SUSTAINABLE TREATMENT AND REUSE OF WATER USING DECENTRALIZED SYSTEMS

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Abstract: Currently natural freshwater resources around the globe are threatened by population growth, worsening food crisis, which battles bioenergy for land and water resources, and climate change, that is disturbing the overall water balance. Skyrocketing energy prices have forced public and private response, and the same is beginning to happen with water. Public utilities are discussing how to restructure water rates to better reflect true costs without causing public harm. There is an impending water crisis, which will threaten crops and cause population movements as water refugees away from places of water scarcity and famine. One of the solutions to this impending crisis is on-site treatment of wastewater with the goal of reuse as irrigation water to promote agriculture and prevent famine. This on-site treatment is possible with decentralized treatment systems instead of large collection systems and centralized wastewater treatment plants. Sustainable, decentralized treatment requires treatment using minimal energy consumption with high treatment efficiencies. In this paper, a low energy consuming method for aeration using micro-nanobubbles will be presented, which when used with a high surface area biomedia, allows efficient treatment in a compact design. The use of self-cleaning filters and membranes allows the treatment system to operate unattended and deliver clear treated water with non-detect suspended solids, BOD and COD less than 10 mg/L with total nitrogen less than 5 mg/L.

Keywords: Sustainable, Decentralized, Wastewater, Treatment, Micro-Nanobubbles, Biomedia

1. Introduction

Decentralized treatment is treatment of wastewater at or near the source(s) with the goal of reusing the treated water, either for local agriculture or other uses at or near the source(s)[1,2]. The goal is reducing net water consumption and allowing treated water to replenish the groundwater table. The conventional approach has been to combine wastewater from several sources (hundreds and thousands of houses, industries, etc.), transport the combined, large flow through a network of underground sewer pipes and then treat the water in a large, centralized treatment plant, and then discharge the treated water into a river, that eventually takes the water into the ocean. This allows groundwater to end up as saltwater resulting in the global decline of groundwater levels. Even with the emergence of large-scale weather events, such as hurricanes, due to climate change, most of the storm water runoff ends up in a river, which takes it back into the ocean.

The traditional decentralized treatment system is a septic tank, which does not treat the wastewater, but merely separates the solids and allows the wastewater to be treated by soil bacteria when it is applied to a drain field. With 40% of U.S. population residing in rural areas, sewage treatment relies mainly on the septic tank, and the effluent is discharged locally in a soil field. The main issue is the clogging of the drain field over time either due to the formation of biomat, and/or in recent years due to the gradual deposition of microplastics, present in sewage water. Due to the global accumulation of plastics, there is the increasing presence of microplastics (plastic particles typically less than 5 microns in size) in our food supply, which are not filtered out by traditional septic tanks. When the water from the septic tank is applied to a soil drain field, it results in the accumulation of these microplastics on the soil surface, gradually decreasing water permeation rate into the ground, and eventually clogging up the soil drain field in a few years. In 2014 a national survey in the U.S. had reported that there were more than 2 million failed soil drain fields. Untreated wastewater from these failed fields have been contaminating drinking water wells for decades, which is jeopardizing the health of millions of people, that rely on well water for their daily needs.

We need to emulate nature which has a water balance cycle, from evaporation and natural transpiration from trees and vegetation to clouds, and then into rain, which regenerates the water in the soil resulting in more biomass growth. Wastewater treatment also needs to follow a circular economy from extraction, converting into wastewater after usage, followed by treatment and reuse. Decentralized treatment at the source is a necessary technology to emulate the natural water cycle, thereby building a circular water economy.

Figure 1. Centralized Treatment and Monitoring versus the future of Wastewater Treatment, which is Decentralized Treatment and Centralized Monitoring.

2. Background

In several European countries, such as Germany, The Netherlands, decentralized treatment systems for 1,000 people have been deployed that incorporate reuse of water, energy and resources [3,4]. The potential of centralized and decentralized wastewater management was studied in Bangkok [5] and it showed that decentralized approach was economical and conducive for sustainable urban growth. It also demonstrated that decentralized treatment was cost-competitive, since it had short sewer lines and simpler technology, which was easy to maintain. Treated water was used locally, mainly for irrigation (30- 100%), while only 5% of treated water from centralized treatment was recycled and reused.

Currently both centralized treatments using large-scale wastewater treatment plants and decentralized systems, such as a septic tank, coexist, with decentralized approach being used mainly for rural areas, and centralized treatment for high population density areas. However, in recent years we have realized that using a network of underground sewer pipes and a large-scale centralized treatment plant was one of the biggest blunders of mankind. Centralized treatment has resulted in a large, expensive to maintain, sewer pipe network, and created a non-sustainable issue of declining groundwater level, since treated water is generally discharged into a river, which eventually converts into salt water in the ocean.

In the United States, about 60 million people use some form of onsite wastewater treatment systems of which about 20 million use the conventional septic tank system [6,7]. In Australia, about 12% of the population uses septic tank systems [7], while in Turkey, about 28% of municipalities are served by septic systems [8].

Traditional decentralized treatment system is a septic tank. Septic tank is a primitive technology which simply separates the solids, allows the wastewater to biodegrade anaerobically in the tank to some extent, and then depends on the soil bacteria in the drain field to treat the wastewater, before it reaches the groundwater table. It is incapable of treating nutrients (nitrogen and phosphorus) adequately, which has resulted in contaminating nearby water bodies, such as lakes, rivers, resulting in toxic algae blooms. With increasing ambient temperatures, due to climate change, the occurrence of toxic algae blooms has increased worldwide, resulting in beach closures and threatening drinking water supplies.

In recent years, on-site aerobic systems have also been used as decentralized treatment, for single family houses and sub-divisions. While these systems can treat the organic load in the influent wastewater, they usually don't effectively reduce the nitrogen and phosphorus, resulting in the discharge of nutrients, that cause algal blooms in nearby water bodies. They are also unable to filter out microplastics, present in domestic wastewater, which eventually clog up the soil drain field

3. Biological Wastewater Treatment Steps

Biological wastewater treatment processes in the following sequential steps [9, 10, 11]:

1. Preliminary Treatment: sedimentation, screening, filtration of large solids, flotation and skimming, and flow equalization.

2. Primary Treatment: chlorination, ozonation, neutralization, coagulation, settling, adsorption, ion exchange.

3. Biological Treatment: This can be further classified as aerobic and/or anaerobic. Aerobic biological treatment includes activated sludge, trickling filters, oxidation ponds, lagoons, and aerobic digestion. Anerobic biological treatment includes anaerobic digesters, septic tanks, lagoons.

4. Tertiary Treatment: This includes disinfection with chemical agents, settling, coagulation and clarification with chemicals, filtration, softening, activated carbon treatment, ion exchange, membrane separation.

4. Decentralized versus Centralized Wastewater Treatment

Centralized treatment using an elaborate network of underground sewer pipes (collection system) followed by a large, central, wastewater treatment plant was the biggest blunder of mankind. It has resulted in the decline of ground water levels around the globe, including many countries, which suffer from water scarcity. It poses the greatest impediment to water reuse, since piping the treated water back to the users would be uneconomical. In addition, maintaining the vast network of sewer pipes is very expensive and has resulted in raw wastewater leaking into the ground and contaminating drinking water wells, with adverse health implications for millions of people.

Decentralized treatment which treats the wastewater at or near the source allows water reuse, at least as non-potable water, thereby preventing the decline of ground water levels, reduces the net consumption of fresh water, and avoids the high installation and maintenance costs of a large collection system. With the growth of wireless connectivity and sensors, these decentralized treatment systems can relay their alarms to a central location, which allows low-cost maintenance of the decentralized treatment systems by a single company or entity. Hence, the world should move from centralized treatment to decentralized treatment with centralized monitoring, as shown in Figure 1.

5. Sustainable Design of Treatment Process

Although septic tanks have major limitations in terms of their capabilities and mainly rely on the soil drain field to treat the wastewater, they fulfill a major advantage of decentralized treatment, which is recycle the water back to the ground water aquifers, thereby perpetuating the availability of fresh water. Septic tanks are passive systems requiring no electrical power connection. However, aerobic systems use electrical power to run blowers (aerators) which provide dissolved oxygen for biotreatment of the wastewater. More advanced treatment systems, such as Membrane Bioreactors, use electrical power to not only aerate the water, but also to operate pumps for pumping the water through membranes. In remote areas, and in many underdeveloped countries, with unreliable electric power availability, treatment systems which require power cannot function effectively.

