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Research Article

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Overview of Biological Treatment Technologies for Greywater Reuse

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Abstract
Greywater is a wastewater discharge originating from kitchen sinks, showers, baths, washing machines and dishwashers. Properly treated greywater can be recycled to meet global water shortages that is expected to affect 2.7 billion people around the world by 2025. Global water shortage can result in a reduction in agricultural land and increased dissertation leading to poverty, faming, war, illegal migration and human trafficking. Greywater contains fewer pathogens than domestic wastewater, is generally safer to handle and easier to treat and reuse onsite for toilet flushing, landscape and crop irrigation. Recycling of grey water provides substantial benefits for both the water supply system by reducing the demand for fresh clean water, and for the wastewater system by reducing the amount of wastewater required to be conveyed and disposed of. In this paper, the existing biological treatment systems for greywater are reviewed. These are: (a) constructed wet land, (b) sequencing batch reactor, (c) vertical flow bioreactor, (d) membrane bioreactor, (e) up-flow anaerobic sludge blanket, (f) rotating biological contractors, (g) trickling filters, (h) aerated lagoons, (i) anaerobic up-flow filter, and (j) expanded bed up-floe reactor. In a biological treatment, the degradation and transformation of greywater constituents are facilitated by the biochemical reactions carried out by microorganisms in the liquid medium. However, the effluent of biologically treated greywater may contain pathogenic microorganisms, requiring a final disinfection step to eliminate the risk of contracting pathogenic diseases. Selection criteria for a disinfectant include: (a) non-toxicity to humans, domesticated animals, and aquatic ecosystems, (b) low cost (c) easy handling, (d) reliable analysis, and (e) a satisfactory residual concentration. Any disinfection process selected (whether chemical oxidants or irradiation treatment is selected) should be evaluated taken into consideration the conditions of the wastewater source and existing biological treatment design.

Keywords: Water Shortage, Greywater, Biological Treatment, Degradation, Disinfection, Reuse

Introduction
Global water resources are worsening, and water shortages will affect 2.7 billion people by 2025 (about one third of the world population) [1]. This will result in a reduction in agricultural land and increased dissertation leading to poverty, faming, war, illegal migration and human trafficking. Reusing greywater can contribute towards solving this water shortage problem. Greywater is a wastewater discharge originating from kitchen sinks, showers, baths, washing machines and dishwashers [2-6]. Table 1 shows the main sources of greywater and their constituents [1,7,8].

Greywater is different from domestic wastewater (wastewater from toilets) which is classified as black water. As grey water contains fewer pathogens than domestic wastewater, it is generally safer to handle and easier to treat and reuse onsite for toilet flushing, landscape and crop irrigation and other non-potable uses [9-16]. Recycling of grey water provides substantial benefits for (a) the water supply system, by reducing the demand for fresh clean water, and (b) the wastewater system, by reducing the amount of wastewater required to be conveyed and treated [17,18] thereby reducing energy use and chemical pollution associated with treatment and disposal [19].

Studies have shown that greywater use for irrigation or toilet flushing appears to be a safe practice and no additional burden of disease being observed among greywater users irrigating their fields arid regions [20-21]. The aim of this study was to examine available biological treatment technologies for greywater reuse and to review their effectiveness in removal of pollutants from greywater including pathogens.

Greywater Characteristics

In an average residence, greywater accounts for 50-80% of the total wastewater produced [22]. The value varies depending on the number of occupants, demographic, and personal habits [16,23,24]. Greywater generation rates reported in different countries are shown in Table 2 [1,6,14,25-35].

Table 1. Greywater sources and their constituents [1,7,8].

Greywater Source	Constituents
Kitchen	Kitchen greywater contains food residues, high amounts of oil and fat and dishwashing detergents. It may occasionally contain drain cleaners and bleach. Kitchen greywater is high in nutrients and suspended solids and may be very alkaline (due to detergent builders). It contains high salt concentrations and bacteria. It has odor, turbidity and high oxygen demand.
Bathroom	Bathroom greywater is regarded as the least contaminated greywater source within a household. It contains soaps, shampoos, toothpaste and other body care products. Bathroom greywater also contains shaving waste, skin, hair, body-fats, lint and traces of urine and faeces. Greywater originating from shower and bath may thus be contaminated with pathogenic microorganisms. It has odor and turbidity and high oxygen demand.
Laundry	Laundry greywater contains high concentrations of chemicals from soap powders (such as sodium, phosphorous, surfactants, nitrogen) as well as bleaches, suspended solids and possibly oil paints, solvents and nonbiodegradable fibers from clothing. Laundry greywater can contain high amounts of pathogens from washing nappies. It has high pH, salinity and turbidity.

Table 2: Greywater generation rates reported in different studies.

Location	Generation (L/p/d)	Reference
Jordon	50	Al-Hamaiedeh and Bino [25], Halalsheh et al. [26], Faraqui and Al-Jayyousi [27]
Africa (Several Countries)	50-160	Morel and Diener [1]
Asia (Several Countries)	72–225	Morel and Diener [1]
South America (Several Countries)	50-170	Morel and Diener [1]
Arizona, USA	123	Casanova et al. [28]
Australia	113	Morel and Diener [14]
Israel	98	Friedler [29]
Malaysia	225	Martin [30]
Mali	30	Alderlieste and Langeveld [31]
Nepal	72	Shresta [32]
Oman	151	Jamrah et al. [33]
South Africa	20	Adendorff and Stimie [34]
Stockholm	65	Ottoson and Stenstrom [6]
Vietnam	80–110	Busser et al. [35]

Table 3 shows some of the physical, chemical, and biological characteristics of greywater [36-42]. Greywater temperature varies within the range of 18–30 oC which is higher than that of clean water due to the use of warm water for personal hygiene and cooking [38,41,43]. The pH levels of greywater fluctuate depending on the source of greywater and is affected by the level of oil and grease [3,26,35].

Greywater contains salts as can be indicated by electrical conductivity. The electrical conductivity of greywater is typically in the range of 300-1500 µS/cm. Important sources of salts are sodium-based soaps, nitrates and phosphates present in detergents and washing powders. In addition to sodium, greywater can also contain calcium, magnesium, chlorine and boron from detergents [40].

The oil and grease concentrations in greywater depends on its source (cooking grease, vegetable oil, food grease) and high concentrations of fate and grease come from kitchen sinks and dishwashers. Oil and grease concentrations in the ranges of 37-78 mg/L and 8–35 mg/L have been observed in bathroom and laundry greywater sources, respectively [44]. However, values as high as 230 mg/L and 2000 mg/L were reported by Al-Jayyousi and Crites and Tchobanoglous respectively. Greywater containing oil and grease above 30 mg/L should be directed to blackwater collection

[10,11,20,46,22,35,45].

The most common surfactants used in household cleansing chemicals are linear alkylbenzene sulfonate (LAS), alcohol ether sulphate (AES) and alcohol ethoxylate (AE). Although non-biodegradable surfactants have been banned in in most Western countries in the 1960s, these environmentally problematic organic chemicals are still used in many developing countries [46,47]. Laundry and automatic dishwashing detergents are the main sources of surfactants in greywater. Other sources of surfactants include personal cleansing products and household cleaners. The concentration of surfactants present in greywater is strongly dependent on type and amount of detergent used [12,29]. Shafran et al [48]. reported surfactant concentrations in greywater in the range of 17-60 mg/L with the highest concentrations observed in greywater from laundry, shower and kitchen sink.

Suspended solids concentrations in greywater depends on the amount of water used with the highest concentrations typically found in kitchen and laundry discharges. Studies on greywater from some countries showed suspended solids loads of 10–30 g/p/d However, high concentrations of 1389-1396 mg/L were reported [38,40].

Table 3: Some characteristics of greywater reported in different studies.

Parameter	Scheumann et al. [36]	Jefferson et al. [37]	Nolde [38]	Friedler et al [39]	Burnat and Mahmoud [40]	Gross et al. [41]	Dallas et al. [42]
Temperature (^{o}C)						18-30	
pH	6.3-7.1				6.7-8.4	6.3-7.0	
Turbidity (NTU)	85				619	32	29
Conductivity (µS/cm)	664-1046				1585	1040-2720	400
Oil and Grease (Mg/L)	7						
BOD (mg/L)	37-69	59-149	50-100	95	590	280-690	167
COD (mg/L)	101-143	92-322	100-200	270	1270	700-980	
BOD/COD	0.36-0.48	0.46-0.64	0.50-	0.35	0.46	0.4-0.70	
TSS (mg/L)			1389		1396	85-286	
TDS (mg/L)			573			102	
TN (mg/L)	11-22	9.6	5-10			25-45	
TKN (mg/L)	9.5-14.3	0-8		11		0.1-0.5	
NH4-N (mg/L)	4.1-9.1			2.7	3.8	17-27	
NO3-N (mg/L)	0-1.8			0.24			
TP (mg/L)	0.45-1.5	0-7	0.2-0.6				
PO4-P (mg/L)	0.6-1.4				4.4		16
BOD/NH_4-N/PO_4-P	1/.06/.001-						
1/.13/.02				1/.006/.007			
Mg (mg/L)		0.11					
Ca (Mg/L)		0.13					
Na (mg/L)	32-35						
Cl (mg/L)	53						
FC(CFU)	$(1.2\text{-}3.6)10^3$		1-10		3.1×10^4	5×10^5	$1.5\text{-}1.6\times10^4$

Suspended solids concentrations in greywater depends on the amount of water used with the highest concentrations typically found in kitchen and laundry discharges. Studies on greywater showed suspended solids loads of 10–30 g/p/d [1]. However, high concentrations of 1389-1396 mg/L were reported [38,40].

Greywater contains biological oxygen demand (BOD), chemical oxygen demand (COD) and nutrients (phosphorous, sulfate, ammonium, sodium, and chloride) which are of typical concern when designing a biological treatment process. The biodegradable proportion of greywater (BOD/COD ratio) and the microbial nutrient available in greywater (COD/NH_4-N/PO_4-P ratio) can result in the deterioration of greywater and production of odor. The ratio of BOD_5/COD in greywater vary between 0.25 and 0.44 [3,37,49] and the average ratio of COD/NH_4-N/PO_4--P is 100/5/1 [28]. The nitrogen in greywater originates from ammonia-containing cleansing products as well as from proteins in meats and vegetables, protein-containing shampoos and other household products [43]. Typical values of nitrogen in mixed household greywater are within a range of 5–50 mg/L [22]. In countries where phosphorous-containing detergents have not been banned, dishwashing and laundry detergents are the main sources of phosphorous in greywater. Average phosphorous concentrations are typically found within the range of 4–14 mg/L in regions where non-phosphorous detergents are used [3]. However, they can be as high as 45–280 mg/L in households where phosphorous detergents are utilized [29,50].

Greywater usually contains some traces of excreta that come from bathing (washing the anal area in the bath and shower) or from the laundry (washing underwear and diapers). Table 4 shows the concentration of microbes reported in greywater [18,51-65]. However, the small traces of faces that enter the grey water stream via effluent from the shower or washing machine do not pose practical hazards under normal conditions [12,18]. Successful reproduction of pathogenic bacteria, fungus and protozoa occurs under warmer conditions where biodegradable matter is available as nutrient in greywater [46]. Greywater stored under warm temperature conditions for longer than 24 h is not recommended for safe use.

Table 4: Microbial contamination of greywater.

Microorganism	Concentration (counts/100 mL)	References
Total coliforms	From 1.2×10^3 To 8.2×10^8	Alsulaili et al. [51], Dwumfour-Asare et al. [52], Mandal et al. [53], Masi et al. [54] Oteng-Peprah et al. [55].
E. coli	Up to 6.5×10^6	Masi et al. [54], Oteng-Peprah et al. [55], Atanasova et al. [56]; Friedler et al. [57], Khalaphallah and Andres [58], Kim et al. [59], Paulo et al. [60].
Faecal coliforms	Up to 1.0×10^6	Halalsheh et al. [26], Mandal et al. [53].
Pseudomonas aeruginosa	Up to 1.4×10^4	Benami et al. [61], Khalaphallah and Andres [62].
Staphylococcus aureus	From 1.2×10^2 To 1.8×10^3	Kim et al. [59], Benami et al. [63], Maimon et al. [64], Shoults and Ashbolt [65].
Salmonella typhi	Up to 5.4×10^3	Kim et al. [59].
Salmonella spp.	Up to 3.1×10^3	Oteng-Peprah et al. [55].
Legionella pneumophila		Blanky et al. [18]

Greywater Biological Treatments

Degradation and transformation of greywater constituents are carried out by biochemical reactions occurring in the liquid medium by the microbial population in the biological treatment. The oxidation of organic compounds in greywater reduces BOD and nutrients (ammonia and phosphate). However, some constituents in greywater may only be partially degraded or not affected at all by biological processes because: (a) the compounds are non-biodegradable, (b) the absence of organisms required for the degradation process and (c) the presence of inhibitors in the medium. The existing biological treatment systems for greywater are: (a) constructed wet land, (b) sequencing batch reactor, (c) vertical flow bioreactor, (d) membrane bioreactor, (e) up-flow anaerobic sludge blanket, (f) rotating biological contractors, (g) trickling filters, (h) aerated lagoons, (i) anaerobic up-flow filter, and (j) expanded bed up-floe reactor.

