Chapter 16

Sludge Processing and Disposal

16.1 INTRODUCTION

Sludge is the relatively concentrated suspension into which the residual solids fraction arising from water or wastewater treatment is concentrated in the course of purification. Sludges are derived from the processes of chemical coagulation and softening at drinking water treatment plants and from the preliminary, primary and secondary stages of wastewater treatment. Most of these sludges are of an unstable organic nature and readily undergo active microbial decomposition with consequent generation of nuisance odours. They all have the common characteristic of a high water content, usually greater than 95% by weight. Sludges derived from wastewaters containing domestic sewage or animal excreta may contain significant concentrations of pathogenic organisms. In this chapter, sludge composition and quantities generated are considered, as well as the available options for processing and disposal.

16.2 WATERWORKS SLUDGE

Waterworks sludge is derived almost exclusively from the chemical coagulation of surface waters to effect the removal of colour and turbidity, using either aluminium or iron salts as chemical coagulants (refer Chapter 3). The yield of sludge solids from chemical coagulation processes can be approximately estimated from the following empirical correlation (WRc, 1992):

$$
\begin{aligned}
\text{Sludge solids (mg l-1 treated water)} \quad = \quad & 2 \text{ x turbidity removed (NTU)} \\
& + 0.2 \text{ x colour removed } (^{o}\text{H}) \\
& + 2.9 \text{ x aluminium precipitated (mg Al } \text{l}^{-1}) \\
& + 1.9 \text{ x iron precipitated (mg Fe } \text{l}^{-1})
\end{aligned}
$$

Significant quantities of calcium carbonate sludge are produced in water softening by the lime/soda precipitation process (refer Chapter 9).

16.2.1 Characteristics of alum sludge

Alum sludge is a bulky gelatinous suspension whose solids fraction consists of aluminium hydroxide floc, fine inorganic particles, colour colloids and other organic debris. The sludge is usually generated in two streams from the sequential processes of clarification and rapid gravity filtration with over 90% of the solids coming from the clarification process.

The sludge residue discharged from clarifiers typically constitutes between 1% and 3% of the plant throughput with an average solids content in the range 0.3-1.0% dry matter by weight. It normally has a relatively low biodegradable organic fraction, typically having a 5-day biochemical oxygen demand (BOD_5) in the range of 30-100 mg l^{-1} (Albrecht, 1972).

The sludge stream arising from the back-washing of rapid gravity filters typically constitutes between 3% and 6% of plant throughput with an average suspended solids concentration in the range 0.004-0.05% by weight, while its BOD_5 is usually less than 5 mg l^{-1}.

16.3 SEWAGE SLUDGE

Sewage sludge is the relatively concentrated suspended solids residue resulting from the purification of municipal and industrial wastewaters. It includes primary sludge, which is composed of the settleable solids fraction of raw wastewater, as separated by sedimentation or flotation processes, and secondary sludges, which are derived from biological or physico-chemical treatment processes.

The solids composition of raw municipal wastewater is quite variable, being influenced by the type of sewer collection system (whether combined or separate) and the relative contributions from domestic and industrial sources. The approximate division of municipal wastewater solids into organic and inorganic fractions, and their further division into settleable, non-settleable and dissolved components, is shown in Table 16.1.

Table 16.1 Average composition of sewage solids expressed as percent by weight

Solids category	Suspended settleable	Suspended non-settleable	Dissolved	Total
Inorganic	10.6	5.3	21.1	37.0
Organic				
Carbohydrates	6.8	3.3	10.5	20.6
Fats	7.4	3.6	8.0	19.0
Proteins	6.8	3.6	13.0	23.4
Organic total	21.0	10.5	31.5	63.0
Total	31.6	15.8	52.6	100.0

Source: Popel (1963)

The preliminary phase of sewage treatment normally consists of screening and grit separation. Screening removes so-called ‘gross’ solids and fibrous materials from the flow. Sometimes screenings are macerated and returned to the flow. Alternatively, they may be collected in skips or sacks and disposed to landfill. Grit is the heavy inorganic particulate fraction of sewage, which separates readily from the lighter organic settleable solids. Removal at a preliminary stage is essential to avoid its accumulation in downstream process units. The grit load may typically vary in the range 25-250 mg l^{-1} (White, 1970), being particularly high in combined sewage at times of heavy rainfall. When grit is washed free of organic matter, it becomes an inert inorganic residue that can be used as a construction material or disposed to landfill.

Primary treatment of sewage normally involves simple sedimentation to remove settleable solids. Secondary treatment consists of a biological process – biofiltration or activated sludge – followed by sedimentation. Tertiary treatment may consist of sand filtration to remove residual suspended solids and/or chemical precipitation of phosphorus. Typical sludge yields from sewage treatment processes are given in Table 2. These ranges should be taken as a rough guide only, as the production of sludge varies considerably from works to works. The lower limits of the ranges may be taken as typical for domestic sewage with the upper limits indicative of a significant industrial contribution.

