Chapter 3

Treatment Mechanisms

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Learning Objectives

- Know the difference between physical, chemical and biological treatment mechanisms.
- Understand how combinations of physical, chemical and biological treatment mechanisms are responsible for faecal sludge treatment.
- Obtain a basic insight into how mechanisms affect the operation and maintenance needs of treatment technologies.
- Understand the mechanisms for key parameters that can be controlled to increase treatment efficiency and meet treatment objectives.

3.1 INTRODUCTION

This chapter presents an overview of the mechanisms on which faecal sludge (FS) treatment processes are based, and highlights those on which the treatment technologies discussed in subsequent chapters rely. Many FS treatment technologies are based on those developed for wastewater and wastewater sludge treatment, but it is important to remember that these technologies cannot be directly transferred. FS characteristics differ greatly from wastewater, and have a direct impact on the efficiency of treatment mechanisms (Spellman, 1997; Kopp and Dichtl, 2001). Important properties of the sludge to consider include stabilisation, organic load, particle size and density, dissolved oxygen, temperature, pH, water content and viscosity. The current understanding of physical, biological and chemical mechanisms in FS management (FSM) is limited and has been acquired via empirical observations over the years. It is important that this lack of knowledge is overcome in order to improve the design and operation of FS treatment technologies. For more detailed background information on wastewater treatment mechanisms, it is recommended that the reader refers to an engineering textbook specific to this topic.

This chapter is divided into three sections, presenting the physical, biological, and chemical mechanisms used for the treatment of FS. Physical mechanisms include dewatering, drying and volume reduction. These are the most widely employed mechanisms in current FS treatment methodologies, and are generally considered to be robust. Biological mechanisms allow the removal and transformation of organic constituents, nutrients and pathogens via the activity of microorganisms. Chemical mechanisms involve employing additives to optimise and control desired reactions, and are mainly used for disinfection and enhanced dewatering.

3.2 PHYSICAL MECHANISMS

One of the most important treatment mechanisms in FSM is dewatering. FS is mainly comprised of water, the proportion of which is dependent on the type of onsite technology. Water is heavy and expensive to transport, and discharging this polluted water to the environment has significant negative impacts. Dewatering is also necessary prior to resource recovery for applications such as composting, or combustion as a fuel. Dewatering is based on physical processes such as evaporation, evapotranspiration, filtration, gravity, surface charge attraction, centrifugal force and pressure.

Water in FS can be available in free or bound forms. This is an important differentiation in understanding treatment mechanisms because the free water is fairly easily removed, while removal of the bound water is much more difficult (Kopp and Dichtl, 2001). Free water (also referred to as bulk water) usually represents the majority of water in untreated sludge. It can be separated from the solid phase by dewatering technologies such as settling or filtration. It is not adsorbed, bound, or influenced by capillary forces. As shown in Figure 3.1, bound water includes interstitial, surface, and intracellular forms of bound water. Interstitial water (also referred to as capillary water) is in pore spaces, but bound to solids through capillary forces. Surface water (also referred to as colloidal water) is bound to solids and microorganisms by adsorption and adhesion. Intracellular water is contained within microorganisms, and is only removed by treatment mechanisms that result in the lysing of cells, thus releasing the liquid. When water is physically bound to solids, it is much more difficult to remove than free water; it requires the addition of chemicals or the use of centrifugation, pressure or evaporation.

3.2.1 Gravity separation

Gravity is probably the most commonly employed method of liquid – solid separation in FSM. It can achieve the separation of suspended particles and unbound water. Particles that are heavier than water settle out under quiescent conditions at rates based on size of particles, suspended solids concentration, and flocculation. These basic fundamentals are used in the design of settling-thickening tanks (Chapter 6) and grit chambers.

The four types of settling mechanisms include discrete particle, flocculent, hindered, and compression. Discrete particle settling occurs in lower concentration waste streams when particles settle out individually without reacting with other particles. Flocculent settling occurs when particles join together and merge, increasing their mass and settling velocity. This is important for smaller particles that are held together through Van der Waals force, resulting in increased settling velocities. Hindered settling occurs in highly concentrated waste streams, where the particles settle out together as a 'blanket'. Compression occurs at the bottom of a settling tank when the sludge blanket is 'squeezed' by the weight of the solids from above, removing more liquid.

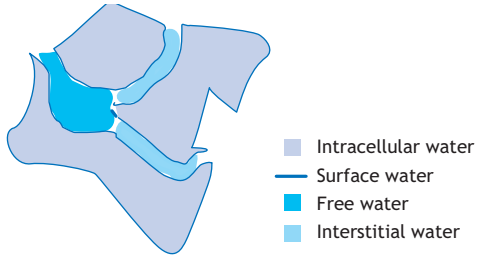

Figure 3.1 Water forms in a sludge floc (adapted from Kopp and Dichtl, 2001).

There are three main forces influencing the settling of a particle; gravity, buoyancy and drag force (or frictional resistance). The gravity force is dependent on the densities of the particle and the fluid, and the volume of the particle. The force due to gravity can be calculated as presented in Equation 3.1.

Equation 3.1: $F_g = \text{particle mass} \cdot g = \rho_p \cdot V_p \cdot g = \rho_p (1/6\ \pi d_p^{\ 3})g$

Where:
F_g = Force due to gravity (N)
ρ_p = Particle density (kg/m^3)
V_p = Particle volume (m^3)
d_p = Diameter of the particle (m)
g = Gravitational constant (9.81 m/s^2)

The force due to buoyancy is in the opposite direction to that of gravity, (represented by the negative sign in Equation 3.2), and it is dependent on the density of the liquid.

