected to remain dominant with an increase from about 45% to about 50% and 65% respectively. In the UK surface disposal is to be eliminated and in Germany landfilling is expected to decline from 35% to 15%. An increasingly significant portion of the sludge is to be incinerated reaching figures to the order of 25% for the UK and 35% for Germany (form 10% and 15% in 1992, respectively). Incineration will become the dominant method in the Netherlands, applied to almost 60% of the sludge, while landspreading and landfilling will decline to 25% and 15%, respectively from 40% and 55% in 1992. The Netherlands represent an exceptional case with respect to landspreading which is expected to decline, for reasons which are associated with the production of excessive amounts of manure.

In both France and Spain, landspreading is to gain in significance and is expected to be applied to almost 55-60% of the produced sludge. However, unlike France where incineration is to be the complimentary method of almost equal importance (for about 45% of the sludge), in Spain, due, amongst other things to strong public reaction against incineration, will later be very limited (if practised at all) and landfilling will remain a significant method (for about 30% of the sludge).

3.2 Landfilling

So far, landfilling has been a major route for sludge disposal. However, it should, at least in the European Union, be a limited outlet in the future because of the European legislation on the landfilling of waste (1999/31/EC) which states that “Member States shall set up a national strategy for the implementation of the reduction of biodegradable waste going to landfills” no later than 16.07.2003.

Landfilling as a method of sludge disposal, should be chosen only in cases where land spreading or other methods of recycling is not feasible (due to heavy concentrations of contaminants in the sludge or unacceptable costs on the basis of local topographical considerations) and no incineration capacity is available on or near the site. It should be noted that landfilling of residuals (ashes or non-recyclable inert material) is considered as an acceptable method for the future.

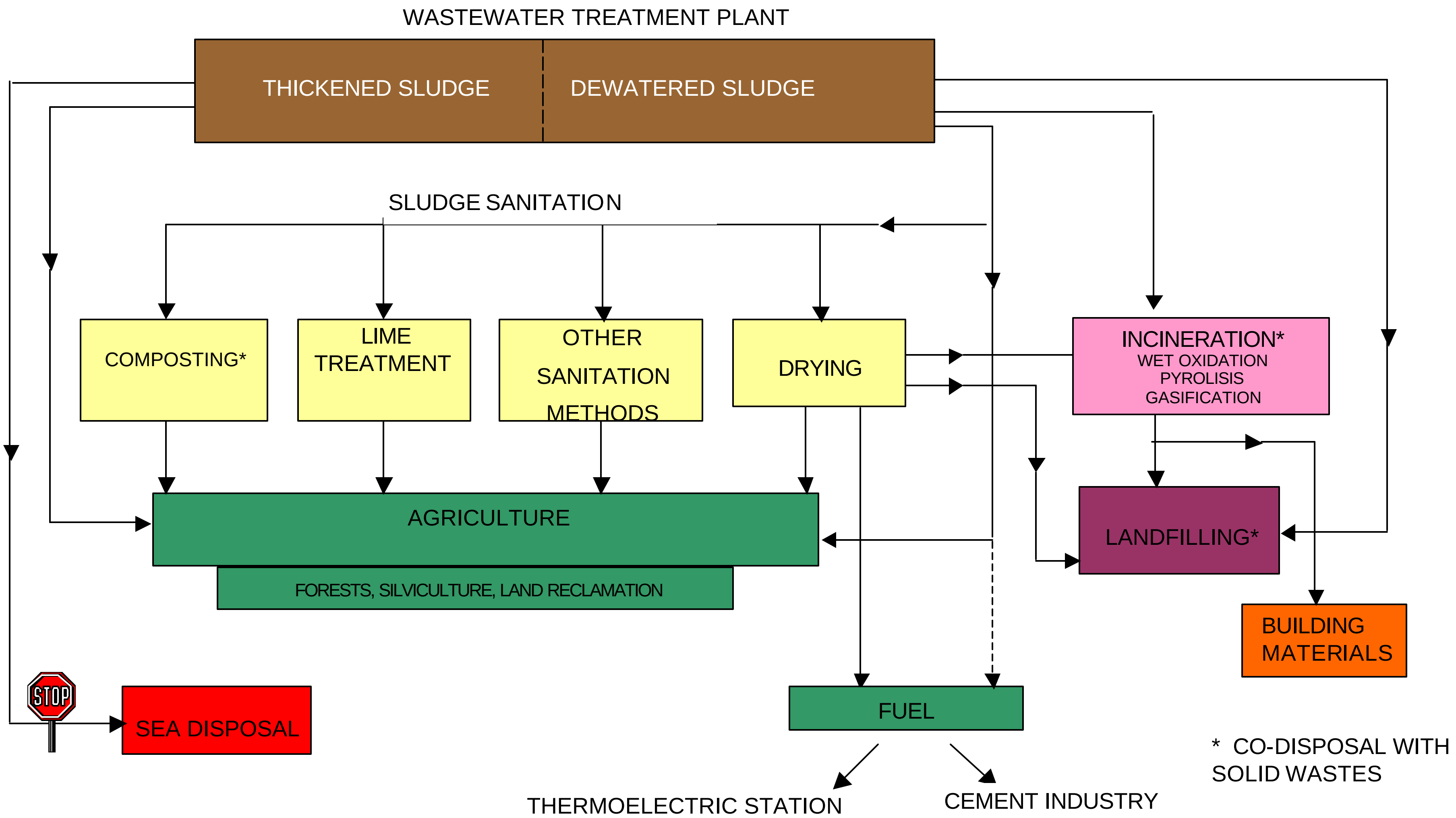

Figure 3.1. Alternative methods of treatment and disposal of sludge (NTUA, 2000).

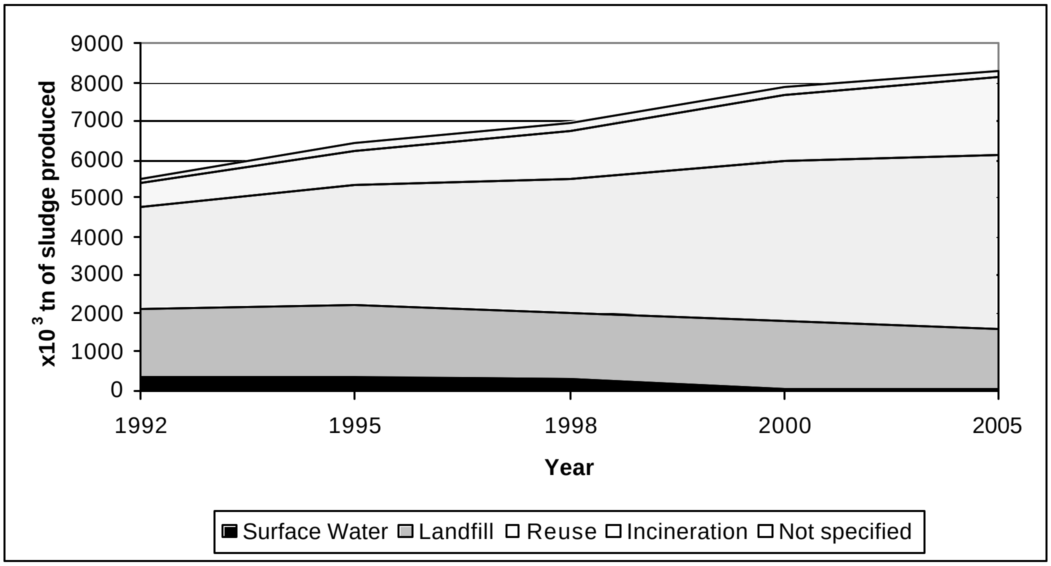

Figure 3.2. Management method of sludge adopted within the EU in relation to time.

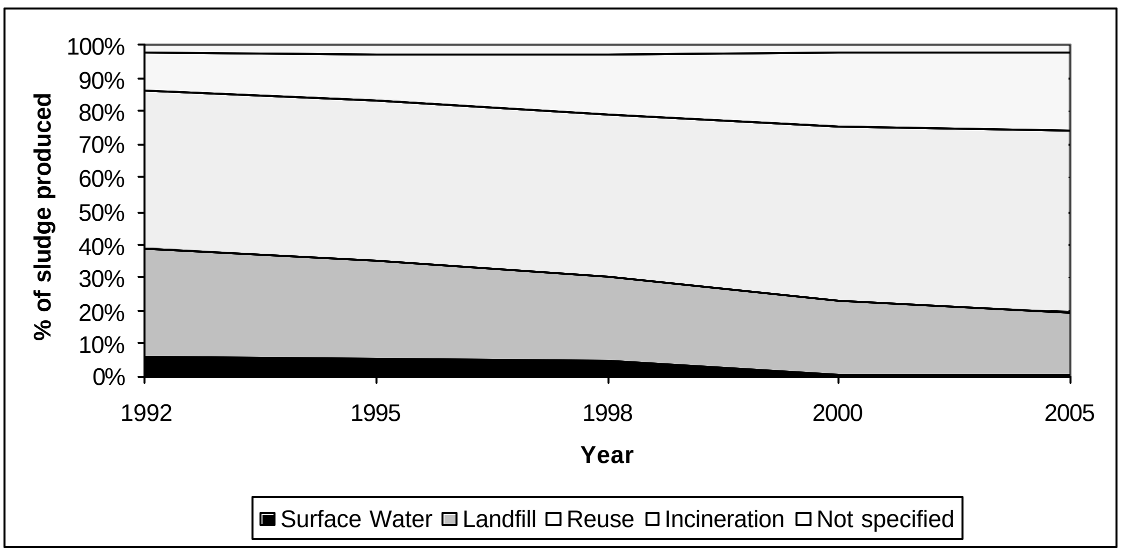

Figure 3.3. Percentile of sludge management alternative (Source :EC Report, 2001).

In a landfill, the following main processes take place:

- initial aerobic phase: the degradation first occurs under aerobic conditions, during which aerobic micro-organisms consume the available oxygen in the deposit. This step is rather short (about 14 days). The organic content of the leachate increases;
- acetogenesis: as the level of oxygen decreases, acetogenic and fermentative bacteria decompose the easily degradable material of the waste. The pH value decreases in the deposit, consequently increasing the solubility of inorganic substances, such as heavy metals. High organic pollution of the leachate is observed; and

- anaerobic methanogenesis: methanogenic bacteria proliferate during this phase, increasing the production of methane. The pH value increases, and the organic content of the leachate decreases. The gas production then reaches a stable composition.

There are two possibilities for landfilling sludge: mono-landfilling, where the landfill is only used for sludge, and mixed-landfilling, when the landfill is also used for municipal wastes. Mixed-landfilling is the most frequently used method, as some of the constraints of mono-landfilling are removed (especially with respect to the moisture content of sludge). In fact, addition of sludge may have a positive effect on the microbial degradation of solid wastes (Vesilind 1979, Pohland 1996, Christoulas et.al 1994), as biological processes are enhanced due to the presence of micro-organisms and the increase of moisture.

When processing a mono-landfilling, the compacted structure of the deposit in the cell is not favourable for gas formation. However, should this happen, its composition would not be very different from usual municipal wastes deposit: 50 to 60% methane, 40 to 50% carbon dioxide, plus trace elements.

3.2.1 Anticipated impacts

Operation of a landfill has several impacts, usually associated with emissions to air, water and soil.

<u>Emissions to air</u>: emissions to air are the release of landfill gas, when this is not recovered on site for energy generation, and dust released during the handling of waste. Other emissions originate from the exhaust gas of engines used on site.

The generation of landfill gas is to the order of 10 m^3 per ton of deposited waste per annum, but may vary according to size, input rate and the characteristics of the disposed wastes. The main components of the landfill gas are methane (between 50 and 60%) and carbon dioxide (between 40 and 50%).

Many other VOCs have been reported as traces, accounting in general for less than 1% of the volume of gas generated. The EEA (2000) reported the presence of 12 halogenated hydrocarbons and about 30 hydrocarbons in landfill gas, with levels ranging from 0,02 mg/m^3 to over 600 mg/m^3. The amount of VOCs released into the atmosphere is lower when landfill gas is used or flared. In this case, however, dioxins may be generated. Volatile substances migrate between the landfill and the atmosphere due to diffusion and pressure difference. VOCs originate from waste, but new substances are also generated by the chemical and biochemical transformations occurring in the deposit.

Carbon dioxide and methane have impacts on the climate, and trace compounds may be toxic and/or carcinogenic, with varying threshold values.

<u>Emissions to water and soil - leachate management</u>: leachate generated within a landfill is emitted to soil and water and the amount generated depends on the climatic conditions and landfill cover. It contains several compounds, such as ions (Ca^{2+}, K^+, Na^+, NH_4^+, CO_3^{2-}, SO_4^{2-}, Cl^-), heavy metals, organic compounds (chlorinated organics, phenol, benzene, pesticides) and micro-organisms. It can also contain dissolved methane, which is present in the landfill gas. Emissions may be reduced when leachate is collected onsite and treated. Leachate may also leach through the soil to the groundwater, or be directly released in surface water, and have impacts on human health and ecosystems. Leaching not only depends on the physical and chemical properties of the compound, but also on the soil properties and environmental factors.

According to Pohland (1996), heavy metals are mobile during the first stage of digestion where leachate pH is reduced, while in the following anaerobic phase the redox potential reduces, resulting in the reduction of SO_4 to SO_2 and the formation of metal-sulfides insoluble compounds. These are the conditions during oxygenises and methanogenesis. The formation and precipitation of insoluble compounds of sulfur, cadmium, nickel, zinc and lead, result in the significant reduction of the metal concentrations in the leachate to undetected levels.

This process of reducing heavy metals is enhanced by recirculation. Thus, with recirculation and appropriate handling of the gas produced, the reduction capacity of the bioreactors of mixed-landfilling is equally effective for toxic organics (Pohland, 1996). This is due to the high hydraulic retention time which enables the acclimatization of the micro-organisms, in relation to the recirculation which allows the contact of biomass with the organics. The result is the bioremediation by redox auto-halogenation. Additionally, with leachate recirculation the moisture content in a landfill stabilizes at levels satisfactory for the biological processes (Christoulas *et al*., 1994, Pohland 1996).

In addition to, or instead of recirculation, treatment may be applied to the collected leachates. This is usually performed by *in-situ* aerobic biological processes which, when properly constructed and operated, can significantly reduce the degradable organic and nitrogen content of the leachate. Alternatively the produced leachate may be transported and treated in a centralized wastewater treatment plant in the vicinity of the landfill.

<u>Other impacts</u>: with proper leachate management (recirculation, collection, treatment), effective and frequent inspection of the watertightness for the protection of groundwater and, finally, collection and appropriate handling of landfill gases, a modern landfill can be an effective biological reactor with controlled operation. According to several researchers, it can ensure satisfactory environmental protection. However, other impacts have to be considered such as:

- noise and dust from the delivery vehicles;
- odours;
- vermin, rats and birds; and
- land use, disturbance of vegetation and landscape.

<u>Sludge treatment requirements</u>: liquid sludge is not an acceptable form for landfilling and methods aiming at reduction of the volume of water abstraction are a prerequisite. Such methods, discussed in detail in Part 6, include thickening (gravity or mechanical) and dewatering. Typically, a minimum so lids content of about 20% is needed prior to landfilling. It is also desirable to reduce the easily degradable organic material of the sludge by stabilization (aerobic or anaerobic as discussed in Part 6).

Thickening, stabilization and dewatering are theusual processes employed for sludge treatment in a conventional wastewater treatment plant, and the typical sludge produced from such treatment plants is suitable for subsequent landfilling. Additional processes aiming at further volume reduction (such as drying, which can achieve a volume reduction to about one third of the volume of the dewatered sludge), involve significant additional costs which, in most cases, can not be justified (with the possible exception in situations characterized by long distance transportation in combination with landfill capacity restrictions).

An issue of concern is the impact of landfills on the health of people living in the neighbourhood of such an installation. Although several studies have been conducted which have not revealed the identification of any sources of exposure or links to the cause, public awareness remains. These concerns and the extensive land areas required, especially when available sites are limited, very often lead to a strong NIMBY (Not in My Back Yard) attitude and opposition to the creation of new landfills. It should also be stressed that due to lack of utilization of useful sludge components, landfilling is not considered as a sustainable method, a consideration reflected in the legislation of the European Union.

3.3 Landspreading

Sewage sludge is of significant agronomic value due to its organic, nitrogen and phosphorus content and can, therefore, be used as a partial substitute for chemical fertilizers and as a general soil conditioner. The addition of sludge to soil increases permeability, humidity, retainability and stability. All sludge types (liquid, semi-solid, solid or dried sludge) can be spread on land. However, the use of each of them induces practical constraints on storage, transport and spreading itself.

Recycling of sludge to agricultural land is probably the most attractive landspreading option, although use of sewage sludge in forestry and silviculture, as well as for land reclamation purposes, are emerging alternatives.

3.3.1 Agricultural application

Sewage sludge contains compounds of agricultural significance, such as nitrogen, phosphorus, potassium, organic matter or calcium, making its use relevant as an organic fertilizer. Moreover, the cost of this route may be cheaper than other disposal routes. However, the presence of pollutants in sludge implies that the practise should be carefully carried out and monitored. For this purpose, in some countries codes of practise and spreading schemes have been established, summarizing the regulatory obligations. Periods for spreading, types of culture and adequate record-keeping are prescribed for management of sanitary and environmental risks. A more detailed discussion of the regulations and codes of practises is presented in Part 4.

Sludge can be applied to the fields by using a trailer tank or umbilical delivery system and may be applied by surface spreading or direct injection into the soil. However, it is important to reduce the formation of aerosols to reduce the risk of odour nuisance. Dried sludge may be supplied by using the same equipment as for solid mineral fertilizers. The spreading equipment has also to be adapted to the type of sludge. Transportation of sludge to the farmland is usually the most expensive aspect of the whole process.

The application of sludge on farmland is seasonal. It can usually be performed twice a year: at the end of summer after harvesting, or in spring before ploughing and sowing. However, sludge production from a wastewater treatment plant is more or less constant throughout the year. Therefore, storage capacity must be available at the wastewater treatment plant or on the farm, either separately or in combination with animal slurry, when national regulations permit. Average storage duration is about 6 months.

Liquid sludge may be stored in concrete tanks (mostly for small wastewater treatment plants) or lagoons. It can be pumped for transportation. Semi–solid sludge may be stored on a platform, which must be waterproof, or in tanks. Sludge pits may also be found. In the majority of cases this type of sludge cannot be pumped and has, therefore, to be conveyed using specific hauling equipment, such as grabs. Odours may arise when sludge is handled for conveyance.

The structure of solid sludge enables storage in piles. Handling implies the use of a crane or a tractor. Dried sludge does not present any specific constraints. However, in this case storage must be monitored in order to prevent any explosion and emission of particles into the air.

As already mentioned sewage sludge contains many compounds of agricultural significance but may also contain pollutants which are potentially harmful to the crops themselves (phytotoxicity), or to animals and humans who consume the crops. The main restrictions on the application of sludge for agricultural purposes are related to heavy metals, organic pollutants and pathogens. Nuisance caused by odour or vector attraction should also be considered.

Heavy Metals: the heavy metal content in sludge depends on the initial quality of the influent sewage to the treatment plant. Whenever industrial wastewaters are treated jointly with municipal sewage, the waste may contain heavy metals in significant concentrations. These metals accumulate in the sludge with the possibility to render it unsuitable for use in agriculture. Most regulations and guidelines make explicit reference to heavy metal, limiting concentrations both in sludge and the receiving soil, as well as to acceptable loading rates (Part 4). In order to avoid the presence of heavy metals in sludge, industrial wastewater influents must be controlled at the source.

Organic Pollutants: there is a large number of potentially toxic micro-organics in sewage sludge, but very little is quantitatively known regarding their fate in the environment and their dose-effects relationship. Considering that acceptability and a sense of safety are prerequisite to extensive reuse application, it is important that attention is paid in the future to organic compounds which might be hazardous to humans or ecosystems when spreading sewage sludge on agricultural land. Organic pollutants, such as PAH, PCB and organochlorine pesticides have been identified as priority substances by some existing or proposed regulations (Part 4).

Pathogens: treatment of sewage sludge to produce a stabilized product is necessary in order to reduce the content of easily degradable organic material, the concentration of pathogens and make sludge more acceptable to farmers. The occasional negative perception of sludge as a soil conditioner/fertilizer is often connected to the presence of pathogens. Their reduction or elimination prior to land application is important to protect operating personnel, the general public, crops intended for human consumption, groundwater and surface water from potential contamination.

Practical elimination of pathogens through proper disinfection technologies to produce sanitized sludge is the safest approach, with the additional benefit of enabling a wider scope for unrestricted sludge reuse possibilities. The cost associated with such treatment may be unacceptably high in some cases. Furthermore, it has been demonstrated that protection of personnel, animals and humans may be achieved through a proper combination of more modest treatment schemes and restrictions with respect to modes of reuse operation. This is reflected in almost all existing regulation and guidelines, which allow restricted reuse (subject to crop selection, land application methods, etc.) of sludge which is not sanitized. However, a minimum reduction of pathogens, which can be achieved though conventional sludge treatment methods (i.e., aerobic or anaerobic stabilization), is normally required.

Odour: sludge odour control is a major consideration because fermentative odours are considered as a nuisance. Strong odours may be produced in cases where non-stabilized sludge is spread. These problems are greater in tourist or highly populated sites and can be reduced with the application of stabilized sludge.

3.3.2 Forestry, silviculture and land reclamation

Forestry and silviculture refer to different kinds of tree plantation and use. The term "forestry" is mainly used when considering amenity forests, or mature forest exploitation. Silviculture, on the other hand, is more specifically used when referring to intensive production, such as energy coppices or poplar plantation.

The application of sewage sludge to forestlands is not yet a common practise but is not only a feasible, but also an attractive option since it offers a disposal route other than to a food or animal fodder crop and thus, hygienic and odour issues are less restrictive. Relatively small areas could permit the spreading of an important part of the sludge production.

Sludge application may be performed at different times during tree growth. Landspreading may be carried out prior to plantation, re-afforestation or plantation of an intensive culture. In silviculture, sludge application may also be performed just after sowing or after each cut. In forests, sludge application can occur practically all year round, in accordance with good practises and local conditions.

From an economical point of view, this route would be attractive if areas were available for spreading in the neighbourhood of the wastewater treatment plant, in order to reduce transportation costs and related pollution. An advantage of forestland utilization is that the municipalities do not usually have to pay for land acquisition. However, it may be difficult to control public access to sludge-amended forestlands, since people are accustomed to free access to forested areas and tend to ignore signposts, fences, etc. Control of public access is needed for up to 12 months after liquid sludge is sprayed on forested areas.

From the agricultural and environmental point of view, even if similarities between landspreading and use in forestry are be observed concerning environmental impacts, great differences exist due, amongst other factors, to the specificity of the species grown, the fauna and flora involved, and the soil types. The issue of sludge recycling to forestry and silviculture has not been addressed to the same extent as its recycling to agricultural land, and much less information is available in literature concerning this outlet.

The use of sewage sludge in land reclamation and revegetation is aimed at restoring derelict land or protecting the soil from erosion, depending on the previous use of the site. Many of the problems of disturbed areas are related to structural damage caused by soil stripping, storing and replacing, and which result in compaction, waterlogging and nutrient-deficient soils. In the case of industrial sites, topsoil may often be absent, and when present, damaged by storage or handling. Soil or soil-forming materials onsite may be deficient in nutrients and organic matter. Other problems may exist, such as toxicity or adverse pH. All these problems create a hostile environment for the development of vegetation (WRc 1999).

