2
Seafood Processing Wastewater Treatment

Joo-Hwa Tay and Kuan-Yeow Show
Nanyang Technological University, Singapore

Yung-Tse Hung
Cleveland State University, Cleveland, Ohio, U.S.A.

2.1 INTRODUCTION

The seafood industry consists primarily of many small processing plants, with a number of larger plants located near industry and population centers. Numerous types of seafood are processed, such as mollusks (oysters, clams, scallops), crustaceans (crabs and lobsters), saltwater fishes, and freshwater fishes. As in most processing industries, seafood-processing operations produce wastewater containing substantial contaminants in soluble, colloidal, and particulate forms. The degree of the contamination depends on the particular operation; it may be small (e.g., washing operations), mild (e.g., fish filleting), or heavy (e.g., blood water drained from fish storage tanks).

Wastewater from seafood-processing operations can be very high in biochemical oxygen demand (BOD), fat, oil and grease (FOG), and nitrogen content. Literature data for seafood processing operations showed a BOD production of 1–72.5 kg of BOD per tonne of product [1]. White fish filleting processes typically produce 12.5–37.5 kg of BOD for every tonne of product. BOD is derived mainly from the butchering process and general cleaning, and nitrogen originates predominantly from blood in the wastewater stream [1].

It is difficult to generalize the magnitude of the problem created by these wastewater streams, as the impact depends on the strength of the effluent, the rate of discharge, and the assimilatory capacity of the receiving water body. Nevertheless, key pollution parameters must be taken into account when determining the characteristics of a wastewater and evaluating the efficiency of a wastewater treatment system. Section 2.2 discusses the parameters involved in the characterization of the seafood processing wastewater.

Pretreatment and primary treatment for seafood processing wastewater are presented in Section 2.3. These are the simplest operations to reduce contaminant load and remove oil and grease from an effluent of seafood processing wastewater. Common pretreatments for seafood-processing wastewater include screening, settling, equalization, and dissolved air flotation.

Section 2.4 focuses on biological treatments for seafood processing wastewater, namely aerobic and anaerobic treatments. The most common operations of biological treatments are also described in this section.

Section 2.5 discusses the physico-chemical treatments for seafood processing wastewater. These operations include coagulation, flocculation, and disinfection. Direct disposal of seafood processing wastewaters is discussed in Section 2.6. Potential problems in land application are highlighted. General seafood processing plant schemes are presented in Section 2.7. Economic considerations are always the most important factors that influence the final decision for selecting processes for wastewater treatment. The economic issues related to wastewater treatment process are discussed in Section 2.8.

2.2 SEAFOOD-PROCESSING WASTEWATER CHARACTERIZATION

Seafood-processing wastewater characteristics that raise concern include pollutant parameters, sources of process waste, and types of wastes. In general, the wastewater of seafood-processing wastewater can be characterized by its physicochemical parameters, organics, nitrogen, and phosphorus contents. Important pollutant parameters of the wastewater are five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total suspended solids (TSS), fats, oil and grease (FOG), and water usage [2]. As in most industrial wastewaters, the contaminants present in seafood-processing wastewaters are an undefined mixture of substances, mostly organic in nature. It is useless or practically impossible to have a detailed analysis for each component present; therefore, an overall measurement of the degree of contamination is satisfactory.

2.2.1 Physicochemical Parameters

pH

pH serves as one of the important parameters because it may reveal contamination of a wastewater or indicate the need for pH adjustment for biological treatment of the wastewater. Effluent pH from seafood processing plants is usually close to neutral. For example, a study found that the average pH of effluents from blue crab processing industries was 7.63, with a standard deviation of 0.54; for non-Alaska bottom fish, it was about 6.89 with a standard deviation of 0.69 [2]. The pH levels generally reflect the decomposition of proteinaceous matter and emission of ammonia compounds.

Solids Content

Solids content in a wastewater can be divided into dissolved solids and suspended solids. However, suspended solids are the primary concern since they are objectionable on several grounds. Settleable solids may cause reduction of the wastewater duct capacity; when the solids settle in the receiving water body, they may affect the bottom-dwelling flora and the food chain. When they float, they may affect the aquatic life by reducing the amount of light that enters the water.

Soluble solids are generally not inspected even though they are significant in effluents with a low degree of contamination. They depend not only on the degree of contamination but also on the quality of the supply water used for the treatment. In one analysis of fish filleting wastewater, it was found that 65% of the total solids present in the effluent were already in the supply water [3].

Odor

In seafood-processing industries, odor is caused by the decomposition of the organic matter, which emits volatile amines, diamines, and sometimes ammonia. In wastewater that has become septic, the characteristic odor of hydrogen sulfide may also develop. Odor is a very important issue in relation to public perception and acceptance of any wastewater treatment plant. Although relatively harmless, it may affect general public life by inducing stress and sickness.

Temperature

To avoid affecting the quality of aquatic life, the temperature of the receiving water body must be controlled. The ambient temperature of the receiving water body must not be increased by more than 2 or 3°C, or else it may reduce the dissolved oxygen level. Except for wastewaters from cooking and sterilization processes in canning factories, fisheries do not discharge wastewaters above ambient temperatures. Therefore, wastewaters from canning operations should be cooled if the receiving water body is not large enough to restrict the change in temperature to 3°C [4].

2.2.2 Organic Content

The major types of wastes found in seafood-processing wastewaters are blood, offal products, viscera, fins, fish heads, shells, skins, and meat "fines." These wastes contribute significantly to the suspended solids concentration of the waste stream. However, most of the solids can be removed from the wastewater and collected for animal food applications. A summary of the raw wastewater characteristics for the canned and preserved seafood processing industry is presented in Table 2.1.

Wastewaters from the production of fish meal, solubles, and oil from herring, menhaden, and alewives can be divided into two categories: high-volume, low-strength wastes and low-volume, high-strength wastes [5].

High-volume, low-strength wastes consist of the water used for unloading, fluming, transporting, and handling the fish plus the washdown water. In one study, the fluming flow was estimated to be 834 L/tonne of fish with a suspended solids loading of 5000 mg/L. The solids consisted of blood, flesh, oil, and fat [2]. The above figures vary widely. Other estimates listed herring pump water flows of 16 L/sec with total solids concentrations of 30,000 mg/L and oil concentrations of 4000 mg/L. The boat's bilge water was estimated to be 1669 L/ton of fish with a suspended solids level of 10,000 mg/L [2].

Stickwaters comprise the strongest wastewater flows. The average BOD_5 value for stickwater has been listed as ranging from 56,000 to 112,000 mg/L, with average solids concentrations, mainly proteinaceous, ranging up to 6%. The fish-processing industry has found the recovery of fish solubles from stickwater to be at least marginally profitable. In most instances, stickwater is now evaporated to produce condensed fish solubles. Volumes have been estimated to be about 500 L/ton of fish processed [2].

The degree of pollution of a wastewater depends on several parameters. The most important factors are the types of operation being carried out and the type of seafood being processed. Carawan [2] reported on an EPA survey with BOD_5, COD, TSS, and fat, oil and grease (FOG) parameters. Bottom fish was found to have a BOD_5 of 200–1000 mg/L, COD of 400–2000 mg/L, TSS of 100–800 mg/L, and FOG of 40–300 mg/L. Fish meal plants were reported to have a BOD_5 of 100–24,000 mg/L, COD of 150–42,000 mg/L, TSS of 70–20,000 mg/L, and FOG of 20–5000 mg/L. The higher numbers were representative of bailwater only. Tuna plants were reported to have a BOD_5 of 700 mg/L, COD of 1600 mg/L,

Table 2.1 Raw Wastewater Characteristics of the Canned and Preserved Seafood-Processing Industries

Effluent	Flow (L/day)	BOD_5 (mg/L)	COD (mg/L)	TSS (mg/L)	FOG (mg/L)
Farm-raised catfish	79.5K–170K	340	700	400	200
Conventional blue crab	2650	4400	6300	420	220
Mechanized blue crab	75.7K–276K	600	1000	330	150
West coast shrimp	340K–606K	2000	3300	900	700
Southern nonbreaded shrimp	680K–908K	1000	2300	800	250
Breaded shrimp	568K–757K	720	1200	800	–
Tuna processing	246K–13.6M	700	1600	500	250
Fish meal	348K–378.5K[a]	100–24M[a]	150–42K[a]	70–20K[a]	20K–5K[a]
All salmon	220K–1892.5K	253–2600	300–5500	120–1400	20–550
Bottom and finfish (all)	22.71K–1514K	200–1000	400–2000	100–800	40–300
All herring	110K	1200–6000	3000–10,000	500–5000	600–5000
Hand shucked clams	325.5K–643.5K	800–2500	1000–4000	600–6000	16–50
Mechanical clams	1135.5K–11.4M	500–1200	700–1500	200–400	20–25
All oysters	53K–1211K	250–800	500–2000	200–2000	10–30
All scallops	3.785K–435K	200K–10M	300–11,000	27–4000	15–25
Abalone	37.85K–53K	430–580	800–1000	200–300	22–30

BOD_5, five day biochemical oxygen demand; COD, chemical oxygen demand; TSS, total suspended solids; FOG, fat, oil, and grease.

[a] Higher range is for bailwater only; K = 1000; M = 1,000,000.

Source: Ref. 2.

TSS of 500 mg/L, and FOG of 250 mg/L. Seafood-processing wastewater was noted to sometimes contain high concentrations of chlorides from processing water and brine solutions, and organic nitrogen of up to 300 mg/L from processing water.

Several methods are used to estimate the organic content of the wastewater. The two most common methods are biochemical oxygen demand (BOD) and chemical oxygen demand (COD).

Biochemical Oxygen Demand

Biochemical oxygen demand (BOD) estimates the degree of contamination by measuring the oxygen required for oxidation of organic matter by aerobic metabolism of the microbial flora. In seafood-processing wastewaters, this oxygen demand originates mainly from two sources. One is the carbonaceous compounds that are used as substrate by the aerobic microorganisms; the other source is the nitrogen-containing compounds that are normally present in seafood-processing wastewaters, such as proteins, peptides, and volatile amines. Standard BOD tests are conducted at 5-day incubation for determination of BOD_5 concentrations.

Wastewaters from seafood-processing operations can be very high in BOD_5. Literature data for seafood processing operations show a BOD_5 production of one to 72.5 kg of BOD_5 per ton of product [1]. White fish filleting processes typically produce 12.5–37.5 kg BOD_5 for every ton of product. The BOD is generated primarily from the butchering process and from general cleaning, while nitrogen originates predominantly from blood in the wastewater stream [1].

Chemical Oxygen Demand

Another alternative for measuring the organic content of wastewater is the chemical oxygen demand (COD), an important pollutant parameter for the seafood industry. This method is more convenient than BOD_5 since it needs only about 3 hours for determination compared with 5 days for BOD_5 determination. The COD analysis, by the dichromate method, is more commonly used to control and continuously monitor wastewater treatment systems. Because the number of compounds that can be chemically oxidized is greater than those that can be degraded biologically, the COD of an effluent is usually higher than the BOD_5. Hence, it is common practice to correlate BOD_5 vs. COD and then use the analysis of COD as a rapid means of estimating the BOD_5 of a wastewater.

Depending on the types of seafood processing, the COD of the wastewater can range from 150 to about 42,000 mg/L. One study examined a tuna-canning and byproduct rendering plant for five days and observed that the average daily COD ranged from 1300–3250 mg/L [2].

Total Organic Carbon

Another alternative for estimating the organic content is the total organic carbon (TOC) method, which is based on the combustion of organic matter to carbon dioxide and water in a TOC analyzer. After separation of water, the combustion gases are passed through an infrared analyzer and the response is recorded. The TOC analyzer is gaining acceptance in some specific applications as the test can be completed within a few minutes, provided that a correlation with the BOD_5 or COD contents has been established. An added advantage of the TOC test is that the analyzer can be mounted in the plant for online process control. Owing to the relatively high cost of the apparatus, this method is not widely used.

Fats, Oil, and Grease

Fats, oil, and grease (FOG) is another important parameter of seafood-processing wastewater. The presence of FOG in an effluent is mainly due to the processing operations such as canning, and the seafood being processed. The FOG should be removed from wastewater because it usually floats on the water's surface and affects the oxygen transfer to the water; it is also objectionable from an aesthetic point of view. The FOG may also cling to wastewater ducts and reduce their capacity in the long term. The FOG of a seafood-processing wastewater varies from zero to about 17,000 mg/L, depending on the seafood being processed and the operation being carried out.