Use of solar power for treating wastewater is necessary for global application of any decentralized treatment systems. This approach does not sacrifice treatment capability and uses sustainable power for its operation.

Constructed Wetlands is considered a sustainable, decentralized treatment system that is being used all over the world. These systems have benefits of low capital and operating costs, less infrastructure, simple design and ease of operation. However, they require more land space (2 m^2 per population equivalent in warm weather conditions and 12 m^2/population equivalent under cold climate conditions, produce biomass which has to be periodically removed and handled, limited nutrient treatment and no disinfection capability [12,13].

Another approach is Nature-Assisted Treatment which uses solar-powered aeration with sprinklers and algae cultivation to enhance dissolved oxygen levels, lower biomass production and increase nutrient treatment levels. However, any treatment depending on nature will be inherently slow and require more land space. In addition, these systems are unable to handle industrial discharges which often contain toxic metals, and other impurities.

Membrane Bioreactors [14] which combine biological treatment with membranes, for separation of solids, has emerged as an effective treatment system for wastewaters. By retaining the biomass within the bioreactor, using membranes to permeate only water from the reactor, they operate at a significantly higher biomass concentration, which decreases the residence time needed to treat the wastewater and increases the Biomass Retention Time, to enhance aerobic digestion of the biomass. However, synthetic membranes require periodic chemical cleaning to maintain an acceptable water flux through the membrane. Another issue with membrane bioreactors is limitations of oxygen transfer at high biomass concentrations.

Membrane aerated biofilm reactors (MABRs) [15,16,17,18] represent a new technology for aerobic wastewater treatment. Oxygen diffuses through a gas permeable membrane into the biofilm where oxidation of pollutants, supplied on the biofilm side of the membrane, takes place. While these biofilm reactors exhibit high oxygen transfer rates, they are limited by the transport of the substrates (contaminants) into the biofilm. In addition, the membrane supporting the biofilm can also clog due to the production of extracellular polysaccharide secretions.

Sustainable design requires that the energy consumption for wastewater treatment is minimized. In a typical wastewater treatment plant (WWTP) there are three stages of treatment and the percent energy consumption in each stage are shown in Figure 2 [19]. The first stage is physical treatment, in which the suspended solids are separated from the wastewater. This stage consumes about 25% of the total energy consumption. The second stage is biological treatment which includes aerobic and anaerobic treatment, which converts the organic matter to carbon dioxide, methane and ammonium nitrogen to nitrogen gas and nitrates/nitrites. This stage consumes about 60 -70% of the total energy consumption. The final stage is sludge handling and dewatering, which consumes 4.1 – 13.9% of the total energy consumption.

Clearly, there is substantial potential for energy reduction in the treatment process and it is well known that energy consumption for treatment per unit volume of wastewater decreases with increasing influent water flowrate. However, this does not account for the fact that smaller-scale wastewater

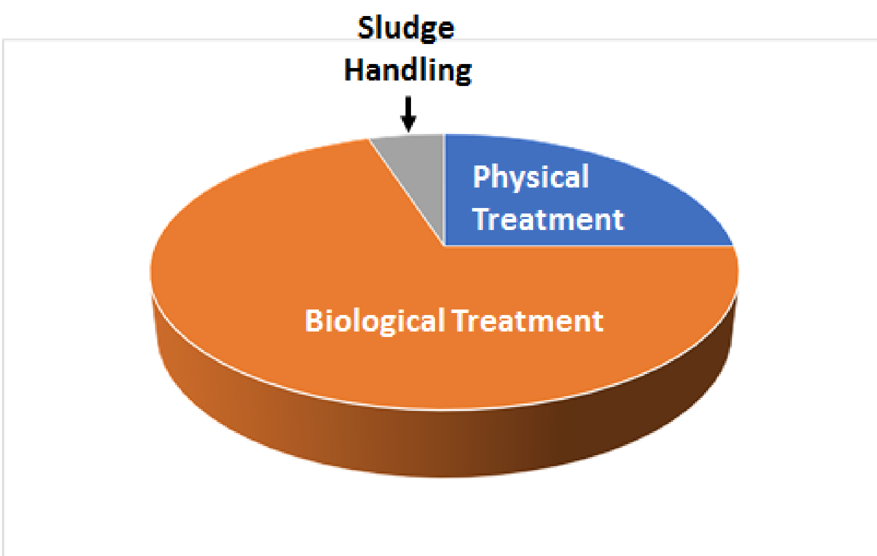


Figure 2. Percent energy consumption in a typical wastewater treatment plant (WWTP).

treatment systems, as in the decentralized approach, can use renewable energy much more easily than large-scale wastewater treatment plants.

6. Filtration of Wastewater

Filtration of suspended solids and biomass is essential for wastewater treatment. In the case of decentralized treatment, self-cleaning filters exhibit low pressure drop and can function without manual intervention. Self-cleaning is achieved either mechanically, using a rotating brush, or by back flushing with filtered water. Two kinds of filter screens are used in self-cleaning filters: (1) thin screen with straight openings from one side to the other side; and (2) porous material with tortuosity of pores within the material. Porous material screens are difficult to clean since material within the tortuous pores is difficult to dislodge. Back flushing usually allows water to flow through the open pores and does not dislodge the material within clogged pores, which gradually allows the filter material to clog. Thin screens with straight openings can be cleaned mechanically using brushes on the dirty side of the screen, since water back flushing tends to flow water through the open spaces rather than clean the openings which are clogged. Additional advantage of using brushes to clean screens with no pore tortuosity is that it can remove biofilms which tend to grow on the screen, when the filter is not being used.

7. Wastewater Aeration

Aeration is essential for biological treatment of wastewater under aerobic conditions and its goal is to maximize the concentration of dissolved oxygen in the aqueous phase. There are two main kinds of aeration systems: (1) natural aeration, which uses no energy and relies on oxygen transfer as the water moves through atmospheric air; and (2) engineered aeration, which can be of several kinds: (1) basin aeration; (2) surface aeration; and (3) in-line aeration. Furthermore, aeration can be achieved with either ambient air or with higher concentrations of oxygen separated from air.

Surface aeration often is unable to oxygenate the entire water column, unless the basin is shallow, which makes the footprint of the aeration basin very large. This stratification of oxygenated water at the top, results in anoxic conditions at the bottom of the basin, resulting in the formation of methane gas, especially in lagoons, where low water velocity creates sedimentation of the suspended solids. Settled solids begin to accumulate at the bottom. Lagoons must be dredged when the sediment thickness increases, and this waste sludge, often rich in nitrogen and phosphorus, has to be treated before disposal.

Most used aeration method is basin aeration, wherein ambient air is bubbled from the bottom of the basin. Oxygen in the air bubbles dissolves into water as the bubbles rise through the water column. To achieve higher levels of oxygen transfer into the water, greater basin depths are used, which reduces the surface area of the basin. Fine bubble aeration is used to increase interfacial surface area between the air bubble and water. However, basin aeration requires the diffusers to be periodically cleaned, to maintain their higher oxygen transfer efficiency. Coarse bubble aeration is often used with fine bubbles to allow the water column to mix, in order to prevent stratification in the basin. Major disadvantage of bubble aeration using deep basins is the high cost of air compression, which makes aeration for biological treatment one of the highest energy consuming steps in wastewater treatment.

The oxygen transfer efficiency is measured in clean water, defined as Standard Oxygen Transfer Efficiency (SOTE) and the fouling potential of the aerators is defined by a parameter α, which is 1.0 for clean water. The actual oxygen transfer efficiency (OTE) is defined as the product of this parameter α with SOTE, measured in clean water. Table 1 lists the values of OTE versus bubble diameter. These values show that while ultrafine bubbles have a higher SOTE/ft compared to larger size bubbles, their fouling parameter, α, is also lower, which gives a lower OTE and the energy consumption to generate ultrafine bubbles is also higher than for larger size bubbles.

In recent years aeration using microbubbles and nanobubbles or micro-nano bubbles (MNB) have been developed and applied for wastewater treatment [20,21]. The critical diameter separating bubble swelling and shrinkage is about 50–65 μm, as shown in Figure 3 [22]. Bubbles larger than the critical value will swell, while smaller bubbles will shrink. Furthermore, surface charge of the MNB's prevents them from coalescing to form larger bubbles. The bubble rise time for different size air bubbles is shown in Figure 4. As the bubble diameter decreases below 30 mm, the bubble rise velocity decreases dramatically, thereby increasing the bubble's residence time from a few minutes to hours and even days. This increases oxygen transfer from the air bubble into the water phase.