Possible solutions include the use of inorganic fertilizers or imported topsoil, which can be very expensive depending on location and availability. An alternative solution is the use of organic wastes such as sewage sludge. There have been a number of successful land reclamation projects involving the use of sludge or sludge compost. Most have been conducted on strip-mined land or mine tailings. This option may be extremely attractive in areas where disturbed and marginal lands exist because of the dual benefit to the municipality in disposing of its sludge, and to the environment through reclamation of practically deserted and useless land areas.

Sludge has several characteristics, which makes it suitable for reclaiming and improving disturbed lands and marginal soils. One of the most important is sludge organic matter which:

- improves soil physical properties by improving granulation, reducing plasticity and cohesion and increasing water-holding capacity;
- increases the soil cation exchange capacity;
- increases and buffers soil pH; and
- supplies plant nutrients.

The natural buffering capacity and pH of most sludge will improve the acidic or moderately alkaline conditions found in many mine soils. Immobilization of heavy metals is pH-dependent, so sludge application reduces the potential for acidic, metal-laden runoff and/or leachates. Sludge is also desirable because the nutrients it contains may substantially reduce commercial fertilizer needs. Furthermore, sludge helps to increase the number and activity of soil micro-organisms.

Disturbed areas, especially old mining sites, often have irregular, excessively eroded terrain. Extensive grading and other site preparation steps may be necessary to prepare the site for sludge application. Similarly, disturbed areas often have irregular patterns of soil characteristics. This may cause difficulties in sludge application, revegetation and future site monitoring.

Plant species selected for use in revegetation should be carefully selected for their tolerance to sludge constituents and their suitability to local soil and climatic conditions. If crops intended for animal or human consumption are planted, the same limitations exist as apply to agricultural utilization of sludge.

In cases where sewage sludge is applied to parks, public gardens, landscaping for industrial, commercial and residential developments, sports fields, recreational areas, etc., consideration should be given to the protection of public health. Thus, sludge must be hygienically safe, must not create odour nuisance, and must have a soil-like consistency. These properties correspond to a high degree stabilized sludge (stored digested - dewatered sludge or composted).

3.3.3 Sludge treatment requirements - sanitization

Although sludge in a liquid form may be directly used for agricultural landspreading, transportation and environmental constraints (costs, runoff potential) must be seriously taken into consideration.

In Europe a stabilized and dewatered sludge is normally required prior to agricultural landspreading, as is the case for landfilling. Storage (typically, for at least 6 months) is the only additional measure required in combination with certain restrictions regarding the uses of the agricultural land, the rates of application and heavy metal concentration of both sludge and soil (discussed in Part 4). Stabilization followed by storage of sludge is considered as an adequate combination for a satisfactory reduction (although not elimination) of pathogens.

Unrestricted landspreading, including use of sludge for forestry, silviculture and land reclamation, requires additional considerations related to the sensitivity of the fauna and flora involved and/or the issue of public health. Additional measures may involve stricter limitations with respect to heavy metals and other toxic compounds and the need to practise elimination of pathogens in the sludge.

As previously mentioned, landspreading of sludge does not necessarily require elimination of pathogens, although a minimum reduction of pathogens which can be achieved though conventional sludge treatment methods, is normally required. In this case, several restrictions with respect to modes of reuse and operation are necessary. Typical processes for partial removal of pathogens include psychrophilic (20°C) and mesophilic (35°C) anaerobic digestion and psychrophilic (20°C) aerobic digestion. Long-term storage of sludge (over one year) may, under favourable conditions, produce a practically sanitized sludge; under the typical conditions of six months storage only partial destruction of pathogens may be expected.

However, practical elimination of pathogens through proper disinfection technologies is an emerging new concept. The additional cost associated with such treatment may, in some cases, be offset by an improved perception by farmers, more flexible modes of operation (e.g., shorter storage periods and fewer restrictions regarding animal grazing) and wider in scope application (e.g., forest silviculture and areas frequently visited by the public).

The main sludge disinfection technologies can be summarized as follows.

<u>Thermophilic anaerobic digestion</u>: thermophilic digestion occurs at temperatures from 50 to 57°C, conditions that enhance the growth of thermophilic bacteria. Thermophilic digestion of sludge achieves, at the same solids retention time, approximately two to four logs greater removal of viruses and bacteria compared to mesophilic digestion (USEPA 1979; Watanabe *et al.*, 1997). Based on field scale and bench scale anaerobic digestion experiments concluded in the mesophilic and thermophilic range Watanabe *et al.*, (1997) concluded that mesophilic anaerobic digestion could not achieve a pathogen reduction level to attain the EPA Class A requirements. Only thermophilic digestion could achieve low enough concentrations of faecal coliforms and *Salmonella* to satisfy the EPA Class A requirements.

<u>Thermophilic aerobic digestion</u>: during aerobic thermophilic digestion, heat is generated when aerobic bacteria degrade the organic matter of the sludge. In adequate conditions, the temperature can rise to over 70°C. By subjecting sludge to these high temperatures for a particular period of time, most harmful organisms are destroyed. It is usual to subject the sludge to a temperature of 50 to 65°C for five to six days. In these conditions, volatile matter is reduced by about 40%. The process is simple in design but has a high energy cost: 5 to 10 times more than anaerobic digestion.

Composting: composting is an aerobic process consisting of aerating sludge mixed with a co-product, such as sawdust or animal manure. Composting produces excess heat which can be used to raise the temperature of the composting mass. The mix then evolves for several weeks. If composting process is sufficiently completed the end product is fully stabilized, has a very low potential for odour generation and may be reused in agriculture as a soil conditioner. The most important feature of the process is heat generation during decomposition of organic material. Approximately 20-30% of the sludge volatile content is converted to carbon dioxide and water with the immediate release of sufficient heat to raise the temperature to high enough levels (50°C – 70°C) to destroy weed seeds and pathogenic micro-organisms.

Composting is one of the sanitation technologies that when properly operated, can achieve a minimum four logs reduction in *salmonella* concentration, inactivate *Ascaris* eggs and render them non-infectious, and produce an end product that contains less than 1,000 FC/g solids and no *salmonella* per g of solids.

<u>Lime treatment</u>: lime treatment consists of the addition of lime to sludge in order to raise its pH to 12, thus destroying or inhibiting the biomass responsible for the degradation of the organic compounds. The treatment also helps to disinfect the sludge, increasing its dry matter content and making handling easier. The dry mass increase depends on the initial dry matter content and the amount of lime supplied. It is usually recommended to add 30% of

lime to the dry mass of sludge, otherwise the treatment would not avoid fermentation in the long-term. Lime can be added either as slaked lime or as quicklime.

Employment of slaked lime is used for sanitation of liquid sludge before its use or for conditioning of sludge before dewatering. In both cases the addition of lime results in an increase of the pH as a function of the amount of lime added and the properties of the sludge. The wet addition of lime as lime milk should be given preference to lime powder, because of the better mixing and sanitizing effect. The limitations of using this type of lime are the low activity against parasite eggs and oocysts. These limitations can be compensated by storing the treated sludge for at least three months. Generally the effect depends on the dry matter content of the sludge.

Besides high pH values, Quicklime also initiates high temperatures by exothermic reactions. By this combination of pH (>12) and high temperatures between 60ºC and 70ºC, *Ascaris* eggs are also destroyed within 24 h (Andreadakis, 2000). The minimum requirements for successful application of quicklime is a treatment by which a pH-value of at least 12 and a temperature of at least 55ºC is kept for at least 2 h and the treated lime must be stored for at least 24 h before further utilization.

Lime is by far the most favoured chemical used for sludge disinfection, although other chemicals such as peracetic acid, sodium hydroxide, formaldehyde and nitrite have been reported as suitable for disinfection purposes.

<u>Pasteurization</u>: pasteurization involves heating the sludge at 65°C for 30 min, followed by cooling and anaerobic digestion. Initially, sludge was pasteurized after mesophilic digestion (post-pasteurization), but international experience showed a high degree of re-infection. Therefore, this technique was regarded as an unreliable disinfection method and finally abandoned.

A break-up of larger particles prior to the pasteurization process is necessary. To ensure that all sludge particles are exposed to the reaction temperature and time, their size may not exceed 5 mm.

Heat pasteurization is a highly effective process in reducing enteric bacteria and inactivating parasitic ova and cysts (Havelaar, 1983), but it may not be cost-effective for small plants with capacities of less than 17000 m^3/d because of high capital costs. Other temperature and heat combinations are: 1) 70°C for 25 min; 2) 75°C for 20 min; 3) 80°C for 10 min; and 4) 55°C for three hours.

<u>High energy irradiation</u>: irradiation is not good for the public but is, in principle, effective in destroying pathogens in sewage sludge (Bohm, 1999). There are very few plants using irradiation to disinfect sludge. According to the German Sludge Ordinance) Strauch 1987), the required radiation doses are 500 krad for liquid sludge and 1000 krad for dewatered sludge. At these doses, complete kill of gram negative bacteria and *Ascaris* ova is achieved (Havelaar, 1983). However, little or no removal is expected for gram positive bacteria. Viruses are removed by 1 - 2 logs.

<u>Drying</u>: thermal drying of wastewater sludge involves the application of heat to evaporate water from sludge. When thermal drying is used as the final sludge treatment process it can achieve a sludge moisture content usually below 10%, thereby significantly reducing the volume and mass of sludge that has to be handled and disposed of. The advantages of the process include reduced transportation costs, improved storage capability and marketability as well as pathogen destruction. Thermally dried sludge with dry solids

content greater than 90% is considered pasteurized and in most cases can be handled with safety in terms of pathogens transmission.

3.4 Incineration and relevant technologies

Sewage sludge incineration can be considered as a valid alternative when other methods of disposal are not applicable, due to the presence of toxic substances (agricultural use) or lack of suitable sites (landfill). The drastic reduction of volume and weight of the wet sludge cake (by approximately 90%), the destruction, reduction or stabilization of toxics and the potential for recovery of energy from waste heat are the main advantages of the process. The main disadvantages are related to the high capital and operating costs, the high maintenance requirements, the need for highly skilled and experienced operators and the potential adverse environmental impacts, particularly those connected to the emission of pollutants to the atmosphere.

Sludge may be incinerated in dedicated incineration plants (mono-incineration) or in combination with household wastes. Sludge may also be used as a fuel in plants whose purpose is the generation of energy or production of material products, such as coal power plants or cement plants. The usual types of furnaces used are multiple hearth furnaces and fluidized beds, although for mono-incineration the latter is usually preferred.

The multiple hearth furnace is a counter-current multistage process where solids flow downward and gases flow upward. Under these conditions, dewatered sludge cake with 20-30% DS will not ignite and can be burned only with auxiliary fuels. Dried cake with dry solids content greater than 30-35%, depending on sludge characteristics and excess air level, is autogenous. Multi-hearth furnaces used for household waste incineration at a moderate cost may be modified in order to receive sludge.

The fluidized bed system consists of a combustion chamber lined with a refractory material, at the base of which a bed of sand is brought to a high temperature and held in suspension by hot air. Sludge is introduced inside or above the bed of sand, and burnt at a temperature of 900°C for a few seconds. Designated incinerators may be installed onsite in a wastewater treatment plant, when handled capacities justify such a cost intensive technique. However, designated incinerators may be shared, and, therefore, burn sludge from different origins. As already mentioned, other possible uses for incineration of sewage sludge are as fuel in coal-fired power plants and cement kilns. When using sludge as a fuel in cement production plants, the maximal sewage sludge feed rate should not be more than 5% of the clinker production capacity.

Full-scale tests have been performed in power plants in Germany, The Netherlands and Belgium. The tests showed that co-combustion had little effect on the emission of gases; a slight increase in the heavy metals content of the ash has been reported.

In all incineration plants flue gas has to be treated to remove acid gases, heavy metals under gaseous and particulate form, and dust. Different treatments are possible for flue gas, such as electrostatic precipitators and bag filters to remove particulate matter, scrubbers and wet processes for acid gases. They may be combined. It is interesting to observe that sludge quality and the fluidized bed incineration technology do not often imply a specific treatment of nitrous oxides and dioxins. The choice of system depends on the emission limits which have to be reached, and the possible recycling of the ashes.

At sludge incineration temperature, dioxins and furans are completely destroyed, so that in the incinerator they are present in negligible concentrations. However, in the flue gas cleaning stages where the gas temperature is below 450°C, new formations of dioxins and furans may take place. It has to be observed that in the case of mono-incineration, the amount of dioxins and NO_X present in the raw flue gas is low enough to avoid the implementation of a specific gas treatment process for these compounds.

Once emitted into the air, pollutants are dispersed in the atmosphere. Their concentration depends on several factors, depending on local conditions (climatic conditions, wind direction, wind speed, distance from the incineration plant) or the physical and chemical properties of the compounds. Atmospheric deposition to the soil can also take place. Lastly, emissions to air may be due to handling of ashes and combustion residues. Emissions to air and especially dust, dioxins, heavy metals, VOC, NOx, CO and SO_2 may have adverse health effects. Those pollutants, as well as CO_2, can also have impacts on ecosystems and climate change. Damages to buildings may also occur, particularly due to particulate matter, NO_X and SO_2.

Water emissions occur due to flue gas treatment, when a wet process is performed. However, water treatment reduces the pollutant content of this wastewater. The pollutants present in this wastewater are mostly the same as those released into the atmosphere with the fumes. Emissions to water may also be caused by leaching of ashes disposed of to landfills. Groundwater as well as surface water is affected by these emissions, which can give rise to adverse health effects and ecotoxicity.

Emissions to soil are due to the disposal of ashes or the flue gas treatment residues to landfill, or the use of ashes in road construction. Bottom or grate ash are largely reused, whilst fly ash and residues from the flue gas cleaning system are generally placed in hazardous waste landfills. They are also the consequence of the atmospheric deposition of the pollutants emitted to the atmosphere. It must be observed that flue gas treatment residues contain much more pollutants than ash.

Disamenity may occur because of the operation of an incineration plant. Among which, noise, dust, odour and visual pollution may be evoked. Operating accidents can also occur, generating an increase in the emissions to air, reducing the energy recovery, and also leading to health impacts on operating personnel.

Several technologies to present an alternative to conventional combustion processes are currently being developed or introduced into the market. These technologies are mainly represented by the wet oxidation process, pyrolysis, and the gasification process. Other technologies may be found that are most often combinations of these three main processes (EC Report, 2001).

<u>Wet oxidation</u>: liquid sludge is put in contact with an oxidative gas such as oxygen, in a wet environment at a temperature of around 250°C and under high pressure (70 to 150 bars) in a continuous process. Temperature and pressure levels, the use of catalyzer, and the gas used (oxygen or air) differentiate the existing processes. Sludge is changed in three main products: a liquid phasecontaining easily degradable organic matter, which is easily treated when sent back at the head of the station; clean combustion gases, which do not have to be treated as the relatively low temperature of the process avoids generation of compounds such as PCDD/F or NOx; and mineral residues in a liquid phase which has to be treated. Organic pollutants are broken down and heavy metals concentrate in the solid residue, except for mercury which is found in the gas. No important preliminary treatment is required before performing wet oxidation and in most cases thickening is sufficient.

<u>Pyrolysis</u>: pyrolysis is a thermal process treatment in absence of oxygen. Waste is not burnt, but brought to a temperature of 300 to 900°C. The process produces two kinds of residues: solids containing mineral matter and carbon, and hot gases. As the products of the process have a calorific value, pyrolysis is considered as a pre-treatment, requiring further valorization of the solids and gases. Analyses of the composition of the gaseous product of

the pyrolysis of sludge have shown that generally H2, CO, CO2 and hydrocarbons are the main compounds found in the gas. The proportions, however, depend of the sludge type. CO is the dominant compound, with hydrocarbons representing in some cases an important part of the gas. Composition of the gas also depends on the temperature of the pyrolysis. The composition of gases implies their treatment and use on site. They can also be cracked, as it facilitates their further use. The main advantages of the process include a reduced gas emission in comparison with incineration (by about 30%), reduced or no emission of PCDD/F, due to the low temperature of the process and a possible separation and valorization of the materials.

Gasification: gasification is a thermal process during which a combustible material is converted with air or oxygen to an inflammable gas and an inert residue. It has been used for a long time to produce gas with coal. This kind of process is performed at high temperatures, between 900°C and 1100°C with air, or between 1000°C and 1400°C with oxygen. Gasification with oxygen, which is the one most often performed, generates a gas containing 55 to 60% N_2, with a calorific value of 4-7 MJ/Nm^3. The gasification process enables the flue gas volume to be drastically reduced since carbon dioxide and water, internally formed, participate in the reaction and the unwanted N_2 may be avoided by supplying pure oxygen. Comparisons given in literature indicate that whereas during mono and co-combustion of sewage sludge, 24-30 m^3 per kg of dry sludge of flue gas are formed, gasification with pure oxygen generates only 1.7 m^3.

3.5 Concluding remarks

Given that sludge disposal to surface waters and the sea is no longer an acceptable method of sludge elimination, landfilling under proper conditions to ensure prevention of air, soil and water pollution is a possible elimination practise not entailing excessive costs (although not insignificant due to the requirements for pollution prevention and ultimate restoration of the site). However, in the context of sustainability, elimination of potentially valuable resources such as sludge is in discrepancy with recent management concepts favouring reuse and recovery practises. Therefore, landfilling lies at the bottom of the prevention ladder endorsed by several countries, including the European Union and a strong preference for methods involving material and for energy recovery is given.

In view of this, landspreading of sludge for agricultural and other purposes (forestry, silviculture, land reclamation) is promoted within the EU as a first priority, followed by methods of energy recovery (i.e., incineration). This priority is reflected in the anticipated overall future decline of landfilling practices from about 40% to 20% and the corresponding increase of landspreading and incineration (from 45% to 55% and from 14% to 25% respectively).

Landspreading is expected to gain in significance and become the main method of sludge management. Furthermore, a land application wider in scope is promoted to include sludge used in forestry, and silviculture and land reclamation, in addition to agricultural landspreading.

It should be stressed that although almost 99% of sludge contains compounds of agricultural value, the presence of pollutants (pathogens, heavy metals and micro-organics) even at the very small fraction of 1%, creates the need for a very careful approach addressing not only technical aspects but also social and economical. A detailed presentation of these issues is presented in Part 4.

As already mentioned, besides landspreading, incineration is also a method of increasing significance. The merits and limitations of this route, therefore, deserve attention and a relevant discussion is presented in Part 5.

4. LANDSPREADING OF SLUDGE

Land use of sludge satisfies the basic ecological principles of recycling and could, at the same time, be considered as financially attractive because due to its organic content sludge can partially substitute the chemical fertilizers and improve the quality of the soil.

In principle, municipal sludge should be applied on land in accordance with the N and/or P requirements of the growing plants. However, it should be stressed that sludge does not contain only valuable compounds, but also contains pathogens, heavy metals and persistent organic pollutants.

For this reason, land application of sludge should be practised under conditions which eliminate unacceptable environmental pollution and risks to animals and humans. Over the last 30 years several municipal sludge land application regulations have been developed specifying safe conditions in terms of standards related to potentially dangerous substances (pathogens, heavy metals and organic substances), acceptable treatment methods and appropriate operational modes.

Recycling of sludge to agricultural land is the most attractive landspreading option and most legislation and guidelines focus on this mode of landspreading. However, use of sludge in forests, silviculture and for land reclamation is emerging as an interesting alternative and due consideration has been given to these modes of landspreading in recent legislations and guidelines.

4.1 Limitations due to pathogens – current practises

Of the three main groups of pollutants, pathogens represent the group which is not persistent in character and decay may be expected over time or by means of appropriate sludge treatment, whilst for the two other groups a reduction to acceptable levels may be achieved only through source control.

Due to the transient character of pathogens, protection may be achieved through proper management techniques allowing for sufficient sludge storage time prior to application, thus inducing reduction through decay. Additional barriers to the transmittance routes (from soil and plants to consumers) may be established in the form of specified restrictions during landspreading. Thus, under such conditions practical elimination of pathogens in the sludge through a sanitization process is not considered to be necessary.

The existing regulatory framework both in EU and in US, follows this guiding principle. Thus, Directive 86/278/EEC on the protection of the environment and, in particular, of the soil when sewage sludge is used in agriculture, does not include specific requirements for pathogen content in sludge. Provision is made so that "*sludge shall be treated before being used in agriculture*". The definition of treated sludge is provided in Article 2 as "*sludge which has undergone biological, chemical or heat treatment, long-term storage or any other appropriate process so as significantly to reduce its fermentability and the health hazards resulting from its use"*. However, according to the same Directive, the use of untreated sludge is possible "*if it is injected or worked into the soil*".

With respect to treatment obligations, most countries have exactly transposed the provisions of the Directive into national legislations. However, discrepancies may be observed among various countries. For example, Belgium-Flanders, Denmark, Finland, Germany, Greece, Italy, The Netherlands, Portugal and Spain prohibit the use of untreated sludge, while other countries have no specific requirements concerning the treatment of sludge. In Finland, sludge must be treated by digestion or lime stabilization before being used in agriculture. In the case of Denmark, even stabilized sludge has to be worked into the soil within 12 hours after application. In France, Ireland, Luxembourg and Sweden legislation permits the use of untreated sludge, in accordance with the Directive. In the United Kingdom, untreated sludge was allowed under certain conditions but as a consequence of a recent agreement between sludge producers (Water UK) and food retailers (BRC) there is a ban the use of untreated sludge in agriculture.

Furthermore, Article 7 of Directive 86/278/EEC provides restrictions concerning the spreading of sludge on grazing and pastureland, and on land on which vegetables and fruits are grown.

These measures include the following restrictions and conditions. The application of sewage sludge in agriculture is not allowed: if the grassland is to be grazed or the forage crops are to be harvested before a three-week period has elapsed; on soil in which fruit and vegetable crops are growing, with the exception of fruit trees; and on soil intended for the cultivation of fruit and vegetable crops which are normally in direct contact with the soil and normally eaten raw, for a period of ten months preceding the harvest of the crops and during the harvest itself.

These measures have been adopted by Member States but in different ways, according to the country (Table 4.1).

In comparison with the EU directive, the EPA regulation places more emphasis on requirements for pathogens (bacteria, viruses) and vector attraction (insects, birds, etc.). The federal Part 503 Rule (40 CFR Part 503) establishes requirements for land application of sewage sludge. In the context of the Part 503 regulation, the pathogen reduction requirements for sewage sludge are divided into two categories known as Class A and Class B. The implicit goal of Class A requirements is to reduce the pathogens in sewage sludge (including *Salmonella* spp., enteric viruses and viable helminth ova) to below detectable limits, while Class B requirements aim to ensure that pathogens have been reduced to levels that are unlikely to pose threat to public health and the environment. Class A sludge can be used in an unrestricted fashion (subject to heavy metals limitations) whilst Class B can be used with the restrictions and specific managerial practises prescribed by the Part 503 regulation.

