

13

Water and Waste Treatment

R. Andrew Wilbey

13.1

Introduction

Most food manufacturers use much more water than the ingredients or raw materials that they are processing. While some water may be used as an ingredient, the greater use will be for cleaning of raw materials, plant cleaning, cooling water and boiler water feed, each use potentially requiring water to a different specification.

This, in turn, creates similar quantities of used water containing variable concentrations of food components, cleaning chemicals, biocides and boiler treatment chemicals. Small enterprises may find it preferable to discharge their waste to the municipal sewage system, though larger process plants normally need to carry out either partial or complete treatment of their trade effluent.

13.2

Fresh Water

Water quality requirements in food processing will vary from product to product. Extreme cases of product-led specifications are to be found with beer and whisky production. For English beer production, a very hard water is needed and product waters are 'Burtonised', i.e. hardened by the addition of salts to approximate the composition of ground water from Burton on Trent. At the other extreme, water used in Scotch malt whisky production is very soft and may contain soluble organic compounds from the peaty highland soils.

Thus the water quality in an area, which is largely determined by its geology, can be a historical determinant of the development of specific sectors within the food industry. Modern water treatment can overcome this constraint by the introduction of a range of physical and chemical treatments, which will be adjusted to the source and end use of the water. These treatments must cope with suspended matter, from trees in floodwater at one extreme to grit and microor-

ganisms at the other, plus dissolved minerals, gases and organic compounds that may give rise to colour, taste and odour problems in the final product.

While most food processors draw their water from the municipal supply and need to carry out very little treatment themselves, some will be required to provide part or all of their water from an untreated source.

13.2.1

Primary Treatment

Surface waters, whether drawn from rivers or lakes, are assumed to contain large suspended matter so intakes must be of robust construction and located away from direct flow so that collision damage may be avoided. The intakes would typically be faced with 15–25 mm vertical mild steel bars, gap width 25–75 mm. Flow through the intake should be $<0.6 \text{ m s}^{-1}$, ideally $<0.15 \text{ m s}^{-1}$, which would minimise drawing in silt [1]. Where ice formation is expected, the intakes must be in sufficiently deep water to permit adequate flow in cold weather despite the surface being frozen over.

Incoming water should then be passed through an intermediate filter, typically with an aperture of 5–10 mm (sometimes down to 1 mm), to remove the smaller debris. Drum or travelling band screens are often used as frequent back washing is needed to prevent blockage. Flow through the screens should be at $<0.15 \text{ m s}^{-1}$. With ground water sources, there should be little suspended matter and only light-duty intermediate screening is needed. The screened water should then be pumped to the treatment plant, the velocity in the pipeline being $\geq 1 \text{ m s}^{-1}$ to avoid deposition in the pipe.

Sedimentation may be employed if there are significant quantities of suspended matter in the water, the process being described by the Stokes Law equation:

$$v = \frac{d^2 g (\rho_s - \rho)}{18\mu} \quad (13.1)$$

where v is the velocity of the particle, d is the equivalent diameter of the particle, g is the acceleration due to gravity, ρ_s is the density of the suspended particle, ρ is the density of the water and μ is the viscosity of the water at the prevailing temperature.

Some water treatment systems use a simple, upflow sedimentation basin for pretreatment. In this case the throughput or surface overflow rate, Q , is given by:

$$Q = uA \quad (13.2)$$

Where u is the upward velocity of the water and A is the cross-sectional area.

Providing that $u < v$, then sedimentation will occur. Brownian motion will prevent very small particles ($\leq 1 \mu\text{m}$ diam.) from separating.

Water that has undergone this primary treatment is adequate for cooling refrigeration plant, for example ammonia compressors, where there is no risk of contact with foodstuffs. For other uses, further treatment is required.

13.2.2

Aeration

Some ground waters may contain gases and volatile organic compounds that could give rise to taints, off flavours and other problems. For instance, while hydrogen sulphide may be regarded as a curiosity in spa waters, it is considered objectionable in potable water. Fresh water may be treated by aeration, mainly using either waterfall aerators or diffusion/bubble aerators. The transfer of a volatile substance to or from water is dependent upon:

- (a) characteristics of that compound
- (b) temperature
- (c) gas transfer resistance
- (d) partial pressures of the gases in the aeration atmosphere
- (e) turbulence in the gas and liquid phases
- (f) time of exposure.

Henry's Law for sparingly soluble gases states that the weight of a gas dissolved by a definite volume of liquid at constant temperature is directly proportional to the pressure. However, there is seldom time for equilibrium to be achieved, so the extent of the interchange will depend on the gas transfer that has occurred in the interfacial film, the diffusion process being described by Fick's First Law. Reducing the bubble size will considerably increase aeration efficiency. With very high surface:volume ratios, such as with a spray nozzle, an exposure time of 2 s may be adequate, while an air bubble in a basin aerator may need to have a contact time of at least 10 s [2]. All aeration systems must be well ventilated, not only to maximise efficiency but also to avoid safety risks, e.g. explosion with methane, asphyxia with carbon dioxide and poisoning with hydrogen sulphide. Hydrogen sulphide can be difficult to remove from water as it ionises, the anions being nonvolatile. However, removal may be accelerated by enriching the atmosphere to over 10% carbon dioxide in order to lower the pH and thus reduce ionisation. In aerating such waters, there is the risk that oxygen could also oxidise the hydrogen sulphide, giving colloidal sulphur which is difficult to remove.

13.2.3

Coagulation, Flocculation and Clarification

Particles of about 1 μm in size, including microbes, are maintained in suspension by Brownian motion. Many particles are also stabilised by their net negative charge within the normal pH range of water. In soft waters, colloidal dispersions of humic and fulvic acids may give rise to an undesirable 'peat stain'. This

suspended matter may be destabilised by addition of salts, sometimes combined with alkali to raise the pH of the water. Trivalent are more effective than divalent cations, which are much more effective than monovalent cations.

Coagulant dosing may be preceded by injection with ozone or hypochlorite to oxidise organic compounds as well as to reduce the microbial loading.

Aluminium sulphate is effective as a coagulant over a pH range of 5.5–7.5, but its popularity in the UK has declined since an accident at Camelford, Cornwall, in 1988. Ferric chloride or ferric sulphate, with an effective pH range of 5.0–8.5, are now more commonly used in the UK. The salts may be used in conjunction with a cationic polymer. Where necessary, lime may be added prior to coagulant addition, allowing about 10 s for mixing, though the pH correction and coagulant dosing may be carried out at the same time if high energy mixing is employed. A wide range of mixing systems has been used, including air injection which, although aiding aeration of the water, can cause scum problems. The principle is that rapid mixing of the chemicals enables a homogeneous mixing so that aggregation of the colloidal particles to form flocs can then progress under low-shear conditions, followed by settlement [3].

Inorganic cations, for example Pb, As, Se, may also be removed from the water, the extent depending on the pH, coagulant and oxidation state of the cation.

Clarification of the water is achieved by allowing the flocs to settle out in settling basins, which may be rectangular or circular and be of downflow or upflow contact design. Figure 13.1 illustrates the principles of a circular clarifier basin.

The pH correcting and coagulating agents may be dosed into the feed line to the clarifier or at the discharge into the basin. In the latter case, higher shear mixing would be needed. Coagulant dosing may be accompanied by the addition of water-softening agents, such as calcium hydroxide and sodium carbonate

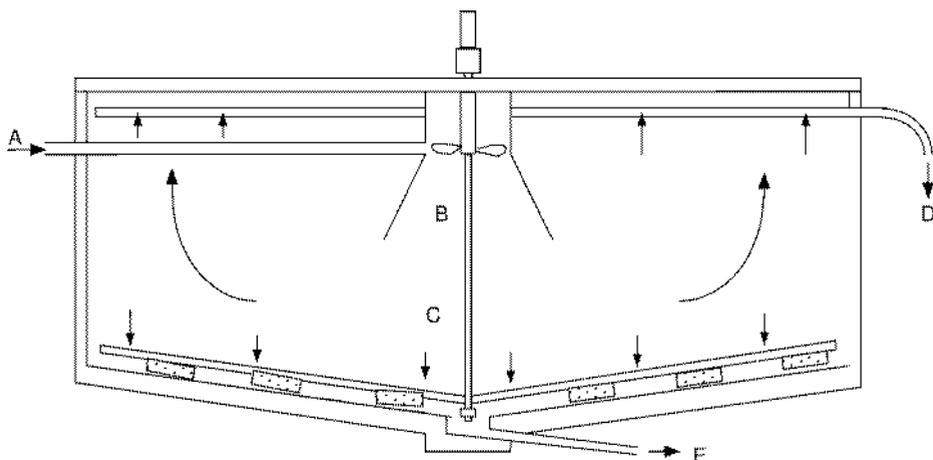


Fig. 13.1 Schematic layout of circular clarifier: A, inlet; B, flocculation zone; C, sedimentation; D, overflow of clarified water; E, sludge removal.

at 100–200 gm⁻³ [4, 5]. Flocculation is encouraged by gentle mixing in the central portion of the basin (shown as B in Fig. 13.1) as the influent water moves downwards within the central ducting. Sedimentation then takes place as the water slowly moves outwards and then up to the radial collecting trays at the surface. The sediment forms a sludge, which is directed down the sloping base of the basin towards the centre by the slowly rotating sweep arms and is then periodically pumped away [6]. This sludge is of no further use for fresh water treatment but may be used to aid flocculation of effluent in waste water treatment.