However, since air has an oxygen concentration of 21%, and mass transfer coefficients for oxygen and nitrogen are very similar, after oxygen in the bubble has been utilized, these nanobubbles of nitrogen serve no function in the bioreactor. In fact, they pose several disadvantages in a bioreactor system. Firstly, nanobubbles are capable of shearing biofilms from the surface of the biomedia, and the functioning of a bioreactor is based on the existence and maintenance of active biofilms. Secondly, the presence of nitrogen bubbles decreases the bulk density of water, which causes activated sludge flocs to float rather than remain immersed in the wastewater. Flotation of activated sludge also occurs due to surface attachment of nanobubbles to the microbial flocs. This also inhibits biological degradation in a suspended culture treatment system, like activated sludge basins.

Table 1. Oxygen Transfer Efficiency (OTE) versus Bubble Diameter in Basin Aeration.

Diffuser Type	Typical SOTE (%/ft)	α (SOTE)/ft	OTE = α x SOTE	Energy Consumption Multiplier
Coarse Bubble (> 4 mm)	0.90	0.75	0.675%	1
Medium Bubble (2.5 mm – 4.0 mm)	1.40%	0.60	0.84%	0.756
Fine Bubble (1 mm – 2.5 mm)	2.1%	0.49	1.03%	0.476
Ultrafine Bubble (0.5 mm – 1.5 mm)	2.6%	0.3	0.78%	1.243

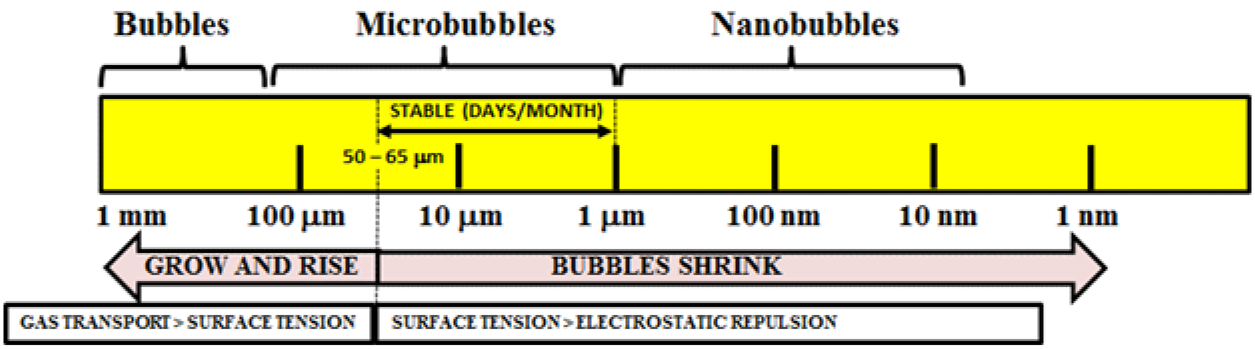


Figure 3. Stability of various air bubble sizes in wastewater.

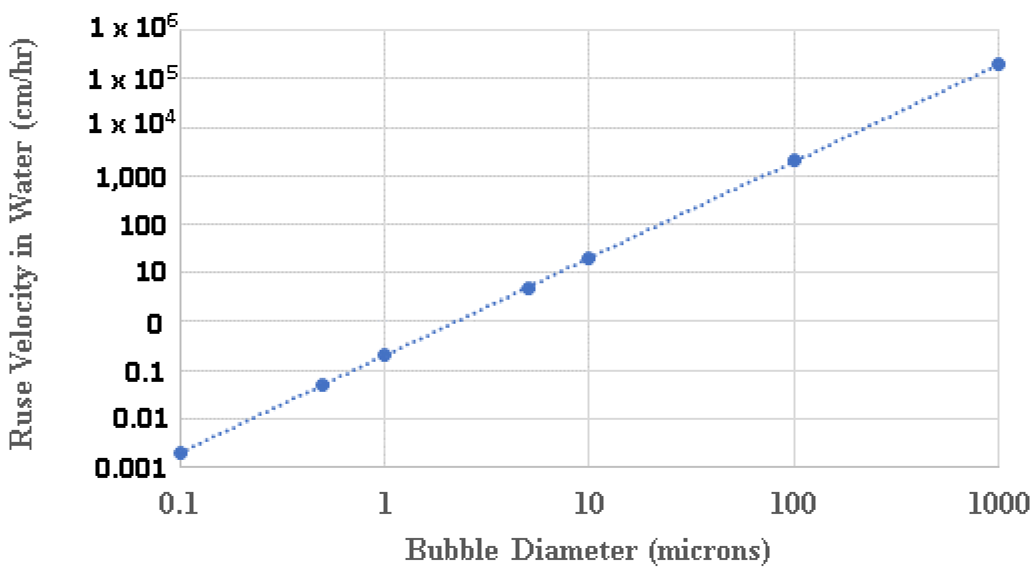


Figure 4. Rise Velocity for Micro and Nanobubbles in Water.

Hence, the optimum bubble size for aeration is in the range of microbubbles, 1-100 µm range, which have a reasonable liquid residence time, ability to coalesce due to low surface charge and ability to deliver oxygen from the air at high oxygen transfer efficiencies.

Liquid mixing is also an important aspect of bioreactor effectiveness, and microbubbles do not possess sufficient drag to effectively mix the wastewater in the bioreactor. The most effective method of mixing the liquid in a bioreactor is to use recycle flow, since it can be scaled up easily for large flow treatment systems. In line aeration achieves this goal of recycling water from the bioreactor and aerating this flow before it re-enters the bioreactor.

The most common way of in-line aeration is achieved by using a venturi [24], which creates a negative pressure when water flow through a narrow region, as described by the Bernoulli equation. In a venturi in-line aeration system, air is introduced into the throat of the venturi, where the water velocity is high, and air is drawn in due to the negative pressure created by the high-water velocity. However, the main issue with venturi aeration is the air-water ratio which can be achieved. Biological treatment requires a specific air/water ratio as shown in Figure 5. This calculation assumes air at 1 atm, 25 deg C, various influent BOD in mg/L and 25% oxygen absorption efficiency. Clearly, for an influent BOD of 500 mg/L the air/water ratio required is 6.65, which is much greater than air/water ratio which can be achieved using a venturi which is typically less than 2.0. Hence, venturi's alone cannot be effectively used for wastewater aeration since they are unable to provide sufficient dissolved oxygen in the water.

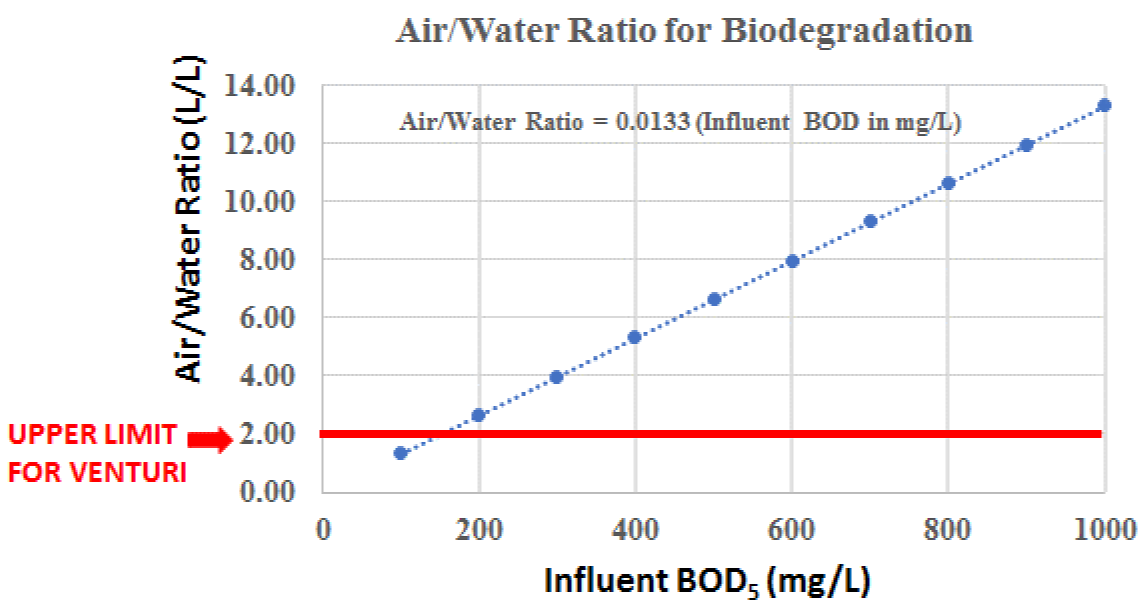


Figure 5. Air/Water Ratio needed for Biodegradation and Limit for Venturis.