For Class A sludge, the pathogen requirements of Table 4.2 are specified (EPA, 1995). In addition to these requirements certain processes for achieving sanitized sludge are prescribed including composting, heat drying, heat treatment, thermophilic aerobic digestion, beta ray Irradiation, gamma ray irradiation and pasteurization. In the case of heat treatment several combinations of temperature and time of exposure can be used. Similarly, in other processes certain combinations of temperature, pH and time may be used (e.g., high pH above 12, for more than 72 hours or high pH above 12 and high temperature above 52°C for at least 12 hours). When one of the recommended methods is properly used the need for analytical work for the determination of specific pathogens in the sludge is limited. Whenever other methods are used or the treatment of sludge is not known, a thorough and time consuming analysis for pathogen detection is required.

Table 4.1

Surfaces on which the use of sludge is prohibited - comparison between requirements of Directive 86/278/EEC (Article 7) and national legislation (EC Report 2001)

Directive 86/278/EEC Requirements of Article 7	**Grassland or forage crops if the grassland is to be grazed or the forage crops to be harvested before a certain period has elapsed. This period, shall under no circumstances be less than three weeks.**	**Soil in which fruit and vegetable crops are growing (with the exception of fruit trees)**	**Ground intended for the cultivation of fruit and vegetable crops which are normally in direct contact with the soil and normally eaten raw, for a period of 10 months preceding the harvest of the crops and during the harvest itself**
Austria	Prohibition on meadows, pasture, alpine pastures.	=	Prohibition on vegetable crops, berries or medicinal herbs; no growing of these crops before the elapse of 1 year.
Belgium (Flandres)	6 weeks delay.	=	=
Belgium (Walloon)	6 weeks delay.	=	=
Denmark		=	=
Finland	Ploughing down compulsory.	=	Potatoes, root crops and vegetables may not be cultivated on arable land before the elapse of 5 years.
	Sludge may be used only on soil on which grain, sugar beet, oil-bearing crops or crops not used for human food or animal feed are cultivated.		
France	=	=	=
Germany	=	Prohibition.	=
Greece	=	=	=
Iceland	=	=	=
Italy	5 weeks delay.	=	=
Luxemburg	4 weeks delay.	=	=
The Netherlands	Prohibition on forage crops, land prohibition during the grazing season on grazing land.	=	=
Portugal	=	=	=
Spain	=	=	=
Sweden	Prohibition on grazing land. In arable land which is to be used for grazing or if fodder crops are to be harvested within ten months of the time the sludge is spread.	=	=
UK	=	=	=
Estonia	2 months for fodder crops	=	1 year delay
Latvia	Prohibition.	No restriction.	Restriction concerning spreading period according to crop type.
Poland	Prohibition.	No restriction.	1 year delay.

= stands for no difference with the Directive.

Table 4.2

Pathogen requirements for all Class A alternatives

Requirements to be met for all six Class A pathogen alternatives
• the density of faecal coliform in the sewage sludge must be less than 1,000 most probable number (MPN) per gram total solids (dry-weight basis); and • the density of *Salmonella* sp. bacteria in the sewage sludge must be less than 3 MPN per 4 grams of total solids (dry-weight basis).
This requirement must be met at one of the following times:
• when the sewage sludge is used or disposed; and • when the sewage sludge is prepared for sale or give-away in a bag or other container for land application.

In many cases of sludge landspreading, Class B sludge may be adequate. This sludge quality can be achieved by more conventional processes aiming to reduce faecal coliforms to less than $2x10^6$ MPN (or CFU) per g TS. There is no requirement for reduction of viable helminth ova in Class B sewage sludge. Unlike Class A sewage sludge which is essentially pathogen free, Class B sewage sludge contains some pathogens, resulting in additional site restrictions during application (Table 4.3).

Table 4.3

Part 503 regulatory restrictions for the Harvesting of Crops and Turf, Grazing of Animals, and Public Access on Sites Where Class B Sewage Sludge is Land Applied

Restrictions for the harvesting of crops and turf:
1. food crops with harvested parts that touch the sewage sludge/soil mixture and are totally above ground shall not be harvested for 14 months after application of sewage sludge; 2. food crops with harvested parts below the land surface where sewage sludge remains on the land surface for 4 months or longer prior to incorporation into the soil shall not be harvested for 20 months after sewage sludge application; 3. food crops with harvested parts below the land surface where sewage sludge remains on the land surface for less than 4 months prior to incorporation shall not be harvested for 38 months after sewage sludge application; 4. food crops, feed crops, and fibre crops, whose edible parts do not touch the surface of the soil, shall not be harvested for 30 days after sewage sludge application; and 5. turf grown on land where sewage sludge is applied shall not be harvested for 1 year after application of the sewage sludge when the harvested turf is placed on either land with a high potential for public exposure or a lawn, unless otherwise specified by the permitting authority.
Restriction for the grazing of animals: 1. animals shall not be grazed on land for 30 days after application of sewage sludge to the land.
Restrictions for public contact: 1. access to land with a high potential for public exposure, such as a park or ball field, is restricted for 1 year after sewage sludge application. Examples of restricted access include posting with no trespassing signs, or fencing; and 2. access to land with a low potential for public exposure (e.g., private farmland) is restricted for 30 days after sewage sludge application. An example of restricted access is remoteness.

4.2 Limitations due to pathogens - trends

As already mentioned, Directive 86/278/EEC does not include specific requirements for pathogen content in sludge used in agriculture. However, in order to reduce possible health risks related to pathogens, several national regulations have added limitations on pathogen content to standard requirements on sludge quality. This is the case in France, Italy, Luxembourg, and in two Länder in Austria (Burgenland and Lower Austria). The most common pathogens which are addressed by legislation are *salmonella* and enteroviruses.

Despite these exceptions it can be concluded that the regulatory requirements on pathogen content in sewage sludge still remains quite limited in national legislations. This can be partly explained by the fact that national codes of practise are considered to sufficiently cover this issue, by providing recommendations on sludge treatment and sludge landspreading.

However, recent developments in the European Union legislation in the framework of an attempt to update Directive 86/278/EEC (EU, 1999) indicate that a more stringent approach with respect to sludge pathogens may be adopted. The proposal becomes more specific with respect to sludge treatment methods aiming at reducing its fermentability, odour nuisance and pathogen content. As a result "treated sludge" is defined as sludge, which has undergone one of the following treatment processes:

- thermal drying (at about 100°C with reduction of water content to less than 30%;
- pasteurization for a minimum of 30 minutes at 70°C);
- thermophilic anaerobic digestion (at a temperature of 55°C ± 5°C with a minimum retention period of 10 days);
- mesophilic anaerobic digestion (at a temperature of 35°C ± 3°C with a minimum retention period of 21 days);
- thermophilic aerobic stabilization (at a temperature of 55°C ± 5°C with a minimum retention period of 10 days);
- composting (at a temperature of at least 40°C for 5 days and for 4 hours during this period at a minimum of 55°C within the body of the pile);
- storage in liquid form or cold fermentation (for at least 3 months as a batch, without admixture or withdrawal during the storage period); and
- conditioning with lime (reaching a pH of 12 or more and maintaining a temperature of at least 55°C for a minimum of 2 hours after the lime addition).

The treatment must reduce the pathogen content of sludge to less than 10^3 MPN/g dm *Salmonella* spp.

Irrespective of treatment, the proposal outlines several additional restrictions.

Sludge should not be used on soils with a pH of less than 5.0, on water saturated, flooded, frozen or snow-covered ground and when weather conditions do not guarantee the minimization of sludge run-off. Sludge should not be spread with mechanical equipment that produces aerosols.

During application to agriculture, sludge should not be used:

- on grassland or forage crops if the grassland is to be grazed or the forage crops are to be harvested before a minimum period three weeks has elapsed;
- on soil in which fruit and vegetable crops are growing, with the exception of fruit trees; and

- on soil intended for the cultivation of fruit and vegetable crops, which are normally in direct contact with the soil and normally eaten raw, for a period of ten months preceding the harvest of the crops and during the harvest itself.

Beneficial use of sludge in silviculture is allowed if there is an agronomic need for nutrients or for the improvement of the content of organic matter in soil and if there is no accumulation of undesirable substances on and in the soil.

Beneficial use of sludge to reclaimed land is allowed in the case of reclaimed land for agriculture, silviculture or forestry, when there is no accumulation of undesirable substances on and in the soil.

For other beneficial outlets such as parks, golf courses, green areas, recreation grounds, sports complexes or the like, sludge can be used in its solid form and without causing odour nuisance. In this case sludge must be treated and its pathogen content reduced to:

- *Salmonella* spp. less than 8 MPN in 10 g of dry matter;
- enteroviruses less than 3 MPN in 10 g of dry matter; and
- Helminth eggs less than 3 MPN in 10 g of dry matter.

The proposal also includes producers' obligations to consumers, duly documented by certificates guaranteeing the suitability of sludge for beneficial use. Member States must ensure that the authorities keep up-to-date records on the following:

- name and address of the producer;
- name and address of the treatment plant from which the sludge originates;
- type of treatment carried out and result of the analysis on *Salmonella* spp.;
- results of the analysis of heavy metal concentrations on the soil on which sludge is applied;
- composition and properties of the sludge in relation to the agronomic parameters;
- results of the analyses of heavy metals and organic compounds in the sludge;
- in the case of use in parks, golf courses, green areas, recreation grounds, sports complexes or the like, results of the analyses of pathogens in sludge.
- name and address of the receiver;
- cadastral location of the parcel on which the sludge is recycled; and
- quantity of sludge supplied for beneficial use.

These records should be collected to provide a basis for the consolidated report, and sent to the Commission. The records should also be available, upon request, to the general public.

It is evident that this EU initiative is intended to broaden the scope of existing regulation by including the management of sludge in beneficial outlets such as silviculture, green areas and reclaimed land, while at the same time indicating specific treatment strategies for the reuse of sludge in order to avoid malpractice. The limits set are more specific and include all the parameters related to the quality of sludge such as heavy metals, pathogen and organic compounds.

4.3 Limitations due to heavy metals and organic pollutants - current practises

Due to the persistent nature of heavy metals and organic pollutants, prolonged application of sludge to land may result in increased concentrations of these pollutants in the soil. If these increased concentrations exceed certain limits, unacceptable human exposure to may occur as the pollutants are transported though several pathways from the soil to the consumer. Several exposure routes have been identified, although the food chain transfer is considered to be the primary route of human exposure.

In order to determine the numerical limits for pollutant input in land application, the process usually starts by establishing the acceptable daily human intake (ADI) for a pollutant and then quantitatively backtracks the pollutant transport through various environmental exposure routes (usually the food chain route) to arrive at an acceptable pollutant concentration for the receiving soil.

The guiding principle involved in this approach has been recently summarized in a WHO report (2002). Proponents of this type of regulation contend that the capacity of soils to detoxify pollutants should be utilized fully. If land application operations are managed properly, the agronomic benefits of wastewater and sewage sludge are realized and the pollutants in the receiving soils are kept at levels that will not be harmful to the exposed individuals. Land application regulations based on very stringent pollutant loading limits, rigorous industrial wastes pre-treatment requirements, and advanced sludge treatment discourage the wastewater treatment plants from considering the land application option. The consequence of favouring other disposal options could be even more undesirable, such as incineration or discharging the wastes into ecologically sensitive water bodies. The land application regulations (or guidelines) should set only the maximum permissible pollutant loading limits and/or maximum permissible pollutant concentrations of the soil and provide the users of this option the flexibility of developing safe and site-specific land application operations.

However, several drawbacks have been recognized:

- the pollutant levels in the soil, are eventually expected to rise to a maximum tolerable level;
- there are multiple environmental pathways through which humans may be exposed to the released pollutants. Thorough knowledge of the exposure pathways and the parameters defining the rate and amount of pollutant transfer in each pathway are necessary to determine the maximum permissible pollutant loading limits. In order for the regulations to be comprehensive, every possible toxicant contained in the wastewater and sewage sludge must be identified, tracked through these pathways, and regulated; and
- in a crop production system, pollutants may also enter the soil and be absorbed by plants through other sources such as atmospheric fallout, irrigation water, and fertilizer and pesticide applications. Limiting the pollutant inputs from land application of wastes does not eliminate and does not account for pollutants from other sources.

Despite these limitations, which have resulted in criticism of this approach in both the US and Europe, the regulatory framework in both cases in essence follows the approach.

In setting the maximum pollutant loading rates for trace elements and metals, USEPA final regulation on land application of sludge (1995) relied on the soil capacity to assimilate pollutants. Although more than 20 organic pollutants were considered during the course of the rule development process, none were eventually regulated, because (1) their concentrations in the sludge were sufficiently low that the maximum permissible annual loading limits are not likely to be exceeded if amounts of sludge applied do not exceed the agronomic rates; or (2) they have been banned or restricted for use or are not longer manufactured for use in the United States. The maximum cumulative metal loading limit (CPLR) and the annual metal loading rates (APLR, on the assumption of an application duration of 20 years) for several heavy metals, are shown in Table 4.4. In the same table the ceiling concentration limits (CCL) and the pollutant concentration limits (PCL) are also shown. CCL is the maximum allowable concentration of a pollutant in sewage sludge applied to land. If the CCL for any one of the regulated pollutants is exceeded, the sewage sludge cannot be applied. These limits were developed to prevent the land application of sewage sludge containing high concentrations of pollutants. PCL is the most stringent pollutant limits, which help ensure a minimum quality of sewage sludge that can be applied on a long-term basis.

Table 4.4

Land application pollutant limits for sewage sludge

Pollutant	**CCL mg/kg (dw)**	**PCL mg/kg (dw) (monthly average)**	**CPLR kg/ha**	**APLR kg/ha/yr**
As	75	41	41	2.0
Cd	85	39	39	1.9
Cr	3 000	1 200	3 000	150
Cu	4 300	1 500	1 500	75
Pb	840	300	300	15
Hg	57	17	17	0.85
Mo Ni	75 420	- * 420	- * 420	- * 21
Se	100	36 *	100	5.0
Zn	7 500	2 800	2 800	140
Applies to:	All sewage sludge that is land applied.	Bulk sewage sludge and bagged sewage sludge.	Bulk sewage sludge.	Bagged sewage sludge.

* EPA is re-examining these limits.

The European Directive 86/278/EEC follows the same basic principle although the respective limits (Table 4.5) are somewhat stricter than the USEPA limits. As in the case of the USEPA regulation, the European Directive does not specify concentration limits for organic pollutants, which is understandable considering the limited information at that time regarding: their transport in the environment and their dose-effect relationships with additional weaknesses in reliable analytical methods; the fact that most of the organic micro-pollutants existing in the sludge are non-polar with very low water solubility and hence, due to their strong hydrophobic nature, most of them tend to be tightly bound to soils.

Apart from the chemical pollutant limitations set, the Directive provides the legal frame for the application of sludge in terms of the producers' responsibility and certification regarding the quality of the product, the provision of the necessary information to the receiver and the responsible authority. Furthermore, producers should have a permit for beneficial use of sludge and should follow codes of good practise.

Table 4.5

Limit values for concentrations of heavy metals in soil and sludge and annual heavy metals loads applied

Elements	Limit values for concentrations of heavy metals in soil (mg/kg dm) 6<pH<7	Limit values for concentrations of heavy metals in sludge for recycling (mg/kg dm)	Limit values for annual loads of heavy metals which may be added to soil, based on a ten year average (kg/ha/y)
Cadmium (Cd)	1 – 3	20 – 40	0.15
Chromium (Cr)	-	-	-
Copper (Cu)	50 – 140	1000 – 1750	12
Mercury (Hg)	1 – 1.5	16 – 25	0.10
Nickel (Ni)	30 – 75	300 – 400	3
Lead (Pb)	50 – 300	750 – 1200	15
Zinc (Zn)	150 – 300	2500 – 4000	30

4.4 Limitations due to heavy metals and organic pollutants - trends

In the US a more cautious approach for land application of sewage sludge has been suggested. It has been argued that because of “the potential for widespread use of sludge on agricultural and residential land, the persistence of many of the pollutants which mayremain in soils for a very long time, and the difficulty of remediation”, it called for a more cautious approach.

They argued that the margin of safety might be compromised, especially at locations having shallow acidic soils and shallow groundwater table. The critics also pointed out that the cropland input of Ba, Be, Tl (thallium), PCB, dioxins, detergent residues and their degradation products (alkylbenzene sulphonates, nonylphenol, and nonylphenol ethoxylates) and di-(2-ethylhexyl)phthalate through sewage sludge application should also be regulated. Consequently, an alternative set of numerical limits (Table 4.6) much closer to those adopted by some European countries, was proposed.

The disagreements in numerical values among the regulations illustrate the dilemma in reaching a consensus in developing land application criteria. A similar situation exists with the various Member States of the European Union. As shown in Table 4.7, although several Member States (Greece, Luxembourg, Ireland, Italy, Portugal and Spain) have set limit values for heavy metals in sludge which are mostly similar to the limit values set by the Directive 86/278/EEC, in other countries (Belgium, Austria, Denmark, Finland, Germany, The Netherlands and Sweden), more stringent national legislations can be found with limit values, in some cases less than 10% of the limit values established by the Directive. A similar conclusion can be drawn with respect to heavy metal concentrations in the soil (Table 4.8), although not to the same extent. It is clear that the rationale behind the more stringent legislation of some of the Member States is based on a different guiding principle than the one already discussed which takes into consideration the capacity of receiving soil to assimilate, attenuate, and detoxify pollutants. Instead, the guiding principle in this case is the consideration that soil is an irreplaceable natural resource for mankind which should be used without any restriction, thus no accumulation of pollutants in the soil can be accepted.

Table 4.6
Metal Concentration Limits for Sewage Sludge-treated Soils-Proposed by Cornell University Waste Management Institute

Element	Maximum Concentration (mg/kg)	Reasons for Setting Limits
As	1 – 10	Ingestion of contaminated soil by children
Cd	2	Harmful through food chain transfer
Cu	40 – 100	Phytotoxicity
Pb	300	Ingestion of contaminated soils by children
Hg	1	Ecotoxicological and groundwater concerns
Mo	4	Toxicity to ruminant animals
Ni	25 - 50	Phytotoxicity
Se	5	Harmful to foraging animals
Tl	1	Groundwater quality and Food chain transfer
Zinc	75 - 200	Phytotoxicity
PCBs	1	Human exposure

Table 4.7
Limit values for heavy metals in sludge (mg/kgDM) (shaded cells represent limit values below those required by Directive 86/278/EEC)

	Cd	Cr	Cu	Hg	Ni	Pb	Zn	As	Mo	Co
Directive 86/278/EEC	20-40	-	1000-1750	16-25	300-400	750-1200	2500-4000	-	-	-
Austria	2[a] 10[b] 10[c] 4[d] 10[e] 0.7-2.5[f]	50[a] 500[b] 500[c] 300[d] 500[e] 70-100[f]	300[a] 500[b] 500[c] 500[d] 500[e] 70-300[f]	2[a] 10[b] 10[c] 4[d] 10[e] 0.4-2.5[f]	25[a] 100[b] 100[c] 100[d] 100[e] 25-80[f]	100[a] 400[b] 500[c] 150[d] 500[e] 45-150[f]	1500[a] 2000[b] 2000[c] 1800[d] 2000[e] 200-1800[f]	20[e]	20[e]	10[a] 100[e]
Belgium (Flanders)	6	250	375[f]	5	100	300	900[f]	150	-	-
Belgium (Walloon)	10	500	600	10	100	500	2000	-	-	-
Denmark -dry matter basis - total phosphorus basis	 0.8 100	 100	 1000	 0.8 200	 30 2500	 120[g] 10000[g]	 4000	 25[h]	 -	 -
Finland	3-1.5[i]	300	600	2 1[i]	100	150 100[i]	1500	-	-	-
France	20[j]	1000	1000	10	200	800	3000	-	-	-
Germany	10	900	800	8	200	900	2500	-	-	-
Greece	20-40	500	1000-1750	16-25	300-400	750-1200	2500-4000	-	-	-
Ireland	20	-	1000	16	300	750	2500	-	-	-
Italy	20	-	1000	10	300	750	2500	-	-	-
Luxembourg	20-40	1000-1750	1000-1750	16-25	300-400	750-1200	2500-4000	-	-	-
The Netherlands	1.25	75	75	0.75	30	100	300	-	-	-
Portugal	20	1000	1000	16	300	750	2500	-	-	-
Spain -soil pH < 7 -soil pH > 7	 20 40	 1000 1750	 1000 1750	 16 25	 300 400	 750 1200	 2500 4000	-	-	-
Sweden	2	100	600	2.5	50	100	800	-	-	-
UK	-	-	-	-	-	-	-	-	-	-
Estonia	15	1200	800	16	400	900	2900	-	-	-
Latvia	20	2000	1000	16	300	750	2500	-	-	-
Poland	10	500	800	5	100	500	2500	-	-	-

[a] Lower Austria
[b] Upper Austria
[c] Burgenland
[d] Vorarlberg
[e] Steiermark
[f] Carinthia

[f] These values are reduced to 125 (Cu) and 300 (Zn) from 31/12/2007
[g] For private gardening, lead value is reduced to 60 mg/kg DM or 5000 mg/kg P
[h] For private gardening
[i] Target limit values for 1998
[j] 15 mg/kg DM from January 1, 2001 and 10 mg/kg DM from January 1, 2004

The objective of regulating land application must be to prevent pollutants from accumulating in the waste-receiving soil. Therefore, land application of sewage sludge should not result in a net increase of the pollutant level in the receiving soils. To achieve this goal, the pollutant input in land application of wastes must be equal to or less than the pollutant output through plant uptake and other losses. If this requirement is met, the soil's ability to sustain any future land use is insured and the food chain transfer of potentially hazardous pollutants may be kept to a minimum. Man is protected from potential adverse health effects by preventing the waste-receiving soils from being contaminated.