Constructed Wetlands

Wetlands are considered a low-cost alternative for treatment of various wastewater streams [66,67]. Constructed wetlands are built specifically for water quality improvement purposes, typically involving controlled outflow and a design that maximizes certain treatment functions [67-69]. The increasing popularity of wetlands can be attributed to several benefits: (a) they can provide removal rates ranging from 60% to 95% for many pollutants, (b) they are much less costly to build and operate than conventional treatment facilities, (c) they provide important functions such as habitat enhancement, and (d) they are a less intrusive and provide more environmentally sensitive approach to pollution abatement [66,70,71].

Some of the disadvantages of wetlands are: (a) they generally require larger land areas than conventional wastewater treatment systems, (b) bioremediation and phytoremediation processes require more time than conventional treatments, (c) monitoring can sometimes prove difficult, and (d) the reliability of wetland treatment systems can be less consistent than that of traditional treatment systems due to the effect of weather [72]. However, there are many good examples of constructed wetland systems used for wastewater treatment. These systems have shown high removal rates of suspended solids and BOD (70-95%) and high fecal coliform and pathogen removal rates (80-99%). The final effluent quality has been generally proven to be safe for non-potable water reuse activities [23].

When designed properly, constructed wetlands are capable of effectively purifying greywater using the same processes carried out in natural wetland habitats by vegetation, soils, and their associated microbial assemblages, but do so within a more controlled environment [66,73]. The specific water quality treatment mechanisms include gravitational settling of suspended matter, facilitation of chemical transformations, and facilitation of bioremediation and phytoremediation processes [71,74]. The application of this type of technology is not a new, as constructed wetlands were frequently utilized for the purpose of pollution abatement by ancient Chinese and Egyptian cultures [75]. European experimentation with phytoremediation techniques began in the early 1950s. The use of treatment wetlands subsequently began to grow, and treatment wetlands can now be found in every continent [70,71,74].

There are two types of constructed wetlands: (a) free water surface (FWS) wetlands and (b) subsurface flow (SSF) wetlands. Both systems consist of a series of cells lined with impermeable material (clay or plastic liners) to limit potential groundwater infiltration. Both support additional substrates composed of soils which are established with some aquatic vegetation such as cattails and reeds (Typha spp. and Phragmites spp.) or water hyacinth (Eichhornia crassipes). Wastewaters enter the wetlands via simple gravitational forces or can be more stringently directed and controlled by pumping mechanisms [76].

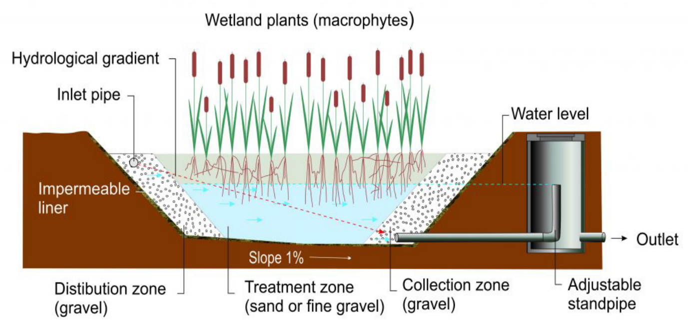
Free Water Surface Wetlands

Free water surface (FWS) wetlands are treatment wetlands in which the surface water flowing through them is exposed to the atmosphere. They consist of several basins or cells with the water surface being 0.15 to 2.00 m above the bottom [70]. They appear much like natural marshes, containing emergent aquatic vegetation. In FWS wetlands, the near-surface layer is aerobic while the deeper waters and substrate are anaerobic. These systems are primarily constructed to treat municipal wastewaters, greywater, mine drainage, urban storm water, agricultural runoff and livestock wastes, and landfill leachate [77]. Free water surface wetlands can be further sub- classified according to their dominant type of vegetation into emergent macrophyte wetlands, free-floating macrophyte wetlands, and submerged macrophyte wetlands. The most common system is the emergent macrophyte based system (Figure 1). These treatment wetlands support a considerable sediment layer above their impervious liners in which emergent macrophytes, such as cattails (Typha spp.), rushes (Juncus spp.) and bulrushes (Scirpus spp.), are planted. Suspended solids are removed by gravitational settling. Nutrients and pollutants absorbed to the settled sediments are then exposed to aerobic rhizome areas created by the macrophytes [71].

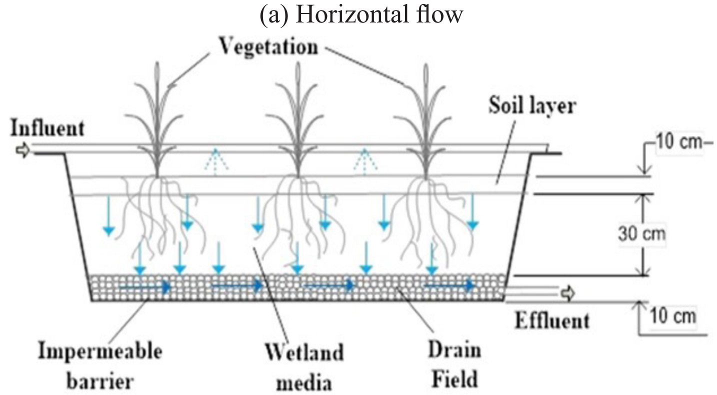
Free floating macrophyte based wetlands (Figure 2) use floating plants such as duckweed (Lemna spp.) and water hyacinth (Eichhornia crassipes) to remove nutrients and other pollutants in wastewater. A floating barrier grid is used to support the growth of floating macrophytes and to reduce wind effects, which would otherwise cause the plants to drift. These systems have the added benefit of being able to mitigate algae growth, a persistent problem in treatment wetlands brought on by the nutrient rich nature of the wastewaters received. In free floating systems, the densely packed floating plants work to block out sunlight, thereby preventing photosynthesis and inhibiting algae growth [70,71].

Little information is available on submerged macrophyte based FWS wetlands (Figure 3) as they are still in the experimental stage. But as their name suggests, these systems would rely on the use of submerged macrophytes such as pondweed (Potamogeton spp.) to remove nutrients and other pollutants from received wastewaters [70,75].

FWS wetlands have many advantages including: (a) simple construction process and lower operating costs, (b) little requirements for mechanical equipment, energy, and skilled operator, and (c) superiority in their abilities to remove BOD, COD, and total suspended solids organics and fecal coliforms. The main disadvantages of FWS wetlands are: (a) they require a larger land area than other systems, (b) the wastewaters are exposed and are therefore accessible to humans and animals, hence it may not prove prudent to establish these wetlands in high-use areas such as parks, playgrounds, or similar public facilities, (c) pollutants such as phosphorus, metals, and some persistent organics can become bound in wetland sediments and accumulate over time, (d) the open water environments of FWS wetlands can attract unwanted pests such as mosquitoes which may ultimately need control and (e) routine harvest and removal of wetland vegetation is typically unnecessary [71,76,77].



(a) Horizontal flow



(b) Vertical flow

Figure 1: Surface flow emergent macrophyte constructed wetland [70].

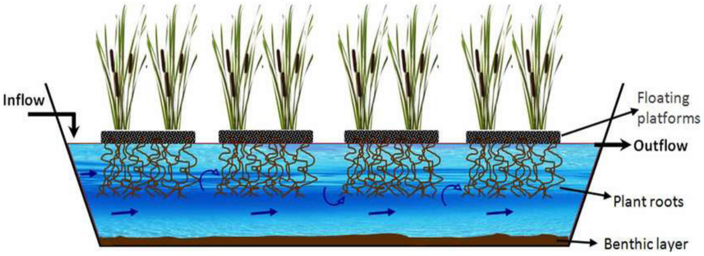


Figure 2: Free water surface floating macrophyte wetland [73].

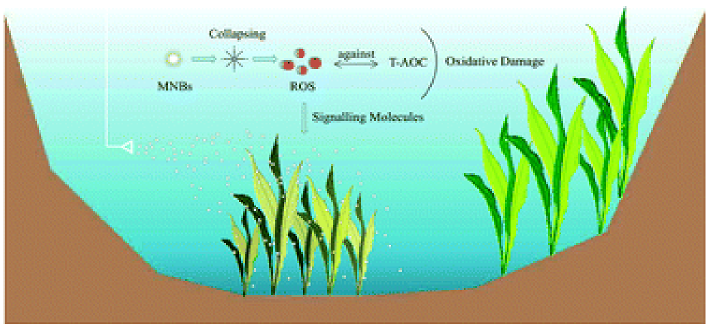


Figure 3: Free water surface submerged macrophyte wetland [79]

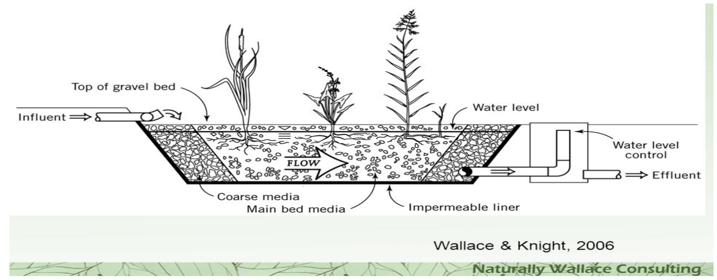


Figure 4: Horizontal subsurface flow constructed wetland [119].

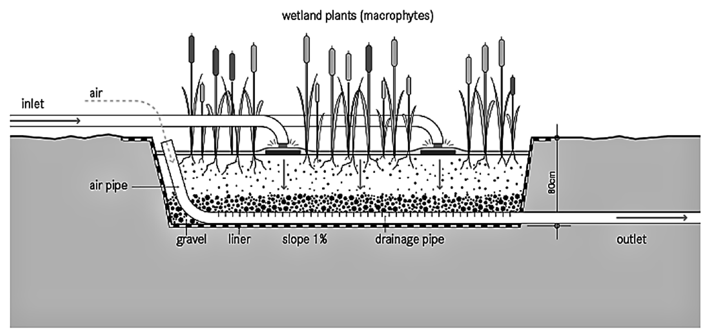


Figure 5: Vertical subsurface flow constructed wetland [119].

Sub-Surface Flow Wetlands

A sub-surface flow (SSF) wetland consists of a sealed basin with a porous substrate of rock, gravel or coarse sand planted with emergent macrophytes such as reeds (*Phragmites* spp.), Eurasian watermilfoil (*Myriophyllum spicatum*), and duckweeds (*Lemna* spp.). The depth of the substrate ranges from 0.3 to 0.9 m. The water level is designed to remain below the top of the substrate allowing the same mechanisms as FWS wetlands to remove contaminants. SSF wetlands are commonly used to treat wastewaters from small-scale sources such as individual homes, schools, apartment complexes, commercial establishments, parks, and other recreational facilities [76].

SSF wetlands can be sub-classified according to their flow patterns to horizontal flow and vertical flow. Horizontal flow SSF wetlands (Figure 4) involve the continuous, horizontal flow of wastewaters through the medium. Oxygen is transferred into the system via atmospheric diffusion through the emergent aquatic plants. In vertical flow SSF wetlands (Figure 5), wastewater is added at timed intervals, and the system drains between dosing. Vertical flow SSF wetlands tend to be less anoxic than horizontal flow wetlands as oxygen diffuses easily from the atmosphere into the drained, porous substrates. However, horizontal flow systems remain by far the more commonly used and documented SSF systems [70].

The major advantages of SSF wetlands are: (a) the rocky substrates provide greater surface area for microbial reactions and therefore SSF wetlands can be smaller in size yet treat larger flow volumes than FWS wetlands, (b) they are often better suited to projects in which the available land area is limited, (c) they are typically more suitable to public areas as contaminated wastewaters are not exposed, and (d) the nature of their substrates and flow regimes allow for better thermal protection and are, therefore, considered to be more effective in colder climates than FWS systems [78].

The disadvantages of SSF wetlands are (a): they are more expensive to construct, maintain and repair, (b) they have problems with clogging and as a result are best suited to wastewaters with low solids concentrations and under uniform flow conditions, (c) as vegetation is the prominent removal mechanism of pollutants in these systems, plants can reach their points of saturation in terms of pollutant absorption, rendering them no longer effective thus requiring costly and time consuming harvesting, (d) they tend to be anoxic, which limits the biological removal of ammonia nitrogen via nitrification unless costly aerators are adopted to mitigate this problem, (e) phosphorus removal rates are inferior to that of FWS systems, (f) they can have problems associated with the accumulation of pollutants in sediments over time, and (g) they provide less habitat value than FWS [70,71,80].

