

CHAPTER 11

BIOLOGICAL TREATMENT PROCESSES

BIOLOGICAL SYSTEMS

Many types of biological processes are active in the breakdown of organic matter.

A nutritional classification of organisms is given in Table 11.1.

TABLE 11.1

Nutritional classification of organisms

Class	Nutritional requirements.
Autotrophic	The organisms depend entirely on inorganic compounds.
Heterotrophic	Organic compounds are required as nutrient.
Phototrophic	Use radiant energy for growth.
Chemotrophic	Use dark redox reaction as energy source.
Lithotrophic	Use inorganic electron donors (e.g. hydrogen gas, ammoniumions, hydrogen sulphate and sulphur).
Organotrophic	Require organic compounds as electron donors.
Strictly aerobic	Cannot grow without molecular oxygen, which is used as oxidant.
Strictly anaerobic	Use compounds other than oxygen for chemical oxidation. Sensitive to the presence of minor traces of molecular oxygen.
Facultative anaerobic	Can grow either in the presence or absence of air.

Most biological systems used to treat organic waste depend upon heterotrophic organisms, which utilize organic carbon as their energy source. As seen in the table, the organisms can be either strictly aerobic, strictly anaerobic or facultative anaerobic. Anaerobic breakdown is used in the treatment of sludge, or denitrification where nitrate is the oxygen source. Chemolithotrophic organisms are also used in biological treatment processes. These comprise specialized groups of bacteria which are able to oxidize inorganic compounds such as hydrogen sulphur, or ammonium.

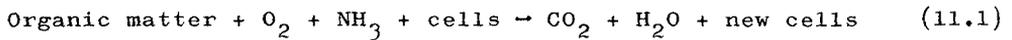
Of the various types of metabolism in which it is the redox reaction that provides the ultimate source of energy, it is customary to recognize three major classes of energy-yielding processes:

Fermentation, which constitutes the class of energy-yielding biological redox reactions in which organic compounds serve as the final electron acceptors;

Respiration (aerobic) constitutes the class of biological oxidation processes in which molecular oxygen is the ultimate electron acceptor;

Respiration (anaerobic) constitutes the class of biological oxidation processes in which inorganic compounds other than oxygen are ultimate electron acceptors.

These reactions can be described by the following overall process



Nitrification results from a two-step oxidation process. First, ammonia is oxidized to nitrite by Nitrosomonas. The second process is oxidation of nitrite to nitrate by Nitrobacter:



Nitrate can be used as an oxygen source for the biological decomposition of organic matter. The reaction that takes place is:



Comparison with molecular oxygen supplied by the aeration method, the use of nitrate as an oxygen source is undoubtedly easier because of its extremely high solubility. Further, it can be expected that satisfactory biodegradation of organic matter may be carried out with microorganisms and waste water containing nitrate. Industrial waste water, especially from petrochemical plants, sometimes contains a large amount of nitrate as well as highly-concentrated organic matter. The application of a biological-treatment method for treating such waste water using nitrate as the oxygen source is therefore attractive.

Studies by Miyajima et al. (1975) have shown that the amount of BOD₅ removed by biological treatment with nitrate as an oxygen source is linearly related to the amount of nitrate removed in the reaction tank. Studies have shown that it is microorganisms such as *Pseudomonas denitrificans*.

Cellular growth

Cellular growth can often be described as a first-order reaction:

$$\frac{dX}{dt} = \mu_m \cdot X \quad (11.5)$$

where

X = concentration of volatile biological solid matter

μ_m = the maximum growth rate

t = time

Integration of this equation where $X = X_0$ and $t = 0$, gives:

$$\ln \frac{X}{X_0} = \mu_m \cdot t \quad (11.6)$$

This equation is only valid during the so-called logarithmic growth phase in which the substrate (the organic matter) is not limiting. When the substrate becomes the limiting factor, the growth rate can be described by means of the following equation:

$$\frac{dX}{dt} = \mu_m X \cdot S \cdot \frac{1}{K_S} \quad (11.7)$$

where S = substrate concentration and K_S = a constant.

These two expressions can be combined by means of the Michaelis-Menten equation:

$$\mu = \mu_m \frac{S}{S + K_S} \quad (11.8)$$

where

μ = growth rate ($= \frac{dX}{dt} \cdot \frac{1}{X}$)

μ_m = maximum growth rate

S = substrate concentration

K_S = Michaelis-Menten constant

As can be seen, when $S \rightarrow \infty$ the equation becomes (11.1) and when $K_S \gg S$ the equation is transformed into (11.7).

