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## Improvements in Biofilm Processes for Wastewater Treatment

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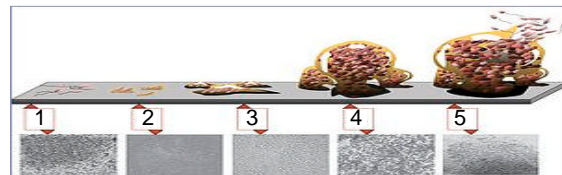
**Abstract:** This review paper intends to provide an overall vision of biofilm technology as an alternative method for treating waste waters. This technology has been gaining popularity through the years, mainly because many wastewater treatment plants, which are still used Activated Sludge Process (AS) are present some shortcomings when exposed to increased hydraulic and organic loads. Fundamental research into biofilms is presented in three sections, Biofilm Types and Characterization, Advantages and Drawbacks and Design Parameters. The reactor types covered in this review are: un-submerged fixed film systems (trickling filters and rotating biological contactors) and submerged fixed film systems (biological aerated flooded filters, submerged aerated filters, biofilm up-flow sludge blanket, fluidized bed, expanded granular sludge blanket, biofilm airlift suspension, internal circulation, moving bed biofilm and membrane biofilm) reactors.

**Key words:** Biofilm, fixed-biofilm reactors, submerged fixed film systems, percolating filters, trickling filters, rotating biological contactors, biological aerated flooded filters, submerged aerated filters, biofilm up-flow sludge blanket, fluidized bed, expanded granular sludge blanket, biofilm airlift suspension, internal circulation, moving bed biofilm, membrane biofilm

### INTRODUCTION

A biofilm is an aggregate of microorganisms in which cells adhere to each other on a surface. These adherent cells are frequently embedded within a self-produced matrix of Extracellular Polymeric Substance (EPS). Biofilm EPS, which is also referred to as slime (although not everything described as slime is a biofilm), is a polymeric conglomeration generally composed of extracellular DNA, proteins and polysaccharides. Biofilms may form on living or non-living surfaces and can be prevalent in natural, industrial and hospital settings. The microbial cells growing in a biofilm are physiologically distinct from planktonic cells of the same organism, which, by contrast, are single-cells that may float or swim in a liquid medium. Microbes form a biofilm in response to many factors, which may include cellular recognition of specific or non-specific attachment sites on a surface, nutritional cues, or in some cases, by exposure of planktonic cells to sub-inhibitory concentrations of antibiotics. When a cell switches to the biofilm mode of growth, it undergoes a phenotypic shift in behavior in which large suites of genes are differentially regulated (Hall-Stoodley *et al.*, 2004). There are five stages of biofilm development as shown in Fig. 1 (Kaplan *et al.*, 2003):

1. Initial attachment:
2. Irreversible attachment:
3. Maturation I:
4. Maturation II:
5. Dispersion:



Five stages of biofilm development: (1) Initial attachment, (2) Irreversible attachment, (3) Maturation I, (4) Maturation II and (5) Dispersion. Each stage of development in the diagram is paired with a photomicrograph of a developing *P. aeruginosa* biofilm. All photomicrographs are shown to same scale.

Fig. 1: Biofilm development stages (Kaplan *et al.*, 2003)

Wastewater is not just sewage. All the water used in the home that goes down the drains or into the sewage collection system is wastewater. This water from baths, showers, sinks, dishwasher, washing machines and toilet. Small businesses and industries often contribute large amounts of wastewater to Sewage collection systems; others operate their own wastewater treatment systems. In combined municipal sewage system, water from storm drains is also added to the municipal wastewater stream and then met in wastewater treatment plant.

Wastewater is about 99.94% water, with only 0.06% of the wastewater dissolved and suspended solid material. The cloudiness of wastewater is caused by suspended particles which in untreated wastewater

ranges from 100 to 350 mg/l. A measure of the strength of the wastewater is biochemical oxygen demand, or BOD<sub>5</sub>. The BOD<sub>5</sub> measures the amount of oxygen microorganisms require in five days to break down wastewater. Untreated wastewater has a BOD<sub>5</sub> ranging from 100 mg/l to 300 mg/l. Pathogens or disease-causing organisms are present in wastewater. Coliform bacteria are used as an indicator of disease-causing organisms. Wastewater also contains nutrients (such as ammonia and phosphorus), minerals and metals. Ammonia can range from 12 to 50 mg/l and phosphorus can range from 6 to 20 mg/l in untreated wastewater (Mancl, 1996; Eckenfelder, 1989).

Conventional wastewater treatment consists of a combination of physical, chemical and biological processes and operations to remove solids, organic matter and sometimes, nutrients from wastewater. General terms used to describe different degrees of treatment, in order of increasing treatment level, are preliminary, primary, secondary and tertiary and/or advanced wastewater treatment. After these treatments municipal wastewater is usually disinfected using chlorine (or other disinfecting compounds, or occasionally ozone or ultraviolet light).

The objective of secondary treatment is the further treatment of the effluent from primary treatment to remove the residual organics and suspended solids. In most cases, secondary treatment follows primary treatment and involves the removal of biodegradable dissolved and colloidal organic matter using aerobic biological treatment processes. Aerobic biological treatment is performed in the presence of oxygen by aerobic microorganisms (principally bacteria) that metabolize the organic matter in the wastewater, thereby producing more microorganisms and inorganic end-products (principally CO<sub>2</sub>, NH<sub>3</sub> and H<sub>2</sub>O). Several aerobic biological processes are used for secondary treatment differing primarily in the manner in which oxygen is supplied to the microorganisms and in the rate at which organisms metabolize the organic matter.

High-rate biological processes are characterized by relatively small reactor volumes and high concentrations of microorganisms compared with low rate processes. Consequently, the growth rate of new organisms is much greater in high-rate systems because of the well-controlled environment. The microorganisms must be separated from the treated wastewater by sedimentation to produce clarified secondary effluent. The sedimentation tanks used in secondary treatment, often referred to as secondary clarifiers, operate in the same basic manner as the primary clarifiers. The biological solids removed during secondary sedimentation, called secondary or biological sludge, are normally combined with primary sludge for sludge processing.

Primary and secondary treatment removes the majority of BOD and suspended solids found in waste waters. However, in an increasing number of cases this level of

treatment has proved to be insufficient to protect the receiving waters or to provide reusable water for industrial and/or domestic recycling. Thus, another techniques like biofilm techniques have been used to provide for further organic and solids removals or to provide for removal of nutrients and/or toxic materials.

Microbial cell aggregates, such as flocs and biofilms, are of great interest in biotechnology. They offer advantages, with respect to suspended single cells, in downstream processing, by facilitating cell-liquid separation by sedimentation or filtration. The term floc is used to refer to an assemblage of individual cells and micro-colonies occurring under specific reactor conditions or after addition of various agents to the medium (Boonaert *et al.*, 1999). A biofilm can be defined as a complex coherent structure of cells and cellular products, like extra-cellular polymers (Characklis, 1990), which either form spontaneously as large, dense granules (Lettinga *et al.*, 1980), or grow attached on a static solid surface (static biofilms) or on suspended carriers (particle supported biofilms) (Heijnen, 1984). Microbial aggregates (either in the form of biofilms, granules or flocs) and the bulk culture medium constitutes two distinct phases. This key feature has three major consequences:

1. Biomass retention can be used to improve reactor volumetric conversion capacity when the conversion is limited by the amount of biomass present. If no biomass retention is applied (e.g. in the standard chemo stat), the biomass concentration depends only on the substrate concentration in the feed and consequently large retention times are required in the presence of diluted feeds. Depending on the settling characteristics of the aggregates, biomass can be readily separated (e.g. by sedimentation) from the bulk liquid and retained in the bioreactor. In this respect, granules and particle-supported biofilms have an extra advantage in that they can be more easily separated than flocs (i.e. higher biomass concentration possible) and have a high reactor specific surface area (i.e. a large mass transfer area than static biofilms).
2. The substrates (e.g. oxygen, carbon and nitrogen sources) have to cross the aggregate-liquid interface and be transported through the aggregate to reach the microbial cells and be consumed. This transport is in general by diffusion and results in a concentration gradient within the aggregate. The penetration depth of substrates in biofilms mainly depends on the porosity of the biofilm, substrate concentration in the bulk liquid, mass transfer at the biofilm-liquid interface and reaction rate in the biofilm. For poorly soluble substrate (e.g. oxygen) the penetration depth is shallow (typically 100-150 mm for oxygen) (Denac *et al.*, 1983).

3. Due to diffusional substrate concentration gradients, a growth rate gradient also exists within the aggregate. In multi-species biofilm systems this will lead to a biofilm with a layered structure, where the organisms with the highest growth rate will be found at the outside of the biofilm, whereas slower growing organisms will be found inside (Heijnen *et al.*, 1989). As a result of this organization, slower-growing organisms will be protected from external shear forces and are less likely to be lost due to detachment and wash-out. In this case not the absolute maximum growth rate of the organisms should be considered but the maximum growth rate under conditions in the reactor (e.g. in the presence of an inhibitor).

The extent to which these features are relevant for a specific system depends, among other factors, on the physical and structural properties (e.g. size, density, porosity, settling velocity, etc.) of the aggregates. Due to a smaller size (typically between 10 and 150  $\mu\text{m}$ ) and higher porosity, diffusional transport is generally faster in flocs than in granules or biofilm (whose size is usually in the range 0.5-3 mm). The substrate concentration and biomass distribution gradients are, therefore, less important in flocs than in biofilms. On the other hand, biofilms and granules present better settling properties than flocs (terminal settling velocity of about 40 and 5 m/h for particle-supported biofilms and flocs, respectively) and can be more easily retained in the bioreactors. The physical and structural properties of particle-supported biofilms and granules are similar and so are their hydrodynamic, mass transfer and reaction characteristics.

For the majority of industrial fermentation processes, where high substrate concentrations are used, biofilm formation is either unnecessary or even disadvantageous and the range of applications of immobilized-cell systems in industry is limited. Several applications of biofilm reactors in various biotechnological processes (e.g. fermentation, production of enzymes, production of primary and secondary metabolites, production of antibiotics and bioconversions), have been reviewed by Furusaki (1988); Schugerl (1989, 1997); Godia and Sola<sup>1</sup> (1995). However, with the exception of wastewater treatment processes, the application of biofilm reactors at full industrial scale is scarce (Godia and Sola<sup>1</sup>, 1995).

An extensive use of biofilms is made within the field of environmental biotechnology for three main reasons:

1. Compared with most other industrial bioprocesses, large volumes of dilute aqueous solutions have to be treated.
2. Natural, mixed populations of microorganisms, which readily form biofilms, are used.

3. The process can be operated at high biomass concentration in the reactor, without the need for settlers for biomass retention and recirculation. A polishing step of the effluent is usually needed to remove remaining suspended (detached) biomass.

**Fixed-biofilm reactors:** It is clear, when examining stones or other submerged objects in an enriched river or the wall of a sewer, that micro-organisms will readily colonize any suitable surface provided that sufficient nutrients are present. This principle has been utilized in fixed-film or attached growth systems where the microbial biomass is present as a film which grows on the surface of an inert and solid medium. Purification is achieved when the wastewater is brought into contact with this microbial film. Because the active biomass is largely retained within the reactor there is no need to recirculate any displaced biomass back to the reactor in order to maintain a sufficient density of micro-organisms, as is the case in the activated sludge process. The required contact between the film and the wastewater is achieved in most fixed-film reactors by allowing the wastewater to pass over the stationary medium in which the film has developed. However, it is not essential for the medium to be stationary and in some reactor designs the medium itself moves through the wastewater. Fixed-film reactors are designed as secondary treatment processes to partially treat (high-rate filter, anaerobic filter) or fully treat settled wastewater (percolating filter, rotating biological contractor, submerged filters) and for tertiary treatment to provide nitrification or denitrification. The most widely used fixed-film reactor is the percolating filter, which is considered in detail below. Rotating biological contactors, biological aerated flooded filters and submerged aerated filters are also considered. These processes are all aerobic. However, anaerobic filters, which involve the development of an anaerobic film on a suitable support medium, are also widely used. Anaerobic filters are used for treating wastewater with a high concentration of soluble BOD (>500 mg/l) and which contain little suspended solids. Under anaerobic conditions, the organic matter is converted to gaseous end-products with little sludge being produced. The most widespread use of anaerobic filters is for the removal of oxidized nitrogen from purified effluents by denitrification, with the nitrogen present reduced to elemental nitrogen and released from the filter as a gas. Denitrification requires a carbon source and in the removal of oxidized nitrogen (nitrite, nitrate) from purified effluents this is usually supplied as methanol (Anderson *et al.*, 1984; Bailey and Thomas, 1975). Anaerobic fixed-film reactors are also known as anoxic filters.

Trickling filters look much like un-submerged filter (i.e. they consist of a tube or tank filled with media through which water is passed). However, they are operated

differently in that the free water surface in the filter is maintained below the media. The culture water is pumped to the top of the filter and uniformly distributed over the top of the filter. As the water trickles down over the media it absorbs oxygen from the air in the filter and supplies ammonia and/or nitrite to the bacteria growing on the media. If properly designed, trickling filters rarely plug, they are quite stable over time, but they require some pumping head (at least the height of the filter). Their primary advantage is the oxygen for the bacteria in the filter comes from the air in the filter. Thus, they are well aerated and the water flow rate through the filter is independent of oxygen supply.