13.2.4

Filtration

For large-scale fresh water filtration, for example with water utility companies, and where extensive land use can be justified, slow sand filters are very effective. The efficiency is due to a combination of physical separation and biological activity. Each tank is approximately 3 m deep, with gravel over a porous base to allow the filtered water to drain away. Fine silica sand, particle size typically 0.2–0.4 mm, is deposited evenly to a depth of up to 1 m, with the influent water being maintained at 1–2 m above the sand. Influent water must be distributed evenly across the filter to avoid disturbing the bed. The top 20–30 mm of the sand rapidly develops a complex biofilm, commonly referred to as zooglear slime or Schmutzdecke. This biofilm is made up from polysaccharide secreted by bacteria such as *Zoogloea ramigera*, which is also colonised by protozoa. Bacteria and fine particles become trapped in the slime and are ingested by the protozoa. The resulting increase in the efficiency of filtration can give a two order (99%) drop in the microbial population of the water, as well as reducing the levels of dissolved organic and nitrogenous matter [5, 7]. The removal of organic matter may be further improved by including a layer of granular activated carbon in the sand bed and by ozone injection prior to filtration. Throughput can be up to 0.71 m⁻² s⁻¹ ($\approx 60 \text{ m}^3 \text{ m}^{-2} \text{ day}^{-1}$). The build up of biofilm and retained debris plus algal growth slowly reduces the throughput of the filter, so that the surface layer needs to be removed and cleaned at intervals of 1–6 months. Complete replacement of the sand is needed at longer intervals, an expensive operation.

In most industrial sites space is at a premium, so either high-rate or pressure filters are employed (see Fig. 13.2). High-rate filters are also commonly used by water utilities, either instead of or in conjunction with slow filters. High-rate filters may be upflow or downflow, the flow rate for the former being limited so that fluidisation of the bed does not occur. These filters are normally built as a series of modules with the pipework so arranged that one module can be taken out of service at a time for regeneration by backflushing, which is usually carried out on a daily basis [8]. Units are normally up to $\approx 200 \text{ m}^2$ in surface area.

High-rate filters use a coarse silica sand, e.g. particle size 0.4–0.7 mm, in a thinner layer (0.4–0.7 m deep) than for the trickling filters. Sometimes dual media beds are used, which can almost double the throughput to $\approx 24 \text{ l m}^{-2} \text{ s}^{-1}$

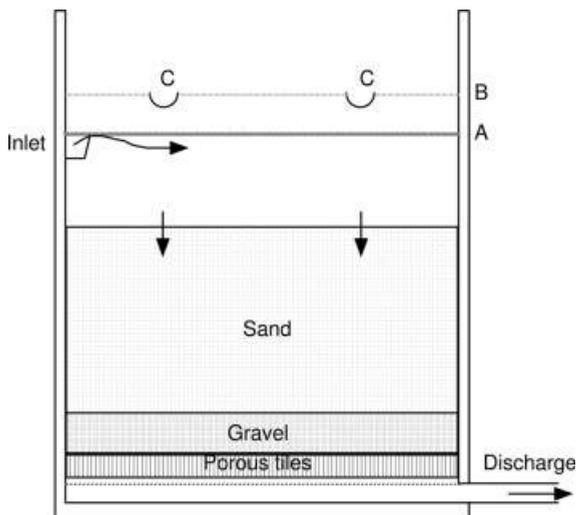


Fig. 13.2 Schematic of high rate down-flow filter: A, normal water level; B, backflushing level; C, washwater collection troughs.

($2000 \text{ m}^3 \text{ m}^{-2} \text{ day}^{-1}$). The faster throughput and more frequent cleaning prevent the build up of a biofilm and bacterial removal is typically less than 80% (less than one order). The sand must be periodically topped up as some is lost during backflushing when the bed is fluidised.

Pressure filters are suitable for smaller-scale operations where space is limited (see Fig. 13.3). These units are normally supplied as prefabricated mild steel pressure vessels, up to 3 m in diameter for ease of transport. The main axis may be horizontal or vertical to suit the site. Operating pressures are higher, with drops of up to 80 kPa across the filter medium. Filter media may range from sand or anthracite to diatomaceous earth on stainless support mesh or plastic formers wound with monofilament. Pressure filters should be installed in a duplex arrangement to allow one to be cleaned while the other remains in operation.

Cartridge filters have been used extensively for medium and small-scale water filtration, using a range of filter media from stainless meshes and sintered materials to plastics and paper filters. While paper filters are single-use, others are more robust and can be cleaned by backflushing. Ceramic filters have been used for small-scale and portable filtration equipment. These can operate down to the micron level and be used for removal of microorganisms, that is for microfiltration, which has been used commercially to provide water with a high microbiological quality. In small-scale personal filters, the ceramic filter may also contain silver to add a bactericidal stage.

Occasionally, where ground water has to be used and is contaminated by heavy metals, the water may be treated by nanofiltration (NF) or reverse osmo-

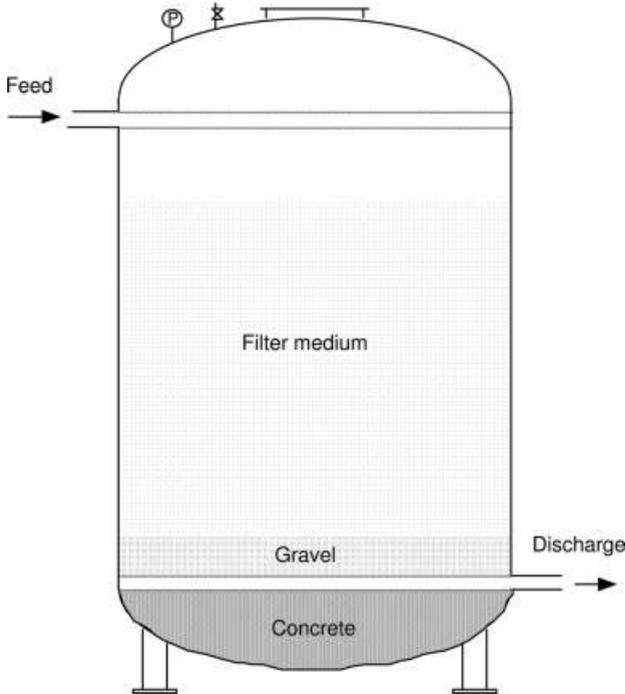


Fig. 13.3 Pressurised vertical filter unit.

sis (RO), the latter also being used for treatment of saline waters (see Chapter 14) [9–11]. In these treatments, the pore size is so small that the filtration is carried out at the molecular level, RO being regarded as a diffusion-based process. With NF there will be appreciable leakage of small ions, e.g. Na^+ , through the membrane, but larger ions and molecules will be retained. The retention of small molecules and ions leads to an increase in the osmotic pressure of the retentate, requiring a higher driving force (>2 MPa) than for other filtration processes [11]. This energy requirement is less than would be required for distillation processes and RO provides an economic alternative for desalination. Ion exchange processes may also be employed for removal of heavy metals [10, 12].

The application of membrane processes for water treatment has been covered in detail by Duranceau [13]. NF has been applied to the treatment of contaminated river water to provide drinking water. At Méry sur Oise, 80% of the water being treated was put through an additional treatment, with MF followed by NF with 230 Da cutoff at a maximum throughput of $180\,000\text{ m}^3\text{ day}^{-1}$. The system achieved a 90% reduction in total organic compounds from a maximum intake value of 3.5 mg l^{-1} to typical output values of $0.1\text{--}0.25\text{ mg l}^{-1}$ [14, 15].

13.2.5

Disinfection

It is assumed that any water used within a food processing plant must, at the very least, be of potable quality. This means freedom from taints, chemical contaminants and pathogenic organisms. Potable water often contains low levels of organisms capable of causing product spoilage problems. The microbiology of water is discussed in detail by McFeters [16]. Disinfection is any process whereby pathogenic organisms are removed or inactivated so that there is no risk of infection from consuming the treated water. Both chemical and physical methods may be used. Chemical methods are the most common and are normally based on chlorine or ozone, while physical methods may include microfiltration, irradiation with ultraviolet (UV) and heat. Since the disinfected water is only safe until it is recontaminated, disinfection should be regarded as the terminal treatment in the process. It is imperative that disinfected water be protected from recontamination. Both pasteurisation and UV treatments have been used for preparing washwaters and additive water where the addition of trace compounds is not desired, e.g. washwater for cottage cheese curd preparation and the dilution of orange concentrates.