2.2.3 Nitrogen and Phosphorus

Nitrogen and phosphorus are nutrients that are of environmental concern. They may cause proliferation of algae and affect the aquatic life in a water body if they are present in excess. However, their concentration in the seafood-processing wastewater is minimal in most cases. It is recommended that a ratio of N to P of 5 : 1 be achieved for proper growth of the biomass in the biological treatment [6,7].

Sometime the concentration of nitrogen may also be high in seafood-processing wastewaters. One study shows that high nitrogen levels are likely due to the high protein content (15–20% of wet weight) of fish and marine invertebrates [8]. Phosphorus also partly originates from the seafood, but can also be introduced with processing and cleaning agents.

2.2.4 Sampling

Of equal importance is the problem of obtaining a truly representative sample of the stream effluent. The samples may be required not only for the 24-hour effluent loads, but also to determine the peak load concentrations, the duration of peak loads, and the occurrence of variation throughout the day. The location of sampling is usually made at or near the point of discharge to the receiving water body, but in the analysis prior to the design of a wastewater treatment, facility samples will be needed from each operation in the seafood-processing facility. In addition, samples should be taken more frequently when there is a large variation in flow rate, although wide variations may also occur at constant flow rate.

The particular sampling procedure may vary, depending on the parameter being monitored. Samples should be analyzed as soon as possible after sampling because preservatives often interfere with the test. In seafood-processing wastewaters, there is no single method of sample preservation that yields satisfactory results for all cases, and all of them may be inadequate with effluents containing suspended matter. Because samples contain an amount of settleable solids in almost all cases, care should be taken in blending the samples just prior to analysis. A case in which the use of preservatives is not recommended is that of BOD_5 storage at low temperatures (4°C), which may be used with caution for very short periods, and chilled samples should be warmed to 20°C before analysis. For COD determination, the samples should be collected in clean glass bottles, and can be preserved by acidification to a pH of 2 with concentrated sulfuric acid. Similar preservation can also be done for organic nitrogen determination. For FOG determination, a separate sample should be collected in a wide-mouth glass bottle that is well rinsed to remove any trace of detergent. For solids determination, an inspection should be done to ensure that no suspended matter adheres to the walls and that the solids are refrigerated at 4°C to prevent decomposition of biological solids. For the analysis of phosphorus, samples should be preserved by adding 40 mg/L of mercuric chloride and stored in well-rinsed glass bottles at −10°C [4].

2.3 PRIMARY TREATMENT

In the treatment of seafood-processing wastewater, one should be cognizant of the important constituents in the waste stream. This wastewater contains considerable amounts of insoluble suspended matter, which can be removed from the waste stream by chemical and physical means. For optimum waste removal, primary treatment is recommended prior to a biological treatment process or land application. A major consideration in the design of a treatment system is that the solids should be removed as quickly as possible. It has been found that the longer the detention time between waste generation and solids removal, the greater the soluble BOD_5 and COD with corresponding reduction in byproduct recovery. For seafood-processing wastewater, the primary treatment processes are screening, sedimentation, flow equalization, and dissolved air flotation. These unit operations will generally remove up to 85% of the total suspended solids, and 65% of the BOD_5 and COD present in the wastewater.

2.3.1 Screening

The removal of relatively large solids (0.7 mm or larger) can be achieved by screening. This is one of the most popular treatment systems used by food-processing plants, because it can reduce the amount of solids being discharged quickly. Usually, the simplest configuration is that of flow-through static screens, which have openings of about 1 mm. Sometimes a scrapping mechanism may be required to minimize the clogging problem in this process.

Generally, tangential screening and rotary drum screening are the two types of screening methods used for seafood-processing wastewaters. Tangential screens are static but less prone to clogging due to their flow characteristics (Fig. 2.1), because the wastewater flow tends to avoid clogging. The solids removal rates may vary from 40 to 75% [4]. Rotary drum screens are mechanically more complex. They consist of a drum that rotates along its axis, and the effluent enters through an opening at one end. Screened wastewater flows outside the drum and the retained solids are washed out from the screen into a collector in the upper part of the drum by a spray of the wastewater.

Fish solids dissolve in water with time; therefore, immediate screening of the waste streams is highly recommended. Likewise, high-intensity agitation of waste streams should be minimized before screening or even settling, because they may cause breakdown of solids rendering them more difficult to separate. In small-scale fish-processing plants, screening is often used with simple settling tanks.

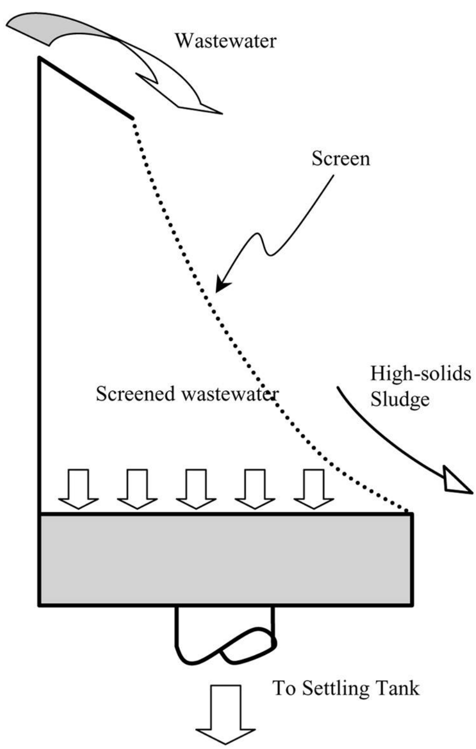

Figure 2.1 Diagram of an inclined or tangential screen.

2.3.2 Sedimentation

Sedimentation separates solids from water using gravity settling of the heavier solid particles [9]. In the simplest form of sedimentation, particles that are heavier than water settle to the bottom of a tank or basin. Sedimentation basins are used extensively in the wastewater treatment industry and are commonly found in many flow-through aquatic animal production facilities. This operation is conducted not only as part of the primary treatment, but also in the secondary treatment for separation of solids generated in biological treatments, such as activated sludge or trickling filters. Depending on the properties of solids present in the wastewater, sedimentation can proceed as discrete settling, flocculent settling, or zone settling. Each case has different characteristics, which will be outlined.

Discrete settling occurs when the wastewater is relatively dilute and the particles do not interact. A schematic diagram of discrete settling is shown in Figure 2.2.

Calculations can be made on the settling velocity of individual particles. In a sedimentation tank, settling occurs when the horizontal velocity of a particle entering the basin is less than the vertical velocity in the tank. The length of the sedimentation basin and the detention time can be calculated so that particles with a particular settling velocity (V_c) will settle to the bottom of the basin [9]. The relationship of the settling velocity to the detention time and basin depth is:

$$V_c = \frac{\text{depth}}{\text{detention time}} \tag{2.1}$$

For flocculent suspension, the formation of larger particles due to coalescence depends on several factors, such as the nature of the particles and the rate of coalescence. A theoretical analysis is not feasible due to the interaction of particles, which depends, among other factors, on the overflow rate, the concentration of particles, and the depth of the tank.

Zone settling occurs when the particles do not settle independently. In this case, an effluent is initially uniform in solids concentration and settles in zones. The clarified effluent and compaction zones will increase in size while the other intermediate zones will eventually disappear.

The primary advantages of using sedimentation basins to remove suspended solids from effluents from seafood-processing plants are: the relative low cost of designing, constructing, and operating sedimentation basins; the low technology requirements for the operators; and the demonstrated effectiveness of their use in treating similar effluents. Therefore, proper design,

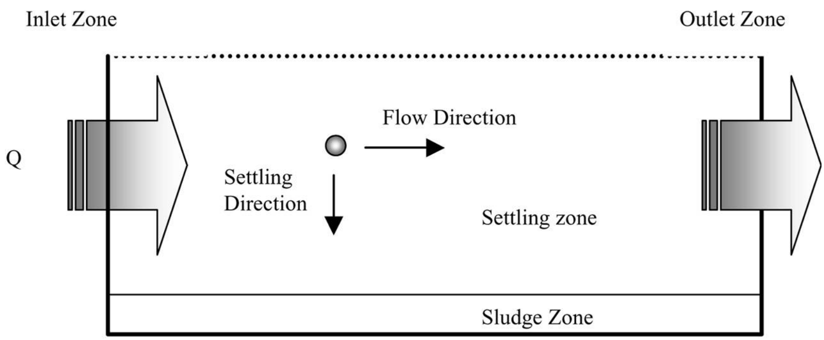

Figure 2.2 Schematics of discrete settling.

construction, and operation of the sedimentation basin are essential for the efficient removal of solids. Solids must be removed at proper intervals to ensure the designed removal efficiencies of the sedimentation basin.

Rectangular settling tanks (Fig. 2.3) are generally used when several tanks are required and there is space constraint, because they occupy less space than several circular tanks. Usually there is a series of chain-driven scrapers used for removal of solids. The sludge is collected in a hopper at the end of the tank, where it may be removed by screw conveyors or pumped out.

Circular tanks are reported to be more effective than rectangular ones. The effluent in a circular tank circulates radially, with the water introduced at the periphery or from the center. The configuration is shown in Figure 2.4. Solids are generally removed from near the center, and the sludge is forced to the outlet by two or four arms provided with scrapers, which span the radius of the tank. For both types of flows, a means of distributing the flow in all directions is provided. An even distribution of inlet and outlet flows is important to avoid short-circuiting in the tank, which would reduce the separation efficiency.

Generally, selection of a circular tank size is based on the surface-loading rate of the tank. It is defined as the average daily overflow divided by the surface area of the tank and is expressed as volume of wastewater per unit time and unit area of settler (m^3/m^2 day), as shown in Eq. (2.2). This loading rate depends on the characteristics of the effluent and the solids content. The retention time in the settlers is generally one to two hours, but the capacity of the tanks must be determined by taking into account the peak flow rates so that acceptable separation is obtained in these cases. Formation of scum is almost unavoidable in seafood-processing wastes, so some settling tanks are provided with a mechanism for scum removal.

Selection of the surface loading rate depends on the type of suspensions to be removed. The design overflow rates must be low enough to ensure satisfactory performance at peak rates of flow, which may vary from two to three times the average flow.

$$V_o = \frac{Q}{A} \tag{2.2}$$

where V_o = overflow rate (surface-loading rate) (m^3/m^2 day), Q = average daily flow (m^3/day), and A = total surface area of basin (m^2).

The area A is calculated by using inside tank dimensions, disregarding the central stilling well or inboard well troughs. The quantity of overflow from a primary clarifier Q is equal to the wastewater influent, and since the volume of the tank is established, the detention period in the tank is governed by water depth. The side water depth of the tank is

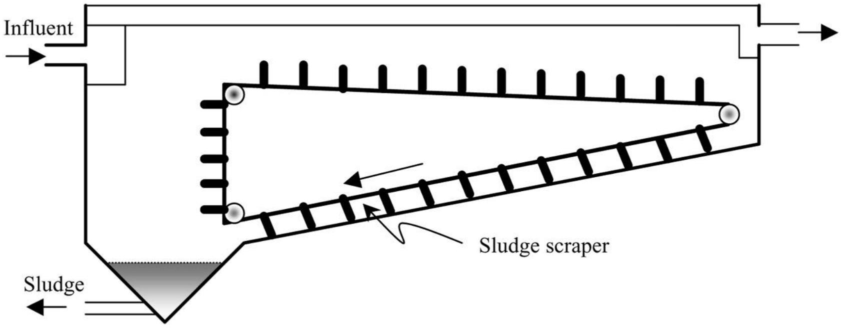

Figure 2.3 Diagram of a rectangular clarifier.

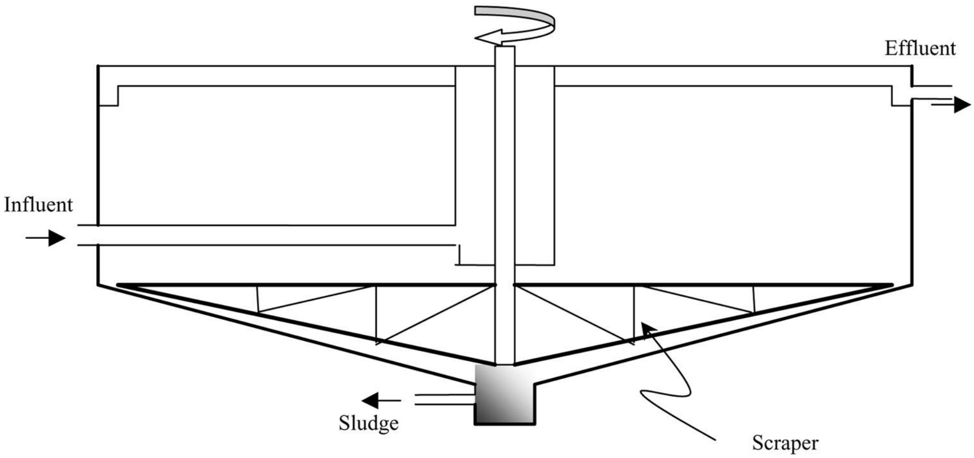

Figure 2.4 Diagram of radial flow sedimentation tank.

generally between 2.5 and 5 m. Detention time is computed by dividing the tank volume by influent flow uniform rate equivalent to the design average daily flow. A detention time of between 1.5 and 2.5 hours is normally provided based on the average rate of wastewater flow. Effluent weir loading is equal to the average daily quantity of overflow divided by the total weir length expressed in m^3/m day.

$$T = \frac{24V}{Q} \tag{2.3}$$

where $T =$ detention time (hour), $Q =$ average daily flow (m^3/day), and $V =$ basin volume (m^3).