Equation 3.2: $F_b = \text{liquid mass} \cdot g = -\rho_L \cdot V_p \cdot g = -\rho_L (1/6\ \pi d_p^{\ 3})g$

Where:
F_b = Force due to buoyancy (N)
ρ_L = Density of liquid (kg/m^3)
V_p = Particle volume (m^3)
g = Gravitational constant (9.81 m/s^2)
d_p = Diameter of the particle (m)

The drag force is dependent on the particle velocity and diameter, the fluid density and viscosity, and a drag coefficient which is a function of the Reynolds number and the flow regime (laminar, transitional, and turbulent). The drag force is also in the opposite direction to that of gravity. For low Reynolds number (non-turbulent flows) and spherical particles, the drag force can be represented by the Stokes' law as shown in Equation 3.3:

Equation 3.3: $F_d = -3\pi\mu d_p$

Where:
μ = Water viscosity (N · s/m^2)
d_p = Diameter of the particle (m)

When the sum of the gravity, buoyancy and drag forces equals to 0, the particle is at its terminal settling velocity. The length of the tank needed for this particle to settle can be calculated based on this velocity, and the design parameter is the superficial area (width times length). Equation 3.4 is called Stokes' Law for settling, where the terms for $F_g + F_b + F_d = 0$ are substituted:

Equation 3.4: $$v = \frac{(\rho_p - \rho l)gd_p^2}{18\mu}$$

Flotation occurs when suspended solids have a similar or lower density to water, for example algal cells, fats, oils, and grease. Air bubbles can attach to particles, and if they have a similar density to water, this is sufficient to float them to the surface. The layer that forms on the surface of the liquid is referred to as 'scum'. In the design of settling tanks and stabilisation ponds for FS, it is important to address the scum layer as there is typically a significant accumulation (Figure3.2).

Figure 3.2 Settling tank at Niayes faecal sludge treatment plant in Dakar, Senegal (photo: Linda Strande).

3.2.2 Filtration

Filtration is also a commonly applied mechanism for liquid – solids separation in FSM. Several filtration media (e.g. membrane, granular) and types (e.g. slow, rapid, gravity driven or pressurised) are applied to water, wastewater and treated sludge (biosolids) processing. However, in FSM the most common types are unplanted and planted drying beds. These processes use filter media to trap solids on the surface of the filter bed, while the liquid percolates through the filter bed and is collected in a drain, or evaporates from the solids. In filter drying beds, slow filtration is occurring with filtration rates of 0.1-0.4 m/h, which requires less operations and maintenance than faster rates.

The parameters that have the greatest impact on slow filtration efficiency are the characteristics of the influent, the type of filtration media, and the filter loading rate (Metcalf and Eddy, 2003). For example, higher suspended solids concentrations in the influent can increase filter clogging, floc strength can impact the solids retained at the surface and the overall performance, and particle size distribution can affect performance as smaller particles are not as effectively removed by filtration.

Variable sizes of filter media can be used. Coarse media (e.g. gravel) has more pore spaces and allows the passage of more solids, whereas finer media provides a greater frictional resistance to the liquid flow and removes more solids. The need for solids removal must be balanced with the solids concentration of loaded FS and the potential for clogging. FS drying beds are usually designed with layers of increasing size media, from sand at the top to gravel at the bottom (Chapters 7 and 8). The flow rate of the liquid fraction that percolates (by gravity) through the bed is dependent on the resistance to flow exerted by the filter. The rate is calculated and reported as the volume passing through the filter in one hour, divided by the filter surface area. The depth of the filter determines the hydraulic retention time, and the head loss of the liquid, or the energy each unit volume needs to flow through the filter.

The main physical mechanisms that are responsible for filtration are shown in Figure 3.3. As these processes cannot be individually quantified, the design of drying beds relies on empirical calculations. Straining is the exclusion of particles based on size because they are larger than the pore size and not able to pass through. Sedimentation onto the media is a result of gravity settling. Interception is when particles come into contact with media as a result of the path of the liquid flow. Adhesion occurs when particles are removed from the liquid flow when they stick to the filter media. Flocculation is the joining of particles to form larger particles which are then removed by one of the above four mechanisms.

Diverse models have been developed based on filtration mechanisms to explain observed behaviors. Darcy's law can be used in the modeling of slow sand filtration, as the flow is considered to be slow enough to ensure laminar conditions. The resistance of the filtering media is best determined in the laboratory. The resistance to flow exerted by a 'clean' filter can be given by Equation 3.5 (Huisman and Wood, 1974):

Equation 3.5: $$H = \frac{v_f}{k} \cdot h$$

Where:
H = Resistance of the clean filter bed or pressure drop (m)
v_f = Filtration rate per unit area of the bed (m/s)
h = Thickness of the bed (m)
k = Permeability coefficient (m/s).

During operation of the filter, the effective pore sizes will become smaller as particles are trapped in the filter, and the growth of a biological biofilm develops on the media. This phenomena, known as 'ripening', results in increased efficiency of the filter with the retention of greater amounts of finer particles. With slow sand filtration, most of the solids are trapped on the surface of the filter. This increases the resistance for liquids to pass the surface of the filter, and result in reduced flow rates as head losses increase. This can result in clogging of the filter and a rapid decrease of filtration efficiency. To prevent this from occurring, care needs to be taken when designing, building and operating the filter (Chapters 7 and 8). It is also important to use washed sand and gravel when building a filter bed, to ensure that fine soil particles do not result in filter clogging.

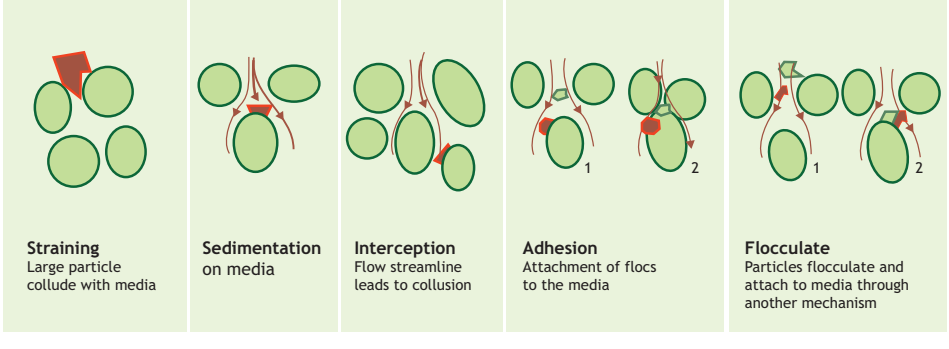

Figure 3.3 Schematic presentation of the mechanisms affecting flow in filter media (modified from Metcalf and Eddy, 2003).

Figure 3.4 Drying beds at Niayes faecal sludge treatment plant, Dakar, Senegal (photo: Linda Strande).

In addition to physical mechanisms, chemical and biological processes also occur within the filter. Chemical processes include attraction processes that result in flocculation or adhesion to filter surfaces. Biological growth happens throughout the filter, but tends to be more intense near the surface, depending on the presence of oxygen, carbon sources and nutrient availability. This can also result in biological removal of nutrients and BOD occurring within the filter (Panuvatvanich *et al.*, 2009).