Chemical disinfection using chlorine or chlorine derivatives is the most commonly applied method for large-scale water treatment, followed by the use of ozone, the latter particularly in Canada, France and Germany.

The Chick-Watson theory remains the principal theory to explain the kinetics of disinfection, where the lethality of the process may be described by the following equation [17]:

$$\ln \frac{N}{N_0} = kC^n t \quad (13.3)$$

where N is the number of pathogens surviving, N_0 is the number of pathogens at t_0 , C is the concentration of disinfectant, t is time, k is the coefficient of specific lethality and n is the dilution coefficient.

Specific lethalities vary considerably between disinfectants and with the target microorganisms. These lethalities also change at different rates with pH and temperature; and thus Table 13.1 should be treated as a semiquantitative indication of the ranking of disinfectants and their effectiveness against groups of organisms.

13.2.5.1 Chlorination

Liquid chlorine was the most commonly used agent for large-scale chlorination. Smaller-capacity plant has also used chlorine dioxide, sodium or calcium hypochlorites.

On dissolution of chlorine in water, chloride ions and hypochlorous acid are generated:

Table 13.1 Specific lethalties of common disinfectants (mg l⁻¹ min⁻¹, n=1). Source: [1], by courtesy of McGraw-Hill.

Disinfectant	Enteric bacteria	Viruses	Spores	Amoebic cysts
Ozone (O ₃)	500.0	5.0	2.0	0.5
Chlorine dioxide (ClO ₂)	10.0	1.5	0.6	0.1
Hypochlorous acid (HOCl)	20.0	>1.0	0.05	0.05
Hypochlorite ion (OCl ⁻)	0.2	0.02	0.0005	0.0005
Monochloramine (NH ₂ Cl)	0.1	0.005	0.001	0.002



The equilibrium is temperature-sensitive, pK_H=3.64 at 10 °C, 3.42 at 25 °C. Ionisation of the hypochlorous acid is pH-sensitive, pK_I ≈ 7.7 at 10 °C, 7.54 at 25 °C [1].

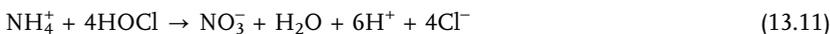
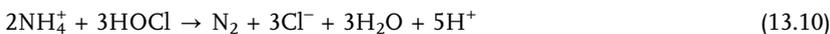


Thus, below pH 7, the un-ionised form predominates and at pH 6–9, the proportion increases rapidly with pH. Chlorine existing in solution as chlorine, hypochlorous acid or hypochlorite ions is known as free available chlorine.

Safety concerns have led to the replacement of chlorine and chlorine dioxide by sodium hypochlorite solutions. Concentrated hypochlorite solutions are both corrosive and powerful oxidising agents, so it is now generated as a dilute solution by electrolysis of brine in all but the smallest water treatment plants [4]. Care must be taken to avoid accumulation of hydrogen.



Hypochlorous acid reacts with ammonia, ammonium compounds and ions to form a range of compounds, e.g.:



Up to 8.4 mg of chlorine could be needed to react with 1 mg of ammonia. Doses of 9–10 mg chlorine per 1 mg ammonia are recommended in order to

guarantee that there would be free residual chlorine in the treated water. The addition level at which the added chlorine has oxidised the ammonia is referred to as the break point. Addition of chlorine beyond the break point enables the disinfection process to continue after treatment and confers some resistance to postprocess contamination. This can be critical in maintaining the safety of some food processes, for example the cooling of canned products.

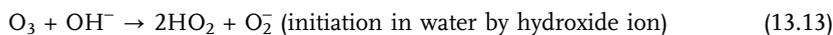
Superchlorination may be used when there are potential problems, e.g. due to a polluted source or breakdown. The water is chlorinated well beyond the break point to ensure rapid disinfection. If the water is left in this state it will be unpalatable, thus the water may be partially dechlorinated by adding sulphur dioxide, sodium bisulphite, ammonia or by adsorption onto activated carbon. Superchlorinated water should be used in the food industry for cooling cans and other retorted products to avoid postprocess contamination due to leakage during the cooling stage.

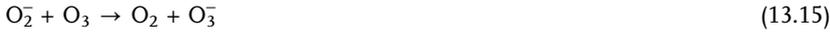
In contrast to superchlorination, the ammonia-chlorine or combined residual chlorination methods make use of chloramines or 'bound chlorine' left by incomplete oxidation. Chloramine is 40–80 times less effective as a disinfecting agent than hypochlorous acid but is more persistent and causes less problems with off flavours and odours than chlorine or dichloramine (≈ 200 times less effective). Thus, any residual chloramine may have a bactericidal effect during distribution. The most common way to apply this principle is to first chlorinate and then add ammonia to the disinfected water [17].

13.2.5.2 Ozone

Ozone treatment has been used widely, both for general supplies and for smaller-scale treatment of water for breweries, mineral water plants and swimming pools. The advantage with ozone is its rapid bactericidal effect, good colour removal, taste improvement and avoidance of problems with chlorophenols. Chlorophenol production can be critical in the reconstitution of orange juice from concentrates; and chlorine-free water is essential if this taint is to be avoided. There is no cost advantage over chlorine.

Ozone is more soluble than oxygen in water and is a more powerful oxidising agent than chlorine. As such, it is frequently used in the pretreatment of water to break down pesticide residues and other organic compounds in the raw water. This has a secondary benefit of inactivating protozoan contaminants such as *Cryptosporidium*, providing a higher quality intermediate for filtration and final treatment. The effectiveness of ozone is related to oxygen, superoxide and hydroperoxyl radicals formed on its autodecomposition [18].





Autodecomposition rates increase with pH, radicals, UV and hydrogen peroxide, but are reduced by high concentrations of carbonate or bicarbonate ions. Ignoring the effect of carbonate, the autodecomposition rate was described [1] as:

$$\frac{[\text{O}_3]_t}{[\text{O}_3]_0} = 10^{-At} \quad (13.19)$$

where $[\text{O}_3]_t$ is the concentration at time t , $[\text{O}_3]_0$ is the concentration at time 0, t is time and the value of A is $10^{(0.636 \text{ pH} - 6.97)}$.

Thus at pH 8, the half-life of ozone is about 23 min, depending upon water quality.

13.2.6

Boiler Waters

Boiler waters must be soft so that total solids build up only slowly in the boiler, but the water should not be corrosive. Simple treatments rely on lime and ferric chloride for softening and/or carbonate reduction with sodium hydroxide addition to give a final pH of 8.5–10.0. Residual carbon dioxide should be removed. Further softening can be achieved by phosphate addition to sequester calcium, by ion exchange or by NF. Dissolved oxygen must be removed, either by de-aeration or by dosing in a scavenger such as sodium sulphite. Dispersants such as tannins or polyacrylates should be added to aid dispersal of the accumulating suspended matter in the boiler, while antifoaming agents can reduce the carry-over of water droplets in the steam [19]. Boiler water demand should be reduced and run time extended by returning condensate to the boiler feed [20]. Where relatively unpolluted evaporator condensate is available, this can be treated to provide either boiler feed or soft cleaning water.

While all boiler water additives must be compatible with food production, any steam being generated for direct injection into food must be of exceptional quality and should be supplied from a dedicated generator.

13.2.7

Refrigerant Waters

Any water used in a heat exchanger for food should be of good microbiological quality, on the assumption that, despite precautions, leakage may occur. The risk of leakage may be reduced by operating the refrigerant at a lower pressure than the product. Any refrigerant must be checked regularly to ensure that it has not been contaminated by leakage of the food product.

For refrigerant waters at 3–8 °C, potable water should be sufficient. Below 3 °C, antifreeze additives may be necessary, becoming essential for operating temperatures below 1 °C unless an extensive ice bank system is used. Such additives are usually covered by national legislation. Glycol solutions are often used and calcium chloride (CaCl₂) solutions have been widely used throughout the food industry, both as a refrigerant and for freezing, e.g. in ice lolly baths.

13.3**Waste Water**

All processes will create waste and byproducts to a greater or lesser degree, as illustrated in Table 13.2. These represent not only a loss of ingredients and hence reduced profit from their conversion but an increased fresh water cost plus the additional cost of disposal of the waste created. It is essential that any manufacturing process should be designed and managed so as to minimise both the amount of fresh water used and the quantity of waste produced.

Waste minimisation must start at raw material delivery. Bulk tankers must be adequately drained before cleaning and a burst rinse should minimise waste before cleaning [20]. As in the factory, cleaning solutions should be collected and reused. Wherever possible, solid waste should be collected for separate disposal

Table 13.2 Examples of effluent loads from food processing.