Rotating Biological Contactors (RBC) typically are designed in one of two configurations. The first consists of a horizontal shaft that has flat or corrugated circular plates attached to it. The plates are typically spaced at least one cm apart along the shaft. The shaft is attached to bearings and mounted above a tank such that about 40 to 45 percent of each plate surface is below the top of the tank. Waste water is pumped into the tank and the RBC is rotated by a motor such that the plates rotate in and out of the water during each revolution. Bacteria grow on the plate surfaces and as the bacteria enter the water they remove ammonia or nitrite and as they rotate through the air they extract oxygen from the air. The second RBC configuration replaces the disks with a drum filled with some light weight media (e.g. plastic) that has a high surface to volume ratio and a high void ratio. As the drum rotates the bacteria on the media are alternately supplied with ammonia or nitrite from the wastewater and oxygen from the air. RBC are generally quite stable in operation, have a high ammonia removal efficiency compared to some other biofilters and they require very little head loss (typically 2 to 3 cm of water). Their primary disadvantage is that they require a power source to turn them and mechanical breakdown can be a problem, particularly with a poorly designed unit (Hochheimer, 1990; Wheaton *et al.*, 1991).

Fluidized bed biological filters consist of a bed of sand or other heavier than water media that is small in diameter. Water is pumped up through the sand at a fast enough velocity to fluidize the sand (i.e. suspend the sand grains in the vertical column of flowing water). Bacteria grow on the sand grains and as the water passes by the fluidized sand the bacteria extract ammonia and/or nitrite. Fluidized sand filters require a small foot print for the size of the filter because the small sand grains provide a very high specific surface area per unit of volume of filter. These filters require continuous pumping and have an essentially constant pressure drop across the filter the pump must overcome (Summerfelt and Cleasby, 1996).

There are several types of bead filters including those using heavier than water and those using lighter than water beads (Timmons and Losordo, 1994; Delos

Reyes and Malone, 1998). Most systems use a small tank specifically designed to provide the flow and operation desired for the bead filter. The tank typically has an up flow configuration and a screen across the top of the bead bed to prevent the beads from exiting the filter. Wastewater is pumped upward through the bead bed. Bacteria on the bead surfaces provide nitrification of the ammonia and nitrite and the beads provide a screening effect that traps considerable solids. Thus, bead filters can be used as solids removal devices or as bio filters (Beecher *et al.*, 1997).

**Percolating filters (Trickling Filters):** Trickling filters (TFs) are used to remove organic matter from wastewater. The TF is an aerobic treatment system that utilizes microorganisms attached to a medium to remove organic matter from wastewater. This type of system is common to a number of technologies such as rotating biological contactors and packed bed reactors (bio-towers). These systems are known as attached-growth processes. In contrast, systems in which microorganisms are sustained in a liquid are known as suspended-growth processes.

The design and function of biological filters has been described by numerous workers (Bruce, 1969; Warren, 1971; Pike, 1978; Bruce and Hawkes, 1983). In its simplest form, the filter consists of a bed of graded hard material, the filter medium, about 2 m deep. The medium has interstices or voids that allow air and applied wastewater to reach all parts of the bed. The filter has a ventilating system to ensure free access of air to the bed and a distributor to regulate the volume and frequency of application of the sewage (influent) over the surface (Fig. 2).

The medium provides the necessary base for the attachment of non-motile micro-organisms, principally bacteria and fungi, which form a film. Motile organisms, both micro- and macroscopic, live in the shelter of the interstices, feeding on and controlling the accumulated film. The action of this grazing fauna prevents heavy film growths blocking the interstices (Hawkes, 1963), which would cause ponding and anaerobic conditions within the filter bed. The accumulation of the film follows a seasonal pattern, becoming thicker during the winter months. The action of the grazing fauna loosens and breaks down the film, resulting in a large removal of film each spring which is known as sloughing. The nutrients in the wastewater promote the growth of the micro-organisms that comprises the film, thus, as the wastewater percolates downwards over the surface of the film-covered medium, biological oxidation and conversion takes place.

Percolating filters are the most widely used secondary treatment process in Europe. They are equally common in other temperate regions including the USA, where they are employed at over 3700 separate municipal

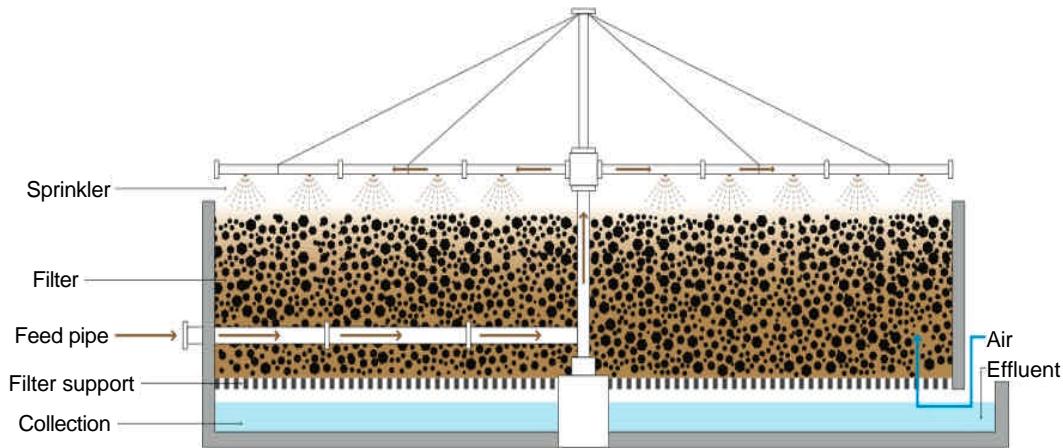


Fig. 2: Basic constructional features of a conventional percolating filter (Tilley *et al.*, 2008)

Table 1: Comparison between the activated sludge and percolating filter processes (Jeger, 1970)

Item	Activated sludge	Percolating filtration
Capital cost	Low	High
Area of land	Low: advantageous where land availability is restricted or expensive.	Large: 10 times more area required.
Operating cost	High	Low
Influence of weather	Works well in wet weather, slightly worse in dry weather, less affected by low winter temperatures.	Works well in summer but possible ponding in winter.
Technical control	High: the microbial activity can be closely controlled; requires skilled and continuous operation.	Little possible except process modifications. Does not require continuous or skilled operation.
Nature of wastewater	Sensitive to toxic shocks, changes in loading and trade waste waters; leads to bulking problems.	Strong waste waters satisfactory, able to withstand changes in loading and toxic discharges.
Hydrostatic head	Small: low pumping requirement, suitable for site where available hydraulic head is limited.	Large: site must provide natural hydraulic head otherwise pumping is required.
Nuisance	Low odour and no fly problems. Noise may be a problem both in urban and rural areas.	Moderate odour and severe fly problem in summer possible. Quiet in operation.
Final effluent quality	Poor nitrification but low in suspended solids except when bulking.	Highly nitrified, relatively high suspended solids.
Secondary sludge	Large volume, high water content, difficult to dewater, less stabilized.	Small volume, less water, highly stabilized.
Energy requirement	High: required for aeration, mixing and maintaining sludge flocs in suspension and for recycling sludge.	Low: natural ventilation, gravitational flow.
Synthetic detergents	Possible foaming, especially with diffuses.	Little foam
Robustness	Not very robust, high degree of maintenance on motors, not possible to operate without power supply.	Very sturdy, low degree of maintenance, possible to operate without power.

sewage treatment plants. Although biofiltration was historically the first process used, it still has certain advantages over the activated sludge process. Filters require virtually no skilled maintenance or close control. In energy terms, percolating filters are more economical than the activated sludge process and are more versatile in responding to changes in flow and character of wastewater (Hawkes, 1963). Filters are more tolerant of continual or shock discharges of certain pollutants compared with activated sludge (Cook and Hering, 1978), including toxic industrial wastes containing heavy metal ions, phenols, cyanides, sulphides and formaldehyde. They are widely used for both total and partial treatment of a wide range of industrial waste waters (Bruce, 1969; Calley *et al.*, 1977; Pike, 1978).

Their major disadvantage is capital cost, being normally uneconomic in serving populations greater than 50000. This is due not only to higher capital costs compared to activated sludge systems but also to the larger area of land they occupy, which is often at a premium in urban areas (Jeger, 1970) (Table 1). For this reason, the activated sludge process predominates at very large sewage treatment works. Although the proportions of the population of England and Wales served by these two bio-oxidation processes are about the same, many more of the 5000 or so sewage treatment plants use percolating filters rather than the activated sludge process (Institute of Water Pollution Control, 1972). Bruce (1969) concluded, in his review on percolating filtration, that there was no indication that the use of

percolating filtration was likely to be outmoded and this remains true even with the introduction of new processes and the development of packaged plants. However, the majority of new domestic and municipal sewage treatment plants built since 1970 have been of the activated sludge type. Small domestic plants serving populations of <2000 are often of the rotating biological contactor or submerged aerated filter design. Reed beds are increasingly being used for secondary and tertiary treatment for small rural plants serving <1000 PE. Parker (1999) examines the future role of percolating filters.

The term percolating filter in Ireland and the UK is still used, although there are many derivations of the name such as biological filter, bacteria bed, percolater and in the USA, trickling filter. However, the term filter often causes confusion as the process is essentially biological, even though there is some physical removal of fine solids. The confusion arises because when the process was first developed it was thought that purification was brought about by physical filtration, as in a sand filter and percolating filtration was developed from such physical filter processes. Experiments were carried out at the Lawrence Experimental Station of the Massachusetts State Board of Health (USA) using small filter beds to test the efficiency of various soils and aggregates, e.g. different grades of sand and gravel, to physically remove solids from wastewater. They found that effective purification of settled sewage could be achieved using quite coarse gravel (19-25 mm grading) and at very high rates of application (0.2-0.5 m<sup>3</sup>/m<sup>2</sup>.d), so that it was not always necessary to use sand filtration or land treatment, both of which required much lower hydraulic loading rates to be as effective (Mills, 1890). Although it was quickly realized that this new type of treatment must involve a degree of biological oxidation, it was not until the process was well established that the physical role of filtration was found to be minor in comparison to the biological function, by which time the terms, trickling and percolating filtration had become established (Stanbridge, 1976; Bruce and Hawkes, 1983).

**Design of percolating filters:** The design of percolating filters has changed very little since they were first introduced and with a working life of 80 years, many of the original filters are only now reaching the end of their useful lives. However, replacement is more often due to an increase in the loading in excess of their original design, rather than structural or mechanical failure. The major factors that need to be taken into account in the design and operation of percolating filters are: medium type and in particular, specific surface area and voidage; depth; area of the bed; organic and hydraulic loading rates.

The depth of filters is arbitrary and when all types of percolating filtration are taken into consideration there is a total range of between 0.9 and 15.0 m. The standard (low-rate) percolating filter in the UK is usually about 1.8 m deep and rarely less than 1.5 m or in excess of 2.5 m. Filter beds in the rest of Europe tend to be deeper than this and in Germany most filters are between 3-4 m in depth. Low-rate filters in the USA are a similar depth to those in the UK, at between 1.5-2.0 m, although shallow beds (<1 m) are used for high-rate filtration. High-rate filters using modular plastic medium are used to treat industrial waste waters. The medium is very light and can be stocked, like building blocks, into a tall tower which is essentially free-standing. These filter towers are housed in lightweight prefabricated material or are built out of breeze blocks and can be constructed as high as 12-15 m.

In terms of treatment efficiency, the majority of BOD and suspended solids removal occurs within the top 750 mm of the filter bed, which is also about the minimum depth when using conventional mineral medium in order to avoid short circuiting of wastewater through the bed. Apart from constructional costs, the major limiting factor in the selection of the depth of filters is the loss of hydraulic head, with the greater the depth the greater the loss of head occurring. Unless the plant is built on a slope, pumping will be required. From experience, it would appear that a depth of 1.8 m, when using conventional medium, is a good compromise between treatment efficiency, loss of hydraulic head and cost.

There appears to be a wide variety in the configuration, or plan, of percolating filters (Learner, 1975), although it is dependent to a large extent on the site and the total loading. They are usually either rectangular or circular in plan. Rectangular filters are normally used for large populations because they are more compact and are cheaper to construct because they share retaining walls. Maximum size of such filters rarely exceeds 10 m in width and 100 m in length. They can also be used for very small populations (<40 people) when the rectangular shape is more compatible with the mechanical distribution system used. This comprises of a tipping trough that discharges the wastewater intermittently into a fixed set of distribution channels that are laid across the filter. Circular plan filters are generally preferred as they allow greater control over the frequency of wastewater application. The maximum size of these filters is limited by the maximum diameter of rotating distributor available, which is 40 m.