3.2.3 Evaporation and evapotranspiration

Evaporation occurs when water is released into the air as a vapour, and transpiration is the process by which plants release water vapour to the air as a part of their metabolic processes. Evapotranspiration is the combination of these two mechanisms. In addition to filtration, dewatering in drying beds is also occurring through evaporation, and with planted drying beds through evapotranspiration. For both mechanisms to occur, the surrounding environment needs to have an evaporative demand, which means that the air is not saturated.

The energy required for evaporation to occur is provided by solar energy (with losses due to convection). Thus, evaporation is strongly influenced by climate, and the available heat and moisture content of air are especially important. The surface from where the evaporation is occurring can also influence the evaporation rate (e.g. free standing water versus water in sludge) (Musy and Higy, 2004). Important parameters are depth and total area of the drying bed. The larger the total mass of an object, the more energy that can be stored, increasing the heat requirement for evaporation. Wind speed also has an effect on the rate of evaporation, as it increases the replacement of saturated air with dry air. As illustrated by Dalton's law of partial pressure, the rate of evaporation depends on wind velocity and the vapor pressure of the air (Equation 3.6):

Equation 3.6: $E_a = f(u) \cdot (e'_a - e_s)$

Where:
E_a = Contribution of mass transfer to evaporation (mm/day)
$f(u)$ = Proportionality constant of the wind velocity
e'_a = Vapor pressure of water at saturation at the temperature of the surface (mm of mercury)
e_s = Effective vapor pressure (mm of mercury)

The Penman formula shown in Equation 3.7 utilises Dalton's law and incorporates empirical factors to calculate evaporation based on local climatic data. This type of information can typically be found on websites such as the Food and Agriculture Organisation (FAO) of the United Nations (www.fao.org), and in documents such as 'Crop evapotranspiration – Guidelines for computing crop water requirements' (Allen *et al.*, 1998).

Equation 3.7: $$E = \frac{+2\gamma}{+\gamma} \cdot E_c - \left(\frac{\gamma^{(2-\lambda)} \cdot E_2}{+2\gamma} \right)$$

Where:
E = Evaporation in (mm)
Δ = Slope of the saturation vapor pressure curve (kPa/°C)
γ = Psychrometric constant (kPa/°C), $\gamma = 0.00163 \cdot P/\lambda$, where P = Atmospheric pressure
E_c = Evaporation measured on a Colorado basin (mm)
λ = Latent heat of vaporisation

As with evaporation, rates of transpiration are influenced by heat, moisture, and wind, but are also dependent on additional factors such as plant species, growth phases, plant density, leaf shape and color, and water availability in the root zone (Stefanakis and Tsihrintzis, 2011). During transpiration, water is transported through the internal circulation system of the plant, and then released by the stoma, pores on the surface of leaves. The rate of evapotranspiration is always greater than the rate of evaporation alone (Musy and Higy, 2004). It has also been observed that temperature variations have a greater influence on evapotranspiration than on evaporation (Stefanakis and Tsihrintzis, 2011). For optimal evapotranspiration to occur with planted drying beds, the sludge load and rainfall data need to be considered to allow maximum biomass production of plants.

There are a few accepted methods for measuring rates of evapotranspiration (Musy and Higy 2004). Potential Evapotranspiration accounts for the theoretical water loss through evapotranspiration assuming that adequate water is available, there is dense plant coverage, and plants are in an intensive-growth phase. Maximal Evapotranspiration can be determined for individual plant species and for each growth phase, assuming optimal growing conditions. Actual Evapotranspiration utilises the recorded evaporation rate, relative humidity, and growth state of plants. This value is always smaller than the Maximal Evapotranspiration. To measure transpiration, water loss is measured in vegetated reference sites. This is more complex than measuring evaporation as the vegetation type and comparison to standard reference types needs to be considered. Calculating evapotranspiration therefore relies on values that are experimentally determined for a defined context and location. Therefore, the extrapolation involves control experiments and adjustments. The Penman-Monteith equation, derived from the Penman equation (as presented in Equation 3.7) is used to evaluate the potential evapotranspiration rate (Allen *et al.*, 1998; Uggetti *et al.*, 2012). This equation (Equation 3.8) allows the comparison of evapotranspiration at different periods of the year, in different locations, and between several types of plants.

Equation 3.8: $$PET = \frac{0.408 \cdot \Delta \cdot (R_n - G) + \gamma \frac{C_n}{T+273} u_2 \cdot (e_s - e_a)}{\Delta + \gamma \cdot (1 + C_d \cdot u_2)}$$

Where:
PET = Reference evapotranspiration in (mm/day)
Δ = Slope of the saturation vapor pressure curve (kPa/°C)
R_n = Net radiation at the crop surface (MJ/m^2/day)
G = Soil heat flux density (MJ/m^2/day)
γ = Psychrometric constant (kPa/°C), $\gamma = 0.00163 \cdot P/\gamma$, where P = Atmospheric pressure

Figure 3.5 Planted drying beds in the garden of a school, Bangkok, Thailand. The large leafs contribute to the evapo-transpiration and dewatering of the sludge (photo: Linda Strande).

C_n = Coefficient: 900 for short vegetation, and 1600 for tall vegetation
T = Mean daily air temperature at 2m height (°C)
u_2 = Daily wind speed at 2m height (m/s)
e_s= Vapor pressure at saturation at the temperature of the surface (kPa)
e_a= Effective vapor pressure (kPa)
C_d = Coefficient: 0.34 for short vegetation, and 0.38 for tall vegetation (mm)

3.2.4 Centrifugation

Centrifugation is mainly used for liquid – solid separation of wastewater sludge, but can also be employed for FS for the partial removal of bound water. Sludge is placed inside the centrifuge while it rotates at a high speed, and the centrifugal forces accelerate the sedimentation process. Solids settle out at the centrifuge walls, where they are pressed and concentrated. The liquid and solid fractions are then collected separately.

This process relies on the fact that when a particle in movement is forced to change its direction, it exerts a force against any obstacle toward its initial movement. The centrifugal force driving the movement from the center of a cylinder to its surface can be calculated by Equation 3.9 (Spellman, 1997):

Equation 3.9: $F_c = Wr(\rho_s - \rho)V$

Where:
Fc = Centrifugal force (N)
W = Angular velocity (radian per second)
r = Radius from the center of rotation to the particle (m)
ρ_s = Density of the particle (kg/m^3)
ρ = Density of the liquid (kg/m^3)
V = Volume of the particle (m)

The parameters influencing the efficiency of the centrifugation are not completely understood, but three important characteristics that were identified to be important with sludge from wastewater treatment plants are the settleability, the scrollability and the floc strength (Kopp and Dichtl, 2001).