Table 4.8

Limit values for heavy metals in soil (mg/kgDM) (shaded cells represent limit values below those required by Directive 86/278/EEC) (EC Report, 2001)

Directive 86/278/EEC (6<pH<7)	Cd 1-3	Cr -	Cu 50-140	Hg 1-1.5	Ni 30-75	Pb 50-300	Zn 150-300	As -	Mo -	Co -
Austria	1.5[a] 1[b] 2[c] 2[d] 2[e] 0.5-1.5[f]	100[a] 100[b] 100[c] 100[d] 100[e] 50-100[f]	60[a] 100[b] 100[c] 100[d] 100[e] 40-100[f]	1[a] 1[b] 1.5[c] 1[d] 1[e] 0.2-1[f]	50[a] 60[b] 60[c] 60[d] 60[e] 30-70[f]	100[a] 100[b] 100[c] 100[d] 100[e] 50-100[f]	200[a] 300[b] 300[c] 300[d] 300[e] 100-200[f]	-	10[e]	50[e]
Belgium (Flanders)	0.9	46	49	1.3	18	56	170	22	-	-
Belgium (Walloon)	2	100	50	1	50	100	200	-	-	-
Denmark	0.5	30	40	0.5	15	40	100	-	-	-
Finland	0.5	200	100	0.2	60	60	150	-	-	-
France	2	150	100	1	50	100	300	-	-	-
Germany	1.5	100	60	1	50	100	200	-	-	-
Greece	1-3	-	50-140	1-1.5	30-75	50-300	150-300	-	-	-
Ireland	1	-	50	1	30	50	150	-	-	-
Italy	1.5	-	100	1	75	100	300	-	-	-
Luxembourg	1-3	100-200	50-140	1-1.5	30-75	50-300	150-300	-	-	-
The Netherlands	0.8	100	36	0.3	35	85	140	-	-	-
Portugal								-	-	-
-soil pH<5.5	1	50	50	1	30	50	150			
-5.5 < soil pH < 7	3	200	100	1.5	75	300	300			
- soil pH > 7	4	300	200	2	110	450	450			
Spain								-	-	-
-soil pH < 7	1	100	50	1	30	50	150			
-soil pH > 7	3	150	210	1.5	112	300	450			
Sweden	0.4	60	40	0.3	30	40	100-150	-	-	-
UK								-	-	-
-5 < soil pH < 5.5	3	-	80	1	50	300	200			
-5.5 < soil pH < 6	3	-	100	1	60	300	250			
-6 = soil pH = 7	3	-	135	1	75	300	300			
-soil pH > 7	3	-	200	1	110	300	450			
Estonia	3	100	50	1.5	50	100	300	-	-	-
Latvia	0.3-1	15-30	10-25	0.1-0.15	8-30	15-30	35-100	-	-	-
Poland	1-3	50-100	25-75	0.8-1.5	20-50	40-80	80-180	-	-	-

[a] Lower Austria
[b] Upper Austria
[c] Burgenland
[d] Vorarlberg
[e] Steiermark
[f] Carinthia

The pros and cons of this approach are summarized in a recent WHO Report (2002). The fundamental concept underlying these types of regulations is in general agreement with the principles of ecology. The advantage of using this approach to develop regulations is that detailed knowledge of exposure pathways and the dose-response relationships for pollutants are not needed. The numerical limits for pollutants may be calculated by employing simply the mass balances by matching pollutant input from waste application with the expected pollutant outputs. As the outcomes of the mass balance would be universally applicable, therefore, one set of rules may apply to all situations. The net cost of waste treatment and disposal, however, will be high because more advanced technologies must be employed in preventing pollutants from entering waste streams and to treat wastewater, or a considerably larger land area is required to accommodate the same amount of waste.

The sludge disposal guidelines of the abovementioned European countries, and more specifically The Netherlands, Denmark, Norway, and Sweden, prescribed the spirit of this approach and set stringent numerical limits on the annual and total input of heavy metals.

A marked difference can also be observed among Member States, regarding organic pollutants. While most countries following Directive 86/278/EEC do not set limits for organic pollutants, specific reference to organic pollutants is made the legislation of some other countries (Table 4.9).

Table 4.9

Limit values for organic compounds (mg/kg dm) and dioxins (ng TE/kg dm) in sludge according to EU and other countries.

	Denmark	Sweden	Germany	Austria	France
Organic compounds					
AOX	-	-	500	500	-
LAS	1300	-	-	-	-
DEHP	50	-	-	-	-
NPE	10	100	-	-	-
PAH	3	3	-	6*	-
PCB	-	0.4	0.2	0.2-1.0	0.8
Toluene	-	5	-	-	-
Fluoranthene	-	-	-	-	4-5
Benzo(b)fluoranthene	-	-	-	-	2.5
Benzo(a)pyrene	-	-	-	-	1.5-2.0
Dioxins					
PCDD	-	-	100	50-100	-
PCDF	-	-	100	50-100	-

*Only the Carinthia region.

The European Union has initiated consultations with governmental experts from member nations, industries, and non-government environmental organizations in general on an update to the Directive 86/278/EEC on land application of sewage sludge.

This process has been carried out through the circulation and discussion of the Working Document on Sludge, 3rd Draft, Brussels, 27 April, 2000 which was already discussed in connection with the pathogens. Regarding heavy metals it seems that the document tends to adopt the no-accumulation concept shared by the European countries with stringent legislation. Furthermore, a proposal for limits on organic pollutants is also included. The proposed limits for heavy metals and organic pollutants are shown in Tables 4.10 and 4.12, and for reasons of comparisons, whenever appropriate the limits of the Directive 86/278/EEC are also included.

Table 4.10
Limit values for concentrations of heavy metals in sludge for recycling

Elements	Limit values (mg/kg dm)			
	Directive 86/278/EEC	Target end 2005	Target about 2015*	Target about 2025*
Cd Cu	20 – 40 1 000 – 1 750	10 1 000	5 800	2 600
Hg	16 – 25	10	5	2
Ni	300 – 400	300	200	100
Pb	750 – 1 200	750	500	200
Zn	2 500 – 4 000	2 500	2 000	1 500

*At least 90% of the sludge produced in Europe should meet these targets

Table 4.11
Limit values for concentrations of heavy metals in soil (mg/kg dm)

Elements	EU 86/278/EEC 6=pH<7	5=pH<6	6=pH<7	pH=7
Cd	1 – 3	0.5	1	1.5
Cu	50 – 140	20	50	100
Hg	1 – 1.5	0.1	0.5	1
Ni	30 – 75	15	50	70
Pb	50 – 300	70	70	100
Zn	150 – 300	60	150	200

Table 4.12
Limit values for concentrations of organic compounds and dioxins
in sludge for recycling

Organic compounds	Limit values (mg/kg dm)
AOX[1]	500
LAS[2]	2 600
DEHP[3]	100
NPE[4]	50
PAH[5]	6
PCB[6]	0.8
Dioxins	**Limit values (ng TE/kg dm)**
PCDD[7]	100
PCDF[8]	100

1. Sum of halogenated organic compounds.
2. Linear alkylbenzene sulphonates.
3. Di(2-ethylhexyl)phthalate.
4. It comprises the substances nonylphenol and nonylphenolethoxylates with 1 or 2 ethoxy groups.
5. Sum of the following polycyclic aromatic hydrocarbons: acenapthene, phenanthrene, fluorene, flouranthene, pyrene, benzo(b+j+k)fluoranthene, benzo(a)pyrene, benzo(ghi)perylene, indeno(1, 2, 3-c, d)pyrene.
6. For each of the polychlorinated byphenils components number 28, 52, 101, 118, 138, 153, 180.
7. Polychlorinated dibenzodioxins.
8. Polychlorinated dibenzofuranes.

4.5 Concluding remarks

There is general agreement on the beneficial aspects of sludge reuse for agricultural purposes, related to an improvement of soils characteristics and addition of valuable nutrients. Application rate in principle should be governed by the N or P requirements of the growing plants, but several constraints leading to reduced rates or even unacceptability of reuse, have been recognized due to the presence of potentially harmful substances in the sludge. In most guidelines and regulations these are grouped in three categories: pathogens, inorganic trace elements and heavy metals and toxic micro-organics.

With respect to inorganic trace elements and heavy metals, the aim of the legislatory framework is to set limits in terms of maximum concentrations in the sludge and the soil, as well as application rates, that will ensure safe practises and to outline the monitoring procedures needed to characterize sludge. Violation of the set limits renders sludge unsuitable for reuse, without practical possibilities for improvement through treatment. Systematic and effective industrial wastewater control is proposed as the only viable method by which a reduction in the concentration of heavy metals in the sewage can be effected, thus leading to an acceptable sludge quality.

The maximum concentration of trace elements and heavy metals in sludge is a parameter considered in all regulations, but with differences regarding the indicative elements and heavy metals and, mainly, the numerical limits.

Another important parameter is the annual metal load in terms of kg/ha year, while in some regulations the maximum concentration of the metals in the soil is also considered, with numerical limits that also vary considerably.

The rationale behind the observed differences can be followed by comparing the regulations adopted by the USEPA on the one hand and countries such as The Netherlands, Denmark and Sweden on the other, representing two different approaches to the problem. The European countries mentioned (as well as others) adopt an approach aiming to prevent pollutant accumulation in the receiving soil. Soil is considered as an irreplaceable natural resource which, if free from pollutant, can be used without any undue restriction in the long term. Prevention of pollutant accumulation can be achieved by balancing the pollutant input via sludge application andthe pollutant output via uptake by harvested plants (surface runoff, leaching and atmospheric loss are limited and in any case not desirable for metals). This leads to very low annual loading rates, creating the need for costly advanced pollution control measures (i.e., industrial wastewater control) and large land areas. By protecting the soil from contamination, human health and safety are ensured without the need for a detailed and inherently ambiguous knowledge of exposure pathways and dose-response relationships.

A different approach is adopted by the USEPA in consideration of the fact that the abovementioned land application regulations, based on stringent pollutant loading limits and/or rigorous industrial wastes control measures, tend to discourage the wastewater treatment plants from considering the land application option and relying on disposal options which maybe more undesirable (incineration, landfilling). The regulation should, according to this approach, avoid excessively stringent standards that may originate from adherence to principles of ecology, and focus on the conditions which are necessary for a documented safe practise. Thus, the concept of zero pollutant accumulation in the soil is challenged and attention is given to specify the levels of pollutants in the soil that are not harmful to the exposed individuals. To determine this maximum tolerable level and the pollutant loading limits, thorough knowledge of the exposure pathways and knowledge of the parameters defining the rate and amount of pollutant transfer in each one of the multiple exposure pathways are necessary (direct ingestion, food chain transfer, etc.). Admittedly, the technical information needed to undertake this task is not readily available and the mathematical models, used to simulate the fate and transport of pollutants, are characterized by many uncertainties. Thus, the scientific basis of the various regulations to be followed for the selection of the numerical values for permissible soil concentrations in the soil or maximum pollutant loadings is often not clear.

The EPA regulations have also been the subject of significant criticism, due to the fact that the maximum permissible concentrations were the product of field data for metal uptake by corn. The direct result was the underestimation of the phytotoxicity thresholds applicable to a wider range of crops, in part because corn is able to root deeply and is metal tolerant. Additionally the decision to use 50% yield reduction and plant top (rather than root) concentrations of heavy metals as phytotoxicity indicators may have obscured incipient toxicity.

The existing EU legislation (86/278) limits, which have been adopted possibly with minor modifications by many countries (Greece, UK, France, Cyprus, etc.), are comparable to the USEPA limits as far as pollutant concentrations in sludge are concerned. However, when it comes to annual loading rates, these are significantly lower in the case of the EU, although still much higher that the limits of the legislations of countries like Denmark, The Netherlands and Sweden. It is interesting to note that the recent new EU proposal further reduces the limits, toward the limits adopted by these countries.

As far as organic compounds are concerned, these are excluded from the USEPA regulation due to their small frequency of appearance and low concentrations. A similar approach is adopted in the 86/278 EU Directive. However, in the new proposal, consideration is given to selected organic compounds including dioxins, as a reflection of the concern expressed and reflected in the legislation of several European countries (Denmark, Sweden, Germany, The Netherlands).

According to the EPA regulations, the type of sewage sludge depends on its quality in terms of pathogen levels and vector attraction reduction control. For each parameter, EPA suggests alternatives for effective control of chemical pollutants, pathogens and vector attraction. With respect to pathogens, sludge is classified as Class A and B, corresponding to complete absence of pathogens and significant reduction to levels that would not pose any threats to public health and the environment. The flexibility provided by the great variety of treatment alternatives specified in order to achieve the requirements of pathogen content is an advantage, but at the cost of intensified monitoring programmes for some methods, in order to prove constant compliance with the required standards. Vector attraction reduction requirements related to the control of the spread of the disease via vectors is an additional feature of the EPA regulation. The combination of these parameters results in the classification of sewage sludge in direct relation to alternative disposal routes, directly suggesting the intention of the EPA to promote sludge reuse.

The concept of sludge classification depending on a combination of hygienic properties and heavy metal concentrations, in a similar manner to the USEPA regulation, can also be found in other countries (e.g., South Africa), although not among the countries of the European Union, at least under the existing regimes.

Of the countries belonging to the south-east Mediterranean region, some follow either EU legislation (e.g., Greece, Cyprus) or rely on the USEPA guideline (e.g., Jordan), while in others (e.g., Tunisia, Egypt) an ongoing process for developing suitable regulations is under way.

5. INCINERATION

5.1 Introduction

Incineration has been practised for many years, applied mainly to municipal and clinical solid wastes and to a lesser degree to sewage sludge. A review of the situation in 1988 (Action Cost 681 of the EC) revealed that at the time incineration did not play a major role in wastewater sludge treatment in European Countries. Belgium and France incinerated the highest proportion of sludge (around 20%) although the largest quantities of sludge were incinerated in Germany followed by France. Finland, Ireland, Sweden, Greece, Luxembourg and Norway did not apply this method. In Italy several multiple hearth incinerators were installed, but only one was practically operating continuously. Three plants (one multiple hearth and two fluidized bed) were in operation in The Netherlands and four plants (three multiple hearth and one fluidized bed) in the United Kingdom. Fluidized bed and rotary kiln furnaces were used in Switzerland; the latter was also used in Spain.

The percentages of sludge incinerated in the US and Japan were much higher than Europe, 27% and 44%, respectively. The multiple hearth furnace was the prevalent type in both countries, although the share represented by fluidized bed furnaces was increasing. The reasons for the wider application of sludge incineration in these countries may be related to the lower cost of auxiliary fuel (US) and the scarcity of land (Japan).

As discussed in Part 3, incineration is expected to gain in significance as a method of sludge management, at least within the EU. The main reasons for this can be summarized as follows:

- technological improvements (dewatering and/or pre-drying, combustion technology, flue gas treatment) give solutions for a safe and efficient, though still expensive, operation of incineration plants;

- landfilling is becoming less attractive, due to the lack of suitable sites and increasing costs as a result of the growing demand for gas and leachate treatment;

- rising public concern (reflected in progressively stricter legislation) that sewage sludge may be a waste product rather than a fertilizer or soil conditioner, may lead to a reduction in the amount of sludge being applied to agriculture, especially in the case of sludge originating from large urban areas; and

- the ban on sludge disposal into sea, which will come into effect at the beginning of 1999, eliminates a significant disposal route for some countries (e.g., UK).

The rising interest in incineration is reflected in the emphasis recently placed on legislation in this field, particularly with respect to emission of air pollutants from incinerators. The limited, at least in Europe, application of incineration can justify the inadequacy of the relevant European legislation before 1990, while the renewed interest realized at the turn of the decade gave birth to a number of new or revised emission standards, both at national and European Union level, aiming to lay down clearly defined and more stringent air pollution control requirements. The emphasis thus placed on the aspects of safety and elimination of adverse environmental impacts reflects the realization that, if incineration is to become a viable and more frequently adopted sludge treatment/disposal method, this is to be accomplished on the basis of safety and environmental rather than economical considerations.

5.2 Methods of sludge incineration

The ultimate scope of incineration is to reduce the amount of sludge to its minimum possible volume, the inert ash. The incineration therefore aims at:

(I) drying the sludge cake;
(II) oxidizing all the volatile and fixed carbon content of sludge cake;
(III) producing a sterile inert residue; and
(IV) utilizing the energy content of sludge.

Methods of incineration of sludge that have been widely applied include the multiple hearth furnace and the fluidized bed furnace. Other types of incinerators, such as electric furnace and slagging furnace, are mostly used for industrial and hazardous wastes with a very limited application to municipal sludge.

Multiple hearth furnaces: the multiple hearth furnace is a counter-current multistage process where solids flow downward and flow gases upward. Under these conditions dewatered sludge cake with 20-30% DS will not ignite and can be burned only with auxiliary fuels. Dried cake with a dry solids content greater than 30-35%, depending on sludge characteristics and excess air level, is autogenous.

The multiple hearth furnace comprises a vertical, cylindrical, refractory-lined stainless steel shell containing a series of shelves or hearths, one above the other. A hollow shaft runs through the centre of the hearths through which air can be circulated for cooling. On each hearth rabble arms are attached to the centre shaft equipped with metal blades set at an angle, so that they can agitate the sludge and break-up any lumps. The teeth on the rabble arms have a dual role. In addition to moving the sludge, they plough the solids, therefore, exposing new surfaces of the cake to the hot gases achieving more efficient heat transport. Sludge is fed at the periphery of the top hearth and is slowly agitated and moved towards the centre shaft, where it drops through holes to the hearth below. From the centre, the rabbles move it to the periphery where it drops through holes to the third hearth. This process continues to the subsequent hearths, initially drying the sludge and eventually, when the dry solids content is high enough, burning it. Air is admitted to the lowest hearth and circulates upward through the drop-holes in the hearths, counter currently to the sludge flow. The exhaust gases exit from the top of the furnace and pass to the air pollution control system.

Multiple hearth furnaces include three zones: the drying zone, combustion zone and cooling zone. The upper hearths of the furnace comprise the drying zone, where sludge water evaporates thus cooling the hot flue gases. The temperature rises through the drying zone where the sludge water evaporates, thus cooling the hot flue gases. The temperature rises through the drying zone from 430 to 760°C. The centre heaths comprise the burning zone where dry sludge combustibles (volatile gases and solids) are burned at temperatures 760 to 850°C. Fixed carbon is burned in the lower hearths of the furnace resulting in inert sludge accumulation. Ash is removed from the bottom of the furnace in a wet or dry form. After incineration the amount of ash is approximately 5-10% of the dewatered sludge.

Multiple hearth furnaces have been applied in wastewater treatment ever since 1935. However over the last years the construction of multiple hearth furnaces has diminished mostly due to air pollution and odour problems, high maintenance and labour requirements, high energy use and complexity of operation. The most critical issue regarding the application of multiple hearth furnaces is their compliance with air quality standards. Gases from multiple hearth furnaces exit the furnace at temperatures between 315 and 480°C and tend to vaporize many volatile compounds found in sludge, such as benzene, toluene, partially oxidized hydrocarbons, carbon monoxide and chlorinated substances, such as pesticides and PCBs. The concern over the presence of volatile organics in the atmosphere has made it necessary to raise the temperature of the exhaust gases in an afterburner to approximately 760°C to 820°C in order to destroy odour and hydrocarbon emissions. Afterburners may be installed either in the top hearth of the furnace or as a separate vessel. The dimensions of the vessel should be adequate to provide for a minimum of 1 to 2 seconds retention time. Due to short-circuiting when the top hearth is used as an afterburner, only 50% of the volume is used to estimate the retention time of the exhaust gases.

The cost of the additional auxiliary fuel needed to achieve high exhaust temperatures puts the multiple hearth furnaces at a disadvantage compared to fluidized bed furnaces. The operating cost for an afterburner may be substantial and can add 35 to 50$ per dry ton of solids.

<u>Fluidized bed furnace</u>: the typical fluidized bed furnace is a two-stage, concurrent process in which solids and flue gases flow upward. Under these conditions dewatered sludge cake with 20-30% DS will not ignite and can be burned only with auxiliary fuels. Dried cake with a dry solids content greater than 30-35%, depending on sludge characteristics and excess air level, is autogenous. The drying and combustion process occur simultaneously in a homogeneous stage known as bed zone. The second stage, called freeboard zone, provides additional time needed to complete combustion and oxidation of all volatile organic chemicals, therefore, reducing significantly odour emissions. The ability of fluidized bed (FB) furnaces to meet stringent air pollution standards and their high process reliability and stability has made FB process the most favourable choice (WEF, 1991).

The fluidized bed system consists of: a combustion vessel; a main air supply; a start up fuel system; a bed fuel system; a particulate collection system; an ash discharge system; and all the required instrumentation. The fluidized bed furnace is a vertical, cylindrical, refractory lined steel shell with a grid structure at the bottom surface to support the sand bed. The shell is divided into three distinct zones: (I) air inlet and distribution; (II) fluidized bed; and (III) freeboard with gas exhaust.

The bed contains sand particles with average diameter between 0.8 to 2 mm and a static bed depth of approximately 0.9 to 1.2 m. Air is supplied through openings at the grid located at the bottom of the vessel at pressures between 21-34 KN/m^2 to fluidize the bed and satisfy the requirements for the combustion of organic content of the sludge and the auxiliary fuel. Air is usually preheated by the exiting flue gas. When fluidized, the bed expands to about double its initial volume and acquires a fluid like behaviour that resembles boiling water. Fluidizing the bed creates a high degree of turbulence and agitation resulting in an increase in the surface contact area, a better mixing of the sludge particles, and an overall improvement in the combustion efficiency of the process.

Dewatered sludge is fed directly from the top or the sides of the bed. To achieve stable and more economic operation of the furnace, sludge feed should be evenly distributed on the bed surface. The sand bed is normally maintained at approximately 700°C. Auxiliary fuel is needed during start-up and sometimes may be used during operation, depending on the sludge feed characteristics. A temperature control system monitors the bed temperature and prevents fuel or sludge from being fed into the furnace if temperature is too high or too low, respectively. In the bed sludge, moisture evaporates and the volatile fraction combusts at temperatures between 760 to 820°C. A portion of the volatized organic matter is burned in the area above the bed causing a temperature increase of 40 to 170°C and resulting in freeboard temperatures between 840 to 900°C. Freeboard temperature is usually controlled by a series of water sprays mounted at the top of the incinerator. In case of significant increases in freeboard temperature the sludge and auxiliary fuel supply are stopped. Sludge ash and some sand particles are removed from the furnace with the exhaust gases. Approximately 5% of the design bed volume is removed by flue gases for every 300 hours of operation. The air pollution control system should be sized to account for this relatively high particulate load.

The high temperatures maintained in the freeboard zone and the size of the vessel provide sufficient retention time (approximately 3-5 sec), so that all the odour emission can be oxidized. However, high temperatures may result in increased NOx emissions and heavy metal fuming. A detailed discussion of the environmental impact of FB furnaces on air quality and the air pollution control systems required is presented in following paragraphs.

The sand bed acts as a heat sink within the furnace providing a significant thermal inertia in the system. Heat loss from the furnace is approximately 5-11^{o}C per hour. In this manner FB systems can handle greater variations in sludge characteristics, such as dry solids content or feed rate, compared to multiple hearth furnaces. In addition, FB furnaces can cope in a more energy-efficient way with temporary short (2-3 days) shutdowns.

Since 1980, fluidized bed furnaces have been used more frequently than multiple hearth furnaces. Reported advantages include the ability to meet stringent air quality standards, higher reliability and stability of operation and high combustion efficiencies in excess of 99%. Some of the operational problems associated with the application of FB furnaces include corrosion of air pollution control equipment and shell heat exchanger due to high operating temperatures and sludge drying within the sludge feeding unit.

5.3 Environmental impact assessment

Environmental assessment of incineration should cover all forms of environmental impact. The main potential impact is the release into the atmosphere of several hazardous pollutants. This aspect and the relevant legislation and control measures will be covered in detail later. Other issues include:

Visual intrusion: the visual impact of an incineration plant depends not only on its bulk but also on its design. The choice of processes, the massing of the individual elements and the extent to which they are exposed rather than hidden behind cladding; the resources devoted to landscaping including screening by earth sloping and tree planting; and the choice of surface materials and colours for buildings and structures.

It is prudent during the design of an incineration plant to devote high priority to its architectural design, even where this incurs increased costs. High stacks enhance the dispersion of the gaseous emissions but are bound to be visible over a wide area. A good balance should be reached with the aid of suitable mathematical models of dispersion. Furthermore, it is often required that visible emissions cannot exceed zero percent opacity. While this plume control requirement is mostly cosmetic, the perception of nothing coming out of the stack usually creates a strong, positive image.

Odours: because compounds that would cause offensive smells are destroyed in an efficient incineration plant, any risk of smells would be associated with fugitive emissions from the plant or from waste awaiting incineration. It is important that operators should take rigorous precautions, including regular inspections for leaks, to prevent the kind of incident which could produce odour problems and/or pose a health risk. It is also advisable that transfer and storage of sludge to be incinerated should take place within buildings that are maintained under reduced air pressure.