Wetland Performance

These wetland systems can be configured differently to suite different needs. Large spaces may be required to accommodate the surface or subsurface flow design to mimic the natural wetland ecosystem. Although, performance of each wetland differs with design, the concept of treatment is similar. An influent pipe diverts water through the wetland and is extracted by gravity drainage into a collection basin or perforated pipe. These systems have a design slope of approximately 1% using a synthetic or clay lining for retaining water in the system [66-68].

Vymazal [81] reported that the constructed wetlands have evolved into a reliable wastewater treatment technology for various types of wastewaters in Germany. Constructed wetlands require very low energy input and, therefore, the operation and maintenance costs are much lower compared to conventional treatment systems. All types of constructed wetlands are very effective in removing organics and suspended solids whereas removals of nitrogen and phosphorous are lower.

Ayaz [82] conducted a study in Turkey to treat tertiary effluent from a wastewater treatment plant using three forms of constructed wetlands: horizontal subsurface fellow (HSSF), free water surface flow (FSF) and surface flow (SF). The systems dimensions were 6.5-7.5 m wide and 20-25 m long. The flow rate into the systems varied from 4.8 to 15.6 m3/d. Samples were taken on a weekly basis during winter and summer months over a period of 3 years and tested for biological oxygen demand (BOD), total organic carbon (TOC), suspended solids (SS), total coliform (TC) and fecal coliform (FC). The wastewater characteristics are sown in Table 5 and the removal efficiencies of the systems are shown in Table 6. All systems had a significant removal rate of BOD, COD, TOC and fecal coliforms. The BOD and COD removal efficiencies were affected by the temperature and the systems removal capabilities were reduced in winter by 25% and 5%, respectively. However, removal efficiency of TOC and coliforms was influenced by the hydraulic retention time (HRT).

Maine [83] conducted a study in Argentina to treat effluent from a tool factory in a free water surface wetland. The system was 50 m long, 30 m wide and 0.6-0.8 m deep and the wastewater flow was in the range of 25-75 m3/d. The wastewater had high concentrations of the heavy metals (Table 7). The study was focused on the efficiency of various plant species to treat heavy metals and concluded that all species used had similar high removal efficiencies of heavy metals. The horizontal flow wetlands have high removal rates of fecal coliforms, BOD, COD, TOC and iron but showed lower removal rates for Mg, K and N-NH4.

Scheumann et al [36]. reported on a small camping site "La Cava" in Arezzo, Italy that was established according to the sustainable water management principles (water saving, reuse, recycling) which was only open during summer months (July-September). The black and grey waters were segregated and treated by a constructed wetland. The treated greywater was recycled for toilet flushing whereas the treated black water was reused for landscaping. The camping complex covered a surface area of about 20,000 m^2 with wood, green terraces, and parking places for a total of 25 cars. The wastewater had to be treated onsite because a sewer connection was located 6 km away. Wastewater was collected by a gravity system and weekly production fluctuated within the range of 0.3-7.0 m^3/d. The greywater was treated in a horizontal fellow wetland at a hydraulic loading rate (HLR) of 8.26 cm/d with a flow of 9.5 m^3/d (passing through a cell of a surface area of 115 m^2). The treated greywater quality complied with the Regional Environmental Protection Agency of Tuscany (TSS of 10 mg/L, COD of 100 mg/L, BOD of 20 mg/L, NH4-N of 2 mg/L, TN of 15 mg/L, TP of 2 mg/L, E. coli of 50 cfu/100 mL).

Table 5: Influent water quality characteristics [82].

Parameter	Average value
pH	7.5
Temperature (^oC)	15
Electric Conductivity (µS/cm)	883
BOD (mg/L)	11
COD (mg/L)	33
TOC (mg/L)	110
SS (mg/L)	15
DO (mg/L)	3.14
Total Coliform (cfu/100 ml)	20,800
Fecal Coliform (cfu/100 ml)	2300

Table 6: Removal efficiencies of three wetlands [82]

Parameter	Season	Influent Concentration	Effluent Concentration (% Removal)		
			HSSF	FWS	SF
BOD (mg/L)	Winter Summer	13 9	5.4 (51) 2.1(77)	5.3 (59) 2.9 (68)	5.4 (58) 3.2 (65)
COD (mg/L)	Winter Summer	36 32	22.5 (38) 13.2 (58)	23.4 (35) 15.2 (52)	20.0 (44) 16.1 (49)
TOC (mg/L)	Winter Summer	10 12	7.3 (28) 8.2 (30)	6.6 (35) 7.9 (33)	6.2 (38) 7.5 (37)
T. Coliform	Winter Summer	29765 15550	1652 (94) 686 (96)	1293 (96) 1021 (93)	2271 (92) 1113 (93)
F. Coliform (cfu/100ml)	Winter Summer	2659 2077	128 (95) 126 (94)	131 (96) 113 (95)	226 (91) 137 (93)

HSSF= horizontal subsurface fellow
FWS= free water surface flow
SF= surface flow

Table 7: Results from wetland remediation of tool factory effluent in Argentina.[83].

Parameter	Influent Loading (g/d)	Concentration		Effluent Loading (g/d)	Concentration		Removal (%)
		Mean	Range		Mean	Range	
Temperature	-	19.1	10.2-28	-	16.7	6.6-26	-
pH	-	9.01	6.5-12.3	-	7.66	6.9-9.1	-
Conductivity	-	3259	480-8500	-	1799	470-5000	-
Alkalinity	18.5	369.1	71.2-1187	14.0	280.1	95.2-475	33
DO (mg/L)	-	1.53	0-7.1	-	0.898	0-7.5	-
SS (mg/L)	137.9	2758.4	699-8550	72.5	1450.9	524-3693	36
Ca (mg/L)	7.90	157.9	24-651	3.33	66.7	17.3-268	34
Mg (mg/L)	0.81	16.2	0.5-59	0.77	15.5	3.3-62	5
SO_4 (mg/L)	62.4	1257.1	98.1-3598	30.5	609.1	158-2238	34
Cl (mg/L)	13.4	268.4	70.4-778	7.70	154.0	38.6-320	34
Na (mg/L)	35.4	708.0	200.2-1680	20.3	405.4	135-1136	34
K (mg/L)	0.878	17.6	7.3-38	0.854	17.1	2.4-39	5
Fe (mg/L)	0.387	7.73	0.05-73.9	0.012	0.237	0.05-0.43	74
Cr (mg/L)	0.0009	0.018	0.001-0.164	0.0002	0.004	0.001-0.015	53
Ni (mg/L)	0.001	0.028	0.002-0.2	0.0006	0.013	0.003-0.10	39
N-NO_2 (mg/L)	0.014	0.285	0.001-1.6	0.001	0.016	0.001-0.30	75
N-NO_3 (mg/L)	0.225	4.53	0.018-16	0.044	0.877	0.07-7.0	68
BOD (mg/L)	3.53	70.7	6.5-360	1.07	21.4	5.0-83.4	66
COD (mg/L)	10.6	211.7	21.8-1082	2.87	57.5	11.2-172.5	72

Madera-Parra [84] used a pilot scale constructed wetland for treatment of landfill leachate which was planted with polycultures of tropical species *Gynerium sagittatum, Colocasia esculenta and Heliconia psittacorum*) sand operated for 7 months at continuous gravity flow of 0.5 m3/d. Three wetlands were divided into three sections (each section was 5.98 m2 and was seeded with 36 cuttings of each species). All cells received pre-treated landfill leachate from a high-rate anaerobic pond operating as primary treatment system. Influent and effluent from each cell were analyzed for COD and heavy metal (Cd, Pb and Hg). Flowering, stem length, chlorophyll and photosynthetic rates in plants were measured. The removal efficiencies varied from 60 to 90% for all parameters, indicating that plant distribution may affect the removal capacity of cells. All plants presented a good physiological response and constant growth.

Rodriguez-Dominguez et al [85]. reported on the state-of-the-art constructed wetlands in the Latin America and Caribbean Region with the aim of bringing updated and sufficient information to facilitate their use for wastewater treatment. They extracted 520 experiences from reviewing 169 documents from 20 countries. About 114 different plant species were used in these wetlands. The data showed that horizontal subsurface flow wetlands were the most reported constructed wetlands in the region (62%), followed by free water surface constructed wetland (17%), vertical flow wetland (9%), intensified constructed wetlands (8%), and finally French wetlands (4%). The COD, total nitrogen, and total phosphorous removal efficiencies varies from 65-83%, 55-72%, and 30-84%, respectively.

Zidan et al [86]. reported on a horizontal subsurface flow constructed wetlands for wastewater treatment in Egypt having three different treatment media (gravel, pieces of plastic pipes, and shredded tire rubber chips). They investigated the change in media porosity of the wetland cells and their effect on the BOD, COD and TSS removal efficiencies. The results showed that after 180 days of operation, the wetland cells had reached steady porosity. The performance of plastic media bed in pollutants reduction was better than gravel and rubber beds and the gravel media was better than the rubber media. Through 218 days from start of operation, the porosity decreases by 16.94% for gravel media, 12.33% for rubber media, and by 9.01% for plastic media. The reduction in porosities was related to the development of reeds roots, growth of biofilm on the bed media surfaces and accumulation of suspended matter. Reductions in TSS were 39–61% while the reductions in the BOD and COD were 20–49% and 19–49%, respectively.

Xu et al [87]. stated that plants play an essential role in methane (CH4) production, transport and release processes in constructed wetlands. In their study, they used plant presence, species richness, plant species-specificity, and harvesting activity information from papers published during the period of 1993-2018 to elucidate the key factors that drive CH_4 emission from constructed wetlands treating wastewater. They noticed that the use of a single plant species not only changed the production and consumption of CH_4 by affecting the functioning of roots but also influenced the process of CH_4 entering the atmosphere under different transport capacities. The CH_4 flux reached 1.0686 g CH_4 /m^2 d from the *Zizania latifolia* system, which is eight times larger than that of the Phalaris arundinacea system. The mixed systems exhibited a positive increase in CH_4 flux with plant species richness due to the complementary effects of the root exudates excreted from different plants. The minimum CH_4 value (0.0084 g CH_4 /m^2 d) was observed in the three-species system (*Oenanthe javanica, Phalaris arundinacea and J. effusus*). These results demonstrate that selecting several species with lower methane fluxes such as *Typha latifolia and C. papyrus* and suitably regulating harvesting in constructed wetlands can be more effective for mitigating the potential of CH_4 emissions while maintaining the efficiency of wastewater purification

Sequencing Batch Reactor

Sequencing batch reactors (SBRs) are one of the preferred technologies for the treatment of greywater for small scale operation due to the low cost and simple operation. Figures 6 shows an SBR setup for treating greywater [88]. The SBR system offers a great operational flexibility for effective nutrient removal. The SBR is a cyclic "fill and draw" system that performs equalization, biological treatment, and secondary clarification in a single tank using a time control sequence. This can be advantageous over other biological reactors that use activated sludge since these systems require separate compartments [89].

The biological treatment in SBRs is achieved by the microorganisms in biologically activated sludge. When wastewater is mixed with a suspension of these microbes, they assimilate pollutants, degrade the biological portion, and then the rest of material settles, at which point it can be separated from the effluent. The SBR systems not only provide compact treatment, but also allow an easy-to-use interface for consumers because the reaction time, retention time, and mixing rate are simplified through computer programming. The required environmental conditions are dependent on the quality of the wastewater influent [90].

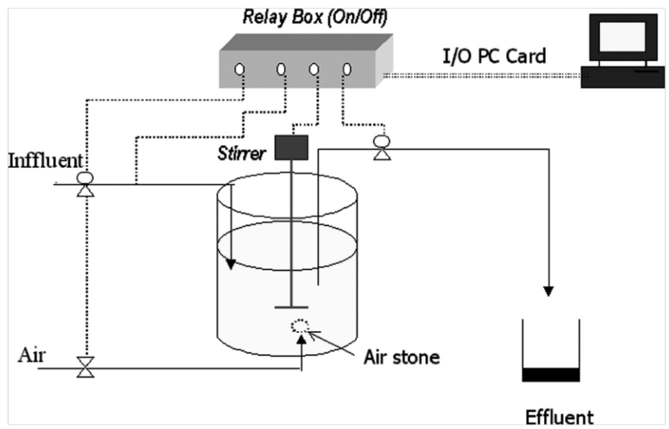


Figure 6: Small Scale SBR and monitoring sys

There are three types of living media in SBR: (a) anaerobic, where organic matter is mineralized into biogas (methane and carbon dioxide) in the absence of oxygen, (b) anoxic, where nitrate is used as the oxidation reagent to produce free nitrogen and other compounds through denitrification and (c) aerobic, where dissolved oxygen is used for oxidation of the carbonaceous material and nitrification. Depending on the living mediums required, these systems could be adapted and modified to suit treatment requirements [91].