Percolating filters are built on a concrete base covered with drainage tiles that provides a raised floor on which the medium rests (Fig. 2). This provides an unrestricted area of under drainage, which takes the final effluent away. It is important to ensure that the under drainage area is deep enough to allow the passage of air from

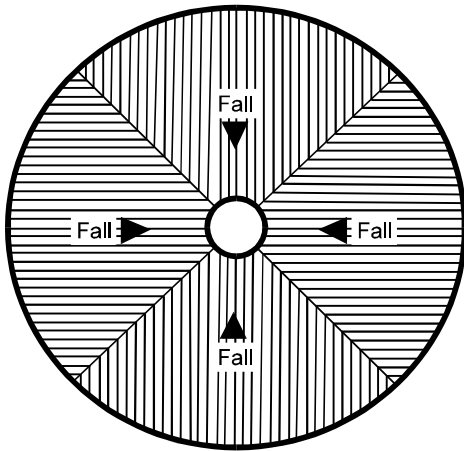


Fig. 3: Typical arrangement of filter floor tiles draining to a central well (IWEM, 2000)

ventilation pipes lining the base of the filter bed. The ventilation pipes are connected to the atmosphere so that there is a free flow of air through the whole filter, providing the necessary aeration. Natural convection causes aeration because of the difference between the air temperature and the temperature within the bed. The concrete base is sloped to at least 1:100, normally 1:50, to ensure a minimum flow of  $0.6 \text{ m s}^{-1}$ , which prevents settlement and the accumulation of solids washed from the medium. Drainage tiles can be replaced by a series of half round field drains laid out in a herringbone pattern in rectangular filters, or arranged radiating out from the centre of circular shaped filters to a central drainage channel or well (Fig. 3) (IWEM, 2000). As is the case with the tiles, the drains must be of sufficient size to allow for adequate ventilation. For conventional single-pass filtration, where settled sewage passes through a single percolating filter bed before having the solids (humus) removed in a secondary sedimentation (humus) tank, loading can be categorized as either low- or high-rate. Hydraulic loading is expressed in cubic metres of settled wastewater per cubic metre of filter medium per day ( $\text{m}^3/\text{m}^3.\text{d}$ ) so that loadings  $<3 \text{ m}^3/\text{m}^3.\text{d}$  are classed as low-rate and  $>3 \text{ m}^3/\text{m}^3.\text{d}$  as high-rate filtration. However, there is no precise demarcation between low and high rate filtration and these figures are only guidelines. Hydraulic loading is not always expressed as a volumetric loading. It is also expressed as a surface loading, which is often referred to as the irrigation rate, in cubic metres of wastewater per square metre of superficial bed area per day ( $\text{m}^3/\text{m}^2.\text{d}$ ). The hydraulic loading rate depends solely on the physical characteristics of the medium and the degree of film accumulation within the bed. The upper hydraulic limit is determined by the onset of flooding because the wastewater is being applied at a faster rate than it can percolate through the bed. Where

this occurs then the interstices of the medium become flooded and the access of air to the interior of the bed is prevented. The lower limit is known as the minimum wetting rate, which is the loading rate at which the medium is kept sufficiently moist to prevent drying and the microbial film to remain active (Winkler, 1981). Organic loading is measured as the BOD per cubic metre of filter medium per day ( $\text{kg BOD}/\text{m}^3.\text{d}$ ) with low-rate filtration classed as filters receiving loads of  $<0.6 \text{ kg BOD}/\text{m}^3.\text{d}$  and high rate  $>0.6 \text{ kg BOD}/\text{m}^3.\text{d}$ . In practice, in order to produce a Royal Commission standard effluent (20 mg/l BOD, 30 mg/l suspended solids) after settlement with a high degree of nitrification, filters treating domestic sewage should receive an organic loading of between  $0.07\text{-}0.10 \text{ kg BOD}/\text{m}^3.\text{d}$  and a hydraulic loading between  $0.12\text{-}0.60 \text{ m}^3/\text{m}^3.\text{d}$ . Generally, increases in organic loading in excess of  $0.10 \text{ kg BOD}/\text{m}^3.\text{d}$  will result in heavier film growths, which may result in ponding. In an attempt to produce more efficient percolating filters, which would operate at much higher loadings, a number of modifications of the basic process have been adopted. By using larger mineral medium, for example, greater loads can be applied to filters without the risk of ponding. Such high rate filtration will produce a 20:30 effluent with an increased hydraulic loading of up to  $1.8 \text{ m}^3/\text{m}^3.\text{d}^{-1}$ , but with little or no nitrification. If a less stabilized effluent is required, such as roughing treatment for strong industrial wastes, then loadings of up to  $12 \text{ m}^3/\text{m}^3.\text{d}$ , with organic loads up to  $1.8 \text{ kg BOD}/\text{m}^3.\text{d}$  will give 60-70% removal.

**Rotating Biological Contactors (RBC):** A Rotating Biological Contactor (RBC) is a biological treatment process used in the treatment of wastewater following primary treatment. The primary treatment process removes the grit and other solids through a screening process followed by a period of settlement. The RBC process involves allowing the wastewater to come in contact with a biological medium in order to remove pollutants in the wastewater before discharge of the treated wastewater to the environment, usually a body of water (river, lake or ocean). A rotating biological contactor is a type of secondary treatment process. It consists of a series of closely spaced, parallel discs mounted on a rotating shaft which is supported just above the surface of the waste water. Microorganisms grow on the surface of the discs where biological degradation of the wastewater pollutants takes place. A Rotating Biological Contractor (RBC) is an attached-growth biological process that consists of one or more basins in which large closely-spaced circular disks mounted on horizontal shafts rotate slowly through waste-water. The disks, which are made of high-density polystyrene or Polyvinyl Chloride (PVC), are partially submerged in the wastewater, so that a bacterial slime layer forms on their wetted surfaces. As the disks rotate,



the bacteria are exposed alternately to waste-water, from which they adsorb organic matter and to air, from which they absorb oxygen. The rotary movement also allows excess bacteria to be removed from the surfaces of the disks and maintains a suspension of sloughed biological solids. A final clarifier is needed to remove sloughed solids. Organic matter is degraded by means of mechanisms similar to those operating in the trickling filters process. Partially submerged RBCs are used for carbonaceous BOD removal, combined carbon oxidation and nitrification and nitrification of secondary effluents. Completely submerged RBCs are used for denitrification.

Rotating Biological Contactors (RBC) became commercially available in 1965, although the system was first developed in the late 1920s (Doman, 1929). They are now widely used throughout the world and although particularly well suited for treating wastewater from small communities, they are now also being used to treat large domestic and industrial loads. RBCs can be operated under aerobic, anoxic, or anaerobic conditions and can be used for a wide variety of applications. However, in practice, they are almost exclusively used for aerobic secondary treatment. RBCs are widely available as packaged plants under a variety of commercial names such as Biosurf, Biodisc, Biodrum and Biospiral. The basic design consists of a series of flat or corrugated discs 2-3 m in diameter mounted on a horizontal shaft and driven mechanically

so that the discs rotate at right angles to the flow of settled wastewater (Fig. 4).

The discs are usually plastic (MDPE), corrugated polythene, PVC, GRP, or expanded polystyrene, although other materials such as asbestos cement and expanded metal are also used. The surface area of the medium is normally between 150-200 m<sup>2</sup>/m<sup>3</sup> with the higher surface area medium used at the outlet end for nitrification. Each disc is 10-20 mm thick and spaced about 20 mm apart (Fig. 5). The mounted discs are placed in a contoured tank that fits fairly closely to the rotating medium so that 35-40% of their area is immersed and are slowly but continuously rotated.

At immersion depths >40%, supplementary air may be required to maintain aerobic conditions. Discs are usually but not exclusively used as support media. Other configurations, such as lattice constructions or wire mesh containers filled with random plastic media have also been used successfully. The flow of wastewater through the tank and the action of the rotating medium produces a high hydraulic shear on the film that ensures efficient mass transfer from the liquid into the film and at the same time, controls excessive film accumulation. Discs are arranged in groups separated by baffles to minimize short circuiting and the effect of surges of flow. Normally, a minimum of four compartments, separated by baffles, are used to simulate plug flow with a small head loss of between 10-20 mm between each compartment. With large installations, plug-flow

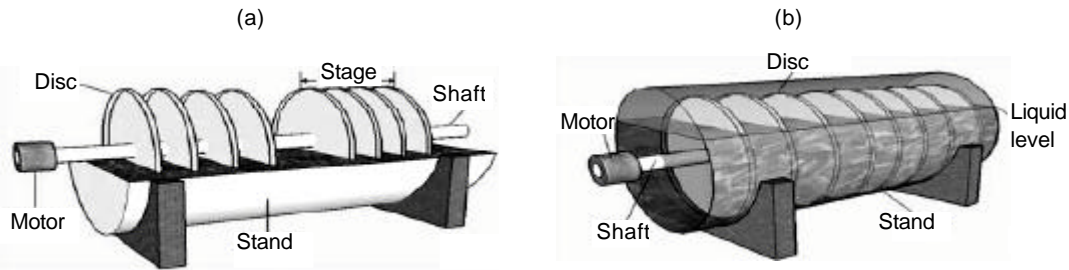


Fig. 4: Schematic diagram of RBC units: (a) conventional RBC with two-stages; (b) single-stage closed RBC with high submergence level (Patwardhan, 2003)

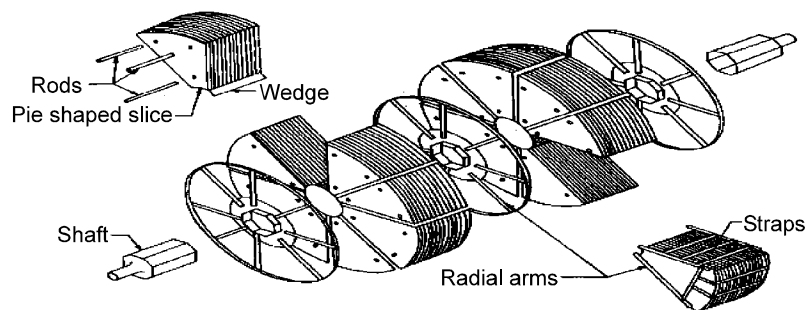


Fig. 5: Schematic view of disc assembly in a rotating biological contactor showing support rods that ensure exact spacing between discs (IWEM, 2000)

conditions are achieved by having several complete RBC units in series (Steels, 1974).

The system consumes very little energy as the lightweight plastic media is evenly balanced throughout the drive shaft and a small motor is used to drive the rotor, which results in a low power consumption for the amount of BOD removed. For example, in a 300 PE unit, the power consumption of the motor is only 0.3 kW. RBC units are invariably covered, with small units covered with a moulded plastic or GRP cover and larger units housed in buildings. Covering the media is important in order to protect the exposed film on the discs from the weather, especially frost and wind which can damage the film. Wind can also increase the load on the motor driving the rotor. Covering insulates the system and reduces heat loss and increases the rate of oxidation. By insulating the cover such units can operate successfully even in arctic conditions. The cover also eliminates fly nuisance and even controls odours to some extent. Using green covers makes such plants inconspicuous, especially as they have the same head loss as a septic tank (<100 mm) and therefore are low form structures. They are ideal for use in open areas where landscape quality is important, for example, golf clubs, hotels and general amenity areas.

As the discs rotate, wastewater enters the spaces between the discs when immersed and is then replaced with air when out of the liquid. In this way, a biological film builds up on the discs in the same way as it develops on the medium in a percolating filter. Therefore, purification occurs with the film alternately adsorbing organic nutrients from the wastewater and then obtaining oxygen from the atmosphere for oxidation. Separation of the discs varies between 15-30 mm which is a compromise between reducing the spacing to maximize surface area per metre length of rotor, while preventing film from bridging the gap between discs so reducing the effective surface area and leading to low dissolved oxygen within the film. This is especially a problem when shearing velocities are too low to dislodge excess film. A survey conducted on 114 RBCs by Severn-Trent Water showed film thickness to vary between 0.5-4.5 mm with significantly thicker films developing on inlet discs compared to those at the outlet (Fig. 6) (Griffin and Findlay, 2000). Allowing a minimum spacing of 15 mm for water to pass between discs and 5 mm or 3 mm for maximum permissible film development on the inlet and outlet discs respectively, they used 25 and 21 mm for inlet and outlet disc separation distances respectively. The thickness of the film is managed by preventing organic overloading. Using these spacings the approximate surface area for a 1.0 m diameter disc unit is 50-80 m<sup>2</sup>/m rotor length rising to 700 m<sup>2</sup>/m for a 3.0 m diameter disc unit. The average depth of disc immersion was 40%.