Fig. 11.1 shows the Michaelis-Menten relationship; the growth rate is plotted against the substrate concentration.

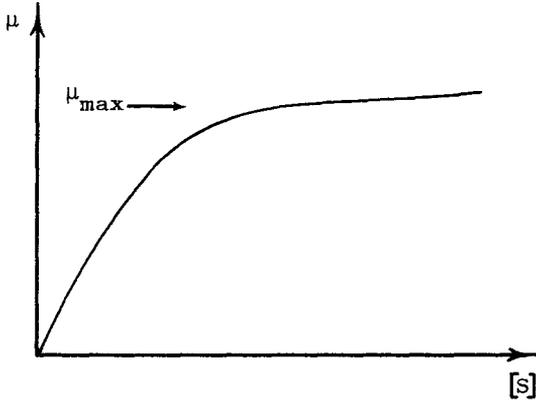


Fig. 11.1. Michaelis-Menten relationship.

It is often convenient to illustrate the Michaelis-Menten equation by means of a Lineweaver-Burk plot. The reciprocals of the growth rate and of the substrate concentration are plotted against each other (see Fig. 6.2 and 11.2).

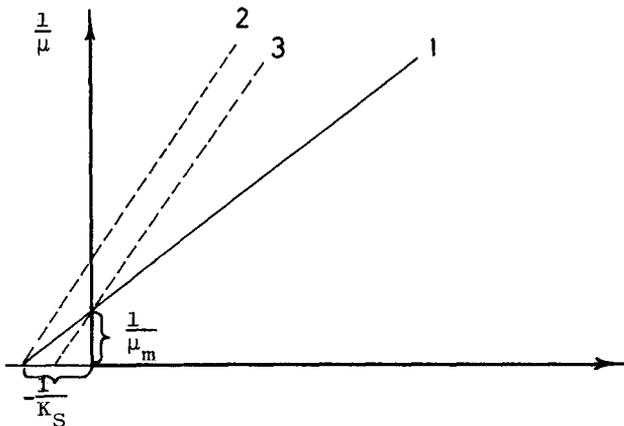


Fig. 11.2. Lineweaver-Burk plot (1). (2) the plot when a toxic compound is present. (3) the plot when a competitive inhibition takes place.

The relationship is linear as can be seen from equation (11.4), which can be transformed to:

$$\frac{1}{\mu} = \frac{K_S}{\mu_m} \cdot \frac{1}{S} + \frac{1}{\mu_m} \quad (11.9)$$

By use of the yield constant, a ($a = \text{mg biomass produced per mg of substrate used}$), dX can be expressed in terms of substrate removal:

$$dX = a \cdot dS \quad (11.10)$$

Combining equations (11.10) and (11.8) gives:

$$\frac{-dS}{dt} = \frac{\mu_m}{a} \cdot \frac{S}{S + K_S} \cdot X \quad (11.11)$$

The set of equations (11.5) to (11.11) is not valid for complex substrate mixtures, but in many cases the equations can be used as good approximations.

The temperature influences the processes significantly. The effect of temperature on the reaction rate can be expressed by the following relationship:

$$\mu_T = \mu_{20^\circ\text{C}} \cdot \theta^{T-20^\circ\text{C}} \quad (11.13)$$

where $\theta = \text{a constant}$

θ is listed in Table 11.2 for various types of processes.

Table 11.2

Temperature effects on biological processes

Process	θ
Activated sludge (low loading)	1.00 - 1.01
Activated sludge (high loading)	1.02 - 1.03
Trickling filter	1.035
Aerated lagoons	1.06 - 1.08
Nitrification	1.143

The reactions are also influenced by the presence of inhibitors. In the case of competitive inhibition, the equation (11.8) becomes changed to:

$$\mu = \mu_m \frac{S}{S + K_S \cdot (K_{S,I} + I)/K_{S,I}} \quad (11.14)$$

where $K_{S,I}$ = inhibition constant, and I = concentration of the inhibitor(s).

Competitive inhibition occurs when the inhibitor molecule has almost the same structure as the substrate molecule, which means that the micro-organism is able to break down the inhibitor and the substrate by the same, or almost the same, biochemical pathway. The resulting influence on the Lineweaver-Burk plot is also shown in Fig. 11.2.

If a toxic compound (non-competitive inhibitor) is present, only the maximum growth rate will be influenced, and it is reduced according to the relationship

$$\mu_m^+ = \mu_m \cdot \frac{K_{S,I}}{K_{S,I} + I} \quad (11.15)$$

Also in this case is the Lineweaver-Burk plot changed (see Fig. 11.2). Heavy metals and cyanide are examples of toxic materials that inhibit non-competitively. The approximate values of $K_{S,I}$ for each inhibitor are given in Table 11.3.

