Chapter 25

Municipal Wastewater Treatment

Charles P. Gerba and Ian L. Pepper

25.1 THE NATURE OF WASTEWATER (SEWAGE)

The cloaca maxima, the "biggest sewer" in Rome, had at one time enough capacity to serve a city of one million people. This sewer, and others like it, simply collected wastes and discharged them into the nearest lake, river or ocean. This expedient made cities more habitable, but its success depended on transferring the pollution problem from one place to another. Although this worked reasonably well for the Romans, it does not work well today. Current population densities are too high to permit a simple dependence on transference. Thus, modern-day sewage is treated before it is discharged into the environment. In the latter part of the nineteenth century, the design of sewage systems allowed collection with treatment to lessen the impact on natural waters. Today, more than 15,000 wastewater treatment plants treat approximately 150 billion liters of wastewater per day in the United States alone. In addition, septic tanks, which were also introduced at the end of the nineteenth century, serve approximately 25% of the U.S. population, largely in rural areas.

Domestic wastewater is primarily a combination of human feces, urine and "graywater." Graywater results from washing, bathing and meal preparation. Water from various industries and businesses may also enter the system. People excrete 100–500 grams wet weight of feces and 1–1.3 liters of urine per person per day (Bitton, 2011). Major organic and inorganic constituents of untreated domestic sewage are shown in Table 25.1.

The amount of organic matter in domestic wastes determines the degree of biological treatment required. Three tests are used to assess the amount of organic matter: biochemical oxygen demand (BOD); chemical oxygen demand (COD); and total organic carbon (TOC).

The major objective of domestic waste treatment is the reduction of BOD, which may be either in the form of solids (suspended matter) or soluble. BOD is the amount of dissolved oxygen consumed by microorganisms during the biochemical oxidation of organic (carbonaceous BOD) and inorganic (ammonia) matter. The methodology for measuring BOD has changed little since it was developed in the 1930s.

The 5-day BOD test (written BOD_5) is a measure of the amount of oxygen consumed by a mixed population of heterotrophic bacteria in the dark at 20°C over a period of 5 days. In this test, aliquots of wastewater are placed in a 300-ml BOD bottle (Figure 25.1) and diluted in phosphate buffer (pH 7.2) containing other inorganic elements

I.L. Pepper, C.P. Gerba, T.J. Gentry: Environmental Microbiology, Third edition. DOI: http://dx.doi.org/10.1016/B978-0-12-394626-3.00025-9

TABLE 25.1 Typical Composition of Untreated Domestic Wastewater

Contaminants	Concentration (mg/L)		
	Low	Moderate	High
Solids, total	350	720	1200
Dissolved, total	250	500	850
Volatile	105	200	325
Suspended solids	100	220	350
Volatile	80	164	275
Settleable solids	5	10	20
Biochemical oxygen demand[a]	110	220	400
Total organic carbon	80	160	290
Chemical oxygen demand	250	500	1000
Nitrogen (total as N)	20	40	85
Organic	8	15	35
Free ammonia	12	25	50
Nitrites	0	0	0
Nitrates	0	0	0
Phosphorus (total as P)	4	8	15
Organic	1	3	5
Inorganic	3	5	10

From Pepper et al. (2006b).
[a]*Five-day test, (BOD_5, 20°C).*

(N, Ca, Mg, Fe) and saturated with oxygen. Sometimes acclimated microorganisms or dehydrated cultures of microorganisms, sold in capsule form, are added to municipal and industrial wastewaters, which may not have a sufficient microflora to enable the BOD test to be carried out. In some cases, a nitrification inhibitor is added to the sample to determine only the carbonaceous BOD.

Dissolved oxygen concentration is determined at time 0, and, after a 5-day incubation, by means of an oxygen electrode, chemical procedures (e.g., Winkler test) or a manometric BOD apparatus. The BOD test is carried out on a series of dilutions of the sample, the dilution depending on the source of the sample. When dilution water is not seeded, the BOD value is expressed in milligrams per liter, according to the following equation (APHA, 1998).

$$\mathrm{BOD(mg/L)} = \frac{D_1 - D_5}{P} \qquad \text{(Eq. 25.1)}$$

FIGURE 25.1 BOD bottle.

where:

D_1 = initial dissolved oxygen (DO), D_5 = DO at day 5, and
P = decimal volumetric fraction of wastewater utilized.

If the dilution water is seeded:

$$\mathrm{BOD(mg/L)} = \frac{(D_1 - D_5) - (B_1 - B_5)f}{P} \qquad \text{(Eq. 25.2)}$$

where:

D_1 = initial DO of the sample dilution (mg/L)
D_5 = final DO of the sample dilution (mg/L)
P = decimal volumetric fraction of sample used
B_1 = initial DO of seed control (mg/L)
B_5 = final DO of seed control (mg/L), and
f = ratio of seed in sample to seed in control = (% seed in D_1)/(% seed in B_1).

Because of depletion of the carbon source, the carbonaceous BOD reaches a plateau called the ultimate carbonaceous BOD (Figure 25.2). The BOD_5 test is commonly used for several reasons:

- To determine the amount of oxygen that will be required for biological treatment of the organic matter present in a wastewater
- To determine the size of the waste treatment facility needed
- To assess the efficiency of treatment processes
- To determine compliance with wastewater discharge permits

The typical BOD_5 of raw sewage ranges from 110 to 440 mg/L (see Example Calculation 25.1). Conventional sewage treatment will reduce this by 95%.

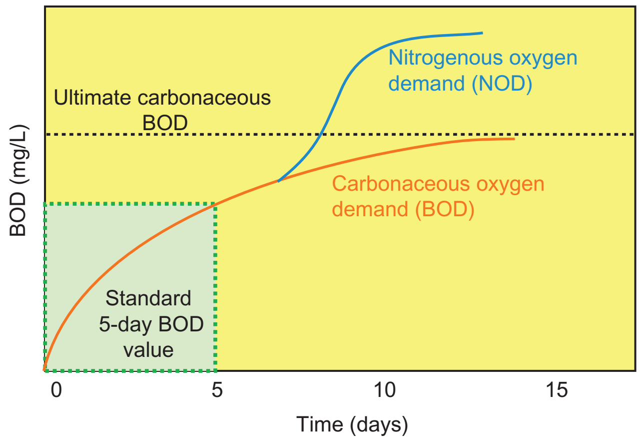

FIGURE 25.2 Carbonaceous and nitrogenous BOD.

Example Calculation 25.1 Calculation of BOD

Determine the 5-day BOD (BOD_5) for a wastewater sample when a 15-ml sample of the wastewater is added to a BOD bottle containing 300 ml of dilution water, and the dissolved oxygen is 8 mg/L. Five days later the dissolved oxygen concentration is 2 mg/L.

Using Eq. 25.1:

$$\text{BOD (mg/L)} = \frac{D_1 - D_5}{P}$$

$$D_1 = 8 \text{ mg/L}$$
$$D_5 = 2 \text{ mg/L}$$
$$P = \frac{15 \text{ ml}}{300 \text{ ml}} = 5\% = 0.05$$

$$BOD_5 = \frac{8-2}{0.05} = 120 \text{ mg/L}$$

Chemical oxygen demand (COD) is the amount of oxygen necessary to oxidize all of the organic carbon completely to CO_2 and H_2O. COD is measured by oxidation with potassium dichromate ($K_2Cr_2O_7$) in the presence of sulfuric acid and silver, and is expressed in milligrams per liter. In general, 1 g of carbohydrate or 1 g of protein is approximately equivalent to 1 g of COD. Normally, the ratio BOD/COD is approximately 0.5. When this ratio falls below 0.3, it means that the sample contains large amounts of organic compounds that are not easily biodegraded.

Another method of measuring organic matter in water is the TOC or total organic carbon test. TOC is determined by oxidation of the organic matter with heat and oxygen, followed by measurement of the CO_2 liberated with an infrared analyzer. Both TOC and COD represent the concentration of both biodegradable and nonbiodegradable organics in water.

TABLE 25.2 Types and Numbers of Microorganisms Typically Found in Untreated Domestic Wastewater

Organism	Concentration (per ml)
Total coliforms	10^5–10^6
Fecal coliforms	10^4–10^5
Fecal streptococci	10^3–10^4
Enterococci	10^2–10^3
Shigella	Present
Salmonella	10^0–10^2
Clostridium perfringens	10^1–10^3
Giardia cysts	10^{-1}–10^2
Cryptosporidium cysts	10^{-1}–10^1
Helminth ova	10^{-2}–10^1
Enteric virus	10^1–10^2

From Pepper et al. (2006b).

Pathogenic microorganisms are almost always present in domestic wastewater (Table 25.2). This is because large numbers of pathogenic microorganisms may be excreted by infected individuals. Both symptomatic and asymptomatic individuals may excrete pathogens. For example, the concentration of rotavirus may be as high as 10^{10} virions per gram of stool, or 10^{12} in 100 g of stool (Table 25.3). Infected individuals may excrete enteric pathogens for several days or as long as a few months. The concentration of enteric pathogens in raw wastewater varies depending on the following:

- The incidence of the infection in the community
- The socioeconomic status of the population
- The time of year
- The per-capita water consumption

The peak incidence of many enteric infections is seasonal in temperate climates. Thus, the highest incidence of enterovirus infection is during the late summer and early fall. Rotavirus infections tend to peak in the early winter, and *Cryptosporidium* infections peak in the early spring and fall. The reason for the seasonality of enteric infections is not completely understood, but several factors may play a role. It may be associated with the survival of different agents in the environment during the different seasons. *Giardia*, for example, can survive winter temperatures very well. Alternatively, excretion differences among animal reservoirs may be involved, as is the case with

TABLE 25.3 Incidence and Concentration of Enteric Viruses and Protozoa in Feces in the United States

Pathogen	Incidence (%)	Concentration in Stool (per gram)
Enteroviruses	10–40	10^3–10^8
Hepatitis A virus	0.1	10^8
Rotavirus	10–29	10^{10}–10^{12}
Giardia	3.8 18–54[a]	10^6 10^6
Cryptosporidium	0.6–20 27–50[a]	10^6–10^7 10^6–10^7

[a]*Children in day care centers.*

TABLE 25.4 Estimated Levels of Enteric Organisms in Sewage and Polluted Surface Water in the United States

Organism	Concentration (per 100 ml)	
	Raw Sewage	Polluted Stream Water
Coliforms	10^9	10^5
Enteric viruses	10^2	1–10
Giardia	10–10^2	0.1–1
Cryptosporidium	1–10	0.1–10^2

From U.S. EPA (1998).

Cryptosporidium. Finally, it may well be that greater exposure to contaminated water, as in swimming, is the explanation for increased incidence in the summer months.

Concentrations of enteric pathogens are much greater in sewage in the developing world than in the industrialized world. For example, the average concentration of enteric viruses in sewage in the United States has been estimated to be 10^3 per liter (Table 25.4), whereas concentrations as high as 10^5 per liter have been observed in Africa and Asia.

25.2 CONVENTIONAL WASTEWATER TREATMENT

The primary goal of wastewater treatment is the removal and degradation of organic matter under controlled conditions. Complete sewage treatment comprises three major steps: primary, secondary and tertiary treatment, as shown in Figure 25.3.

25.2.1 Primary Treatment

Primary treatment is the first step in municipal sewage treatment and it involves physically separating large solids from the waste stream. As raw sewage enters the treatment plant, it passes through a metal grating that removes large debris, such as branches and tires (Figure 25.4). A moving screen then filters out smaller items such as diapers and bottles (Figure 25.5), after which a brief residence in a grit tank allows sand and gravel to settle out. The waste stream is then pumped into the primary settling tank (also known as a sedimentation tank or clarifier), where about half the suspended organic solids settle to the bottom as sludge (Figure 25.6). The resulting sludge is referred to as primary sludge. Microbial pathogens are not effectively removed from the effluent in the primary process, although some removal occurs.