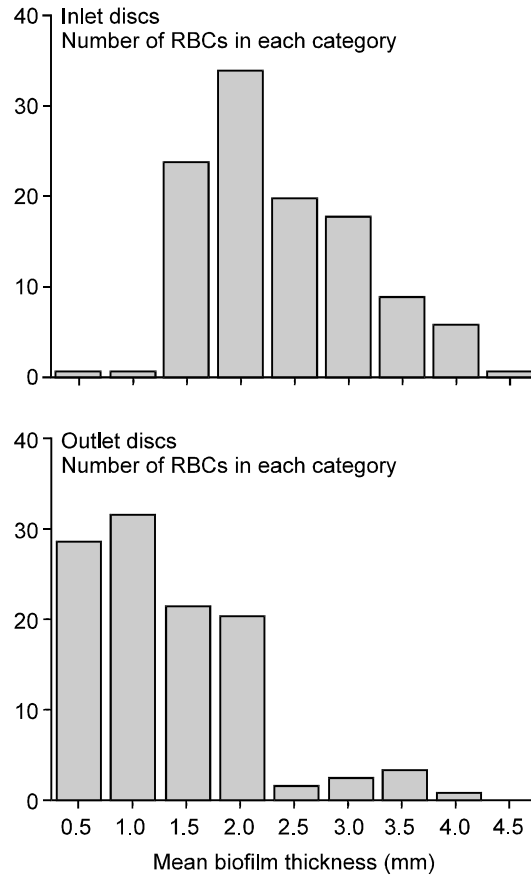


Fig. 6: Mean film thickness on the inlet and outlet discs at 114 rotating biological contactor sites (Griffin and Findlay, 2000)

RBC systems due to their advantages (Table 2) constitute a very unique and superior alternative for biodegradable matter and nitrogen removal. Depending on the organic loading, nitrification can occur on the first group of discs, although nitrifying activity is usually limited. Like percolating filters, nitrifying bacteria in RBCs are found towards the end of the system on the discs in the third and fourth compartments of a four-compartment unit. Due to the plug flow nature of the system there is a tendency for nitrite to accumulate in the compartment where nitrification first occurs. There is a net loss of nitrogen through the system which approaches 20% because of denitrification at the media-film interface (Ellis and Banaga, 1976). Some designs incorporate recirculation of final effluent back to the first stage. The idea behind this is to supply combined oxygen in the heavily loaded first stage to prevent septicity and to allow denitrification. The nitrogen oxidation rate is similar to that found in filters at approximately 1 g NH<sub>3</sub>-N/m<sup>2</sup>.d at 20°C (Antonie, 1976). Major variables affecting nitrification include total disc area, wastewater temperature, influent nitrogen and

Table 2: General advantages and drawbacks of the RBC process (Cortez, 2008)

Advantages	Drawbacks
Land requirement relatively small	Difficult scale-up
Easy construction and expansion	Slow process start-up
Compact design with separate compartments	Adequate primary treatment and secondary clarifier required
Simple process control and monitoring	Limited process flexibility
Low operating and maintenance cost	
Short HRTs	
High OTE	
High biomass concentration per volume reactor	
Low sludge volume index values in the second clarifier	
No requirement of sludge recirculation	
Resistance to shock and toxic loads	
No problems with malodours and flies	

BOD concentrations and flow rate. For example, based on the Van't Hoff-Arrhenius relationship, nitrification at 5°C may require 2.5 times more medium surface area than at 13°C. The speed of rotation is limited by the shearing of the film at the periphery of the discs, but the normal range is between 0.5-10 rpm, typically 0.5-1.5 rpm depending on the diameter of the disc. Film is sloughed off continuously and will remain in the liquid phase adding to the overall treatment process. Eventually, sloughed film is carried away to a separate sedimentation tank although in smaller units the secondary settlement tank is built into the chamber. Most units also have a digestion chamber below the discs to reduce the load to the secondary settlement tank, thereby reducing the quantity and sludge produced and the frequency of desludging.

**Design of Rotating Biological Contactors (RBC):** Design of RBC is very similar to other biofilters. The object is to get the waste water to move past the RBC so the nitrifying bacteria can remove the ammonia and the nitrite from the water. The factors noted above and by several other authors (Hochheimer, 1990; Hochheimer and Wheaton, 1991; Wheaton *et al.*, 1991) effect the operation of an RBC and the bacteria on the RBC the same as they do any other nitrifying bacteria. Because an RBC rotates through both an aqueous and an air phase, the oxygen is supplied by the air and the ammonia and nitrite by the water. The RBC is typically operated with a 35-45% submergence. Thus slightly less than one-half of the time the bacteria will be in the water and slightly more than one-half of the time the bacteria will be in the air. There are several constraints on the rotational speed of the RBC. The bacteria grow on the plate or media surfaces of the RBC. If the rotational speed gets too high the shear forces generated by the plates moving through the water will exceed the adhesion of the bacteria for the plate surface and the bacteria will be stripped off of the plates. Thus maximum rotation speed (i.e. Revolutions per Minute (RPM)) is generally limited by the lineal velocity of the fastest moving part of the plate as it moves through the water. This maximum velocity is ill defined because it is

dependent on the plate surface characteristics (which is a function of the construction materials and the geometric design of the plate or media surfaces), the health and age of the bacteria and other factors. It should also be noted that the maximum lineal velocity is a fixed value for a given application, but one of the design variables is the diameter of the RBC plates or drum. The larger the diameter, the greater the lineal velocity of the outer rim of the plates for a given number of Revolutions per Minute (RPM) of the RBC. For example, a two meter diameter RBC operating a 4 RPM has a much higher lineal speed at the rim of the plates than a 1 meter diameter RBC operating at the same RPM.

Another limit on the speed of rotation of the RBC is related to the oxygen concentration in the wastewater and the drying rate of the air. Any one bacteria must not be left in the water phase so long that it runs out of oxygen before reemerging into the air. Similarity the period of time the bacteria is in the air must not be sufficient to dry the bacteria out so it can no longer function. These two factors place a lower limit on the speed of the RBC.

Between the two limitations discussed above is a wide range of rotational speeds that can be used in design of RBC. Selection of the optimum RPM of the rotor appears to be more of an art than a science. However, Antonie (1976) showed that ammonia removal by RBC was enhanced at peripheral speeds up to 0.305 m/sec (1 ft/sec), but above this value the ammonia removal was constant with increased peripheral speeds. Wortman (1990) in his biodrum studies used 10.37 cm/sec (0.34 ft/sec) peripheral speed. Paloini (1985) showed when RBCs were used for COD (Chemical Oxygen Demand) removal, the removal of soluble BOD, influent BOD and RBC rotational speed interacted. For a given influent BOD with all other variables remaining constant increased rotational speed increased COD removal. Paloini (1985) also concluded that under limiting oxygen transfer conditions, the maximum COD removal rate is an approximately linear function of the square root of the disk rotational speed, regardless of the wastewater type and the RBC system used. Friedman *et al.* (1979)

showed that RBC removal leveled off, the value where the removal leveled off was a function of the influent COD. Weng and Molof (1974) found that nitrification by a RBC increased when speed was increased from 0.1-0.34 m/sec (0.3-1.1 feet/sec). Gilbert *et al.* (1986) found in commercial installations most RBC were driven at a peripheral speed of about 0.3 m/sec (1 ft/sec), but this varied somewhat over the 105 units they surveyed.

The performance of RBC depends upon several design parameters. Particularly significant are: rotational speed, organic and hydraulic loading rates, Hydraulic Retention Time (HRT), RBC media, staging, temperature, biofilm characteristics, Dissolved Oxygen (DO) levels, effluent and medium submergence.

**Rotational speed:** The rotational speed of the RBC media is a very important parameter that affects nutrient and oxygen mass transfer in the biofilm and consequently substrate removal. Usually an increase on the speed of rotation increases the dissolved oxygen concentration available to the microorganisms and as a result they are able to degrade the substrate at a higher rate. However, increasing the rotational speed leads to higher power consumption, which may not be economical for wastewater treatment applications (Ramsay *et al.*, 2006). Besides, if the rotational speed gets too high, the microorganisms will be stripped off the media, deteriorating the effluent quality and lowering the biodegradation rate in the reactor.

Thus, the guiding principle is to adopt the minimum speed commensurate with acceptable treatment. According to Mathure and Patwardhan (2005), typically rotational speeds are 1-10 rpm for RBC media in disc form with discs with 1-4 m diameter mounted on shafts around 5-10 m long.

**Organic loading:** The organic loading of a RBC reactor must be accurately defined during planning and designing. The variation of the organic loading rate is generally accomplished by changing the inlet flow rate or the HRT, which also results in a change in the hydraulic loading (Najafpour *et al.*, 2005). Under normal operating conditions, carbonaceous substrate is mainly removed in the first-stage of the RBC. To avoid oxygen transfer limitations the first stage design load must be limited to a BOD<sub>5</sub> load of about 30 g BOD<sub>5</sub>/m<sup>2</sup> d or to a soluble BOD<sub>5</sub> load of 12-20 g BOD<sub>5</sub>/m<sup>2</sup> d according to WEF and ASCE (WEF and ASCE, 1998). The use of higher first-stage organic loadings will increase the probability of developing problems such as excessive biofilm thickness, depletion of dissolved oxygen, deterioration of process performance, appearance of H<sub>2</sub>S odours and excessive growth of nuisance organisms such as Beggiatoa (Tchobanoglous and Burton, 1991; Grady *et al.*, 1999). Overloading problems can be overcome by removing baffles between the first and second-stages to

reduce surface loading and increase oxygen transfer. Other approaches include supplemental air systems, step-feed, or recycle from the last stage. The organic loading affects nitrification in a RBC unit. In the initial stages, where the organic load is high, heterotrophic bacteria offer strong competition to nitrifiers displacing them within the bioreactor (Brazil, 2006). Nowak (2000) has investigated nitrification in full-scale RBCs (with discs and plastic packages) and proposed that the surface loading rate should not exceed 2.5 g BOD<sub>5</sub>/m<sup>2</sup> d to keep the effluent ammonia concentration below 5 mg NH<sub>4</sub> N/l; at temperatures above 13°C. In the same investigation nitrification rates of 1.5 g N oxidized/m<sup>2</sup> d at 8°C and of 1.8 g N oxidized/m<sup>2</sup> d at 13°C were obtained in tertiary full-scale RBCs with ammonia effluent concentrations mostly below 4 mg NH<sub>4</sub> N/l.

**Hydraulic loading:** Hydraulic loading rates vary widely depending on the design, the substrate being removed and the effluent concentration desired (Hochheimer and Wheaton, 1998). Some RBC manufacturers developed design relationships for municipal wastewater in which effluent quality is plotted as a function of hydraulic loading, at a given temperature. These relationships are very useful for characterizing full-scale RBC facilities performance. However, since in these relationships the intrinsic biodegradation constants and hydrodynamics of the system are not taken into consideration and equipment manufacturers provide optimistic estimates, care should be exercised in the selection and application of such empirical relationships (Grady *et al.*, 1999).

**Hydraulic retention time:** Studies with RBC systems have revealed that longer contact times improve the diffusion of the substrate into the biofilm and its consequent removal of the influent (Hanhan *et al.*, 2005; Najafpour *et al.*, 2006).

Too short a HRT will result in low removal rates, whereas too long a HRT will not be economically feasible. In order for a biological system to compete successfully with conventional physicochemical methods of treatment, the shortest possible HRT associated with the most efficient removal rates is required (Costley and Wallis, 2000).

**RBC media:** RBC systems have evolved considerably from the original design of several rotating discs. Many variations now exist, ranging from simple flat discs through corrugations to cellular meshes all of which are designed to give extra surface area per unit volume (Fig. 7). However, as the supporting medium gets more complex its cost increases (Ware *et al.*, 1990).

The media used for RBCs are actually produced from Styrofoam, polycarbonate sheets or High-Density Polyethylene (HDPE). HDPE containing UV inhibitors

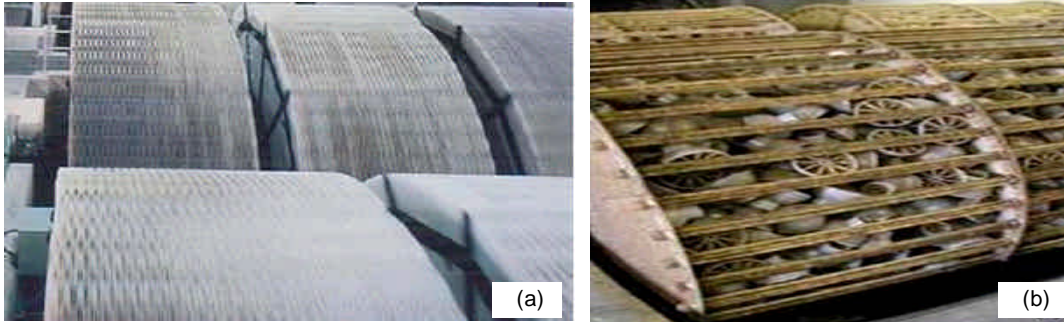


Fig. 7: (a) Closely spaced discs in a full-scale RBC (<http://www.dmw.co.jp>); (b) RBC with a random packed medium (<http://www.wateronline.com>)

such as carbon black is the material most commonly used and is provided in different configurations or corrugation patterns (Ware *et al.*, 1990; Rodgers and Zhan, 2003). Corrugations enhance structural stability, improve mass transfer and increase the available surface area (Grady *et al.*, 1999). The types of biofilm supporting media are classified on the basis of surface area provided and are commonly termed low- (or standard-) density, medium-density and high-density. Standard-density media are defined as having a surface area of about  $115 \text{ m}^2/\text{m}^3$  of reactor, with larger spaces between media layers and are normally used in the lead stages of a RBC process train. Medium and high-density media have surface areas of about  $135\text{-}200 \text{ m}^2/\text{m}^3$  of reactor and are used typically in the middle and final stages of a RBC system where thinner biological growth occurs (Tchobanoglous and Burton, 1991; Patwardhan, 2003). Standard-density media must be used in the first two stages that are highly loaded or where Beggiatoa growth is possible because excess biological growths are more difficult to remove from high-density media (Grady *et al.*, 1999).

**Staging:** Staging of RBC media is recommended to maximize removal of BOD<sub>5</sub> and ammonia nitrogen (NH<sub>4</sub>\_N). Stages are accomplished by using baffles in a tank or using a series of tanks. Typical RBC staging arrangements are illustrated in Fig. 8. In secondary treatment applications, RBCs shall be designed and operated in a series of three stages per flow. For combined BOD<sub>5</sub> and NH<sub>4</sub>\_N removal a minimum of four stages is recommended per flow. For small plants, multiple stages are acceptable on a single shaft oriented in parallel to the direction of flow. In larger installations, shafts are mounted perpendicular to flow with several stages in series (Tchobanoglous and Burton, 1991).

