Excessive amounts of volatile organic compounds (VOCs) and odorants discharged into the environment are highly dangerous to human health as well as to ecological systems. Biological treatments of waste gas streams, called biofiltration, containing VOCs and odorous compounds has gained much attention because biofilters are more cost effective and environmentally friendly than conventional air pollution control technologies. This review provides an overview of biotrickling filtration, which is a type of biofiltration including continuous trickled-water flow inside filter media, for VOC and odor abatement. The configuration, design, cost effectiveness, removal capacity and environmental impact of this techniques and the future research and development needs in this area are all considered.

Key words: Biofilter, VOC, Odor, Biotrickling filtration, Biological treatment

1. INTRODUCTION

Various types of anthropogenic activities including kraft pulping, animal rendering, and wastewater treating/composting are sources of reduced sulfur compounds (RSCs: e.g., hydrogen sulphide (H₂S), dimethyl sulphide (DMS), methanethiol (MT), and dimethyl disulphide) (Syed et al., 2006; Sercu et al., 2005). These compounds are well known for their unpleasant smell and low odor thresholds (e.g., DMS 1.2 ppb and MT 2.4 ppb) (Verschueren, 2001). With ever-increasing global population and industrialization levels, the demand for sustainable VOC odor control technologies thus becomes more important in order to ensure nuisance-free air in and around the emission sources. However, in order to establish a better control strategy for VOCs, it is imperative to accurately characterize their atmospheric behavior and emissions.

It is widely acknowledged that VOCs can play a crucial role in the formation of surface ozone as well as secondary organic aerosols (SOAs) (Yue et al., 2017; Kroll and Seinfeld, 2008; Seinfeld and Pandis, 2006). These properties of VOCs have been brought into the light of the recent atmospheric research as most urban regions around the world are facing severe pollution associated with their production and emission (Sun et al., 2014). In this respect, a proper VOC emission inventory is the need of the hour, but current VOC emission inventories suffer from large uncertainties (e.g., as high as 100%) (Zhong et al., 2017). Zheng et al. (2017) attempted to quantitatively assess the industrial VOC emissions in China. These authors presented a sectoral VOC emission contributions with four processes using the spatial distribution of VOCs using GIS based emission factors and related data inventory (Fig. 1(a)). These authors pointed out that the VOC-containing products are the fastest growing sector towards the emission of VOCs with an average annual growth rate of 57.2%. They have also projected the average industrial VOC emission from 2000 to 2050 (Fig. 1(b)) using the currently available emission inventory. Accordingly, surface coating industries are also known to be a major culprit for the atmospheric emission of VOCs. Fig. 1(c) provides a comparison of VOCs emission in autocoating industry. Interestingly, VOCs released from the vehicle evaporative emissions also contribute significantly to photochemical air pollution, with toluene, isopentane/n-pentane, and 2,2,4-trimethylpentane as the dominant components (Yue et al., 2017).

In recent years, the sludge generated due to the mechanical, biological, and chemical treatment of wastewater has often been viewed as a useful raw
material to be composted owing to its high content of organic matter, nutrients, and other micro elements (Kosobucki et al., 2000). However, sludge treatment facilities have faced a wide social rejection due to the emission of unpleasant odors during composting (e.g., VOCs and ammonia) (Komilis et al., 2004; Goldstein, 2002). The introduction of more stringent environmental regulations implemented by government agencies has forced polluters to adopt more effective air pollution treatments (Giri et al., 2010). As a consequence, development of biological techniques which are more environmentally friendly and have higher pollution removal efficiency, has been gaining great attention as it may overcome various limitations in conventional techniques (Munoz et al., 2012). For instance, the traditional physicochemical processes such as incineration, employed for treating VOCs and other organic sulfur compounds require relatively high energy with high chemical use and disposal costs (Wani et al., 2008).

Biofiltration relies on aerobic microorganisms immobilized on solid particles in a bed media such as peat, compost, wood chips, or polyurethane foam packed in a column (Kumar et al., 2011). The biofilter is generally a fixed-film bioreactor that provides a large contact area between the gas stream and the microorganisms attached to porous media surface. As the polluted gas stream passes through the filter media, VOCs or odorous compounds in the gas are partitioned into the biofilm where biological oxidation occurs under aerobic conditions (Kumar et al., 2011). The main advantage of biofiltration is that the pollutants are converted into harmless end-products. Relatively low costs and excellent operational stability are also recognized as the advantages of biofiltration approaches (Rene et al., 2012).