Temperature effects are normally not an important consideration in the design. However, in cold climates, the increase in water viscosity at lower temperatures retards particles settling and reduces clarifier performance.

In cases of small or elementary settling basins, the sludge can be removed using an arrangement of perforated piping placed at the bottom of the settling tank [10]. The pipes must be regularly spaced, as shown in Figure 2.5, to be of a diameter wide enough to be cleaned easily in case of clogging. The flow velocities should also be high enough to prevent sedimentation. Flow in individual pipes may be regulated by valves. This configuration is best used after screening and is also found in biological treatment tanks for sludge removal.

Inclined tube separators are an alternative to the above configurations for settling [11]. These separators consist of tilted tubes, which are usually inclined at 45–60°. When a settling particle reaches the wall of the tube or the lower plate, it coalesces with another particle and forms a larger mass, which causes a higher settling rate. A typical configuration for inclined media separators is shown in Figure 2.6.

2.3.3 Flow Equalization

A flow equalization step follows the screening and sedimentation processes and precedes the dissolved air flotation (DAF) unit. Flow equalization is important in reducing hydraulic loading in the waste stream. Equalization facilities consist of a holding tank and pumping equipment designed to reduce the fluctuations of the waste streams. The equalizing tank will store excessive

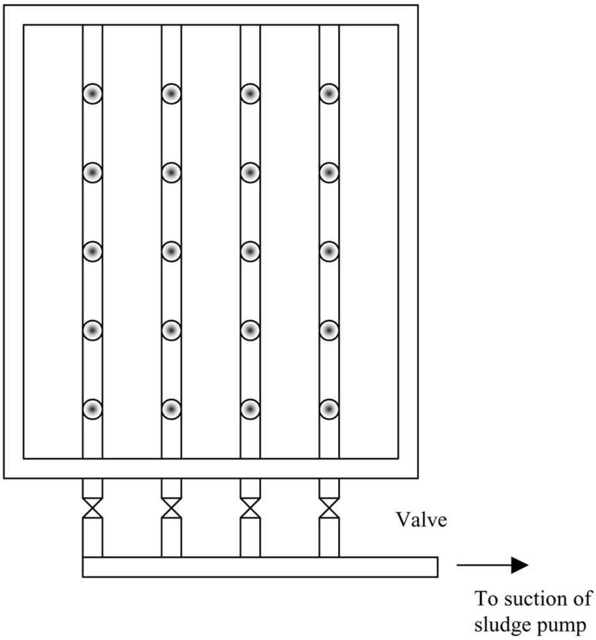

Figure 2.5 Pipe arrangement for sludge removal from settling tanks.

hydraulic flow surges and stabilize the flow rate to a uniform discharge rate over a 24-hour day. The tank is characterized by a varying flow into the tank and a constant flow out.

2.3.4 Separation of Oil and Grease

Seafood-processing wastewaters contain variable amounts of oil and grease, which depend on the process used, the species processed, and the operational procedure. Gravitational separation may be used to remove oil and grease, provided that the oil particles are large enough to float towards the surface and are not emulsified; otherwise, the emulsion must be first broken by pH adjustment. Heat may also be used for breaking the emulsion but it may not be economical unless there is excess steam available. The configurations of gravity separators of oil–water are similar to the inclined tubes separators discussed in the previous section.

2.3.5 Flotation

Flotation is one of the most effective removal systems for suspensions that contain oil and grease. The most common procedure is that of dissolved air flotation (DAF), which is a waste-treatment process in which oil, grease, and other suspended matter are removed from a waste stream. This treatment process has been in use for many years and has been most successful in removing oil from waste streams. Essentially, DAF is a process that uses minute air bubbles to remove the suspended matter from the wastewater stream. The air bubbles attach themselves to a discrete particle, thus effecting a reduction in the specific gravity of the aggregate particle to less than that of water. Reduction of the specific gravity for the aggregate particle causes separation from the carrying liquid in an upward direction. Attachment of the air bubble to the particle induces a vertical rate of rise. The mechanism of operation involves a clarification vessel where

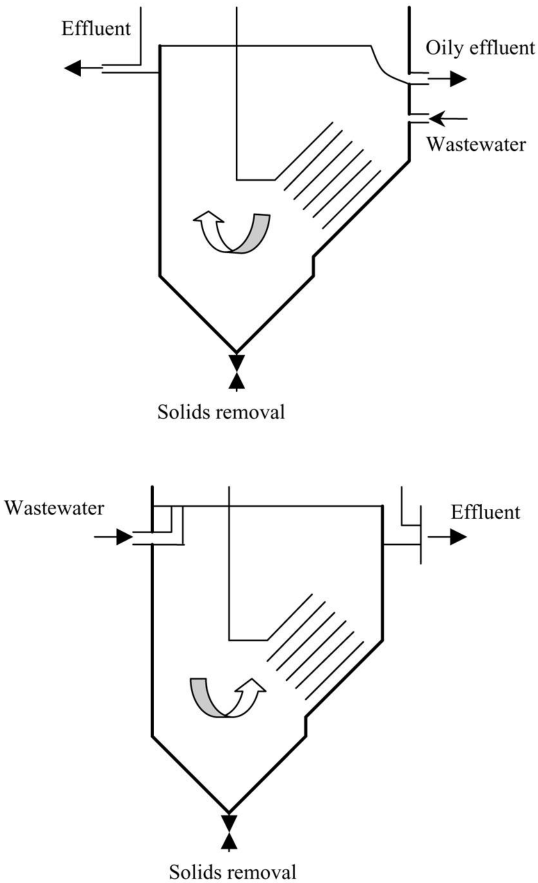

Figure 2.6 Typical configurations for inclined media separators.

the particles are floated to the surface and removed by a skimming device to a collection trough for removal from the system. The raw wastewater is brought in contact with a recycled, clarified effluent that has been pressurized through air injection in a pressure tank. The combined flow stream enters the clarification vessel and the release of pressure causes tiny air bubbles to form and ascend to the surface of the water, carrying the suspended particles with their vertical rise. A schematic diagram of the DAF system is shown in Figure 2.7.

Key factors in the successful operation of DAF units are the maintenance of proper pH (usually between 4.5 and 6, with 5 being most common to minimize protein solubility and break up emulsions), proper flow rates, and the continuous presence of trained operators.

In one case, oil removal was reported to be 90% [12]. In tuna processing wastewaters, the DAF removed 80% of oil and grease and 74.8% of suspended solids in one case, and a second case showed removal efficiencies of 64.3% for oil and grease and 48.2% of suspended solids. The main difference between these last two effluents was the usually lower solids content of the second [13]. However, although DAF systems are considered very effective, they are probably not suitable for small-scale, seafood-processing facilities due to the relatively high cost. It was reported that the estimated operating cost for a DAF system was about US$250,000 in 1977 [14].

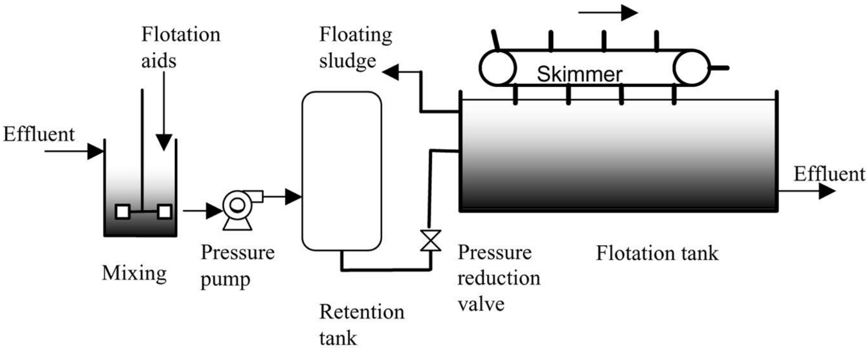

Figure 2.7 Diagram of a dissolved air flotation (DAF) system.

2.4 BIOLOGICAL TREATMENT

To complete the treatment of the seafood-processing wastewaters, the waste stream must be further processed by biological treatment. Biological treatment involves the use of microorganisms to remove dissolved nutrients from a discharge [15]. Organic and nitrogenous compounds in the discharge can serve as nutrients for rapid microbial growth under aerobic, anaerobic, or facultative conditions. The three conditions differ in the way they use oxygen. Aerobic microorganisms require oxygen for their metabolism, whereas anaerobic microorganisms grow in absence of oxygen; the facultative microorganism can proliferate either in absence or presence of oxygen although using different metabolic processes. Most of the microorganisms present in wastewater treatment systems use the organic content of the wastewater as an energy source to grow, and are thus classified as heterotrophes from a nutritional point of view. The population active in a biological wastewater treatment is mixed, complex, and interrelated. In a single aerobic system, members of the genera *Pseudomonas*, *Nocardia*, *Flavobacterium*, *Achromobacter*, and *Zooglea* may be present, together with filamentous organisms. In a well-functioning system, protozoas and rotifers are usually present and are useful in consuming dispersed bacteria or nonsettling particles.

Biological treatment systems can convert approximately one-third of the colloidal and dissolved organic matter into stable endproducts and convert the remaining two-thirds into microbial cells that can be removed through gravity separation. The organic load present is incorporated in part as biomass by the microbial populations, and almost all the rest is liberated gas. Carbon dioxide (CO_2) is produced in aerobic treatments, whereas anaerobic treatments produce both carbon dioxide and methane (CH_4). In seafood-processing wastewaters, the nonbiodegradable portion is very low.

The biological treatment processes used for wastewater treatment are broadly classified as aerobic and anaerobic treatments. Aerobic and facultative microorganisms predominate in aerobic treatments, while only anaerobic microorganisms are used for the anaerobic treatments.

If microorganisms are suspended in the wastewater during biological operation, this is known as a “suspended growth process,” whereas the microorganisms that are attached to a surface over which they grow are said to undergo an “attached growth process.”

Biological treatment systems are most effective when operating continuously 24 hours/day and 365 days/year. Systems that are not operated continuously have reduced efficiency because of changes in nutrient loads to the microbial biomass. Biological treatment systems also generate a consolidated waste stream consisting of excess microbial biomass, which must be properly disposed. Operation and maintenance costs vary with the process used.

The principles and main characteristics of the most common processes used in seafood-processing wastewater treatment are explained in this section.

2.4.1 Aerobic Process

In seafood processing wastewaters, the need for adding nutrients (the most common being nitrogen and phosphorus) seldom occurs, but an adequate provision of oxygen is essential for successful operation. The most common aerobic processes are activated sludge systems, lagoons, trickling filters and rotating disc contactors. The reactions occurring during the aerobic process can be summarized as follows:

$$\text{Organic} + O_2 \longrightarrow \text{cells} + CO_2 + H_2O$$

Apart from economic considerations, several factors influence the choice of a particular aerobic treatment system. The major considerations are: the area availability; the ability to operate intermittently is critical for several seafood industries that do not operate in a continuous fashion or work only seasonally; the skill needed for operation of a particular treatment cannot be neglected; and finally the operating and capital costs are also sometimes decisive. Table 2.2 summarizes these factors when applied to aerobic treatment processes.

The considerations for rotating biological contactors (RBC) systems are similar to those of trickling filters.

Activated Sludge Systems

In an activated sludge treatment system, an acclimatized, mixed, biological growth of microorganisms (sludge) interacts with organic materials in the wastewater in the presence of excess dissolved oxygen and nutrients (nitrogen and phosphorus). The microorganisms convert the soluble organic compounds to carbon dioxide and cellular materials. Oxygen is obtained from applied air, which also maintains adequate mixing. The effluent is settled to separate

Table 2.2 Factors Affecting the Choice of Aerobic Processes

(A) Operating characteristics			
System	Resistance to shock loads of organics or toxics	Sensitivity to intermittent operations	Degree of skill needed
Lagoons	Maximum	Minimum	Minimum
Trickling filters	Moderate	Moderate	Moderate
Activated	Minimum	Maximum	Maximum
(B) Cost considerations			
System	Land needed	Initial costs	Operating costs
Lagoons	Maximum	Minimum	Minimum
Trickling filters	Moderate	Moderate	Moderate
Activated	Minimum	Maximum	Maximum

Source: Ref. 10.

biological solids and a portion of the sludge is recycled; the excess is wasted for further treatment such as dewatering. These systems originated in England in the early 1900s. The layout of a typical activated sludge system is shown in Figure 2.8.