3.2.5 Heat drying

Heat drying is used to evaporate and dewater wastewater sludge (biosolids) beyond what can be achieved with other more passive, or conventional methods. Currently, heat drying is applied more for wastewater sludge processing than for FS, but this technology should be transferable, and further information can be obtained from manufacturers and pilot studies.

Heat drying achieves both weight and volume reduction, as water is lost in the form of vapour. The temperature of the sludge is increased through energy transferred from an external heat source, which allows the evaporation of free water at the sludge surface, at a rate that depends on the ambient air temperature, humidity, flow and pressure, and the exposed sludge surface. As heat is continuing to be transferred, internal moisture is also being transferred to the surface and evaporated at a rate that depends on the physical characteristics of the sludge, the temperature and the moisture content. Heat drying involves convection, conduction, radiation, or a combination of these processes. Convection is employed in direct drying systems, conduction in indirect drying systems, and radiation in infrared drying systems (Chapter 5).

The amount of heat required depends on the specific heat capacity of the FS, which is the amount of energy required to raise the temperature of a unit mass by 1 °C. For example, the specific heat capacity of water at 25 °C is 4.18 kJ/kg/°C, which means that 4kJ are needed to raise the temperature of 1kg of water by 1 °C. No literature values were found for the heat capacity of FS, but the heat capacity of solids in wastewater sludge is reported to be 1.95 kJ/kg/°C (Kim and Parker, 2008).

3.2.6 Screening

Screening is another important physical mechanism in FSM. Bar screens at the influent of a FSTP are imperative to remove municipal waste and large solid objects from the FS, thereby preventing clogging and pump failures, and enhancing the value of treatment endproducts. Bar screens installed in a vertical or inclined position against the incoming flow make a physical barrier that retains coarse solids (Figure 3.6). The distance between the bars are set such that the liquid and small solid particles can flow through while the larger solids are trapped.

The velocity of the flow of FS through the bars influences the screen performance. A low velocity allows an increased removal of solids, but involves a greater solids deposition in the channel leading up to it, which should be avoided. Therefore, the flow velocity should reach, at a minimum, the self-cleansing velocity (greater than 0.3 m/s for wastewater). The flow should also not exceed 1 m/s in order to avoid coarse wastes being pulled through the bars due to the strength of the flow (Mara, 1976). The bars create a head loss that depends on the quantity and type of solid wastes retained.

Figure 3.6 Bar screen at Niayes faecal sludge treatment plant, Dakar, Senegal (photo: Linda Strande).

3.3 BIOLOGICAL MECHANISMS

In FSM, biology is essential in the achievement of treatment objectives through transformation of organic matter and nutrients. Biology is also important in understanding mechanisms of pathogen reduction. Pathogens of concern are presented in Chapter 2, while mechanisms of inactivation are covered in the following section.

Biological treatment harnesses the metabolism and growth rate of microorganisms in naturally occurring processes, and employs them in controlled situations to optimise the desired outcomes. Treatment systems usually rely on complex populations of microorganisms. As the microbes grow, they are dynamically altering the system, by modifying forms of organic matter, and releasing and binding up nutrients. They also release gases and other byproducts that can affect the environment.

The biodegradable organic matter in FS varies depending on the source, but usually needs to be stabilised prior to final enduse or disposal. Stabilisation involves the degradation of putrifiable, readily degradable material, leaving behind more stable, less degradable organics. This is important in order to reduce the oxygen demand, produce stable and predictable characteristics, reduce odors, and allow for easy storage and manipulation (Vesilind, 2001). 'Stabilised' organic matter does not have an exact agreed upon scientific definition, but in general refers to resistance to further biodegradation. Stabilised sludge consists of particles like cellulose, lignin, inorganic matter, and cellular matter of microorganisms

that consumed readily degradable organics, whereas unstabilised sludge contains easily degradable compounds such as carbohydrates, proteins, and sugars. Volatile solids are used as a measure for stabilisation, as they are considered to be composed of readily degradable organic matter. Equation 3.10 can be used to assess the level of degradation that has occurred during FS treatment. For 'raw' or 'fresh' FS (unstabilised FS) $\rho = 0$; when fully digested and stabilised, $\rho = 1$ (Kopp and Dichtl, 2001).

Equation 3.10: $\rho_{VSS} = (1-(VSS_1/VSS_0)) \cdot 100$

Where:
ρ = Percentage of degradation
VSS_1 = Volatile Suspended Solids (g/L) at time 1
VSS_0 = Volatile Suspended Solids (g/L) at time 0

3.3.1 Metabolism

For growth to occur, microorganisms need energy and carbon sources. As illustrated in Figure 3.7, bacteria are grouped together based on their metabolic properties, including energy source, carbon source, and electron receptors (e.g. aerobic or anoxic). Energy can be provided through solar energy or chemical forms (i.e. phototrophs, and chemotroph organisms), the chemical forms can be either organic or inorganic (i.e. chemoorganotrophs and chemolithotrophs). The carbon source used for the synthesis of new cells can be obtained from organic matter or carbon dioxide. Essential nutrients for growth include nitrogen, phosphorus, sulfur, potassium, magnesium, iron and calcium.