As the wastewater flows through the system, each subsequent stage receives an influent with an organic concentration lower than the previous stage. Because heterotrophic bacteria grow faster than nitrifies the first

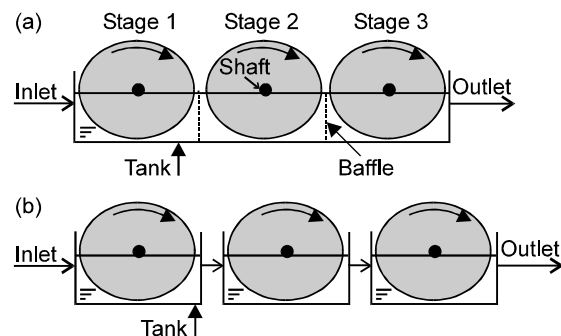


Fig. 8: Staging arrangements of RBC units accomplished by (a) baffles in a tank; (b) using a series of tanks, both with flow perpendicular to shaft (Cortez, 2008)

stage tends to be primarily an organic removal device, unless the wastewater organic content is very low. As the wastewater moves to the second and subsequent stages the RBC tends to first removing ammonia and then nitrite with the final product being nitrate, assuming that the RBC is sized and operated correctly (Hochheimer and Wheaton, 1998). When there is recycling of wastewater from the last stage to the first one, denitrification may be achieved in the first stage, where there is high organic loading and low dissolved oxygen content.

Experimental results of Banerjee (1997a) justify the use of staging in a RBC reactor, since mixing decreases gradually along the reactor, better approximating the system to the plug-flow regime. Radwan and Ramanujam (1997) concluded that staging in the design of RBC systems is especially important at higher organic loadings and also if high effluent treatment quality is required. Moreover, according to Tawfik *et al.* (2002) staging of RBC decreases the detrimental effect of shock load on the performance of the system.

The number of stages to be used depends on the organic content of the influent, flow rate and several other variables (Hochheimer and Wheaton, 1998).

Staging calculations, based on COD and  $\text{NH}_4\text{-N}$  effluent concentrations, can be done using literature tables (Grady *et al.*, 1999), with the appropriate adaptations.

**Dissolved oxygen levels:** In an aerobic RBC system the biofilm is allowed to form on the medium, which is partly submerged in the wastewater and partly exposed to the air. The rotation alternately exposes this biofilm to atmospheric oxygen and wastewater. Oxygen transfers from the air to the RBC unit in three ways: by oxygen absorption at the liquid film over the biofilm surface when the biofilm is in the air; by direct oxygen transfer at the air-water interface and by direct oxygen absorption by the microorganisms during the air exposures (Grady, 1982).

Usually, as a consequence of an active respiration in the initial stages, the oxygen concentration reaches minimal levels, increasing along the reactor where substrate concentration is low.

Israni *et al.* (2002) and Mathure and Patwardhan (2005) evaluated the performance of pilot-scale RBC systems in terms of the Oxygen Transfer Efficiency (OTE). They observed that the OTE per unit energy consumed decreased rapidly with an increase in rotational speed and increased with a decrease in hydraulic loading rate. Mathure and Patwardhan (2005) also compared the oxygen transfer efficiencies of a conventional RBC and a RBC with different packings such as rings, superintalox saddles and a wiremesh spiral bundle. The OTE values for the typical RBC were found to be 1-2 kg/kWh, which were poor in comparison with the values found with packings (2-5 kg/kWh).

Dissolved oxygen is very important in carbonaceous oxidation and nitrification being the most important design factor for aerobic RBCs. During operation oxygen levels must be properly controlled and to prevent from becoming a limiting factor, initial stages should have at least 2 mg DO/l (Nowak, 2000). If the rotating motion does not supply sufficient oxygen, a supplemental aeration system should be installed (Surampalli and Baumann, 1997; Rodgers and Zhan, 2003). Usually this promotes a remarkable performance of the RBC, with an established thinner and active aerobic biomass, allowing considerable cost savings in design and construction of RBC units.

**RBC medium submergence:** The percentage of RBC medium submergence depends on several factors, namely the operation type, microorganisms and characteristics of the effluent to be treated. Typically in aerobic processes of municipal wastewater treatment the submergence is about 40%, although in nutrients removal it can attain 60%. However, due to the diversity of industrial wastewater there is no reference value for disc submergence.

Increased submergence was developed to reduce shaft and bearing loads and to improve equipment reliability (Tchobanoglous and Burton, 1991). Submerged Biological Contactors (SBCs), as are called, operate at 70-90% submergence providing the advantages of larger medium volume available and fewer SBC units required (Cortez, 2008). Submergence in excess of 50% will decrease the rate of oxygen transfer in the system, thereby if the SBC is used to treat wastewater aerobically, additional air drive units to provide oxygen and rotation must be used (Rodgers and Zhan, 2003). The increased submergence combined with the air drive rotation of the SBC has dramatic economic and operating benefits (Cortez, 2008).

**Temperature:** Temperature is one of the most important factors that affect the rate of biological processes and consequently influences RBCs performance. At limited conditions, an increase in the influent temperature leads to an increase in the microbial activity and a higher substrate removal can be observed in all RBC stages (Banerjee, 1997b; Israni *et al.*, 2002). Low influent temperatures can adversely affect biofilm establishment, particularly in its early stages (Costley and Wallis, 2000). When wastewater temperatures less than 13°C are expected, organic and nitrogen removal rates may decrease.

Temperature correction factors need to be taken into account in design criteria and can be obtained from the equipment manufacturers or from pilot studies. Generally, when the temperature drops from 13 to 5°C, nearly 2.5 times more media surface area is required for achieving the same performance (Rodgers and Zhan, 2003).

**Biofilm characteristics:** A biofilm is a living microbial system composed mainly of microorganisms, extracellular polymers and water. The spatial distribution of these components within the biofilm matrix may influence the biofilm functions and the relationship to the immediate aquatic environment. This, in turn, depends on the operating conditions. For example, biofilm thickness depends on applied organic loading and shearing forces (Griffin and Findlay, 2000).

To optimize the removal of organic matter and nitrogen compounds from wastewater in a RBC, an adequate understanding of the dynamic nature and characteristics of the biofilm, the major constituent of the process, is essential.

Observations of full-scale RBCs biofilms treating municipal waste waters report that biofilms from the initial stages have a gelatinous aspect, being usually greyish and may present some white zones probably due to filamentous bacteria like *Beggiatoa*. Biofilms of the last stages appear more compact: are always

thinner than in the first stages and have a brown like colour or sometimes reddish. In addition, the main limiting factor of microfauna growth is the degree of pollution in the influent expressed in terms of COD or BOD<sub>5</sub>. As long as this parameter decreases along the RBC, its effect as a limiting factor decreases too, resulting in an increase in the majority of existing species. Initial stages are almost entirely constituted by species of ciliates, whereas the last stages show more diversified communities, not only in species of ciliates but also in flagellates, amoebae and metazoan (Salvado *et al.*, 2004).

**Submerged fixed film systems:** One of the limiting factors to increasing the efficiency of percolating filtration is the limited surface area available for film development without reducing the size of the interstices so that they become blocked with film growth and so restrict flow and aeration. While the introduction of plastic media improved the situation, the total area of fixed active biomass still remains much lower in percolating filters than in the activated sludge process per unit volume of reactor. In the 1980s the use of very fine media, such as expanded clay nodules or even sand and passing the wastewater upwards through the media at a sufficient velocity to fluidize the media was proposed. In this way, the interstices could not become blocked no matter how much biomass developed. The small media offered specific surface areas of up to 3,500 m<sup>2</sup>/m<sup>3</sup> allowing biomass concentrations equivalent to an MLSS concentration of 30-40,000 mg/l. The use of such fine media providing very high specific surface areas allowed more compact and versatile systems to be designed. From this original concept two further submerged systems emerged. Biological Aerated Flooded Filters (BAFF) and Submerged Aerated Filters (SAF) both use submerged solid medium with oxygen provided by diffusers or other mechanical devices. There is considerable confusion between the two systems and the terms are often used interchangeably. This is because there is such a wide variation between individual systems that the term BAFF or SAF is often not sufficient on its own. However, most authors define BAFF technology as those filters that achieve solids separation by back washing, while SAF technology are filters that require a settlement tank for solids separation. The submerged fixed film systems include five main modes of operation:

1. Biological aerated flooded filters.
2. Submerged aerated filters.
3. particulate biofilm reactors.
4. Moving bed biofilm reactors.
5. Membrane biofilm reactors.

**Biological aerated flooded filters:** Although the correct name for the process is Biological Aerated Flooded

Filters (BAFF), this name has been universally shortened to Biological Aerated Filter (BAF). Experiments in aerating flooded fixed film reactors have been going on since 1913, although the process as we recognize today was first introduced in the late 1980s (Hodkinson *et al.*, 1999). With the introduction of the Urban Wastewater Treatment Directive, there is pressure to up rate existing sewage treatment plants, especially at coastal sites, where land is at a premium, in order to meet new standards. The potential of BAFF technology was immediately recognized because of its compactness, the quietness of operation and the low odour potential of the process. There are many proprietary BAFF systems available, all of which are slightly different in design. However, the principle of the process is common to all (Lilly *et al.*, 1992).

BAFF technology can be used in the same way as the activated sludge process for carbonaceous removal, nitrification, denitrification and biological phosphorus removal (Yoon and Suzuki, 1991; Smith and Hardy, 1992). In fact, whenever activated sludge is appropriate then BAFF should also be considered.

Primary sedimentation or fine screening (1-3 mm) is required prior to secondary treatment by BAFFs. They are constructed above ground in either concrete or steel and must be strong enough to support the media when expanded during back washing. The system is made up of a number of individual cells or compartments. Mineral or artificial granular medium, generally in the range of 1-5 mm in diameter, is used. Granular media with a density less than water is termed floating (e.g. polyurethane, polystyrene) while media with a density greater than water is termed sunken (e.g. sand, pozzolanic stone). The selection of the medium is a compromise between maximum surface area and retention efficiency of the medium during back washing. Like fluidized beds, the granular mineral media in BAFFs are subject to abrasion so that fine particles formed are lost during back washing. This leads to media losses of 2-5% per annum. Some proprietary systems employ fixed lattice type medium, more commonly associated with Submerged Aerated Filters (SAF). The advantage of this modular is that there is lower head loss and no loss of media. However, modular media has significantly lower surface areas, poorer solids removal and is unable to produce high quality (<10 mg/l BOD) final effluents. Multi-layer granular media beds, similar to rapid sand filters are employed where primary sedimentation is omitted (Fig. 9). Screened and degrittled raw wastewater enters the bottom of the filter and rises through graded layers of media. Solid removal is achieved by gravity flushing, up flow air and water scouring and flushing.

The depth of the medium varies from 2-4 m for sunken granular media used in down flow mode (now largely replaced by up flow systems), 3-5 m for up flow granular

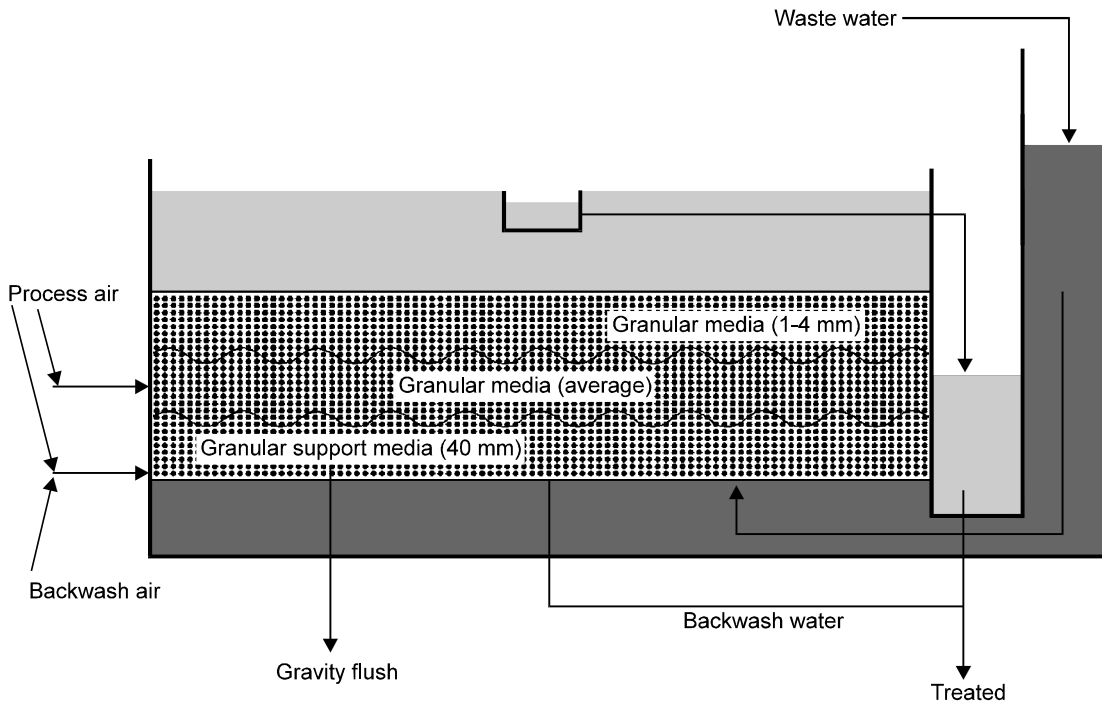


Fig. 9: A typical multi-layer biological aerated flooded filter (BAFF) (IWEM, 2000)

media systems and 4-6 m for fixed lamina media, making all BAFFs tall units that are highly visible. A separate backwash water holding tank is required to store the final effluent used for back washing and another tank for holding the dirty backwash water which is fed back to the inlet is also required. Air blowers are used for both aeration and air scouring during the backwash cycle. The direction of wastewater flow and back washing can be in either direction and varies widely (Fig. 10).