	COD (mg l⁻¹)	SS (mg l⁻¹)	Water:product ratio (W:P)	Source
Beet sugar refining	1600	1015	–	[21]
Bread, biscuits, confectionery	5100 (275–9500)	3144	–	[22]
Brewing	2105 (1500–3500)	441	4	[22]
Cocoa, chocolate confectionery	9500 (up to 30 000)	500	4	[22]
Fruit and vegetables	3500 (1600–11 100)	500	15–20	[22]
Meat, meat products, poultry	2500 (500–8600)	712	10	[22]
Milk and milk products	4500 (80–9500)	820	1.5	[22]
Milk: liquid processing	ca. 700	–	12	[23, 25]
Milk: cheesemaking	≥2000	–	3	[23, 25]
Potato products	2300	656	–	[22]
Starch, cereals	1900	390	–	[22]

or recycling and certainly not flushed into the drains to add to effluent disposal problems.

The level of contamination of waste water is normally measured in terms of the biochemical oxygen demand over 5 days (BOD_5). This is defined as the weight of oxygen ($mg\ l^{-1}$, or $g\ m^{-3}$) which is absorbed by the liquid on incubation for 5 days at $20^\circ C$. The method is time-consuming and may underestimate the potential for pollution if the sample is deficient in microflora capable of degrading the materials present. Measurement of the chemical oxygen demand (COD), based on the oxidation by potassium dichromate in boiling sulphuric acid, provides a more rapid measure of the capacity for oxygen uptake. Sometimes the milder oxidation by potassium permanganate may be used to yield a permanganate value. These values are typically lower than the total oxygen demand based on incineration and CO_2 measurement [21]. There is no direct link between BOD_5 and COD, since the relationship depends on the biodegradability of the components in the waste stream. For readily biodegradable wastes there is an empirical relationship: 1 BOD_5 unit \approx 1.6 COD units, rising for less-biodegradable materials. The variability in these values may be less important if citing the difference between influent and effluent values for a treatment step or process, as these differences are based on changes in readily metabolised components.

It can be argued that the role of a food factory is to produce food and not to become involved with a nonproductive issue such as effluent disposal, which should be left to specialists. This could certainly be true of small enterprises but for larger factories, it may prove more cost-effective to undertake either partial or complete treatment of its trade effluent. In the UK, the charge for treatment of effluent is calculated by the Mogden formula, based primarily on the volume, COD and total suspended solids (TSS) [22].

Where an enterprise is to treat its own trade waste, the treatment plant should be located as far away from the production plant as possible, downwind and yet not being a nuisance to neighbours.

In most cases where effluent treatment is undertaken, this is kept separate from sewage, which poses greater public health problems and is normally dealt with on a community basis. Occasionally a plant may be built as a joint operation to handle sewage as well as trade waste, in which case more rigorous isolation from the food plant is essential.

13.3.1

Types of Waste from Food Processing Operations

The types of waste water produced by food processing operations reflect the wide variety of ingredients and processes carried out. Washing of root vegetables, including sugar beet, can give rise to high TSS levels in the effluent. Further processing of vegetables, involving peeling and/or dicing, increases the dissolved solids, as is also the case with fruit processing where sugars are likely to be the major dissolved component. Cereals processing and brewing create a carbohydrate-rich effluent, while effluent from processing legumes contains a

higher level of protein. The processing of oilseeds results in some loss of fats, usually as suspended matter. Milk processing creates an effluent with varying proportions of dissolved lactose and protein plus suspended fat. Meat and poultry processing gives rise to effluents rich in both protein and fat.

In most of these examples, there is particulate waste, i.e. particles greater than 1 mm in size, in addition to the fine suspended matter. These should be removed by screening prior to disposal of the plant effluent into the drain and strainers should be fitted into each drain to collect those particles that bypass the screens. Both screens and strainers must be cleaned daily.

Material recovered from screens within the process plant may be suitable for further processing. If screens are not fitted into the process plant, then particles collected on screens at the entry to effluent treatment cannot be reclaimed within the processing plant and must go to solid waste disposal.

13.3.2

Physical Treatment

Sedimentation and/or flotation usually form the first stage of effluent treatment, depending on the particular effluent. Both processes are applications of Stoke's Law (see Section 13.2.1). Where the fat may be recovered and recycled within the process, the flotation must be carried out within the production area. Centrifugation provides a rapid and hygienic technique.

The simplest flotation technique is to use a long tank. Waste water enters at one end over a distribution weir. Flows at $\geq 0.3 \text{ m s}^{-1}$ prevent sedimentation of fine suspended matter, but a residence time of about 1 h can be needed. Flotation can be hastened by aeration, fine gas bubbles being introduced at the base of the tank by air or oxygen injection, or by electrolysis. Fat globules associate with the gas bubbles, forming larger, less-dense particles that rise more rapidly to the surface. Bubbles 0.2–2.0 mm in diameter rise at $0.02\text{--}0.2 \text{ m s}^{-1}$ in water. The presence of free, that is unemulsified, liquid fat acts as an antifoam and prevents excessive foaming, as the fat agglomerates into a surface layer. This can be scraped off, dewatered and, for instance, sold off for fatty acid or soap production.

Production processes starting with dirty raw materials such as root vegetables produce an effluent with high TSS, some of which may get past the primary screening within the plant. In this instance a sedimentation or grit tank is needed. A rectangular tank is often used, with flows $\geq 0.3 \text{ m s}^{-1}$ to allow grit and mineral particles to sediment without loss of suspended organic material. The grit may be removed from the tank by a jog conveyor and dumped into a skip for disposal, either back onto the farmland, if relatively uncontaminated, or by landfill.

Following either of these pretreatments, the effluent should be collected into balance tanks. These serve to even out the fluctuations in pH, temperature and concentration throughout the day. Some form of mixing is desirable, both to aid standardisation and to maintain an aerobic environment, thus reducing off

odour generation. This minimises the use of acids and alkali to standardise the pH of the waste water and render it suitable for subsequent treatments. Lime (calcium hydroxide) or sodium hydroxide has been used to raise pH, while hydrochloric acid is a common acidulant. With dairy wastes, the use of sodium hydroxide as the principal cleaning agent normally results in an alkaline effluent, while plants handling citrus products could expect an acid waste stream.

13.3.3

Chemical Treatment

Most of the organic contaminants remaining in the waste water are either in solution or in colloidal dispersion. At around neutral pH, these colloidal particles usually have a net negative charge, so the addition of polyvalent cations, for instance aluminium sulphate at pH 5.5–7.5, or ferric chloride (or sulphate) at pH 5.0–8.5, promotes the formation of denser agglomerates that can be sedimented and recovered as sludge. Chemical addition is usually by dosing a solution into the waste stream followed by rapid mixing to ensure even distribution. The treated waste water is then allowed to stand, to permit formation of the flocs and their sedimentation. Sedimentation may be carried out in rectangular or circular basins. This process is similar to that employed for fresh water treatment (see Section 13.2.3), but the quantities of sludge settling out are much higher. Effluent leaves the settling tank via an overflow weir, which should be protected in order to prevent any surface fat and scum from overflowing too. Such fat should be scraped off periodically.

Sludge from settlement vessels typically contains about 4% solids and must be pumped over to an additional settlement tank where about half of the volume can be removed as supernatant and returned to the beginning of the treatment process.

The effluent from the settlement tank may then either be discharged to the sewer as partially treated effluent, incurring a much lower disposal charge, or else taken on to biological treatment.

13.3.4

Biological Treatments

Biological treatments may be divided into aerobic and anaerobic processes. In aerobic processes, oxygen acts as the electron acceptor so the primary products are water and carbon dioxide. In anaerobic treatment, the primary products are methane and carbon dioxide, with sulphur being reduced to hydrogen sulphide.

While properly run aerobic treatments produce the less polluting effluents, anaerobic treatment has great potential for large-scale treatment of sludge and highly polluted waste waters. Relative advantages are summarised in Table 13.3.

In general, smaller plants opt for aerobic treatment while the larger plants may use a combination of aerobic and anaerobic methods.

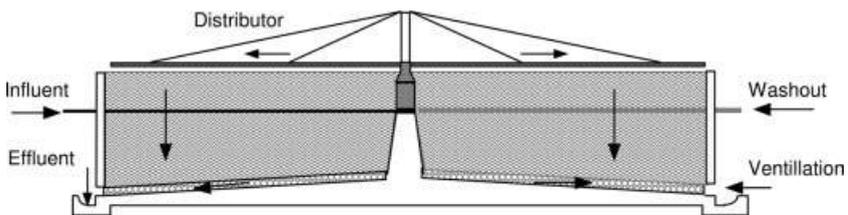
Table 13.3 Comparison of aerobic and anaerobic processes.