TABLE 11.3

Effect of inhibitors

Non-competitive inhibitor	$K_{S,I}$ (mg/l)
Hg	2
Ag	5
CO	10
Cu	20
Ni	40
Cr ⁶⁺	200
CN ⁻	200-2000

It is possible, however, according to Neufeld et al. (1975), to grow cultures of activated biota in the presence of mercury, cadmium and zinc levels that are higher than those that would previously have been thought possible. Mercury, cadmium and zinc are

rapidly removed from aqueous solutions by biological flocculation. Although the eventual equilibrium was only achieved after about 2-3 weeks, three hours of contact were sufficient to produce an almost complete approach to that equilibrium. The ratio of the weight of metal in the biological phase of flocculation to the weight of metal in the surrounding aqueous phase for the metals mercury, cadmium and zinc at equilibrium, ranges from 4000 to 10000.

Metal uptake by the biomass depends on several factors, including pH and the concentration of organic matter and metals present in the system. A higher initial concentration of metal ions or sludge increases the overall uptake. In general, the uptake capacity increases with increasing pH up to a value at which metals are precipitated. Although the affinity of the biomass for the metal ions is relatively less than that of strong chelating agents, it is much higher than that of competing organics in a supernatant. The large-scale accumulation of heavy metals by activated sludge with its subsequent removal in a secondary clarifier explains the significant reduction of these metals observed in many treatment plants.

The influence of pH on the biological processes is expressed by the equation:

$$\mu_m^+ = \mu_m(\text{pH}_{\text{opt}}) \frac{K_{S,I}}{K_{S,I} + I} \quad (11.16)$$

where

$\mu_m(\text{pH}_{\text{opt}})$ = maximum growth rate at optimum pH

$K_{S,I}$ = inhibition constant

I = concentration of the inhibitor

I can be expressed as:

$$I = 10^{\text{pH}_{\text{opt}} - \text{pH}} - 1 \quad (11.17)$$

The approximate value of $K_{S,I} = K_{S,\text{pH}}$ is about 30 for biological treatment processes and the optimum pH is in the range 7-8.

However, the best method to determine the constants μ_m and K_S is by extensive laboratory studies. Corresponding values of μ and X are measured and K_S and μ_m found by a Lineweaver-Burk plot.

Biological treatment processes

The various biological treatment processes are summarized as follows:

(1) The conventional activated-sludge process is defined as a system in which flocculated biological growth are continuously circulated and contacted with organic waste water in the presence of oxygen, which is usually supplied in the form of air bubbles injected into the liquid sludge mixture. The process involves an aeration step followed by sedimentation. The separated sludge is partly recycled back to be removed with the waste water. During aeration there occurs

- (a) Rapid adsorption and flocculation of suspended organics,
- (b) Oxidation and decomposition of adsorbed organics, and
- (c) Oxidation and dispersion of sludge particles.

Sometimes, depending on the retention and amount of oxygen introduced, ammonium ions are oxidized to nitrate by nitrifying organisms. This is seen particularly during the summer, and is due to the influence of the temperature on the rate constant for the nitrification process.

Activated sludge usually provides an effluent with a soluble BOD_5 of approximately 20 mg/l. The process necessitates the treatment of excess sludge before disposal.

(2) The extended aeration process works on the basis of providing sufficient aeration time for oxidizing the biodegradable portion of the sludge produced from the organics removed from the process. Fig. 11.3 shows the process schematically. The excess sludge in the process contains only non-biodegradable residues remaining after total oxidation. The total BOD_5 provided by this process is 20 mg/l or less.

(3) In the contact stabilization process the waste water is aerated with stabilized sludge for a short period of 1/2-1 hour. The mixed liquid is then separated by sedimentation and when settled the sludge is transferred to a sludge stabilization tank where aeration is continued to complete the oxidation. This process is used with advantage when a high percentage of the BOD is removed rapidly by biosorption after contact with the stabilized sludge. The extent of removal is dependent upon the characteristics of the sludge and of the waste water. As a general rule the process should give an efficiency of 85% BOD_5 removal.