The performance of the BSR is influenced by reaction time, retention time, rate of mixing and flow rate in the system. The SBR systems require proper start up times that include inoculating the system with activated sludge and sustaining environment conditions for microbial growth. Once the system is configured, it can produce effluent suitable for non-potable reuse applications [88,89,91].

Lamine et al [88]. conducted a study on greywater treatment using SBR in a student house in Tunisia. The system used both anoxic and aerobic mediums to reduce the COD and BOD of the wastewater. The performance of the treatment process was assessed at various hydraulic retention times (HRTs). Tables 8 and 9 show the raw water characteristics and results of the study. Removal rates of BOD and COD were greater than 90%.

Jamrah et al [92]. Used SBR to treat greywater collected from various households in the area surrounding the University of Jordan in Aman, Jordan. The study focused on varying operating times and conditions as shown in Table 10. The optimal operating time was 6 h with a fill and react time of 2-3 h. The COD removal was over 90 %.

Scheumann and Kraume [93] used a pilot scale SBR in Germany to treat greywater at varying retention times. The removals of COD, NH_4-N and TN were sufficient to meet discharge reuse guidelines. The COD was reduced from 250 to 18.9 mg/L, the NH_4-N was reduced from 11.9 to 4.1 mg/L and the TN was reduced from 17.1 to 0.37 mg/L, all being below the mandatory values for reuse applications.

Table 8: SBR influent characteristics [88].

Parameter	Samples	Minimum	Maximum	Average	SD
pH	11	7.5	7.9	7.6	0.4
TTS (mg/L)	16	23.0	50.0	33.0	16.0
COD (mg/L)	23	25.0	300.0	102.0	86.0
BOD (mg/L)	11	15.0	140.0	97.0	56.0
TOC (mg/L)	13	12.0	67.0	32.6	32.0
NH_4-N (mg/L)	14	1.2	15.2	6.7	5.6
NO_2-N (mg/L)	14	0.0	0.2	0.0	0.2
NO_3-N (mg/L)	14	0.0	1.2	0.2	0.1
PO_4-P (mg/L)	14	2.8	11.3	3.5	4.8
TKN (mg/L)	10	4.2	20.0	8.1	3.7

Table 9: SBR effluent characteristics [88].

Parameter (mg/L)	HRT=0.6 d			HRT=2.5 d		
	Maximum	Average	SD	Maximum	Average	SD
TTS	38.0	23,0	16.0	40.0	23.0	16.0
COD	25.0	12.0	10.0	38.0	20.0	16.0
BOD	15.0	7.0	6.0	16.0	7.0	6.0
NH4-N	10.3	6.2	4.8	0.9	0.3	0.6
NO2-N	0.2	0.1	0,1	0.1	0.05	0.1
NO3-N	6.2	5.4	6.5	16	10	5.6
PO4-P	17.6	8.7	7.8	5.9	4.9	2.3

Table 10: Variable parameters for SBR and design considerations [92].

Total Cycle Time (h)	6.0	6.0	6.0	6.0	6.0
Fill and React Time (h)	5.0	5.0	5.0	5.0	5.0
Fill to React Time (h)	0.5:4.5	1:4	1.5:3.5	2:3	2.5:2.5
Anoxic Fill (h)	0.25	0.5	0.5	0.5	0.5
Aerated Fill (h)	0.25	0.5	1.0	1.5	2.0
React (h)	4.5	4.0	3.5	3.0	2.5
Settle (h)	0.75	0.75	0.75	0.75	0.75
Draw and Idle (h)	0.25	0.25	0.25	0.25	0.25
Maximum Reactor Volume (L)	20.0	20.0	20.0	20.0	20.0
Fill/Withdraw Volume (L/cycle)	6.0	6.0	6.0	6.0	6.0
SBR Volume Exchange Ratio (%)	30.0	30.0	30.0	30.0	30.0
Biomass Seeding (%)	35.0	35.0	35.0	35.0	30.0

Krishnan et al [94]. investigated the performance of greywater treatment from residential houses in Malaysia using an aerobic SBR at fixed hydraulic retention time of 36 h. They explored the effectiveness of the SBR in treating nutrient-deficit and nutrient-spiked dark greywater for agricultural reuse. The dark greywater had a COD:N:P ratio of 100:1.82:0.76, while the preferred ratio for biological oxidation is 100:5:1. The aerobic oxidation of nutrient-deficit and nutrient-spiked dark greywater with a COD:N:P ratio of 100:2.5:0.5, 100:3.5:0.75 and 100:5:1 resulted in outlet COD values of 64, 35, 15 and 12 mg/ L, with a corresponding BOD value of 37, 22, 10 and 8 mg/ L which complied with the Malaysian Discharge Standards for Agricultural Activities.

Hernandez Leal et al [95]. compared an aerobic SBR with an up-flow anaerobic sludge blanket reactor and a combined anaerobic-aerobic treatment (up-flow anaerobic sludge blanket reactor + sequencing batch reactor) in treating greywater at hydraulic retention times of 12–13 hours. The aerobic conditions resulted in a COD removal of 90%, which was significantly higher than the 51% removal by anaerobic treatment. The low removal in the anaerobic sludge blanket reactor may have been caused by high concentration of anionic surfactants in the influent (43.5 mg/L) and the poor removal of the colloidal fraction of the COD. The combined aerobic-anaerobic treatment achieved a COD removal of 89% which is very close to that of the aerobic treatment.

Vertical Flow Bioreactors

Vertical flow bioreactors (VFBR) use similar concepts to horizontal flow constructed wetlands. Wastewater enters through an influent source and is subjected to treatment in the system. Typical systems use two basins stacked vertically (Figure 7), one acts as the working mechanism while the other retains wastewater and operates as a retention basin. The first container is comprised of various layers of organic soils, plastic media and limestone pebbles. Water passes through these various layers and the contaminants are filtered out. Holes are evenly spaced along the bottom of the container and allow water to drain into a drainage basin. Water can then be sent to the distribution system. The VFBR systems can vary in size depending on the flow rate [96].

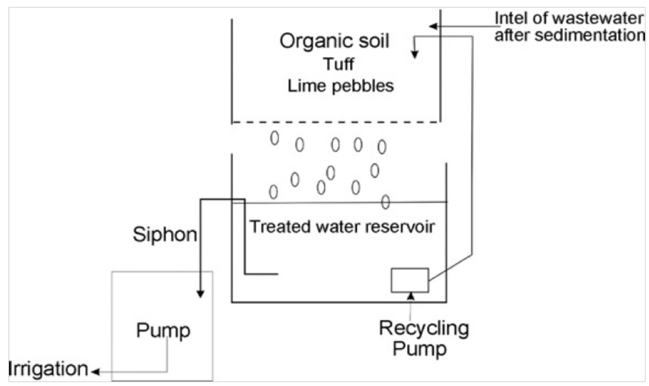


Figure 7: Wastewater flow through VFBR reactor [96].

Gross et al [96]. reported on a VFBR system that treated 213 m3/ year of greywater from various households. The dimensions of the system were 1 m long by 1 m wide and 0.5 m in depth. The system was comprised of a three-layer bed: (a) the first layer consisting of 15 cm planted organic soil, (b) the second layer consisting of 30 cm of plastic media and (c) the third layer consisting of 5 cm of limestone pebbles. The greywater entered through the root structure of the system and then passed through the medium of evenly spaced holes to the reservoir below. This system had a centrifugal pump that recycled water from the reservoir to the VFBR to be retreated. The system produced high removal rates of contaminants as shown in Table 11.

Kanawade [97] developed a modular system for recycled RVFB for the removal of contaminants from synthetic greywater, enriched with wastes from a dining hall. The greywater was recirculated for 3 days, after which time half of the greywater was removed from the system and replaced with fresh greywater. The RVFB system reduced the effluent concentrations of NO_3-N, NH_4-N, NO_2-N, TSS, boron, and anionic surfactants to below the levels considered acceptable for either recreation or irrigation.

Al-Zubi et al [98]. treated ablution greywater in a low cost and easy to operate modified RVFB for use in landscape irrigation in Al-Balqa' Applied University in Amman, Jordon. The treatment system adequately removed BOD_5, COD, TSS, chloride, and Na by 94, 88, 90, 48, and 33%, respectively. Concentrations of Mg, Ca, and K were increased by up to 29, 63, and 95%, respectively. Nitrate concentration increased but remained less than the maximum allowable limit. Electrical conductivity, total dissolved solids, and SO_4 were much less than the maximum allowable limits of the Jordanian Guidelines. The treated greywater was suitable for irrigation of ornamentals, fruit trees, and fodder crops according to the WHO Guidelines.

Ammari et al [99]. evaluated a modified RVFB for greywater treatment under arid conditions in Jordon. The BOD5, COD, PO4, TSS, NO3, Cl, and SO4 removal efficiencies were 97%, 94%, 100%, 90%, 45%, and 55%, respectively. Total coliform and Escherichia coli were reduced by 2.5 and 2.3 log, respectively. Treated greywater was suitable for irrigation of ornamentals, fruit trees and fodder crops. The treatment system demonstrates great potential for treating low quality greywater in rural areas.

Table 11:Containment's reductions in wastewater treated in VSBR [96].

Parameter (mg/L)	Initial (mg/L)	Final (mg/L)	Reduction (%)
COD	339	46.6	86.25
Anionic Surfactant	12.3	0.2	98.37
TSS	46.o	3.0	93.48
NO3-N	3.5	1.8	48.57
NH4-N	1.2	1.0	16.67
NO2-N	1.3	0.04	96.92
Boron	0.1	0.15	(+50.00)
Total P	1.9	0.5	73.68

Membrane Bioreactor

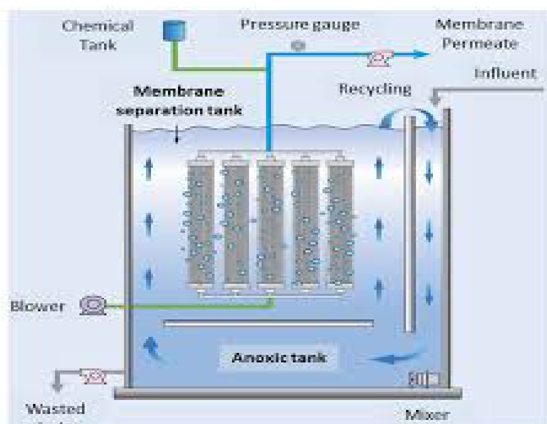
A membrane bioreactor (MBR) is a combination of biological, microfiltration and ultrafiltration systems. It is an appropriate solution for greywater treatment in densely urbanized areas (where space has high value) due to its compact size. The MBR can be operated under aerobic or anaerobic conditions as shown in Figure 8.

Atanasova et al [100]. studied the performance of an MBR treating greywater from a hotel in Spain. The COD removal efficiency ranged from 80 to 95%, and the COD concentration in the effluent was below the quantification limit of 30 mg/L based on the Spanish Legislation for Water Reuse. Ammonia and TN removal were 80.5 and 85.1%, respectively.

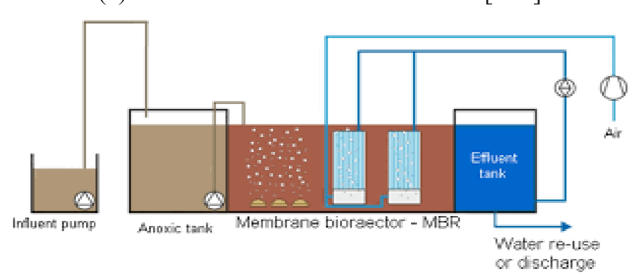
Chae et al [101]. Investigated the characteristics of membrane fouling in a laboratory scale A/O (anoxic/oxic) series MBR treating synthetic wastewater. The high concentrations of extracellular polymeric substances, high viscosity and a high sludge volume index corresponded to high membrane resistance indicating severe membrane fouling in the MBRs. As hydraulic retention time decreased from 10 to 4 h, the concentrations of extracellular polymeric substances increased and the average particle size increased, leading to reduced settling of the sludge and increased membrane fouling. It was found that air backwashing was more efficient for fouling mitigation than was air scouring.

Merz et al [103]. evaluated the performance A 3L-laborastory scale MBR treating shower effluent from a sports club in Rabat, Morocco. The MBR was operated with a hollow fibre membrane for 137 consecutive days and the removal performance and membrane behaviour were assessed. The permeate was of excellent aesthetic quality, free from odours and complied with the Standards for Domestic Reuse except for bacterial contamination. Non-detectable levels of faecal coliform could not be continuously guaranteed due to bacterial re-growth in the pipe from the open permeate storage tank. To always guarantee zero faecal coliform levels, disinfection of the permeate was necessary.