Most of the activated sludge systems utilized in the seafood-processing industry are of the extended aeration types: that is, they combine long aeration times with low applied organic loadings. The detention times are 1 to 2 days. The suspended solids concentrations are maintained at moderate levels to facilitate treatment of the low-strength wastes, which usually have a BOD_5 of less than 800 mg/L.

It is usually necessary to provide primary treatment and flow equalization prior to the activated sludge process, to ensure optimum operation. A BOD_5 and suspended solids removals in the range of 95–98% can be achieved. However, pilot- or laboratory-scale studies are required to determine organic loadings, oxygen requirements, sludge yields, sludge settling rates, and so on, for these high-strength wastes.

In contrast to other food-processing wastewaters, seafood wastes appear to require higher oxygen availability to stabilize them. Whereas dairy, fruit, and vegetable wastes require approximately 1.3 kg of oxygen per kg of BOD_5, seafood wastes may demand as much as 3 kg of oxygen per kg of BOD_5 applied to the extended aeration system [2].

The most common types of activated sludge process are the conventional and the continuous flow stiffed tanks, as shown in Figure 2.8, in which the contents are fully mixed. In the conventional process, the wastewater is circulated along the aeration tank, with the flow being arranged by baffles in plug flow mode. This arrangement demands a maximum amount of oxygen and organic load concentration at the inlet. A typical conventional activated sludge process is shown in Figure 2.9. Unlike the conventional activated sludge process, the inflow streams in the completely mixed process are usually introduced at several points to facilitate the homogeneity of the mixing such that the properties are constant throughout the reactor if the mixing is completed. This configuration is inherently more stable in terms of perturbations because mixing causes dilution of the incoming stream into the tank. In seafood-processing wastewaters the perturbations that may appear are peaks of concentration of organic load or flow peaks. Flow peaks can be damped in the primary treatment tanks. The conventional configurations would require less reactor volume if smooth plug flow could be assured, which usually does not occur.

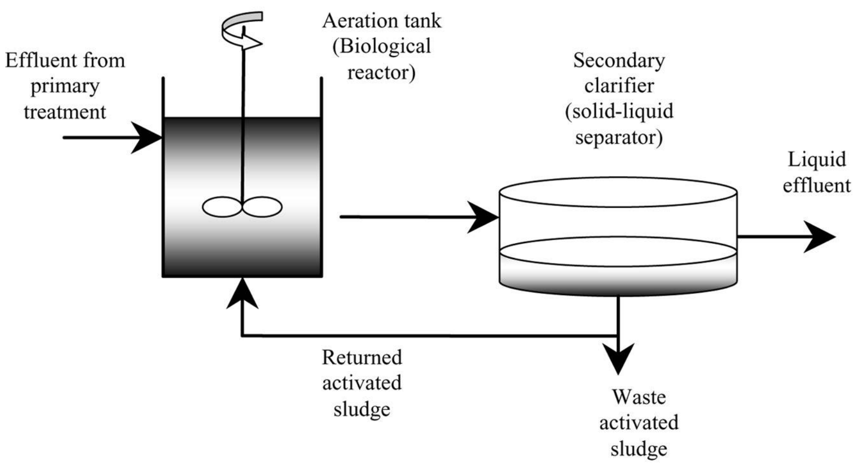

Figure 2.8 Diagram of a simple activated sludge system.

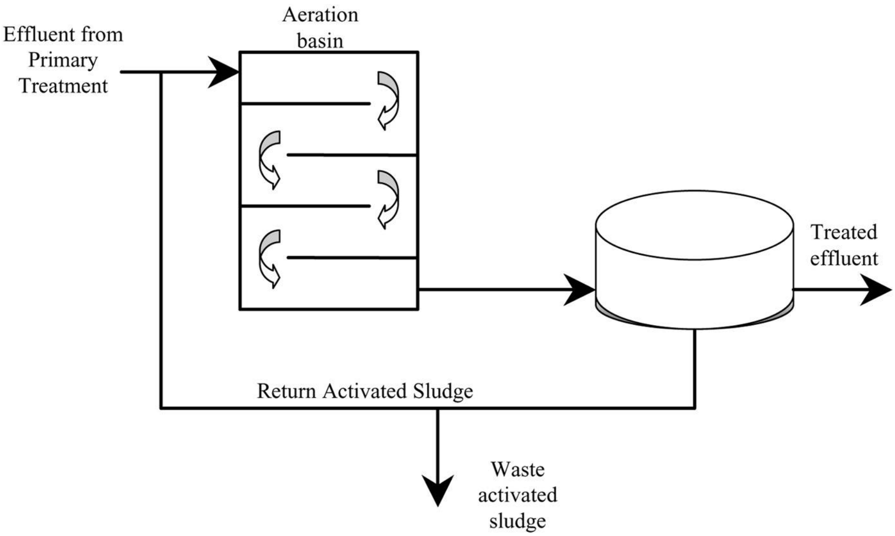

Figure 2.9 Diagram of a conventional activated sludge process.

In activated sludge systems, the cells are separated from the liquid and partially returned to the system; the relatively high concentration of cells then degrades the organic load in a relatively short time. Therefore, there are two different resident times that characterize the systems: one is the hydraulic residence time (θ_H) given by the ratio of reactor volume (V_R) to flow of wastewater (Q_R):

$$\theta_H = \frac{V_R}{Q_R} \tag{2.4}$$

The other is the cell residence time (θ_C), which is given by the ratio of cells present in the reactor to the mass of cells wasted per day. Typical θ_H values are in the order of 3–6 hours, while θ_C fluctuates between 3 and 15 days.

To ensure the optimum operation of the activated sludge process, it is generally necessary to provide primary treatment and flow equalization prior to the activated sludge process. Pilot- or laboratory-scale studies are required to determine organic loadings, oxygen requirements, sludge yields, and sludge settling rates for these high-strength wastes. There are several pieces of information required to design an activated sludge system through the bench-scale or pilot-scale studies:

- BOD_5 removal rate;
- oxygen requirements for the degradation of organic material and the degradation of dead cellular material (endogenous respiration);
- sludge yield, determined from the conservation of soluble organics to cellular material and the influx of inorganic solids in the raw waste;
- solid/liquid separation rate: the final clarifier would be designed to achieve rapid sedimentation of solids, which could be recycled or further treated. A maximum surface settling rate of 16.5 m^3/m^2 day has been suggested for seafood-processing wastes [2].

Typically, 85–95% of organic load removals can be achieved in activated sludge systems. Although used by some large seafood-processing industries that operate on a year-round basis, activated sludge may not be economically justified for small, seasonal seafood processors because of the requirement of a fairly constant supply of wastewater to maintain the microorganisms.

Aerated Lagoons

Aerated lagoons are used where sufficient land is not available for seasonal retention, or land application and economics do not justify an activated sludge system. Efficient biological treatment can be achieved by the use of the aerated lagoon system. It was reported to have removal efficiency of 90–95% of BOD_5 in seafood-processing wastewater treatment [2]. The major difference with respect to activated sludge systems is that the aerated lagoons are basins, normally excavated in earth and operated without solids recycling into the system. The ponds are between 2.4 and 4.6 m deep, with 2–10 days retention and achieve 55–90% reduction in BOD_5. Two types of aerated lagoons are commonly used in seafood-processing wastewater treatment: completely mixed lagoons and facultative lagoons. In the completely mixed lagoon, the concentrations of solids and dissolved oxygen are uniformly maintained and neither the incoming solids nor the biomass of microorganisms settle, whereas in the facultative lagoons, the power input is reduced, causing accumulation of solids in the bottom that undergo anaerobic decomposition, while the upper portions are maintained in an aerobic state (Fig. 2.10).

The major operational difference between these lagoons is the power input, which is in the order of 2.5–6 W/m^3 for aerobic lagoons, while the requirement for facultative lagoons is of the order 0.8–1 W/m^3. Reduction in biological activity can occur when the lagoons are exposed to low temperatures and eventually ice formation. This problem can be partially alleviated by increasing the depth of the basin.

If excavated basins are used for settling, care should be taken to provide a residence time long enough for the solids to settle, and provision should also be made for the accumulation of sludge. There is a very high possibility of offensive odor development due to the decomposition of the settled sludge, and algae might develop in the upper layers causing an increased content of suspended solids in the effluent. Odors can be minimized by using minimum depths of up to 2 m, whereas algae production can be reduced with a hydraulic retention time of fewer than 2 days.

Solids will also accumulate all along the aeration basins in the facultative lagoons and even at corners, or between aeration units in the completely mixed lagoon. These accumulated

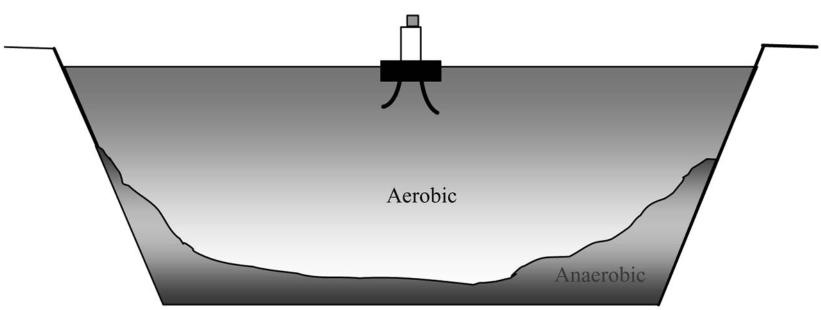

Figure 2.10 Diagram of facultative aerated lagoon.

solids will, on the whole, decompose at the bottom, but since there is always a nonbiodegradable fraction, a permanent deposit will build up. Therefore, periodic removal of these accumulated solids is necessary.

Stabilization/Polishing Ponds

A stabilization/polishing ponds system is commonly used to improve the effluent treated in the aerated lagoon. This system depends on the action of aerobic bacteria on the soluble organics contained in the waste stream. The organic carbon is converted to carbon dioxide and bacterial cells. Algal growth is stimulated by incident sunlight that penetrates to a depth of 1–1.5 m. Photosynthesis produces excess oxygen, which is available for aerobic bacteria; additional oxygen is provided by mass transfer at the air–water interface.

Aerobic stabilization ponds are 0.18–0.9 m deep to optimize algal activity and are usually saturated with dissolved oxygen throughout the depth during daylight hours. The ponds are designed to provide a detention time of 2–20 days, with surface loadings of 5.5–22 g BOD_5/day/m^2 [2]. To eliminate the possibility of shortcircuiting and to permit sedimentation of dead algal and bacterial cells, the ponds usually consist of multiple cell units operated in series. The ponds are constructed with inlet and outlet structures located in positions to minimize shortcircuiting due to wind-induced currents; the dimensions and geometry are designed to maximize mixing. These systems have been reported achieving 80–95% removal of BOD_5 and approximately 80% removal of suspended solids, with most of the effluent solids discharged as algal cells [2].

During winter, the degree of treatment decreases markedly as the temperature decreases and ice cover eliminates algal growth. In regions where ice cover occurs, the lagoons may be equipped with variable depth overflow structures so that processing wastewater flows can be stored during the winter. An alternative method is to provide long retention storage ponds; the wastes can then be treated aerobically during the summer prior to discharge.

Aerobic stabilization ponds are utilized where land is readily available. In regions where soils are permeable, it is often necessary to use plastic, asphaltic, or clay liners to prevent contamination of adjacent groundwater.

Trickling Filters

The trickling filter is one of the most common attached cell (biofilm) processes. Unlike the activated sludge and aerated lagoons processes, which have biomass in suspension, most of the biomass in trickling filters are attached to certain support media over which they grow (Fig. 2.11).

Typical microorganisms present in trickling filters are *Zoogloea*, *Pseudomonas*, *Alcaligenes*, *Flavobacterium*, *Streptomyces*, *Nocardia*, fungi, and protozoa. The crux of the process is that the organic contents of the effluents are degraded by these attached growth populations, which absorb the organic contents from the surrounding water film. Oxygen from the air diffuses through this liquid film and enters the biomass. As the organic matter grows, the biomass layer thickens and some of its inner portions become deprived of oxygen or nutrients and separate from the support media, over which a new layer will start to grow. The separation of biomass occurs in relatively large flocs that settle relatively quickly in the supporting material. Media that can be used are rocks (low-rate filter) or plastic structures (high-rate filter). Denitrification can occur in low-rate filters, while nitrification occurs under high-rate filtration conditions; therefore, effluent recycle may be necessary in high-rate filters.