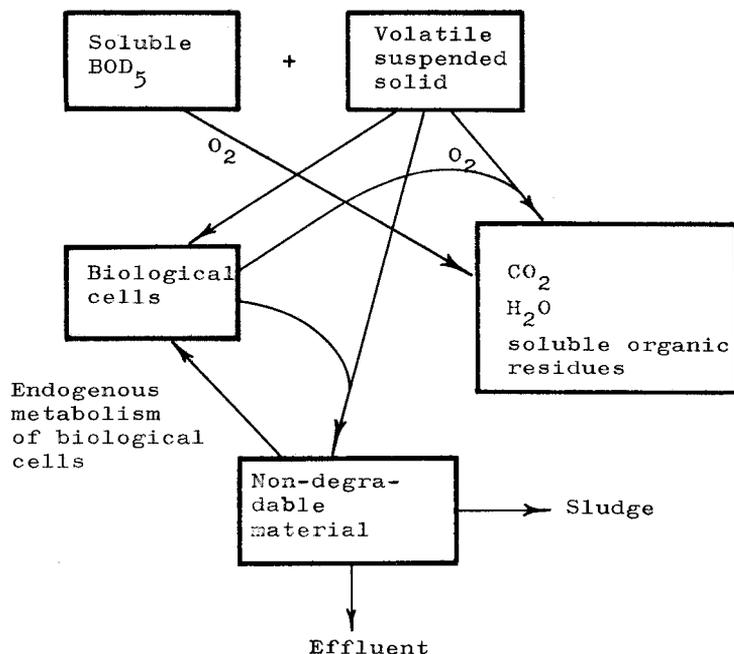


Fig. 11.3. Extended aeration process.

(4) A trickling filter is a bed packed with rocks, although, more recently, plastic media have been used. The medium is covered with a slimy micro-biological film. The waste water is passed through the bed, and oxygen and organic matter diffuse into the film where oxidation occurs. In many cases recirculation of the effluent improves the BOD removal, especially when the BOD of the effluent is relatively high. A high-rate trickling filter provides an 85% reduction of BOD for domestic sewage, but 50-60% is the general figure for BOD₅ reduction in the treatment of organic industrial waste water.

A plastic-packed trickling filter will require substantially less space than stone-packed trickling filters. According to Wing et al., (1970), plastic media packed to a depth of 6.5 m in a trickling filter, will require less than one fifth of the land required by trickling filters packed with stones to the usual depth of 2-4 m.

(5) Lagoons are the most common method of organic waste treatment, when sufficient area is available. They can be divided into four classes:

- (a) Aerobic algal ponds, which depend upon algae to provide sufficient oxygen.
- (b) Facultative ponds, which can be divided into an aerobic surface and an anaerobic bottom.
- (c) Anaerobic ponds, which are loaded to such an extent that anaerobic conditions exist throughout the liquid volume.
- (d) Aerated lagoons are basins in which the oxidation is accomplished by mechanical or diffused aeration units and by induced surface aeration. The turbulence is usually insufficient to maintain solids in suspension, thus most inert solids settle to the bottom where they undergo anaerobic decomposition. The basin (2-4 m deep) might include a sedimentation compartment to yield a more clarified effluent. If the turbulence level in the basin is increased to maintain solids in suspension, the system becomes analogous to an activated-sludge system.

An aerated lagoon might provide an effluent with less than 50 mg/l BOD₅, depending on the temperature and the characteristics of the waste water. Posttreatment is necessary when a highly clarified effluent is desired, and large areas are required for the process compared with the activated sludge process.

The design factors for lagoons are summarized in Table 11.4.

TABLE 11.4

Design factors for lagoons

	Aerobic	Facultative	Anaerobic	Aerated
Depth (m)	0.2-0.4	0.75-2	2.5-4	2-4
Retention time (days)	2-6	7-30	30-50	2-10
BOD loading (kg/ha/day)	100-200	20-50	300-500	depending upon waste characteristics and aeration
BOD removal (%)	80-90	75-85	50-70	50-90

(6) Anaerobic digestion. The anaerobic breakdown of organic matter to harmless end-products is very complicated. Fig. 11.4 summarizes some of the more general processes.

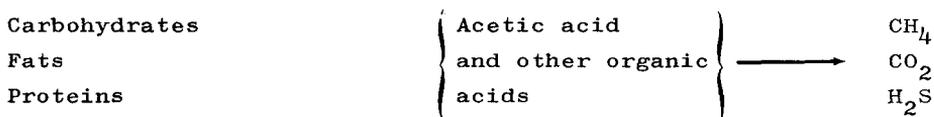


Fig. 11.4. Anaerobic degradation of organics.