Dissolved air flotation (DAF) is a more recent innovation for removing suspended solids from sewage, which is now being introduced into new wastewater treatment plants as an alternative to conventional primary sedimentation processes. DAF clarification is achieved by dissolving air in the wastewater under pressure, and then releasing the air at atmospheric pressure in a flotation tank or basin. This occurs in the front end of the DAF tank known as the "contact zone." The resulting air bubbles that form attach to floc particles and suspended solids. Frequently, coagulants are added to the wastewater prior to the DAF tank to produce the flocs. The floc–bubble aggregates are then carried by water into the second DAF zone known as the "separation zone." Here, free bubbles and floc–bubble aggregates rise to the surface of the tank forming a concentrated sludge blanket that can be removed by skimming devices (Edzwald, 2010). DAF clarifiers remove suspended solids more rapidly than does conventional primary sedimentation and are cost effective from an engineering standpoint.

25.2.2 Secondary Treatment

Secondary treatment consists of biological degradation, in which the remaining suspended solids are decomposed by microorganisms, and the number of pathogens is reduced. In this stage, the effluent from primary treatment usually undergoes biological treatment in a trickling filter bed (Figure 25.7), an aeration tank (Figure 25.8) or a sewage lagoon (see Section 25.3). A disinfection step is generally included at the end of the treatment.

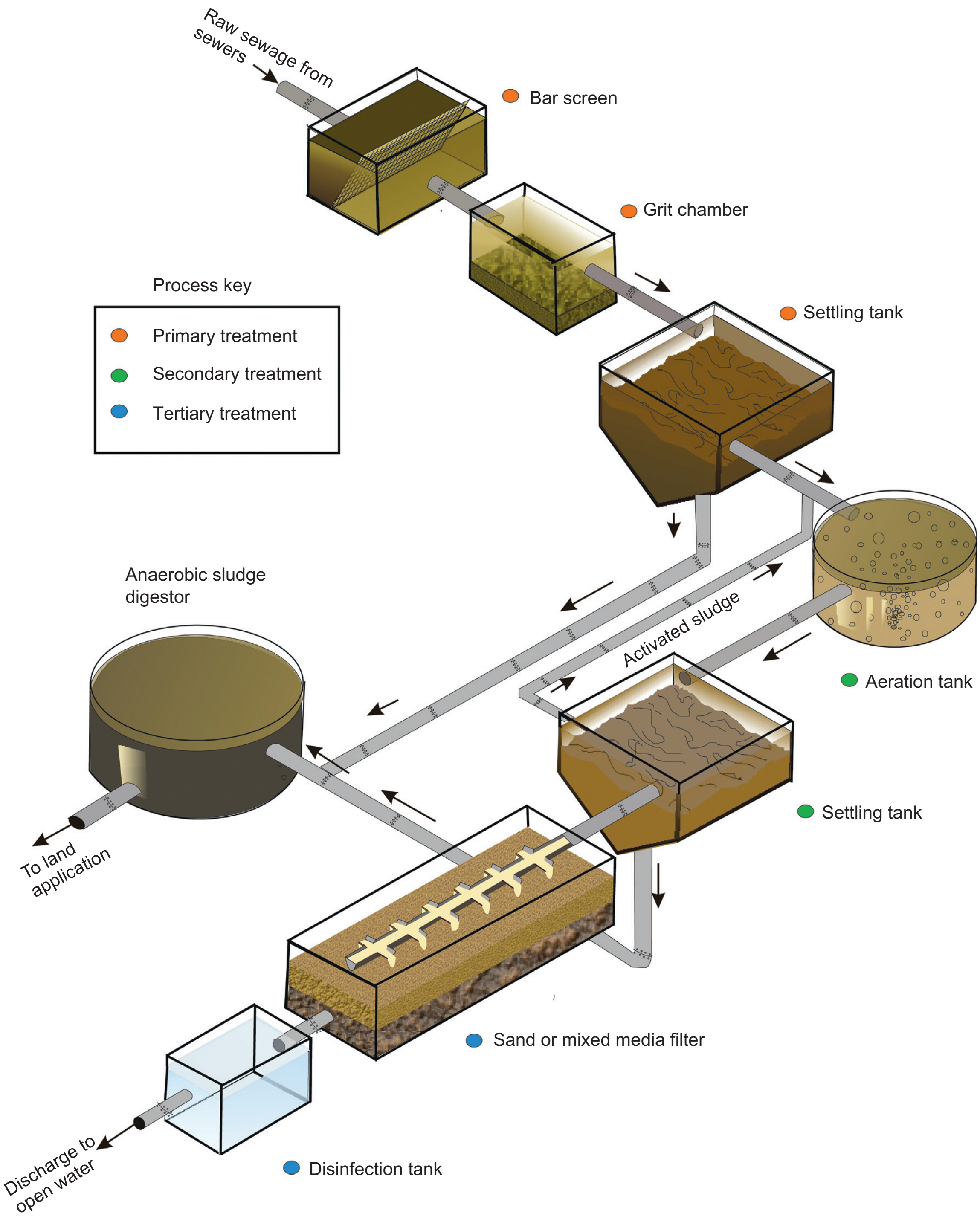

FIGURE 25.3 Schematic of the treatment processes typical of modern wastewater treatment.

25.2.2.1 Trickling Filters

In modern wastewater treatment plants, the trickling filter is composed of plastic units (Figures 25.9 and 25.10). In older plants, or developing countries, the filter is simply a bed of stones or corrugated plastic sheets through which wastewater drips (see Figure 25.7). This is one of the earliest systems introduced for biological waste treatment. The effluent is pumped through an overhead sprayer onto the filter bed, where bacteria and other microorganisms have formed a biofilm on the filter surfaces. These microorganisms intercept the organic material as it trickles past and decompose it aerobically.

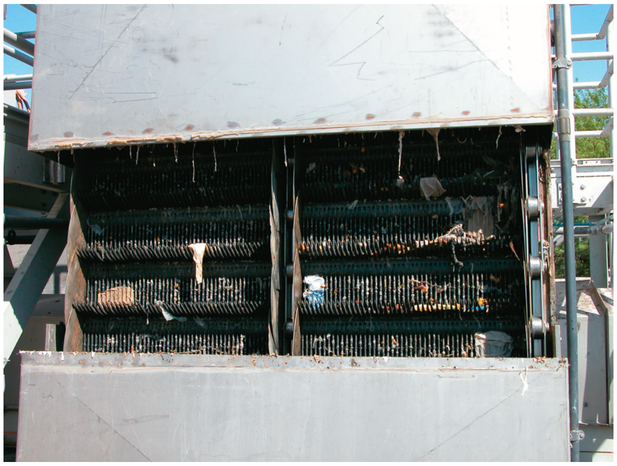

FIGURE 25.4 Removal of large debris from sewage via a "bar screen."

FIGURE 25.5 Removal of small debris via a "moving screen."

FIGURE 25.6 Three clarifiers (foreground—blue) where suspended organic solids settle out as primary sludge. Also see the two anaerobic sludge digestors in the background (white).

FIGURE 25.7 A trickling filter bed. Here, rocks provide a matrix supporting the growth of a microbial biofilm that actively degrades the organic material in the wastewater under aerobic conditions.

FIGURE 25.8 Secondary treatment: an aeration basin.

The media used in trickling filters may be stones, ceramic material, hard coal or plastic media. Plastic media of polyvinyl chloride (PVC) or polypropylene are used today in high-rate trickling filters. As the organic matter passes through the trickling filter, it is converted to microbial biomass, which forms a thick biofilm on the filter medium. The biofilm that forms on the surface of the filter medium is called a zooleal film. It is composed of bacteria, fungi, algae and protozoa. Over time, the increase in biofilm thickness leads to limited oxygen diffusion to the deeper layers of the biofilm, creating an anaerobic environment near the filter medium surface. As a result, the organisms eventually slough from the surface and a new biofilm is formed. BOD removal by trickling filters is approximately 85% for low-rate filters (U.S. EPA, 1977). Effluent from the trickling filter usually passes into a final clarifier to further separate solids from effluent.

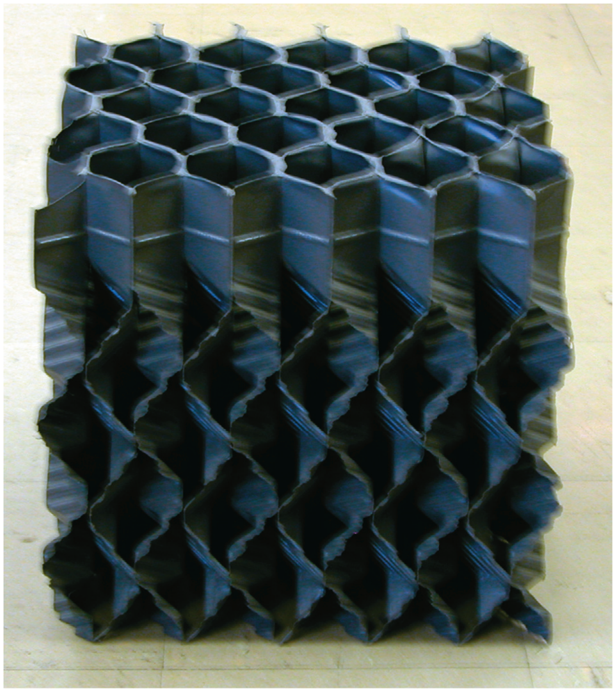

FIGURE 25.9 A unit of plastic material used to create a biofilter (trickling filter). The diameter of each hold is approximately 5 cm. From Pepper *et al*. (2006a).

FIGURE 25.10 A trickling biofilter or biotower. This is composed of many plastic units stacked upon each other. Dimensions of the biofilter may be 20 m diameter by 10−30 m depth. From Pepper *et al.* (2006a).

25.2.2.2 Conventional Activated Sludge

Aeration-tank digestion is also known as the activated sludge process. In the United States, wastewater is most commonly treated by this process. Effluent from primary treatment is pumped into a tank and mixed with a bacteria-rich slurry known as activated sludge. Air or pure oxygen pumped through the mixture encourages bacterial growth and decomposition of the organic material. It then goes to a secondary settling tank, where water is siphoned off the top of the tank and sludge is removed from the bottom. Some of the sludge is used as an inoculum for primary effluent. The remainder of the sludge, known as secondary sludge, is removed. This secondary sludge is added to primary sludge from primary treatment, and is subsequently anaerobically digested to produce biosolids (Chapter 26). The concentration of pathogens is reduced in the activated sludge process by antagonistic microorganisms as well as adsorption to or incorporation in the secondary sludge.

An important characteristic of the activated sludge process is the recycling of a large proportion of the biomass. This results in a large number of microorganisms that oxidize organic matter in a relatively short time (Bitton, 2011). The detention time in the aeration basin varies from 4 to 8 hours. The content of the aeration tank is referred to as the mixed-liquor suspended solids (MLSS). The organic part of the MLSS is called the mixed-liquor volatile suspended solids (MLVSS), which is the nonmicrobial organic matter as well as dead and living microorganisms and cell debris. The activated sludge process must be controlled to maintain a proper ratio of substrate (organic load) to microorganisms or food-to-microorganism ratio (F/M) (Bitton, 2011). This is expressed as BOD per kilogram per day. It is expressed as:

$$\frac{\mathrm{F}}{\mathrm{M}} = \frac{Q \cdot \mathrm{BOD}_5}{\mathrm{MLSS} \cdot V} \qquad \text{(Eq. 25.3)}$$

where:

Q = flow rate of sewage in million gallons per day (MGD)
BOD_5 = 5-day biochemical oxygen demand (mg/L)
MLSS = mixed-liquor suspended solids (mg/L)
V = volume of aeration tank (gallons)

F/M is controlled by the rate of activated sludge wasting. The higher the wasting rate, the higher the F/M ratio. For conventional aeration tanks the F/M ratio is 0.2−0.5 lb BOD_5/day/lb MLSS, but it can be higher (up to 1.5) for activated sludge when high-purity oxygen is used. A low F/M ratio means that the microorganisms in the aeration tank are starved, leading to more efficient wastewater treatment.