Factor	Aerobic	Anaerobic
Capital cost	(Lower)	Higher
Energy cost	Medium-high	Net output
Influent quality	Flexible	Demanding
Sludge retention	High	Low
Effluent quality	Potentially good	Poor

13.3.4.1 Aerobic Treatment – Attached Films

The trickling or percolating filter provides a simple and flexible means of oxidising dilute effluents. It takes the form of a circular (typically 7–15 m diam.) or rectangular concrete containment wall, 2–3 m high, on a reinforced concrete base which includes effluent collecting channels (see Fig. 13.4). The infill is preferably a light, porous material (about 50% voidage) with a high surface area (up to $100 \text{ m}^2 \text{ m}^{-3}$), for example coke or slag. Solid rock may also be used. The particle size varies over 30–50 mm, sometimes up to 75–125 mm for pretreatment prior to discharge to a sewer. The use of less dense, synthetic filter media allows deeper beds to be constructed.

The influent of clarified waste water is spread over the top surface of the filter by nozzles, mounted either on rotating arms for circular filters or on reciprocating bars for the rectangular beds. The surface must be evenly wetted, the liquid then trickling down through the filter. Filamentous algae often grow on the surface while, within the bed, the medium provides a physical support for a complex ecosystem. This biofilm contains bacteria, protozoa, fungi, rotifers, worms, and insect larvae. *Zoogloea ramigera* is the predominant colonising bacterial species, producing an exopolysaccharide-based support medium, the thickness of which increases with the richness of the nutrients in the influent liquor. The slime may be less than 1 mm thick with lean wastes but reach several millimetres with concentrated wastes, with a higher proportion of fungal mycelium in the latter. The biofilm also contains a range of other heterotrophic species, including *Pseudomonas*, *Flavobacterium* and *Alcaligenes* spp, which absorb soluble nutrients from the influent [21, 24]. Fungi and algae are also present. Fine suspended matter is trapped by the slime and ingested by protozoa, while the bur-

**Fig. 13.4** Section through a percolating filter.

rowing activity of the larvae helps maintain the flow through the biomass on the filter. Most of the BOD_5 reduction occurs in upper layers, while the oxidation of ammonia to nitrate take place in the lower portion of the bed.

With such a complex ecosystem, trickling filters take time to adapt to changes of feedstock. Initiation time can be reduced by seeding the filter with material from another filter running on similar effluent. Care must be taken to avoid feeding inhibitory materials or making sudden, major changes to the feedstock.

Careful management of the filter is essential. Since the biofilm builds up when fed rich effluents, blockage of the channels can occur with ponding of feedstock on the top of the filter. Algae can also grow on the surface and sometimes weeds can grow too, requiring the surface layers to be periodically turned over with a fork. The biofilm can be most easily managed by using two filters in series, alternating their position at intervals of 10–20 days, as illustrated in Fig. 13.5. The filter with a rich biofilm is then supplied with a much leaner nutrient stream and loses biofilm, which is sloughed off into the effluent at a higher rate than before. This system is referred to as alternating double filtration (ADF). In some small effluent plants a pseudoADF system is used, where the effluent from the trickling filter is collected and then passed back through the filter a second time, while the primary effluent is held back. The pseudo-ADF system could be run on a daily cycle basis.

The hydraulic loading on the filter should ideally be less than $1 \text{ m}^3 \text{ m}^{-3} \text{ day}^{-1}$ (or doubled with an ADF system), with a BOD loading of less than $300 \text{ g m}^{-3} \text{ day}^{-1}$, normally $\approx 60 \text{ g m}^{-3} \text{ day}^{-1}$. Filter effluent contains flocs of biofilm, which

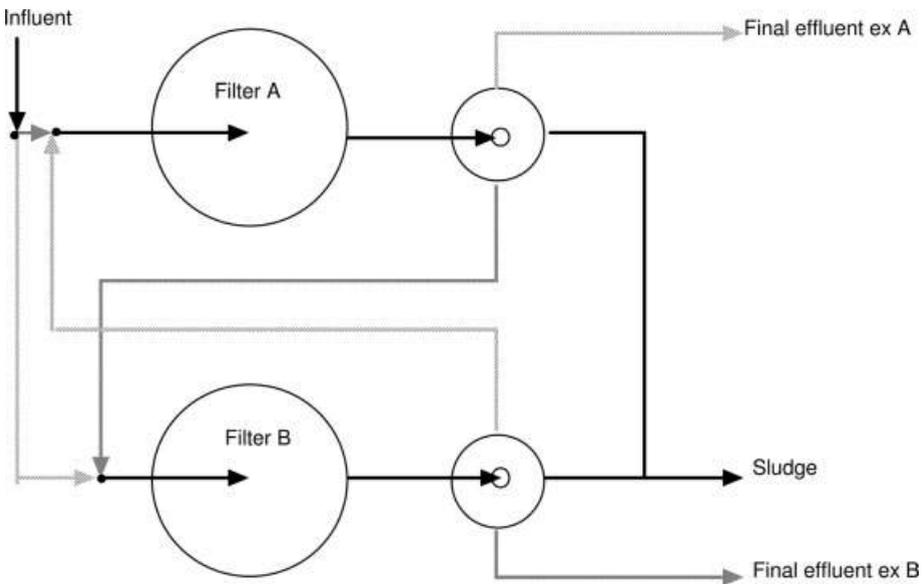


Fig. 13.5 Illustration of flows with alternating double filtration.

must be removed by sedimentation. Clarification is achieved by passing the effluent through a settling tank, for example as shown in Fig. 13.6. The sludge from the settling tank may be combined with that from earlier settling processes. The settled effluent BOD from the primary filter should be less than 30 mg l^{-1} , while that from the final filter should be less than 15 mg l^{-1} , typically $\approx 4 \text{ mg l}^{-1}$.

While trickling filters are relatively tolerant of varying effluent quality, their oxidative capacity is fairly low. In all but the smallest effluent plants, there can be an advantage in using higher rate aerobic systems, if only for pretreatment.

High-rate aerobic filters use a very low-density medium, e.g. plastic tube section, with high voidage (up to 90%) and specific surface areas up to $300 \text{ m}^2 \text{ m}^{-3}$. Being very light, high-rate filters can be mounted above ground level, sometimes being used as modifications mounted above preexisting treatment plants. The standardised, pretreated effluent is pumped over the filter at a relatively high rate. The hydraulic load is typically $5\text{--}10 \text{ m}^3 \text{ m}^{-2} \text{ day}^{-1}$ with high recirculation rates, to give more than 50% removal of BOD_5 [24].

Various types of disc and other rotating contactors have been used for effluent treatment, although initially these were more successful for general sewage than for food industry wastes. Rotating contactors use slowly rotating surfaces, which are less than half immersed in the effluent. As the device rotates, the damp film is taken up into the air and oxygenated, enabling the surface biofilm to metabolise the effluent. The build-up of biomass must be removed by peri-

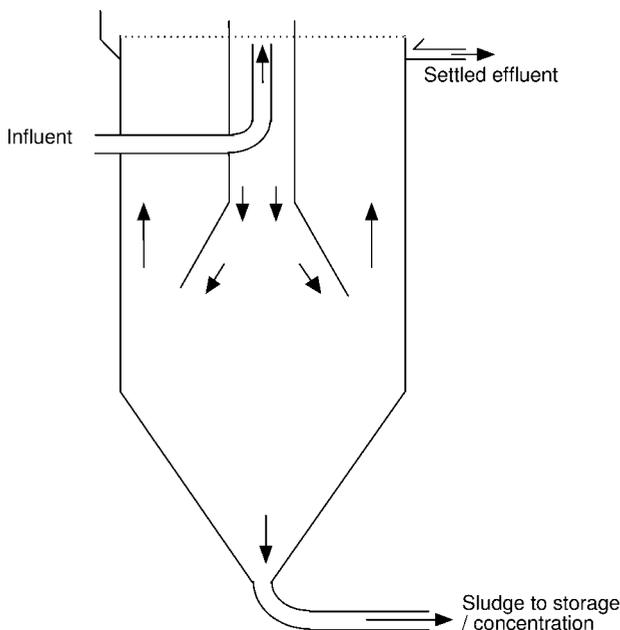


Fig. 13.6 Sludge settling tank.

Table 13.4 Examples of aerobic treatments used for food effluents. Source: [19]

Sector	Organisation	Location	Reactor	Capacity (t day ⁻¹ BOD ₅)	COD ^{a)}
Brewery	San Miguel	Philippines	Trickling filter + activated sludge	14.0	22.4
Dairy	Entrement	Malestroit, France	Extended aeration	3.0	4.8
Vegetables	Findus	Beauvais, France	Activated sludge	7.5	12.0

a) COD converted from BOD, assuming $1.6 \text{ COD} \approx 1 \text{ BOD}_5$,
to aid comparison with data in Table 13.5

odic flushing. Greater success has been achieved with submerged filters, operating with forced aeration in either upflow or downflow modes. The method has found use in smaller plants where activated sludge treatment may be difficult to use.

Some examples of aerobic treatments are given in Table 13.4.

13.3.4.2 Aerobic Treatment – Suspended Biomass

Activated sludge processes have been adapted successfully to the treatment of food wastes, and are attractive for larger plants ($>8 \text{ t COD day}^{-1}$) despite the higher energy costs, as they take up less land than trickling filters [25].