Methane organisms convert long-chain volatile acids to methane, carbon dioxide and other volatile acids having a short carbon chain, which are then fermented in a similar fashion. Acetic acid is directly converted into carbon dioxide and methane. The rate of methane fermentation controls the overall reaction rate. Sufficient time must be available in the reactor to permit growth of the organisms or they will be washed out of the system. This means that the retention time must be greater than that corresponding to the growth rate of the methane organisms. It is possible, by the use of extracellular enzymes, to cut down the resistance time considerably, but the use of such enzymatic processes is still only in its infancy.

Many factors such as the composition of the sludge and the waste water, pH and temperature influence the reaction rate, but it is generally shown that the overall rate is controlled by the rate of conversion of volatile acids to methane and carbon dioxide. Digestion fails to occur when there is an imbalance in the rate of the successive processes, which might result in a build-up of volatile acids. The optimum conditions can be summarized as follows: pH, 6.8-7.4; redox potential, -510 to -540 mV; concentration of volatile acids, 50-500 mg/l; alkalinity, (as calcium carbonate) 1500-5000 mg/l; temperature, 35-40°C.

It should be possible to obtain an effective digestion with a retention period as low as 5 days, but increasing the retention time to 10 days should assure 90% degradation of organic matter. Anaerobic digestion is used for the treatment of sludge from biological processes as well as for the treatment of industrial waste water with an extremely high BOD₅, e.g. industrial waste water from the manufacture of yeast.

The major part of the gas produced by anaerobic treatment processes comes from the breakdown of volatile acids. The gas is composed of methane, carbon dioxide, hydrogen sulphide and hydrogen. The higher the residence time, the lower the percentage of carbon dioxide and the higher will be the percentage of methane in the gas produced.

Lawrence and McCarty (1967) have shown that methane gas production, at a good approximation, is 0.4 m^3 gas per kg COD removed. This value must be considered as the maximum obtained by complete conversion of the solid into methane.

Nitrification and denitrification

Effective nitrification occurs when the age of the sludge is greater than the growth rate of the nitrifying micro-organisms.

Bernhart (1975) has demonstrated that it is possible to oxidize ammonia in a complex organic effluent by biological nitrification.

The sludge age is defined as

$$G = \frac{X}{\Delta X} \quad (11.18)$$

where X = the mass of biological solid in the system and

ΔX = the sludge yield.

Shorter sludge ages will result in the organism being washed away.

The relationship between nitrification and sludge age is shown in Fig. 11.5. The data are based on those of Wuhrmann (1956) and Ford (1965).

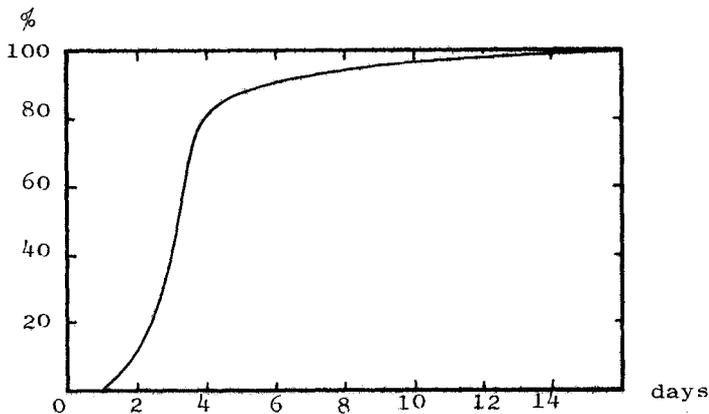


Fig. 11.5. Relationship between nitrification and sludge age.

Temperature exerts a profound effect on nitrification.

Downing (1966) has reported that the influence of temperature on the rate coefficient can be expressed as follows:

$$K_N = 0.18 \cdot 1.128^{T-15} \quad (11.19)$$

As can be seen, K_N is 0.18 at 15°C.

The optimum pH range for Nitrosomonas is 7.5-8.5 and for Nitrobacter 7.7-7.9. The rate seems to be dependent on the ammonium-ion concentration at concentrations in excess of 0.5 mg/l, which is considerably lower than those generally found in industrial waste water containing ammonium ions. Heavy metals are toxic at rather low concentrations. Toxic levels of about 0.2 mg/l are reported for chromium, nickel and zinc.

Nitrate can be reduced to nitrogen and dinitrogen oxide by many of the heterotrophic bacteria present in activated sludge, but the process requires anaerobic conditions. The pH affects the process rate, the reported optimum being above 7.0.

As the denitrifying organisms are heterotrophic, they require an organic carbon source. It is possible either to add the carbon source, methanol or molasses are used, or to use the endogenous by-product as the food supply.