Two sets of air blowers are required: fine bubble diffusers supply processed air at a constant rate to satisfy the oxygen demand of the biomass, while coarse bubble diffusers supply higher rates of air for short periods during back washing. This often requires two sets of pipe work and diffusers and even two sets of blowers. In smaller plants, both fine and coarse bubble aeration can be supplied using perforated latex rubber sheets covered by perforated metal diffuser plates. The size of the bubble and the rate air is supplied is controlled by varying the pressure within the pipe work which causes the pores within the latex to expand or contract, altering the size of the bubble created.

Back washing is very important in BAFF systems as it reduces excess biomass from the filter, prevents head loss due to solids retention and collects the solid for treatment. The frequency of back washing is controlled by the accumulation rate of the film which results in head loss through the cell. Typical backwash cycles vary from 12-24 h for carbonaceous BAFFs to 4-5 d for nitrification

only systems. Back washing takes between 30-60 minutes and has four phases. (i) Influent feed to the cell is shut off. (ii) Water and air is passed upward through the bed of granular medium expanding it 1.3-1.5 times its operational depth. (iii) The medium is agitated by air scouring to detach excess biomass. (iv) Backwash water is passed upward through the medium to remove the detached biomass. Phases (ii) and (iii) are repeated several times to ensure adequate solids removal. The velocity of backwash water is 5-30  $m^3/m^2 \cdot h$  for short periods of 5-20 min, the larger the media the higher the rate. The backwash air flow varies from process to process, but a typical range is 20-60  $m^3$  air  $m^2$  of bed area per backwash. The volume of backwash water used ranges from 5-15% of the average flow, although 10% is typical. It is similar in nature to surplus activated sludge being a mixture of biomass and trapped solids, but with a much lower dry solids content (0.2-0.5%). The backwash water is pumped to the primary sedimentation tank for solids separation, or where there is no primary sedimentation then the backwash water is thickened and the sludge treated as normal. Where medium losses are high, the backwash water should be returned up stream of the grit separation stage of the plant.

Typical organic loadings are between 2-3 kg BOD/ $m^3 \cdot d$ , although this can be increased to 5 kg BOD  $m^3/d$  for high rate treatment producing lower effluent quality (70% BOD). High hydraulic loadings are beneficial because of the improved physical kinetics within the cell, although



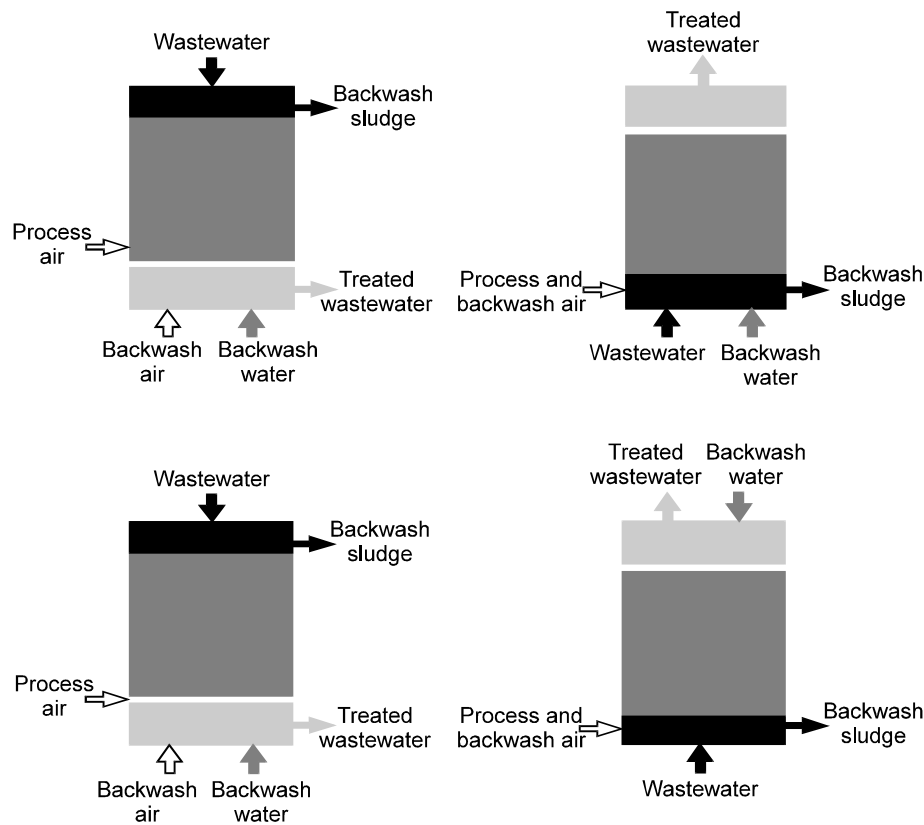


Fig. 10: Directions of flow and backwash in BAFFs (IWEM, 2000)

if the bed becomes fluidized, there will be a significant loss in effluent quality. Assuming one cell out of operation for back washing, then hydraulic loadings of 4-12 m<sup>3</sup>/m<sup>2</sup>.h are normal. Sludge production is between 0.6-1.0 kg suspended solids per kg BOD removed, while the oxygen requirement of BAFFs is similar to that of activated sludge at 1.0-1.5 kg O<sub>2</sub> per kg BOD removed. Design criteria for nitrification are 1.0 kg amm.N/m<sup>3</sup>.d with an additional oxygen requirement of 4.3 kg O<sub>2</sub> kg amm.N removed. The current practice is to nitrify in a two-stage BAFF as conditions are more appropriate for nitrifying bacteria in the second stage. An example of a ten cell steel BAFF unit producing a fully nitrified effluent with an average BOD of 2.7 mg/l and a suspended solids concentration of 3.7 mg/l is described by Robinson *et al.* (1994).

**Submerged aerated filters:** Submerged Aerated Filter (SAF) and Biological Aerated Flooded Filter (BAFF) technologies are similar, i.e. the media is submerged and air is supplied via blowers to maintain the biomass on the medium surface, except that SAF systems do not employ back washing but require a sedimentation tank for solids separation and removal. The medium used is normally modular plastic, 3-6 m in depth, which reduces head loss and prevents blockages (Fig. 11). The surface

area of the modular medium is significantly lower than the granular media used in BAFFs, resulting in lower solids retention which eliminates back washing. Typical specific surface areas of SAF modular media varies from 250-350 m<sup>2</sup>/m<sup>3</sup>, with the higher surface areas are being used for nitrification. Air is supplied by diffusers and although only process air is required some designs incorporate the provision of air scouring at similar pressures as used during back washing of BAFFs. This provides greater control of sludge removal as otherwise the biomass sloughs off in large clumps causing problems during final settlement.

Submerged aerated filters are packaged units designed for typical organic loadings of 3 kg BOD/m<sup>3</sup>.d and hydraulic loadings of 2-8 m<sup>3</sup>/m<sup>2</sup>.h. Oxygen requirements and sludge production rates are similar to those for BAFF. If a 95 percentile effluent quality <25 mg/l BOD and 35 mg/l suspended solids is required, then tertiary treatment must be used.

Simple to operate and robust, SAFs are ideal packaged units for small treatment plants (<5,000 PE). They require little maintenance making them suitable for unmanned and isolated works. SAF units are used to up rate existing works, for example by reducing the loads to existing percolating filters in order to promote nitrification. The period for set up is much shorter than

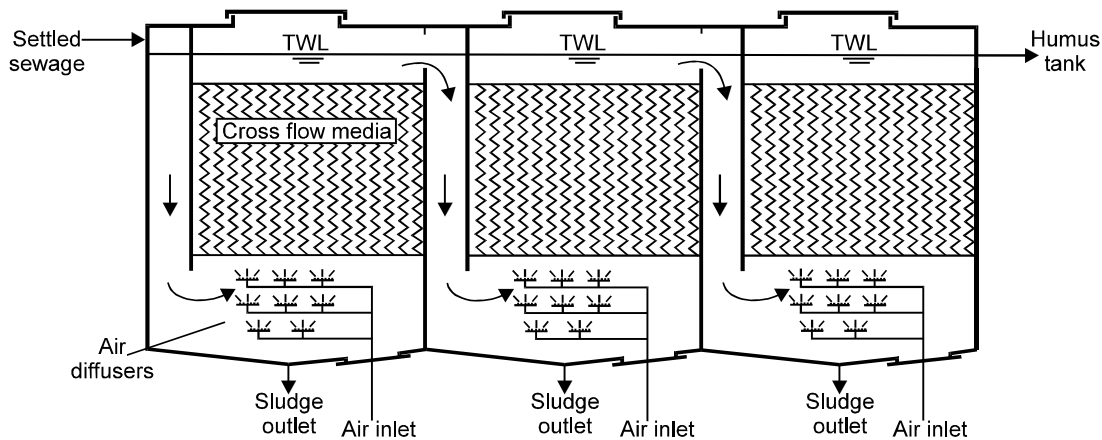


Fig. 11: Schematic diagram of submerged aerated filters in series (IWEM, 2000)

for activated sludge, being only 3-8 d for BOD removal, making them very flexible in terms of expanding treatment capability. Schlegel and Teichgraber (2000) describe the successful application of SAFs for the pre-treatment of industrial waste waters including waste waters from food processing, carpet dyeing, pharmaceutical and tar processing factories. An up flow SAF system was used by Timur (2001) for denitrification with maximum denitrification rates achieved at COD:NO<sub>3</sub>-N ratios of 5-6, independent of hydraulic loading rate or influent NO<sub>3</sub>-N concentrations. Denitrification followed half-order kinetics, with removal efficiencies of 71-99% reported. An Up-flow Anaerobic/Aerobic Fixed Bed (UA/AFB) combined reactor was used by Moosavi *et al.* (2005) for simultaneous organics and nutrients removal by using a synthetic wastewater was prepared in concentrations which were close to those found in municipal waste waters. The reactor was operated at 5 different HRTs ranging from 5 to 24 h, the obtained results showed that the HRT of 7 h was suitable for simultaneous removal of COD, nitrification and denitrification. In this HRT efficiencies are 95.4, 94 and 94.5% for COD removal, nitrification and denitrification, respectively. The reactor did not show good performance in phosphorus removal.

**Particulate biofilm reactors:** The main reactor types applicable for the suspension of particulate biofilms in wastewater treatment processes are Biofilm Up-flow Sludge Blanket (USB), Fluidized Bed (BFB), Expanded Granular Sludge Blanket (EGSB), Biofilm Airlift Suspension (BAS) and Internal Circulation (IC) reactors (Fig. 12). In USB, BFB and EGSB reactors (Fig. 12a, b, c, respectively), particles are kept fluidized by the up-flowing influent. In BAS reactors (Fig. 12d) an airlift suspension is obtained by pumping air into the system, whilst in IC reactors (Fig. 12e) the gas produced in the system drives the circulation and mixing of liquid and solids in the reactor. The advantages and

disadvantages of particulate biofilm reactors are presented in Table 3 (Heijnen *et al.*, 1989; Bryers and Characklis, 1990; Harremoës and Henze, 1995; Wright and Raper, 1996).

Although the development of water and wastewater biological reaction systems using a fluidized bed of biomass can be traced back to 1940 in the UK (Mishra and Sutton, 1991), development of particle-supported biofilm reactors did not occur until the early 1970s. Researchers at Manhattan College in New York were granted a US patent in 1976 (Jeris *et al.*, 1976) for the application of Biofilm Fluidized Bed (BFB) process configuration to denitrifying wastewater. The first commercial-scale application for the BFB process configuration occurred at Pensacola (USA) in the mid-1970s, with the installation of a system for the denitrification of nitrified municipal wastewater (Mishra and Sutton, 1991). The first commercial-scale application of the BFB technology to industrial treatment occurred in the late 1970s when an Ecolotrol HY-FLO system was installed at a soft drink bottling plant in Birmingham (USA) (Jeris, 1983). Since the early 1980s, the BFB process has been used on an industrial scale for most conventional-sewage treatment processes including carbonaceous oxidation (Jeris *et al.*, 1977; Cooper and Wheeldon, 1982), nitrification (Dunn *et al.*, 1983; Nutt *et al.*, 1984), denitrification (Melcer *et al.*, 1984; Nutt *et al.*, 1984) and anaerobic treatment (Hickey and Owens, 1981; Jewell *et al.*, 1981; Jeris, 1983).