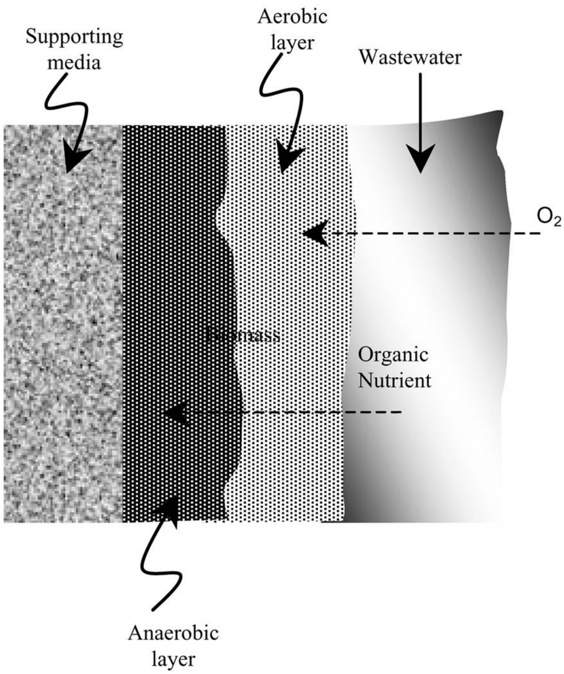

Figure 2.11 Cross-section of an attached growth biomass film.

In order to achieve optimum operation, several design criteria for trickling filters must be followed:

- roughing filters may be loaded at a rate of 4.8 kg $BOD_5/day/m^3$ filter media and achieve BOD_5 reductions of 40–50%;
- high-rate filters achieve BOD_5 reductions of 40–70% at organic loadings of 0.4–4.8 kg/$BOD_5/day/m^3$; and
- standard rate filters are loaded at 0.08–0.4 kg/$BOD_5/day/m^3$ and achieve BOD_5 removals greater than 70% [2].

The trickling filter consists of a circular tank filled with the packing media in depths varying from 1–2.5 m, or 10 m if synthetic packing is used. The bottom of the tank must be constructed rigidly enough to support the packing and designed to collect the treated wastewater, which is either sprayed by regularly spaced nozzles or by rotating distribution arms. The liquid percolates through the packing and the organic load is absorbed and degraded by the biomass while the liquid drains to the bottom to be collected.

With regard to the packing over which the biomass grows, the void fraction and the specific surface area are important features; the first is necessary to ensure a good circulation of air and the second is to accommodate as much biomass as possible to degrade the organic load of the wastewaters. Although more costly initially, synthetic packings have a larger void space, larger specific area, and are lighter than other packing media. Usually, the air circulates naturally, but forced ventilation is used with some high-strength wastewaters. The latter may be used with or without recirculation of the liquid after the settling tank. The need for recirculation is dictated by the strength of the wastewater and the rate of oxygen transfer to the biomass. Typically, recirculation is used when the BOD_5 of the seafood-processing wastewater to be treated exceeds 500 mg/L. The BOD_5 removal efficiency varies with the organic load imposed but usually fluctuates between 45 and 70% for a single-stage filter. Removal efficiencies of up to 90% can be achieved in two stages [4]. A typical unit of a trickling filter is shown in Figure 2.12.

Rotating Biological Contactors (RBC)

Increasingly stringent requirements for the removal of organic and inorganic substances from wastewater have necessitated the development of innovative, cost-effective wastewater treatment alternatives in recent years. The aerobic rotating biological contactor (RBC) is one of the biological processes for the treatment of organic wastewater. It is another type of attached growth process that combines advantages of biological fixed-film (short hydraulic retention time, high biomass concentration, low energy cost, easy operation, and insensitivity to toxic substance shock loads), and partial stir. Therefore, the aerobic RBC reactor is widely employed to treat both domestic and industrial wastewater [16–18]. A schematic diagram of the rotating biological contactor (RBC) unit is shown in Figure 2.13; it consists of closely spaced discs mounted on a common horizontal shaft, partially submerged in a semicircular tank receiving wastewater. When water containing organic waste and nutrients flows though the reactor, microorganisms consume the substrata and grow attached to the discs' surfaces to about 1–4 mm in thickness; excess is torn off the discs by shearing forces and is separated from the liquid in the secondary settling tank. A small portion of the biomass remains suspended in the liquid within the basin and is also responsible in minor part for the organic load removal.

Aeration of the culture is accomplished by two mechanisms. First, when a point on the discs rises above the liquid surface, a thin film of liquid remains attached to it and oxygen is transferred to the film as it passes through air; some amount of air is entrained by the bulk of liquid due to turbulence caused by rotation of discs. Rotation speeds of more than 3 rpm are seldom used because this increases electric power consumption while not sufficiently increasing oxygen transfer. The ratio of surface area of discs to liquid volume is typically 5 L/m^2. For high-strength wastewaters, more than one unit in series (staging) is used.

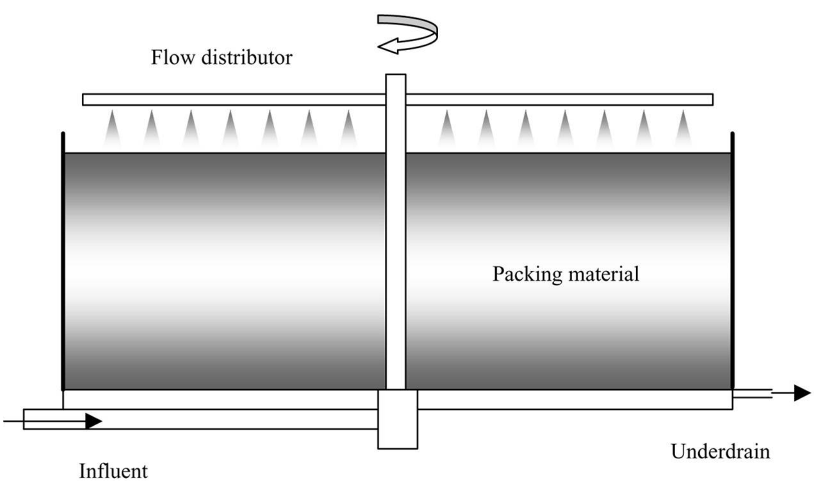

Figure 2.12 Sketch of a trickling filter unit.

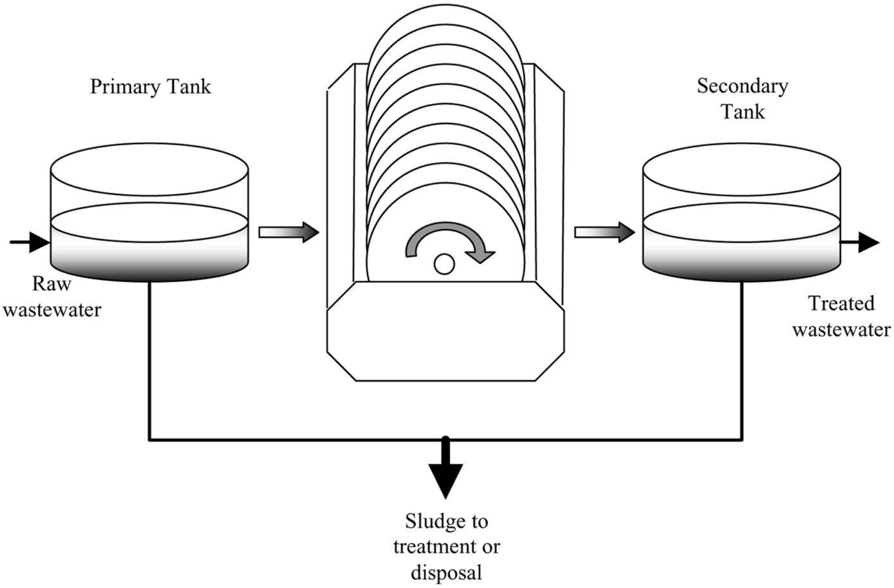

Figure 2.13 Diagram of a rotating biological contactor (RBC) unit.

2.4.2 Anaerobic Treatment

Anaerobic biological treatment has been applied to high BOD or COD waste solutions in a variety of ways. Treatment proceeds with degradation of the organic matter, in suspension or in a solution of continuous flow of gaseous products, mainly methane and carbon dioxide, which constitute most of the reaction products and biomass. Its efficient performance makes it a valuable mechanism for achieving compliance with regulations for contamination of recreational and seafood-producing wastes. Anaerobic treatment is the result of several reactions: the organic load present in the wastewater is first converted to soluble organic material, which in turn is consumed by acid-producing bacteria to produce volatile fatty acids, plus carbon dioxide and hydrogen. The methane-producing bacteria consume these products to produce methane and carbon dioxide. Typical microorganisms used in this methanogenic process are *Metanobacterium*, *Methanobacillus*, *Metanococcus*, and *Methanosarcina*. These processes are reported to be better applied to high-strength wastewaters, for example, blood water or stickwater. The scheme of reactions during anaerobic treatment is summarized in Figure 2.14.

Digestion Systems

Anaerobic digestion facilities have been used for the management of animal slurries for many years, they can treat most easily biodegradable waste products, including everything of organic or vegetable origin. Recent developments in anaerobic digestion technology have allowed the expansion of feedstocks to include municipal solid wastes, biosolids, and organic industrial waste (e.g., seafood-processing wastes). Lawn and garden, or "green" residues, may also be included, but care should be taken to avoid woody materials with high lignin content that requires a much longer decomposition time [19]. The digestion system seems to work best with a feedstock mixture of 15–25% solids. This may necessitate the addition of some liquid,

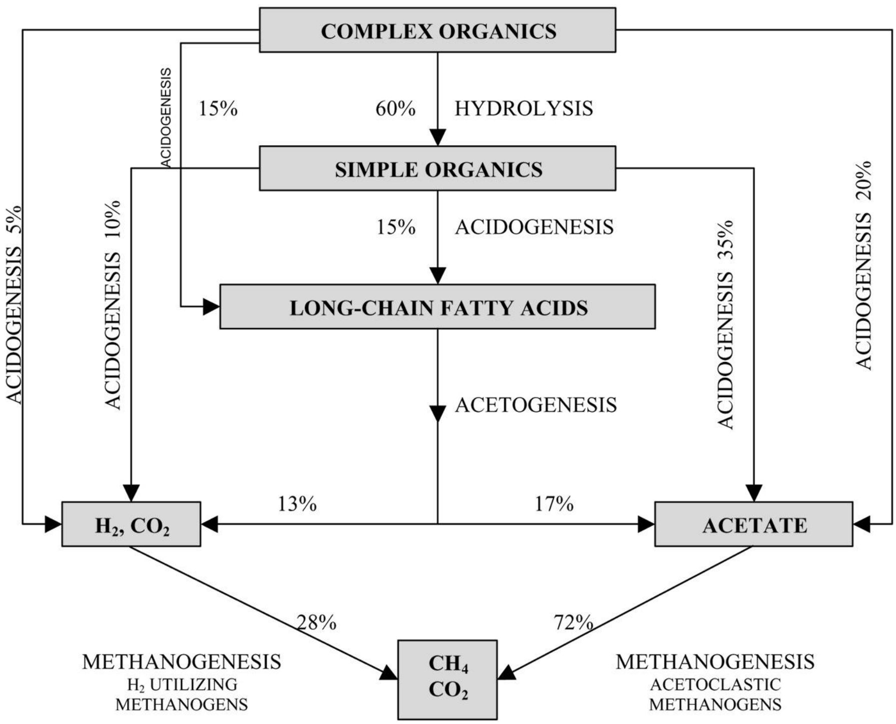

Figure 2.14 Scheme of reactions produced during anaerobic treatment.

providing an opportunity for the treatment of wastewater with high concentrations of organic contaminants. A typical anaerobic system diagram is shown in Figure 2.15.

The flow of anaerobic digestion resembles that of an activated sludge process except that it occurs in the absence of oxygen. Therefore, it is essential to have a good sealing of the digestion tanks since oxygen kills some of the anaerobic bacteria present and presence of air may easily disrupt the process. From the anaerobic digester the effluent proceeds to a degasifier and to a settler from which the wastewater is discharged and the solids are recycled. The need for recycling is attributed to the fact that anaerobic digestion proceeds at a much slower rate than aerobic processes, thereby requiring more time and more biomass to achieve high removal efficiencies. The amount of time required for anaerobic digestion depends upon its composition and the temperature maintained in the digester, because anaerobic processes are also sensitive to temperature. Mesophilic digestion occurs at approximately 35°C, and requires 12–30 days for processing. Thermophilic processes make use of higher temperatures (55°C) to speed up the reaction time to 6–14 days. Mixing the contents is not always necessary, but is generally preferred, as it leads to more efficient digestion by providing uniform conditions in the vessel and speeds up the biological reactions.