The rate of denitrification increases with increasing concentrations of available carbon and of nitrate. Francis et al. (1975) report a successful denitrification of waste water, which contained more than 1000 ppm nitrate nitrogen.

Design of biological waste-water treatment facilities for industrial waste water

A mass balance of the substrate leads to the following equation:

$$Q \cdot S_i + V \frac{dS}{dt} = Q \cdot S \quad (11.20)$$

where

V = volume of the tank

S = substrate concentration in the tank

Q = flowrate of the waste water

S_i = substrate concentration of inflowing waste water

Dividing by Q the equation yields:

$$S_i - S = -\frac{V}{Q} \cdot \frac{dS}{dt} \quad (11.21)$$

If we combine this equation with equation (11.11):

$$-\frac{dS}{dt} = \frac{\mu_m}{a} \cdot \frac{S}{S + K_S} \cdot X$$

we obtain the following expression:

$$S_i - S = \frac{V}{Q} \cdot \frac{\mu_m}{a} \cdot \frac{S}{S + K_S} \cdot X \quad (11.22)$$

Often this expression can be transformed to:

$$\frac{S_r}{X \cdot T_r} = \frac{S_i - S}{X \cdot T_r} = \frac{\mu_m}{S + K_S} \cdot \frac{S}{a} \quad (11.23)$$

where $T_r = \text{retention time} = \frac{Q}{V}$, and $S_r = S_i - S$.

For a first-order reaction:

$$\frac{S_r}{X \cdot T_r} = K_1 \cdot S \quad (11.24)$$

$K_1 = \text{the rate coefficient}$ and can be found by laboratory experiments as mentioned previously (see p. 169).

If this is not possible an approximate K_1 value must be used. For most organic compounds K_1 is in the range 0.00012-0.00076, average 0.00024 1/mg h.

The distribution of K_1 values for several organic compounds which are present in some types of industrial waste water is shown in Fig. 11.6. Here the K_1 value is plotted against the percentage probability and it is possible, on the basis of general rules for the biodegradability of different organic compounds, to estimate an approximate K_1 value based on this plot.

The design of a biological treatment plant is based on the organic loading, which is defined as

$$OL = \frac{24 \cdot S_i}{X \cdot T_r} \quad (11.25)$$

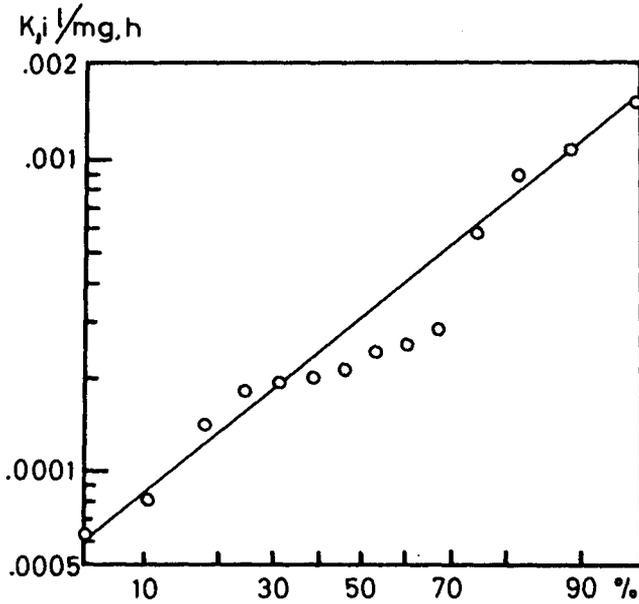


Fig. 11.6. The distribution of K_1 values for several organic compounds.