The biomass, consisting primarily of bacteria, is suspended in the medium as flocs. Protozoa are present in the flocs as well as free-swimming ciliated species [21]. Oxygen is introduced via compressed air injection, oxygen injection or by rigorous stirring using surface impellers. Surface mixing is less efficient but simpler and is used in smaller plants. The suspended biomass requires a constant influent composition for optimal operation. The efficiency in removing BOD depends on the rate of oxygenation, which can be reduced as the medium flows through the tank. Where phosphate reduction is also desired, part of the tank is run anaerobically to encourage additional phosphate uptake into the biomass once aerobic conditions are reintroduced. Partial denitrification is achieved by recycling biomass, a 1:1 recycling ratio being associated with 50% denitrification as a result of anoxic conditions being set up in the first stage of the digestion. Figure 13.7 illustrates the basic principles of an activated sludge plant. Where air or oxygen injection is employed, a tall cylindrical shape may also be used.

Though activated sludge plants are effective in reducing BOD₅, energy costs and sludge production are relatively high. A typical BOD₅ reduction of 95% can be achieved with hydraulic retention times of 10–20 h and biomass retention times of 5–10 days [25]. Higher BOD₅ removal can be achieved by reducing the substrate concentration and throughput, the longer residence time also resulting in lower sludge production. Increasing the substrate loading and/or throughput increases sludge production while giving a lower % BOD₅ removal.

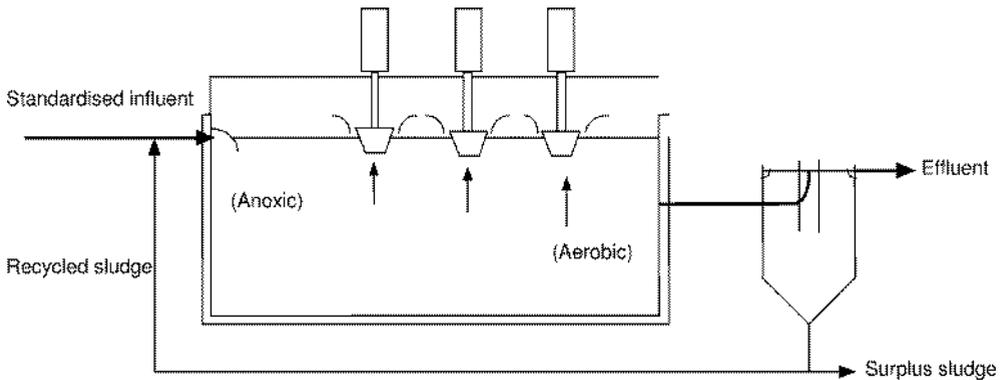


Fig. 13.7 Activated sludge fermentation plant.

With dilute effluents, similar results can be achieved using an oxidation ditch where, again, the biomass is largely suspended. Waste water is circulated under turbulent conditions ($\geq 0.3 \text{ m s}^{-1}$) around a channel, 1–2 m deep. The propulsion and aeration is carried out by a series of rotating brushes, as illustrated in Fig. 13.8.

Waste water is constantly fed to the ditch, the overflow passing through a settling tank. Retention times vary over 1–4 days, with sludge retention times of 20–30 days. Part of the sludge can be fed back to the ditch, giving a relatively low, net sludge production. While the oxidation ditch is simpler to operate than the activated sludge system, more land is required.

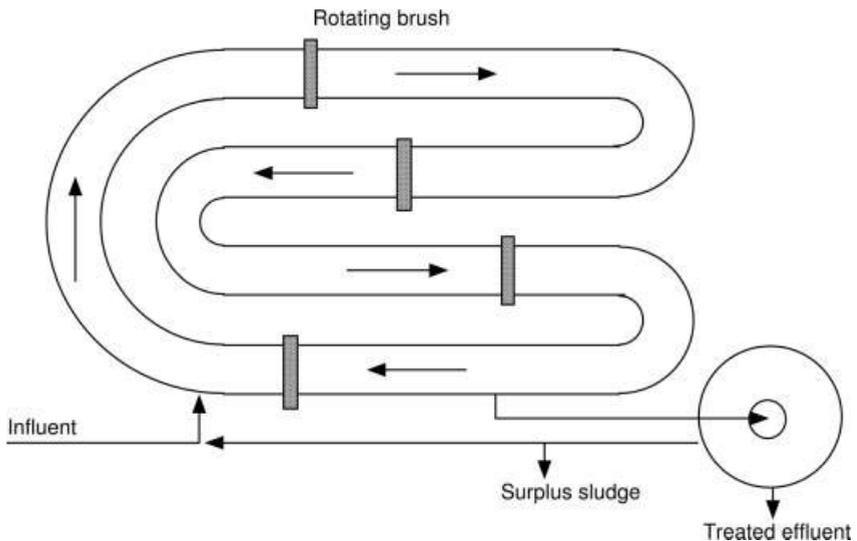


Fig. 13.8 Plan view of an oxidation ditch.

13.3.4.3 Aerobic Treatment – Low Technology

In some areas, where land is inexpensive and rainfall moderate or low, it may be possible to simply use a shallow lagoon, typically 0.9–1.2 m deep, for slow oxidation and settlement of effluents prior to irrigation [8]. Some lagoons may be up to 2 m deep, in which case anaerobic conditions will occur in the lower levels [26]. The lagoon should be lined with clay or other impermeable material to minimise the loss of polluted water into the surrounding soil. This approach has been used for effluent from seasonal canning operations and some dairy wastes. The ecosystem in such lagoons is extremely complex, with bacteria, protozoa and invertebrates plus algae and aquatic plants. Though these systems tolerate high organic shock loadings, care must be taken to site such a lagoon well away from the factory, to avoid overloading its oxidation capacity and to prevent leakage into any watercourse. Such partially treated effluent can be used for irrigation, using low pressure sprays to minimise drift. Up to 25 mm ($1 \text{ l m}^{-2} \text{ day}^{-1}$, including rainfall) have been used per 25-day irrigation cycle on grassland.

The lagoon approach can be improved by providing a number of ponds operating in series, providing a residence time of up to 3 weeks for stabilisation of BOD_5 levels [21].

Treatment of effluents has been achieved by trickling through beds of reeds and/or other semiaquatic plants, where the root structure supports a complex aerobic ecosystem, similar to that found in lagoons. Soil based wetland has been found to be more stable than gravelbased systems, which can block up. Findlater et al. [27] reported 70–80% BOD_5 removal at loadings of $4\text{--}20 \text{ g m}^{-2} \text{ day}^{-1}$, while Halberl & Perfler [28] reported 80–90% BOD_5 removal at slightly lower loadings of $1.5\text{--}15 \text{ g m}^{-2} \text{ day}^{-1}$. Extending the loading to $2\text{--}25 \text{ g m}^{-2} \text{ day}^{-1}$ gave a still wider range of BOD_5 removal (56–93%), the higher loading giving a reduced percent removal [29]. A reduced reduction in BOD_5 was also noted when feeding the reed beds with treated waste water where the BOD_5 was $\approx 50 \text{ mg l}^{-1}$. In experiments with meat processing effluent in gravelbed wetland trenches (18 m^2) and soilbased surfaceflow beds (250 m^2), van Oostrom & Cooper [30] found that COD removal rates increased with loadings up to $\approx 20 \text{ g m}^{-2} \text{ day}^{-1}$. BOD_5 reduction was 79% with untreated effluent from balance tanks at a loading of $117 \text{ g m}^{-2} \text{ day}^{-1}$, rising to 84% for partially treated, anaerobic effluent fed at $24 \text{ g m}^{-2} \text{ day}^{-1}$. These data suggest a bed requirement of $50 \text{ m}^2 \text{ kg}^{-1} \text{ BOD}_5 \text{ day}^{-1}$ with feed rates of $\approx 50 \text{ l m}^{-2} \text{ day}^{-1}$, depending upon influent strength. A complex of beds in both parallel and series would be desirable to consistently produce a low BOD_5 effluent, with an annual harvest of the aboveground biomass.