Noise: incinerators, in common with all process plants using mechanical equipment, will create some noise (for example, from fans in the gas cleaning and emissions systems). This noise process can be reduced to unobtrusive levels by careful design (e.g., sound absorbing partitions) and proper specification of equipment.

Traffic: journeys to bring in raw waste are more frequent than journeys to carry the ash for disposal to landfill. In cases where the production of dewatered sludge occurs in the vicinity of the incineration plant, the movement of vehicles is limited.

Socio-economic factors: while it has been recognized that an incineration plant may have a positive effect by adding to the employment and economic activity in a particular area, it has also been argued that it may have a negative effect on property values and some local industries, such as food processing. In view of the stringent emission standards, a well operated incineration plant should have little effect on the environmental quality of the locality or on property values, or should it cause any contamination of food produced or processed in the locality.

Storage of chemicals and fuels: the major chemical handling and storage systems at a wastewater treatment plant relate to liquid processing and sludge dewatering. A similar programme, assuring that there can be no damage to the environment from inadvertent chemical spills (solvents), is needed in the case of incineration. When fuel oil is used as a supplemental fuel, measures should be taken to prevent oil spills. The oil storage systems must have adequate berms, a synthetic liner and a leak detection system. These systems will increase the cost of fuel oil storage facilities but will greatly lessen the chance of an impact on the environment.

Fugitive dust: fugitive dusts most commonly result when the incinerator operates at a positive pressure, or when dry ash systems are not sealed properly and dust escapes. In these instances the ash can produce localized concentrations within or nearby the facilities, which can be harmful to individuals. In addition, transporting dry ash over surface roads can create fugitive dust if vehicles are not properly sealed. The burial of dry ash at landfills must be done carefully so as not to create dust during the discharge or covering. For these reasons it is recommended that a wet type ash system is adopted.

Carbon dioxide and the green house effect: sludge incineration, as well as drying, leads to the emission of carbon dioxide, which contributes to the greenhouse effect. However, when energy recovery from these processes is practised and used in substitution for fossil fuels, no net addition of carbon dioxide emissions results. It should also be taken into consideration that landfilling, an alternative disposal method, produces both carbon dioxide and methane. Methane has a particularly strong effect on the greenhouse phenomenon, about 7.5 times greater than the effect of an equivalent amount of carbon as carbon dioxide. Even with methane collection in landfills, approximately 50% of the gas leaks into the air and the impact on the greenhouse effect is far more significant than the impact of incineration.

Disposal of ash: while incineration provides the greatest reduction in the volume of sludge, there is still a significant quantity of material to be disposed of, and the quality of this product is of primary concern.

During incineration, the organic compounds in the sludge are destroyed. Experience shows that organics in the ash normally represent less than 1%, with PCDD's and PCEF's normally below detectable limits. The ash consists mainly of oxides of silicium, aluminium and iron. Heavy metals remain in the ash to the extent that they do not escape with the gases. Mercury fixation in the ash is very low (0.5%) because it remains in vapour from even at low temperatures. For most of the other heavy metals (Zn, Cu, Pb, Cr, Ni, Cd) it has been found that the percentages captured in the ash are high, between 60 - 100%, provided that an efficient flue gas treatment systems is in operation. Bearing in mind that there is a significant reduction of sludge mass during incineration, the concentrations of heavy metals in the ash are usually higher than the corresponding concentrations in the dewatered sludge (in terms of mg metal/kg sludge DS).

Despite the fact that the ash has increased concentrations of heavy metals, it has been found that landfilling is an acceptable disposal method which does not create significant pollution problems due to leachate, or in any case comparable to the problems created by sludge and solids waste landfilling. This is due to the fact that incineration transforms the mobile heavy metals into more mineralized, non-mobile, stable forms, thus preventing excessive leachate generation, as confirmed by several leach tests. Arsenic, cadmium, copper, mercury, nickel, lead and zinc were found to be washed out in amounts corresponding to less that 1% of the total content in the sludge ash, while larger amounts were found for calcium and molybdenum.

Two factors are of importance with respect to the wash-out of heavy metals. The one is related to the use of chemicals (Fe or Al) during wastewater treatment. It has been found that addition of these elements significantly increase solubility of Cd, Cr, N and Pb and possibly of other metals. The second factor is related to the adopted method of landfilling. Co-disposal of ash with municipal waste may promote leaching of heavy metals, due to the formation of organic acids resulting from anaerobic decomposition of the municipal waste (although the production of hydrogen sulphide and formation of insoluble sulphides of the metals partly counter-balances the effect of organic acids).

In view of the above, it is recommended that co-disposal should be avoided and that the ash from the incineration plant is disposed of to a separate landfill. However, some alternative possibilities could also be explored. Recovery of metals from the ash by chemical leaching can not be justified economically, but fusing ash using heat and cement may prove to be an attractive long term solution to control the heavy metal content of sludge ash. Several methods for melting sludge ash and production of slag in crystallized form are commercially available (O' Conor *et. al.*, 2001). The slag can subsequently be usedfor the production of materials suitable for the building industry, such as cement tiles, industrial abrasives, asphalt aggregates and bricks. It has been reported that, due to its high self-porosity, sludge ash works as a pore-forming agent, thus the potential benefit of this characteristic behaviour of sludge as an additive has to be evaluated in the light of the purpose of the building product itself (Weibusch *et. al.*, 1999).

Liquid effluents: these are mainly generated from the gas scrubbers and consist of water containing used reagents and some particles. Wastewater also comes, less heavily contaminated, from the tank used to cool hot ashes. These effluents can be returned to the inlet of the sewage treatment works, although it may be more prudent to provide a small "trade effluent" type plant to treat the spent scrubber liquid and avoid build ups of metals in the sewage works.

5.4 Emission of air pollutants - legislation and control

5.4.1 Main pollutants

Gaseous pollutants associated with sludge incineration include those solids that are vaporized because of the heat of combustion in the incinerator, or are compounds formed by the chemical reactions that occur during combustion in the incinerator. The main pollutants can be grouped as follows:

Products of incomplete combustion (PICs): these include compounds such as carbon monoxide, unburned hydrocarbons, volatile organic compounds and polycyclic organic matter such as dioxins and furans. The emission of these compounds is the result of the incomplete combustion of carbon in the sludge and auxiliary fuel, due to low temperatures and/or inadequate air for combustion.

Carbon monoxide can be absorbed in the human bloodstream and prevent oxygen uptake. Significantly high levels can cause death. Apart from dioxins (PCDD) and furans (PCDF), there are no regulatory limits on emissions of individual organic compounds. Instead, limits are placed on the overall concentrations of organic material in emissions as total organic carbon (TOC) or volatile organic carbon (VOC). The major component of TOC in stack gases is aliphatic compounds, which are unlikely to pose a threat to health. Volatile organic compounds contribute to the formation of ground-level ozone, although incineration is estimated to account for only a negligible portion of manmade emissions of these substances, about a tenth of the proportion is contributed by landfills. Concern has been expressed that several hundred aromatic compounds, which are potentially toxic, have also been identified, although from carcinogenicity studies it seems that organic compounds other than dioxins and furans account for only a very small part of the risk associated with an incineration plant.

Dioxins and furans (polychlorinated dibenzo-para-dioxins and polychlorinated dibentofurans) are formed as trace by-products in combustion of high temperature processes involving chlorine and organic compounds, including waste incineration. They can be detected in emissions from motor vehicles, power stations, domestic and accidental fires and the smelting of scrap metal. They are also formed during the manufacture of certain herbicides and wood preservatives. In industrialized countries the levels of dioxins and furans in urban areas are approximately four times higher than in rural areas. Municipal waste and sludge incinerators can contribute significantly to the total manmade releases. In the UK it was estimated that in 1989 about 20% of the total manmade releases of dioxins and furans was related to incinerators. However, it should be stressed that this significant contribution was the result of using old technology incineration systems without careful consideration of emission standards. The adoption of modern systems in compliance with the current strict emission standards can reduce the releases by more than 95%.

Dioxins and furans are stable and appear to break down only slowly in the environment. They can reach man by inhalation or absorption through the skin, but these pathways may be important only when released in close proximity to people. The background level of dioxins and furans in humans is almost wholly attributable to intake from food. Both volatile and particulate-bound dioxins and furans can be deposited either on plants grazed by farm livestock, from which they can enter meat or dairy products, or on fruit or vegetables for human consumption. If they are deposited on soil, however, they bind tightly to soil particles and do not readily enter the food chain. Within the human body, dioxins and furans are fat-soluble and are retained for long periods, especially in the liver and fatty tissue.

Assessments of the toxicity of dioxins and furans to humans are based on findings from studies of laboratory animals and records of accidental or occupational exposure. In animals, dioxins and furans are known carcinogens which modify cell growth and differentiation and appear to promote, rather than initiate, tumours. There is also evidence that they can adversely affect reproduction and the immune system of laboratory animals. Most of the occupational exposure studies on humans produced results difficult to interpret due to the fact that exposure has been to mixtures containing other chemicals as well. It has, however, been suggested that it is possible that exposure to dioxins and furans may be associated with the development of soft-tissue sarcomas. All these studies indicate that the toxicity of dioxins and furans to humans is not as high as has sometimes been suggested, although in summarizing the evidence the International Agency for Research on Cancer has classified the most toxic dioxin (2,3,7,8-TCDD) as a possible carcinogenic substance in humans. However, they have not been shown to be acutely or chronically toxic to humans in the concentrations likely to have been produced by emissions from incineration plants. The reduction in emissions that result from the new standards will enlarge the safety margins.

Nitrogen Oxides (NO_x): nitrogen oxides can take many forms, with the nitrogen atom joining with one, two or more oxygen atoms. NO_x emissions can result from the following two combustion reactions. During combustion at high temperatures, sludge-bound nitrogen can combine with excess oxygen to form NO_x. In addition, thermal nitrogen oxides are formed by atmospheric nitrogen reacting with excess oxygen at high temperatures, generally greater than 1100°C. The most harmful effect of NO_x is the formation of acid gas in the atmosphere when the gas combines with moisture. It also is a factor in smog formation.

Acid gases: sulphur dioxide is formed when the trace amount of sulphur present in the sludge is oxidized during incineration. Sulphur dioxide then combines with moisture to form sulphuric acid. Emissions from incineration plants are only a very small proportion of the total emissions due to other combustion processes. Hydrochloric acid is formed during incineration from the chlorine present in sludge. Additional acid gases formed in lesser quantities include hydrofluoric and hydrobromic acids. In health terms the main significance of acidic acids is that they can cause respiratory irritation, especially in susceptible groups such as asthmatics.

Heavy metals: metal emissions form a portion of the total particulate emissions from incinerators. The source of metals emissions is the ash content of the sludge and may be increased by chemical addition to the liquid treatment processes at the plant. Depending on the metal and its vapour pressure, the metals may leave the incinerator in a gaseous or solid form. Mercury, and to a lesser extent cadmium, are released mostly in the gaseous phase. The combustion process makes some metals more biologically active. For example, it converts some cadmium to the soluble chloride and sulphate salts and largely converts chromium to the hexavalent state, which is carcinogenic.

The primary route for human exposure to heavy metals released by incineration is the food chain. Mercury is deposited on the surface of water and soil. Emissions of cadmium to air are non-particulate material; after deposition on soil it may pass into vegetables and crops. Health effects reported have occurred in situations involving either high or sustained exposure to a particular metal. The evidence that lower doses can have specific effects on health is much less clear. Although no effects on health have been linked to the release of heavy metals from incineration plants, risk analyses have shown that the risks from metals in incinerators are higher than those from organic compounds. Despite the absence of direct evidence for health effects, there is a general policy of reducing exposures to heavy metals, which is reflected in the stringent current legislation concerning emissions of metals from incineration plants.

Particulates: solid pollutants consist of the particulate matter that is exhausted from incinerators. A portion of the ash, or all of it in the case of a fluidized bed furnace, leaves the furnace as fly ash. Suspended particulate matter can cause respiratory problems in humans and also has the effect of reducing outdoor visibility. Basic particulate emission standards were applied even to the old incinerators built 20 or 30 years ago, but the standards in the more recent legislation have become much stricter.

5.4.2 Indicative legislation

As already mentioned, plants built in the 1960s and 1970s were only required to achieve basic particulate emission standards and it was customary in most countries to regard sludge incineration as being environmentally safe but expensive. Increasing awareness by the general public of the environmental issues related to combustion processes in the 80s led to several revisions of environmental policies regarding incineration.

In Europe, Germany tends to be looked on by other European States as a major driving force behind EU Directives aimed at minimizing atmospheric pollution. It is worthwhile, therefore, to review the recent (over the last ten years or so) European legislation in connection to the corresponding German legislation.

The EC Council Directive on the prevention of air pollution from industrial plants (84/360/EEC, issued in 1984), was intended to facilitate the removal of disparities in national legislations concerning air pollution from industrial installations. The Directive requires that plants in certain categories, including sewage sludge incinerators, be authorized by the competent authorities at the design stage. Application for authorization will have to include a description of the plant, containing all information necessary for the authority to establish that appropriate preventative measures against air pollution have been taken, using the best available technology, provided that this does not entail excessive cost. The authority must also be satisfied that the plant will not cause significant air pollution, particularly by the emission of a number of listed substances which include SO_2, NO_x, CO, hydrocarbons, heavy metals, dust, chlorine and fluorine and their compounds. Subsequent “daughter” directives were to be issued to introduce requirements and emission limits relating to specific types of plant or more general standards.

At approximately the same time, the basic enabling law in Germany was implemented through an ordinance, which in the form of the 1986 revision is known as the Technical Instruction on Air Quality Control: TA Luft. One of the main principles of the Ordinance, the principle of precaution (Vorsorgeprinzip), is implemented through setting limit values to some of the key operating conditions of a combustion process and the concentrations of named pollutants in the stack gas emitted into the atmosphere. These limiting values are reflective of what was considered to be achievable using the then state of the art technology. A listing of the limiting standard values of greater relevance to sewage sludge incineration is provided in Table 5.1. With respect to operating conditions, it is required that a minimum flue gas temperature of 850^0C and an oxygen content of 6% (v/v) are maintained in order to ensure that emissions of products of incomplete combustion are kept at low levels. As can be seen from Figure 5.3, gas composition depends to a large extent on the operating temperature and it is clear that a temperature of approximately 850^0C is an integrated optimum. The flue gas temperature standard is a major influential factor affecting process selection. Whilst flue gas temperatures of 800^0 - 900^0C can quite readily be achieved using fluidized bed technology, this is not the case with the conventional design of a multiple hearth furnace from which the flue gases will emerge at lower temperatures, typically 300^0 - 500^0C. The flue gases can be raised to the higher temperatures required by afterburning, but at the expense of a large supplementary fuel burden. There is little doubt, therefore, that compliance with the emission standards for products of incomplete combustion, as set by TA Luft, shifts the balance heavily towards the adoption of a fluidized bed as opposed to a multiple hearth incineration technology.

TA Luft Ordinance has been the basis for several European national standards concerning sludge incineration during the second half of the 80s (Holland, Switzerland, etc.). Even more so due to the lack of a relevant legislation on an EU level. Following the 1984 EU “Framework Directive”, a EU directive “on the prevention of air pollution from new municipal waste incineration plants” was issued in 1989 (89/369/EEC). Furthermore, a draft proposal “on the incineration of hazardous waste” was issued in 1992 (92/130/01). The emission standards of the directive (for large plants) and the proposal are shown in Table 5.1.

It is interesting to note that neither the directive nor the proposal explicitly cover the case of sludge incineration. Since sludge has not been characterized as a hazardous waste, it is logical to adopt the 1989 incineration of municipal solids standards for sludge as well. This is the road taken by several countries (e.g., Denmark) although in others, intermediate standards between the 1989 EU Directive and 1992 proposal for hazardous wastes have been set. Two typical examples are the cases of Germany and UK. The new German standards used for sewage sludge and hazardous waste incineration are presented in the Seventh Ordinance 17. Blm Schv of 1990 (Table 5.1). These standards are significantly stricter than the EU directive of 1989, although not as strict as the 1992 EU proposal for hazardous wastes. On the other hand, the UK standards for sludge (HMIP, IPR5/3 of 1992, also presented in Table 5.1) do not consider sewage sludge as a hazardous material and are closer to the 1989 EU directive, although they go beyond EU requirements by setting limits on emissions of nitrogen oxides and dioxins, and setting more stringent limits for hydrogen chloride and metals.

5.4.3 Air pollution control systems

Despite the differences in the emissions standards either from country to country or between types of incineration plants (municipal solids, sewage sludge, hazardous waste), the requirements in all cases are stringent, thus calling for control systems along the lines of best available technology.

Control of odours and gases such as CO and NO_x as well as organics (hydrocarbons, dioxins and furans), is likely to be achieved through the selection of an appropriate furnace design (e.g., fluidized bed) and operation under suitable conditions (e.g., flue gas temperatures in the range 850 - 900^{0}C). This range of temperatures, as shown earlier, achieves low concentrations of CO, hydrocarbons and NO_x. Dioxins are also destroyed at temperatures over 850^{0}C. Unfortunately, the dioxins tend to reform in the gas stream at temperatures between 200 and 400^{0}C. Quick cooling of the gases (quench cooling) from 400^{0}C to 200^{0}C minimizes dioxin reformation. However, this means that the amount of recoverable energy in the waste heat boiler is reduced, since the heat recovery is limited to 400^{0}C instead of 200^{0}C in order to avoid dioxin reformation on the heat exchange tubes. As mentioned earlier NO_x control is possible through proper selection of furnace design and operational conditions. Post NO_x control is also possible through <u>catalytic decomposition and reduction</u>. Catalytic decomposition involves exposing the gases to a catalyst, resulting in the decomposition of NO_x to N_2. Similarly, exposure of the gases to a metal or carbon-based catalyst and use of ammonia as the reducing agent, reduces NO_x to N_2. Catalytic processes are capable of up to 90% NO_x removal. Disadvantages of the method include high cost and potential poisoning (fouling) by heavy metals. A more economical system, which can be used in incinerators if post NO_x control is required, is the thermal system which involves ammonia or liquid urea as a reducing agent but without the addition of catalysts. These systems are capable of 40 - 60% NO_x removal.

For dust control, three types of control systems can be used: <u>cyclonic separation, electrostatic precipitation and fabric filtration</u>. However, the last two methods can more easily and reliably achieve the required stringent limits with respect to particulates emission. Electrostatic precipitates use alternating rows of charging and collecting electrodes, which negatively charge the particulate and collect it via electrostatic attraction, on the adjacent row of positively charged electrodes. Once collected, the particulate is dislodged by mechanically striking the electrode, fluidization of the particulate and collection in storage hoppers below. Major factors that affect the performance of an electrostatic precipitator include particle size, loading and resistivity. In applications where dust resistivity and size, or other factors require an electrostatic precipitator that is too large and expensive, fabric filtration is a good alternative. The most common type of fabric filters are the tubular “bag style” filters that employ an internal support cage and filter on the outside surface of the cloth. For cleaning, compressed air is usually injected on the inside of the tube, shaking the accumulated dust of the filter rag, and into a collection hopper. Unlike electrostatic precipitators, fabric filters are not sensitive to the resistivity of the dust. Both electrostatic precipitators and fabric filters are considered best available technology by EU directives.

Table 5.1

Gas emission limits for sludge incineration

Parameter mg/Nm^3	Germany TA/LUFT (1986)	Switzerland	EC (89/369/EEC) (solid wastes)	EC for toxic sewage (1992)	UK HMIP,IPR5/3 (1992)	Germany 1,7 Blm Schv (1990)
Total Particulate matter	30	50	30	5	20	10
Carbon Monoxide	100		100	50	100	50
VOC	20(TOC)		20	5	20	10
Sulphur Dioxide	100	500	300	25	300	50
Hydrogen Chloride	50	30	50	5	30	10
Hydroxide Fluoride	2	5	2	1	2	1
Nitrogen Oxides (as NO_2)	500	1300	-	-	650	200
Cadmium			-		0.1	
Cadmium and Thallium			-	0.05		0.05
Cadmium and Mercury	0.2	0.2	0.2			
Mercury			-	0.05	0.1	0.05
Nickel and Arsenic	1	1*	1			
Other Heavy Metals	5	5	5	0.5	1	0.5
Dioxins			-	0.0001	0.001**	0.0001

* Nickel + Arsenic + Chromium + Cobalt + Selenium + Tellurium.
** HMIP's guidance note says that the emission of dioxins should be reduced as far as possible by progressive techniques with the aim of achieving a guide value of 0,0001 mg/Nm^3.

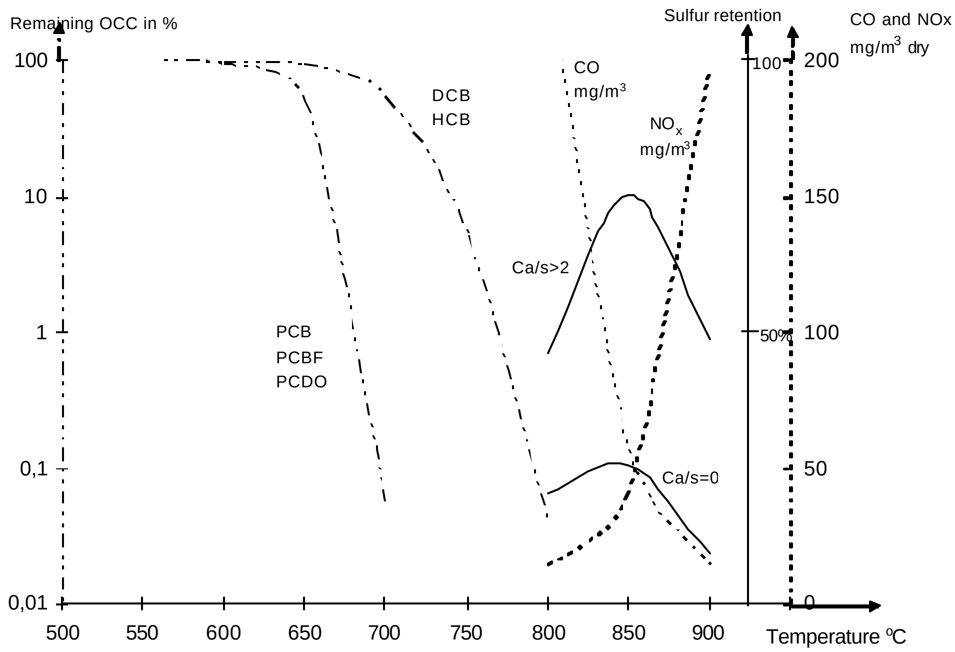

Figure 5.3. Variation of NO_x, SO_x, and OCCs in flue gases.