Perhaps the most significant recent development at the commercial-scale level is the use of Up flow fluidized bed reactors containing granules and no bed media. The USB reactor was developed in The Netherlands in the late 1970s for anaerobic treatment of low-strength wastes and the first full-scale plant (200 m<sup>3</sup>) was installed in 1978 at a sugar-beet factory in Halfweg (The Netherlands) (Lettinga *et al.*, 1980). The EGSB and the IC reactors are the most recent evolutions of the USB concept. Biothane Biobed EGSB (Lourens and

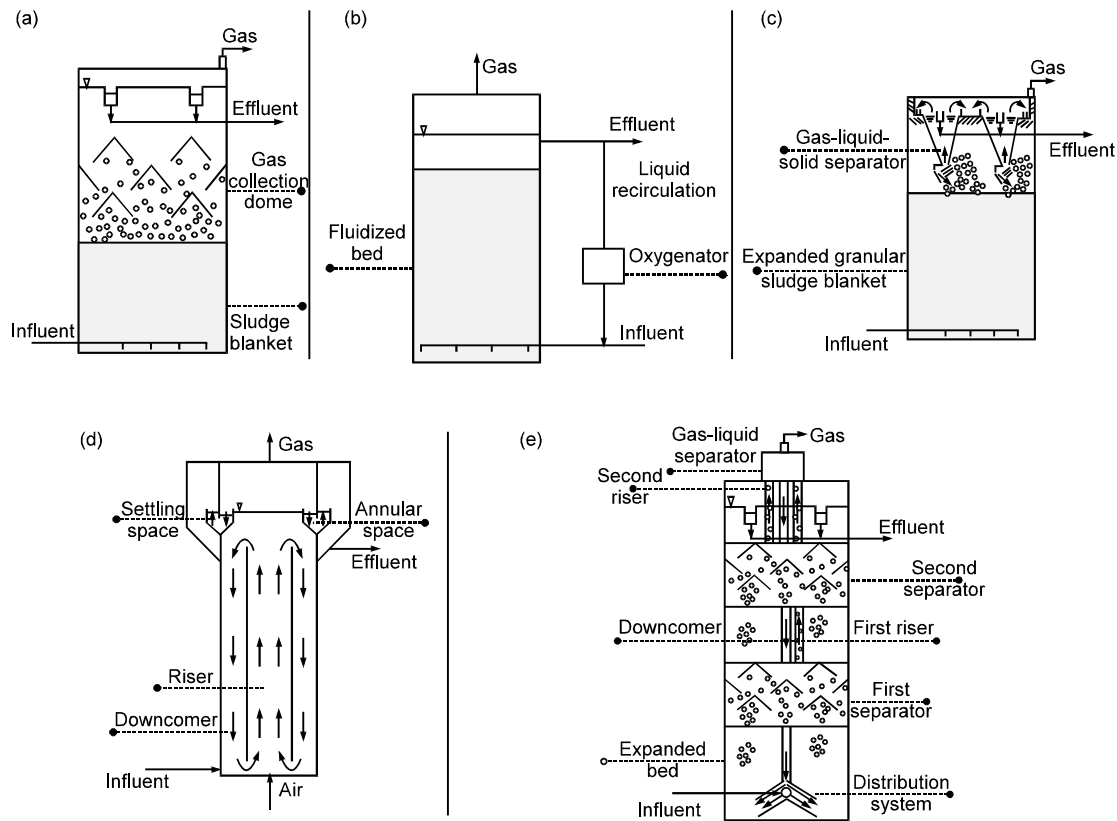


Fig. 12: Biofilm reactor configurations. (a) USB; (b) BFB; (c) EGSB; (d) BAS; (e) IC (Heijnen *et al.*, 1989; Bryers and Characklis, 1990)

Table 3: Advantages and disadvantages of particulate biofilm reactors (Heijnen *et al.*, 1989; Bryers and Characklis, 1990; Harremoens and Henze, 1995; Wright and Raper, 1996)

Advantages	Disadvantages
High terminal settling velocity of solids (50 m/h; for flocculated sludge: 5 m/h), leading to possible elimination of external clarification/separation stages.	Biofilm formation on carriers poses problems leading to long start-up times.
High reactor concentration (30 kg m <sup>-3</sup> ; for flocculated sludge systems: 3 kg m <sup>-3</sup> ).	Control of biofilm thickness is difficult.
High biofilm surface area (3000 m <sup>2</sup> m <sup>-3</sup> ; in trickling filters 300 m <sup>2</sup> m <sup>-3</sup> ).	Overgrowth of biofilms leads to elutriation of particles.
High biomass concentration and mass transfer area result in high conversion capacities (for oxygen, 20 kg m <sup>-3</sup> day <sup>-1</sup> ; in activated sludge and trickling filter processes: 3 kg m <sup>-3</sup> day <sup>-1</sup> ).	Liquid distributors for fluidized systems are costly for large-scale reactors and pose problems with respect to clogging and uniform fluidization.
Compact reactor with small area requirements High biomass age (several weeks) and minimization of excess sludge production.	

Zoetemeyer, 1992) and Paques IC (Vellinga, 1986) systems operate at much higher liquid up flow velocities than the conventional USB.

Fundamental and applied research on Biofilm Airlift Suspension (BAS) reactors in the late 1980s (Heijnen, 1985; Heijnen *et al.*, 1991; 1993), including research at Gist-Brocades, TNO (Dutch Organisation for Applied Scientific Research) and Delft University of Technology (The Netherlands), led to the concept of CIRCOX airlift reactor, developed and patented by Gist-Brocades and commercialized by Paques.

High rate anaerobic pre-treatment followed by aerobic post-treatment using biofilm reactors (Heijnen *et al.*, 1991) has been shown on numerous occasions to be a very cost effective way to remove organic compounds from waste waters (Fig. 13). The IC anaerobic reactor followed by the CIRCOX airlift aerobic technology has been recently selected to treat the effluent from a large brewery in The Netherlands (Gorur *et al.*, 1995). After a year of operation, this innovative combination was reported to have overall total and soluble COD removals of 80 and 93.5%, respectively. Current volumetric loading



Fig. 13: Full scale Paques CIRCOX (front, 140 m<sup>3</sup>) and IC (back, 385 m<sup>3</sup>) reactors at a brewery in Brazil (Heijnen *et al.*, 1991)

rates have averaged 14 kg/m<sup>3</sup>. day in the IC and 10 kg/m<sup>3</sup>. day in the airlift system.

**Up flow Sludge Blanket (USB) reactor:** The principle of operation of the Up flow Anaerobic Sludge Blanket (USB) reactor (Fig. 12a) (Lettinga *et al.*, 1980) is similar to that of biofilm fluidized bed reactors, but hydrodynamic conditions are more quiescent due to lower superficial liquid velocities. USB processes are based on the development of dense granules (1-4 mm) in the reactor. Wastewater enters the bottom of the reactor vessel through the inlet distribution system and passes upward through the dense anaerobic sludge bed (biomass concentration: 60-70 kg/m<sup>3</sup>). Soluble COD is readily converted to biogas, which is rich in methane and an upward circulation of water and gas borne sludge is established. A settler section allows effective degasification to occur. The dense, granular sludge particles now devoid of attached gas bubbles, sink back to the bottom, establishing a return downward circulation. The upward flow of gas borne sludge through the blanket combines with the return downward flow of degassed sludge and creates continuous convection. This insures effective sludge to wastewater contact. The design of the reactor allows a highly active biomass concentration and thereby high loading rate (10-15 kg COD/m<sup>3</sup>.day), i.e. short hydraulic retention time (less than 48 h for most applications).

Suspended solids in the influent, which accumulate in the reactors, pose a major problem to the operation of USB and reduce the reactor capacity. To overcome these limitations, the BFB and later, the EGSB and the IC concepts were developed.

**Biofilm Fluidized Bed (BFB) reactor:** When designing a recirculating system it is essential to consider the most cost effective filtration system for the desired water quality. For "oligotrophic" (low nutrient levels) systems, where extremely high water quality must be attained, biofilters with low cost per cubic foot of media and high surface area per unit volume must be considered. This is where fluidized bed filters really stand apart from other filters.

In Biofilm Fluidized Bed (BFB) reactors (Fig. 12b), the liquid to be treated is pumped through a bed of small media (typically sand with a particle size range of 0.2-0.8 mm) at a sufficient velocity to cause fluidization. In the fluidized state the media provide a large specific surface for attached biological growth and allows biomass concentrations in the range 10-40 kg/m<sup>3</sup> to develop (Cooper and Sutton, 1983). For aerobic treatment processes the reactor is aerated. This is done by recirculating the liquid from the reactor to an oxygenator where air, or possibly oxygen, is bubbled (Cooper, 1981). To overcome problems related to high recirculation rates, needed when there is a high oxygen demand in the reactor, the reactor might be aerated directly (three-phase BFB reactor) (Fan *et al.*, 1986; Tang and Fan, 1987; Trinet *et al.*, 1991).

The BFB technology is typically most useful for treatment of streams contaminated with organic or inorganic compounds (e.g. ammonia) requiring long solids residence time conditions (longer than 15 days) for biological oxidation or reduction and low (less than 100 mg/l) concentrations of suspended solids (Sutton and Mishra, 1990).

Biothane B.V. has built several anaerobic two stage BFB plants for purification of their fermentation process wastewater (Heijnen *et al.*, 1989) treating wastes at COD volumetric loading rates in excess of 20 kg/m<sup>3</sup>.day. The first two plants were built in the mid-1980s at Delft in The Netherlands and Prouvy in France.

Degremont S.A. developed the ANAFLUX process (Durot and Prevot, 1987; Bernard and Rovel, 1989; Oliva *et al.*, 1990), an up flow anaerobic reactor using a mineral bacterial support, in the mid-1980s. The first industrial reactor was built in 1986 for the treatment of brewery wastewater. Since then, Degremont S.A. has constructed over 25 full-scale ANAFLUX reactors treating, world-wide, a variety of industrial waste waters (Holst *et al.*, 1997). The system is highly efficient due to the large reactor biomass concentration (30-90 kg/m<sup>3</sup>) and to the intimate contact between biomass and substrate created by high up flow liquid velocities (5-10 m/h).

**Expanded Granular Sludge Bed (EGSB) reactor:** The Expanded Granular Sludge Bed (EGSB) reactor (Fig. 12c) (Frankin *et al.*, 1992; Zoutberg and Frankin, 1996) combines both characteristics of USB and BFB processes. Biomass is present in a granular form, but conditions with respect to the up-flow velocities for liquid (10 m/h) and gas (7 m/h) approach those of the BFB system. EGSB reactors can be operated as ultra high loaded anaerobic reactors (up to 30 kg COD/m<sup>3</sup>.day) to treat effluents from chemical, biochemical and biotechnological industries (Zoutberg and de Been, 1997). Biothane B.V. have built dozens of Biobed EGSB units for various types of wastewater (food, chemical, pharmaceutical industry) in all parts of the world.

**Biofilm Airlift Suspension (BAS) reactor:** Airlift reactors (Chisti, 1989) consist of two connected sections, a riser and a down comer. Different configurations are possible, including internal loop and external loop reactors. The principle of operation is the same for both configurations.

A gas is spared at the bottom, moves upward and exits at the top of the riser section. In internal-loop airlift reactors, air may recirculate through the down comer section and provide aeration throughout the reactor. The difference in density between riser and down comer, due to the difference in gas hold-up, drives the liquid to circulate between the two sections. When the liquid velocity is sufficiently high, small particles will be suspended and recirculated with the liquid. This results in a thorough mixing of both particles and liquid throughout the reactor. The airlift technique has found two major applications in wastewater treatment processes, the Biofilm Airlift Suspension (BAS) reactor (Fig. 12d) for aerobic treatment and the gas-lift reactor for anaerobic treatment. The BAS technology was

originally developed for aerobic purification of anaerobically treated industrial waste waters (Heijnen, 1984; Heijnen *et al.*, 1990, 1993). In anaerobic processes, the circulation in the reactor is induced by recirculating the gas produced by the biochemical reactions (gas-lift reactors; van Houten *et al.*, 1994; 1997).

CIRCOX, a commercial embodiment of the BAS reactor, is an aerobic system which ensures high biological load (4-10 kg/m<sup>3</sup>), short liquid residence time (0.5-4 h), high biomass settling velocities (50 m/h) and high biomass concentration (15-30 kg/m<sup>3</sup>). The CIRCOX airlift technology has been applied to treat municipal wastewater at a sewage plant in The Netherlands (Frijters *et al.*, 1997). Two CIRCOX reactors, an airlift reactor and an airlift reactor with integrated denitrification, were used to treat pre-settled influent. High BOD and nitrogen removal rates could be obtained resulting in a good effluent quality. Approximately 15 such systems are presently in operation.

TURBOFLO is a biofilm reactor for secondary and tertiary wastewater treatment developed by Cie Lyonnaise des Eaux (France) using the concept of an internal circulating airlift reactor (Lazarova and Manem, 1996; Lazarova *et al.*, 1997; Mousseau *et al.*, 1998). The reactor comprises a rectangular column filled with high density polyethylene granules (size: 0.5-2.5 mm; density 860 kg/m<sup>3</sup>). An industrial-scale reactor was operated at a wastewater treatment plant at Evry (France).

An industrial prototype of an external loop airlift reactor, the BIOLIFT process, was developed by OTV S.A. (France) and installed at the Maxeville (France) wastewater treatment plant (Badot *et al.*, 1994).