The important parameters controlling the operation of an activated sludge process are: organic loading rates; oxygen supply; and control and operation of the final settling tank. This tank has two functions: clarification and thickening. For routine operation, sludge settleability is determined by use of the sludge volume index (SVI) (Bitton, 2011).

SVI is determined by measuring the sludge volume index, which is given by the following formula:

$$\mathrm{SVI} = \frac{V \cdot 1000}{\mathrm{MLSS}} \qquad \text{(Eq. 25.4)}$$

where V = volume of settled sludge after 30 minutes (ml/L).

The microbial biomass produced in the aeration tank must settle properly from suspension so that it may be wasted or returned to the aeration tank. Good settling occurs when the sludge microorganisms are in the endogenous phase, which occurs when carbon and energy sources are limited, and the microbial specific growth rate is local (Bitton, 2011). A mean cell residence time of 3–4 days is necessary for effective settling (Metcalf and Eddy, 2003). Poor settling may also be caused by sudden changes in temperature and pH, absence of nutrients and presence of toxic metals and organics. A common problem in the activated sludge process is filamentous bulking, which consists of slow settling and poor compaction of solids in the clarifier. Filamentous bulking is usually caused by the excessive growth of filamentous microorganisms. The filaments produced by these bacteria interfere with sludge settling and compaction. A high SVI (>150 ml/g) indicates bulking conditions. Filamentous bacteria are able to predominate under conditions of low dissolved oxygen, low F/M, low nutrient and high sulfide levels. Filamentous bacteria can be controlled by treating the return sludge with chlorine or hydrogen peroxide to kill filamentous microorganisms selectively.

25.2.2.3 Nitrogen Removal by the Activated Sludge Process

Activated sludge processes can be modified for nitrogen removal to encourage nitrification followed by denitrification. The establishment of a nitrifying population in activated sludge depends on the wastage rate of the sludge, and therefore on the BOD load, MLSS and retention time. The growth rate of nitrifying bacteria (μ_n) must be higher than the growth rate (μ_h) of heterotrophs in the system. In reality, the growth rate of nitrifiers is lower than that of heterotrophs in sewage; therefore, a long sludge age is necessary for the conversion of ammonia to nitrate. Nitrification is expected at a sludge age greater than 4 days (Bitton, 2011).

Nitrification must be followed by denitrification to remove nitrogen from wastewater. The conventional activated sludge system can be modified to encourage denitrification. Two such processes are:

- Single sludge system (Figure 25.11A). This system comprises a series of aerobic and anaerobic tanks in lieu of a single aeration tank.
- Multisludge system (Figure 25.11B). Carbonaceous oxidation, nitrification and denitrification are carried out in three separate systems. Added methanol or settled sewage serves as the source of carbon for denitrifiers.

25.2.2.4 Phosphorus Removal by the Activated Sludge Process

Phosphorus can also be reduced by the activity of microorganisms in modified activated sludge processes. The process depends on the uptake of phosphorus by the microbes during the aerobic stage and subsequent release during the anaerobic stage. One of several systems in use is the A/O (anaerobic/oxic) process. The A/O process consists of a modified activated sludge system that includes an anaerobic zone (detention time 0.5–1 hour) upstream of the conventional aeration tank (detention

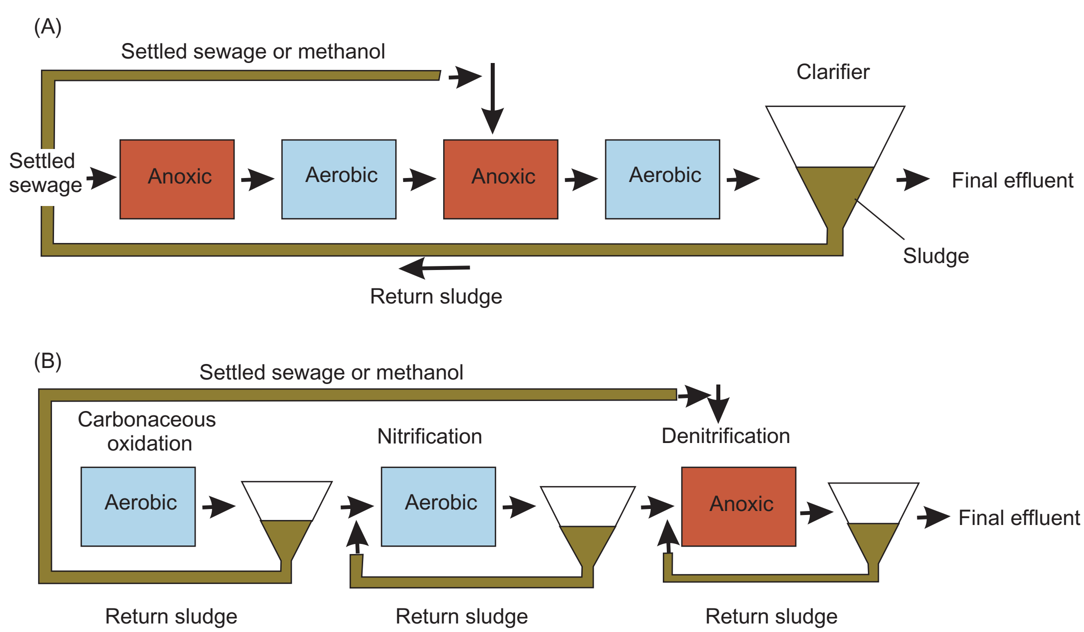

FIGURE 25.11 Denitrification systems: (A) single-sludge system; (B) multisludge system. Modified from Curds and Hawkes (1983).

time 1–3 hours). Figure 25.12 illustrates the microbiology of the A/O process. During the anaerobic phase, inorganic phosphorus is released from the cells as a result of polyphosphate hydrolysis. The energy liberated is used for the uptake of BOD from wastewater. Removal efficiency is high when the BOD/phosphorus ratio exceeds 10 (Metcalf and Eddy, 2003). During the aerobic phase, soluble phosphorus is taken up by bacteria, which synthesize polyphosphates, using the energy released from BOD oxidation.

25.2.2.5 The Bardenpho Process

The Bardenpho process is an advanced modification of the activated sludge process, which results in nutrient removal of nitrogen and phosphorus via microbial processes that occur in a multistage biological reactor (Figure 25.13). This reactor removes high levels of BOD, suspended solids, nitrogen and phosphorus.

- Fermentation stage: Activated sludge is returned from the clarifier and undergoes microbial fermentation and phosphate is released.
- First anoxic stage: Mixed liquor containing nitrates from the third stage is recycled here, and mixed with conditioned sludge from the fermentation stage, in the absence of oxygen. Heterotrophic denitrifying bacteria reduce BOD by utilizing carbonaceous substrate while using nitrate as a terminal electron acceptor, which is reduced to gaseous nitrogen.
- Nitrification stage: Oxygen is introduced allowing for heterotrophic aerobic respiration, which further oxidizes BOD. At the same time ammonia is aerobically nitrified to nitrate, and phosphate is taken up and utilized by microbes. Mixed liquor containing the nitrates is recycled back to the first anoxic stage.
- Second anoxic stage: The remaining liquor from the nitrification stage is passed into this second anoxic stage, where nitrate (in the absence of oxygen) is again reduced to nitrogen gas. This results in low effluent nitrate concentrations.
- Re-aeration stage: This is an aerobic environment that ensures that phosphate taken up microbially is not released in the final clarifier.

Overall the Bardenpho process results in an effluent that is low in nitrates and phosphates. Bardenpho pro-

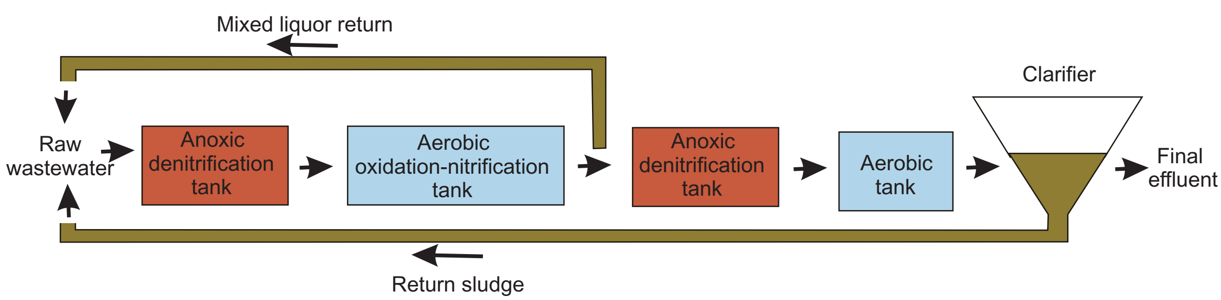

FIGURE 25.12 Microbiology of the A/O process.

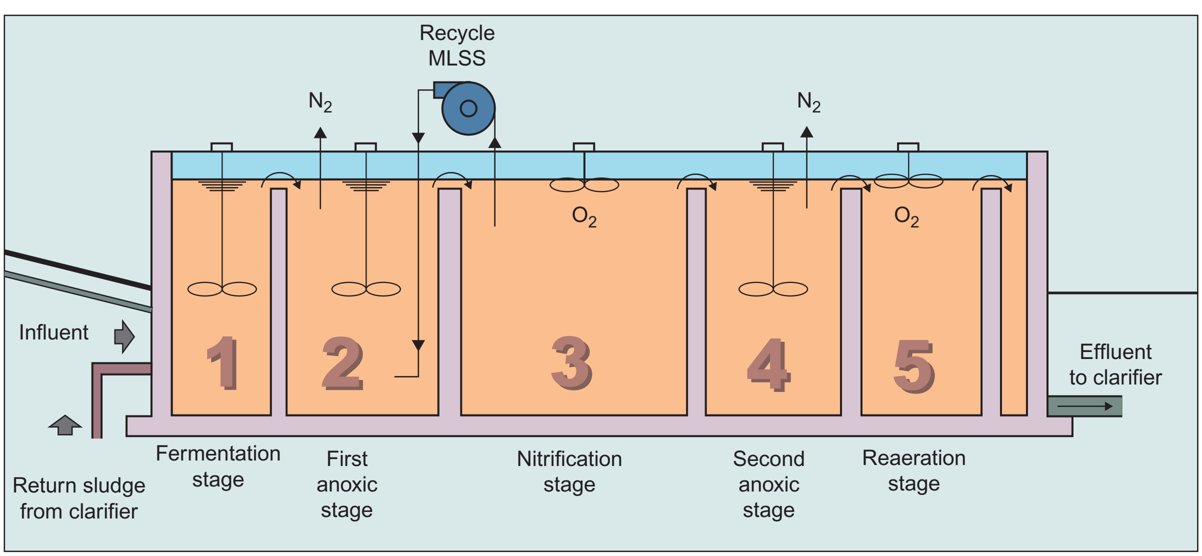

FIGURE 25.13 The five-stage Bardenpho process for microbial nutrient removal.

cesses utilizing autotrophic denitrification have also been evaluated utilizing "spent caustic" as an electron donor. Spent caustic is produced in petrochemical processes and contains adsorbed hydrogen sulfide which acts as substrate for autotrophic denitrifiers (Park *et al*., 2010).

25.2.2.6 Membrane Bioreactors

Membrane bioreactors (MBR) are a combination of biological treatment with membrane separation by microporous or ultrafiltration membranes. The process consists of a tank and a membrane unit either located external to the bioreactor or submerged directly within it (Figure 25.14). The membranes act to retain suspended solids and maintain a high biomass concentration within the bioreactor thereby functioning as a replacement for sedimentation. Membranes come with various pore sizes and can be dense or porous. Separation by dense membranes relies on physicochemical interactions between the permeating components and the membrane material and is known as reverse osmosis or nanofiltration. Porous membranes have larger pore size, separate particles mechanically and are referred to as ultrafiltration or microfiltration. The microbial bioreactor is normally maintained aerobically, but can be operated anaerobically, or with alternating aerobic/anaerobic phases to enhance microbial nitrification followed by denitrification.