Control of acid gases can be achieved by dry-dry, semi-dry and wet scrubbing systems. Dry-dry systems contact a basic sorbent, usually lime, with the flue gas in a venturi where the dry sorbent reacts with the acids in the gas. The products of reaction and excess sorbent are trapped on the surface of a downstream fabric filter. Dry-dry systems are the simplest and lowest capital cost systems but have fairly low removals (especially for SO_2 analysis) and require large amounts of reagents. Due to their low efficiency it is doubtful whether they can meet the required for incineration plants emissions standards. Semi-dry systems are more efficient with removal of SO_2 over 90% and of HCl and HF over 99%. Semi-dry systems expose finely atomized lime slurry to the flue gas in a reactor vessel. The lime slurry, which is atomized using compressed air, has a large surface area where the acid gases are effectively adsorbed into the droplets, allowing reaction of the lime slurry with the acid gases. The product is a dry powder consisting principally of calcium salts which is collected downstream with the acid of an electrostatic precipitator or fabric filter. Wet scrubbing systems are even more efficient (with removals of SO_2 over 98% and of HCl and HF over 99%) and are less expensive than semi-dry systems. These are probably the reasons why wet scrubbers, despite the drawback of producing a liquid effluent requiring treatment, are widely used for incinerator flue gas treatment. Wet scrubbing systems re-circulate absorbent slurry through the scrubber vessel exposing large quantities of absorbent slurry to the gas. The large excess of water exposed to the gas lowers the gas temperature to its adiabatic saturation value. At this temperature all absorption of acid takes place in the efficient wet phase. High efficiencies can be achieved with only 3-5% excess reagent usage. Two commonly used reagents are lime slurry and the more expensive caustic soda (NaOH).

Control of emission of heavy metals can be quite dependent on their speciation and volatility. Metals of low volatility (copper, nickel, chromium etc.), may be evenly distributed in the fly ash and removed by the ash control devices. More volatile elements such as lead, cadmium, arsenic and especially mercury can be vaporized at incineration temperatures. At post boiler lower temperatures, some of these metals agglomerate into sufficiently large particles for efficient capture in the particulate control devices. The very volatile compounds, mercury in particular, are the more difficult to remove. Oxidized mercury (as $HgCl_2$) is highly soluble in water and, therefore can be removed in a wet scrubber operating at saturation temperatures in the range of 60 - 65^{0}C.

However, practically no elemental mercury is retained in the wet scrubbers and only a minor degree within the fly ash. A separate process for removing elemental mercury can be used, based on activated carbon adsorption. Reformed dioxins may also be removed through this process.

In order to prevent the formation of a steam plume at the top of the stack, the saturated gases are usually reheated. A gas-to-gas heat exchanger can be used, with the hot flue gases on the heating side.

6. SLUDGE TREATMENT PROCESSES

6.1 Introduction

Irrespective of the method of sludge disposal or reuse, some form of prior sludge treatment is needed. In Parts 3, 4 and 5 where the alternative sludge management practises are discussed, the treatment requirements for each alternative scheme are briefly presented.

This part is devoted to a more analytical description of the treatment processes/stages which can be suitably combined in order to provide the required overall treatment for each management scheme.

The various sludge treatment processes are grouped in three categories depending on their main function. The first category refers to methods which are mainly used for water removal and subsequent sludge volume reduction. The second category refers to methods which are responsible for organic matter destruction and partial pathogen removal (stabilization), and the last category refers to methods mainly used for practical elimination of pathogens (sanitization).

Incineration and other related technologies are not included in this part since they have been described in Part 5.

6.2 Treatments for water removal and volume reduction

6.2.1 Thickening

The purpose of thickening is to reduce sludge liquid volume and thereby reduce subsequent treatment and handling costs. The overall cost benefit of thickening is likely to be greatest where sludge is disposed of in liquid from. However, even in countries where sludge is normally dewatered to sludge cake before disposal, thickening is regarded as a benefit in reducing treatment and handling costs. The volume reduction obtained by sludge concentration is beneficial to the subsequent treatment process because: 1) the required tank volume and equipment capacity downstream are decreased; 2) the quantities of chemicals are reduced; and 3) the amount of tank volume required for the digester is lower. Thickening was initially practised by gravity (i.e., gravity thickeners). However, the presence of waste activated sludge in gravity thickeners significantly reduces the amount of sludge thickening. Thus, when both sludges are to be thickened before further treatment, it is more common to thicken primary and secondary sludge separately. Mechanical thickening methods such as flotation, centrifugal thickening, gravity belt thickeners are nowadays widely employed for WAS or for mixture of primary and WAS. However, it should be underlined that these alternative thickening processes are characterized by higher operational costs, require more skilled operators and have greater maintenance requirements compared to gravity thickeners.

Gravity thickeners: sedimentation thickening in circular tanks with either batch or continuous flow operation is often the most cost effective method for thickening of primary sludges and lime sludges. The presence of waste activated sludge may create problems due to the often poor settling properties of biological sludge and the increased biological activity. Many gravity thickeners may often experience odour problems and are often covered and provided for with odour control measures.

Gravity thickeners range up to 24 m in diameter with average depths from 3 to 4 m. The critical design parameter is the solids loading rate that depends on the type of sludge in the feed. For example, solids loading for primary sludges range from 100-150 kgSS/m^2-day whereas for mixtures of primary and WAS solids, loading should not exceed 70 kgSS/m^2-day. Gravity thickeners can concentrate solids in primary sludge by a factor of two times and can achieve thickened primary sludge solids content in the 5-10% range.

Dissolved air flotation: dissolved air flotation (DAF) thickening, concentrates sludge by releasing small bubbles of 50-100 µm in diameter that attach to the suspended particles and carry the sludge to the top of the tank. The floated solids, which have a thickness of several centimetres, are continuously removed by skimmers. Simultaneously, the heavier solids are removed from the bottom of the tank. To introduce small bubbles in the thickener tank a recycle flow, usually DAF effluent, is pressurized at 40-80 psi, before tank entry. Because air solubility increases with pressure, significant quantities of air can be dissolved. As the pressurized recycle is then pumped in the thickener, where its surface is at atmospheric pressure, the pressure release from the recycle forms small air bubbles. The principal use of DAF thickener has been to concentrate waste biological sludge from suspended growth processes or aerobically digested sludge. The float solids concentration usually ranges between 3-6% and depends mostly on air-to-solids ratio, sludge characteristics and particularly SVI, solids loading rate and polymer application. Polymer doses of 2-5 kg/ton of dry solids are common. For primary sludge and trickling filter sludge that have better settling characteristics, DAF is rarely employed because gravity thickening is a simpler and more economical method.

Centrifugal thickening: centrifugal thickening is a process in which sedimentation is accelerated through the use of centrifugal force (usually 500-3,000 times the force of gravity). Suspended particles under the influence of centrifugal force are removed from the surrounding liquid. Thickening by centrifugation is normally applied to waste activated sludge and other biological sludge. Primary sludge is rarely thickened with centrifuges because of its good settling characteristics that allow the use of more economic gravity thickening and because of the tendency to contain abrasive materials that can be detrimental to centrifuges. Centrifugal thickening can produce thickened sludge with a solids content from 2 to 10%. Although not required centrifuge performance and especially solids capture can be improved by polymer addition at doses that range from 0.5-5 g polymer/kg of dry solids. Generally the high maintenance and power costs for centrifugal thickening as well as the requirements for highly trained personnel make this thickening process attractive only to large WWTP facilities and in areas where space is very limited.

Gravity belt thickeners: gravity belt thickeners are a recent development that resembles the operations of dewatering belt filter presses. Its operation is based on the principle of separation of free water from the sludge by gravity drainage through a porous horizontal belt. Efficient operation of gravity belt thickeners requires addition of polymer at doses that range from 2-5 g/kg dry solids. The critical design parameter for the selection of appropriate belt thickeners is the solids loading rate that ranges from 100-500 kg SS/m-hr. Gravity belt thickeners have a wide range of applications for thickening primary sludge, waste activated sludge, digested sludge and even difficult to thicken chemicals sludge. Through proper adjustment of operating parameters, waste activated sludge and digested sludge can be thicken to a solids content of 4 to 8% and 4 to 10%, respectively.

6.2.2 Dewatering

Dewatering is the process employed to remove water from liquid sludge resulting in a non-fluid end product with a solids content of 15 to 35%. Dewatering is performed primarily to decrease the volume of sludge to be further processed or disposed of and thus to decrease the capital and operating costs of the subsequent stages of sludge treatment and disposal. Specifically, the following reasons constitute sludge dewatering one of the most frequently used methods in wastewater treatment plants: 1) sludge disposal costs are significantly lower due to a 60 to 80% volume reduction of sludge; 2) dewatering increases the energy content of sludge and can significantly reduce the operating cost of incinerators; 3) dewatered sludge is easier to handle than liquid sludge; and 4) moisture removal is required prior to certain disposal and reuse methods (e.g., composting, landfilling in monofills, etc.), and in some cases may be required to reduce the risk of odour generation.

There are a number of mechanical and natural methods of sludge dewatering that include centrifuges, belt filter presses, pressure filters, drying beds and lagoons. In developed areas with temperate or cold climate, natural methods for sludge drying are becoming unpopular and are usually replaced by mechanical methods. Natural methods are mostly employed in developing countries where they lack technical and economic resources to employ more expensive methods. The advantages and disadvantages of the various methods are summarized in Table 6.3 and discussed briefly in the following paragraphs.

Mechanical Dewatering: the main mechanical dewatering systems include vacuum filters, belt-filter presses, filter presses and centrifuges. The use of vacuum filters of sludge dewatering is constantly in decline because of high operating costs, relatively low efficiency and inability to produce high solids cakes, and system complexity.

Belt filter presses are continuously-fed sludge dewatering devices that achieve sludge dewatering through a combination of gravity drainage and compression. Although belt filter press performance depends on many parameters, such as polymer dose and sludge type, the process is capable of producing a cake with a solids content ranging from 25 to 30% using polymer doses in the 1-10 kg/tn of dry solids range. Total solids recovery is approximately 80-95% for sludge loading rates that range from 50 to 550 kgSS/m-hr. Typical design data for belt filter presses are shown in Table 6.1

Pressure filtration is a strong contender as a method of dewatering when cake production with a high solids content is important. In cases where cake solids greater than 35% are desired, filter presses can be the most cost-effective dewatering option. Pressure filtration is a batch process that separates solids from liquid by exerting a high pressure ranging from 100 to 225 psi. This way the liquid is forced through recessed plate pressure filters and the concentrated sludge is left trapped between the plates. Following the pressure application stage, the liquid drains into internal piping and is removed and the dewatered cake drops from the plates into a conveyor belt once the plates are opened. Advantages claimed for pressure filtration, in addition to high cake solids content, are high solids capture and filtrate clarity. The main disadvantages of filter presses are high capital cost, high operating cost, the need for chemical conditioning, complexity of operation and sludge storage requirements.

Dewatering by centrifugation and specifically by solid bowl centrifuges, is particularly popular in some developed countries (e.g., Denmark, Finland, Sweden, etc.), and is generally accepted as a feasible alternative for sludge dewatering. The principal of centrifugation is similar whether centrifuges are employed for thickening or dewatering and has been described in previous paragraphs. Advantages of the centrifuges compared to other mechanical dewatering devices include minor odours, high average cake solids, low average maintenance and high safety record. Disadvantages are the high operating cost and the rapid wear of the centrifuge caused by abrasive materials (e.g., sand). Typical dewatering performance of solid bowl centrifuges is presented in Table 6.2 (Metcalf & Eddy, 2002). Thickening performance depends on the sludge characteristics, with primary sludge generally exhibiting better dewatering characteristics compared to waste activated sludge. The average cake solids content for digested primary and WAS range from 20 to 25%. As reported in literature (WPCF, 1987) chemical conditioning may not always be required. However, with high polymer doses performance can be significantly enhanced and cake solids concentration can reach as high as 35 to 40%.

Table 6.1

Typical performance data for belt filter presses (WEF, 1991)

Type of sludge	Dry feed solids %	Loading per metre belt width		Dry polymer[a], g/kg dry solids	Cake solids, %	
		L/s	kg/h		Typical	Range
Raw primary (P)	3-7	1.9-3.2	360-550	1-4	28	26-32
Waste activated sludge (WAS)	1-4	0.6-2.5	45-180	3-10	15	12-20
P + WAS (50:50)[b]	3-6	1.3-3.2	180-320	2-8	23	20-28
P + WAS (40:60)[b]	3-6	1.3-3.2	180-320	2-10	20	18-25
P + Trickling filter (TF)	3-6	1.3-3.2	180-320	2-8	25	23-30
Anaerobically Digested						
Primary	3-7	1.9-3.2	360-550	2-5	28	24-30
WAS	3-4	0.6-2.5	45-135	4-10	15	12-20
Primary + WAS	3-6	1.3-3.2	180-320	3-8	22	20-25
Aerobically Digested:						
Primary+WAS (1:1) Unthickened	1-3	0.6-3.2	135-225	2-8	16	12-20
Primary+WAS (1:1) Thickened	4-8	0.6-3.2	135-225	2-8	18	12-25
Oxygen activated WAS	1-3	0.6-2.5	90-180	4-10	18	15-23

[a]Polymer needs based on high molecular weight polymer (100% strength, dry basis).
[b]Ratio is based on dry solids for the primary and WAS.

Table 6.2

Typical performance of solid bowl centrifuge (WEF, 1991)

Sludge type	**Feed solids Concentration (%)**	**Average cake solids concentration (%)**	**Dry polymer used per dry feed solids**	**Solids recovery (%)**
Raw primary	5-8	25-36	0.5-2.5	90-95
	5-8	28-36	0	70-90
Anaerobically digested primary	2-5	28-35	3-5	98+
	9-12	30-35	0	65-80
	9-12	25-30	0.5-1.5	82-92
Waste activated	0.5-3	8-12	5-7.5	85-90
Aerobically digested waste activated	1-3	8-10	1.5-3	90-95
Thermal conditioned primary and	9-14	35-40	0	75-85
waste activated	13-15	29-35	0.5-2	90-95
Primary and trickling filter	7-10	35-40	0	60-70
		30-35	1-2	98+
High lime	10-12	30-50	0	90-95
Raw primary and waste activated	4-5	18-25	1.5-3.5	90-95

Natural drying systems: when land is available and climatic conditions are favourable, sludge dewatering by natural methods may be an attractive alternative due to low capital cost, simplicity of operation and low energy requirements. However, over the last years there is a general decline in the use of drying beds, especially in urban areas, because of: 1) the intensive labour requirements; 2) the uncontrollable factors (rainfall, temperature, etc.) that affect dewatering by natural systems; and 3) the large areas needed. Two dewatering methods are considered natural: drying beds and drying lagoons.

Sand drying beds have been used for more than seventy years in many developed and developing countries. In many cases and especially for small and medium size treatment plants sand drying may be a preferable dewatering method compared to mechanical dewatering. Table 6.3 lists some of advantages and disadvantages of the drying beds.

The major type of sand drying is conventional sand beds. Conventional sand beds are usually employed for cities with less than 20,000 p.e. Conventional sand beds are usually rectangular with side walls enclosing sand and gravel. Drying beds are equipped with under-drainage piping to collect the water draining from sludge. In some cases depending on the weather drying beds may be covered with greenhouse types of enclosures.

Area requirements range from 140 m^2/1000 p.e. for primary sludge to 240 m^2/1000 p.e. for digested primary and waste activated sludge. The accepted solids loading criteria range from 50 to 125 kgSS/m^2/year for open beds and 60 to 200 kgSS/m^2/year for enclosed beds. The time required to achieve a cake with sufficient solids content varies according to climatic conditions, initial sludge solids content and drainage system. For relatively hot and dry climates, drying time may be as low as 10 to 15 days.

Table 6.3

Comparison of alternative sludge-dewatering methods(Metcalf & Eddy, 2002)

Dewatering method	Advantages	Disadvantages
Centrifuge	Clean appearance, minimal odour problems, fast start-up and shut-down capabilities. Easy to install. Produces relatively dry sludge cake. Same machines can be used for both thickening and dewatering. Chemical conditioning may not be required. Very flexible in meeting process requirements. Excellent results for difficult to dewater sludge.	Scroll wear potentially a high-maintenance problem. Requires grit removal and possibly a sludge grinder in the feed stream. Skilled maintenance personnel required. Moderately high suspended-solids content in centrate. Limited size capacity. Consumes more energy per unit of sludge dewatered. High capital cost. Vibration.
Belt filter press	Low energy requirements. Relatively low capital and operating costs. Less complex mechanically and easier to maintain. High-pressure machines are capable of producing very dry cake. Minimal effort required for system shutdown.	Hydraulically limited in throughput. Requires sludge grinder in feed stream. Very sensitive to incoming sludge feed characteristics. Short media life as compared to other devices using cloth media. Automatic operation generally not advised.
Recessed plate filter press	Highest cake solids concentration. Low suspended solids in filtrate.	Batch operation. High equipment cost. High labour cost. Special support structure requirements. Large floor area required for equipment. Skilled maintenance required. Additional solids due to large chemical addition require disposal.
Sludge drying beds	Lowest capital cost method where land is readily available. Low requirements for operator attention and skilled personnel. Low energy consumption. Low to zero chemical consumption. Less sensitive to sludge variability. High solids content in some cases.	Requires large area of land. Requires stabilized sludge. Design requires consideration of climatic effects. Sludge removal is labour-intensive.
Sludge lagoons	Low energy consumption. No chemical consumption. Organic matter is further stabilized. Low capital cost where land is available. Least amount of skill required for operation.	Potential for odour and vector problems. Potential for groundwater pollution. More land-intensive than mechanical methods. Appearance may be unsightly. Design requires consideration of climatic effects.

Other types of drying beds, though with limited use, are: 1) paved drying beds with a centre sand drainage strip with, or without heating pipes and with or without covering; 2) artificial media drying beds employing artificial media such as stainless steel wire or high density polyurethane; and 3) vacuum assisted drying beds equipped with a vacuum system to accelerate dewatering.

According to EPA guidelines for sludge disinfection, anaerobic digestion followed by dewatering on sand drying beds may provide sufficient pathogen inactivation to render the final product suitable for unrestricted reuse, assuming that the sludge contains no other contaminants (e.g., heavy metals, etc.). Based on a survey conducted by the Thames Water

Authority, approximately 33% of samples from drying bed cakes contained *salmonellas* with average counts of 46/100 mL (Pike, 1983). *Salmonella* occurrence appeared to be a function of drying time with complete inactivation of *salmonella* being obtained after 75 days.

Drying lagoons are similar to drying beds. Sludge is placed in pits of 0.75 - 1.3 m, where it is allowed to dewater and dry through evaporation for a period of one to three years. Lagoons are not suitable for dewatering untreated sludge, limed sludge or sludge in general of high organic content because of the potential odour and nuisance problems that may be created. Sludge is usually removed mechanically at a solids content from 25 to 30%. Typical solids loading rates range from 35 to 40 kgSS/m^3-year that approximately correspond to 2 m^3/capita.

Pathogen reduction has been recognized for years as a side benefit of sludge storage in lagoons. A 99.9% reduction of faecal coliform concentration has been obtained after 30 days of storage. Similarly it has been reported (EPA, 1979) that two to three order of magnitudes reduction of faecal coliforms can be achieved during long-term storage of anaerobically digested primary and waste activated sludge.

6.2.3 Conditioning

A conditioning stage is usually employed in many sludge processing schemes in order to improve sludge dewatering characteristics and enhance water removal. Conditioning is most often applied by chemical addition prior to sludge thickening or sludge dewatering facilities.

Chemical conditioning results in the coagulation of sludge solids and release of absorbed water. Chemical conditioning involves use of inorganic chemicals, such as ferric chloride, alum, lime and organic polyelectrolytes or both. The application of organic polyelectrolytes over the last 20 years has allowed the use of very efficient dewatering equipment that can dewater sludge up to 20-35% solids content. Although much more expensive polyelectrolytes are frequently preferred to inorganic chemicals because of the small doses required, 1 -10 kg/ton of dry solids versus 20-100 kg inorganic chemicals per ton of dry solids, their efficiency and the small increase in solids production obtained with their use.

6.3 Treatment for organic matter reduction and partial pathogen removal (sludge stabilization)

The main purpose of sludge stabilization is to make sludge less odorous, reduce or eliminate the potential for putrefaction and reduce the sludge pathogen content. The success of a process in achieving these objectives can be evaluated by measuring two critical parameters: 1) the per cent of volatile solids destruction; and 2) the pathogen indicator organism reduction. The most frequent methods used for sludge stabilization are:

- anaerobic mesophilic digestion (biological); and
- aerobic psychrophilic digestion (biological).

Chemical treatment by lime addition is often referred to as a stabilization process, although organic matter reduction does not take place and the purpose is to make the sludge less odorous and destroy pathogens. On the other hand, methods such as thermophilic anaerobic and thermophilic aerobic digestion, as well as composting, though resulting in significant organic matter reduction are mainly used for practical elimination of pathogens. Therefore, none of these methods are considered in this section and are discussed under the section related to sludge sanitization methods.

In selecting a sludge stabilization process for a wastewater treatment facility, the engineer has to take into account many factors: 1) the objectives of the process; 2) the method(s) of ultimate sludge disposal or reuse and the relevant national or local regulations; 3) the available technical and economic resources of the owner; 4) the proximity to neighbours and the potential for odour production; and 5) the environmental concerns of the public. All these factors are vital to the proper selection of a sludge handling system.

6.3.1 Anaerobic digestion

Anaerobic digestion is the biological degradation of organic substances in the absence of oxygen. It involves complex biochemical processes in which facultative and anaerobic bacteria breakdown organic matter, release energy and produce methane gas, carbon dioxide and water as end products. Anaerobic digestion, though a relatively complex method requiring proper design and careful operation, is the most common process of sludge stabilization. Its wide use over the other stabilization processes is due to the following factors:

- low bacteria yield and, therefore, low sludge production;
- production of energy in the form of methane gas;
- high volatile solids reduction ranging from 30 to 60%; and
- production of well stabilized sludge, suitable for use as soil conditioner with a relatively low risk of odour nuisance.

On the other hand, the following disadvantage of the process may result in selection of alternative methods of sludge stabilization (EPA, 1979):

- high capital cost;
- susceptibility to upsets and process failure from inhibition and solids overload;
- production of a supernatant stream with high organic, nitrogen and phosphorous content that is usually recycled to the wastewater process stream and may overload the plant; and
- large reactor volume requirements in order to achieve high solids retention time. Due to the low growth rate of methanogenic bacteria, larger digesters are required to provide sufficient sludge retention time.

The main parameters for the design and operation of anaerobic digesters are solids retention time (?c) and temperature. According to these two design parameters anaerobic digesters can be classified into:

- low rate, cold digestion (?c = 1-2 years, T = 10-20°C);
- mesophilic digestion (?c = 15-25 days, T = 34-36°C); and
- thermophilic digestion (?c = 5-15 days, T = 54-57°C).

Thermophilic digestion is discussed under the section dealing with sanitization processes.

<u>Low rate cold anaerobic digestion</u>: cold anaerobic storage of sludge is often used as a low tech treatment to stabilize primary and waste activated sludge, or even as an additional stabilization and disinfection method for digested sludge. Anaerobic cold digestion can be performed either in large tanks that are not equipped with mixing and heating devices or in lagoons. During storage, besides volatile solids destruction, the viability and infectivity of most pathogens also declines.

Cold anaerobic digester and anaerobic lagoons employ similar microbiological processes. Both treatments operate at ambient temperatures normally not exceeding 20-25°C. The differences between the two processes are mostly related to the design of each system. The retention periods for both systems are usually many months so that methanogenesis and microbial breakdown of organics that occur slowly can be completed, and ultimately be approximately the same as those obtained in mesophilic digestion. Anaerobic lagoons are normally relatively shallow, earth-banked or concrete faced reservoirs for long term storage of liquid sludge. The average retention time is two years or even longer. Horizontal mixing is poor and the retention time is difficult to control because of dead spaces. Cold digesters are usually constructed as deep concrete tanks and mixing occurs only due to methane gas production by bacterial action. Under summer temperatures stabilization may take about six months and retention times of nine months are typical.