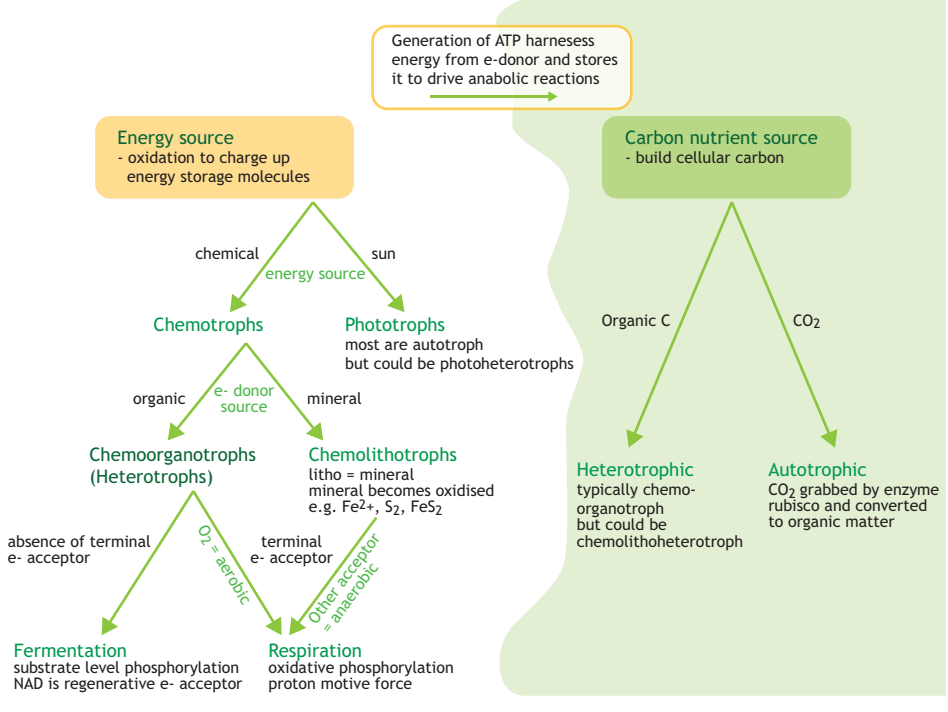

Figure 3.7 Nomenclature of microorganisms based on their energy and carbon requirements (figure: Linda Strande).

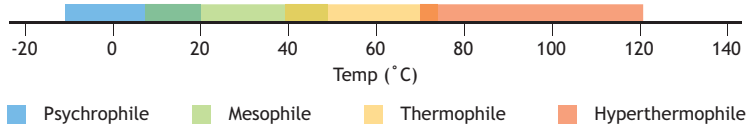

Figure 3.8 Partition of the different types of organisms based on their optimum temperature.

3.3.2 Temperature

The growth rate of microorganisms is also heavily reliant on temperature. Biological activity often doubles for every 10˚C increase in temperature within a given growth range for each organism. Each organism has a minimum temperature, defined as the point below which the organism cannot grow; an optimum temperature range, where enzymatic reactions happen at their greatest possible rates; and a maximum temperature, above which microorganisms can no longer grow due to denaturation of proteins. As shown in Figure 3.8, there are four types of organisms which can be defined depending on their optimal temperature range, namely: psychrophilic (optimal temperature at 15°C or lower), mesophilic (optimal temperature 20-45°C), thermophilic (optimal temperature 45-80°C) and hyperthermophilic (optimal temperature at 80°C or greater).

3.3.3 Types of microorganisms

All living organisms have a cellular structure that is either prokaryotic or eukaryotic. Prokaryotes consist of bacteria and archaea, are single celled, structurally less complex than eukaryotes, and their DNA is not enclosed in a nucleus. Bacteria are 0.5-1 µm in size, and have the shape of bacilli (rods), spirilla (spirals), or cocci (sphere). Archaea differ from bacteria in their evolutionary history. They are all chemotrophic, and many archea grow in extreme environmental environments (e.g. high temperature or salt content).

The cells of eukaryotes contain complex structures enclosed within membranes, and have a membrane bound nuclei. In FSM, the eukaryotes of greatest importance for treatment mechanisms are protozoa, fungi and algae, while pathogenic protozoa and helminthes determine pathogenic risk. Protozoa are unicellular, eukaryotic organisms. They are often motile and larger than bacteria, and are frequently predators of bacteria. Protozoa lack chlorophyll and cell walls. They also play an important role in waste stabilisation and maturation ponds, and can reduce pathogens through predation. Fungi are a large group of organisms, comprised of molds, yeasts, and mushrooms. They are chemoorganotrophs, can be aerobic or anaerobic, and live in a large variety of environments. Fungi are important in the stabilisation of more recalcitrant organic molecules (e.g. in composting). Algae are photoautotrophic, getting their energy from the sun, and carbon from CO_2. Algae use chlorophyll in a similar fashion to plants, and produce oxygen. They play an important role in waste stabilisation and maturation ponds.

Viruses are smaller than bacteria (20-300 nanometers), and consist of a core of RNA or DNA in a protein capsid. They are able to infect plants, animals, and bacteria, and are not able to replicate without a host. They are typically not considered living organisms. In FSM, they are primarily a concern as a pathogen risk.

3.3.4 Aerobic treatment

Aerobic environments refer to the presence of oxygen, and aerobic organisms rely on oxygen for their respiration. Microorganisms can be either purely aerobic requiring oxygen to grow, or facultative meaning they are also able to survive in anaerobic conditions. Typical aerobic treatment processes in wastewater treatment include activated sludge, sequencing batch reactors, trickling filters and

facultative or maturation ponds. Aerobic processes occur in any solid or liquid treatment process where oxygen is present, including FS drying beds and composting.

Aerobic growth phases include oxidation and synthesis during rapid growth, and endogenous respiration. As shown in Equation 3.11 during oxidation organic matter is consumed, CO_2 released, and new cells are synthesised:

Equation 3.11: Organic matter + O_2 + nutrients → CO_2 + H_2O + $C_5H_7NO_2$ + other products

Where $C_5H_7NO_2$ are the newly synthesised cells.

Endogenous respiration corresponds to the period where readily degradable organic matter is depleted, and microorganisms consume cellular content to maintain metabolic reaction. This is shown in Equation 3.12:

Equation 3.12: $C_5H_7NO_2 + 5O_2 \rightarrow 5CO_2 + 2H_2O + NH_3$ + energy

The dissolved oxygen content in FS is very low due to microbial activity which rapidly depletes available oxygen, and the low solubility of oxygen in water. For processes to remain aerobic, they typically rely on aeration or mixing, which can be energy intensive.

3.3.5 Composting

Composting is the controlled process by which biological decomposition of organic matter occurs by the same organisms that naturally degrade organic matter in the soil. The resulting endproduct is a dark, rich, humus-like matter that can be used as a soil amendment. Humus is defined as the stable fraction of the soil organic matter remaining after the major portions of added plant and animal residues have decomposed. Important mechanisms that govern this process are the oxidation of organic compounds, the release and immobilisation of nutrients, and the microbial synthesis of new compounds.

In thermophilic composting, the system goes through a three phase process. During the first phase, bacteria are growing rapidly while consuming readily degradable compounds (e.g. sugar, starch, protein). During this period, the temperature is also increasing due to the rapid rate of growth (due to exothermic catabolic reactions), which is faster than the rate at which heat can escape. In the second phase, thermophilic temperatures of 50-75 °C are achieved and thermophilic bacteria become active, further decomposing the organic matter. During this phase pathogen reduction and inactivation of plant seeds (e.g. weeds) occurs as a result of the high temperatures. In the third phase, stabilisation is being reached as the last of the readily degradable substrates are depleted, bacterial activity slows down, and the temperature lowers. During this phase actinomycetes and fungi are further degrading more recalcitrant organic molecules such as cellulose and lignin.