Membrane bioreactors have several advantages including a much smaller area needed than conventional activated sludge and a high quality effluent. A membrane bioreactor effectively displaces three individual process steps in a conventional treatment plant (primary settling, activated sludge and reduces the need for disinfection). The major advantages of MBRs include: good quality effluent, reduced reactor volume and net sludge production. The major disadvantages include high operating costs and membrane fouling (Chang *et al*., 2002). Operating costs can be reduced by integrating microbial fuel cells and membrane bioreactors (Wang *et al*., 2012).

25.2.3 Tertiary Treatment

Tertiary treatment of effluent involves a series of additional steps after secondary treatment to further reduce organics, turbidity, nitrogen, phosphorus, metals and pathogens. Most processes involve some type of physicochemical treatment such as coagulation, filtration, activated carbon adsorption of organics, reverse osmosis and additional disinfection. Tertiary treatment of wastewater is practiced for additional protection of wildlife after discharge into rivers or lakes. Even more commonly, it is performed when the wastewater is to be reused for irrigation (e.g., food crops, golf courses), for recreational purposes (e.g., lakes, estuaries) or for drinking water.

25.2.4 Removal of Pathogens by Sewage Treatment Processes

There have been a number of reviews on the removal of pathogenic microorganisms by activated sludge and other wastewater treatment processes (Leong, 1983). This information suggests that significant removal especially of enteric bacterial pathogens can be achieved by these processes (Table 25.5). However, disinfection and/or advanced tertiary treatment are necessary for many reuse applications to ensure pathogen reduction. Current issues related to pathogen reduction are: treatment plant reliability; removal of new and emerging enteric pathogens of concern; and the ability of new technologies to effect pathogen reduction. Wide variation in pathogen removal can result in significant numbers of pathogens passing through a process for various time periods. The issue of reliability is of major importance if the reclaimed water is intended for recreational or potable reuse, where short-term exposures to high levels of pathogens could result in significant risk to the exposed population.

Compared with other biological treatment methods (i.e., trickling filters), activated sludge is relatively efficient in reducing the numbers of pathogens in raw wastewater. Both sedimentation and aeration play a role in pathogen reduction. Primary sedimentation is more effective for the removal of the larger pathogens such as helminth eggs, but solid-associated bacteria and even viruses are also removed. During aeration, pathogens are inactivated by antagonistic microorganisms and by environmental factors such as temperature. The greatest removal probably occurs by adsorption or entrapment of the

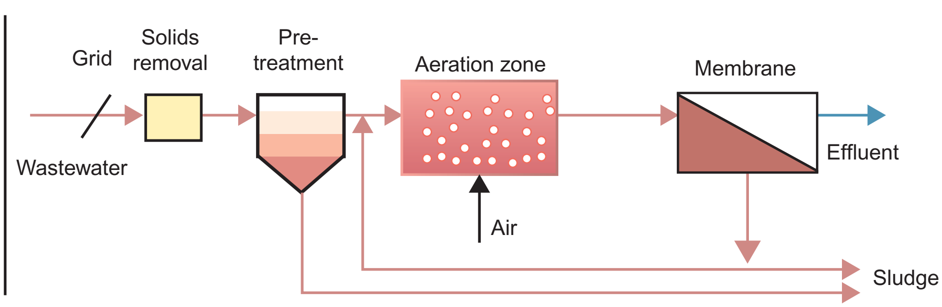

FIGURE 25.14 Membrane bioreactor treatment train showing an externalmembrane unit.

TABLE 25.5 Pathogen Removal during Sewage Treatment

	Enteric Viruses	*Salmonella*	*Giardia*	*Cryptosporidium*
Concentration in raw sewage (per liter)	10^5−10^6	5000−80,000	9000−200,000	1−3960
Primary treatment[a] % removal	50−98.3	95.8−99.8	27−64	0.7
Number remaining (per liter)	1700−500,000	160−3360	72,000−146,000	
Secondary treatment[b] % removal	53−99.92	98.65−99.996	45−96.7	
Number remaining (per liter)	80−470,000	3−1075	6480−109,500	
Secondary treatment[c] % removal	99.983−99.9999998	99.99−99.999999995	98.5-99.99995	2.7[d]
Number remaining (per liter)	0.007−170	0.000004−7	0.099−2951	

[a]*Primary sedimentation and disinfection.*
[b]*Primary sedimentation, trickling filter or activated sludge, and disinfection.*
[c]*Primary sedimentation, trickling filter or activated sludge, disinfection, coagulation, filtration, and disinfection.*
[d]*Filtration only.*

TABLE 25.6 Average Removal of Pathogen and Indicator Microorganisms in a Wastewater Treatment Plant, St. Petersburg, Florida

	Raw Wastewater to Secondary Wastewater		Secondary Wastewater to Postfiltration		Postfiltration to Postdisinfection		Postdisinfection to Poststorage		Raw Wastewater to Poststorage	
	Percentage	$\log_{10}$	Percentage	$\log_{10}$	Percentage	$\log_{10}$	Percentage	$\log_{10}$	Percentage	$\log_{10}$
Total coliforms	98.3	1.75	69.3	0.51	99.99	4.23	75.4	0.61	99.999992	7.1
Fecal coliforms	99.1	2.06	10.5	0.05	99.998	4.95	56.8	0.36	99.999996	7.4
Coliphage[a]	82.1	0.75	99.98	3.81	90.05	1.03	90.3	1.03	99.999997	6.6
Enterovirus	98.0	1.71	84.0	0.81	96.5	1.45	90.9	1.04	99.999	5.0
Giardia	93.0	1.19	99.0	2.00	78.0	0.65	49.5	0.30	99.993	4.1
Cryptosporidium	92.8	1.14	97.9	1.68	61.1	0.41	8.5	0.04	99.95	3.2

[a]*Escherichia coli* host *ATCC 15597.*

organisms within the biological floc that forms. The ability of activated sludge to remove viruses is related to the ability to remove solids. This is because viruses tend to be solid associated, and are removal along with the floc. Activated sludge typically removes 90% of the enteric bacteria and 80−99% of the enteroviruses and rotaviruses (Rao *et al*., 1986). Ninety percent of *Giardia* and *Cryptosporidium* can also be removed (Rose and Carnahan, 1992), being largely concentrated in the sludge. Because of their large size, helminth eggs are effectively removed by sedimentation, and are rarely found in sewage effluent in the United States, although they may be detected in the sludge. However, although the removal of the enteric pathogens may seem large, it is important to remember that initial concentrations are also large (i.e., the concentration of all enteric viruses in 1 liter of raw sewage may be as high as 100,000 in some parts of the world).

Tertiary treatment processes involving physicochemical processes can be effective in further reducing the concentration of pathogens and enhancing the effectiveness of disinfection processes by the removal of soluble and particulate organic matter (Table 25.6). Filtration is probably the most common tertiary treatment process. Mixed-media (sand, gravel, coal) filtration is most effective in the reduction of protozoan parasites. Usually, greater removal of *Giardia* cysts occurs than of *Cryptosporidium* oocysts because of the larger size of the cysts (Rose and Carnahan, 1992). Removal of enteroviruses and indicator bacteria is usually 90% or less. Addition of coagulant can increase the removal of poliovirus to 99% (U.S. EPA, 1992a).

Coagulation, particularly with lime, can result in significant reductions of pathogens. The high pH conditions (pH 11−12) that can be achieved with lime can result in significant inactivation of enteric viruses. To achieve removals of 90% or greater, the pH should be maintained above 11 for at least an hour (Leong, 1983). Inactivation of the viruses occurs by denaturation of the viral protein coat. The use of iron and aluminum salts for coagulation can also result in 90% or greater reductions in enteric viruses. The degree of effectiveness of these processes, as in other solids separating processes, is highly dependent on the hydraulic design and, in particular, coagulation and flocculation. The degree of removal observed in bench-scale tests may not approach those seen in full-scale plants, where the process is more dynamic.

Reverse osmosis and ultrafiltration are also believed to result in significant reductions in enteric pathogens. Removal occurs by size exclusion. Removal of enteric viruses in excess of 99.9% can be achieved (Leong, 1983).

25.2.5 Removal of Organics and Inorganics by Sewage Treatment Processes

In addition to nutrients such as nitrogen and phosphorus, and microbial pathogens, there are other constituents within sewage that need to be kept at low concentrations. These include inorganics exemplified by metals, and organic priority pollutants. Metals and organics are normally associated with the solid fraction of sewage, and neither is significantly removed by sewage treatment. However, when point source control mechanisms are implemented to prevent industrial discharges, the concentration of metals and organics within sewage can be significantly reduced. In particular, over the past 15 years in the United States this has resulted in decreased metal concentrations. More recently, there has been concern over the presence of pharmaceuticals such as endocrine disruptors in sewage.

25.3 OXIDATION PONDS

The next two sections discuss several alternatives to large-scale modern wastewater treatment process discussed in Section 25.2. The first of these are sewage lagoons and are often referred to as oxidation or stabilization ponds. These are the oldest of the wastewater treatment systems. Usually no more than a hectare in area and just a few meters deep, oxidation ponds are natural "stewpots," where wastewater is detained while organic matter is degraded (Figure 25.15). A period of time ranging from 1 to 4 weeks (and sometimes longer) is necessary to complete the decomposition of organic matter. Light, heat and settling of the solids can also effectively reduce the number of pathogens present in the wastewater.

FIGURE 25.15 An oxidation pond. Typically these are only 1−2 meters deep, and small in area.

The following four categories of oxidation ponds are often used in series:

- Aerobic ponds (Figure 25.16A), which are naturally mixed, must be shallow (up to 1.5 m) because they depend on penetration of light to stimulate algal growth that promotes subsequent oxygen generation. The detention time of wastewater is generally 3 to 5 days.
- Anaerobic ponds (Figure 25.16B) may be 1 to 10 m deep, and require a relatively long detention time of 20 to 50 days. These ponds, which do not require expensive mechanical aeration, generate small amounts of sludge. Often, anaerobic ponds serve as a pretreatment step for high-BOD organic wastes rich in protein and fat (e.g., meat wastes) with a heavy concentration of suspended solids.
- Facultative ponds (Figure 25.17) are most common for domestic waste treatment. Waste treatment is provided by both aerobic and anaerobic processes. These ponds range in depth from 1 to 2.5 m and are subdivided into three layers: an upper aerated zone; a middle facultative zone; and a lower anaerobic zone. The detention time varies between 5 and 30 days.
- Aerated lagoons or ponds (Figure 25.18), which are mechanically aerated, may be 1−2 m deep and have a detention time of less than 10 days. In general, treatment depends on the aeration time and temperature, as well as the type of wastewater. For example, at 20°C an aeration period of 5 days results in 85% BOD removal.