Anaerobic sludge storage may be regarded as a passive method of disinfection. At a temperature of 20°C or greater, 90% reduction of bacteria, parasite ova and viruses, required one month, six months and more than two months, respectively. At lower temperatures these times were increased to greater than six months, at least three years and greater than eight months. Table 6.2 shows that under temperate climates, lagooning of sludge can be quite effective in *salmonella* reduction, particularly as a supplementary disinfection step following anaerobic digestion of sludge. According to a study conducted in the UK only 4% of the sludge samples contained any *salmonella* after storage of longer than two years. However, storage time is critical as according to the same study, when sludge was stored anaerobically for less than two years *salmonella* was detected in 25% of the sludge samples. Similar findings were reported that ova of *Ascaris* and *Toxocara* canis survived during storage in sewage sludge and remained infectious for 16 to 25 months.

Mesophilic digestion: mesophilic digestion at around 35°C is the most commonly employed method of sludge stabilization, especially for medium and large wastewater treatment plants. During the 60s this process went through a phase of unpopularity due to some cases of process failure caused by toxic organic and inorganic chemicals or inefficient operation of the digester. However, understanding of the inhibition effects that various substances exert on anaerobic digestion, as well as improved design and control of the process, have resulted in turning anaerobic digestion into the most often employed method of sludge stabilization. Furthermore, because of emphasis on energy conservation and recovery, the economics of using methane gas for onsite energy generation are favourable and, therefore, create an additional reason for selecting this process.

The oldest type of mesophilic digesters, frequently called standard rate or conventional, were unmixed. As a result, the contents of the tank are stratified and the four zones form a scum layer, a liquid supernatant, a layer of actively digesting solids and a layer of thickened digested sludge and inert solids. Because only a small part of the digester is active, large reactor volumes are required and typical solids retention times are high, ranging from 30 to 60 days. Furthermore, because operating conditions are not controlled the digestion process can be unstable and inefficient. Therefore, most designers avoid the use of standard rate anaerobic digester for primary process of sludge stabilization and very few facilities remain in use in Europe and the US.

Table 6.2

Effect of anaerobic lagooning on geometric mean counts of *salmonella* at five works

	Types of sludge treatment	Period of sampling	*Salmonella* per 100 mL (and no of samples)		Removal %
			Input	Output	
1.	Mixed primary and activated sludge, mainly domestic. Retention up to 1 year; sludge is mostly removed in late spring and summer.	Over 12 months	770 (87)	23 (87)	97
2.	Primary sludge, mainly domestic, retention up to 6 months.	April-September	140 (17)	12 (14)	91.2
3.	Domestic; 5 lagoons in series retention up to 1 year.	-	827 (4)	26 (4)	96.9
4.	Domestic sludge, mesophilically digested 10-15 d, lagooned 6-8 weeks.	-	325 (4)	2.6 (4)	99.2
5.	Mixed domestic and industrial. Sludge circulated between two lagoons, retention up to 6 months.	-	89 (4)	9.0 (4)	99

Nowadays, high rate digestion is almost exclusively employed as a method of anaerobic mesophilic digestion. High rate digestion is characterized by supplemental heating and mixing, uniform feeding rates and thickening of the raw sludge. These four features usually create a steady and uniform environment were biological processes can be optimized. As a result, the tank volume required for adequate digestion is reduced and the stability of the process improved.

Mixing is very important for the efficient operation of the digester because it promotes contact between raw sludge and active biomass, it dilutes any potential inhibitory substances through the entire volume of the tank, it eliminates any temperature and solids stratification and controls formation of a surface scum layer. Mixing systems for high rate digesters are typically designed to achieve a digester working volume of 85 to 95%. The available types of mixing systems include: mechanical mixers, pump circulation and compressed gas release. The heating system of an anaerobic digester is one of the most important parts of the treatment process as temperature influences significantly the rate of stabilization. Mesophilic digesters operate between 34°C and 38°C. From a design point of view, both establishing and maintaining the design operating temperature are critical as temperature variations greater than 1.0°C can result in process failure. This is because anaerobic organisms may easily be inhibited by small temperature changes. The most frequently used heating methods are external heat exchanger through which sludge is circulated, followed by internal exchanger through coils placed inside the tank, stream injection and direct flame heating.

To maintain constant conditions in a digester it is advisable to feed the digester at a constant rate. Although continuous feeding is seldom practised, design of the sludge feeding system should be such that it allows for sludge to be added to the digester on a frequent basis (e.g., every two hours). The feeding protocol can significantly influence the inactivation rate obtained during digestion. Feeding and mixing the digester for a short period before withdrawing sludge appeared to increase the concentration of the pathogen organism indicator by 10 to 50 times compared to withdrawing digested sludge before feeding.

Sludge thickening reduces flow through the digester so that the design solids retention time can be achieved with smaller tank volumes. However, there is a point beyond which further thickening of feed sludge may have a negative impact on digestion efficiency because: 1) mixing is impaired at solids concentrations greater than 6%; and 2) increased concentration of inorganic and organic potentially inhibitory substances.

The key parameters for the design and sizing of anaerobic digesters are solids residence time (SRT) and volatile solids leading. Both these parameters directly influence the size of the anaerobic digester, which should be sufficient to ensure that SRT never falls below a certain critical value so that the desired level of sludge stabilization will be maintained. The required volatile solids loading rate and solids retention time in mesophilic digesters depend on many factors, such as type and digestibility of sludge, temperature pH, presence of potentially toxic substances, etc. As shown in Table 6.3, typical design values for volatile solids loading and SRT are in the range from 1.9 to 2.5 kgVS/m^3-day and from 15 to 30 days, respectively.

Table 6.3

Typical design criteria for sizing mesophilic anaerobic sludge digesters (EPA, 1979)

Parameter	Low-rate digestion	High-rate digestion
Volume criteria, m^3/1000 capita		
Primary sludge	56-84	36
Primary sludge + Trickling filter humus	112-140	76-92
Primary sludge + Activated sludge	112-168	76-112
Solids loading rate, kg VSS/day/ m^3	0.64-1.6	2.4-6.4
Solids retention time, days	30-60	10-20

Assuming that a mesophilic digester is designed and operated properly it should achieve two primary functions: sludge stabilization and pathogen reduction. Sludge stabilization is normally determined as the percent of volatile solids destruction. Biological sludge is usually more difficult to degrade compared to primary sludge that contains more readily degradable organic matter. As a result, higher volatile solids destruction efficiencies ranging from 50-60% are obtained with primary sludge VS 20-25% with secondary sludge. Volatile solids destruction of 40-60% are typical with mixtures of primary and biological sludge. The final product of volatile solids destruction is anaerobic digester gas that typically consists of methane (55-78%), carbon dioxide (25-45%), nitrogen (2-6%), hydrogen (10.1-2%) and hydrogen sulfide. Approximately 750-1000 L of methane gas, with an average value of 5,500 Kcal/m^3 are produced for every kg VS destroyed.

Well operated anaerobic digesters can achieve significant reduction in virus and bacteria levels but are less effective against parasitic cysts. Solids retention time, temperature and method of operation are apparently the most important factors effecting virus, bacteria and parasite removal. Pathogen occurrence in raw and anaerobically digested sludge is shown in Table 6.4 (EPA, 1979). Anaerobic digestion has been shown to reduce the concentration of detectable viruses by the order of one to four magnitudes. Similarly, bacterial counts during mesophilic anaerobic digestion are removed by two to three logs.

According to literature (EPA, 1979), the upper range of faecal coliforms and *Salmonella* in anaerobic digested sludge are 6×10^6/100 mL and 62/100 mL, respectively.

There is a wide variation in the parasite content of digested sludge depending on the type of parasite. Protozoa cysts are inactivated during anaerobic digestion. Stadterman et al (1995) reported 99.9% elimination of *Cryptosporidium* cysts by anaerobic thermophilic digestion after 24 hrs. They also reported complete inactivation of *cryptosporidium*, obtained in mesophilic anaerobic digesters operating at a 15d SRT. On the other hand, as illustrated in Table 6.4 only limited reduction of *Ascaris* and Helminths ova can occur during mesophilic anaerobic digestion.

Table 6.4

Pathogen occurrence in liquid wastewater sludge

Pathogen	**Name of species**	**Concentration, number/100 mL**	
		Unstabilized raw sludge [a]	**Digested sludge** [a,b]
Virus	Various	2.5×10^3 - 7×10^4	100 - 10^3
Bacteria	*Clostridia sp.*	6×10^6	2×10^7
Bacteria	*Faecal coliform*	10^9	3×10^4 - 6×10^6
Bacteria	*Salmonella* spp.	8×10^3	BDL [c] - 62
Bacteria	*Streptococcus faecalis*	3×10^7	4×10^4 - 2×10^6
Bacteria	Total coliforms	5×10^9	6×10^4 - 7×10^7
Bacteria	*Mycobacterium tuberculosis*	10^7	10^6
Parasites	*Ascaris lumbricoides*	200 - 1,000	0 - 1,000
Parasites	Helminth eggs	200 - 700	30 - 70

[a] Type of sludge usually unspecified.
[b] Anaerobic digestion; temperature and detention times varied.
[c] BDL is below detection limits, < 3/100 mL.

6.3.2 Aerobic digestion

Aerobic digestion of municipal wastewater sludge is based on the principle that cells under aerobic conditions and with inadequate external food sources will oxidize their own protoplasm to obtain energy for cell maintenance. Aerobic digestion products are mainly carbon dioxide, water, non-degradable organic matter and oxidized forms of nitrogen. Some advantages of the aerobic digestion are: 1) volatile solids reduction almost as high as the one obtained anaerobically; 2) lower BOD concentrations in supernatant; 3) low capital cost; 4) simple and safe operation; and 5) production of a relatively inoffensive and biologically stable end product. The major disadvantages of aerobic digestion include: 1) high operating costs associated with maintaining aerobic conditions in the digester; 2) poor dewatering of aerobically digested sludge; 3) reduced efficiency at low temperatures; and 4) loss of a potential energy source (methane gas).

Aerobic digesters receive waste activated sludge, or mixtures of waste activated sludge or trickling filter sludge and primary sludge. Generally, as the percentage of biological sludge increases, the tank volume and aeration requirements decrease. In temperate climates, retention of 15-30 days and in cold climates up to 50 days may be necessary to provide a volatile solids reduction as much as 35 to 50%. Typical design criteria for aerobic digesters are presented in Table 6.5. Due to high operating costs aerobic digestion is mostly used in wastewater treatment plants with a capacity of less than 18,900 m^3/day or 100,000 p.e.

Table 6.5

Design criteria for aerobic digesters (Metcal & Eddy, 2002)

Parameter	Value
Hydraulic retention time, at about 20°C, d[a]	
Waste activated sludge only	10-15
Activated sludge from plant without primary settling	12-18
Primary plus waste activated or trickling-filter sludge[b]	15-20
Solids loading, kg volatile solids/m^3.d	1.6 - 4.8
Oxygen requirements, Kg O_2/Kg solids destroyed	
Cell tissue	~2.3
BOD_5 in primary sludge	1.6-1.9
Energy requirements for mixing	
Mechanical aerators, kW/10^3 m^3	20 – 40
Diffused air mixing, $mt^3/10^3$ m^3.min	20-40
Dissolved oxygen residual in liquid, mg/L	1-2
Reduction in volatile suspended solids, %	40-50

[a] Detention times should be increased for operating temperatures below 20°C.
[b] Similar detention times are used for primary sludge alone.
[c] Ammonia produced during carbonaceous oxidation oxidized to nitrate.

The principal of aerobic digestion has been used successfully in extended aeration activated sludge systems. In many of these systems the sludge age is maintained higher than 15 days so that bacterial mass is in endogenous r espiration stage and aerobic digestion occurs within the liquid process stream. In many cases where primary treatment is omitted, sludge stabilization can occur within the aeration basin and, therefore, there is no need for separate sludge stabilization facilities. Aerobic stabilization as part of secondary sewage treatment is practised in many European countries like The Netherlands, Denmark, Greece and others.

Aerobic digestion of sludge at ambient temperatures can achieve some degree of pathogen inactivation, which is usually lower compared to pathogen inactivation obtained in anaerobic thermophilic digesters. According to Farrell and Stern (1975) average faecal coliforms and *Salmonella* concentrations in aerobically digested sludge were 7 x 10^7/100 mL and 1.5 x 10^4/100 mL, respectively. These values are higher than the maximum concentrations reported for anaerobically digested sludge. Additionally, as reported by Pike (1983) it is unlikely that the decay of parasite ova will be accelerated during aerobic digestion.

A more promising process in terms of bacterial inactivation, is the thermophilic (autothermic) aerobic digestion in which the heat produced from oxidation of sludge organic matter is conserved to a sufficient degree to produce a large increase in operating temperature. This process is discussed under the section relating to sanitization of sludge.

6.4 Treatment for sludge disinfection

The aim of sludge disinfection is to minimize the chance of pathogenic micro-organism transmission and thus minimize public health risk through sludge disposal and reuse methods. An additional concern is to minimize the exposure of domestic animals to pathogenic micro-organisms from grazing on pastureland receiving sludge. The commonly employed stabilization methods, described in previous paragraphs, achieve a certain degree of pathogen inactivation. These sludge treatment methods are partial barriers to the transmission of pathogens but in some cases, depending on the type of sludge reuse method, additional barriers may be justified in order to completely eliminate public health risks. The methods employed to achieve pathogen destruction beyond that achieved by the typical stabilization methods are the following: anaerobic thermophilic digestion, aerobic thermophilic digestion, composting, quicklime treatment at high pH and temperature, thermal drying, pasteurization and high energy irradiation.

An important aspect arising from the need to evaluate process efficiency and from formulating regulations for sludge disposal and reuse is to determine the monitoring requirements and type of indicator organizing that should be used. What pathogens are relevant will differ from one situation to another and consequently monitoring requirements may be case specific. However, it is generally accepted that *Salmonella* and infective parasite eggs are of primary importance in evaluating a sludge management scheme. Absence of these organisms is a very safe indicator of a tolerable risk (Strauch, 1987 and Havelaar, 1983).

6.4.1 Anaerobic thermophilic digestion

Thermophilic digestion occurs at temperatures between 50-57°C, conditions that enhance the growth of thermophilic bacteria. Operating in the thermophilic range accelerates the digestion process, thereby potentially reducing the solids retention time and consequently the digester volume requirements. Advantages claimed for thermophilic digestion include increased sludge processing capability, improved sludge dewatering, increased pathogen removal and increased scum digestion. However, because of higher energy requirements for heating, lower process stability, odours, and poorer quality of supernatant, thermophilic digestion is used infrequently.

Thermophilic digestion of sludge achieves, at the same solids retention time, approximately two to four logs greater removal of viruses and bacteria compared to mesophilic digestion. Based on field scale and bench scale anaerobic digestion experiments in the mesophilic and thermophilic range Watanabe *et al.*, (1997) concluded that mesophilic anaerobic digestion could not achieve a pathogen reduction level to attain the EPA Class A requirements. Only thermophilic digestion could achieve low enough concentrations of faecal coliforms and *Salmonella* to satisfy the EPA Class A requirements.

6.4.2 Aerobic thermophilic digestion

Thermophilic aerobic digestion can be used to achieve volatile solids removal up to 70%, at solids retention time of 3 to 4 days and operating temperature in the 35-50°C range. Based on European experience of thermophilic aerobic digestions, the key to successful operation of these systems are efficient aeration, adequate pre-thickening of feed sludge up to 2.5-5%, sufficient tank insulation, good mixing and foaming control. Compared to anaerobic mesophilic digestion, aerobic mesophilic digestion has a lower capital cost but a higher operating cost. Reported advantages of this process are good pathogen inactivation, low tank volume requirements in order to achieve a certain degree of VS destruction, process stability and a final end product with good dewatering characteristics. Thermophilic aerobic digestion can produce a hygienically unobjectionable end product and, according to the German Ordinance of Sewage Sludge, is one of the sanitation technologies that when properly operated can achieve a minimum four logs reduction in *salmonella* concentration, inactivate ascaris eggs and render them non-infectious, produce an end product that contains less than 1,000 FC/g solids and no *salmonella* per g of solids.

Another modification of aerobic digestion is a dual digestion process in which sludge is heated to 50-65°C for approximately one day and then subjected to anaerobic mesophilic digestion for 8 to 10 days. The advantages of using aerobic thermophilic digestion followed by anaerobic mesophilic digestion are: 1) satisfactory degree of pathogens kill as the first aerobic step is a type of sludge stabilization; 2) improved overall volatile solids destruction; 3) sufficiently stabilized end product; and 4) production of methane gas that may be utilized for energy generation.

6.4.3 Composting

Composting is an aerobic process in which biodegradable organic material is converted into a stable end product. If the composting process is sufficiently completed the end product is fully stabilized, has a very low potential for odour generation and may be reused in agriculture as a soil conditioner. The most important feature of the process is heat generation during decomposition of organic material. Approximately 20-30% of the sludge volatile content is converted to carbon dioxide and water with the immediate release of sufficient heat to raise the temperature to high enough levels (50°C – 70°C) to destroy weed seeds and pathogenic micro-organisms.

A variety of process factors can influence the composting system performance, which include: temperature, retention time, pH, moisture content, bed porosity, aeration efficiency and nutrient content. For efficient stabilization the material undergoing composting should have a moisture ratio between 20:1 and 30:1, a pH of 7 to 9 and should be uniformly porous to allow air distribution. Moisture content exceeding 30% will reduce the free pore space and result in anaerobic conditions. Aerobic conditions must be maintained throughout the sludge for rapid and odourless composting. On the contrary moisture content below 40% may adversely affect microbial activity. The carbon to nitrogen ratio (C/N) is a very important parameter for efficient composting. A C/N ratio exceeding 30 tends to inhibit the rate of organic decomposition resulting in an inadequately stabilized product. On the other hand, low C/N ratio of less than 20 results in ammonia release. To adjust the initial moisture content at the optimum 50-60% range, a bulking agent such as wood chips, leaves, sawdust and others, is added to dewatered sludge. In many cases, vegetative materials may act as a supplemental carbon source to increase the C/N ratio.

There are three types of aerated composting processes, namely: windrow, static pile and in-vessel (mechanical). These processes are differentiated by mechanical turning, container structure and air flow. All of these systems include some or all of the following steps:

- dewatered sludge is mixed with a bulking agent to increase porosity, create a porous material, achieve the desired moisture content and provide a supplemental carbon source;
- the mixture is aerated for a period of 15 –30 d, during which heat is generated as a result of microbial degradation of organic substances resulting in water evaporation and pathogen inactivation. Aeration supplies oxygen, controls temperature and removes water vapour;
- the bulking agent may be recovered by screening; and
- the compost is cured for an additional time to complete the process.

Windrow composting**:** the windrow system is the least complex of the composting systems. In the windrow system, a mixture of biosolids and bulking agent is placed in long rows (windrows) that are turned periodically using mobile equipment. The turning effect mixes the wet cake and dry compost and bulking agent, increases porosity in the windrow to maintain aerobic conditions, promotes drying of the sludge by exposure to air and sun and ensures that the sludge is subjected to the higher temperatures achieved at the interior of the windrow. The composting period is about 21 to 28 days.

When the material has reached at least 60% total solids (less than 40% moisture) and the volatile fraction has been reduced below 40%, the material is dry and stable enough to be used as a solid conditioner. The process takes several weeks, depending on the frequency of turning the windrow and the climate. Winter climate and inconsistent turning noticeably reduce maximum internal temperatures in the windrow, increasing the likelihood of pathogens survival.

<u>Aerated static pile composting</u>: the aerated static pile system employs a grid of aeration, or exhaust piping, placed below the pile to maintain aerobic conditions during composting. Typical land requirements for aerated static pile systems are 0.067 - 0.040 ha/ton dry solids composting per day (WEF, 1991). Static pile composting is comprised of four unit operations which are: mixing, aeration, curing and screening.

1. <u>Mixing</u>*:* mixing involves the combination of dewatered sludge and a bulking agent by mechanical means like a front end loader or a similar type of mobile equipment or a fixed blender.
2. <u>Composting</u>*:* following mixing the material is ready for the active composting operation. This operation includes stocking the material in piles and aerating. The goal in stocking the material is to produce a pile of uniform height that will allow proper aeration and the maximization of aerobic performance. The pile is aerated to supply the aerobic micro-organism with sufficient oxygen to accomplish organic stabilization and pathogen destruction. This composting procedure lasts approximately 21 days. Aeration is performed through piped laid in the ground connected to a fan.
3. <u>Curing</u>*:* after active composting the material is removed from the aeration system and placed in a curing pile. During curing, solids decomposition continues and temperatures initially remain elevated before falling.
4. <u>Screening</u>*:* screening involves the separation of the bulking agent from compost with the recovery of bulking agent for water use. The screening operation should commence when the material reaches the desired moisture content of less than 50%.

<u>*In*-vessel composting systems</u>: in-vessel composting is accomplished inside an enclosed container or vessel. The major goal is to provide optimum conditions for biological growth. In-vessel systems are designed to minimize odours and process time by controlling environmental conditions such as air flow, temperature. Another benefit lies in the fact that the climate does not significantly affect the determination. The detention time in the reactor varies from 10 to 21 days, depending on system supplier recommendations, regulatory requirements and costs.

<u>Pathogen inactivation</u>: the temperatures that can be reached in the composting process depend on many factors, but in general will be between 40 and 60 ^{0}C if the process is carried out in piles or up to 80 ^{0}C in bioreactors. Current EPA requirements for enclosed and aerated static pile systems stipulate that, the mass undergoing d ecomposition shall be subjected to a minimum temperature of 55°C for a period not less than 3 days to achieve essentially complete pathogen inactivation. Windrow systems require a minimum of five turnings and a minimum temperature of 55°C for 15 days. Under these high temperatures, composting can be effective in eliminating pathogens in the final end product. According to the German Ordinance of Sewage Sludge a composting is one of the sanitation technologies that when properly operated can achieve a minimum four logs reduction in *salmonella* concentration, inactivate ascaris eggs and render them non-infectious, and produce an end product that contains less than 1,000 FC/g solids and no *salmonella* per g of solids.

Table 6.6 shows pathogens and indicator organisms (faecal coliforms) in finished compost from three full-scale facilities. All the systems consistently produced an effectively pasteurized product with low or not detected pathogen and faecal coliform levels.

Table 6.6

Pathogens and indicator organisms in finished compost from selected composting facilities

Location	Salmonella MPN/g dry solids	Faecal coliform, MPN/g dry solids	System type
Schenectady, N.Y.	-	3-11	In-vessel
Montgomery Co., Md	0	0-4000	Static pile
Austin, Tex.	-	<4 - <33	Windrow

6.4.4 Treatment with lime

Lime is a readily available alkali, which is widely used in sewage treatment. Utilization of lime has the following benefits: a) conditions all types of sludge; b) precipitates toxic metals and removes nutrients; c) destroys pathogenic agents; d) reduces the biochemical and biological oxygen demand and suspended solids; and e) eliminates offensive odours. Depending on the point of lime addition within the sludge process scheme, lime can be used for conditioning, stabilizing and disinfecting sludge. Therefore, treatment with lime may be an attractive alternative for many small to moderate treatment plants in the Mediterranean region that may lack the technical and economic resources to proceed with other high tech alternatives (NTUA,1999 and Andreadakis, 2000).