Microbubble generation using microporous membranes is a simple, energy-efficient method for generating microbubbles wherein a high gas/liquid ratio, as required for biodegradation of wastewater, as shown in Figure 4. Bubble size depends on the shear stress exerted by the liquid flow and the hydrophobicity of the membrane [25,26, 27,28]. This method allows the air/water ratio to be controlled independently, and the microbubbles can be generated in-line, thereby allowing the recycled liquid to mix the bioreactor. Furthermore, the gas and liquid phase pressure drops are small, compared to other methods of microbubble generation such as venturi, high pressure dissolution, etc.

8. Suspended vs Immobilized Biomass

Wastewater treatment in domestic WWTPs is usually conducted using an activated sludge process, which uses suspended cultures. The main disadvantages of suspended cultures are low biomass concentration, typically 2,000 – 3,000 mg/L, high impact of biocidal chemicals, washout of the suspended biomass at high water flowrates, and net generation of biomass, which is either digested subsequently in aerobic digesters or used to generate biogas. Immobilized biomass or biofilms allow a substantially higher concentration of active biomass, typically 10,000 – 50,000 mg/L, less adverse impact of biocidal chemicals [29], no washout of the active biofilms, and robust performance even at low operating water temperatures. Immobilized biomass or biofilms are achieved using biomedia, which presents a high surface area for active biofilms to attach and grow on its surface. Various kinds of biomedia have been used, such as plastic pieces of various shapes and sizes, open-cell polyurethane foam pieces, porous ceramic media and even rocks.

Biomedia can be used as moving media in the water phase or in a static packed bed with trickling flow of wastewater.

Flexible polyurethane foams have gained relevance as microbial carriers for their good mechanical properties, high porosity, large adsorption surface, resistance to organic solvents and microbial attack, easy handling, allow biofilms to slough-off the surface and cost effectiveness [15]. In general, the high rates of sorption of positive charge and hydrophobic character of the polyurethane foam, allow interaction with most microbial cell surfaces [29,30, 31].

The main issue with open-cell polyurethane foam is its ability to get clogged with suspended solids, typically present in wastewater. To protect the open cell foam, it is enclosed in a spherical, plastic mesh which protects the foam structure from large biomass flocs and other suspended solids. Figure 6 shows a photograph of such a piece of Biomesh Biomedia™, which consists of the plastic mesh that encloses a piece of open-cell, polyurethane foam. The foam piece provides a very large surface area for immobilization of active biofilms, and the main characteristics of this biomedia are given in Table 2.

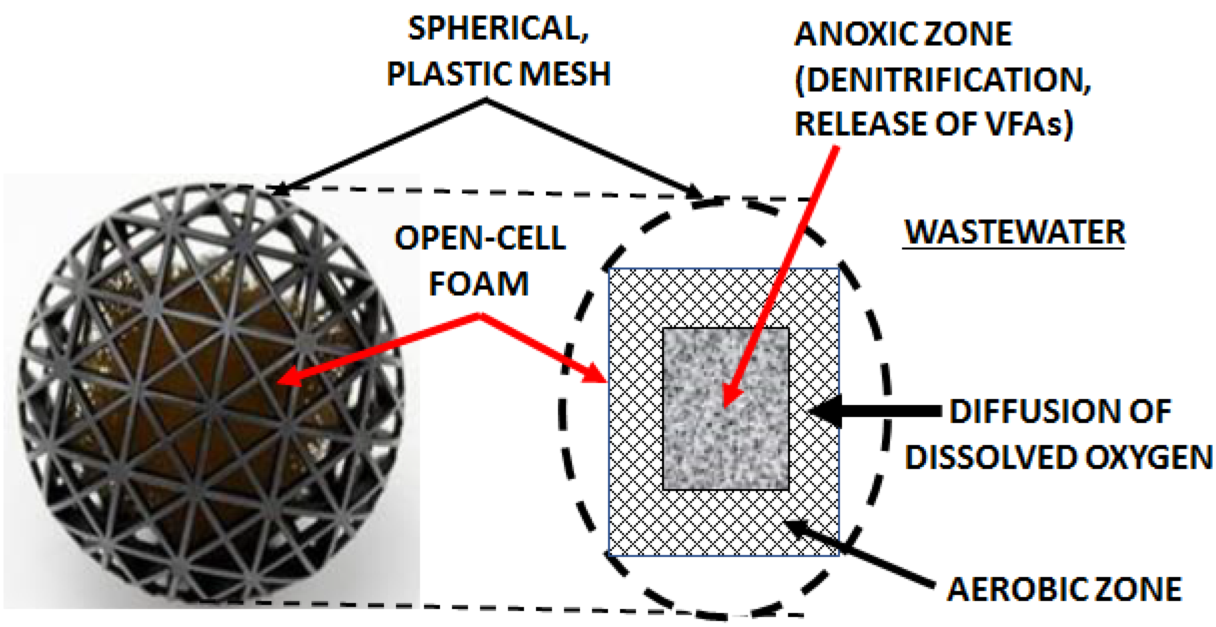


Figure 6. Photograph of Biomesh Biomedia and the formation of aerobic and anoxic zones within the open-cell foam piece, inside the plastic mesh.

Table 2. Characteristics of Biomesh Biomedia.

Biomedia Characteristics	
Material	Non-toxic polypropylene, Black
Bulk Density with no biofilms	3.5 lbs/ft3
Surface Area for biofilms	2,200 ft^2/ft^3
Diameter	2 inches
Maximum Temperature	140 deg F
Typical volume % in Bioreactor	10% – 25%
BOD_5 Oxidation Rate	0.45 lbs BOD_5/ft^3.day
Ammonium Nitrification Rate	0.02 lbs NH_4-N/ft^3.day
Denitrification Rate	0.05 lbs NO_x-N/ft^3.day

One of the central issues with any biomedia, whether stationary or moving, is clogging due to biomass growth [32]. Most plastic medias currently on the market clog due to biomass growth. Also, the surface area in a biomedia can be sub-divided into two types: (1) protected; and (2) unprotected. Protected surface area represents surface area in a biomedia in which the biofilm is not subjected to any action which may slough-off the biofilm from the biomedia. The outside surface area of any biomedia which is subjected to either water velocity that exerts frictional force on the biofilm or where the biomedia can rub against each other to abrade the biofilm off the surface, is unprotected surface area. Biofilms in the protected areas of the biomedia tend to grow significantly thicker while biomedi's surface area in unprotected areas are unable to sustain any active biofilms. This allows the biomedia to begin to clog in the protected areas while having no biofilm in the unprotected areas, and this significantly reduces the total effective surface area of the biomedia.