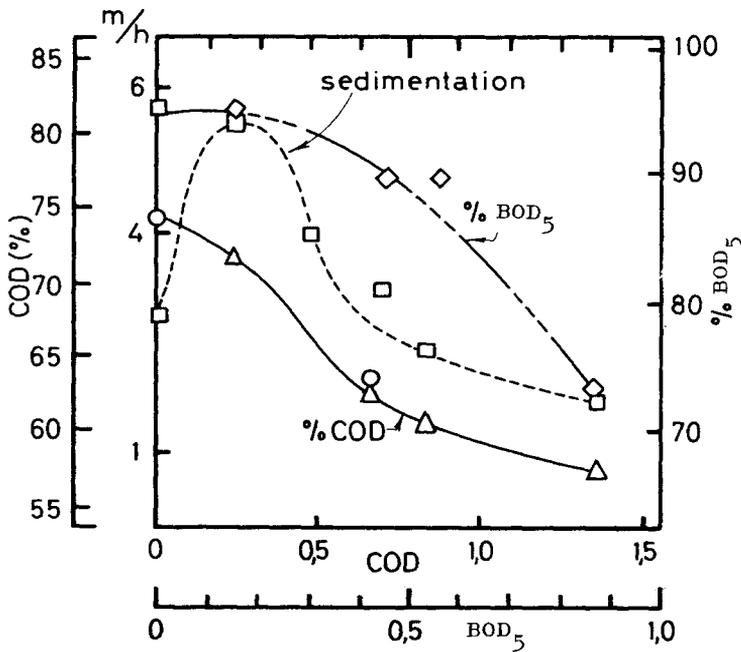


Fig. 11.7. Organic loading (COD or BOD_5) is plotted against the settling rate and efficiency.

The value of the organic loading should be selected to give a good efficiency in the removal of BOD_5 , and, simultaneously, a sludge able to settle. The relationship between the settling rate and the efficiency on the one hand and the organic loading on the other is shown in Fig. 11.7. As can be seen, an organic loading of 0.2-0.5 should be chosen (based on BOD_5). If the organic loading is increased above this limit, the settling rate and the efficiency will decrease rapidly. If the value of the organic loading is chosen below this limit, the sedimentation will be too slow.

The design procedure based on equation (11.24) and (11.25) is:

1. On the basis of a known S_i value and K_1 value, and from the required S value, $X \cdot T_R$ is calculated (equation 11.24).
2. X and T_R are chosen in accordance to both equations (11.24) and (11.25). X will generally be in the range 2000-4000 mg/l, and the higher the value of S_i the higher must X be.
3. If the organic loading found from equation (11.25) is not of a suitable value (use Fig. 11.7), new values of X_1 , T_R and S must be selected until an acceptable organic loading is achieved.

The production of sludge (per 24h, for example) can be found from the following equation:

$$\Delta X = S_0 + a \cdot S_r - b \cdot X_d \quad (11.26)$$

where

a and b = coefficients (see Table 11.4)

S_0 = the influx of volatile suspended solids not degraded (e.g. expressed as kg $BOD_5/24h$)

S_r = BOD_5 removed (expressed in kg removed per 24h)

X_d = the biodegradable mixed-liquor suspended solids per 24h. With good approximation X_d can be replaced by X multiplied by $m^3/24h$, except when considering the extended aeration process

The oxygen consumption per 24h can be calculated from the following equation:

$$O = a_1 \cdot S_r + b_1 \cdot X_d \quad (11.27)$$

where a_1 and b_1 = coefficients (see Table 11.5).

As mentioned previously, the sludge age can be found from equation (11.18).

TABLE 11.5

Coefficients and constants for the biological treatment of industrial waste water

Activated sludge	Range	Average
K_1 (1/mg · h)	0.00012-0.00076	0.00024
a	0.31-0.72	0.52
a_1	0.31-0.76	0.53
b	0.02-0.18	0.07
b_1	0.10-0.24	0.17
OL	0.2 -0.5	0.3
Aerated lagoons:		
$K_{20}^{24h^{-1}}$	0.5 -1.0	0.75
q	1.085-1.100	1.098

Aerated lagoons can be designed using the following relationship:

$$S = \frac{S_i}{1 + K_1 \cdot T_r} \quad (11.28)$$

where

S = COD or BOD₅ for the effluent

S_i = BOD₅ or COD for the influent

K_1 = coefficient, which is in the range of 0.5-1.0/24h
(see Table 11.3)

T_r = the average retention time for the waste water

The K value is greatly influenced by the temperature:

$$K_1 = K_{20} \cdot q^{t-20} \quad (11.29)$$

The values of q and the constant K_{20} can be found from Table 11.5.

By substituting the known value for the temperature, K_{20} , S_i and S , T_r can be calculated.

BOD₅ removal in a trickling filter is related to the area of biological slime surface available and to the time of contact of the waste water with that surface.

The general relationship for the biological reaction is:

$$\frac{S}{S_0} = e^{-K_1 \cdot X \cdot t} \quad (11.30)$$

In a trickling filter the mean time of contact, t , has been found to be:

$$t = \frac{m \cdot D}{Q^n} \quad (11.31)$$

where D = depth, Q = flow and m and n = characteristic constants for the filter media.

Since the biological solid concentration, X , in a trickling filter is proportional to the specific area of surface, which is constant for a specific packing, equation (11.30) can be modified for a trickling filter:

$$\frac{S}{S_0} = e^{-K^1 \cdot D/Q^n} \quad (11.32)$$

where K^1 = a new constant.

K^1 and n can be found from experiments. In cases where experimental data are not available, Table 11.6 can be used.