Table 16.2 Sludge production in sewage treatment

Process	Quantity (g PE^{-1} d^{-1})
Primary sedimentation	45-55
Biofiltration of settled sewage	13-20
Activated sludge, SRT 5d, pre-settled sewage	25-40
Activated sludge, SRT 25d, raw sewage	30-60
Tertiary sand filtration	3-5
Phosphorus precipitation (Al or Fe)	8-12

16.3.1 Sewage sludge characteristics

Primary sludge is a grey slimey suspension with an offensive odour, typical of active putrefaction. Since it contains wastes of enteric origin it may contain significant numbers of pathogenic oreganisms. Activated sludge is a flocculent suspension with a characteristic inoffensive earthy odour, when fresh.

It is essentially a microbial biomass consisting mainly of bacteria but also including fungi and protozoa. The stability of activated sludge is a function of the AS process SRT. Extended aeration activated sludges, particularly those with an SRT > 20d (e.g. oxidation ditch sludge) are relatively stable while non-nitrifying activated sludges readily become septic in the absence of oxygen. Sewage-derived activated sludges may contain some surviving pathogenic organisms (van Gils, 1964). In general, biofiltration sludges are similar to activated sludges.

Sewage sludges exhibit non-Newtonian flow behaviour and have thixotropic properties. Their specific gravities reflect the specific gravities of their solid fractions and also depend on solids concentration.

16.4 INDUSTRIAL WASTEWATER SLUDGES

The principal industrial activities that give rise to sludge-producing wastewaters are:

- Food and related process industries, including milk, meat, fish, fruit and vegetables processing.
- Beverage industries including malting, brewing, distilling and soft drinks production.
- Pulp and paper, chemicals, biochemicals, textiles production
- Mining and quarrying.

With the exception of mining and quarrying, where the derived sludge residues are entirely inorganic, and certain relatively toxic residues from the chemical and biochemical sectors, industrial wastewaters are mainly of a biodegradable nature and are amenable to the same range of treatment processes as municipal sewage. The distribution of solids by size and composition varies considerably from industry to industry. The general physical characteristics of primary and secondary sludges derived from processing of organic industrial wastewaters are broadly the same as those described for sewage sludges.

16.5 PROCESSING OF SEWAGE SLUDGE

Water and wastewater sludges, as discharged from the process units in which they are produced, vary greatly in composition and water content. The degree to which they are subsequently processed depends largely on their disposal destination, which may be to agriculture, landfill or incineration. The range of process options is outlined in Fig 16.1. The optimal process route for a particular sludge is determined by technical, economic and environmental considerations.

16.6 SOLIDS/WATER SEPARATION PROCESSES

The water fraction of sludge can be divided into three categories: (1) free water, which is not intimately associated wit sludge solids; (2) capillary and boundary layer water retained by surface forces; and (3) intracellular and chemically bound water.

The ease with which the free water can be drained from a sludge depends on its hydraulic permeability. Sludges that have a flocculent solids fraction have relatively large drainage channels through which the water can escape while the opposite holds for sludges of a gelatinous nature. In general, organic sludges have a high affinity for water and require artificial flocculation ('conditioning') prior to thickening and dewatering. Flocculation causes an aggregation of solids particles, in consequence of which a considerable fraction of the water is transformed into the free drainage category.

Capillary and surface-held water can only be removed by the application of pressure gradients that exceed the counter-gradients generated by holding forces such as surface tension.

The following laboratory measurements are used: sludge volume index (SVI), specific resistance to filtration (SRF) and capillary suction time (CST).

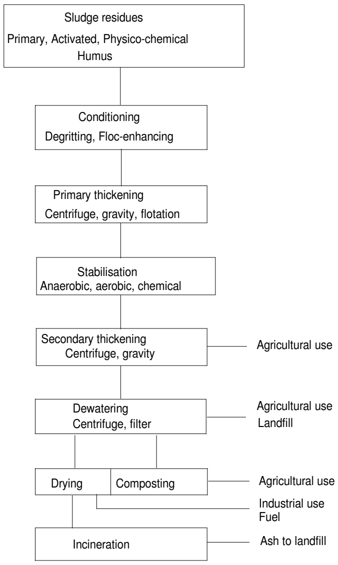

Fig 16.1 **Sludge processing flow sheet**

SVI is the sludge volume, as ml per g of sludge, after 1 litre of sludge is allowed to settle for 30 minutes in a standard graduated cylinder. It essentially measures sludge settleability and is particularly relevant to sedimentation and thickening processes. An activated sludge with good settling properties typically has an SVI less than 100 ml g^{-1}, while a 'bulking' sludge may have an SVI value in excess of 200 ml g^{-1}.

Sludge SRF is a measure of the impermeability of a sludge layer deposited on a filter medium to which a vacuum has been applied. The rate of filtration can be expressed as follows:

$$\frac{dV}{dt} = \frac{p}{\mu} \cdot \frac{A}{R} \tag{16.1}$$

where dV/dt is the filtration rate, p is the total pressure difference across the retained solids layer and the filter medium (driving force), μ is the viscosity of the filtrate, and A is the plan area of the filter.