13.3.4.4 Anaerobic Treatments

Anaerobic treatments have been applied to both the sludges from aerobic treatment and to treat highly polluted waste streams from food plants. Aerobic treatments can produce 0.5–1.5 kg of biomass per 1 kg of BOD_5 removed, so potentially large quantities of sludge may need to be treated. Much of the sludge is

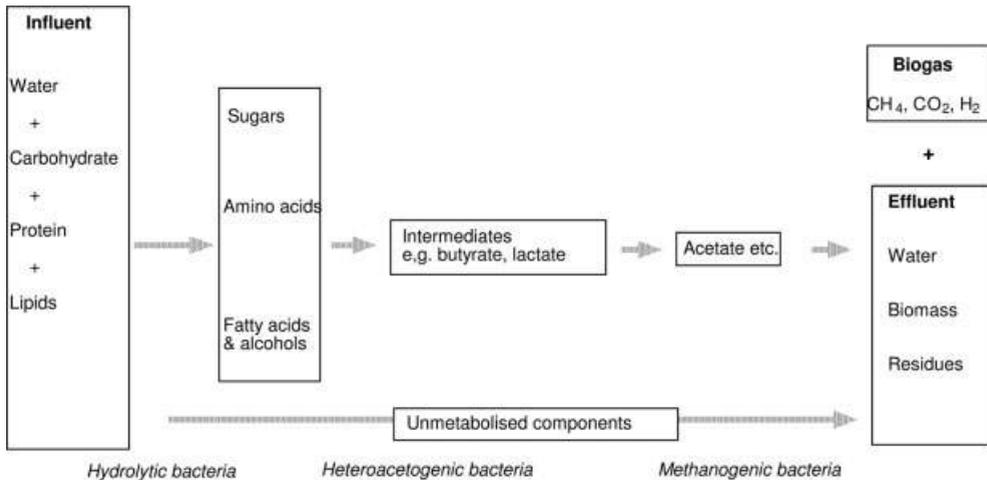


Fig. 13.9 Anaerobic catabolism.

low in solids and should first be concentrated, by settlement or centrifugation, to at least 8–10% dry matter.

Most anaerobic reactors are run at $35 \pm 5^\circ\text{C}$, to ensure methanogenesis. The catabolism of the food components is summarised in Fig. 13.9. While the main gaseous products are methane and carbon dioxide, there are also small quantities of hydrogen sulphide and other noxious compounds. The gases are normally collected and used on site, e.g. to drive a combined heat and power (CHP) plant, with scrubbing of the waste gasses where necessary.

There are five main types of anaerobic reactor:

- stirred tank reactor
- upflow sludge blanket
- upflow filter
- downflow filter
- fluid bed.

The stirred tank reactor is similar to the sludge fermenters used for domestic sewage, but with part of the effluent sludge recycled to the influent (see Fig. 13.10). Sludge from domestic sewage treatment is increasingly being heat-treated before fermentation to remove pathogens; and similar treatment may be required for some food wastes such as slaughterhouse effluent. Mixing may also be achieved by returning biogas from the gas separator. Heat from the CHP plant or another low-grade source is used to preheat the influent and to maintain the temperature at 35°C in the reactor. Biogas may be recovered from both the reactor and separator.

The upflow sludge anaerobic blanket (USAB) reactor does not use mixers but relies on the evenly distributed upflow of the influent (see Fig. 13.11), plus the bubbles from gas generation during fermentation. The bulk of the biomass

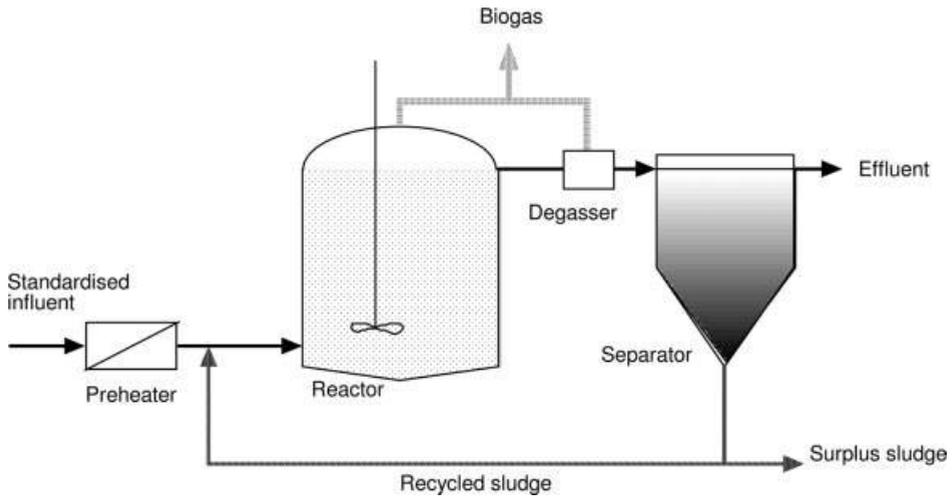


Fig. 13.10 Stirred tank reactor. Mixing may also be carried out by recycling biogas.

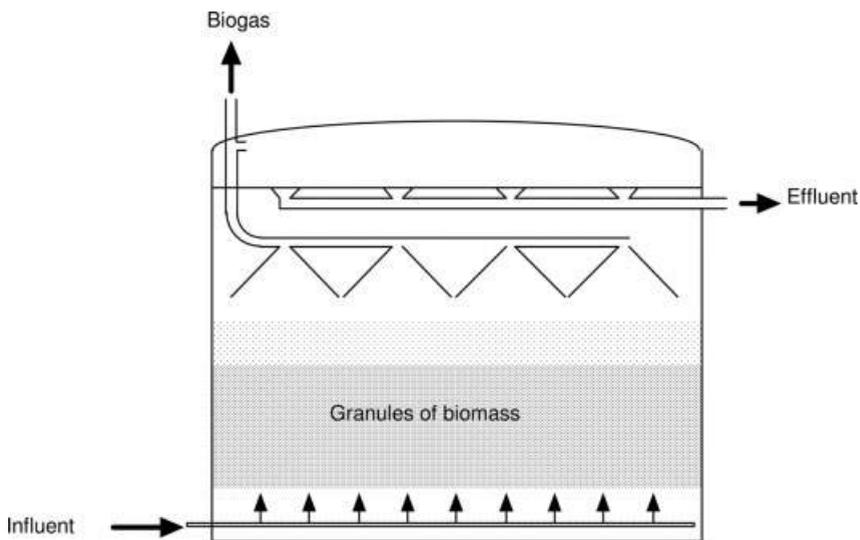


Fig. 13.11 Upflow stirred anaerobic blanket (USAB) reactor.

forms a granular floc in the lower layer of the reactor, encouraged by a high proportion of short chain fatty acids (which may be produced by prefermentation), the presence of Ca^{2+} and $\text{pH} > 5.5$, preferably $\text{pH} \approx 7.5$. For many influents, dosing with $\text{Ca}(\text{OH})_2$ is needed. A lighter floc of biomass also covers the granular floc. The influent quality demands have restricted the application of this type of reactor to wastes from yeast, sugar beet and potato processing

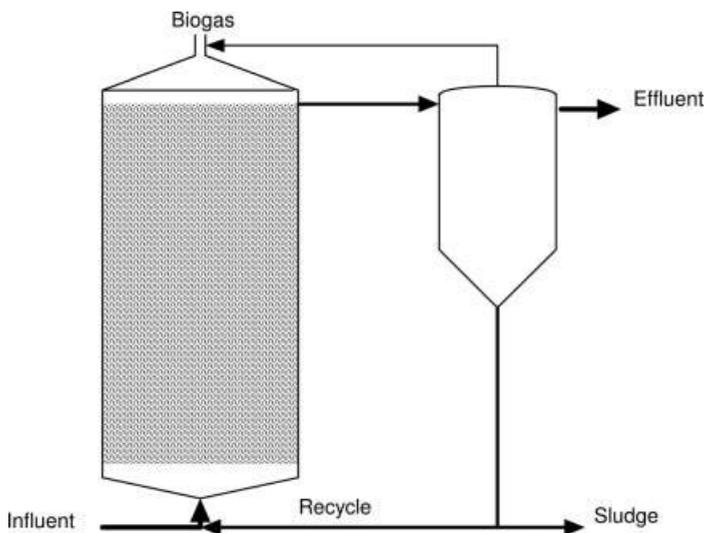


Fig. 13.12 Upflow anaerobic reactor.

wastes. Problems have been encountered with abattoir, dairy, distillery and maize processing wastes [31].

Anaerobic upflow filter reactors (see Fig. 13.12), contain a media fill. Crushed rock is the lowest-cost fill, similar to that with the aerobic trickling filters, with 25–65 mm rock giving approximately 50% void. About half of the biomass is attached to the medium, the rest being in suspension in the voids. More expensive media can give up to 96% void but with less biomass attached to the medium. The high proportion of suspended biomass limits the throughput as excessive biomass can be lost from the reactor. Similarly, the risk of excessive biomass growth also limits the influent concentration.

Anaerobic downflow reactors tend to use random packed high-void media. Though the bulk of the biomass is in suspension, the downward flow of the influent is opposed by the upward movement of the gas bubbles (see Fig. 13.13). This upward movement both buoys up the biomass and promotes mixing within the reactor.

Fluid bed reactors provide an improvement on the USAB and upflow filter reactors, combining some of their properties. Fine particulate material, typically sand (particle size ≤ 1 mm) is fluidised by the upflow of the influent liquid plus the gas evolved, expanding the bed volume by 20–25%. Upflow velocity is critical, typically being $3\text{--}8 \times 10^{-3} \text{ m s}^{-1}$. The sand and attached biomass are retained within the reactor by reducing the upflow velocity in the wider section at the top of the reactor, suspended biomass being returned to the primary vessel, as shown in Fig. 13.14.