Anaerobic processes have been applied in seafood-processing wastewaters, obtaining high removal efficiencies (75–80%) with loads of 3 or 4 kg of COD/m^3 day [20,21].

In total, 60–70% of the gas produced by a balanced and well-functioning system consists of methane, with the rest being mostly carbon dioxide and minor amounts of nitrogen and hydrogen. This biogas is an ideal source of fuel, resulting in low-cost electricity and providing steam for use in the stirring and heating of digestion tanks.

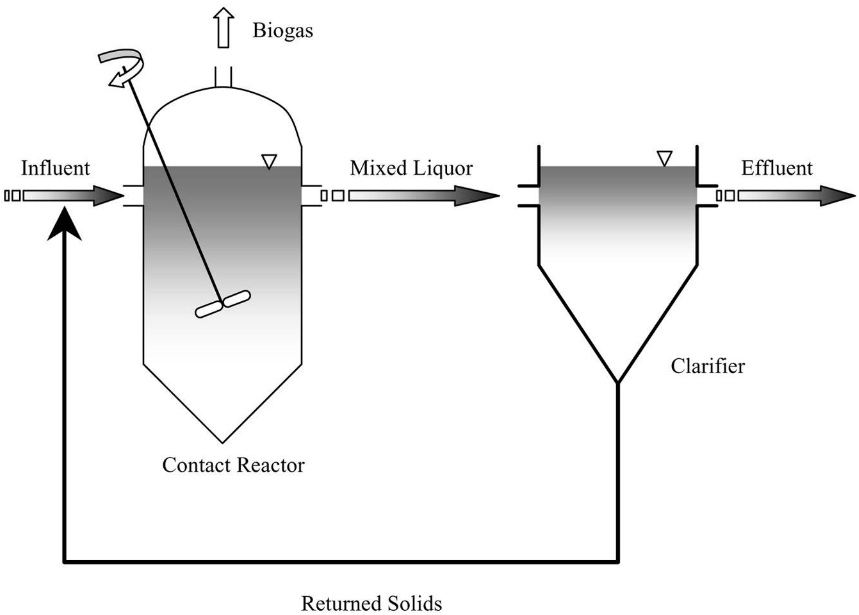

Figure 2.15 Diagram of an anaerobic digestion process.

Imhoff Tanks

The Imhoff tank is a relatively simple anaerobic system that was used to treat wastewater before heated digesters were developed. It is still used for plants of small capacity. The system consists of a two-chamber rectangular tank, usually built partially underground (Fig. 2.16).

Wastewater enters into the upper compartment, which acts as a settling basin while the settled solids are stabilized anaerobically at the lower part. Shortcircuiting of the wastewater can

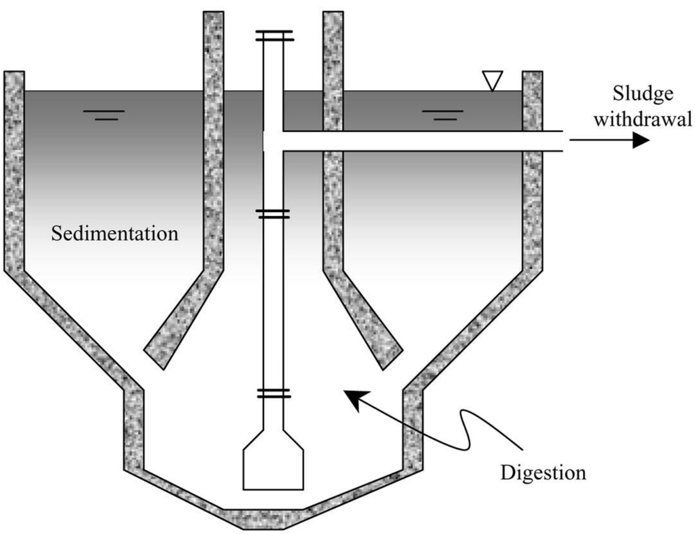

Figure 2.16 An Imhoff tank.

be prevented by using a baffle at the entrance with more than one port for discharge. The lower compartment is generally unheated. The stabilized sludge is removed from the bottom, generally twice a year, to provide ample time for the sludge to stabilize, although the removal frequency is sometimes dictated by the convenience of sludge disposal. In some cases, these tanks are designed with inlets and outlets at both ends, and the wastewater flow is reversed periodically so that the sludge at the bottom accumulates evenly. Although they are simple installations, Imhoff tanks are not without inconveniences; foaming, odor, and scum can form. These typically result when the temperature falls below 15°C and causes a process imbalance in which the bacteria that produce volatile acids predominate and methane production is reduced. This is why in some cases immersed heaters are used during cold weather. Scum forms because the gases that originate during anaerobic digestion are entrapped by the solids, causing the latter to float. This is usually overcome by increasing the depth in the lower chamber. At lower depths, bubbles form at a higher pressure, expand more when rising, and are more likely to escape from the solids. Odor problem is minimal when the two stages of the process of acid formation and gas formation are balanced.

2.5 PHYSICOCHEMICAL TREATMENTS

2.5.1 Coagulation/Flocculation

Coagulation or flocculation tanks are used to improve the treatability of wastewater and to remove grease and scum from wastewater [9]. In coagulation operations, a chemical substance is added to an organic colloidal suspension to destabilize it by reducing forces that keep them apart, that is, to reduce the surface charges responsible for particle repulsions. This reduction in charges is essential for flocculation, which has the purpose of clustering fine matter to facilitate its removal. Particles of larger size are then settled and clarified effluent is obtained. Figure 2.17 illustrates the coagulation/flocculation and settling of a seafood-processing wastewater.

In seafood processing wastewaters, the colloids present are of an organic nature and are stabilized by layers of ions that result in particles with the same surface charge, thereby increasing their mutual repulsion and stabilization of the colloidal suspension. This kind of wastewater may contain appreciable amounts of proteins and microorganisms, which become charged due to the ionization of carboxyl and amino groups or their constituent amino acids.

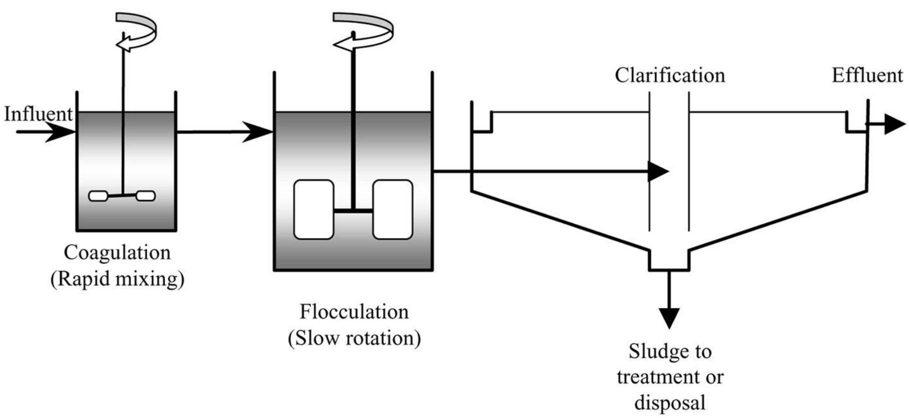

Figure 2.17 Chemical coagulation process.

The oil and grease particles, normally neutral in charge, become charged due to preferential absorption of anions, which are mainly hydroxyl ions.

Several steps are involved in the coagulation process. First, coagulant is added to the effluent, and mixing proceeds rapidly and with high intensity. The purpose is to obtain intimate mixing of the coagulant with the wastewater, thereby increasing the effectiveness of destabilization of particles and initiating coagulation. A second stage follows in which flocculation occurs for a period of up to 30 minutes. In the latter case, the suspension is stirred slowly to increase the possibility of contact between coagulating particles and to facilitate the development of large flocs. These flocs are then transferred to a clarification basin in which they settle and are removed from the bottom while the clarified effluent overflows.

Several substances may be used as coagulants. The pH of several wastewaters of the proteinaceous nature can be adjusted by adding acid or alkali. The addition of acid is more common, resulting in coagulation of the proteins by denaturing them, changing their structural conformation due to the change in their surface charge distribution. Thermal denaturation of proteins can also be used, but due to its high energy demand, it is only advisable if excess steam is available. In fact, the "cooking" of the blood–water in fishmeal plants is basically a thermal coagulation process.

Another commonly used coagulant is polyelectrolyte, which may be further categorized as cationic and anionic coagulants. Cationic polyelectrolytes act as a coagulant by lowering the charge of the wastewater particles, because wastewater particles are negatively charged. Anionic or neutral polyelectrolyte are used as bridges between the already formed particles that interact during the flocculation process, resulting in an increase of floc size.

Since the recovered sludges from coagulation/flocculation processes may sometimes be added to animal feeds, it is advisable to ensure that the coagulant or flocculant used is not toxic.

In seafood-processing wastewaters there are several reports on the use (at both pilot plant and working scale) of inorganic coagulants such as aluminum sulfate, ferric chloride, ferric sulfate, or organic coagulants [22–25].

On the other hand, fish scales are reported to be used effectively as an organic wastewater coagulant [26]. These are dried and ground before being added as coagulant in powder form. Another marine byproduct that can be used as coagulant is a natural polymer derived from chitin, a main constituent of the exoskeletons of crustaceans, which is also known as chitosan.

2.5.2 Electrocoagulation

Electrocoagulation (EC) has also been investigated as a possible means to reduce soluble BOD. It has been demonstrated to reduce organic levels in various food- and fish-processing waste streams [27]. During testing, an electric charge was passed through a spent solution in order to destabilize and coagulate contaminants for easy separation. Initial test results were quickly clarified with a small EC test cell – contaminants coagulated and floated to the top. Analytical test results showed some reduction in BOD_5, but not as much as originally anticipated when the pilot test was conducted. Additional testing was carried out on site on a series of grab samples; however, these runs did not appear to be as effective as originally anticipated. The pH was varied in an attempt to optimize the process, but BOD_5 reductions of only 21–33% were observed. Also, since metal electrodes (aluminum) were used in the process, the presence of metal in the spent solution and separated solids posed a concern for byproduct recovery. Initial capital outlays and anticipated operating costs were not unreasonable (US$140,000 and US$40,000, respectively), but satisfactory BOD_5 reductions could not be achieved easily. It was determined that long retention times would be needed in order to make EC work effectively.

2.5.3 Disinfection

Disinfection of seafood-processing wastewater is a process by which disease-causing organisms are destroyed or rendered inactive. Most disinfection systems work in one of the following four ways: (i) damage to the cell wall, (ii) alteration of cell permeability, (iii) alteration of the colloidal nature of protoplasm, or (iv) inhibition of enzyme activity [9,15].

Disinfection is often accomplished using bactericidal agents. The most common agents are chlorine, ozone (O_3), and ultraviolet (UV) radiation, which are discussed in the following sections.

Chlorination

Chlorination is a process commonly used in both industrial and domestic wastewaters for various reasons. In fisheries effluents, however, its primary purpose is to destroy bacteria or algae, or to inhibit their growth. Usually the effluents are chlorinated just before their final discharge to the receiving water bodies. For this process either chlorine gas or hypochlorite solutions may be used, the latter being easier to handle. In waste solutions, chlorine forms hypochlorous acid, which in turn forms hypochlorite.

$$Cl_2 + H_2O \longrightarrow HOCl + H^+ + Cl^-$$
$$HOCl \longrightarrow H^+ + OCl^-$$

A problem that may occur during chlorination of fisheries effluents is the formation of chloramines. These wastewaters may contain appreciable amounts of ammonia and volatile amines, which react with chlorine to give chloramines, resulting in an increased demand for chlorine to achieve a desired degree of disinfection. The proportions of these products depend on the pH and concentration of ammonia and the organic amines present. Chlorination also runs the risk of developing trihalomethanes, which are known carcinogens. Subsequently, the contact chamber must be cleaned regularly.

The degree of disinfection is attributed to the residual chlorine present in water. A typical plot of the breakpoint chlorination curve with detailed explanation is shown in Figure 2.18.

Initially, the presence of reducing agents reduce an amount of chlorine to chloride and makes the residual chlorine negligible (segment A–B). Further addition of chlorine may result in the formation of chloramines. These appear as residual chlorine but in the form of combined chlorine residual (segment B–C). Once all the ammonia and organic amines have reacted with the added chlorine, additional amounts of chlorine result in the destruction of the chloramines by oxidation, with a decrease in the chlorine residual as a consequence (segment C–D). Once this oxidation is completed, further addition of chlorine results in the appearance of free available chlorine. Point D on the curve is also known as "breakpoint chlorination." The goal in obtaining some free chlorine residual is to achieve disinfection purpose.