Equation (16.1) defines R, the total resistance to flow through the deposited cake and filter medium. The total resistance to flow may be divided into the cake resistance R_c and the medium resistance R_m, and equation (16.1) may be written as follows:

$$\frac{dV}{dt} = \frac{p}{\mu}\left(\frac{A}{R_c + R_m}\right) \qquad (16.2$$

R_c is not a constant because, as filtration proceeds, the depth of cake increases as does its hydraulic resistance. It has been found convenient to directly correlate this increase in resistance to w, the mass of solids deposited per unit area of cake. Hence equation (16.2) becomes:

$$\frac{dV}{dt} = \frac{p}{\mu}\left(\frac{A}{SRF \cdot w + R_m}\right) \tag{16.3}$$

Equation (16.3) defines the SRF value as the hydraulic resistance of a cake having unit mass of dry solids per unit area of filtration surface, with units of m kg^{-1}.

To use equation (16.3) for practical purposes, w is expressed in terms of the volume of filtrate per unit area by writing $w = cV/A$, where c is the suspended solids concentration in the sludge sample being filtered. Substitution in equation (16.3) yields:

$$\frac{dV}{dt} = \frac{p}{\mu}\left(\frac{A^2}{SRF \cdot cV + R_m A}\right) \tag{16.4}$$

If the assumption is made that the terms in equation (16.4), other than V and t, are constants, the equation can be integrated to give

$$\frac{t}{V} = \left(\frac{\mu \cdot SRF \cdot C}{2A^2 p}\right)V + \left(\frac{\mu R_m}{Ap}\right) \tag{16.5}$$

The data required for computing SRF, using equation (16.5), are obtained from a sludge filtration test (Vesilind, 1975). The ratio t/V is plotted as a function of filtrate volume to give a straight line plot of slope ($\mu \cdot SRF \cdot C/2A^2p$), in accordance with equation (16.5). The value of SRF is computed from the measured slope of the plotted line.

Typical SRF values for a variety of sludges are given in Table 16.3.

Table 16.3 Specific resistance of sewage sludges and other materials

Suspension	SRF value (10^{13} m kg^{-1})	Filtration pressure (kN m^{-1})
Raw sewage sludges (mixed primary and secondary) from 8 works	10-20	49
The same sludges after anaerobic digestion	3-30	49
Activated sewage sludges	0.1-1000	49
Thixotropic mud	15	550
Gelatinous ferric hydroxide	1.5	173
Gelatinous aluminium hydroxide	2.2	173
Gelatinous magnesium hydroxide	0.3	173
Colloidal clay	0.5	173
Precipitated calcium carbonate	0.02	173
Sludges from:		
Stream peeling of carrots	4.6	49
Lime neutralisation of mine water	0.3	49
Vegetable tanning	0.15	49
Alum coagulation of water	5.3	49

Source: Gale (1971)

CST is the time (seconds) taken for the wetting front generated by a sludge sample in contact with thick blotting paper to travel 10 mm along the paper. It is measured in a standard apparatus (Baskerville and Gale, 1968), permitting rapid evaluation. The CST is a measure of sludge impermeability in a low hydraulic gradient environment. Because of its simplicity and rapidity, it is a very useful test for evaluating the relative merits of the conditioning chemicals used in sludge thickening and dewatering.

16.6.1 Sludge conditioning

The dewatering characteristics of sludges can be improved by chemical and thermal conditioning processes as well as by elutriation. Chemical conditioners include the trivalent salts of aluminium and iron as well as organic polyelectrolytes. They effect a breakdown of the natural barriers to particle aggregation or flocculation.

Cationic polymers have been found (Baskerville et al., 1978) to be particularly effective in this regard, using doses ranging from 2 to 5 kg per tonne of dry solids. The optimum chemical dose must be determined empirically for a particular sludge.

Thermal conditioning of sludge may be by heat treatment or freezing (Vesilind, 1975). In heat treatment, the sludge is brought to about 200 ^{o}C for 20-30 min at a pressure of about 18 atm. In addition to reducing SRF, this also effects sterilisation. One disadvantage is that some organic matter is resolubilized under the extreme conditions. A freeze/thaw cycle also improves sludge drainability. Thermal conditioning processes have a high energy demand, which in some cases can be satisfied by the use of biogas produced from digestion of the sludge.

Elutriation is a term applied to the washing of sludge to remove fines and lower its alkalinity, thereby improving its dewaterability. It is sometimes applied to anaerobically digested sludges prior to dewatering. The recycling of sludge fines in the elutriate has been found (Vesilind, 1975) to cause problems in upstream processes.

Sludges may be conditioned at the thickening and/or dewatering stages.

16.6.2 Sludge thickening

Thickening theory

Sludge thickening is a solids concentration process in which the readily separated water is removed either by sedimentation or flotation. Thickening produces a slurry or concentrated sludge and is conveniently distinguished from ‘dewatering’ which produces a sludge ‘cake’, i.e. a residue with dry solid handling characteristics. Thickening can effect a considerable volume reduction; for instance, an increase in solids concentration from 1% to 4% by a thickening process reduces the sludge volume to a quarter of its original volume.