Determining the maximum biofilm thickness is critical for any biomedia [32], since it determines the opening size for the protected areas. The maximum biofilm thickness must be 50% or less than the size of the openings in the protected areas of the biomedia. Using F as the influent flowrate in million gallons per day, C_{in} mg/L as the maximum influent soluble BOD in most domestic wastewater treatment applications, 0.8 as the fraction of volatile suspended soids to the mixed liquor suspended solids, and Y (lb/lb) as the biomass yield, the total amount of biomass that will result from complete treatment of this wastewater is given by the following equation:

$$\text{Rate of biomass growth (lbs/day)} = (F \times C_{in} \times Y \times 8.34)/0.8 = 10.425F \times C_{in} \times Y \quad \ldots\ldots\ldots\ldots\ldots \quad (1)$$

For steady-state to be achieved in the bioreactor, the rate of growth of biomass must be equal to its decay rate. When this steady-state is achieved, the biofilm thickness would have attained a constant value. This also assumes that there is no removal of biofilms due to slough-off from the protected areas of the biomedia. At this steady-state condition, the decay rate of biomass is given by the decay rate, which is in the range of 0.025 – 0.075 day^{-1}, with an average value of 0.06 day^{-1}. Using this average decay rate and W lbs is the total amount of biomass in the bioreactor, we get the following equation when biomass growth and decay are equal:

$$0.06 \times W = 10.425 \times C_{in} \times Y \quad \ldots\ldots \quad (2)$$

This gives the total amount of biomass is the bioreactor, W lbs as follows:

$$W\ (lbs) = 173.75 \times C_{in} \times Y \quad \ldots\ldots \quad (3)$$

Using density of water (62.5 lbs/ft3) as density of biomass, we get the total volume of biomass in the bioreactor, VB (ft3) as follows:

$$VB\ (ft^3) = 2.78 \times C_{in} \times Y \quad \ldots\ldots \quad (4)$$

If the fractional protected area of the biomedia (protected area/total surface area) is AB,, and A_{tot} is the total surface area of all the biomedia pieces in the bioreactor, then the maximum biofilm thickness is given by the volume of biomass dived by the total protected area, i.e.,

$$\text{Maximum biofilm thickness (ft)} = VB\ (ft^3)/(AB \times Atot) = 2.78 \times Cin \times Y/(AB \times A_{tot}) \ldots\ldots \quad (5)$$

The above equation gives the maximum biofilm thickness since we are assuming that all biomass growth is on the biomedia and there are no suspended cultures in the liquid phase. Clearly, if the maximum opening size dimension, t (ft) is less than or nearly equal to the maximum biofilm thickness, then the biomedia will clog.

$$t \leq 2.78 \times C_{in} \times Y/(AB \times A_{tot}) \text{ Biomedia will clog} \quad \ldots\ldots \quad (6)$$

$$t >> 2.78 \times C_{in} \times Y/(AB \times A_{tot}) \text{ Biomedia will not clog} \quad \ldots\ldots \quad (7)$$

9. Disinfection of Treated Water

The goal of disinfection is to destroy and/or inactivate pathogenic organisms to minimize the spread of water-borne diseases. The dose of a disinfectant chemical, CT is defined as its concentration in water multiplied by the contact time between the disinfectant and wastewater. A very high concentration of disinfectant in contact with the pathogen for a very short time may result in the same effectiveness as a low concentration of disinfectant in contact with the wastewater for a long time. Pathogens are usually associated with solids, and with effective separation of total suspended solids, most of the pathogens are removed from the wastewater. When the treated water is discharged into a soil drain field, any residual of a chemical disinfectant is undesirable, since it prevents the active bacteria in the soil from effectively treating the water. However, with the use of advanced decentralized treatment systems, which effectively treat the wastewater before discharge, disinfectants such as chlorine, which can maintain a residual concentration, can harm the natural biota in the soil drain field.

The disinfectant chemical which has several advantages over other chemicals, such as chlorine, peracetic acid, etc. is ozone. Ozone can be produced on-site with oxygen from ambient air. The disinfecting mechanisms of dissolved ozone in water includes direct oxidation/destruction of the cell wall, damage to the cell constituents, such as nucleic acids, reactions with radical by-products of ozone decomposition, such as hydrogen peroxy and hydroxyl, and breakage of nitrogen-carbon bonds leading to depolymerization. Ozone eventually decomposes to oxygen and water, leaving no toxic by-products.

For decentralized treatment, ozone is a better disinfectant than UV light since UV light requires a high degree of filtration to separate the suspended solids. Typically, the concentration of ozone generated from dried, ambient air is in the rage of 30 – 60 mg/L in air. At maximum water temperature of 35 deg C the solubility of ozone from an ozone/air mixture will be in the range of 2.57 mg/L – 5.15 mg/L, compared to 8.37 mg/L – 16.7 mg/L at 10 deg C. Hence, water temperature has a significant impact on ozone solubility in water. At the higher temperature ozone absorption is substantially reduced.

The disinfection effectiveness of ozone compared to other chemicals has been summarized in the literature [33]. For *E.Coli* reduction by 99%, the CT value (Concentration in water in mg/L x time in minutes), is 0.02 at a pH of 6-7 [33], which is the pH of treated water. Earlier the ozone concentration of 3.15 mg/L was calculated in the treated water after ozone absorption, for 10% ozone absorption efficiency and 5 gms/hr ozone generating capacity operating at 80% of this generating capacity. Hence, to kill 99% of the organisms typically present in wastewater, the time needed will be 2.0/3.15 = 0.64 minutes.

10. Treatment of Nutrients

Several of the seventeen Sustainable Development Goals (SDGs) are related to managing nutrients in wastewater. Nutrients (nitrogen and phosphorus compounds), normally present in domestic wastewater, when released into water bodies result in the growth of toxic algal blooms. Nutrients also stimulate eutrophication in lakes and rivers, and ammonium-nitrogen in wastewater can deplete water bodies of dissolved oxygen, resulting in fish kills. Eutrophic conditions are a major risk to human health, resulting from consumption of shellfish contaminated with algal toxins or direct exposure to waterborne toxins. Algal blooms have caused major problems in water bodies used to supply drinking water, since chlorine, used to disinfect drinking water, reacts with organic compounds to form disinfection byproducts, which are potential carcinogens and regulated by the U.S. Environmental Protection Agency.

Release of nutrients from decentralized systems, such as a septic tank and even on-site aeration units has been well documented. While treatment of nutrients in centralized treatment systems has been well studied and will not be reviewed here, application of these technologies for decentralized systems is usually difficult and uneconomical.

As discussed earlier, the use of moving biomedia with high surface area enables simultaneous nitrification and denitrification. In the Biomesh Biomedia, the inner section of the spherical biomedia with open-cell foam inside, operates under anoxic conditions due to the inability of dissolved oxygen to penetrate through the aerobic section, near the outer surface of the biomedia. In the aerobic section, ammonium-nitrogen is converted to nitrates and nitrites, while in the anoxic section, within the moving biomedia, nitrates and nitrites are converted to nitrogen gas. This simultaneous nitrification and denitrification, which occurs when using Biomesh Biomedia, enables the ammonium-nitrogen to be converted to nitrogen gas.

The second strategy which can be employed in decentralized treatment systems is the use of adsorption and chemical complexation to remove ammonium-nitrogen, nitrates, nitrites, and phosphates. Recovery and recycle of nutrients is essential since it creates a circular economy. Furthermore, adsorption and chemical complexation can remove nitrogen and phosphorus compounds from wastewater to very low levels. Biological nitrification and denitrification and treatment of phosphates by Polyphosphate Accumulating Organisms (PAOs) can reduce nitrogen and phosphorus levels but achieving very low concentrations is difficult without advanced controls. In decentralized treatment systems, adsorption and chemical complexation are the lowest cost technologies and allow recovery and reuse of the nutrients.

Magnesium salts have been extensively used to remove phosphorus from wastewater. Water insoluble magnesium salts, such as magnesium and cerium carbonates have been used to complex phosphates in wastewater streams. Blast furnace slag (BF slag) is an industrial by-product from steel plants derived from the slag forming minerals, mainly limestone, during iron production in the blast furnace [34]. The cooling process of the molten slag affects the properties of the solidified slag. Cooling by water results in an amorphous and glassy slag, whereas a slow air-cooled slag is a half crystallized, rock-like material [34]. Both adsorption and precipitation mechanisms contribute to the phosphorus sorption by BF slag. Khelifi et al. [35] found adsorption processes to account for about 20% of the total removal of phosphate. When metals of the BF slag were washed away with HCl and distilled water, the phosphorus retention capacity was drastically reduced. Johansson and Gustavsson (2000) investigated crystalline and amorphous BF-slags and found that the major mechanism for phosphorus retention was the precipitation of hydroxyapatite. However, Grüneberg and Kern [36] performed sequential extraction tests showing that the major fraction of the sorbed phosphorus was defined as readily available, loosely bound phosphorus or phosphorus associated with free Ca or Mg carbonates. Johansson [37] reported similar results.

Wollastonite is a calcium metasilicate ($CaSiO_3$) with a theoretical composition of 48.3% calcium oxide and 51.7% silicon dioxide [38]. Wollastonite belongs to the pyroxenoid group, often appearing with a needle shaped structure [39,40]. It is built up by Si-O tetrahedra and Ca atoms situated inside the tetrahedra [41].