Chlorination units consist of a chlorination vessel in which the wastewater and the chlorine are brought into contact. In order to provide sufficient mixing, chlorine systems must have a chlorine contact time of 15–30 minutes, after which it must be dechlorinated prior to discharge. A schematic diagram of the systems is presented in Figure 2.19.

The channels in this contact basin are usually narrow in order to increase the water velocity and, hence, reduce accumulation of solids by settling. However, the space between the channels should allow for easy cleaning. The levels of available chlorine after the breakpoint should comply with the local regulations, which usually vary between 0.2 and 1 mg/L. This value strongly depends on the location of wastewater to be discharged, because residual chlorine in treated wastewater effluents was identified, in some cases, as the main toxicant suppressing

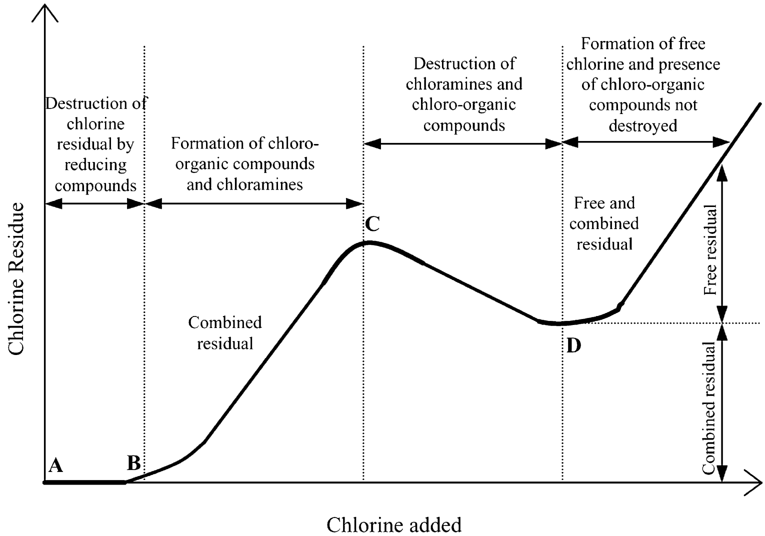

Figure 2.18 Breakpoint chlorinating curve (from Ref. 9).

the diversity, size, and quantity of fish in receiving streams [28]. Additionally, the chlorine dosage needed to achieve the residual effect required varies with the wastewater considered: 2–8 mg/L is common for an effluent from an activated sludge plant, and can be about 40 mg/L in the case of septic wastewater [6,7].

Ozonation

Ozone (O_3) is a strong oxidizing agent that has been used for disinfection due to its bactericidal properties and its potential for removal of viruses. It is produced by discharging air or oxygen across a narrow gap with application of a high voltage. An ozonation system is presented in Figure 2.20.

Ozonation has been used to treat a variety of wastewater streams and appears to be most effective when treating more dilute types of wastes [29]. It is a desirable application as a

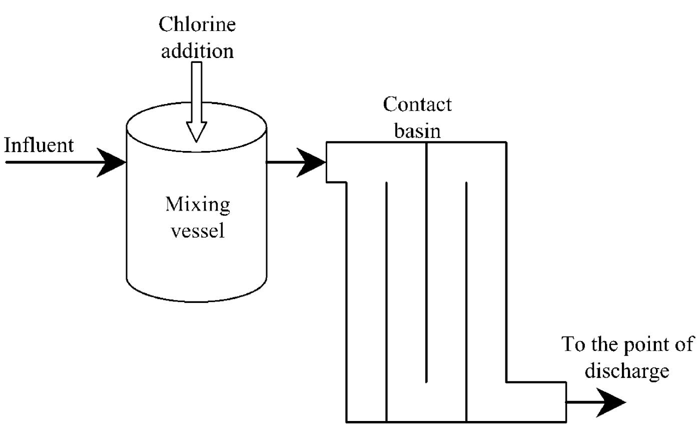

Figure 2.19 Schematics of a chlorination system.

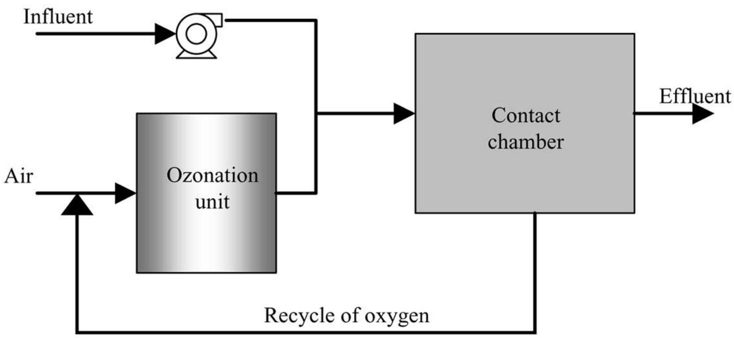

Figure 2.20 Simplified diagram of an ozonation system.

polishing step for some seafood-processing wastewaters, such as from squid-processing operations, which is fairly concentrated [30].

Ozone reverts to oxygen when it has been added and reacted, thus increasing somewhat the dissolved oxygen level of the effluent to be discharged, which is beneficial to the receiving water stream. Contact tanks are usually closed to recirculate the oxygen-enriched air to the ozonation unit. Advantages of ozonation over chlorination are that it does not produce dissolved solids and is affected neither by ammonia compounds present nor by the pH value of the effluent. On the other hand, ozonation has been used to oxidize ammonia and nitrites presented in fish culture facilities [31].

Ozonation also has limitations. Because ozone's volatility does not allow it to be transported, this system requires ozone to be generated onsite, which requires expensive equipment. Although much less used than chlorination in fisheries wastewaters, ozonation systems have been installed in particular in discharges to sensitive water bodies [4,32,33].

Ultraviolet (UV) Radiation

Disinfection can also be accomplished by using ultraviolet (UV) radiation as a disinfection agent. UV radiation disinfects by penetrating the cell wall of pathogens with UV light and completely destroying the cell and/or rendering it unable to reproduce.

However, a UV radiation system might have only limited value to seafood-processing wastewater without adequate TSS removal, because the effectiveness decreases when solids in the discharge block the light. This system also requires expensive equipment with high maintenance [34]. Nevertheless, UV radiation and other nontraditional disinfection processes are gaining acceptance due to stricter regulations on the amount of residual chlorine levels in discharged wastewaters.

2.6 LAND DISPOSAL OF WASTEWATER

Land application of wastewater is a low capital and operating cost method for treating seafood-processing wastes, provided that sufficient land with suitable characteristics is available. The ultimate disposal of wastewater applied to land is by one of the following methods:

- percolation to groundwater;
- overland runoff to surface streams;
- evaporation and evapo-transpiration to the atmosphere.

Generally, several methods are used for land application, including irrigation, surface ponding, groundwater recharge by injection wells, and subsurface percolation. Although each of these methods may be used in particular circumstances for specific seafood-processing waste streams, the irrigation method is most frequently used. Irrigation processes may be further divided into four subcategories according to the rates of application and ultimate disposal of liquid. These are overland flow, normal irrigation, high-rate irrigation, and infiltration — percolation.

Two types of land application techniques seem to be most efficient, namely infiltration and overland flow. As these land application techniques are used, the processor must be cognizant of potential harmful effects of the pollutants on the vegetation, soil, surface and groundwaters. On the other hand, in selecting a land application technique one must be aware of several factors such as wastewater quality, climate, soil, geography, topography, land availability, and return flow quality.

The treatability of seafood-processing wastewater by land application has been shown to be excellent for both infiltration and overland flow systems [2]. With respect to organic carbon removal, both systems have achieved pollutant removal efficiencies of approximately 98 and 84%, respectively. The advantage of higher efficiency obtained with the infiltration system is offset somewhat by the more expensive and complicated distribution system involved. Moreover, the overland flow system is less likely to pollute potable water supplies.

Nitrogen removal is found to be slightly more effective with infiltration land application when compared to overland flow application. However, the infiltration type of application has been shown to be quite effective for phosphorus and grease removal, and thus offers a definite advantage over the overland flow if phosphorus and grease removal are the prime factors. [One factor that may negate this advantage is that soil conditions are not favorable for phosphorus and grease removal and chemical treatment is required.]

Irrigation is a treatment process that consists of a number of segments:

- aerobic bacterial degradation of the deposited suspended materials and evaporation of water and concentration of soluble salts;
- filtration of small particles through the soil cover, and biological degradation of entrapped organics in the soil by aerobic and anaerobic bacteria;
- adsorption of organics on soil particles and uptake of nitrogen and phosphorus by plants and soil microorganisms;
- uptake of liquid wastes and transpiration by plants;
- percolation of water to groundwater.

The importance of these processes depends on the rate of application of waste, the characteristics of the waste, the characteristics of soil and substrata, and the type of cover crop grown on the land.

2.6.1 Loading Rates

Application rates should be determined by pilot plant testing for each particular location. The rate depends on whether irrigation techniques are to be used for roughing treatment or as an ultimate disposal method.

This method has both hydraulic and organic loading constraints for the ultimate disposal of effluent. If the maximum recommended hydraulic loading is exceeded, the surface runoff would increase. Should the specified organic loading be exceeded, anaerobic conditions could develop with resulting decrease in BOD_5 removal and the development of odor problem. The average applied loadings of organic suspended solids is approximately 8 g/m^2; however, loadings up to

22 g/m^2 have also been applied successfully [2]. A resting period between applications is important to ensure survival of the aerobic bacteria. The spray field is usually laid out in sections such that resting periods of 4–10 days can be achieved.

2.6.2 Potential Problems in Land Application with Seafood-Processing Wastewater

Two potential problems may be encountered with land application of seafood-processing wastewaters: the presence of disease-producing bacteria and unfavorable sodium absorption ratios of the soil. A key to minimizing the risk of spreading disease-producing bacteria can be accomplished by using low-pressure wastewater distribution systems to reduce the aerosol drift of the water spray. With respect to unfavorable sodium absorption ratios associated with the soil type, the seafood processor should be aware that clay-containing soils will cause the most serious sodium absorption problem. Sandy soils do not appear to be affected by unfavorable sodium absorption ratios and seem to be the best suited for accepting the high sodium chloride content found in most meat packing plant wastewaters.

As seafood-processing plant wastewaters are applied to land, certain types of grasses have been found to be compatible with these wastewaters. These are Bermuda NK-32, Kentucky-31 Tall Fescue, Jose Wheatgrass, and Blue Panicum [2]. In addition, it was reported that the southwestern coast of the United States, with its arid climate, mild winters, and vast available land areas, presents ideal conditions for land application treatment systems.

In some cases, the use of land application systems by today's seafood processors is feasible. However, in many cases, land disposal of seafood-processing wastes must be ruled out as a treatment alternative. Coastal topographic and soil characteristics, along with high costs of coastal property are the two major factors limiting the use of land application systems for treating seafood-processing wastes.

2.7 GENERAL SEAFOOD-PROCESSING PLANT SCHEMES

Seafood processing involves the capture and preparation of fish, shellfish, marine plants and animals, as well as byproducts such as fish meal and fish oil. The processes used in the seafood industry generally include harvesting, storing, receiving, eviscerating, precooking, picking or cleaning, preserving, and packaging [2]. Figure 2.21 shows a general process flow diagram for seafood processing. It is a summary of the major processes common to most seafood processing operations; however, the actual process will vary depending on the product and the species being processed.

There are several sources that produce wastewater, including:

- fish storage and transport;
- fish cleaning;
- fish freezing and thawing;
- preparation of brines;
- equipment sprays;
- offal transport;
- cooling water;
- steam generation;
- equipment and floor cleaning.