Huelgas and Funamizu [104] treated greywater using a laboratory scale MBR under varying pressure. A 10 L lab-scale submerged membrane bioreactor (subMBR) was operated with a flat-plate membrane for 87 days using a mixture of washing machine and kitchen sink wastewater at a constant flux of 0.22 m^3/m^2 d and an HRT of 13.6 h. permeate was intermittently withdrawn at constant transmembrane pressure induced by water level difference and without pump requirement. The COD removal was 96% and a Permeate COD of 26 mg/L was obtained. The total linear alkylbenzene sulfonate removal was > 99%, indicative of its non-inhibited degradation even at influent concentration of 30.8 mg/L.



(a) Anaerobic membrane bioreactor [101].



(b) Aerobic membrane bioreactor [102].

Figure 8: Membrane bioreactors for removal of organic matter

and nutrients from wastewater

Jong et al [105]. used a laboratory scale anaerobic-anoxic-oxic MBR to treat greywater in Korea. A submerged MF (0.45 µm pore size membrane) was installed in the reactor to maintain activated sludge biomass. An analysis of pathogenic microorganism and microbial communities in treated graywater was performed. Pathogenic microorganisms *Escherichia coli, Coliform, Staphylococcus aureus and Salmonella* were detected in the effluent. These systems could achieve very good effluent that meets Regulatory Standards for Reuse. However, the MF membrane in the MBR system could not perfectly remove microorganisms.

Friedlee et al [39]. investigated the reliability of an on-site MBR system for greywater treatment and reuse using IWA ASM1 simulation model to describe biological and physical mechanisms for greywater treatment by the MBR. Model results were found to agree well with experimental data from a greywater treatment pilot plant. Then, the calibrated model was used in a Monte Carlo mode for generating statistical data on the MBR system performance under different scenarios of failures and inflow loads variations. The reliability analysis considered four major types of hardware failures: aeration system circulation pump, power and membrane texture. The effect of excess application of cleaning agents was studied too. Membrane texture failure was found to have the most significant negative effect on effluent quality since mixed liquor which leaks through the membrane is mixed with the permeate resulting in very high effluent COD, BOD and TSS values. This failure also resulted in significant washout of biomass from the aeration basin. Excess application of cleaning agent had the second most severe effect on effluent quality as significant proportion of the biomass was lost due to enhanced decay. When a failure in the circulation pump and a power cut off occurred no effluent was produced, and biomass was washed out from the aeration basin. Effluent TSS, COD and BOD concentrations were found to be quite insensitive to influent quality while effluent nitrogen species (TKN, NO3, NO2 and NH4) were found to be more sensitive. Oxygen concentration in the aeration basin was found to have medium sensitivity to these fluctuations.

Jabornig [106] investigated two processes of recycling greywater (shower, washbasin, washing machine) for reuse in households. The first process was a combination of biofilm carrier material with downstream membrane filtration while the second does not use any carrier material, but a hollow fibre membrane with high specific surface was used as combined growth area for biofilm and membrane filtration. In both cases, the aim was to reach the quality requested in International Guidelines for Greywater Reuse. The investigation focus was on the reduction of investment, operation cost and maintenance cost. The results showed that it was possible to reduce the power consumption of both small plants to less than 1.5 kWh/m³ treated water at constantly high effluent quality.

Up-flow Anaerobic Sludge Blanket

The up-flow anaerobic sludge blanket (UASB) is one of the most widely used wastewater treatment system for various types of wastewaters (Figure 9). It works on an anaerobic process, retains a high concentration of active suspended biomass and produces better settleable sludge than other treatment systems [107].

Bal and Dhagat [108] stated that the key to the UASB process is the understanding that anaerobic sludge inherently has superior flocculation and settling characteristics which allows for a high solid retention time at high HRT with separation of the gas from the sludge solids. The UASB process is a combination of physical and biological processes. The main feature of physical process is separation of solids and gases from the liquid and the feature of biological process is degradation of decomposable organic matter under anaerobic conditions. No separate settler with sludge return pump is required as is the case in activated sludge bioreactor. There is no loss of reactor volume through filter or carrier material, as the case with the anaerobic filter and fixed film reactor types. There is no need for high-rate effluent recirculation and concomitant pumping energy, as in the case with fluidized bed reactor. Anaerobic sludge inherently possesses good settling properties and for this reason mechanical mixing is omitted in UASB-reactors. At high organic loading rates, the biogas production guarantees sufficient contact between substrate and biomass and as a result UASB reactor approaches the completely mixed reactor.

Hemandez et al. [95] treated greywater from 32 houses in the Netherlands using UASB system and compared it to an aerobic sequencing batch reactor (SBR) at a hydraulic retention time of approximately 12–13 h. Aerobic conditions resulted in a COD removal of 90%, which was significantly higher than 51% removal by anaerobic treatment. The low removal in the anaerobic reactor was caused by high concentration of anionic surfactants in the influent (43.5 mg/L) and a poor removal of the colloidal fraction of the COD in the up-flow anaerobic sludge blanket reactors.

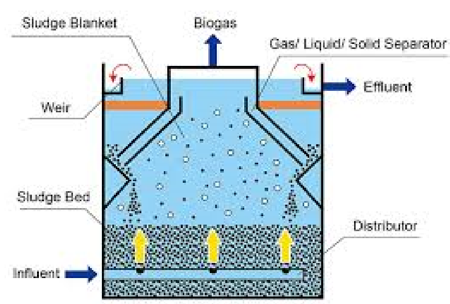


Figure 9: An up-flow anaerobic sludge blanket [107].

Elmitwalli et al [109]. Used a UASB system in Lubeck Germany for the treatment of greywater at varying retention times (8, 12 and 20 h) and ambient temperatures (14-24 oC). A COD removal of 31-41% was achieved which was significantly higher than that achieved by a septic tank (the most common system for grey water pre-treatment) of 11-14% at HRTs of 2-3 days. The relatively lower removal of total COD in the UASB reactor was mainly due to a higher level of colloidal COD in the greywater. The UASB reactor removed 24-36% and 10-24% of total nitrogen and total phosphorus in the greywater, respectively. The sludge characteristics of the UASB reactor showed that the system had stable performance at an HRT of 12 h.

Abdel-Shafy et al [110]. evaluated the efficiency of a UASB treating greywater for unrestricted use in Egypt. The raw greywater had average concentrations of 95, 392, 298, 10.45, 0.4, 118.5 and 28 mg/L for TSS, COD, BOD_5, TP, nitrates, oil and grease and TKN, respectively. After treatment, the effluent had average concentrations of 76.65, 165.4 96.85 and 19.31 mg/L for TSS, COD, BOD5 and oil and grease respectively. This represents removal efficiencies of 19.3% for TSS, 57.8% for COD, 67.5% for BOD5 and 83% for oil and grease. The characteristics of the treated effluent complied with the Egyptian Guidelines for Unrestricted Water Reuse.

Laguna et al [111]. evaluated a granulometry procedure based on manual humid sieving for the determination of the particle size distribution of UASB sludge. No solid loss occurred during the screening and the particle size profiles were reproducible when performed with sludge samples of 5, 10, 25 and 150 ml. Sieving could be performed on sludge samples stored for up to 50 days in a refrigerator and tap water could be use for the wash and backwash operations without any impact on the particle size profile. The granulometry obtained by image analysis was not comparable to that given by sieving.

Isik and Sponza [112] treated a simulated wastewater containing sizing agents, azo dyes, salts and other additives using a lab-scale UASB reactor at different hydraulic retention times. The COD removal efficiency decreased from 80 to 29.5% when the HRT was decreased from 100 to 6 h. The colour removal efficiency was 90 and 95% for HRTs of 100 and 6 h.

Bressani et al [113]. stated that since the high-rate anaerobic treatment of sewage using UASB reactor only removes organic carbon, a cost-effective post-treatment such as trickling filters (TF) is required to remove nitrogen, besides residual organic carbon, thereby assuring low sludge production, low operational costs and maintenance simplicity. They reviewed the experience of the last 20 years of research, design and operation of UASB/TF systems. Three main topics were addressed: (a) the development of trickling filters for UASB reactor effluent treatment, building on first experiences with TF preceded by primary settlers, (b) the design criteria, performance and empirical models for predicting the efficiency of TF post UASB reactor and (c) the future challenges associated with elimination of secondary settlers and nitrogen removal in sponge-bed trickling filter (SBTF).

Rotating Biological Contactors

A rotating biological contactor (RBC) is a biological fixed-film treatment process used in the treatment of wastewater following primary treatment. The primary treatment process involves removal of grit, sand and coarse suspended material through a screening process, followed by settling of suspended solids. The RBC allows the wastewater to be in contact with a biological film to remove pollutants in the wastewater before discharge to a river, lake or ocean. The RBC consists of a series of closely spaced, parallel discs mounted on a rotating shaft which is supported just above the surface of the wastewater (Figure 10). Microorganisms grow on the surface of the discs where biological degradation of pollutants takes place. The microbes are alternatively exposed to the atmosphere allowing both aeration and assimilation of dissolved organic pollutants and nutrients for degradation [114-119].

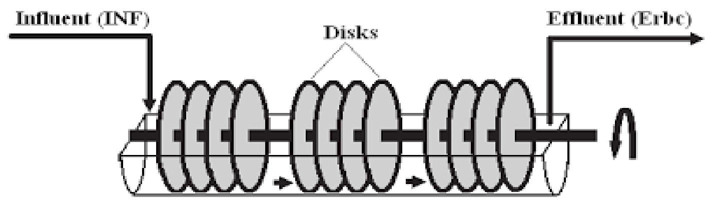


Figure 10: Rotating biological contactors [117].

Pathan et al [120]. studied the performance of a single-stage laboratory scale RBC treating greywater in Pakistan. The RBC tank was made of plastic sheets and the disks were made from textured plastic. An electric motor equipped with gear box to control the rotations of the disks was mounted on the tank and the system was run at of 1.7 rpm. The disc area was immersed about 40% in the greywater. The removal of BOD5 and COD were 53% and 60%, respectively.

Friedler et al [121]. studied the potential chlorination and UV irradiation of RBC treatment effluent in removing indicator bacteria (faecal coliforms, heterotrophic bacteria) and specific pathogens (*Pseudomonas aeruginosa sp., Staphylococcus aureus sp*.). The study concluded that RBC removed 88.5–99.9% of all four bacteria groups. However, the effluent had to be disinfected. Most of the chlorine was consumed during the first 0.5 h, after which its decay rate decreased significantly. The remaining residual after 6 h was sufficient to prevent regrowth of bacteria in the stored greywater effluent. Under exposure to low UV doses (≤69 mJ/cm2), faecal coliform was the most resistant bacteria group, followed by heterotrophic bacteria, *Pseudomonas aeruginosa and Staphylococcus aureus*. Exposure to higher doses (≤439 mJs/cm2) completely inactivated faecal coliforms, *Pseudomonas aeruginosa and Staphylococcus*, and no further heterotrophic bacteria inactivation was observed.

Gilboa and Friedler [122] evaluated the effectiveness of RBC for removal of faecal coliforms, *Staphylococcus aureus, Pseudomonas aeruginosa and Clostridium perfringens* in greywater followed by sedimentation. The raw greywater had 1.6 x 10^7 cfu/ml heterotrophic plate count, and 3.8 x 10^4, 9.9 x 10^3, 3.3 x 10^3 and 4.6 x 10^1 cfu/100 ml faecal coliforms, *Staphylococcus aureus, Pseudomonas aeruginosa. and Clostridium perfringes*, respectively. The RBC systems performed well with respect to pH, BOD_5, COD, reduced microbial loads and produced effluents that meet discharge guidelines. The system removed up to 99% of all microorganisms found in the raw greywater.

Hassard et al [123]. reported that the RBC was used to remove organic matter from wastewater with loading rates of up to 120 g/m2 d with an optimum loading rate at around 15 g/m2 d (combined BOD and ammonia). Full nitrification was achievable with oxidation rates of 6-14 g/m2 d for nitrogen rich wastewaters. A biological total phosphorus removal of 70% and up to 99% of fecal coliforms and most other pathogens were achieved.

Tawfik et al [124]. evaluated the RBC treatment process of domestic wastewater at temperatures of 12–24°C. The RBC was a two-stage system connected in series and operated at different organic loading rates (OLR) and hydraulic retention times (HRT). The overall removal efficiencies for COD_{total}, $COD_{suspended}$ and $COD_{colloidal}$ significantly decreased when decreasing the HRT from 10 to 2.5 h and increasing the OLR from 11 to 47 g COD/m^{2}d. Most of the COD was removed in the first stage and nitrification took place in the second stage. The overall nitrification efficiency was 49% at total OLR of 11 g COD/m^{2}d. At HRTs of 10, 5 and 2.5 h, the *Escherichia coli* concentration was reduced by 1.6, 1.5 and 0.8 log10, respectively.