TABLE 11.6

Trickling filter coefficients

Waste Water	K^1	n
Sewage	0.079	0.5
Fruit canning	0.018	0.5
Boxboard	0.020	0.5
Textile	0.016-0.040	0.5
Steel coke plant	0.021	0.5
Pharmaceutical	0.029	0.5
Slaughterhouse	0.025	0.5

The percentage BOD removal in a trickling filter is in accordance to equation (11.32) given as:

$$\% \text{BOD}_{\text{removal}} = (1 - e^{-K^1 \cdot D/Q^n}) \cdot 100 \quad (11.33)$$

Investigations on several kind of packings have shown, that

$$m = m^1 \cdot A_v^p \quad (11.34)$$

m^1 has a value of 0.7 for rocks and plastic media without slime, and p has a value of 0.75 (Eckenfelder et al., 1963). A_v is the specific surface area.

Equations (11.31) and (11.34) can be combined to give a general expression for the mean retention time for any type of filter packing.

The mean retention time increases substantially if filter slime is present (see Eckenfelder et. al., 1963).

Example 11.1

The design data for an activated-sludge plant treating 100 m³ industrial waste water per hour are required.

The BOD₅ of the effluent is 1000 mg/l.

If possible, an effluent BOD₅ of 50 mg/l would be desirable.

Solution

From equation (11.24), we find that $X \cdot T_r = 160000$.

If we choose $X = 6400$ mg/l; $T_r = 25$ hours.

$$OL = \frac{24 \cdot 1000}{160,000} = 0.15.$$

From Fig. 11.7 it can be seen that this is a suitable value.

$$\text{Control of OL: } \frac{250}{6400} = 0.15.$$

Example 11.2

Find the design data for an aerated lagoon treating 200 m³ waste water per 24 hours.

The influent BOD₅ is 800 mg/l and the effluent BOD₅ is 200 mg/l.
 $T = 10^\circ\text{C}$.

The data from Table 11.5 can be used.

Solution

$$K_{10} = 1.098^{-10} \cdot 0.75 = 0.3,$$

$$200 = \frac{800}{1 + 0.3T_r},$$

$$T_r = 10 \text{ days.}$$

The volume of the aerated lagoon must be 2000 m³.

REFERENCES

- Basu, A.K., 1970. Contribution to Studies on Treatment of Distillery Waste Water. Doctoral Thesis, University of Liege.
- Bernhart, E.L., 1975. Nitrification in industrial treatment works. 2nd International Congress on Industrial Waste Water and Wastes, Stockholm, Feb. 4-7, 1975.
- Ching, M.H., Parrison, J.W. and Minear, R.E., 1975. Heavy metal uptake by activated sludge. J. W. P. Contr. Fed., 47: 362.
- Downing, A., 1966. Advances in water quality improvement, 1: University of Texas Press, Austin, Texas, April 1966.
- Downing, A., 1966. Population Dynamics in biological systems. Proc. 3rd Intern. Conf. Water Pollution Research, Munich, 1966, Water Pollution Control Federation, Washington D.C.
- Eckenfelder, W.W. and Barnhart, E.L., 1963. J. Water Poll. Contr. Fed., 35: 12, 1535.
- Eckenfelder, W.W., 1969. Manual Treatment Processes. Part 1. Environmental Science Inc., U.S.A.
- Ford, D. and Reynolds T., 1965. Proc. 3rd Ind. Waste Conf., Dallas, Texas.
- Francis C.W. and Callahan, M.W., 1975. Biological Denitrification and its Application in Treatment of High-Nitrate Waste Water. J. Environ. Qual., 4: 153-163.
- Lawrence A. and McCarty P.L., 1967. Kinetics of Methane Fermentation in Anaerobic Waste Treatment. Tech. Rep. 75, Department of Civil Engineering, Stanford University, Stanford, California, Feb. 1967.
- Miyaji, I. and Cato, K., 1975. Biological treatment of industrial waste water by using nitrate as an oxygen source. Water Research, 9: 95.
- Neufeld, R.D. and Hermann, E.G., 1975. Heavy metal removal by acclimated activated sludge. J. W. P. Contr. Fed., 47: 310.
- Wing, A. Bruce and Steinfeld, William M., 1970. Comparison of stone-packed and plastic-packed trickling filters. J. W. Poll. Contr. Fed., 42: 255.
- Wuhrmann, K., 1956. In Biological Treatment of Sewage and Industrial Waste, Vol. I, J. McCabe and W.W. Eckenfelder, eds., Reinhold, New York.