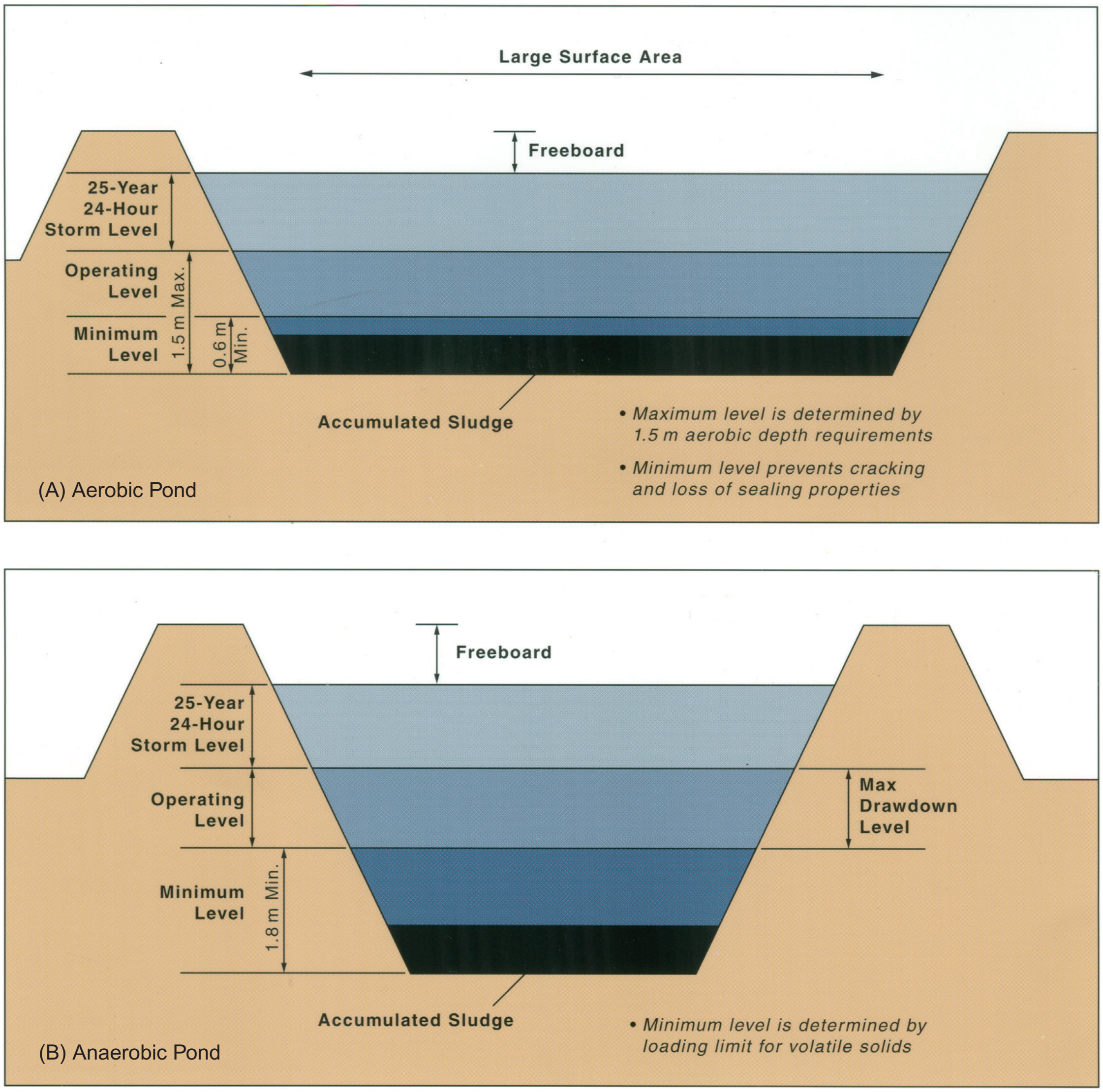

FIGURE 25.16 Pond profiles: (A) aerobic waste pond profile, and (B) anaerobic waste pond profile.

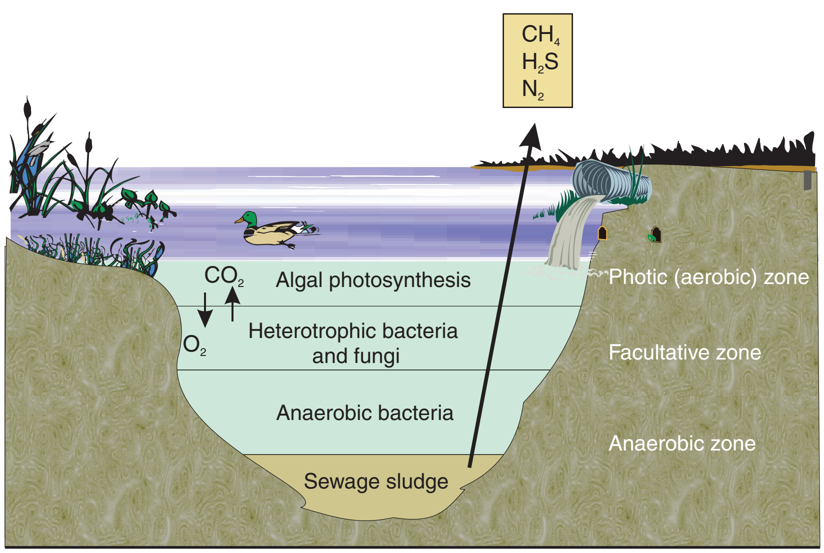

FIGURE 25.17 Microbiology of facultative ponds. Modified from Bitton (1980).

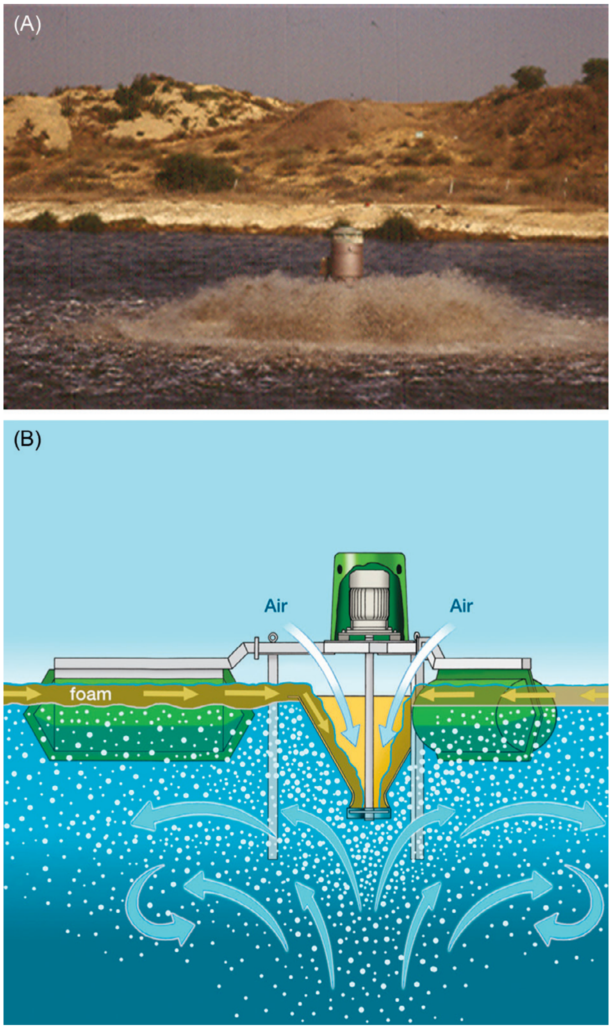

FIGURE 25.18 (A) Aerated lagoon, and (B) floating aeration device.

Because sewage lagoons require a minimum of technology and are relatively low in cost, they are most common in developing countries. However, biodegradable organic matter and turbidity are not as effectively reduced as during activated sludge treatment.

Given sufficient retention times, oxidation ponds can cause significant reductions in the concentrations of enteric pathogens, especially helminth eggs. For this reason, they have been promoted widely in the developing world as a low-cost method of pathogen reduction for wastewater reuse for irrigation. However, a major drawback of ponds is the potential for short-circuiting because of thermal gradients even in multi-pond systems designed for long retention times (i.e., 90 days). Even though the amount of short-circuiting may be small, detectable levels of pathogens can often be found in the effluent from oxidation ponds.

Inactivation and/or removal of pathogens in oxidation ponds is controlled by a number of factors including: temperature; sunlight; pH; bacteriophages; predation by other microorganisms; and adsorption to or entrapment by settleable solids. Indicator bacteria and pathogenic bacteria may be reduced by 90–99% or more, depending on retention times.

25.4 SEPTIC TANKS

Until the middle of the twentieth century in the United States, many rural families and quite a few residents of towns and small cities depended on pit toilets or "outhouses" for waste disposal. In rural areas of developing countries these are still used. These pit toilets, however, often allowed untreated wastes to seep into the groundwater, allowing pathogens to contaminate drinking water supplies. This risk to public health led to the development of septic tanks and properly constructed drain fields. Primarily, septic tanks serve as repositories where solids are separated from incoming wastewater, and biological digestion of the waste organic matter can take place under anaerobic conditions. In 2007, 20% (26.1 million) of the homes in the United States depended on septic tanks. Approximately 20% of all new homes constructed use septic tanks. Most septic tanks are located in the eastern United States (Figure 25.19). In a typical septic tank system (Figure 25.20), the wastewater and sewage enter a tank made of concrete, metal or fiberglass. There, grease and oils rise to the top as scum, and solids settle to the bottom. The wastewater and sewage then undergo anaerobic bacterial decomposition, resulting in the production of a sludge. The wastewater usually remains in the septic tank for just 24–72 hours, after which it is channeled out to a drain field. This drain field or leach field is composed of small perforated pipes that are embedded in gravel below the surface of the soil. Periodically, the residual sludge in the septic tank known as septage is pumped out into a tank truck, and taken to a treatment plant for disposal.

Although the concentration of contaminants in septic tank separate is typically much greater than that found in domestic wastewater (Table 25.7), septic tanks can be an effective method of waste disposal where land is available and population densities are not too high. Thus, they are widely used in rural and suburban areas. But as suburban population densities increase, groundwater and surface water pollution may arise, indicating a need to shift to a commercial municipal sewage system. (In fact, private septic systems are sometimes banned in many suburban areas.) Moreover, septic tanks are not appropriate for every area of the country. They do not work well, for example, in cold, rainy climates, where the drain field may be too wet for proper evaporation, or in areas where the water table is shallow. High densities of septic tanks can also be responsible for nitrate contamination of

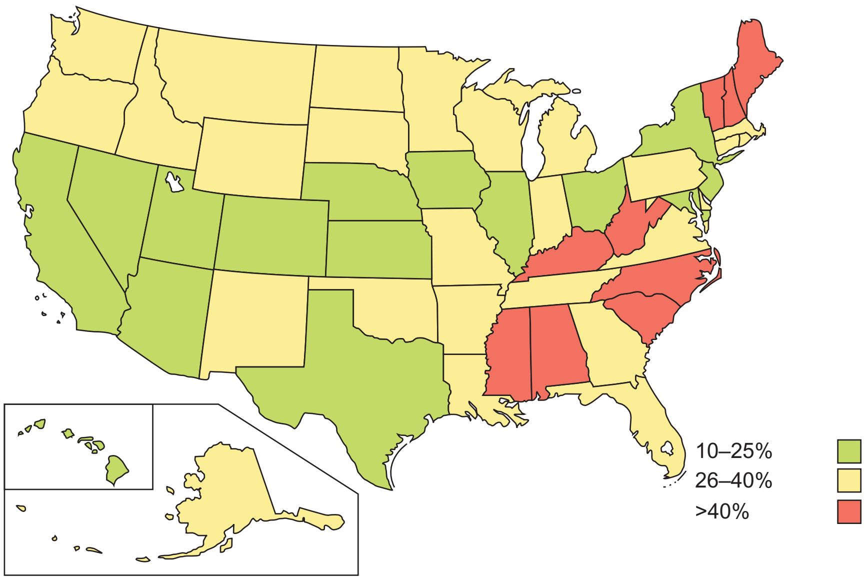

FIGURE 25.19 Percentage of U.S. residents utilizing septic tanks for onsite wastewater treatment. *Source:* U.S. Census Bureau, 1990.