Trickling Filters

A trickling filter is a wastewater treatment system consisting of a fixed bed of rocks, coke, gravel, slag, foam, sphagnum peat moss, ceramic, or plastic media over which wastewater flows downward and causes a layer of microbial slime (biofilm) to grow and cover the bed media. Aerobic conditions are maintained by splashing, diffusion, and either by forced air flowing through the bed or natural convection of air if the filter medium is porous (Figure 11). Typically, wastewater flow enters at a high level and flows through the primary settlement tank. The supernatant from the tank flows into a dosing device, often a tipping bucket which delivers flow to the arms of the filter. The flush of water flows through the arms and exits through a series of holes pointing at an angle downwards. The liquid is distributed evenly over the surface of the filter media. Some are uncovered and freely ventilated to the atmosphere. The removal of pollutants from the wastewater stream involves absorption and adsorption of organic and inorganic (nitrate and nitrite) compounds by the microbial biofilm. Passage of the wastewater over the media provides the dissolved oxygen required for the biochemical degradation of the organic compounds and the releases carbon dioxide gas, water and other oxidized end products. As the biofilm layer thickens, it eventually sloughs off into the liquid flow and subsequently forms part of the secondary sludge that can be removed

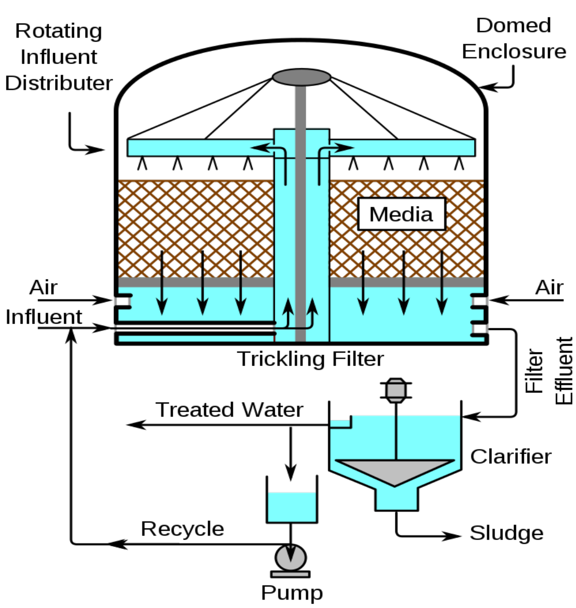


Figure 11: A trickling filter [125].

by a clarifier or sedimentation. The biofilm contains many species of bacteria, ciliates, protozoa, annelids, roundworms and insect larvae as well as many other microfauna. Within the thickness of the biofilm, both aerobic and anaerobic zones can exist supporting both oxidative and reductive biological processes [4,5,6,125].

Diagger and Boltz [126] reported that the major components of modern trickling filter are: (a) rotary distributors with speed control, (b) modular plastic media (typically cross-flow media unless the bioreactor is treating high-strength wastewater, which warrants the use of vertical-flow media), (c) a mechanical aeration system that consists of air distribution piping and low-pressure fans, (d) influent/recirculation pump station and (e) covers that aid in the uniform distribution of air and foul air containment (for odor control) [126]. The covers may be equipped with sprinklers that can spray in-plant wash water to cool the media during emergency shut down periods.

Logan et al [127]. Developed a computer model to examine biochemical oxygen demand (BOD) removal in plastic media trickling filters. The performance of trickling filters was predicted using first order microbial kinetics and equations of substrate transport in the thin fluid film. Various geometries of trickling filter support media caused different fluid hydraulics. The trickling filter model was calibrated and verified using the results of several laboratory, pilot plant, and full-scale studies.

Vianna et al [128]. treated domestic sewage in trickling filters in laboratory pilot plants in which the peeled dehydrated fruits of *Luffa cyllindrica* were used as a support medium for microbiological growth, The capacity of the system to remove organic matter, measured in terms of (BOD_5,20) and COD as well as suspended and settleable solids,, was evaluated. When comparing the results to those obtained from similar pilot plant using stones as supporting medium, it appeared that this support medium can be used as a substitute to the traditional support media.

Matsumoto and Weber [129] reported that a fine-grained, shallow-bed, air-pulsed filter was used successfully to treat primary greywater effluents at several locations. Suspended solids and BOD_5 removals were 50-70% and 25-45%, respectively [129]. The information obtained from the study showed how the use of primary effluent filtration can alter the design and operating characteristics of trickling niters.

Naz et al [130] assessed selected packing media for biological trickling filters (BTFs) and developed a simplified model for describing the capacity of BOD removal in BTFs. BTFs with four different media (rubber, polystyrene, plastic and stone) were evaluated at two temperature ranges (5–15°C and 25–35°C) [130]. The average removasl of COD and BOD were higher than 80 and 90% at the temperature ranges of 5–15 and 25–35°C, respectively. The geometric mean of faecal coliforms in BTF using polystyrene, plastic, rubber and stone as filter media at the low temperature range of 5–15°C was reduced by 4.3, 4.0, 5.8 and 5.4 $\log_{10}$, respectively. At the higher temperature range of 25–35°C, the faecal coliform count was reduced by 3.97, 5.34, 5.36 and 4.37 $\log_{10}$ by the polystyrene, plastic, rubber and stone media, respectively. The model showed that highly efficient BTFs are capable of treating organic loading rates of more than 3 kg BOD/m3 day.

Zylka et al [131] investigated the possibility of using a trickling filter for treatment dairy wastewater after dissolved air flotation (DAF). The results confirmed the possibility of high efficiency treatment of dairy waste with DAF and trickling filter technologies. The average efficiency of DAF treatment was 59.3% for BOD, 49.0% for COD and 80.0% for total phosphorus, while the average treatment efficiency of TF without recirculation was 87.3%, 78.3% and 27.9% and 95.2%, 85.5% and 42.0% with 100% recirculation applied, for BOD, COD, total phosphorus respectively.

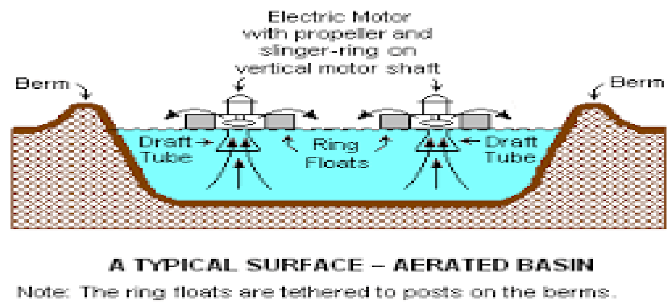
Dhokpande et al [132]. reviewed research on application of trickling filters for removal of various pollutant from wastewater from mining, textile and other industries and concluded that the trickling filter processes are very efficient in handling many types of water pollutants with COD removal up to 90 % and nitrogen removal up to 99 %. The removal of heavy metals (copper, lead and nickel) and citrate was reported to be around 90 %.

Aerated Lagoons

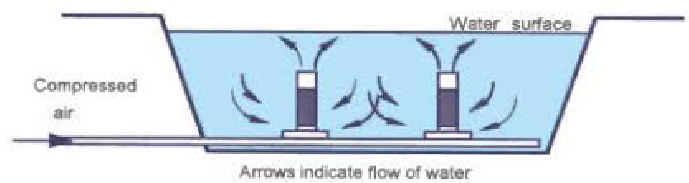
An aerated lagoon is a suspended-growth process in wastewater treatment unit. The aerated lagoon wastewater treatment system consists of a large earthen lagoon or basin that is equipped with mechanical aerators to maintain an aerobic environment and to prevent settling of the suspend biomass. It is provided with inlet at one end and outlet at the other end to enable the wastewater to flow through and to be retained for a specified detention time. The microbial population in an aerated lagoon is much lower than that in an anaerobic sludge blanket (ASP) because there is no sludge recycle and thus a longer residence time is required. Also, the biological oxidation processes are sensitive to temperature and the reaction rates increase with increases in temperature within the range of 4-32 °C [23, 133].

Aerated lagoons can be aerated from the surface using floating aerators or from subsurface using submerged aerators (Figure 12). In a surface-aerated system, the aerators provide two functions: (a) transfer the air into the liquid required by the biological oxidation reactions, and (b) provide the mixing required for dispersing the air and for contacting the reactants (oxygen, wastewater, and microbes). The floating high speed surface aerators deliver the oxygen at a rate of 1.0-2.5 kg O_2 kWh. Aerated lagoons using floating surface aerators achieve 80-90% removal of BOD with retention times of 1-10 days. They may range in depth from 1.5 to 5.0 metres [134,135]. The submerged aerator is essentially a form of a diffuser grid inside the lagoon. These systems utilize medium bubble diffusers to provide aeration and mixing to the wastewater. The diffusers can be suspended slightly above the lagoon floor or rest on the bottom. A flexible airline supplies air to the diffuser unit [133,136-137].

There are two types of aerated lagoons based on how the microbial mass of solids in the system is handled: (a) suspended growth aerated lagoon and (b) facultative aerated lagoons. Suspended growth aerated lagoons are relatively shallow earthen basins varying in depth from 2 to 5 m and are provided with mechanical aerators (on floats or fixed platforms) to provide oxygen for the microorganisms as well as to keep the biological solids in suspension and maintain fully aerobic conditions from top to bottom. No settlement occurs in such lagoons and under equilibrium conditions, the new microbial solids produced in the system equal to the solids leaving the system. Because the aerated lagoon is a complete mix reactor without recycle, the SRT is equal to HRT and vary from 3 to 6 days. In facultative aerated lagoon, the aeration power is sufficient for oxygenation and not for keeping solids in suspension and as a result some solids leave with the effluent, and some settle down in the lagoon. Therefore, the lower part of facultative lagoons may be anaerobic while the upper layers are aerobic. Facultative aerated lagoons have been more commonly used because of their simplicity in operation, minimum need of machinery and require much less land compared to oxidation ponds. Facultative aerated lagoons can provide 70–80% BOD removal from readily degradable wastes such as domestic and grey wastewaters [23,138-142].



(a) Surface aerated lagoon



(b) Subsurface aerated lagoon

Figure 12. Aerated lagoon [140].

The advantages of aerated lagoons are: (a) they are simple and rugged in operation, the only moving piece of equipment being the aerator, (b) the removal efficiencies and the power input are comparable to the other aerobic treatment methods, (c) construction mainly entails earthwork and land requirement is not excessive (5-10 % as much land as stabilization ponds) and (d) they are used frequently for the treatment of industrial and domestic wastewaters as well as animal wastewater [133,138,141].

Lagoons typically have 50-200 mg/L dry weight biomass compared to activated sludge systems which typically have 1000-5000 mg/L and function 10-20 times slower than activated sludge systems. The microorganisms responsible for biological treatment in lagoons are interrelated. Bacteria decompose the organic material and convert it into new bacterial cells and carbon dioxide. The carbon dioxide produced by this process and atmospheric carbon dioxide is used by algae to generate new alga cells and oxygen during the sunlight period. Herbivores (microscopic animals) graze on the algae and bacteria. Carnivores (larger animals) graze on the herbivores. Most of the microorganisms in aerated systems convert food to energy in the presence of free dissolved oxygen. Anaerobes obtain oxygen from chemically bound oxygen compounds such as nitrate and sulfate. Facultative organisms use either free dissolved oxygen or chemically bound oxygen [137-138].

Rich [143] discussed the impact of effluent algae and nitrification in the BOD 5 test on the performance of aerated lagoons and the ways of improving their performance. He modified the procedure for the design of dual-power multicellular (DPMC) aerated lagoon systems which included the use of: (a) a steady-state model for the hydrolysis of the fraction of influent biodegradable materials and (b) a steady-state algal growth model. Parameter values were applied to the models and the results showed that the modifications improve performance with respect to effluent quality.

Fonade et al [144]. reported on treating industrial wastewaters by aerated lagoons, with high flowrates. They developed a methodology which gave the best fit between the biological reactions and the ideal hydrodynamic behaviour of the lagoon, based on the real kinetics of the degradation process This, lead to the minimum volume and the ideal behaviour of the lagoon needed to reach the degree of conversion required to meet the discharge regulations. This original approach is suitable for modifying existing aerated lagoons, or for upgrading natural lagoons to aerated lagoons, to successfully meet the new legislation on effluent quality.

Andiloro et al [145]. investigated the treatment of olive oil mill (OMW) wastewater using aerated lagoon due to its low cost and easy management. The effects of the aeration rates, concentration of polyphenols and nitrogen shortage on depuration performance of lagoon were explored. The removal rates of COD and PP, and variations of pH in the treated OMW were determined. Compared to the non-aerated systems, aeration of OMW increased the removal rates from 61% to 90% for COD and from 52% to 64% for PP. Permanent aeration was more advisable compared to intermittent air flow rates. Increasing concentrations of PP noticeably reduced the COD removal rates, which were halved at a 4-fold PP concentration. In contrast, the PP removal rate was constant at every concentration experimented. A shortage in nitrogen availability (COD:N higher than 400:5.) reduced COD removal by about 20–25% and PP removal by 25%. The pH was less influenced by the variations in aeration rates, PP concentration and COD:N ratio.