The composting process is controlled through the optimisation of the carbon to nitrogen mass ratio (C:N), moisture content, and oxygen supply. The empirically observed optimal C:N is between 20 and 30, based on the ratio that microbes utilise carbon and nitrogen during their growth. There needs to be a balance of enough carbon for cell synthesis and energy extraction, with nitrogen for synthesis of amino acids, enzymes and DNA. If the C:N is lower than 20 there will be excess nitrogen in the system that will be lost following mineralisation due to nitrate leaching or ammonia volatilisation. If the C:N is greater than 30, then the nitrogen is 'locked up' in organic matter and not bio-available.

The optimal moisture content is 40-60% by weight. Moisture is necessary for biological growth to occur, and to transport nutrients throughout the compost pile. If the moisture content is greater than 60%, then it can impede microbial growth by creating anaerobic conditions.

The free pore space should represent at least 20% of the volume, with an optimal oxygen content of the air greater than 10% to ensure that aerobic decomposition is occurring. To achieve this, it is important to have a mix of different textures of materials to allow for oxygen to pass through the pile, and to turn the pile at certain intervals to introduce oxygen, and redistribute partially decomposed matter. Turning the pile lowers the temperature as ambient air is introduced, but is followed by a rapid increase in temperature as biological activity speeds back up again. This process continues until the third stabilisation phase of the composting process is reached.

3.3.6 Anaerobic treatment

Anaerobic conditions are characterised by the lack of oxygen. Anaerobic degradation occurs anywhere in FSM systems where oxygen has been depleted, for example anaerobic and facultative waste stabilisation ponds, septic tanks, and settling tanks. Anaerobic fermentation can also be employed for the treatment of sludge. Anaerobic digesters can provide a beneficial method of stabilising FS, as it also results in the production of biogas that can be used for energy generation. Biogas is a mixture of mainly methane (55-75%) and carbon dioxide (30-45%) (Arthur *et al.*, 2011). Due to the less energetically favorable nature of anaerobic metabolisms, less sludge (i.e. microbial biomass) is produced during anaerobic digestion.

Anaerobic digestion is a complex process characterised by hydrolysis, fermentation, acidogenesis, acetogenesis and methanogenesis. Hydolysis is an enzymatic process through which particulate matter and more complex organic compounds are degraded and become more bioavailable. At the same time, proteins, lipids, and polysaccharides are converted into amino acids, fatty acids, and monosaccharides. During fermentation (or acidogenesis) acidogenic microorganisms further degrade amino acids, sugars, and fatty acids to methogenic substrates (e.g. H_2, CO_2, formate, methanol, methylamines, and acetate). Organic molecules are used as both electron donors and accepters. Therefore, methanogen archea can be characterised as chemoorganotrophs (Figure 3.7). During methanogenesis, one group of archea split acetate into methane and carbon dioxide, while another group produces methane through the use of hydrogen and carbon dioxide. Methanogenesis occurs more readily at mesophilic (30- 38 °C) and thermophilic (49-57 °C) temperatures. Methanogenic processes are presented in Equation 3.13 to Equation 3.15 (Madigan *et al.*, 2003):

Equation 3.13: $4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$

Equation 3.14: $CH_3OH + H_2 \rightarrow CH_4 + H_2O$

Equation 3.15: $CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^-$

Acidogenic and methanogenic microorganisms have a syntrophic relationship. The methanogens use the hydrogen produced by acidogens, maintaining an optimal partial pressure for the acidogens. Hence, the slow growth rate of methanogens is the limiting step in the process. If this process slows down, then the volatile fatty acids produced by acidogens will build up in the reactor, resulting in a lowered pH, and a further disruption of the methanogenic activity. When this occurs, it is referred to as the digester going 'sour'. Because of this carefully balanced microbial relationship, it is important to ensure that consistent operation and monitoring are occurring, with pH monitoring being the most convenient and useful method. Methanogens are also strongly inhibited by the presence of oxygen, free ammonia, heavy metals, and sulfides.

Figure 3.9 Biogas reactors at 2iE in Ouagadougou, Burkina Faso (photo: Linda Strande).

3.3.7 Nitrogen cycling

Biological nitrogen cycling is an important aspect of FSM, as FS tends to be very high in ammonia nitrogen. Nitrogen is an essential nutrient that can be captured for beneficial enduse, and is also a potential pollutant that should not be indiscriminately discharged into the environment. Inorganic forms of nitrogen are available for microorganisms to use during growth. As shown in Figure 3.10, once it is utilised, nitrogen is immobilised and no longer bioavailable as it is bound up into organic molecules such as microbial cellular components and structures. Nitrogen is then mineralised and released back into bioavailable forms as organisms die off and the organic matter is degraded. The majority of nitrogen in FS is present as ammonia that is released during this hydrolysis process.

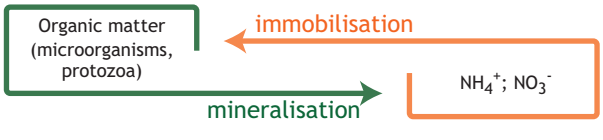

Figure 3.10 The process of mineralisation and immobilisation of nitrogen in the environment.

Nitrification

Ammonia nitrogen that is released during mineralisation, can be oxidised to nitrate through biological nitrification, which is an aerobic, autotrophic process. Ammonia oxidising bacteria oxidise ammonia to nitrite, rapidly followed by nitrite oxidising bacteria oxidising nitrite to nitrate, as shown in Equation 3.16 and Equation 3.17. This is a sensitive biological process and it is thus important to consider total nitrogen concentration, biochemical oxygen demand (BOD) concentration, alkalinity, pH, temperature, and potential for toxic compounds when designing systems that rely on nitrification (Metcalf and Eddy, 2003). The optimal temperature for nitrification is 28°C, with the process becoming inefficient below 10°C. The optimal pH range is between 7.5 and 8.0. Reasonable rates of nitrification occur at neutral pH (7.0), but become restricted below pH 6.8. Since nitrification is an aerobic process, it should be ensured that the dissolved oxygen concentration is greater than 1 mg/L. The nitrification process requires 7.14 g of alkalinity as calcium carbonate ($CaCO_3$) for each gram of ammonia nitrogen (as nitrogen) converted to nitrate (Metcalf and Eddy, 2003). Free ammonia concentrations above 100 mg/L at pH 7 can also inhibit the nitrification process (Metcalf and Eddy, 2003).