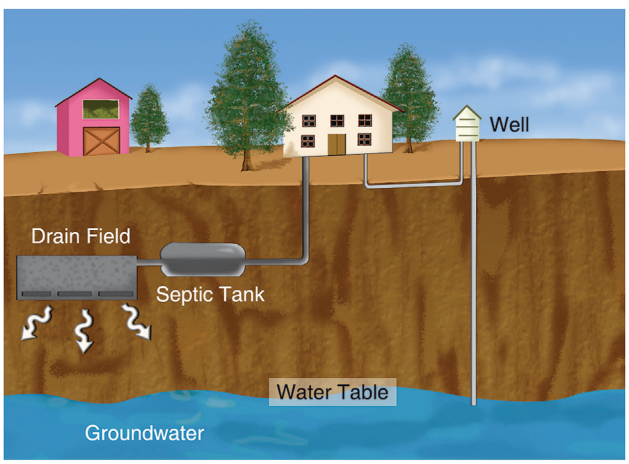

FIGURE 25.20 Septic tank (on-site treatment system). *Source:* Pepper *et al.* (2006).

groundwater. Finally, most of the waterborne disease outbreaks associated with groundwater in the United States are thought to result from contamination by septic tanks.

TABLE 25.7 Typical Characteristics of Septage

Constituent	Concentration (mg/L)	
	Range	Typical Value
Total solids	5000–100,000	40,000
Suspended solids	4000–100,000	15,000
Volatile suspended solids	1200–14,000	7000
BOD_5, 20°C	2000–30,000	6000
Chemical oxygen demand	5000–80,000	30,000
Total Kjeldahl nitrogen (as N)	100–1600	700
Ammonia, NH_3 (as N)	100–800	400
Total phosphorus (as P)	50–800	250
Heavy metals[a]	100–1000	300

From Pepper et al. (2006b).
[a]*Primarily iron (Fe), zinc (Zn), and aluminum (Al).*

25.5 LAND APPLICATION OF WASTEWATER

Although treated domestic wastewater is usually discharged into bodies of water, it may also be disposed of via land application for crop irrigation, or as a means of additional treatment and disposal. The three basic methods used in the application of sewage effluents to land include: low-rate irrigation; overland flow; and high-rate infiltration. Characteristics of each of these are listed in Table 25.8. The choice of a given method depends on the conditions prevailing at the site under consideration (loading rates, methods of irrigation, crops and expected treatment).

With low-rate irrigation (Figure 25.21A), sewage effluents are applied by sprinkling or by surface application at a rate of 1.5 to 10 cm per week. Two-thirds of the water is taken up by crops or lost by evaporation, and the remainder percolates through the soil matrix. The system must be designed to maximize denitrification in order to avoid pollution of groundwater by nitrates. Phosphorus is immobilized within the soil matrix by fixation or

TABLE 25.8 General Characteristics of the Three Methods Used for Land Application of Sewage Effluent

Factor	Application Method		
	Low-Rate Irrigation	Overland Flow	High-Rate Infiltration
Main objectives	Reuse of nutrients and water, wastewater treatment	Wastewater treatment	Wastewater treatment, groundwater recharge
Soil permeability	Moderate (sandy to clay soils)	Slow (clay soils)	Rapid (sandy soils)
Need for vegetation	Required	Required	Optional
Loading rate	1.5−10 cm/week	5−14 cm/week	>50 cm/week
Application technique	Spray, surface	Usually spray	Surface flooding
Land required for flow of 10^6 liters/day	8−66 hectares	5−16 hectares	0.25−7 hectares
Needed depth to groundwater	About 2 cm	Undetermined	5m or more
BOD and suspended solid removal	90−99%	90−99%	90−99%
N removal	85−90%	70−90%	0−80%
P removal	80−90%	50−60%	75−90%

From Pepper et al. (2006b).

precipitation. The irrigation method is used primarily by small communities and requires large areas, generally on the order of 5−6 hectares per 1000 people.

In the overland flow method (Figure 25.21B), wastewater effluents are allowed to flow for a distance of 50−100 m along a 2−8% vegetated slope and are collected in a ditch. The loading rate of wastewater ranges from 5 to 14 cm a week. Only about 10% of the water percolates through the soil, compared with 60% that runs off into the ditch. The remainder is lost as evapotranspiration. This system requires clay soils with low permeability and infiltration.

High-rate infiltration treatment is also known as soil aquifer treatment (SAT) or rapid infiltration extraction (RIX) (Figure 25.21C). The primary objective of SAT is the treatment of wastewater at loading rates exceeding 50 cm per week. The treated water, most of which has percolated through coarse-textured soil, is used for groundwater recharge, or may be recovered for irrigation. This system requires less land than irrigation or overland flow methods. Drying periods are often necessary to aerate the soil system and avoid problems due to clogging. The selection of a site for land application is based on many factors including: soil types; drainable and depth; distance to groundwater; groundwater movement; slope; underground formations; and degree of isolation of the site from the public.

Inherent in land application of wastewater are the risks of transmission of enteric waterborne pathogens. The degree of risk is associated with the concentration of pathogens in the wastewater and the degree of contact with humans. Land application of wastewater is usually considered an intentional form of reuse, and is regulated by most states. Because of limited water resources in the western United States, reuse is considered essential. Usually, stricter treatment and microbial standards must be met before land application. The highest degree of treatment is required when wastewater will be used for food crop irrigation, with lesser treatment for landscape irrigation or fiber crops. For example, the State of California requires no disinfection of wastewater for irrigation and no limits on coliform bacteria. However, if the reclaimed wastewater is used for surface irrigation of food crops and open landscaped areas, chemical coagulation (to precipitate suspended matter), followed by filtration and disinfection to reduce the coliform concentration to 2.2/100 ml, is required. In some cities excess effluent is disposed of in river beds that are normally dry. Such disposal can create riparian areas (Figure 25.22).

Because high-rate infiltration may be practiced to recharge aquifers, additional treatments of secondary wastewater may be required. However, as some removal of pathogens can be expected, the treatment requirement may be less. The degree of treatment needed may be influenced by the amount or time it takes the reclaimed water to travel from the infiltration site to the point of extraction, and the depth of the unsaturated zone. The greatest concern has been with the transport of viruses, which, because of their small size, have the greatest chance of traveling large distances within the subsurface.

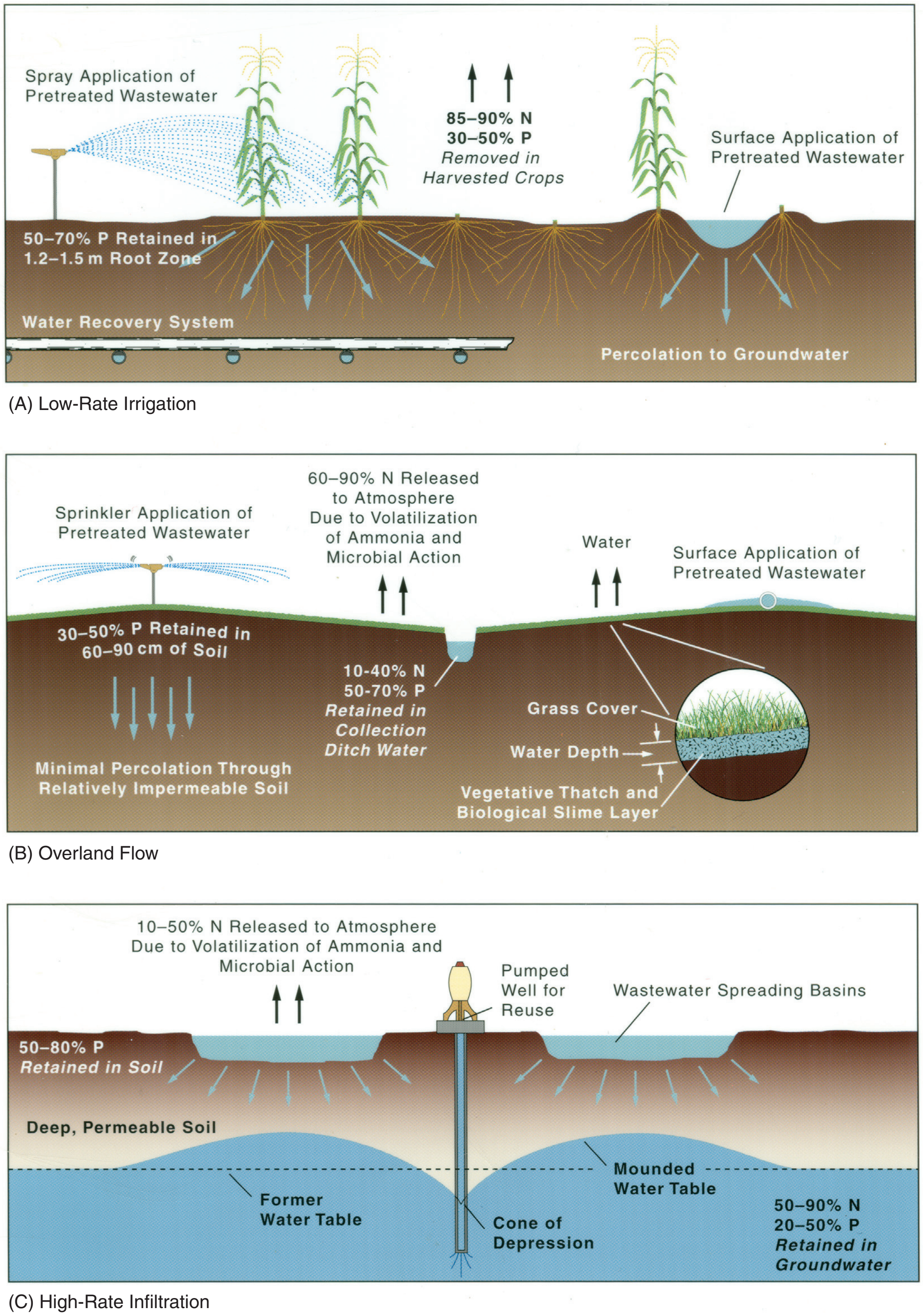

FIGURE 25.21 Three basic methods of land application of wastewater.

Factors that influence the transport of viruses are discussed in Chapter 15. Generally, several meters of moderately fine-textured, continuous soil layer are necessary for virus reductions of 99.9% or more (Yates, 1994).

25.6 WETLANDS SYSTEMS

Wetlands, which are typically less than 1 m in depth, are areas that support aquatic vegetation and foster the growth of emergent plants such as cattails, bulrushes, reeds, sedges and trees. They also provide important wetland habitat for many animal species. Recently, wetland areas have been receiving increasing attention as a means of additional treatment for secondary effluents. The vegetation provides surfaces for the attachment of bacteria, and aids in the filtration and removal of such wastewater contaminants as biological oxygen and excess carbon. Factors involved in the reduction of wastewater contaminants are shown in Table 25.9. Although both natural and constructed wetlands have been used for wastewater treatment, recent work has focused on constructed wetlands because of regulatory requirements. Two types of constructed wetland systems are in general use: (1) free water surface (FWS) systems; and (2) subsurface flow systems (SFS). An FWS wetland is similar to a natural marsh because the water surface is exposed to the atmosphere. Floating and submerged plants, such as those shown in Figure 25.23A, may be present. SFS consist of channels or trenches with relatively impermeable bottoms filled with sand or rock media to support emergent vegetation.

FIGURE 25.22 Effluent outfall of the Roger Road Wastewater Treatment Plant in Tucson, Arizona. Here, extensive growth of vegetation due to the effluent produces a riparian habitat.

During wetland treatment, the wastewater is usable. It can, for instance, be used to grow aquatic plants such as water hyacinths (Figure 25.23B) and/or to raise fish for human consumption. The growth of such aquatic plants provides not only additional treatment for the water but also a food source for fish and other animals. Such aquaculture systems, however, tend to require a great deal of land area. Moreover, the health risk associated with the production of aquatic animals for human consumption in this manner must be better defined.