Anaerobic Up-flow Filters

A biofilter is a bed of media on which microorganisms attach and grow to form a biological layer called biofilm. Thus, biofiltration is usually referred to as a fixed–film process used for air pollution control, water treatment and wastewater treatment. Generally, the biofilm is formed by a community of different microorganisms (bacteria, fungi and protozoa) and extracellular polymeric substances. The water to be treated can be applied intermittently or continuously over the media, via up-flow or downflow [146]. The anaerobic up-flow filter (Figure 13) represents a significant advance in anaerobic waste treatment since the filter can trap and maintain a high concentration of biological solids. By trapping these solids, long SRT's could be obtained at large waste flows, necessary to anaerobically treat low strength wastes at low temperatures economically. The anaerobic up-flow filter has been successfully used for the treatment of different types of wastewaters including domestic wastewater, aquaculture water for recycling, greywater and carwash water as a way to minimizing water replacement while maintaining environmental quality [147-149].

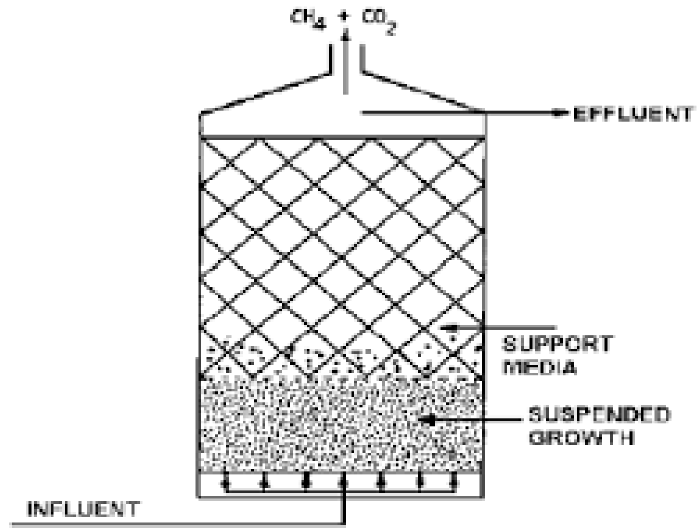


Figure 13: Anaerobic up-flow filter [149].

Organic matter and other water components diffuse into the biofilm where the treatment occurs by biodegradation process under anaerobic conditions, which means that microorganisms do not require oxygen for their metabolism. The main factors influencing the efficiency of biofilter are the water composition, the biofilter hydraulic loading, the type of media, the feeding strategy (percolation or submerged media), the age of the biofilm and temperature. Biological filters internal hydrodynamics and the microbial biology and ecology are complex and variable, characteristics that confer robustness to the process and give it the capacity to maintain its performance or rapidly return to initial levels following periods of no flow, intense use, toxic shocks or media backwash [150].

The structure of the biofilm protects microorganisms from difficult environmental conditions and retains the biomass inside the process, even when conditions are not optimal for their growth. Other advantages include: the development of microorganisms with relatively low specific growth rates because microorganisms are retained within the biofilm, the biofilters are less subject to variable or intermittent loading and to hydraulic shock, operational costs are usually low, final treatment results ares less influenced by biomass separation since the biomass concentration at the effluent is much lower than that in suspended biomass and attached biomass becomes more specialized (higher concentration of relevant organisms). However, because filtration and growth of biomass leads to an accumulation of matter in the filtering media, this type of fixed-film process is subject to bio-clogging and flow channeling. However, depending on the type of application and the media used for microbial growth, bio-clogging can be controlled using physical and/or chemical methods such as backwash using air and/or water to disrupt the bio-mat and recover flow or using oxidizing chemicals (Peroxide and ozone) or biocide agents [151].

Young and Yang [152] stated that anaerobic filters represent a treatment technology suitable for treatment of wastewaters containing soluble biodegradable organic materials. There was a good agreement between results obtained from laboratory and full-scale anaerobic filters. The most critical design factors affecting performance are hydraulic retention time, media type, and flow direction. Media surface area affects performance slightly, with higher efficiencies associated with higher specific surface areas. However, the difference is so small that the use of media having high specific surface area may not be economically justified. Treatment performance is not affected significantly by influent wastewater having COD values above about 3 000 mg/L. Reactor height seems to have no significant effect on performance, but minimum media heights of 2 m are recommended for full-scale anaerobic filters.

Kavittha [153] reported that anaerobic reactors have been successfully installed in full-scale plants world-wide for treating high-strength industrial wastewater and domestic wastewater over the years. Initially, it was thought that this was not practical as methane fermentative process was considered too slow to be able to treat the increasing volume of domestic sewage at a high rate but with technological advances, better understanding of anaerobic microbial characteristics, and good control of the biological process these barriers can be overcome.

Pak and Chang [154] tested a two-biofilter system operated under alternating anaerobic/aerobic conditions to remove nutrient and organics from wastewater generated from car washing facility. The wastewater was characterized by relatively low organic and high phosphorus contents. The operational parameters examined in this study were hydraulic retention time, organics, suspended solid and total nitrogen loading rates. The factors affecting phosphorus removal in the biological filter appeared to be influent COD, nitrogen and the COD/TP, BOD/COD and SS/TP ratios.

Expanded Bed Up-flow Reactor

The expanded bed up-flow reactor (Figure 14) is a variant of the up-flow anaerobic sludge blanket digestion concept (Figure 15) for anaerobic wastewater treatment. The distinguishing feature is that a faster rate of upward-flow velocity is designed for the wastewater passing through the sludge bed. The increased flux permits partial expansion (fluidization) of the granular sludge bed, improving wastewater-sludge contact and enhancing segregation of small inactive suspended particle from the sludge bed. The expanded bed up-flow reactor relies on the development of biomass on the surfaces of a media.

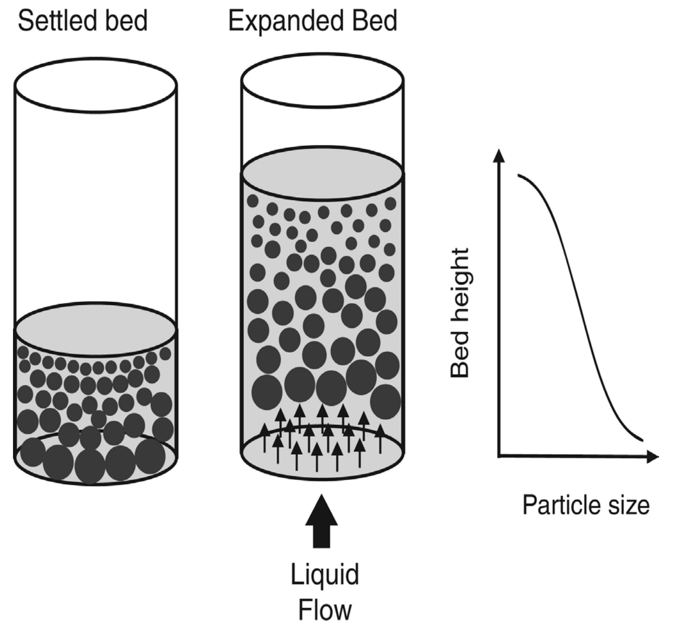


Figure 14: Expanded bed up-flow bioreactor [156].

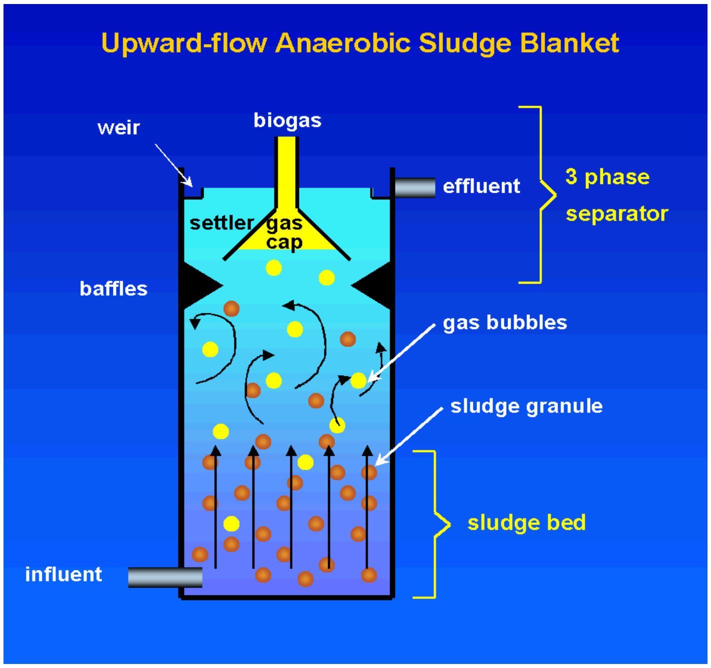


Figure15: Moving bed biofilm reactor [157].

The primary concept of the process consists of passing wastewater up through a bed of inert sand sized particles at sufficient velocities to fluidize and partially expand the sand bed. The system design is appropriate for low strength soluble wastewaters (less than 1-2 g soluble COD/L) or for wastewaters that contain poorly biodegradable suspended particles which should not be allowed to accumulate in the sludge bed. [155-157].

Moharram et al [158]. studied the performance of an anaerobic up flow fluidized bed reactor as a primary treatment unit in domestic wastewater treatment with unexpected industrial water flows at different operational temperatures (14–25 °C), different organic loading rates (OLR) and different HRT (6, 4, 2.5 h). The best methane yield rate (0.285 l/g COD total) and COD removal rate (70.82%) were achieved at a temperature of 19 °C, OLR of 7.76 kg COD/m^3/day and HRT of 6 h. On the low temperature operation, the average COD removal was 55.28% and 50.33% for HRT of 4 h and 2.5 h respectively. The methane production dropped to 0.1623 and 0.0988 L CH4/g COD with average OLR of 5.34 and 10 kg COD/m^3/day for HRT of 4 h and 2.5 h, respectively. The total nitrogen removal ranged between 2.23 and 10.83% with an apparent decrease during the low temperature. Nitrite removal was in the range of 23–77%, with up to the 2 mg/L in the effluent water when obtaining high organic loading and warm temperature.

Switzenbaum and Jewell [189] found the anaerobic attached film expanded bed reactor (AAFEB) to be effective for the treatment of low strength soluble organic wastes anaerobically at reduced temperatures, short retention times, and high organic loading rates [159]. The process consists of inert particles (approximately 500 microns in apparent diameter) packed in a cylindrical column which expanded slightly with the upward flow of liquid through the column. Three reactors fed a soluble synthetic waste consisting of glucose and nutrient salts at concentrations ranging from 50 to 600 mg/L COD were monitored over a period of 9 months of start-up and six months of operation. The effects of temperature, influent substrate concentration, and hydraulic flow rate on process efficiencies were measured. The AAFEB permitted the maintenance of high solids retention times (SRT) with low hydraulic retention times (HRT). This study presented an analysis of the key process variables which affect AAFEB operation and presented two simplified first order equations relating the process efficiency.

Yoochatchaval et al [160]. operated a laboratory scale expanded granular sludge bed (EGSB) reactor at 20°C with low strength wastewater (0.6-0.8 g COD/L) for over 200 d. The reactor was inoculated with mesophilic granular sludge. The up-flow velocity was set to 5 m/h by effluent recirculation. The COD loading was increased up to 12 kg COD/m3/d until the day 76, resulting in hydraulic retention time of 1.5 h. The settleability and diameter of retained sludge tended to deteriorate during the first 2-3 months However, sludge settleability kept sufficiently in the later part of experiment due to the reconstruction of granular sludge. The growth yield (Yg) of retained sludge (0.13 g VSS/g COD) was about two times higher than mesophilic and thermophilic granular sludge processes while the endogenous decay constant (K_d) was very low (0.0001/day) as compared with those processes. The sludge retention time was reduced from 100 to 40 d by the reduction of hydraulic retention time from 4 to 1.5 h. Maintenance of 40 days of sludge retention time caused the stable retainment of biomass and the significant increase of methanogenic activity of the retained sludge.

Jaafari et al [161]. studied the effect of up-flow velocity on performance and biofilm characteristics of an anaerobic fluidized bed reactor (AFBR) treating wastewater at various loading rates. The reactor was made of a plexiglass column of 60 mm diameter and 140 cm height with a volume of 3.95 L. The AFBR system could handle an exceptionally high organic loading rate. At organic loading rates of 9.4-24.2 kg COD/m^3 at steady state, the reactor performances with up-flow velocities of 0.5, 0.75 and 1 m/min were 89.3-63.4, 96.9-79.6 and 95-73.4 %, respectively. The average biomass concentration per unit volume of the AFBR (as gVSSatt/L expended bed) decreased with the increase of up-flow velocity in the range of 0.5?1m/min at all applied organic loading rates. The total biomass in the reactor increased with increases in the organic loading rate. The peak biomass concentration (as gVSSatt L expended bed) was observed at the bottom part of the reactor, then it dropped off slowly towards the top. The biofilm thickness increased from the bottom to the top of the reactor representing a stratification of the media in the AFBR. The bed porosity increased from the bottom to the top of the reactor.