The attraction mechanisms for positively charged ions to wollastonite have not been thoroughly described. However, Xie and Walther [42] studied alkali and alkaline earth metal-bearing silicates, e.g. wollastonite, and found that cations (e.g. Ca^{2+}) were released when added to pure water, leaving a negatively charged mineral surface. In the same study, silicon (Si^{4+}) of wollastonite was released into solution, also contributing to the negative surface of the wollastonite. The negative wollastonite surfaces attracted positively charged ions such as H^+. Cations such as K^+, Na^+, and Ca^{2+} as well as NH^{4+} were adsorbed to the wollastonite surfaces and exchanged H^+, which was most easily bound to the negatively charged surface [42].

Wollastonite has been investigated regarding its adsorption of positively charged metal ions [43-47. Lind et al. [48] studied adsorption of ammonium in human urine on wollastonite, with about 50% of the ammonium in the solution being adsorbed to the wollastonite. In addition, wollastonite has been investigated regarding removal of phosphorus. Wollastonite was suggested as a filter material for phosphorus removal in con-structed wetlands in surveys [49,50]. Palacios and Timmons [51] obtained promising results when investigating wollastonite as a filter material to treat recirculating water in an aquaculture.

Clinoptilolite is a natural zeolite that belongs to a group of hydrated aluminosilicate minerals containing alkali and alkaline-earth-metals. The chemical formula of clinoptilolite is

$(Na,K,Ca_{0.5},Mg_{0.5},)_6(AlO_2)_6(SiO_2)30 \cdot 24H_2O$ [51]. Clinoptilolite has a structure consisting of a three-dimensional framework of SiO_4 and AlO4 tetrahedra, with the Si^{4+} or Al^{3+} ions located at the centers of the tetrahedra [53]. Substituting aluminum for silicon in the mineral lattice of clinoptilolite creates a negative charge of the lattice balanced by positively charged ions, such as sodium, calcium and potassium [52]. These ions are located in the relatively large cavities of the outer framework and are exchangeable by other cations [52]. Furthermore, clinoptilolite has an ion sieving capability, since the framework structure forms narrow ring channels with dimensions of 3x4.4 Å and 3.5x7.9 Å [53]. The positively charged ions located within the clinoptilolite framework can be replaced relatively easily with other positive ions, known as "ion exchange" or "cation exchange". The concept "adsorption" can be used for the same phenomenon, referring to positively charged ions that can be adsorbed to the negatively charged surfaces of the clinoptilolite. The term adsorption is connected to "desorption", since the adsorbed ions can be removed/desorbed by other ions.

Ames [54,55] conducted experiments to rank cations according to their affinity to clinoptilolite and developed the following order:

$$Cs^+ > Rb^+ > K^+ > NH_4^+ > Ba^{2+} > Sr^{2+} > Na^+ > Ca^{2+} > Fe^{3+} > Al^{3+} > Mg^{2+} > Li^+$$

As can be seen by the above affinity sequence, clinoptilolite has a high affinity for ammonium in solutions as studied by e.g. [56 – 59]. The ammonium adsorption capability of clinoptilolite makes it interesting for wastewater treatment applications [60 – 70]. Many factors influence the adsorbed amount of ammonium on clinoptilolite in practical applications. Studied aspects are the origin and clinoptilolite concentration of the ore sample used [71,72], transformation of the clinoptilolite to a homoionic form [57, 60, 63, 65], grain size [54, 58, 66, hydraulic load [63, 65, 66, 67], ammonium concentration [57, 58, 59], competition with other cations [60, 56, 73, 74], occurrence of suspended solids and organic matter in the wastewater [58, 75], pH [59,60], temperature [60, 76], and scale of system [58].

Desorption of ammonium ions from the clinoptilolite is of interest to enable recovery of the ammonium ions or make the exchange sites available for new ions. Another method for de-sorption of ammonium ions is chemical regeneration with NaCl brine solutions, as studied by, e.g., [60, 77, 61, 58] and means that adsorbed ammonium ions are desorbed/exchanged by Na ions when the regeneration solution is flushed through the clinoptilolite. Chemical regeneration can be combined with nitrification to desorb ammonium ions, called biological regeneration [77].

More than 200 reactive filter systems using Filtralite P have been constructed in Norway during recent years, which makes it the probably most used reactive filter material in full-scale applications. About 70-80 compact reactive filter systems with Nordkalk Filtra P have been built in Finland during the last 2-3 years. A reactive filter system with blast furnace slag for small scale wastewater treatment was constructed in Luleå in 2005 [78]. A larger reactive filter system to treat urine separated wastewater was built in 2003 at a highway rest stop with toilet facilities in Ångersjön, Hudiksvall. In one of the treatment lines, a filter bed with blast furnace slag for phosphorus sorption was located after a limestone filter [79]. In Turkey, a reactive filter bed using blast furnace slag and planted with Phragmites australis to treat domestic wastewater (3 m^3/d) was built in 2001 [80]. In a Canadian system designed for about 100 persons, blast furnace slag was investigated at an experimental plant with reactive filter beds to treat lagoon and wetland effluents [81]. In New Zealand, a pond system for a population of about 6,000 was upgraded 1993 with steel-melter slag filters to increase the phosphorus removal [82].

The phosphorus treatment efficiencies of large Norwegian filter bed systems using about 40 m^3 of filter material have been high. Systems using porous filter materials with high phosphorus sorption capacity have consistently removed more than 90% of the phosphorus for more than 10 years. These systems were designed with a total surface area of 7-12 m^2/person [83]. The compact filter systems built in Finland using Nordkalk Filtra P reduced phosphorus concentrations of the wastewater by more than 90% during a period of 1-2 years. In these systems, a 1 m^3 tank filled with the Nordkalk Filtra P was loaded with wastewater from one family [84].

11. Types of Biofilms in Bioreactors

There are several types of bioreactors which have been used for wastewater treatment. While the various types of bioreactors will not be reviewed here, the use of immobilized biomass for decentralized treatment is preferred since it has a smaller footprint (significantly higher concentrations of active biomass), less net generation of waste sludge due to higher sludge retention times, and ability to withstand presence of biocides in the influent wastewater. However, immobilized biomass bioreactors can be either liquid phase filled, with moving biomedia or a trickling filter, with a packed bed of biomedia and gravity-driven trickling flow of wastewater through the packed bed.

From a biofilm point of view, bioreactors can be described in terms of the biofilm characteristics, as shown in Figure 7. In Figure 7(a), biomedia in the liquid phase supports the active biofilm, and the contaminant, dissolved oxygen and nutrients diffuse from the bulk liquid phase into the biofilm. This situation occurs in liquid-phase bioreactors with either moving or stationary biomedia. In this case, there is a maximum thickness of biofilm which can be achieved, since the organic contaminants and dissolved oxygen are being consumed as a function of increasing biofilm depth, and at the maximum biofilm thickness, the organic contaminant(s) and/or dissolved oxygen concentration is reduced to zero. In Figure 7(b), there is trickling flow of wastewater on the surface of the active biofilm, supported by the biomedia. This is the case in Trickling Filters and packed bed bioreactors, in which the dissolved oxygen in the trickling flow of wastewater, contaminants and nutrients simultaneously diffuse into the biofilm. The biofilm thickness is also limited as in the case of Figure 7(a). In Figure 7(c) the wastewater flows on the biofilm side, which is supported on a porous membrane, and the oxygen, present in air, dissolves into the liquid-phase present in the biofilm through the porous membrane. This occurs in Membrane-Assisted Biofilm Reactors (MABR), wherein dissolved oxygen and contaminant(s)/Nutrients diffuse into the biofilm in opposite directions. This results in thicker biofilms, since dissolved oxygen diffuses on the back side of the biofilm which is supported by the porous membrane. In this case, the maximum biofilm thickness is mainly limited by the diffusion flux of the contaminant(s) through the biofilm.

Figure 7(d) shows a moving biomedia bioreactor except in this case aeration is achieved using micro nanobubbles, rather than standard aeration. Microbubbles in the size range of 1 - 100 microns have a significantly higher surface area than fine bubble aeration and this allows a significantly higher oxygen transfer rate into the biofilm. In addition, nanobubbles, which are less than 1 micron in diameter, stay in the water for days and attach themselves to the biofilm surface, since the buoyancy force is cancelled by their weight, and they have a hydrophobic surface, which enables them to adsorb on the biofilm's surface.

Treatment of contaminants in the bulk liquid phase, as in Figure 7(a), is limited by the mixing of the liquid, which controls the mas transfer rates from the bulk liquid phase into the active biofilm, and the concentration of dissolved oxygen in the bulk liquid. However, in this case, the liquid residence time in the bioreactor can be designed since it depends on the volume of liquid in the bioreactor and the influent wastewater flowrate. By increasing the size of the bioreactor, the liquid residence time in the treatment system can be designed and implemented.