There has been increasing interest in the use of natural systems for the treatment of municipal wastewater as a form of tertiary treatment (Kadlec and Wallace, 2008). Artificial or constructed wetlands have a higher degree of biological activity than most ecosystems; thus transformation of pollutants into harmless by-products or

TABLE 25.9 Principal Removal and Transformation Mechanisms in Constructed Wetlands Involved in Contaminant Reduction

Constituent	Free Water System	Subsurface Flow	Floating Aquatics
Biodegradable organics	Bioconversion by aerobic, facultative, and anaerobic bacteria on plant and debris surfaces of soluble BOD, adsorption, filtration	Bioconversion by facultative and anaerobic bacteria on plant and debris surfaces	Bioconversion by aerobic, facultative, and anaerobic bacteria on plant and debris surfaces
Suspended solids	Sedimentation, filtration	Filtration, sedimentation	Sedimentation, filtration
Nitrogen	Nitrification/denitrification, plant uptake, volatilization	Nitrification/denitrification, plant uptake, volatilization	Nitrification/denitrification, plant uptake, volatilization
Phosphorus	Sedimentation, plant uptake	Filtration, sedimentation, plant uptake	Sedimentation, plant uptake
Heavy metals	Adsorption to plant and debris surfaces	Adsorption to plant roots and debris surfaces, sedimentation	Absorption by plants, sedimentation
Trace organics	Volatilization, adsorption, biodegradation	Adsorption, biodegradation	Volatilization, adsorption biodegradation
Pathogens	Natural decay, predation, UV irradiation, sedimentation, excretion of antimicrobials from roots of plants	Natural decay, predation, sedimentation, excretion of antimicrobials from roots of plants	Natural decay, predation, sedimentation

From Pepper et al. (2006b).

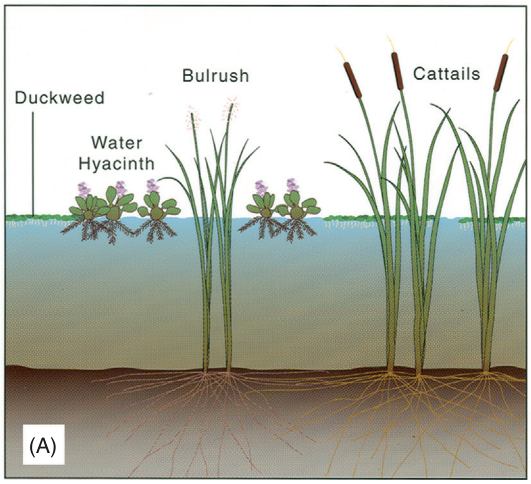

FIGURE 25.23 (A) Common aquatic plants used in constructed wetlands. (B) An artificial wetland system in San Diego, California, utilizing water hyacinths.

essential nutrients for plant growth can take place at a rate that is useful for the treatment of municipal wastewater (Case Study 25.1). Most artificial wetlands in the United States use reeds or bull rushes, although floating aquatic plants such as water hyacinths and duckweed have also been used. To reduce potential problems with flying insects, subsurface flow wetlands have also been built (Figure 25.25). In these types of wetlands all of the flow of the wastewater is below the surface of a gravel bed containing plants tolerant of water-saturated soils. Most of the existing information on the performance of these wetlands concerns coliform and fecal coliform bacteria. Kadlec and Wallace (2008) have summarized the existing literature on this topic. They point out that natural sources of indicators in treatment wetlands never reach zero because wetlands are open to wildlife. Reductions in fecal coliforms are generally greater than 99%, but there is a great deal of variation, probably depending on the season, type of wetland, numbers and type of wildlife and retention time in the wetland. Volume-based and area-based bacterial die-off models have been used to estimate bacterial die-off in surface flow wetlands (Kadlec and Wallace, 2008).

In one study of a mixed-species surface flow wetland with a detention time of approximately 4 days, several other types of microorganisms were examined. Results showed that *Cryptosporidium* was reduced by 53%, *Giardia* by 58% and enteric viruses by 98% (Karpiscak *et al.*, 1996).

25.7 SLUDGE PROCESSING

Primary, secondary and even tertiary sludges generated during wastewater treatment are a major by-product of the treatment process. These sludges, in turn, are usually subjected to a variety of treatments. Raw sludge is sometimes subjected to screening to remove coarse materials including grit that cannot be broken down biologically. Thickening is usually done to increase the solids content of the sludge. This can be achieved via centrifugation which increases the solids content to approximately 12%. Dewatering can further concentrate the solids content to 20−40%. This is normally achieved via filtration, or by the use of drying beds. Conditioning enhances the separation of solids from the liquid phase. This is usually accomplished by the addition of inorganic salts such as: alum; lime; ferrous or ferric salts; or synthetic organic polymers known as polyelectrolytes. All of these processes reduce the water content of the sludge, which ultimately reduces transportation costs to the final disposal and/or utilization site.

Finally, stabilization technologies are available, reducing both the solids content of the sludge and inactivating pathogenic microbes present in the sludge.

25.7.1 Stabilization Technologies

25.7.1.1 *Aerobic Digestion*

This consists of adding air or oxygen to sludge in a 4- to 8-foot-deep open tank. The oxygen concentration within the tank must be maintained above 1 mg/L to avoid the production of foul odors. The mean residence time in the tank is 12−60 days depending on the tank temperature. During this process, microbes aerobically degrade organic substrate, reducing the volatilize solids content of the sludge by 40−50% (U.S. EPA, 1992b). Digestion temperatures are frequently moderate or mesophilic (30−40°C). By increasing the oxygen content, thermophilic digestion can be induced (>60°C). By increasing

Case Study 25.1 Sweetwater Wetlands Infiltration–Extraction Facility in Tucson, Arizona

Tucson, Arizona, is located in the Sonoran Desert in the southwestern United States. Because of limited water supplies reclamation of wastewater is critical to meet water needs in the region. To meet these needs a system was built to provide tertiary effluents derived from an activated sludge/trickling filter system of sufficient quality to be used for landscape irrigation. The system is composed of several components that allows for various treatments and storage of tertiary effluent (Figures 25.22 and 25.23). A tertiary treatment plant filters the secondary effluent (to reduce turbidity and microorganisms) and then provides additional disinfection. The backwash from the filters is then discharged into an artificial wetland for treatment. When the water exists the wetland it is discharged into infiltration basins where it is further treated. In times of low reclaimed water demand (winter) the tertiary effluent may be discharged into the infiltration basins. The subsurface aquifer is then used as a storage facility, the water then being pumped to the surface (extraction) when needed during periods of peak demand.

The multiple barriers of conventional and natural technologies are design to enhance the removal of chemical and microbial contaminates. Filtration of the secondary wastewater during tertiary treatment allows for reduction of the larger protozoan parasites (which are more resistant to disinfection than enteric bacteria and viruses) and more effective disinfection. In the wetlands protozoan parasites settle out and bacteria and viruses are reduced by inactivation by sunlight (UV light) and microbial antagonism. Infiltration of the water through the soil results in further removal of pathogens by filtration and adsorption to soil particle (especially viruses) (see Figure 25.24).

FIGURE 25.24 Aerial view of Sweetwater Reclamation Facility. Numbered blue areas are infiltration basins. Photo courtesy Water Reuse Association.

the temperature and the retention time, the degree of pathogen inactivation can be enhanced. Pathogen concentrations ultimately determine the treatment level of the product. Class B biosolids can contain many human pathogens (see Chapter 26). Class A biosolids, which result from more stringent and enhanced treatment, contain very low or nondetectable levels of pathogens. The degree of treatment, for Class A versus Class B, has important implications on the reuse potential of the material for land application (see Chapter 26). Aerobic digestion generally results in the production of Class B biosolids.

25.7.1.2 Anaerobic Digestion

This type of microbial digestion occurs under low redox conditions, with low oxygen concentrations. Carbon dioxide is the major terminal electron acceptor used (see Chapter 3), and results in the conversion of organic substrate to methane and carbon dioxide. This process reduces the volatile solids by 35–60% (Bitton, 2011), and results in the production of Class B biosolids. The advantages and disadvantages of anaerobic digestion relative to aerobic digestion are shown in Information Box 25.1.

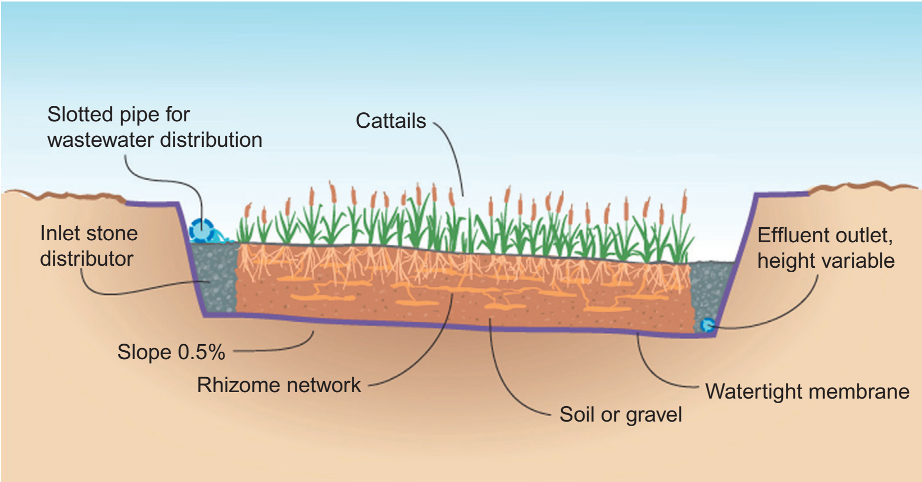

FIGURE 25.25 Cross-section of a subsurface wetland.

Information Box 25.1 Advantages and Disadvantages of Anaerobic Digestion

Advantages

- No oxygen requirement, which reduces cost
- Reduced mass of biosolids due to low energy yields of anaerobic metabolism (see also Chapter 3)
- Methane produced, which can be used to generate electricity
- Enhanced degradation of xenobiotic compounds

Disadvantages

- Slower than aerobic digestion
- More sensitive to toxics

Adapted from Bitton (2011).

FIGURE 25.26 Wood bulking agent for composting. The wood is shredded to increase the surface area of bulking agent for composting.

Information Box 25.2 Factors Affecting Efficient Composting

Temperature.Adequate aeration and moisture must be maintained to ensure temperatures reach 60°C, to inactivate microbial pathogens.
Aeration.Air must be provided via blowers or by turning.
Moisture. Conditions must be neither too moist, which promotes anaerobic activity, nor too dry, which limits microbial activity.
C:N ratio.The C:N ratio of the substrate should be maintained around 25:1, to ensure adequate but not excessive amounts of nitrogen for the microbes.
Surface area of bulking agent.Shredded material should be used to increase substrate surface area for microbial metabolism.

Source: Pepper et al. (2006a).

25.7.2 Sludge Processing to Produce Class A Biosolids

Class B biosolids that arise following digestion can be further treated to Class A levels prior to land application. The three most important technologies to achieve this goal are: composting; lime treatment; and heat treatment.

25.7.2.1 Composting

Composting consists of mixing sludge with a bulking agent that normally has a high C:N ratio (Figure 25.26). This is necessary because of the low C:N ratio of the sludge. The mixtures are normally kept moist but aerobic. These conditions result in very high microbial activity, and the generation of heat that increases the temperature of the composting material. Factors affecting the composting process are shown in Information Box 25.2. There are three main types of composting systems:

- The aerated static pile process typically consists of mixing dewatered digested sludge with wood chips. Aeration of the pile is normally provided by blowers during a 21-day composting period. During this active composting period, temperatures increase to the mesophilic range (20–40°C) where microbial degradation occurs via bacteria and fungi. Temperatures subsequently increase to 40–80°C, with microbial populations dominated by thermophilic (heat tolerant) and spore-forming organisms. These high temperatures inactivate pathogenic microorganisms, and frequently result in a Class A biosolid product. Subsequently, the compost is cured for at least 30 days, during which time temperatures within the pile decrease to ambient levels.

- The windrow process is similar to the static pile process except that instead of a pile, the sludge and bulking agent are laid out in long rows of dimensions: 2 m × 3 m × 80 m (Figure 25.27). Aeration for windrows is provided by turning the windrows several times a week. Once again, if the composting process is efficient, Class A biosolids are produced.
- In enclosed systems the composting is conducted in steel vessels of size 10–15 m high by 3–4 m diameter. For this type of composting, aeration via blowers and temperature of the composting are carefully controlled. This results in a high quality Class A compost, with little or no odor problems. However, costs of enclosed systems are higher.