**Internal Circulation (IC) reactor:** The IC reactor (Fig. 12e) (Vellinga, 1986) consists of two USB-like reactor compartments on top of each other, one highly loaded and one low loaded. The first reactor compartment contains an expanded granular sludge bed, where most of the COD is converted to biogas. The biogas produced in this compartment contains an expanded bed, is collected by the lower level phase separator and is used to generate a gas lift by which water and sludge are carried upward via the riser pipe to the gas/liquid separator on the top of the reactor. Here the biogas is separated from the water: sludge mixture and leaves the system. The water: sludge mixture is directed downwards to the bottom of the reactor via the concentric downer pipe, resulting in the internal circulation flow. In the lower section, the upward liquid velocity varies from 10 to 20 m/h. The effluent from the first compartment is post-treated in the second, low loaded compartment, where remaining biodegradable COD is removed. The liquid velocity in this compartment normally varies from 2 to 10 m/h. The IC system consists of a slender vertical reactor with a height of up to 25 m and a relative small surface area.

Paques has several full-scale IC systems in operation in Europe. One of the largest systems is located at a brewery in The Netherlands (Pereboom and Vereijken, 1994) and consists of six IC reactors each 162 m<sup>3</sup> in volume. The reactors are designed at a COD loading rate of 24 kg/m<sup>3</sup>.day. The largest reactors (22 m high, 9.5 m in diameter and approximately 1500 m<sup>3</sup> in volume) are presently at ADM in Ireland and at Zhongya Chemicals in China.

**Moving Bed Biofilm Reactor (MBBR):** The MBBR was developed in Norway at the Norwegian University of Science and Technology in cooperation with a Norwegian company Kaldnes Miljøteknologi (now AnoxKaldnes AS). The first MBBR was installed in 1989. In United States the first MBBR was introduced in 1995, now there are over 400 installations worldwide in both the municipal and industrial sectors with over 36 in North America. The MBBR is a complete mix, continuous flow through process which combines the benefits of fixed film and suspended growth processes. Benefits include increased treatment capacity, improved settling characteristics, enhanced process stability and reduced sludge production (Metcalf and Eddy, 2003). The process can either be used as:

1. Pre-treatment system ahead of an existing activated sludge system for increased organic matter removal.
2. Stand alone biological treatment process for BOD removal, nitrification and/or denitrification.
3. A retrofit of an existing activated sludge processes to help increase overall nitrification capacity of the existing system. MBBRs incorporate benefits provided by both fixed film and activated sludge processes.

The MBBR process is a continuous flow process which uses small High Density Polyethylene (HDPE) carrier elements 10-20 mm in diameter to provide sites for active bacteria attachment in a suspended growth medium. The density of the medium is critical as it allows it to move freely within the reactor even at filling fractions (i.e. the volume occupied by the medium in an empty reactor) of 70%. The carrier elements allow a higher biomass concentration to be maintained in the reactor compared to a suspended growth process, such as activated sludge. This increases the biological treatment capacity for a given reactor volume. The carrier elements can be installed in either an anoxic reactor or aeration basin. A screen or sieve assembly with 5 mm slot openings is used to retain the carrier elements in the reactor. The effective open area of the screen is sized to provide less than 2 inches of head loss. However, the process does not require back washing of the retention screens which retain the carriers. Figure 14

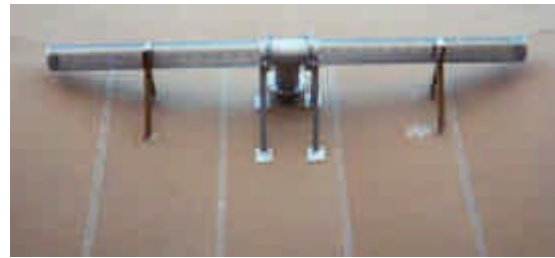


Fig. 14: Effluent sieve assembly for retaining carrier elements (Zimmerman *et al.*, 2005)

shows a photograph of a sieve assembly on effluent pipe for an MBBR.

The carrier elements are continuously kept in suspension by either a mixer or an aeration system. The agitation pattern in the reactor is designed to provide an upward movement of the carriers across the surface of the retention screen which creates a scrubbing effect to prevent clogging, so that the whole reactor volume is biologically active resulting in higher biomass activity. This allows a higher concentration of active biomass to be maintained in the reactor for biological treatment without increasing the reactor size. The result is more treatment capacity within a given reactor volume, resulting in a smaller footprint compared to a conventional activated sludge process. In addition, unlike the conventional activated sludge process MBBRs do not require return activated sludge pumping.

Coarse bubble and jet aeration are typically used to provide oxygen for an aerobic reactor. Reports using fine bubble aeration in an MBBR system indicate the media cause the bubbles to coalesce, thereby reducing oxygen transfer efficiency (Zimmerman *et al.*, 2005). Jet aeration is recommended for new installations due to its intense mixing and improved oxygen transfer efficiency when compared to coarse bubble aeration. Ideal depths for the jet aeration system range between 16 to 24 feet.

There are several different sizes and designs of carrier elements used in the MBBR process. The main four carriers shape are shown in Fig. 15. The KMT carrier K1 is the original kindles carrier that is mostly used. All carriers were made of high-density polyethylene (density 0.95 g/cm<sup>3</sup>) in order to avoid influence buoyancy differences. The surface areas given in Table 4 are estimations to the best of our ability. The total surface area consist of both inner and outer surfaces, while the effective surface area is that where biofilm seems to attach. The effective surface area of the KMT K1 and the AWT carriers were calculated as the whole inner area plus the area of the outer fins. The area between the fins was not included since visual inspection did not show any sign of growth here. For the ANOX carrier, the effective area is calculated as the inner area since there are no fins with outer area.

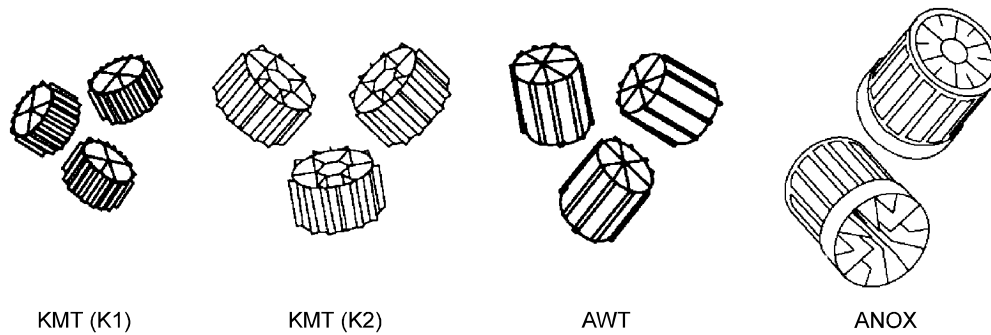


Fig. 15: The main four biofilm carriers (Odegaard *et al.*, 1994)

Table 4: Characteristic data for the four different carriers (Odegaard *et al.*, 1994)

Specific surface area	KMT carrier K1	KMT carrier K2	AWT carrier	ANOX carrier
Estimated surface area (mm <sup>2</sup> /piece)	Total : 670	Total : 3465	Total : 2200	Total : 10000
	Effective : 490	Effective : 1910	Effective : 1500	Effective : 7700
Bulk carriers (number/liter)	1030	159	203	24
Specific surface area (m <sup>2</sup> /m <sup>3</sup> )	Total : 690	Total : 550	Total : 450	Total : 240
	Effective : 500	Effective : 315	Effective : 310	Effective : 190

**Membrane biofilm reactors:** Many microbial processes in environmental engineering can use dissolved gases as substrates for microbial growth. The most obvious example is dissolved oxygen, which serves as an electron acceptor for aerobic degradation. Other examples include hydrogen, an electron donor for autotrophic denitrification and methane, which supports cometabolic Trichloroethene (TCE) oxidation. Low aqueous solubility limits the use of many gaseous substrates. For example, the BOD of typical waste waters (100-400 mg/L) greatly exceeds oxygen's solubility of 8 mg/L (for air at 1 atmosphere). In activated sludge, the solubility problem is resolved by continuously bubbling air (sparging) to replenish oxygen. However, sparging is problematic when applied to other gases, such as hydrogen or methane: it is wasteful, due to loss of excess gas to the atmosphere and can cause safety hazards. In addition, sparging requires large amounts of energy and can vent Volatile Organic Compounds (VOCs) to the atmosphere. The Membrane Biofilm Reactor (MbFR) a novel system that provides dissolved gas directly to a biofilm growing on the membrane surface, avoiding the need for sparging.

MBFRs are not Membrane Bioreactors (MBRs). An MBR is a biological treatment process where a membrane is used to separate biomass from the effluent water, substituting for a clarifier. Because MBRs act as filters, they are susceptible to fouling by biofilms or other materials that accumulate at the membrane surface. In contrast, in MBFRs a gaseous substrate moves across the membrane, while the naturally-forming biofilm on the outer surface catalyzes desired reactions. Since the pores of the membrane are hydrophobic, water and bacteria do not penetrate and block them. The

combination of a membrane for gas delivery as well as for biofilm support led to the name Membrane- Biofilm Reactor.

One of the key elements of an MBFR is the membrane. Membranes may be made from organic or inorganic materials and can be configured in sheet or hollow-fiber geometries. Hollow-fiber membranes are commonly used for MBFRs because, with outside diameters as small as 0.1 mm, they provide high surface-to-volume ratios. Hydrophobic materials are preferred because their pores remain dry and gas molecules diffuse much more quickly through dry pores than through liquid-filled pores (Yang and Cussler, 1986). Dry pores also eliminate the potential for fouling.

A key feature of hydrophobic hollow-fiber membranes is that they can be operated at high gas pressures without bubbling. Higher gas pressures improve mass transfer by providing a greater driving force for dissolution. When membrane pores are fairly large, such as with silicon membranes, bubbles begin to form when the gas pressure slightly exceeds the hydrostatic pressure of the liquid (Ahmed and Semmens, 1992). In contrast, when the pores are small, the water surface tension on the pores can provide a significant resistance to the formation of bubbles, allowing much higher applied pressures.

Figure 16 shows a schematic bundle of hollow fiber membranes and a section of a single hollow fiber. As shown on the right side of the figure, the fibers are collected into a gas-supplying manifold at one end and are sealed at the opposite end. On the left side of the figure, pressurized gas in the lumen (interior) of the fiber diffuses through the dry pores and into the biofilm coating the fiber. 100% of the gas supplied to the MBFR passes into the biofilm.

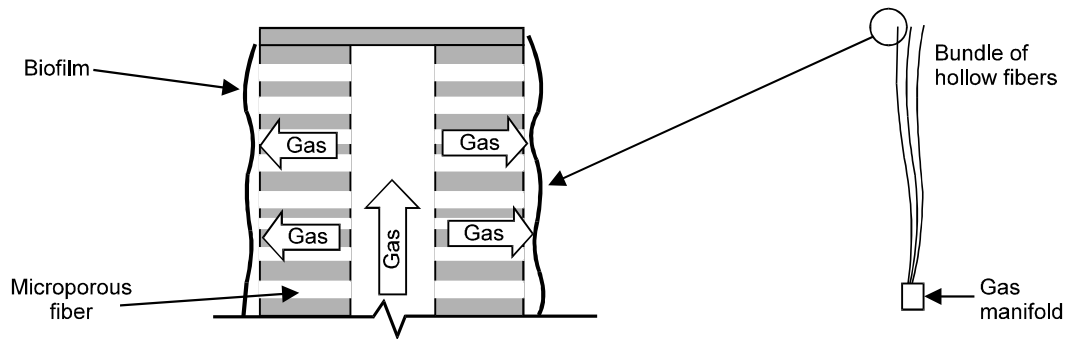


Fig. 16: Section of fiber (left) and schematic of hollow fiber membrane bundle (right) (Essila *et al.*, 2000)

A major advantage of the MBfR is that virtually all of the gas passing through the membrane can be utilized within the biofilm. This is due to the counter-current transport of dissolved gas and substrate from the bulk liquid, as discussed above. Nearly 100% use of the gas means no unproductive loss of gas in the effluent. It also avoids potential hazards with explosive or toxic gases. A second advantage is that the gas moves across the membrane wall when the bacterially catalyzed reaction in the biofilm creates a concentration gradient. If the biochemical demand for dissolved gas declines, the gradient of and demand for gas also decline. If the demand increases, the gradient and demand also increase. Thus, to a certain degree the MBfR operates as a self-regulating, on-demand system that modulates its gas supply rate to the contaminant load. This prevents wasting gas or having an under-supply. A third advantage is that hollow-fiber membranes provide a large specific surface area for biofilm accumulation. A high specific surface area allows a high density of contaminant-reducing bacteria in the MBfR. This means that the detention time for the reactor can be small, thereby minimizing capital costs and the system's footprint. It is ideal for treatment-plant retrofits, as well as for new construction.

**Conclusion:** Fixed-film biological systems have been used widely in the treatment of wastewater, particularly in the attainment of secondary effluent standards and nitrification. Biofilm systems continue to draw significant research attention. While this review is not an all-encompassing documentation for the biofilm processes, it does provide an opportunity to reflect on what biofilm and hybrid biofilm systems may still have to offer the wastewater and environmental research and engineering community. Future research will likely extend the focus of biofilm systems application in the areas of nutrient control, metals and other contaminants and microbial fuel cell technology.

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