Gravitational thickening is the separation of water from sludge by particle sedimentation. It is particularly applicable to dilute sludges, where very large volume reductions can be achieved, as illustrated in Fig 16.2. A schematic outline of a continuous thickener is shown in Fig 16.3. The design of such units is based on the concept of solids flux or the downward movement of solids within the thickening unit.

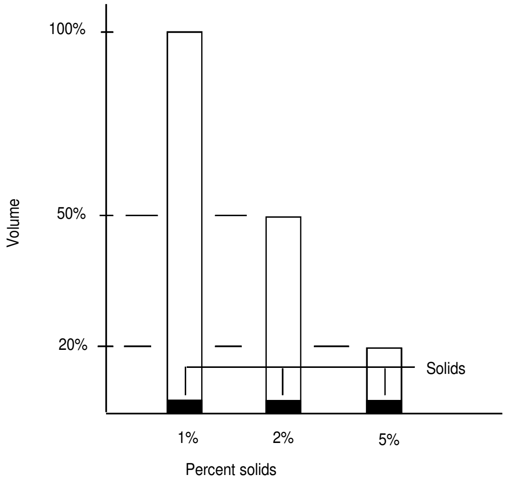

Fig 16.2 **Sludge volume reduction by thickening**

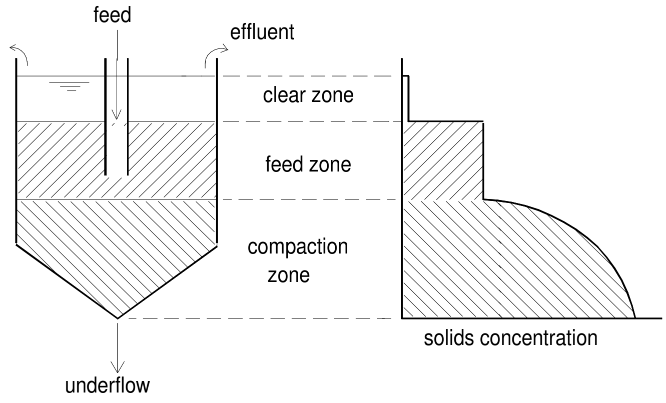

Fig 16.3 **Continuous gravitational thickening process**

According to the Kynch theory (Chapter 2), which is based on the assumption that the settling rate is a function of concentration only, the settling rate at any concentration can be obtained from a single test. In practice, however, it has been found that the Kynch theory does not hold for highly compressible materials such as wastewater sludges. The settling rate at a particular concentration must therefore be determined by carrying out a settling test at that concentration, yielding a settling curve similar to that illustrated in Fig 16.4(a). The slope of the linear section of the curve gives the settling rate for that concentration. A typical settling rate/concentration correlation is shown in Fig 16.4(b). The product of concentration and settling rate gives the solids flux, expressed as kg m^{-2} h^{-1}. A typical solids flux/concentration correlation is shown in Fig 16.4(b). The product of concentration and settling rate gives the solids flux, expressed as kg m^{-2} h^{-1}. A typical solids flux/concentration correlation is shown in Fig 16.4(c).

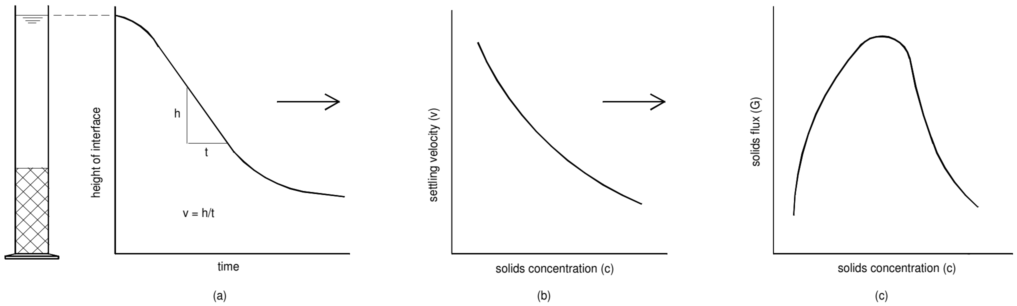

Fig 16.4 Development of the relation between solids concentration and solids flux.

In a continuous thickener the total solids flux is the sum of two components, one due to the settling rate of the sludge and the other due to its bulk downward movement. Thus, in a continuous thickener the total flux Gi at any level i is the sum of two components:

$$G_i = C_i(v_i + u) \qquad (16.6)$$

Where C_i is the solids concentration at level i, v_i is the settling velocity at concentration C_i and u is the bulk downward velocity. A typical total flux curve is shown in Fig 16.5.