FIGURE 25.27 Biosolid composting via the windrow process. Here three windrows are illustrated.

25.7.2.2 Lime and Heat Treatment

Lime stabilization involves the addition of lime as $Ca(OH)_2$ or CaO, such that the pH of digested sludge is equal to or greater than 12 for at least 2 hours. Liming is very effective at inactivating bacterial and viral pathogens, but less so for parasites (Bitton, 2011). Lime stabilization also reduces odors, and can result in a Class A biosolid product.

Heat treatment involves heating sludge under pressure to temperatures up to 260°C for 30 minutes. This process kills microbial pathogens and parasites, and also further dewaters the sludge.

25.7.2.3 The Cambi Thermal Hydrolysis Process

The Cambi process utilizes thermal hydrolysis as a pretreatment to anaerobic digestion. This increases the

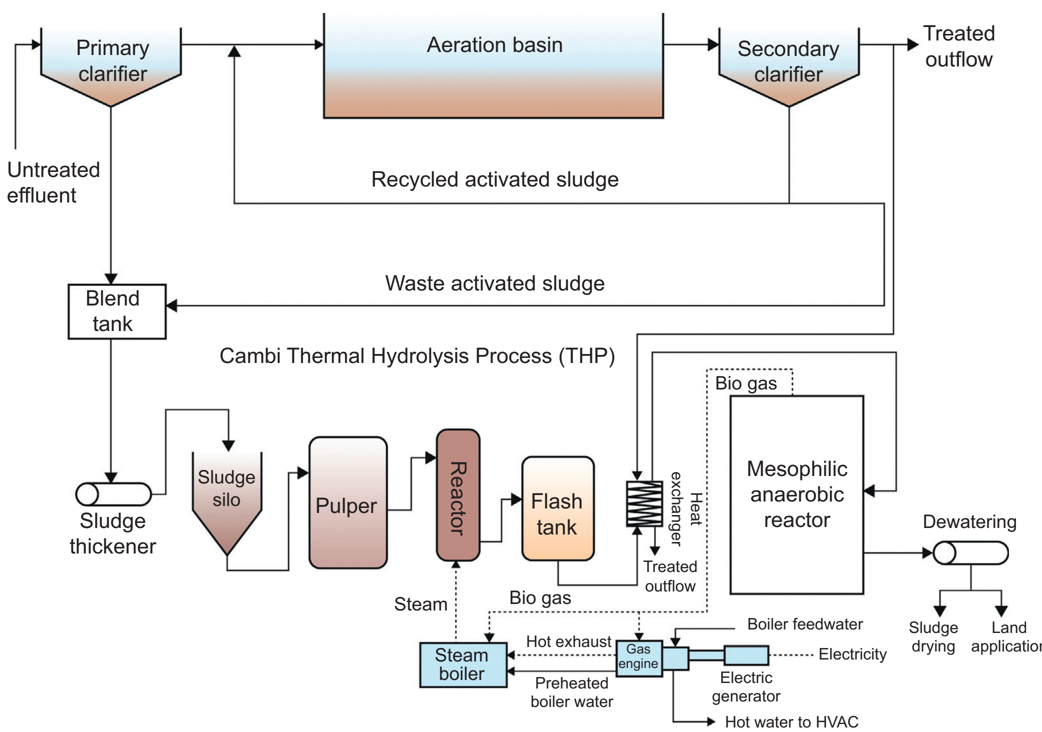

FIGURE 25.28 Cambi thermal hydrolysis process (THP).

FIGURE 25.29 Cambi thermal hydrolysis—preferred technology for Cotton Valley and Whitlingham. © Mott MacDonald Group Limited 2013.

microbial degradation of organic volatile solids and increases the amount of biogas obtained. This process also facilitates a higher degree of separation of solid and liquid phase after digestion. The process is depicted in Figure 25.28. Sludge generated during primary and secondary treatment is dewatered to approximately 15−20% dry solids content, preheated to 100% in the pulper tank, then heated to 150 to 170°C under 8−9 bar pressure in the reactor. In the flash tank the sludge cools to about 100°C and the released steam is recirculated. Further cooling to 35°C occurs via the heat exchangers where more energy is recycled via the production of hot water. Following this, the sludge is subjected to mesophilic anaerobic digestion, leading to the production of biogas and Class A biosolids, since no pathogens can survive the steam treatment. A typical plant is shown in Figure 25.29. Cambi is now well established in several countries within Europe, particularly the United Kingdom and Norway. The first thermal hydrolysis plant to be built in the U.S. is the Blue Plains treatment plant in Washington DC, with startup in 2014. This plant will treat up to 450 dry tons of sludge per day, and the biogas produced will cover the entire steam needs of the plant saving $20 million a year from the energy saved.

Overall, 50% of all biosolids is land applied in the U.S. with most of it currently being Class B (see Chapter 26).

QUESTIONS AND PROBLEMS

1. What are the three major steps in modern wastewater treatment?
2. Why is it important to reduce the amount of biodegradable organic matter and nutrients during sewage treatment?
3. When would tertiary treatment of wastewater be necessary?
4. What are some types of tertiary treatment?
5. What are the processes involved in the removal of heavy metals from wastewater during treatment by artificial wetlands.
6. What are the three types of land application of wastewater? Which one is most likely to contaminate the groundwater with enteric viruses? Why? What factors determine how far viruses will be transported in groundwater? How does nitrogen removal occur? Phosphorus removal?
7. What is the major contaminant in groundwater associated with the use of on-site treatment systems?
8. What factors may determine the concentration of enteric pathogens in domestic raw sewage?
9. Five milliliters of a wastewater sample is added to dilution water in a 300-ml BOD bottle. If the following results are obtained, what is the BOD after 3 days and 5 days?

Time (days)	Dissolved oxygen (mg/L)
0	9.55
1	4.57
2	4.00
3	3.20
4	2.60
5	2.40
6	2.10

10. List some advantages and disadvantages of the wetland treatment of sewage.
11. What is the major mechanism of pathogen removal during activated sludge treatment?
12. What treatment process would you need to obtain an 8-$\log_{10}$ reduction of (i) enteric viruses from raw sewage, and (ii) *Giardia*?
13. How effective do you think sunlight is in killing *Cryptosporidium*? Enteric viruses?

REFERENCES AND RECOMMENDED READING

APHA (1998) "Standard Methods for Water and Wastewater," American Public Health Association, Washington, DC.

Bitton, G. (2011) "Wastewater Microbiology," 4th ed. Wiley-Liss, New York.

Chang, I. S., LeClech, P., Jefferson, B., and Judd, S. (2002) Membrane fouling in membrane bioreactors for wastewater treatment. *J. Environ. Eng.* **128**, 1018−1029.

Curds, C. R., and Hawkes, H. A. (1983) Ecological aspects of used water treatment, In "The Processes and their Ecology, **vol 3. Academic Press, London, 1-113.**

Edzwald, J. K. (2010) Dissolved air flotation and me. *Wat. Res.* **44**, 2077−2106.

Kadlec, R. H., and Wallace, S. (2008) "Treatment Wetlands," CRC Press, Boca Raton, FL.

Karpiscak, M. M., Gerba, C. P., Watt, P. M., Foster, K. E., and Falabi, J. A. (1996) Multi-species plant systems for wastewater quality improvements and habitat enhancement. *Wat. Sci. Technol.* **33**, 231–236.

Leong, L. Y. C. (1983) Removal and inactivation of viruses by treatment processes for portable water and wastewater—a review. *Wat. Sci. Technol.* **15**, 91–114.

Metcalf and Eddy, Inc. (2003) "Wastewater Engineering," McGraw-Hill, New York.

Neilson, J. W., Josephson, K. L., Pepper, I. L., Arnold, R. B., DiGiovanni, G. D., and Sinclair, N. A. (1994) Frequency of horizontal gene transfer of a large catabolic plasmid (pJP4) in soil. *Appl. Environ. Microbiol.* **60**, 4053–4058.

Newby, D. T., Gentry, T. J., and Pepper, I. L. (2000b) Comparison of 2,4-dichlorophenoxyacetic acid degradation and plasmid transfer in soil resulting from bioaugmentation with two different pJP4 donors. *Appl. Environ. Microbiol.* **66**, 3399–3407.

Park, S., Seon, J., Byun, I., Cho, S., Park, T., and Lee, T. (2010) Comparison of nitrogen removal and microbial distribution in wastewater treatment process under different electron donor conditions. *Bioresour. Technol.* **101**, 2988–2995.

Pepper, I. L., Brooks, J. P., and Gerba, C. P. (2006a) Pathogens in Biosolids. *Adv. Agron.* **90**, 1–41.

Pepper, I. L., Gerba, C. P., and Brusseau, M. L. (2006b) "Environmental and Pollution Science," 2nd ed. Academic Press, San Diego, CA.

Rao, V. C., Metcalf, T. G., and Melnick, J. L. (1986) Removal of pathogens during wastewater treatment. In "Biotechnology" (H. J. Rehm, and G. Reed, eds.), vol. 8, VCH, Berlin, pp. 531–554.

Rose, J. B., and Carnahan, R. P. (1992) "Pathogen removal by full scale wastewater treatment," Report to Florida Department of Environmental Regulation, Tallahassee, Fl.

Rose, J. B., and Gerba, C. P. (1991) Use of risk assessment for development of microbial standards. *Wat. Sci. Technol.* **24**, 29–34.

Rusin, P., Enriquez, C. E., Johnson, D., and Gerba, C. P. (2000) Environmentally transmitted pathogens. In "Environmental Microbiology" (R. M. Maier, I. L. Pepper, and C. P. Gerba, eds.), Academic Press, San Diego, pp. 447–489.

U.S. Census Bureau, 1990. Historical Census of Housing Tables: Sewage Disposal. Available from <http://www.census.gov/hhes/www/housing/census/historic/sewage.html>.

U.S. EPA (U.S. Environmental Protection Agency) (1977) Wastewater treatment facilities for sewered small communities. EPA–62511–77–009, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency) (1992a) Guidelines for water reuse. EPA/625/R–92/004, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency) (1992b) Technical support document for land application of sewage sludge, vol. I EPA 822/R–93–001A.

U.S. EPA (U.S. Environmental Protection Agency) (1994) A plain English guide to the EPA part 503 biosolids rule. EPA 832/R–93/003 U.S. Environmental Protection Agency.

U.S. EPA (U.S. Environmental Protection Agency) (1996) Technical support document for the round two sewage sludge pollutants. EPA–822–R–96–003.

U.S. EPA (U.S. Environmental Protection Agency) (1997) Exposure factors handbook, EPA/600/P–95/002, August 1997.

U.S. EPA (U.S. Environmental Protection Agency) (1998) Volume I—General factors, exposure factors handbook. Update to exposure factors handbook, EPA/600/8–89/043, May 1989.

U.S. EPA (U.S. Environmental Protection Agency) (1999a) Environmental regulations and technology: control of pathogens and vector attraction in sewage sludge. EPA/625/R–92/013. Available online: <http://www.epa.gov/ttbnrmrl/625/R-92/013.htm>.

U.S. EPA (U.S. Environmental Protection Agency) (1999b) "Environmental regulation and technology. Control of pathogens and vector attraction in sewage sludge EPA/652/R–92/013. Revised 1999," Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC, 177 pp.

Wang, Y-P., Liu, X-W., Li, W-W., Wang, Y-K., Shery, G. P., Zeng, R. J., and Yu, H-Q. (2012) A microbial fuel cell-membrane bioreactor integrated system for cost-effective wastewater treatment. *Appl. Energy* **98**, 230–235.

Yates, M. V. (1994) Monitoring concerns and procedures for human health effects. In "Wastewater Reuse for Golf Course Irrigation," CRC Press, Boca Raton, FL, 143–171.