Equation 3.16: $2NH_2 + 3O_2 + \text{Nitrosomonas} \rightarrow 2NO_2^- + 2H_2O + 2H^+$

Equation 3.17: $2NO_2^- + O_2 + \text{Nitrobacter} \rightarrow 2NO_3^-$

Denitrification

Biological nitrogen removal happens in anoxic environments with the reduction of nitrate to nitrogen gas thereby releasing nitrogen to the air. Anoxic environments are low in oxygen, and nitrate is used as an electron receptor. Dissolved oxygen concentrations greater than 0.1-0.5 mg/L inhibit the anoxic process and the optimal pH range is 7.0-8.0. The process occurs with both heterotrophic and autotrophic bacteria, many of which are facultative aerobes. The process happens through a series of intermediate gaseous nitrogen oxide products. Denitrification generally proceeds through some combination of the intermediate forms shown in Equation 3.18.

Equation 3.18: $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$

Where:
NO_3^- = nitrate
NO_2^- = nitrite
NO = nitrogen oxide
N_2O = nitrous oxide
N_2 = nitrogen gas.

When designing a system that includes both nitrification and denitrification, it is important to ensure there is adequate BOD for denitrification to occur. More precise values can be calculated, but it is estimated that 4 g of BOD is needed per g of nitrate reduced (Metcalf and Eddy, 2003). In addtion 3.57 g of alkalinity as $CaCO_3$ is regained during denitrification which should be taken into account when calculating the total alkalinity requirement for nitrifying – denitrifying systems. Simultaneous nitrification and denitrification can also occur in anaerobic conditions without BOD with anammox bacteria that can oxidise NH_4^+ to N_2, utilising NO_2^- as an electron acceptor, which also gets reduced to N_2.

Phosphorus cycling

As with nitrogen, phosphorus is an essential nutrient that can be captured for beneficial enduse, and is also a potential pollutant that should not be indiscriminately discharged into the environment. Phosphorus in FS and excreta is mostly present as phosphates; molecules comprised of the acid or base

form of orthophosphoric acid (H_3PO_4) or phosphate (PO_4^{3-}), or as organically bound phosphorus (e.g. nucleic acids, phospholipids and phosphorylated proteins).

The fate of phosphorus in treatment processes depends on factors such as sorption, precipitation, complexation, sedimentation, mineralisation, pH, plant uptake, and redox potential. During degradation of organic material, bound phosphates are mineralised and released. Phosphate is not lost due to off gassing or leaching like nitrogen, as the soluble inorganic form is adsorbed in the sludge. During biological treatment processes, about 10-30% of phosphorus is taken up by microorganisms. This can be increased through biological dephosphatation, or through chemical precipitation with $FeCl_3$ or $Al_2(SO_4)_3$ or $FeSO_4$ which are used for wastewater treatment. The greatest loss of phosphorus during FS treatment is due to removal by plants in planted drying beds.

3.3.8 Pathogen reduction

In this section, the mechanisms that result in biological die-off of pathogens from physical, biological, and chemical mechanisms are covered. Types of pathogens are presented in more detail in Chapter 2. It is important to have an understanding of all of these interrelated mechanisms, to ensure that pathogen reduction is achieved during FS treatment. They affect all biological processes, which need to be considered to ensure that treatment processes function as designed.

Temperature

Most pathogens are inactivated above temperatures of 60°C when cell proteins and nucleic acids are denatured. This is achieved in processes such as thermophilic co-composting as shown in Figure 3.11, as well as lime treatment. As the temperature increases, less time is needed for pathogen inactivation.

Figure 3.11 Pilot scale co-composting facility with faecal sludge and municipal waste in Bangalore, India (photo: Chris Zurbrügg).

Time

The duration of treatment (e.g. planted drying beds) or the storage of treated sludge can result in pathogen reduction, as they have a limited survival time in adverse conditions. In faeces, most bacteria can only survive between 1 week and 2 months. For example, Salmonella spp. survives on average for 30 days and Faecal coliforms for 50 days (Feachem *et al.*, 1983). Helminth eggs however are very persistent, and can maintain viability for several months to years. The required storage duration for pathogen reduction also depends on the ambient temperature. For example, Niwagaba (2009) recommends storage time of FS for up to one year at an ambient temperature of 35°C, and two years at 20°C. Storage at temperatures less than 10°C does not result in adequate inactivation (Weemaes and Verstraete, 2001).

Sorption

Helminth eggs tend to sorb or settle, and hence partition with the solids fraction in FS systems. In settling and thickening tanks, about 50% of the helminth eggs are separated from the liquid fraction due to settling (Heinss *et al.*, 1998). In filtration that occurs with drying beds, the majority of Helminth eggs remain with the solid fraction, as does 90% of indicator bacteria (Pepper *et al.*, 2008). However, indicator bacteria are not necessarily representative of all possible pathogens (e.g. viruses, different types of bacteria, protozoan cysts). Although the majority of Helminth eggs partition with the solids fraction, the fate of all pathogens must be considered.

Desiccation

Evaporation resulting in desiccation or dehydration reduces active pathogens, as microorganisms need water for survival. Water activity is represented by the ratio of water vapour pressure of the sludge to the water vapour pressure of pure water under the same conditions. Pure water has a water activity of 1, and most pathogens cannot survive under a water activity of 0.9, while some yeast and eggs survive in much drier conditions (Carrington, 2001). All dewatering technologies therefore contribute to the die-off of pathogens (e.g. drying beds) if the water content gets below a certain point where desiccation has an effect. Further storage also contributes to disinfection due to the reduction of the available water.

UV

Solar/UV radiation in the range of 300-400 nm effectively inactivates pathogens by denaturing DNA molecules via photochemical reactions (Borrely *et al.*, 1998). UV light has been shown to effectively inactivate *E. coli* in waste stabilisation ponds (Maïga *et al.*, 2009). However, it is important to remember that for this mechanism to be effective, the light rays must be able to penetrate the FS during treatment. This mechanism is therefore most likely only occurring at the surface, as the high organic matter and turbidity prevents penetration of UV radiation.

pH

Most microorganisms can only survive and grow within a range of 2-3 pH units, and very few can survive below pH 3 and above pH 10. In this way, chemical addition for pH control can result in pathogen reduction. However, the pH can also upset composting and anaerobic digestion processes, and it is therefore important to consider downstream treatment steps when employing pH control for pathogen reduction.