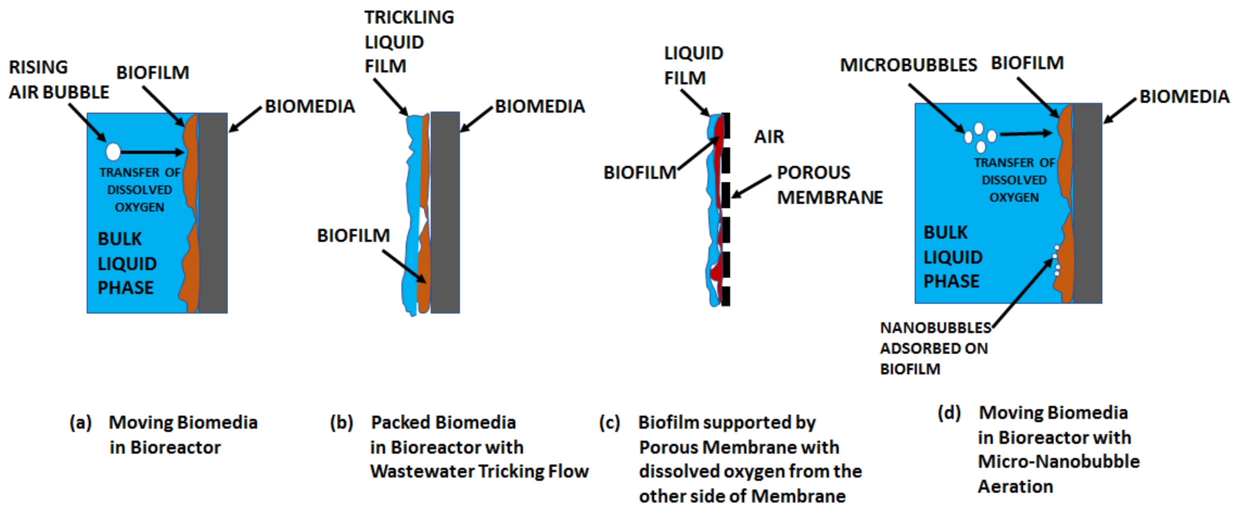


Figure 7. Biofilm Characteristics in Various Bioreactors.

When the wastewater is trickled down the bed of biomedia, as shown in Figure 7(b), the liquid residence time is limited by the time it takes for the liquid to trickle down by gravity through the biomedia packing. To increase this liquid residence time in the packed bed, recycle of the effluent flow from the bottom of the packed bed to the spray heads located at the top, is conducted at the expense of electric power consumption by the recycle pumps. A high recycle ratio, which is the ratio of the recycle flowrate divided by the influent flowrate before addition of the recycle flow stream, makes the packed bed behave hydrodynamically as a completely mixed, liquid-phase bioreactor. However, the thin flowing liquid film on the surface of the biomedia, with immobilized active biofilms, allows higher mass transfer rates than in the moving bed bioreactor. In traditional trickling bed bioreactors, the air movement outside the trickling liquid film is achieved by natural convection caused by a temperature difference between the ambient air and the wastewater. However, in summertime, when the wastewater coming into the trickling beds is warm, this temperature difference is small, which causes stagnant air within the trickling flow bioreactor, causing poor oxygen transfer into the water.

The advantage of the Membrane Assisted Biofilm Reactor (MABR) is the availability of dissolved oxygen due to diffusion across the porous membrane and the high surface area of the membrane pores. However, aerobic biofilms produce polysaccharides (slime) which is likely to clog the membrane pores over time. The second big advantage is the power consumption needed to provide adequate dissolved oxygen in the wastewater. In moving biomedia bioreactors, wherein the biofilm immobilized on the biomedia's surface is completely submerged in the liquid, aeration consumes a significant amount of energy, as shown in Figure 1, since ambient air has to be compressed in order to bubble it at the bottom of the bioreactor. In trickling beds, although air flows under natural convection, recycle of the liquid consumes energy. However, in a MABR system, air flows on the other side of the membrane at very low pressure drop and the liquid trickles down under gravity or at a small pressure difference. When micro nanobubbles are generated at low energy consumption using porous membranes, high rates of oxygen transfer are achieved, with no membrane clogging issues as in the MABR case.

The rate of biodegradation depends on three factors: (1) concentration of contaminant(s); (2) concentration of dissolved oxygen; and (3) concentration of nutrient(s). This is shown in Figure 8. The figure corresponding to Figures 7(a), 7(b) and 7(d), the concentration of the contaminant(s), nutrients and dissolved oxygen are high at the liquid-biofilm interface and declines within the depth of the biofilm. This results in a high biodegradation rate at the liquid-biofilm interface and declines as the concentrations of the contaminant(s), nutrients and dissolved oxygen decrease with increasing depth of the biofilm. In a MABR system, the highest concentration of dissolved oxygen is when the concentration of the contaminant(s) and nutrients is lowest, as shown in Figure 8 corresponding to Figure 7(c), which gives a lower biodegradation rate, since this rate is the product of all three concentrations. In Figure 8 corresponding to Figure 7(d), using micro nanobubble aeration, higher rates of biodegradation are achieved at the liquid-biofilm interface compared to cases in Figures 7(a) and 7(b), since the dissolved oxygen concentration is higher compared to fine bubble aeration in moving bed bioreactors and in trickling filters. The rate of biodegradation in the MABR system depends on the product of decreasing contaminant(s) and nutrient concentrations and varying dissolved oxygen concentration, as shown in Figure 8.

Sustainability requires low energy consumption and high biodegradation rates which can be achieved in a moving bed bioreactor system using high surface area biomedia and micro nanobubble aeration..

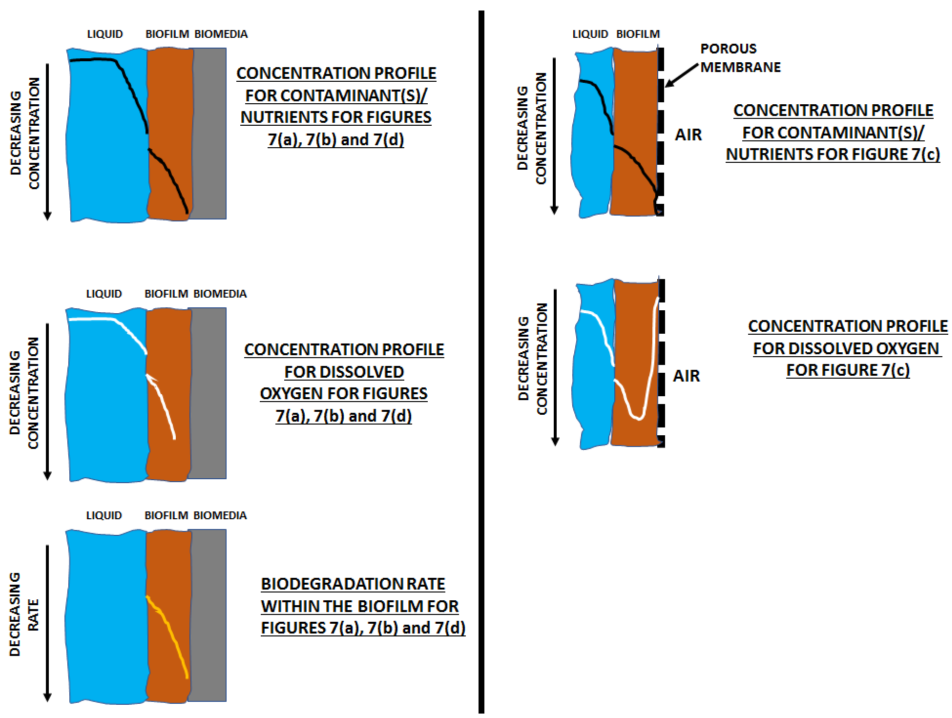
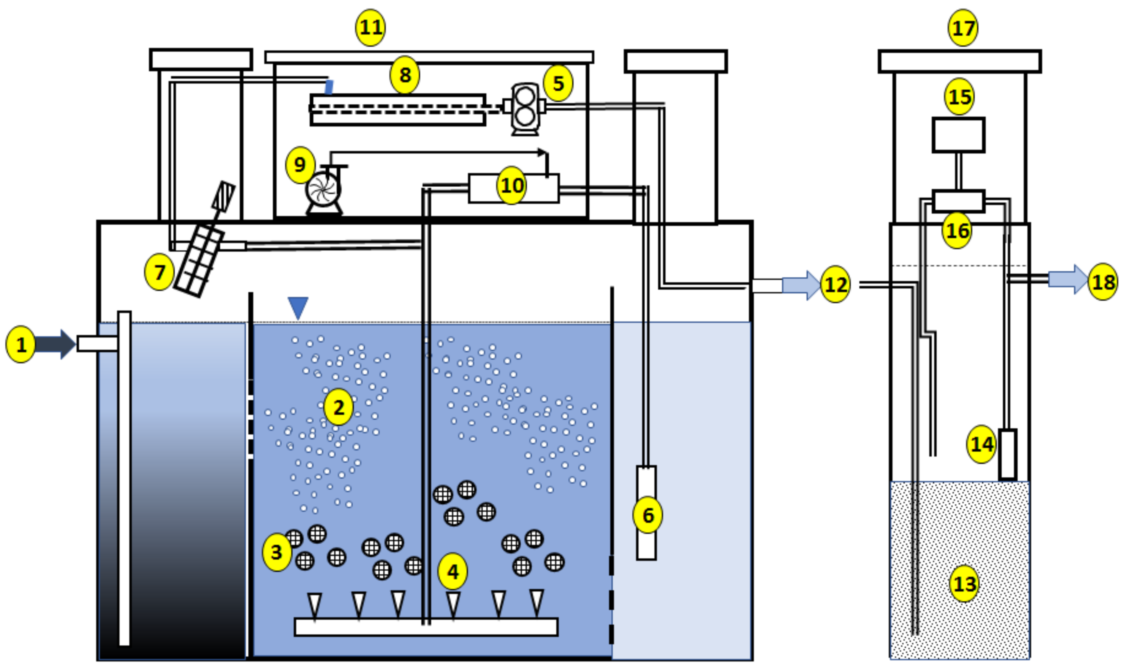