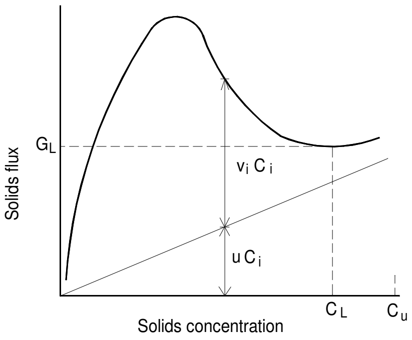

Fig 16.5 Typical total flux curve

The underflow rate Q_u in a continuous thickener of plan area A, operating at steady state, can be equated to the total flux G_i at any level i:

$$G_iA = Q_uC_u = AuC_u \tag{16.7}$$

Also

$$u = \frac{G_i}{C_u} \tag{16.8}$$

Combining equations (16.6) and (16.8) the following is obtained:

$$G_i = \frac{C_i v_i}{\left(1 - \frac{C_i}{C_u}\right)} \tag{16.9}$$

For a particular value of the underflow concentration Cu there is a limiting flux G_L at sludge concentration C_L, that determines the minimum thickener area required. This critical flux rate can be determined graphically (Vesilind, 1974) from the batch flux curve, as shown in Fig 16.6.

It follows from the geometry of Fig 16.6 that:

$$\frac{G_L}{v_iC_i} = \frac{C_u}{C_u - C_i} \tag{16.10}$$

from which it can be shown that:

$$G_L = \frac{C_i v_i}{\left(1 - \frac{C_i}{C_u}\right)}$$

which is identical to equation (16.9). Also

$$A = \frac{QC}{G_L} \tag{16.11}$$

where Q is the sludge output rate (m^3d^{-1}) and C is its solids concentration (kg m^{-3}).

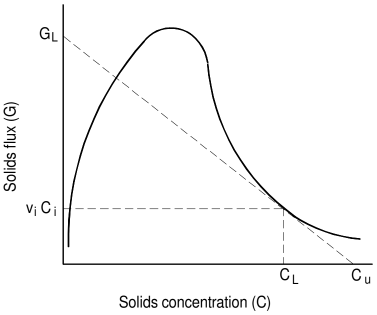

Fig 16.6 Graphical determination of critical flux G_L

Thickening Technology

Two methods of thickening are commonly employed: gravity thickening and flotation thickening. A schematic outline of a continuous gravity thickener is shown in Fig 16.3. Gravity thickener design is based (Dick, 1972; Fitch, 1975) on the concept of solids flux, expressed as kg h^{-1} m^{-2} of thickener plan area. Typical design values for various sludges are given in Table 16.4. For very dilute feed sludges, such as waterworks sludge, the overflow rate (≅G/C) may be critical. Ideally, the overflow rate in continuous gravity thickeners should not exceed 1.5 m h^{-1}.

Table 16.4 Typical design solids flux rates for gravity thickeners

Type of sludge	Solids flux range G (kg h^{-1} m^{-2})
Activated sludge	0.6-1.0
Trickling filter humus	1.4-1.8
Raw primary sludge	4.0-5.0
Mixture of primary and activated sludges	1.5-4.0
Pure oxygen activated sludge	1.0-2.0
Waterworks alum sludge	3.0-4.0

The depth of the unit must accommodate a clearwater zone, a feed zone and a compaction zone, as shown in Fig 16.3. Experimental evidence indicates that the effect on ultimate concentration of increasing the compaction zone depth beyond 0.9m is insignificant. An overall depth of 2.5-3.0m should therefore prove satisfactory.

Wherever possible, the thickening characteristics of a sludge should be assessed experimentally prior to design. Slowly rotating paddles or picket fences are sometimes used in gravity thickeners as an aid to water release from the compaction zone. In some instances (Lockyear, 1977) they have been found to effect a considerable increase in the solids concentration of the thickened sludge. Typical solids concentrations in gravity-thickened sludges are set out in Table 16.5.

Table 16.5 Solids concentration of gravity-thickened sludges

Type of sludge	Thickened sludge solids concentration (% DM)
Primary sludge	5.0-10.0
Mixed primary and activated sludges	4.0-6.0
Extended aeration activated sludge	3.0-5.0
Mixture of primary and activated sludges	1.5-4.0
Digested sludge (primary + activated)	6.0-10.0
Heat-treated activated sludge	10.0-15.00

Light flocculent sludges, such as activated sludge and waterworks alum sludge, can be thickened by dissolved air flotation processes (Sarfert, 1976; Bratby and Marais, 1977). A schematic outline of the flotation process is shown in Fig 5.4, Chapter 5. Thickening is accomplished in the lighter-than-water sludge float owing to the downward movement of water in accordance with the pressure gradient in the air/particle/water mixture. Since the concentration of solids varies within the depth of the float – being highest at the upper surface – it is normal to operate the scraping mechanism on an intermittent basis, removing the top layer with each pass. The basic design parameters for flotation thickeners are:

(1) solids flux rate (kg m^{-2} h^{-1}), on the basis of which the tank surface area is determined, and

(2) the air/solids ratio, which determines the required air release rate.