Disinfiction

In biological treatments, the degradation and transformation of greywater constituents are facilitated by the biochemical reactions carried out by microorganisms in the liquid medium. However, the effluent of biologically treated greywater may contain pathogenic microorganisms. Therefore, a final disinfection step may be needed to eliminate the risk of contracting diseases bya microorganisms living in the treated effluents. Three of the most common hazard species causing gastroenteritis in humans through fecal-oral contamination are *Escherichia coli, Cryptosporidium spp.,* and *Giardia lamblia*. Selection of a disinfectant to remove pathogens is of great importance for public health [23].

The selection criteria for a disinfectant include: (a) non-toxicity to humans, domesticated animals and aquatic ecosystems, (b) low cost (c) easy handling, (d) reliable analysis, and (e) a satisfactory residual concentration. When evaluating any disinfection process selected (chemical oxidants, irradiation, or thermal process), the conditions of the wastewater source and existing treatment design into which the disinfectant will be added must be taken into consideration [162-146].

Chemical Disinfectants

Chemical disinfectants are quite inexpensive, leave a residual concentration and when properly dosed are effective in killing pathogens. Chlorine is generally used as a disinfectant because it is inexpensive and easier to handle. Chlorine is effective as free chlorine in the form of Cl2, OCl-, and HOCl. The dissociation of chlorine is pH and temperature dependent. At a pH of < 5.0, chlorine presents as HOCl, and at a pH >10.0 presents as OCl-. TOHCL is about 80-200 times as strong as the OCL-, thus making the disincentive properties extremely pH dependent. Disinfection concentrations of total chlorine at most treatment plants vary from 0.2 to 40 mg/L depending on the demands of the source being treated but the maximum dosage rarely exceeds 15 mg/L. However, the use of chlorine as disinfectant can result in the formation of the carcinogenic byproduct trihalomethanes (THMs) when chlorine reacts in the presence of organic matter. Chloramines, chlorine dioxide, permanganate, ozone, bromine, bromine chloride, and iodine are all available alternative disinfectants [162].

There are several factors that influence the rate at which microorganisms are killed or inactivated by chemical agents which must be considered in the application of any chemical agent to destroy microbial populations. These include type of microorganisms and physiological state of cells, chemical concentration, time of contact, temperature, nature of material bearing the organisms, surface tension and pH [162.166].

Microorganisms differ in their susceptibility to chemical agents. Different species of microbes especially bacteria show much greater variation in their susceptibility to chemical agents than they do to sterilization by physical agents. The physiological state of the organisms may influence their susceptibility to antimicrobial agents. Generally, the growing vegetative cell have the most susceptibility to chemical agents whereas the spore forms are extremely resistance. Young actively metabolizing cells are easily damaged through interference with synthesis while nongrowing cells would not be severely affected. [163,164].

The concentration of disinfectant used depends on the material being disinfected, and on the organisms to be destroyed. In general, a higher concentration will be bactericidal whereas a weaker concentration will be only bacteriostatic. Therefore, concentration plays a major role in determining the effectiveness of a chemical against microbes. In a strong concentration, a given compound may quickly kill vegetative cells and spores but when somewhat diluted the same compound may function as a disinfectant killing only the sensitive organisms. When further diluted so that it is no longer injurious to tissue, it may be used as an antiseptic inhibiting bacterial growth. However, when diluted to trace amounts, the compound may stimulate the growth of microbes [165].

A germicidal agent rarely kills microbes on instant contact. Extremely sensitive bacteria may be killed in a minute or even in seconds. However, hours may elapse before spores succumb to many of the germicides. Most chemical agents in common use as disinfectants are germicidal but are not sporicidal, they do not kill all spores present. The process of disinfection is a gradual one and a few microorganisms survive longer than the majority do. Therefore, to be effective, the disinfectant must be applied for a length of time sufficient to destroy all microorganisms. This often means 18-24 h [163].

An increase in temperature speeds up the rate of a chemical action. Generally, an increase of 10 ^{o}C would double the rate of disinfection. Therefore, the higher the temperature at which the disinfectant is applied the more effective it is. Small amount of chemical at an elevated temperature will accomplish the same results as a large amount of the same substrate at a lower temperature [163-165].
The physical and chemical properties of the medium or substance carrying the organisms have a profound influence on the rate as well as the efficiency of microbial destruction. The consistency of material (aqueous or viscous) will markedly influence the penetration of agent. High concentration of carbohydrates generally absorb the chemical agent and some chemical agents may be unable to penetrate organic matter. As general rule, there is a slowing down of the destructive effect of the disinfectant in the presence of organic matter. A disinfectant added to organic matter-microorganisms mixture may result in combining of the disinfectant with the organic material to form a new product that precipitate, thus denying the disinfectant from possible combination with microorganisms. In addition, accumulation of the organic matter on the microbial cell surface provides coating which will impair the contact between the disinfectant and cell [165].

Surface tension determines the penetration that takes place and the ability of the disinfectant to be brought into contact with the organisms. The pH of the medium may greatly influence the disinfection process. The pH alone may determine whether an agent is only inhibitory in action or is lethal [166].

Several authors reported on the use of chemical disinfection process after biological treatment. Bernstein et al [167]. reported on onsite chlorination of greywater in a vertical flow constructed wetland. Wang et al [168]. reported on disinfection of hospital wastewater using liquid chlorine, sodium hypochlorite, chlorine dioxide and ozone. Collivignarelli et al [169]. reported on the use of chlorine-based disinfectants for disinfecting wastewater. De Souza et al [170]. used peracetic acid alone and in combination with UV radiation to disinfect wastewater. Winward et al [171]. examined the impact of organics and particles on chlorine disinfection of grey water (measured by total coliform inactivation) and found the efficacy of disinfection was most closely linked with particle size. Larger particles shielded total coliforms from inactivation and disinfection efficacy decreased with increasing particle size. The organic concentration of grey water affected chlorine demand but did not influence the disinfection resistance of total coliforms when a free chlorine residual was maintained...

UV radiation Disinfectant

This well-developed method is used for the disinfection of drinking water, food processing water, municipal and industrial wastewater, swimming pools and different liquid products. The radiation Intensity loss is about 30% at 40 cm below the liquid surface because of lack of penetration [172]. Allowing the turbulent flow of liquid during the disinfection process can overcome the insufficient contact of all particles with the UV radiation. Continuous-wave mercury vapor lamps in either low pressure (monochromatic at 253.7nm) or medium-pressure (polychromatic in the UV and visible light range) formats are the conventional technology which has been used in water and wastewater disinfection by UV radiation [173].

Proteins, RNA and DNA in microorganisms absorb ultraviolet radiation. Disruption of the cell membrane is the result of the absorption of ultraviolet light by protein in membranes at high UV doses which lead to the death of the cell. Adsorption of lower doses of ultraviolet light by DNA (or RNA in some viruses) can disturb the ability of the microorganisms to replicate [174,175]. The principal inactivation effect of UV irradiation is the formation of photoproducts in the DNA, the most important of which is the pyrimidine dimer formed between adjacent pyrimidine molecules on the same strand of DNA, which can interrupt both DNA transcription and translation [176].

Several authors reported on the use of UV radiation for treatment of wastewater. Wang et al [168]. reported on disinfection of hospital wastewater using ultraviolet irradiation. De Souza et al [170]. used UV radiation to disinfect wastewater. Snowball and Hornsey discussed the purification of water supplies using UV light. Bohrerovaa et al [172]. reported on the comparative disinfection efficiencies of pulsed and continuous-wave UV irradiation technologies for wastewater. Shoults and Ashbolt [177]. examine the efficacy of *Staphylococcus spp*. as an endogenous surrogate for greywater pathogen reduction performance testing, by evaluating UV-C irradiation of hand-rinse greywater, and the variability in UV resistance between different wild *Staphylococcus spp.* Fenner and Komvuschara [178]. presented a model for UV disinfection of greywater which incorporated variations in micro-organism sensitivity to UV radiation, the variation of dose received in the UV reactor chamber, and the shielding effect of part of the microbial population by the presence of particulates. The model could predict the asymptotic decay observed in bacterial survival curves when the organisms were exposed to a UV dose in a greywater matrix. Deck et al. [179] exposed greywater samples to monochromatic (253.7 nm) ultraviolet (UV) light and found a UV dose of 100 mJ/cm^2 completely inactivated *Enterococci* and reduced total coliforms to the required California Drinking Water Standards (achieved 5-log reduction of MS2).

Conclusion

Global water shortages will affect 2.7 billion people by 2025 resulting in reduction in agricultural land and increased dissertation leading to poverty, faming, war, illegal migration and human trafficking. Reusing greywater can contribute towards solving this water shortage problem. Greywater is a wastewater discharge originating from kitchen sinks, showers, baths, washing machines and dishwashers Greywater contains fewer pathogens and can be treated and reused onsite for toilet flushing, landscape, crop irrigation and other non-potable uses. Degradation and transformation of greywater constituents can be facilitated by biochemical reactions occurring in the liquid medium. The existing biological treatment systems for greywater are: (a) constructed wet land, (b) sequencing batch reactor, (c) vertical flow bioreactor, (d) membrane bioreactor, (e) up-flow anaerobic sludge blanket, (f) rotating biological contractors, (g) trickling filters, (h) aerated lagoons, (i) anaerobic up-flow filter, and (j) expanded bed up-floe reactor.

The constructed wetlands offer a low capital cost for treatment of greywater, but they occupy large areas and require higher HRTs to treat effluent efficiently. The sequencing batch bioreactors have low capital and operational costs and can be a viable form for the treatment of greywater, producing effluent with low TSS. However, the performance of the system is influenced by HRT and implementing process control is more complicated and requires a highly trained operator. The vertical flow bioreactor achieves high removal rates of BOD, COD, TSS and fecal coliforms and other pathogens. The system has a low capital cost., requires a little maintenance and can be operated by untrained personnel. The membrane bioreactor achieves high removal rates of BOD, COD and TSS, and the effluent is free from odours. However, the system is subject to severe membrane fouling and disinfection of the permeate is necessary to guarantee zero faecal coliform levels.

The up-flow anaerobic sludge blanket achieves a COD removal efficiency of 90%, does not require separate settler with sludge return pump or effluent recirculation and there is no loss of reactor volume. The rotating biological contractors process has low energy costs, high removal of BOD and nutrients, high process stability, high resistant to shock hydraulic or organic loading, short contact periods, low space requirement, well drainable excess sludge and low sludge production. However, it requires high investment and high operation and maintenance costs and must be protected against sunlight, wind and rain. Trickling filters have far greater void space and porosity within the media, which allows for a higher hydraulic loading and promotion of a heavier biological growth on the media, low maintenance requirements, resistance to upset from variations in wastewater volume and strength, operational simplicity, resistance to toxic and shock loads, and low energy requirements. However, there is a lower limit on the mass of oxygen demand that can be removed, and they are susceptible to nuisance conditions that are primarily caused by macro fauna. Aerated lagoons are simple and rugged in operation, their construction is very simple, land requirement is not excessive, the removal efficiencies and the power input are comparable with other aerobic treatment methods. The anaerobic up-flow filter can trap and maintain a high concentration of biological solids that allows for long SRT's that is necessary to anaerobically treat low strength wastes at low temperatures economically. The efficiency of this biofilter is influenced by wastewater composition, the biofilter hydraulic loading, the type of media, the feeding strategy, the age of the biofilm and temperature. However, this type of fixed-film process is subject to bio-clogging and flow channeling. Then expanded bed up-flow has a faster rate of upward-flow velocity of the wastewater passing through the sludge bed, causing partial expansion of the granular sludge bed, improving wastewater-sludge contact and enhancing segregation of small inactive suspended particle from the sludge bed. The system is appropriate for low strength soluble wastewaters and the removal efficiency and methane yield depends on temperature, HRT and organic loading rate.

In biological treatment, the degradation and transformation of greywater constituents are facilitated by the biochemical reactions carried out by microorganisms in the liquid medium. However, the effluent of biologically treated greywater may contain pathogenic microorganisms and require a final disinfection step is needed to eliminate the risk of contracting pathogenic diseases. The selection criteria for a disinfectant include: (a) non-toxicity to humans, domesticated animals, and aquatic ecosystems, (b) low cost (c) easy handling, (d) reliable analysis, and (e) a satisfactory residual concentration.

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Authors' contributions

This work was carried out in collaboration among all authors. All authors read and approved the final manuscript.

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