3.4 CHEMICAL MECHANISMS

Chemicals can be mixed with FS to improve the performance of other physical mechanisms (e.g. addition of a cationic polymer to increase the flocculation and the settling efficiency), or to inactivate pathogens and stabilise FS. The addition of chemicals can represent a significant increase in the overall cost of treatment, and the benefits therefore need to be carefully weighed.

3.4.1 Alkaline stabilisation

Alkaline additives, such as lime, can be used for the stabilisation of FS, either pre- or post-dewatering. If carried out prior to dewatering, more additives will be required. Lime is also used to precipitate phosphorus from liquid streams in wastewater treatment plants, and to polish effluents. The addition of adequate quantities of lime to FS raises the pH to 12, which stops the microbial activity. This results in an odor reduction due to factors that cause putrification, and in a reduction of pathogens. This chemical reaction also hydrolyses fats, carbohydrates, and proteins, as well as ammonia from amino acids. If quick lime (CaO) is used, it will also result in an exothermic reaction that can increase the temperature up to 60°C (Andreasen, 2001), thereby increasing pathogen reduction, and inactivating Helminth eggs. The reaction has also been documented to increase settling efficiency. However, after the initial reaction, the pH will lower again, therefore requiring that lime is added in excess. It should also be noted that regrowth of bacterial pathogens can occur over time. Concerns with this process include ammonia odors and lime scaling.

3.4.2 Ammonia treatment

It is well established that aqueous ammonia is effective at inactivating microorganisms, but the exact mechanisms are not yet fully understood. As described in Vinnerås (2013), possible mechanisms for bacterial inactivation are that NH_3 denaturates proteins, destroys membrane potentials, or causes rapid alkalinisation of the cytoplasm resulting in a critical loss of potassium (K). Viral inactivation is possibly due to the cleavage of RNA, but for larger organisms, such as helminths, the mechanisms are still not fully understood. Ammonia disinfection has been shown to be effective in urine (Vinnerås *et al.*, 2008), sewage sludge (Pecson *et al.*, 2007), and compost (Adamtey *et al.*, 2009), but applications for FS are still in the research phase of development.

It is aqueous NH_3 that is responsible for microbial inactivation, not the ammonium ion (NH_4^+). The pK_a of ammonia is 9.25 (the pH where 50% is NH_3 and 50% is NH_4^+), and the percent NH_3 concentration based on pH can be determined by Equation 3.19.

Equation 3.19:
$$NH_3,\% = \frac{100}{1+[H^+]/K_a}$$

The total aqueous NH_3 concentration will also depend on temperature and total ammoniacal nitrogen concentration (NH_3+NH_4^+). For NH_3 disinfection to be effective, the pH has to be above 8.5 (Vinnerås, 2013). Ammonia can be added as as aqueous NH_3 solution, or urea ($CO(NH_2)_2$), which is rapidly enzymatically transformed to NH_3. The treatment needs to be carried out in a confined space to avoid loss of gaseous NH_3. At this stage, the time for inactivation of microorganisms needs to be determined empirically for each specific organism of concern. As long as the pH remains stable, the aqueous NH_3 will remain constant and pathogen regrowth will not occur (Vinnerås, 2013). If the treated FS is applied to the soil it will result in a decrease of pH, thereby increasing the concentration of NH_4^+, which is beneficial as a fertiliser.

3.4.3 Coagulation and flocculation

Colloidal particles that are not removed through gravity settling tend to be negatively charged, making them stable in suspension. In coagulation and flocculation additives are added that destabilise particles, allowing them to come in contact with each other, form larger flocs and settle, thereby achieving enhanced sedimentation. These additives are chosen based on the hydrophobic or hydrophilic characteristics of the particles, together with their surface charge.

Coagulation and flocculation are achieved by adding polymers that form a bridge between particles, or by adding potential determining ions (strong acid or base) that reduce the total surface charge. Polymers

can be natural or synthetic based chemicals. They work by either forming a bridge between the anionic and non-ionic ends of the polymer to particles, or by forming a bridge with high molecular weight polymers that are adsorbed to particles.

3.4.4 Conditioning

Chemical conditioning is based on the same physical properties as coagulation and flocculation, and can be carried out prior to physical forms of dewatering as described in Section 3.4.3 to enhance performance. Common additives include ferric chloride, lime, alum, and organic polymers. Iron salts and lime can increase the total solids of dried sludge (increasing bulk), whereas polymers do not increase the total solids. To select the appropriate chemical to use, important aspects to consider are sludge age, pH, source, solids concentration, and alkalinity. In general, the dosage is determined in the laboratory with simple settling jar tests. The information that is currently available relates to wastewater sludge treatment. To transfer this technology to FS, information from manufacturers, laboratory, and pilot-scale testing is necessary.

3.4.5 Disinfection of liquid effluents

Disinfection of liquid effluents is not covered in detail in this book, as it is not specific to FS, and is covered in detail in wastewater and water treatment references (e.g. ponds, wastewater treatment). Liquid effluents from settling tanks or drying beds typically require further treatment prior to disinfection. Disinfection should be considered as a 'polishing' step to achieve a final reduction in pathogens and is not a primary form of FS treatment. Liquid effluent treatment also needs to be considered in the context of adequate and appropriate levels of treatment for the intended enduse, as discussed in Chapter 10. Disinfection refers to a reduction of pathogens, not a total elimination (which is referred to as sterilisation). Chemical forms of disinfection include chlorination, ozonation, and UV, but it is also achieved through mechanical means such as filters or membranes.

Chlorination is the most widely used method of disinfection, and can be applied in either a solid or liquid form. Important design parameters include contact time, chlorine concentration, pathogen load, temperature, and other constituents in effluent (e.g. remaining organic load). Chlorine is toxic to microorganisms, as it has a high capacity for oxidation, which damages cell membranes. The oxidation process is not specific to microbes, and it is therefore important to consider the total organic load. Chlorination is not effective for disinfecting FS, or liquid effluents that contain high organics, as the chlorine will be used up in the oxidation of these other constituents.

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End of Chapter Study Questions

1. Give two examples each of physical, chemical and biological mechanisms, and what treatment technologies they are relevant to.

2. Does composting rely on physical, chemical or biological treatment mechanisms? What are three conditions that are required for efficient composting?

3. Which mechanisms are responsible for pathogen reduction and in which treatment technologies do they exist?