The tank depth is usually in the range 2.0-2.5m. Air/solids ratios are typically in the range 0.02-0.05 w/w, while the solids flux rate may be of the order of 10 kg m^{-2} h^{-1}. The design values for a particular application are best determined experimentally at laboratory or pilot plant level.

16.6.3 Sludge dewatering

Dewatering in this context means the removal of water to the degree that the remaining sludge residue effectively behaves as a solid for handling purposes. The minimum solids content at which this is achieved can vary between 16% and 30%. The wide spread of this value range is due to the fact that sludges that have a low SRF loose their fluidity at lower solids levels than those with high SRF values.

Dewatering can be accomplished by spreading on open air drying beds, vacuum filtration, pressure filtration and centrifugation.

The practice of using open air sand beds for the dewatering of stabilised sludges at small wastewater treatment works, which was once widely applied, has to a large extent been replaced by mechanical dewatering devices. Sludge drying beds typically consist of a 150-200 layer of fine sand, supported on a graded san/gravel underdrainage system, contained within a shallow concrete tank structure. Sludge is discharged onto the sand bed to a depth of 150-300mm and is left until it can be lifted as a semi-dry cake. In temperate climates typical design solids loading rates in the range 30-50 kg m^{-2} y^{-1} have been applied (Swanwick, 1972; WPCF, 1959). Dewatering is effected by passive drainage and evaporation, the relative amounts removed by each process depending on the sludge permeability, as shown in Fig 16.7.

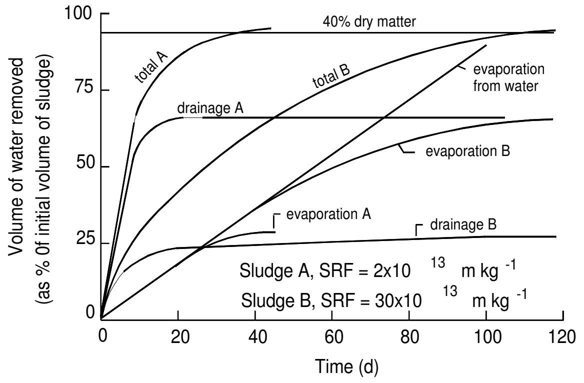

Fig 16.7 Effect of sludge SRF on dewatering on drying beds (Swanwick and Baskerville, 1966)

A schematic outline of a vacuum filter of the type used for sludge dewatering is shown in Fig 16.8. The main feature is a slowly rotating drum covered by a filter cloth. As the drum rotates, partly immersed in the sludge, an applied suction draws water inwards leaving a sludge layer attached to the drum surface. This layer continues to be dewatered until it reaches the discharge zone of the cycle where the vacuum is released and the cake removed by scraper. Typical vacuum filter performance data are presented in Table 16.6

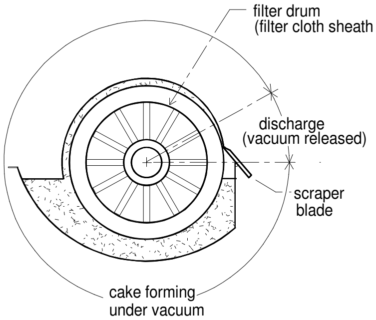

Fig 16.8 Schematic arrangement of a vacuum filter

Table 16.6 Typical vacuum filter performance data

Sludge type	Chemical conditioning	Filter yield ($kg\ m^{-2}\ h^{-1}$)	Cake solids (% DM)
Digested primary	Ferric chloride and lime	35	27
	Polymer	20-75	34-26
Mixed digested	Ferric chloride and lime	20-40	21.5
	Polymer	20	32-24

Two types of pressure filter are employed in sludge dewatering: the plate press and the belt press.

An outline of the plate press is shown in Fig 16.9. The parallel plates are covered by a filter cloth and are surface profiled to permit a flow of filtrate. Dewatering by the plate press process involves filling, pressing and cake removal. Performance depends on sludge SRF, pressing time and final cake thickness. The pressing time required to achieve a given solids concentration is very much influenced by the initial solids concentration (Gale, 1971); since all the expelled water has to be forced through the forming cake, the higher the initial solids concentration, the lower will be the amount of water to be discharged in this manner as filtrate.

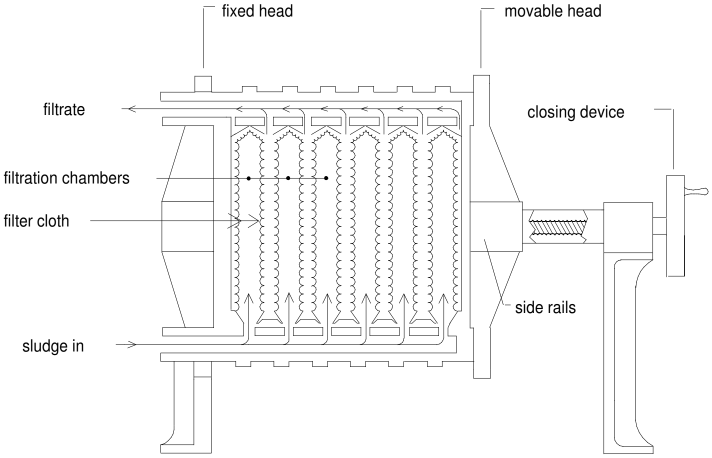

Fig 16.9 Schematic outline of a plate press

A schematic outline of the belt press is shown in Fig 16.10. Preconditioned sludge is fed on to the horizontal drainage section of the unit, which removes up to 50% of the drainage water. This sludge layer is then sandwiched between the carrier belt and a cover belt and is subjected to a varying compression by the roller system. Only readily drainable sludges can be effectively dewatered by the belt press, which means that conditioning is almost invariably required. The dewatered cake solids concentration may vary in the range 12-25% by weight.