The various types of reactor have been increasingly adopted for large-scale processing of food processing wastes, particularly where these wastes are rela-

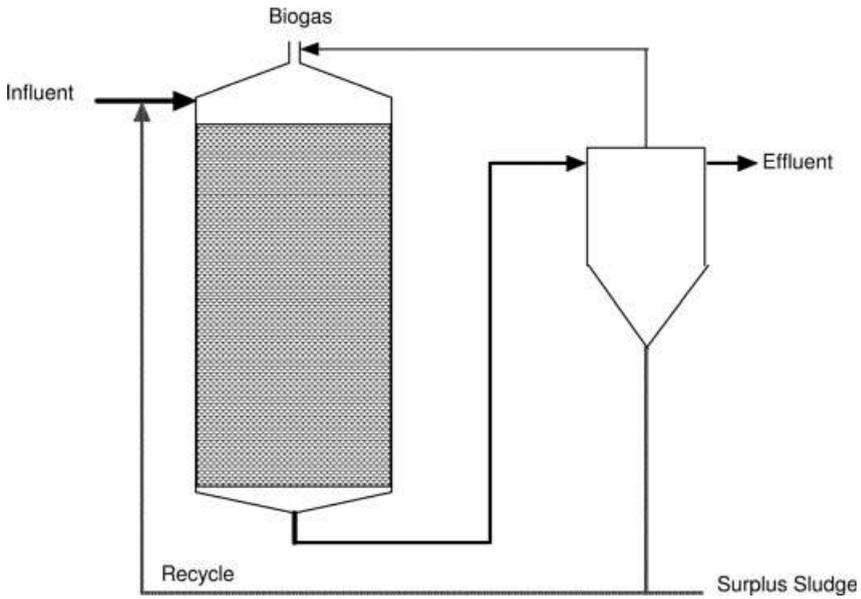


Fig. 13.13 Downflow anaerobic reactor.

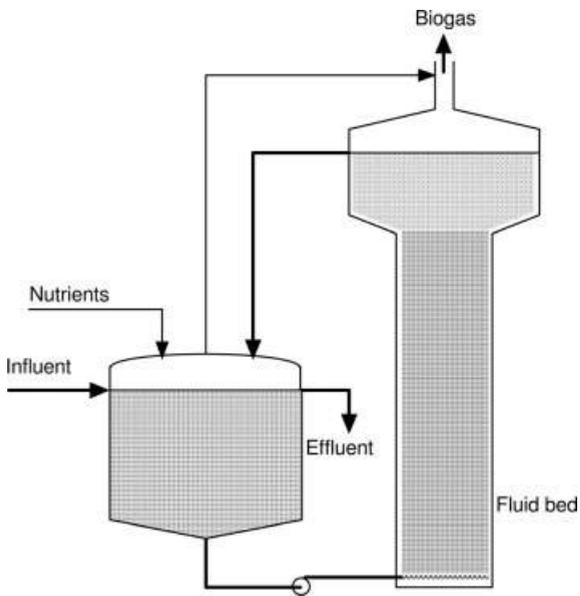


Fig. 13.14 Fluidised bed.

Table 13.5 Examples of anaerobic treatments used for food effluents. Source: [18]

Sector	Organisation	Location	Reactor	Capacity (t day ⁻¹ COD)
Brewing	El Aguila	San Sebastian de Los Reyes, Spain	Fluidised bed	50.0
Brewing	Sébastien-Artois	Armentières, France	Sludge blanket	10.0
Canning	Bonduelle	Renescue, France	Digester	18.0
Dairy	St Hubert	Magnières, France	Upflow filter	2.8
Distillery	APAL	Paraguay	Digester	54.0
Distillery	DAA	Ahausen, Germany	Upflow filter	12.0
Sugar	Julich	Julich, Germany	Digester + activated sludge	30.0
Sugar	Südzucker	Platting, Germany	Digester	30.0–38.0
Winery	Canet C.C.	Canet, France	Fluidised bed	4.2

Table 13.6 Comparison of anaerobic reactor designs. Sources: [19, 25, 32]

Type	Loading (kg COD m ⁻³ day ⁻¹)	Feed (g l ⁻¹)	Retention time (h)	COD removal (%)
Stirred tank	0.2–2.5	5–10	24–120	80–90
Filter	2–15	1	10–50	70–80
USAB	2–15	10	10–50	70–90
Fluid bed	2–60	2.5	0.5–24.0	70–80

tively concentrated (Table 13.5). A comparison between types is given in Table 13.6. Typically, 10–30% of the influent COD remains in the effluent so, irrespective of the type of reactor, further processing of the effluent is required. This entails separation of the sludge and final aerobic treatment of the water.

13.3.4.5 Biogas Utilisation

Gas collected from anaerobic treatments contains primarily methane and carbon dioxide in ratios varying from 1:1 to 3:1, with traces of hydrogen, hydrogen sulphide and other volatiles, depending on the substrate and operating conditions [32]. It is normally collected in a floating-dome gas holder at relatively low pressure, 1.0 ± 0.5 kPa. The capacity of the gas holder depends on the output from the digester and whether the gas is used constantly or only during part of the day. Gas may be used in boilers to raise steam or in a CHP engine. The power produced by the CHP should exceed that used in the anaerobic process, the heat being used to maintain the fermentation temperature in the reactor(s). A flare may be used to automatically burn off surplus gas in the event of a breakdown.

13.4

Sludge Disposal

Sludge production is a major problem with both aerobic and anaerobic processes, in terms of both their immediate offensive nature and the potential for pollution. Sludge disposal at sea is no longer permitted, so the choice is between disposal on land or incineration. Sludges, particularly those from primary settlement, can be highly putrescible but have an advantage over municipal sewage sludge in that their levels of heavy metals and organic contaminants are likely to be low.

If agricultural land is nearby, it may be economic to dispose of the unconcentrated sludge direct to the land, preferably by injection below the surface to minimise nuisance and avoid runoff [33]. In many cases, it is necessary first to concentrate or thicken the sludge. This may be carried out by gravity settlement for 2–5 days, sometimes aided by the addition of polyelectrolytes. The supernatant must be fed back to the effluent treatment plant. Disposal of sludge from food processing by landfill is uncommon in the UK but has been used widely in other parts of the EU.

Sludge has been used to aid bioremediation of contaminated land, where its nutrients and humus help raise the activity of the soil bacteria. It has also been useful in raising the productivity of the poor soils often used for forestry. With further dehydration, such as by belt drying, there is potential for mixing the sludge with straw and composting. The composting process is an aerobic batch process, during which the temperature can rise to 70 °C so that further dehydration occurs. The resulting compost is suitable for horticultural use as a soil conditioner.

Exceptionally, dehydration by belt press could be carried out to give solids in excess of 30%, for instance with fibrous sludges from vegetable processing. These high-solids sludges could be disposed of by high-temperature incineration. As with other waste incineration, the waste gases must be brought up to 800–900 °C to destroy any volatile organic compounds, particularly dioxins, that may be formed at lower temperatures in the initial stages of combustion. Within the EU, Germany is the greatest user of incineration for sludge disposal [25]. Incineration still yields a solid waste, the ash which makes up to 30% of the original solids normally being sent for landfill as a hazardous waste.

13.5

Final Disposal of Waste Water

In most cases, final disposal of treated waste water is into a water course where it will be diluted by the existing flow. General requirements are covered by regulations, in the EU based on the Urban wastewater directive (91/271/EC), usually complemented by consent limits based on avoidance of pollution. The EU approach is now complemented by a move to integrated pollution prevention

Table 13.7 Simplified example of a river classification. BOD₅ (mg ml⁻¹) to the 90th percentile, DO (min. % oxygen saturation) to the tenth percentile, NH₃ (mg N l⁻¹) to the 90th percentile. Source: adapted from [35].

Class	Description	BOD ₅	DO	NH ₃	Biology
A	Very good	2.5	80	0.25	No problems
B	Good	4.0	70	0.6	No significant problems
C	Fairly good	6.0	60	1.3	Some restriction to fish species
D	Fair	8.0	50	2.5	Extraction for potable water after advanced treatment
E	Poor	15.0	20	9.0	Only low-grade abstraction, eg cooling water
F	Bad	>15.0	<20	>9.0	Very polluted, severely restricted ecosystem and potential nuisance

and control [34] under Directive 96/61/EC. While it may be desirable to recycle water within a factory [35], with food processing such recycling would be constrained by aesthetic as well as cost considerations.

Measurement of river quality is complex, as the river is effectively an aerobic fermenter, with the flow rate and hence oxygenation playing a vital part in the natural bioremediation processes [36]. An illustration of a simplified river classification is given in Table 13.7.

Discharge licences may include maxima for flow, temperature, suspended solids, dissolved solids, BOD₅, nitrogen, phosphorous and turbidity. One processor's waste stream may subsequently (certainly eventually) be another's water source.

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