Figure 8. Concentration profiles for contaminant(s), nutrients and dissolved oxygen as a function of Biofilm thickness for cases shown in Figure 7.

12. Decentralized Treatment Process

A compact, decentralized treatment process has been developed for single family homes and sub-divisions in the U.S. using micro-nanobubble aeration, recycle of water to mix the bioreactor and Biomesh Biomedia™ to immobilize the active biofilms [20]. A schematic of the decentralized, sustainable wastewater treatment system is shown in Figure 9. Influent wastewater from a single-family house flow by gravity into the treatment system, which consists of a 1,730-gallon standard septic tank, with 1,500 gallons capacity for water. It has two baffles, the first baffle that has opening below the water surface but well above the bottom of the tank and a second baffle which has openings only at the bottom of the baffle. In the first compartment, influent wastewater from the house with solids enters the tank and the solids begin to settle down. This compartment is anoxic to allow the biosolids to break down slowly. Water from the first compartment then flows into the second compartment, which is aerated with micro nanobubbles and has moving biomedia. Biological treatment of the organic load and conversion of ammonium to nitrates and nitrites occurs in this compartment.

Micro nanobubbles (MNB) are created by using a membrane generator which takes air from the blower and water from the pump, located in the third compartment, and creates an air-water flow with the air present as micro nanobubbles. This air-water mixture is then introduced into the bottom of the second compartment using eductors. The liquid flow flowing through the eductors mixes the water in this compartment and the micro nanobubbles provide dissolved oxygen. The use of eductors prevents any possibility of fouling, as occurs in the case of membrane diffusers.



1. Influent 2. Micro-Nano Bubbles 3. Biomesh Biomedia
4. Eductors 5. Reversible Pump 6. Pump 7. Brush Filter
8. Microfiltration Membrane 9. Blower 10. Membrane MNB Generator
11. Technology Box 12. Water into second tank 13. Adsorptive Media
14. Pump 15. Ozone Generator 16. Membrane MNB Generator
17. Second Tank with Riser 18. Treated, Disinfected Water for reuse

Figure 9. Sustainable Biotreatment System for Wastewater.

Moving biomedia is the Biomesh Biomedia, described earlier in this paper, which provides a very high surface area for the active biofilms to grow and biodegrade the contaminants. This media allows simultaneous nitrification and denitrification, as explained earlier in this paper. This allows the ammonium nitrogen in the influent to be converted to nitrogen gas. The inner section of this moving media also produces volatile fatty acids, which allows the growth of polyphosphate Accumulating Organisms (PAOs) which sequester phosphates.

As explained earlier, biological treatment of nutrients is unable to achieve very low levels of total nitrogen and phosphorus, especially in decentralized treatment systems. Further removal of nutrients will be achieved by adsorption and chemical complexation in the second tank.

Water from the third compartment is also pumped through a self-cleaning filter with a 50-micron screen which has a slowly rotating brush inside to keep the screen free of any solids and biofilms. The solids swept from the surface of the screen fall back into the first compartment and filtered water flows into a stainless-steel membrane. This membrane has an average pore size of 5 microns and is periodically backflushed by a reversible pump. Filtered water through this membrane then flows into the second tank.

The second tank, which is 200 gallons in volume is field in the lower section by an adsorbent, which is capable of adsorbing and chemical complexing nutrients and achieve very low levels of nitrogen and phosphorus. This media eventually after a few years will get saturated and can be tilled into the ground as a fertilizer. This allows the nutrients to be recycled back to the land, where they are needed for growing food and other plants.

Above the adsorptive media is water which is recirculated by a second pump through a membrane MNB generator in which micro nanobubbles of ozone/air mixture are created to disinfect the water. Disinfected water is also pumped out periodically as treated water, which is free of organic contaminants, nutrients, pathogens, bacteria and viruses.

Field performance of the sustainable wastewater treatment system is summarized in Table 3. Two sets of data have been presented. One of single-family homes and the second set for Recreational Vehicle (RV) camps, where the influent wastewater parameters are significantly higher. The field results show that the sustainable treatment system is able to treat the wastewater adequately to allow the treated water to be discharged into an existing water body.

Table 3. Summary of Field Performance of Sustainable Treatment System.

Wastewater Parameter	Influent	Effluent
Single Family House		
Average Daily Flowrate	500 – 700 gallons per day	Same as influent
BOD5	510 mg/L ± 10 mg/L	9.8 mg/L ± 2 mg/L
TSS	310 mg/L ± 12 mg/L	Non-Detect
Ammonium-N	65 mg/L ± 4.7 mg/L	< 2 mg/L
Nitrates/Nitrites	34 mg/L ± 3.2 mg/L	< 1 mg/L
Phosphates	12 mg/L ± 2 mg/L	< 5 mg/L
RV Park		
Average Daily Flowrate	3,000 gallons per day	Same as influent
BOD5	1530 mg/L ± 15 mg/L	9.8 mg/L ± 2 mg/L
TSS	745 mg/L ± 15 mg/L	Non-Detect
Ammonium-N	120 mg/L ± 9.10 mg/L	< 2 mg/L
Nitrates/Nitrites	64 mg/L ± 5.2 mg/L	< 1 mg/L
Phosphates	32 mg/L ± 2 mg/L	< 5 mg/L

13. Conclusions

Decentralized treatment is the future of wastewater treatment. While centralized treatment dominates the current status of wastewater treatment, it has several disadvantages, the most important being that it takes groundwater and eventually discharges it into a creek or river, which ends up in the ocean. This converts fresh water into salt water. This has resulted in the global decline of groundwater levels. Currently, the most common decentralized treatment system is a septic tank, followed by an aerobic treatment unit. These systems are unable to treat nutrients, which is causing toxic algal blooms in water bodies, and are incapable of preventing the release of microplastics into the receiving water bodies and/or the soil drain field, causing it to clog prematurely. Sustainable wastewater treatment requires minimum energy consumption and using renewable energy. In this paper, an on-site, decentralized, sustainable wastewater treatment is presented, which uses micro-nanobubble aeration and high surface area Biomesh Biomedia to effectively treat the wastewater. In addition, it uses self-cleaning filters and membranes to effectively filter the water resulting in non-detect suspended solids, clear water with low level of contaminants. It can be scaled up for on-site wastewater treatment for RV sites, housing sub-divisions, etc.

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