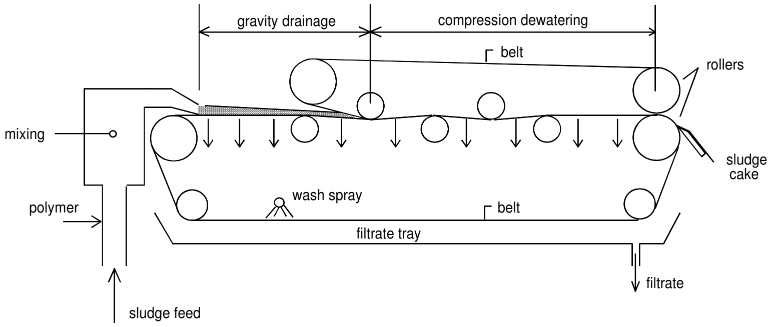

Fig 16.10 Schematic outline of a belt press

Centrifugation is a widely used industrial solids/water separation process. It is also extensively used for sludge thickening and dewatering where solid bowl centrifuges are employed. Performance is influenced by sludge particle size and specific gravity. The sludge is fed into the rotating bowl where it is separated into a dewatered solids stream and a dilute centrate stream. Since the latter may contain a relatively high concentration of light solids, it must normally be returned for reprocessing. Sludge conditioning effects a reduction in centrate solids concentration and an increase in the dewatered sludge solids concentration. The latter may lie in the range 10-25% by weight.

16.7 SLUDGE STABILIZATION

Many wastewater sludges possess high concentrations of biodegradable material, whose decomposition, in the absence of oxygen, is accompanied by the production of nuisance odour levels. Sludge can be stabilized by the biological conversion of its organic fraction to stable end products using aerobic or anaerobic digestion processes or by chemically inhibiting microbial activity in the sludge mass. The latter is usually effected by raising the sludge pH by the addition of lime.

16.7.1 Anaerobic digestion

The reader is referred to Chapter 15 for a detailed discussion of the theory and technology of anaerobic digestion processes. Anaerobic digestion is widely used for sludge stabilization and the treatment of high-strength organic wastes. It can effect a 40-60% reduction in organic solids, improve dewaterability, reduce odour potential and produce methane as a useful byproduct. As shown in Fig 15.2, the rate of anaerobic decomposition is strongly influenced by temperature. Digestion proceeds rather slowly between 7 ^{o}C and 16 ^{o}C. Mesophilic (16-38 ^{o}C) digestion is a more rapid process with optimal performance in the temperature range 30-35 ^{o}C. Thermophilic digestion (45-65 ^{o}C) is a still faster process and has an optimum around 55 ^{o}C. The influence of temperature on digestion time for primary sludge is shown in Fig 16.11.

In addition to being slow-growing, methane bacteria are sensitive to inhibition. Organic over-loading, leading to the accumulation of volatile acids and therefore a reduction in pH, may inhibit methane production. It is found that digestion proceeds satisfactorily in the pH range 6.6-7.6. Certain toxic components of wastewater sludges, including anionic detergents, chlorinated hydrocarbons, heavy metals as well as light metal cations at high concentrations (Swanwick, 1971), also inhibit methane bacteria. Mesophilic digestion is commonly adopted as the primary mode of sludge stabilization at medium- and large-sized works. Tanks are designed with a detention time in the range 15-25d with a mean suspended solids concentration in the range 3-6% dry matter. The process may be carried out in a single unit or in two stages, the second stage being used for settling and gas collection. Tanks are usually cylindrical with conical hopper bases and fixed or floating roofs, and are thermally insulated.