Two forms of application of lime are usually distinguished: a) as unslaked (quick-) lime - CaO; and b) as slaked lime - $Ca(OH)_2$. Furthermore, during treatment of sludge, lime can be added to the sludge before thickening, before dewatering, or after dewatering. No significant difference has been observed between the effect of hydrated lime and quicklime when treating sludge with high water content, as is the case of non-dewatered sludge (Tullander, 1983). In reaction with water CaO forms $Ca(OH)_2$ within a few minutes. For this reason, if quicklime is used in sludge with a high water content, the same effect will be attained in practise as with hydrated lime, but with a lower dose of chemicals (1:1.3). As the handling of quicklime is more complicated, hydrated lime is normally used in small treatment plants and quicklime in large plants. When quicklime is slaked to the hydrated form, energy is emitted - 1160 kJ/kg CaO. Theoretically, 350-400 kg CaO/m^3 water can bring the temperature near boiling point. However, as the quantities of lime used in practise are normally related to the dry solids content of the sludge, the treatment of non-dewatered sludge will not result in any significant rise in temperature.

The effects of liming non-dewatered sludge, usually at the dose of 10-20 kg/m^3, can be summarized as follows: i) improvement of sludge dewatering properties (in some cases in combination with addition of ferrous sulphate or ferric chloride); ii) pH increase to about 11.5-12, which last for about two weeks; iii) no increase of temperature; iv) inactivation of bacterial and viral pathogens, but limited effect on parasites; and v) re-growth of bacterial pathogens. In the case of dewatered sludge, addition of quicklime results in a significant temperature rise and high dry solids content due to evaporation. This in turn leads to improved sludge handling characteristics and long lasting disinfection (EU, 1999).

With addition of the appropriate quantity of CaO, a pH increase to approximate 12.5 can be obtained followed by a significant temperature increase. For dewatered sludge with TS content in the range 20-30%, respective increases of 3.4 to 3.9°C for each percent CaO dose (1% CaO = 10 kg CaO per tonne of sludge) can be theoretically expected. In practise, due to heat losses and quicklime quality (less than 100% active CaO), the observed temperature increases are limited to 60-82% of the theoretical values. The time needed to effect the temperature increase is approximately one to two hours after mixing, depending on lime quality. However, it should be noted that unless excess CaO is used, a consequent reduction of the pH can be observed due to the reaction of the produced $Ca(OH)_2$ with the CO_2 of the atmosphere or that produced due to biological activity. Another factor that has to be taken into consideration is the reaction with external CO_2 of the atmosphere and/or CO_2 produced by fungi activity at the surface of the sludge. However, with proper storage (low surface area / volume ratios, low temperatures) this factor can be minimized.

The effect of CaO on selected bacteriological parameters after mixing is shown in Table 6.7, which indicates that substantial reduction of pathogens can be achieved even for a dose of 2% CaO.

Table 6.7

Influence of CaO on bacteriological parameters four hours after mixing, 20°C (Carl Bro S/A, 1997)

CaO Dose	**Coliforms**	**Temperature resistant Coliforms**	***Streptococcus***	***Clostridium perfrigens***	***Salmonella***
%	number/g	number/g	number/g	number/g	number/g
0	$23x10^4$	$49x10^3$	$11x10^4$	$30x10^3$	Traced in 10 g
2	$33x10^1$	-	<100	$18x10^2$	-
4	$13x10^1$	-	<100	$14x10^2$	-
6	$33x10^1$	-	<100	$13x10^2$	-
8	$13x10^1$	-	<100	$9x10^2$	-
10	$13x10^1$	-	<100	$2x10^2$	-

The effects of the duration under high pH and of the temperature seem to vary with the pathogen type. Prolonged exposure, over several days, of coliforms to a high pH environment enhances their removal, while increased temperatures appear to be effective only in the case vegetative *Clostridium perfrigens*. Quicklime treatment of sludge is also effective with respect to removal of parasites. Comparative studies of *Ascaris* ova growth with (10% CaO) and without CaO addition, reveal the presence of multi-cell ova and development of fully grown larva up to 71% in the reference sample, while practically only unicellular ova are observed with quicklime treatment.

The long-term effect of CaO treatment on sludge characteristics is shown in Table 6.8. Upon quicklime addition, an increase in total solids content and pH is observed, while volatile solids content, phosphorus and nitrogen and ammonia contents are reduced. With storage over seven months, total solids content tends to gradually increase while pH is slightly reduced, but remains, however, above 12. It is only after a storage period of two years that a drastic reduction of pH and alkalinity are observed. The total solids content increases significantly, while the ratio of total solids volatile to total solids is reduced due to degradation of the organic material. This degradation also explains the higher phosphorus content observed. Sludge obtains a wood-like texture with slight but not unpleasant odour.

Table 6.8

Long-term effects of the CaO treatment (10%) on sludge characteristics
(Carl Bro S/A, 1997)

Parameter	Units	Days						
		0*	**0**	**14**	**45**	**120**	**210**	**720**
pH		7.1	12.5	12.5	12.5	12.3	12.1	8.4
TS	g/kg	168	283	292	291	332	303	531
VS	g/kg	101	87	86	90	93	104	103
VS/TS	%	60	31	29	31	28	34	19
Tot-P	g/kg TS	17	9.8	9.6	-	9.3	-	17
Tot-N	g/kg TS	32	19	17	17	17	17	13
NH_3-N	g/kg	1.5	0.27	0.21	0.27	0.31	0.18	-
Alkalinity	mmol/kg	-	-	2390	2700	3080	2700	1340
Odours			-	-	-	-	-	-

Before CaO addition.

Addition of quicklime to dewatered sludge and subsequent storage under a pH of over 12 for at least three months ensures a high degree of sludge sanitation. This sludge can be used as a soil conditioner and fertilizer without any restrictions as far as pathogens are concerned. Even after prolonged storage, there is a very limited reduction of nitrogen, while the availability of phosphorous for plant growth is high (over 90%) under conditions of neutralised pH, which is very quickly established upon mixing of the sludge with the soil (Akrivos *et al*., 1999). Finally, quicklime treatment improves the handling characteristics of the sludge and allows for long-term storage without development of odour. Quicklime addition is considered as an acceptable sanitation method to produce sludge for reuse in agriculture.

6.4.5 Thermal drying

Thermal drying of wastewater sludge involves the application of heat to evaporate water from sludge. When thermal drying is used as the final sludge treatment process it can achieve a sludge moisture content usually below 10%, thereby significantly reducing the volume and mass of sludge that has to be handled and disposed of. The advantages of the process include reduced transportation costs, improved storage capability and marketability, as well as pathogen destruction. Thermally dried sludge can be easily marketed as a fertilizer or soil conditioner. The product produced by the process is pasteurized and can be handled safely and, therefore, offers a much more attractive alternative than handling liquid sludge or sludge cake. In order to increase the bulk density of the dried sludge and to improve the marketability of the product dried sludge is usually converted to pellets.

Thermal drying is a high-energy demanding process. The required thermal energy for drying 1 ton of sludge from 20% to 90% dry solids is approximately 2.5 - 3×10^6 KJ. The efficiency of dewatering facilities can significantly influence the energy consumption because energy requirements decrease as the dry solids content of the feed sludge increase.

An important aspect that should not be overlooked in the design of drying processes is associated with air pollution and odour control. The two most critical issues are particulates (fly ash) and volatile organics removal. Cyclone separators or wet scrubbers are usually employed for the removal of particulates. Odours due to volatile organics are best removed by complete oxidation in a boiler or a furnace at a minimum temperature of 730°C.

Thermal dryers can be grouped into two categories based on the method of transferring heat to the wet solids: i) direct; and ii) indirect. In direct drying the energy to heat the sludge is supplied directly by means of heated oil or steam. In indirect drying the energy is supplied by means of a heat exchanger. A brief description of each process is presented in the following paragraphs.

Direct Dryers: in direct drying systems the wet sludge directly contacts the heat transfer medium, usually hot gases. In general, direct drying requires much greater quantities of hot gases and, therefore, a greater volume of air polluting gases are generated by direct dryers. Direct drying can be divided into the following groups:

flash drying involves drying of sludge in a cage mill in the presence of hot gases. The flash dryers consist of a furnace, mixer, cage mill, cyclone separator, vapour fan and an air pollution control system; and

the rotary dryer consists of a rotating cylindrical steel shell usually mounted with its axis on a slight slope from the horizontal to facilitate solids flow, a mixing tank where dried sludge mixes with wet sludge, furnace where gases are heated up to temperatures of 260-480°C, cyclone and air pollution control system.

Indirect Dryers: the paddle or disk dryer system consists of a horizontal vessel with a rotor through which a heat transfer medium (usually steam or oil) circulates. The vessel contains a group of large hollow disks mounted on a central hollow shaft. The external surfaces of the paddle or disks provide 88 to 100% of the total heat transfer. Often, scraper bars are mounted on the vessel extending vertically between each disk to achieve better mixing and prevent sludge build up on the surface of the disks.

The multiple effect evaporation (Carver - Greenfield) process extracts water from wet sludge by evaporation. The advantage of this process compared to single stage evaporation systems such as flash dryers is that heat is reused to improve efficiency and decrease energy consumption. Steam is used only to evaporate water from sludge at the last stage, whereas vapour water produced at the last stage is used instead of steam to heat solids in the preceding evaporator stages.

A disadvantage of the process is its great complexity that results in a high construction and maintenance cost and necessitates the employment of highly trained personnel. The process resembles a petrochemical plant more than a wastewater treatment plant and, therefore, a municipality employing this process would have to make a considerable investment for the training of skilled operators.

Steam Fluidized Bed drying is a relatively new drying process that has been used for drying of municipal sludge since the beginning of the 90s. The process consists of a sludge grinder, fluidized bed vessel, closed drying loop containing steam, steam boiler, dry sludge conveyor, dry sludge cooling system and a water vapour condenser. The drying process is designed as a completely closed system in order to eliminate odour emissions.

Environmental Concerns: the impact on air quality is a critical issue in the selection of the most appropriate technology especially in highly populated areas. During thermal drying of sludge steam that comes into contact with sludge can strip the sludge of certain volatile organic chemicals that are odorous and sweep away, depending on gas velocity, sludge particles. Because indirect dryers utilize small quantities of gases as carriers or sweep gases that come in direct contact with dry sludge, they generate much smaller quantities of non-condensable air polluting gases. Therefore, it appears that in the case of direct drying it will be more difficult to achieve odour and particulate-free air emissions and it will be necessary to install and maintain an efficient air pollution control system that will include both a scrubber for particulate removal and an afterburner for odour control.

An important issue in the design of all dryer types is the provision of sufficient storage area for wet and dry sludge, to allow for mechanical shutdowns and fluctuation in product demand. Dry sludge is usually stored in silos sealed from the outside environment.

Based on operating experience it appears that some drying processes more often suffer from operational problems. For example, flash dryers appear to be vulnerable to severe abrasion by dried sludge. From the 50 municipal flash dryer facilities installed in the US only five or six are still operating. Another important issue in selecting the most appropriate technology is the flexibility and complexity of operation; it appears that the complexity of some drying systems, especially the Carver-Greenfield drying system, requires experienced and highly trained staff.

Thermally dried sludge with a dry solid content greater than 90% is considered pasteurized and in most cases can be handled with safety in terms of pathogens transmission. For a successful sludge-marketing programme, dried sludge should be as dust free as possible with the sludge particles durable enough to withstand handling and transportation. Direct dryers tend to produce a less dusty product. However, in view of the need to convert sludge into pellets and compact sludge particles in most drying methods in order to improve marketability of the final product, ease of sludge handling and also to reduce potential fire hazards, this does not appear to be a critical issue in the selection of the appropriate technology.

6.4.6 Pasteurization

The pasteurization method used in Europe and mostly in German and Switzerland, involves heating the sludge at 65°C for 30 min, followed by cooling and anaerobic digestion. Based on European experience, heat pasteurization is a highly effective process in reducing enteric bacteria and inactivating parasitic ova and cysts (Havelaar, 1983). Heat pasteurization may not be cost-effective for small plants with capacities of less than 17000 m^3/d, because of high capital costs. Other temperature and heat combinations are: 1) 70°C for 25 min; 2) 75°C for 20 min; 3) 80°C for 10 min;and 4) 55°C for 3 hours.

6.4.7 High energy irradiation

There are very few plants using irradiation to disinfect sludge. Accordin g to the German Sludge Ordinance (Strauch, 1987) the required radiation doses are:

- 500 krad for liquid sludge; and
- 1000 krad for dewatered sludge.

At these doses, complete/kill of gram negative bacteria and *Ascaris ova* is achieved (Havelaar, 1983). However, little or no removal is expected for gram positive bacteria. Viruses are removed by one to two logs.

Table 6.12

Summary of sludge disinfection processes (Havelaar, 1983)

Type of process	Lethal factors[1)]	Effect against micro-organisms[2)]				Product stability[3)]	Remarks
		Bacteria	Viruses	Parasite eggs	Spores		
Pasteurization	Heat 30 min 70°C	Good	Variable	Good	Poor	Post-past: poor Pre-past.: good	Must be combined with stabilization.
Irradiation	Ionizing radiation 300 krad	Good	Poor	Good / moderate	Poor	Variable	Must be combined with stabilization.
Aerobic thermophilic stabilization - oxygen - air	Heat 60-80°C 40-60°C	Good Poor/good	Good /var. Poor /var.	Good Poor/good	Poor Poor	Variable	Effect depends on mixing characteristics and regime of filling drawing.
Composting - windrows - bioreactors	Heat 40-60°C 50-80°C	Poor /good Good	Poor /var. Poor /good	Poor/good good	Poor Moderate	Good	Windrows: effect depends on turning +climate.
Lime treatment - slaked lime - quick lime	High pH and/or free ammonia pH up to 12 as above, heat up to 80°C	Good Good	Moderate/ good Good	Variable Good	No data	Good if pH remains > 10	Infectivity of some parasitic eggs (e.g., Taenia) may be destroyed by high pH.

[1)]Process conditions given as a typical example of operating range; [2)]good: no survival; moderate: significant reduction; poor: no significant reduction; variable: dependent on species; and [3)]poor: re-growth of enterobacteria occurs; good: re-growth does not occur; variable; re-growth of some species but not of others occurs.

Table 6.13

Overview of sludge treatment processes

Processes	Water removal / Volume reduction	Organic matter reduction (stabilization)	Pathogen reduction	Remarks
Gravity thickening	+			Low initial and operating costs. Not very efficient with activated sludge.
Mechanical thickening	++			Moderate initial capital. Need for personnel. Polymers required (4-5 kg/tn dry solids). Efficient thickening (4-5% solids)
Drying beds - lagoons	+++			Inexpensive. Area demanding. Emission of odours. Sludge removal involves unpleasant work.
Mechanical dewatering	+++			Moderate to high initial capital. Employed in combination with mechanical thickening.
Anaerobic mesophilic digestion		+	PD	Typical process especially when primary sludge is produced. Biogas production in excess of thermal requirements.
Anaerobic thermophilic digestion		+	D	Used for sludge disinfection. Thermal requirements in excess of by biogas production. Energy input required.
Aerobic psychrophilic digestion		+	PD	Typical process for extended aeration systems (without primary treatment). Energy demanding.
Aerobic thermophilic digestion		+	D	Used for sludge disinfection. Energy demanding.
Composting		+	D	Usually in combination with other coarser organic material. Otherwise additives are needed. Used for sludge disinfection.
Slaked lime treatment			PD	Used for sludge disinfection, although at least three-month storage must follow. Low initial cost. High operational cost due to chemical additive.
Quick lime treatment			D	Effective for sludge disinfection. Low initial cost. High operational cost due to chemical additive.
Pasteurization			D	Energy demanding. Must be followed by anaerobic digestion.
Sludge irradiation			D	Not well established method.
Drying	++++		D	Very effective in drastic volume reduction (by about 70%) and sludge disinfection. Fairly high initial cost and need for energy input.
Incineration	+++++	+++++	D	Expensive. Potential air pollution. Limited if any energy recovery.

PD: Partial destruction   D: Practical elimination (disinfection).

7. REFERENCES

A.D. Andreadakis, "Treatment and disinfection of sludge using quicklime", (2000), Proceedings of the European Commission Workshop on Problems around Sludge, p. 31-37

ADEME 2001, Hygienisation des bio-dechets: Validation du caractere hygienisant de procedes de traitment. Deportes I et Gay J. Rapport ADEME/OSER, 86p.

ADEME, 1994, Les germes pathogenes dans les boues residuaires de la station d'epuration urbaines. Elissade N, Ganiere JP , L'hostis M, Legeas M, Demillac R, Carre J, Wiart J, Feix I, 72 p.

ADEME, Agences de l'eau; 1995; Les différents procédés de stockage des boues d'épuration avantvalorisation en agriculture.

ADEME, ENSAIA, INRA Nancy; 1996; La valeur phosphatée des bouesrésiduaires urbaines

ADEME, ENVN, FNDAE et ENSP; 1994; Les germes pathogènes dans les boues d'épuration urbaines. Collection "valorisation agricole des boues de station d'épuration".

ADEME, INAPG, ADEPRINA; 1995; La valeur azotée des boues résiduaires u rbaines.

Adriano DC, 1986. Trace Elements in the Terrestrial Environment. Springer-Verlag, New York, NY.

Akrivos J, Mamais D, Katsara K and Andreadakis A. (1999). Agricultural utilization of lime treated sewage sludge. Proceedings of the Specialized Conference on Disposal and Utilization of Sewage Sludge: Treatment Methods and Application Modalities, pp. 393-400.

Carl Bro A/S, (1997), "Treatment of Sludge with Lime", Research Report in the framework of the research project Hygienic Management of Sludge for Agricultural Utilization (Sponsored by EU-DGXVI, 1997-2000).

Christoulas D, et. al., (1994) "Leahate production for sanitary landfills and the effect of leachate recirculation", Final report of an NTUA research project sponsored by General Secretariat of Research and Technology.

EPA, "Process Design Manual, Surface Disposal of Sewage Sludge and Domestic Sewage" /625/R-95/002, September 1995..

Epstein, E. (2002) Human pathogens: hazards, controls and precautions in compost. In: "Compost utilization in horticultural cropping systems". Lewis Publishers, Boca Raton, USA, 361-380.

European Commission JRC (1999), Proceedings of the workshop "Problems around sludge", Stresa, Italy.

European Commission DG Environment–B/2. 'Disposal and Recycling of Sewage Sludge', Part 1 - Sludge use acceptance, October 2001, Part 2 - Regulatory report, October 2001, Part 3- Scientific and technical subcomponent report, October 2001, Part 4-report, January 2002.

EUROPEAN ENVIRONMENT AGENCY; 2000; Down to earth: Soil degradation and sustainable development in Europe, Environmental Issues Series n° 16, http://themes.eea.eu.int/toc.php/state/soil?doc=39303&l=en

European Union (1986) Council Directive 126/1986 on the protection of the environment and in particular of the soil when sewage sludge is used in agriculture, 86/278/EEC, Off. J. of the Eur. Comm. No L186/6-12, 4/7/1986.

European Union, (1999), Draft Proposal for Sewage Sludge Management, XI.E.3/LM

FAO/OMS; Codex Alimentarius; 1984.

Havelaar A.H. (1983). “Disinfection of Sewage Sludge: A review of methods applied in the European Communities”, Processing and Use of Sewage Sludge, Commission of the European Communities, L’ Hermite (eds), D. Reidell Publishing Company.

Implementation of Council Directive 91/271/EEC of May 21 1991 concerning urban wastewater treatment, as amended by Commission Directive 98/15/EEC of 27 February 1998 (in the process of adoption by the Commission)’.

Jones, K. C. & G. L. Northcott (2000): Organic Contaminants in Sewage Sludges: A Survey of UK Samples and a Consideration of their Significance - Final Report to the Department of the Environment, Transport and the Regions. Water Quality Division.

Metcalf & Eddy. (2002). Wastewater Engineering, Treatment and Reuse, Fourth Edition.

Muse JK, Mitchell CC and Mullins GL. Land Application of Sludge, Environmental Education Series, ANR-607, Auburn University, February 1991.

NTUA Research Project (2000) Treatment and reuse of sewage and sludge in Middle East and South Mediterranean Countries (sponsored by the Ministry of Environment and Public Works, DAPE/DAC)

NTUA Research Project (1999) Hygienic Management of Sludge for Agricultural Utilization (Sponsored by EU-DGXVI, 1997-2000)

NTUA Research Project (1996) Technico-economical and Environmental study for the Management of the Sludges Produced in Psytallia WWTP (Sponsored by the Ministry of Environment and Public Works)

O’ Connor et. al., (2001), “Minergy glass aggregate technology”, Proceedings of Specialized conference on “Sludge Management: regulation, treatment, utilization and disposal”, Acapulco, Mexico 2001

Pike E.B. (1983). “Long-term Storage of Sewage Sludge”. Processing and Use of Sewage Sludge, Commission of the European Communities, L’ Hermite (eds), D. Reidell Publishing Company.

Pohland F., “Landfill bioreactors: fundamentals and practise”, Water Quality International Sept/Octob. 1996, 1996, pp.18-22.

Smith S. R.; 1996; Agricultural recycling of sewage sludge and the environment 367 pp.

Stadterman, K.L., A.M. Sminsky, Sykora, J.L. and Jakubowski, W. (1995). "Removal and Inactivation of *Cryptosporidium* Oocysts by Activated Sludge Treatment and Anaerobic Digestion", Wat. Sci. Tech., Vol 31, (5/6), 97-104.

Strauch, D. (1987). "Federal Republic of Germany", Hygienic Aspects of the Treatment and Use of Organic Sludge and liquid Agricultural Wastes, Hannan J.H. and P.L'Hermite, Commission of the European Communities.

Tullander V. (1983). "Quicklime Treatment of Sludge", Environmental Effects of Organic and Inorganic Contaminants in Sewage Sludge, Davis, Hucker and L' Hermite (eds), D. Reidell Publishing Company.

U.S. Environmental Protection Agency. (1993) 40CFR503. Standards for the use or disposal of sewage sludge. *Federal Register* 58(32), 9248–9415.

USEPA (1979). Process Design Manual: Sludge Treatment and Disposal. EPA 625/1-79-011.

Vesilind P.A., "Treatment and Disposal of Wastewater Sludges", Ann Arbur Science, 1979.

Watanabe H., Kitamura T., Ochi, S. and Ozaki, M. (1997). "Inactivation of Pathogenic Bacteria under Mesophilic and Thermophilic Conditions", Wat. Sci. Tech., Vol. 36 (6/7), 25-32.

WEF - ASCE (1991). Design of Municipal Wastewater Treatment Plants. WEF Manual of Practise No 8.

Wiebusch B, et. al., (1999), Proceedings of Specialized conference on "Disposal and utilization of sewage sludge: treatment methods and application modalities", Athens, Greece, 1999

WHO (2002); Developing human health-related chemical guidelines for reclaimed wastewater and sewage sludge applications in agriculture.

WPCF (1987). Operation and Maintenance of Sludge Dewatering Systems. Manual of Practise No. OM-8.

WRc (1999), Manual of good practise for the use of sewage sludge in land reclamation

Web sites (Glossary):
http://glossary.eea.eu.int/EEAGlossary
http://www.wef.org/publicinfo/newsroom/wastewater_glossary.jhtml