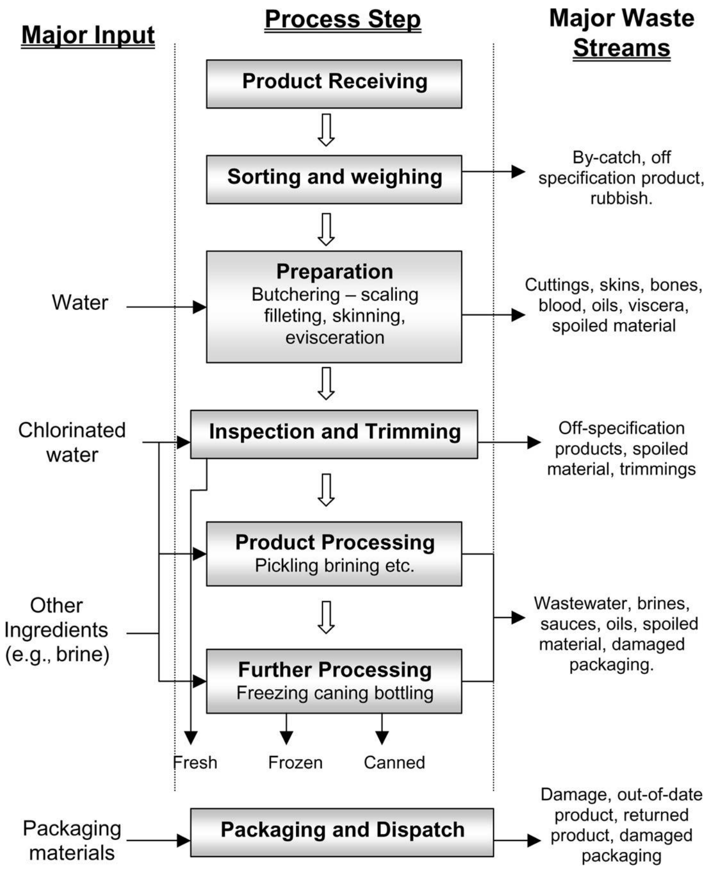

Figure 2.21 General process flow diagram for seafood processing operations.

Organic material in the wastewater is produced in the majority of these processes. However, most of it originates from the butchering process, which generally produces organic material such as blood and gut materials. The volume and quality of wastewater in each area is highly dependent on the products or species being processed and the production processes used.

Most seafood processors have a high baseline water use for cleaning plant and equipment. Therefore, water use per unit product decreases rapidly as production volume increases. Reducing wastewater volumes tends to have a significant impact on reducing organic loads as these strategies are typically associated with reduced product contact and better segregation of high-strength streams.

Water consumption in seafood-processing operations has traditionally been high to achieve effective sanitation. Industry literature indicates that water use varies widely throughout the sector, from 5–30 L/kg of product. Several factors affect water use, including the type of product processed, the scale of the operation, the process used, and the level of water minimization in place [1]. General cleaning contributes significantly to total water demand so smaller scale sites tend to have significantly higher water use per unit of production. Thawing operations can also account for up to 50% of the wastewater generated. A figure for water use of around 5–10 L/kg is typical of large operations with dedicated, automated, or semi-automated equipment that have implemented water minimization practises.

2.8 ECONOMIC CONSIDERATIONS OF SEAFOOD-PROCESSING WASTEWATER TREATMENT

Economic considerations are always the most important parameters that influence the final decision as to which process should be chosen for wastewater treatment. In order to estimate cost, data from the wastewater characterization should be available together with the design parameters for alternative processes and the associated costs. Costs related to these alternative processes and information on the quality of effluent should also be obtained prior to cost estimation in compliance with local regulations.

During the design phase of a wastewater treatment plant, different process alternatives and operating strategies could be evaluated by several methods. This cost evaluation can be achieved by calculating a cost index using commercially available software packages [36,37]. Nevertheless, actual cost indices are often restrictive, since only investment or specific operating costs are considered. Moreover, time-varying wastewater characteristics are not directly taken into account but rather through the application of large safety factors. Finally, the implementation of adequate control strategies such as a real-time control is rarely investigated despite the potential benefits [38,39]. In order to avoid these problems, a concept of MoSS-CC (Model-based Simulation System for Cost Calculation) was introduced by Gillot et al. [40], which is a modeling and simulation tool aimed at integrating the calculation of investment and fixed and variable operating costs of a wastewater treatment plant. This tool helps produce a holistic economic evaluation of a wastewater treatment plant over its life cycles.

2.8.1 Preliminary Costs of a Wastewater Treatment Plant

Several methods may be used to assess the preliminary costs of a wastewater treatment plant to facilitate a choice between different alternatives in the early phase of a process design. One method is cost functions [41–45]. Examples of different investment and operating cost functions are presented in Tables 2.3, 2.4, and 2.5. These cost functions were developed for the MoSS-CC modeling tool.

Another method was developed by EPA to estimate the construction costs for the most common unitary processes of wastewater treatment, as presented in Table 2.6. This was developed for municipal sewage treatment and may not be entirely applicable for small wastewater treatment plants, but it is useful for preliminary estimation and comparison among alternatives [4].

2.8.2 Cost of Operation and Maintenance

Several main factors influence the costs of operation and maintenance, including energy costs, labor costs, material costs, chemical costs, and cost of transportation of sludges for final disposal and discharge of treated wastewater. The relative importance of these items varies significantly depending on the location, the quality of the effluent discharged, and on the specific characteristics of the wastewater being treated [4].

The total operating cost of a wastewater treatment plant may be related to global plant parameters (e.g., average flow rate, population equivalent), generally through power laws [46–48]. However, such relationships apply to the average performance of plants and often suffer from a high uncertainty, unless very similar plant configurations are considered [40].

Table 2.3 Examples of Investment Cost Functions

Unit	Item	Cost function	Parameter	Parameter range	Reference	Cost unit
Influent pumping station	Concrete	$2334Q^{0.637}$	Q = flow rate (m^3/hour)	250–4000	45	Euro of 1998
	Screws	$2123Q^{0.540}$				
	Screening	$3090Q^{0.349}$				
Any unit	Excavation	$2.9(\pi/4D^2H)$	D = diameter (m)	Not defined	44	Can$ of 1995
	Compaction	$24.1 \times 0.4(\pi/4D^2)$	H = height (m)			
	Concrete base	$713.9 \times 0.5(\pi/4D^2)$				
	Concrete wall	$933.6 \times 0.5\pi DH$				
Oxidation ditch	Concrete	$10304V^{0.477}$	V = volume (m^3)	1100–7700	45	Euro of 1998
	Electromech.[a]	$8590OC^{0.433}$	OC = oxygen capacity (kgO_2/hour)			
Settler	Concrete	$2630A^{0.678}$	A = area (m^2)	175–1250	45	Euro of 1998
	Electromech.[a]	$6338A^{0.325}$				
	Concrete	$150(A/400)^{0.56}$	A	60–400	41, 42	Can$*1000 of 1990
		$150(A/400)^{1.45}$		400–800		
	Electromech.[a]	$60(A/220)^{0.62}$		60–7000		
Sludge pump	Electromech.[a]	$9870IQ^{0.53}$	Q, I = Engin. Index[b]	Not defined	52	US$ of 1971
	Electromech.[a]	$5038Q^{0.304}$	Q	35–2340	45	Euro of 1998

[a] Electromech. = electromechanical equipment; [b] Engineering News Record Index = index used to update costs in United States.
Source: Ref. 40

Table 2.4 Examples of Fixed Operating Cost Functions

Cost item	Cost function			Reference
	Formula	Symbols	Units	
Normal O&M	$L = U_c PE$	L = labor U_c = unit cost PE = population equivalent	man-hour/year man-hour/year/PE —	53
Clarifier mechanism	$P = \theta A^b$	P = power θ, b = constant A = area	kW — m^2	44
Mixers	$P = P_s V$	P = power P_s = specific power V = volume	kW kW/m^3 m^3	53
Small equipment (supplies, spare parts . . .)	$C = U_c PE$	C = cost U_c = unit cost PE = population equivalent	Euro/year Euro/year/PE —	5
Analyses	$C = U_c PE$	C = cost U_c = unit cost PE = population equivalent	Euro/year Euro/year/PE —	

Source: Ref. 40

In terms of cost functions evaluations, some possible models in generic form for the fixed and variable operation costs are illustrated in Tables 2.4 and 2.5, respectively.

Capital Costs

These comprise mainly the unit construction costs, the land costs, the cost of the treatment units, and the cost of engineering, administration, and contingencies. The location should be carefully evaluated in each case because it affects the capital costs more than the operating costs [4]. When comparing different alternatives, special attention should be paid to the time and space scales chosen [38], since it may influence the choice of the implemented cost functions [49]. At best, an overall plant evaluation over the life span of the plant should be conducted [40].

Estimation of Total Costs

The total cost of a plant is normally determined by using the present worth method [50]. All annual operating costs for each process are converted into their corresponding present value and added to the investment cost of each process to yield the net present value (NPV). The net present value of a plant over a period of n years can be determined as:

$$\mathrm{NPV} = \sum_{k=1}^{N} IC_k + \left[\frac{1-(1+i)^{-n}}{i}\right] \sum_{k=1}^{N} OC_k \tag{2.5}$$

Table 2.5 Example of Variable Operating Cost Functions

Cost item	Cost function			Reference
	Formula	Symbols	Units	
Pumping power	$P = Qwh/\eta$	Q = flow rate P = power w = specific liquid weight h = dynamic head η = pump efficiency	m^3/s kW N/m^3 m^3/s —	54
Aeration power (fine bubble aeration)	$q_{air} = f(K_L a_f)$ $P = f(q_{air})$	q_{air} = air flow rate P = power $K_L a_f$ = oxygen transfer coefficient in field conditions	Nm^3/hour kW 1/hour	53, 55
Sludge thickening dewatering and disposal	$C = U_c TSS$	C = cost U_c = unit cost TSS = excess sludge	Euro/year Euro/t TSS t	5
Chemicals consumption	$C = U_c C_n$	C = cost U_c = unit cost C_n = consumption	Euro/year Euro/kg kg	40
Effluent taxes (organic matter and nutrient)	$L = U_c^* (k_{org} \cdot N_{org} + k_{nut} \cdot N_{nut})$	U_c = unit cost $N_{org} = f(Q, \text{BOD, TSS, COD})$ $N_{nut} = f(Q, N, P)$	Euro/unit	38

Source: Ref. 40.

where IC_k represents the investment cost of a unit k, and OC_k the operating cost, i is the interest rate, and N is the number of units. The results could also be expressed as equivalent annual worth (AW):

$$\mathrm{AW} = \frac{i(1+i)^n}{(1+i)^n - 1} \sum_{k=1}^{N} IC_k + \sum_{k=1}^{N} OC_k \qquad (2.6)$$

For small wastewater treatments plants, an initial estimate of the total cost can be obtained from the cost of a similar plant with a different capacity, a relationship derived from costs relationships in chemical industries. The cost of plants of different sizes is related to the ratio of their capacity raised to the 0.6 power [4]:

$$\text{Capital}_2 = \text{Capital}_1 \times \left(\frac{\text{Capacity}_2}{\text{Capacity}_1}\right)^{0.6} \qquad (2.7)$$

where $\text{Capital}_{1,2}$ = capital costs of plants 1 and 2, and $\text{Capacity}_{1,2}$ = capacity of plants 1 and 2.

The operation and maintenance costs can be estimated by a similar formula:

$$\text{OM}_2 = \text{OM}_1 \times \left(\frac{\text{Capacity}_2}{\text{Capacity}_1}\right)^{0.85} \qquad (2.8)$$

where $\text{OM}_{1,2}$ = operation and maintenance costs of plants 1 and 2, $\text{Capacity}_{1,2}$ = capacity of plants 1 and 2.

Table 2.6 Construction Costs for Selected Unitary Operations of Wastewater Treatment

Liquid stream	Correlation
Preliminary treatment	$C = 5.79 \times 10^4 \times Q^{1.17}$
Flow equalization	$C = 1.09 \times 10^5 \times Q^{0.49}$
Primary sedimentation	$C = 1.09 \times 10^5 \times Q^{1.04}$
Activated sludge	$C = 2.27 \times 10^5 \times Q^{0.17}$
Rotating biological contactor	$C = 3.19 \times 10^5 \times Q^{0.92}$
Chemical addition	$C = 2.36 \times 10^4 \times Q^{1.68}$
Stabilization pond	$C = 9.05 \times 10^5 \times Q^{1.27}$
Aerated lagoon	$C = 3.35 \times 10^5 \times Q^{1.13}$
Chlorination	$C = 5.27 \times 10^4 \times Q^{0.97}$
Solids stream	**Correlation**
Sludge handling	$C = 4.26 \times 10^4 \times Q^{1.36}$
Aerobic digestion	$C = 1.47 \times 10^5 \times Q^{1.14}$
Anaerobic digestion	$C = 1.12 \times 10^5 \times Q^{1.12}$
Incineration	$C = 8.77 \times 10^4 \times Q^{1.33}$

C represents the cost in USD and Q represents the flow rate of the wastewater to be treated.
Source: EPA, 1978.

An alternative procedure for developing cost models for wastewater treatment systems includes the preparation of kinetic models for the possible treatment alternatives, in terms of area and flow rates at various treatment efficiencies, followed by the computation of mechanical and electrical equipment, as well as the operation and maintenance costs as a function of the flow rates [51].

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