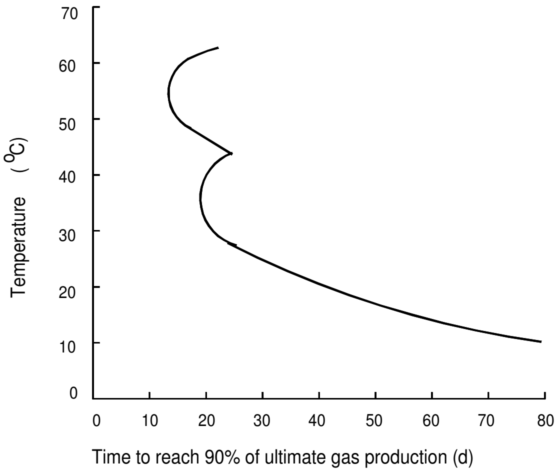

Fig 16.11 Influence of temperature on digestion of primary sewage sludge (Fair et al., 1968)

Adequate mixing is essential to avoid short-circuiting and ensure effective solids degasification. Methods of mixing include mechanical stirring, sludge recirculation and gas recirculation. The heat required for the process is derived from combustion of the biogas produced. The quantity of gas produced generally varies in the range 0.9-1.4 m^3 per kg volatile solids removed. Its composition is typically 65-70% methane and 30-35% carbon dioxide with traces of hydrogen sulphide.

Experience shows that the mesophilic anaerobic digestion process, when applied to domestic sewage sludge, is potentially self-sufficient in energy, including that required for process heating, with some to spare.

16.7.2 Aerobic digestion

Aerobic stabilization refers to the separate aeration of primary, secondary or mixed sewage sludges resulting in the microbial oxidation of organic material. The primary oxidized end products produced are carbon dioxide and water. The process requires an input of oxygen through an aeration system. The amount of oxygen consumed depends on the volatile solids composition and has been found (Kambhu and Andrews, 1969) to vary from 1.2 to 1.5 kg oxygen per kg volatile solids oxidized. The process is an exothermic one, the heat of reaction being approximately equal to the heat of combustion of the volatile solids (fair and Moore, 1932).

The rate at which aerobic digestion proceeds is influenced by many factors, including the concentrations of microorganisms, biodegradable solids and inorganic nutrients, temperature, mixing intensity, etc. It has been found that conversion rates increase by about 3-7% per °C rise in temperature. Thus, aerobic digestion proceeds very rapidly in the thermophilic temperature range (45-65 °C) where the evolution of heat can be sufficient to make the process thermally self-sustaining (Matsch and Drnevich, 1977).

The process, as generally employed, operates at ambient temperatures. The design is usually based on the single parameter of solids residence time, which is normally in the range 10-20d.

Recommendations in the United States (Weston, Inc. 1971) for aerobic digester design are summarised in Table 16.7.

Table 16.7 Aerobic digestion design parameters

Parameter	Value	Remarks
Hydraulic retention time	15-20d	Waste activated sludge alone
	20-25d	Primary + waste activated sludge
Diffused air aeration - air requirements	0.02-0.035	Waste activated sludge alone
	0.06	Primary + waste activated sludge.
	$m^3 min^{-1} m^{-3}$	Sufficient to keep solids in suspension
Surface aeration systems	26-33 W m^{-3}	This level is governed by mixing requirements. Most mechanical aerators in aerobic digesters require bottom mixers for solids concentrations greater than 8000 mg l^{-1}, especially if deep tanks are used (>4m).
Temperature	15 °C	If sludge temperatures are lower than 15 °C, then additional retention time should be provided so that stabilization will occur at the lower biological reaction rates.
Volatile solids reduction	40-50%	
Tank design		Aerobic digestion tanks are open and generally require no special heat transfer equipment or insulation. For small treatment systems the tank design should be flexible enough so that the digester tank can also act as a sludge thickening unit. If thickening is to be utilised in the aeration tank, then membrane-type diffusers should be used to minimize clogging.

Source: Westin (1971)

16.7.3 Chemical stabilisation

Lime effects stabilization by raising the sludge pH to a level that inhibits all microbial activity. It would appear (Vesilind, 1975) that a sustained pH value in the region 11-12.2 achieve this objective, including the destruction of pathogenic organisms. The addition of lime to primary sludge has been found to reduce its specific resistance to filtration (Farrell et al., 1974). The method, however, does not achieve a permanent stabilisation because during long-term storage in lagoons, the pH gradually drops to a level that permits a resumption of microbial activity and so eventually risking the generation of nuisance odour conditions.

16.8 SLUDGE DISPOSAL

The selection of a specific disposal destination for sludge is one of the most important decisions to be made in wastewater treatment system design, particularly for larger works. The choice rests between disposal to agriculture, landfill or incineration with landfill disposal of the ash residue. It is influenced by the quantity and composition of the sludge and the location of the works relative to potential disposal sites. The extent of processing prior to disposal to any of these sites is determined by technical, economic and environmental considerations. Economic and environmental criteria are particularly important determinants. Sludge processing and disposal may account for over 50% of both the capital investment and operational costs of conventional activated sludge-based wastewater treatment systems. Environmental protection considerations impose constraints on disposal and exclude disposal options that are likely to cause environmental damage.

When sludge is used in agriculture it should be processed in such a manner as to retain the desired nutrients. Where sludge is to be transported in liquid form, considerable economy can usually be achieved by the use of a thickening and/or dewatering process to reduce its water content. Disposal to agriculture in thickened slurry form would appear to be an attractive solution, where it is economically and environmentally feasible. This method of disposal is, of course, climate-dependent and requires the provision of on-site storage to cater for those periods of the year where application to land is not feasible.

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