Although various configurations exist, the main types of conventional gas-stream biological reactors include biofilters, biotrickling filters, and bioscrubbers. Among the recently developed reactors, membrane reactors have been used for the abatement of VOC and odor (Kumar et al., 2008a, b; Shareefdeen and Singh, 2005). Although the basic mechanisms of pollutant removal are similar to each other, differences exist in water flow types inside packing media. Fig. 2 presents an overview of the broad range of pollutants and applications for which the biological techniques are being used at present.

Fig. 1. (a) Sectoral VOC emission contributions with four processes (2013), (b) Industrial VOC emissions from 2000 to 2050, and (c) Comparison of VOCs emissions in auto coating industry [Adapted from Zheng et al. (2017) and Zhong et al. (2017)].
In this review, we focus particularly on the biotrickling filter (Fig. 3). The continuous supply of trickled-water provides a suitable treatment of hydrophilic VOCs. Nevertheless, as intimate contact exists between microorganisms and the pollutant stream, solubility restrictions are less stringent than for other biofilters such as bioscrubbers (e.g., dimensionless Henry’s law coefficient \( \frac{[C]_a}{[C]_g} < 0.1 \), where \([C]_a\) and \([C]_g\) are the aqueous-phase and gas-phase concentration of the species respectively) (Cox and Deshusses, 1999; Van Groenestijn and Hesselink, 1993). Also the continuous infusion (Fig. 3) of the nutrient solution facilitates control of the microbial activities and other operating parameters such as pH buffering.

This review has been organized to provide an overview of the biotrickling filter employed for the control of VOCs and odors, its merits and drawbacks, its important operational parameters, and future research and development needs in this area.

### 2. BIOLOGICAL GAS TREATMENT TECHNIQUES

The biological techniques for the treatment of VOCs include biofilters, biotrickling filters, bioscrubbers, and membrane bioreactors. In these methods, the pollutants are biologically degraded by aerobic microorganisms to stable end products like CO\(_2\), H\(_2\)O, sulfate, microbial biomass, etc. (Delhomenie and Heitz, 2005; Kim and Deshusses, 2005) (Figs 4-6 show the schematics of a Membrane Bioreactor, a Biofilter and a Bioscrubber, respectively).

#### 2.1 Biofilters

Biofiltration is the most typical type of biological air pollution control process, initially developed in the late 1970s (Leson and Winer, 1991). It is now emerging as a sustainable alternative for the treatment of air contaminated with VOCs and odorous compounds. In biofiltration, the polluted air is forced through a bed of packing media covered with a layer of aerobic microorganisms. The microorganisms are immobilized on the surface of the packing media. The primary role of the packing material in biofilter media bed is to support the microbial community through the attachment of microbial biofilm to the surface of packed media. Bohn (1992) established that an ideal biofilter bed
should have a high specific surface area (\(>300 \text{ m}^{-1}\)) for the proper development of the micro flora which can induce high gas-to-biofilm mass transfer. However, specific surface area as high as 1000 \(\text{m}^{-1}\) has also been reported for polyurethane-based beds (Mudliar et al., 2010). High porosity and water retention capacity are also highly desirable to facilitate homogenous distribution of gas flow and avoid media drying, respectively. The widely used packing materials for biofiltration include soil, compost, and wood chips. These materials are advantageous, as they satisfy the basic requirements stated above and are cost effective (Mudliar et al., 2010). In order to avoid bed crushing and compaction and to improve many other properties such as moisture hold up and microbial growth, several authors have suggested the use of advanced packing materials with complex blending such as a mixture of compost and wood chips or compost mixed with hard plastic (Taghipour et al., 2008). The pollutants are transferred from air to the water layer adhering to the bacterial growth on the media to be biologically metabolized (Upadhyay and Kumar, 2004). Biofiltration is energy efficient and cost effective while producing minimal quantities of toxic end-product. This technology has been successfully used for removing a wide range of pollutants such as VOCs, ammonia, mercaptans, and sulfurous compounds (Giri et al., 2010; Galera et al., 2008; Ho et al., 2008). The drawbacks of this technique are excessive pressure drops and gradual accumulation of acidic by-products due to dry-out, rapid degradation, and clogging; difficulty in controlling the biological operating parameters; clogging due to the accumulation of large amount of biofilm and reduced treatment efficiency at high pollutant concentrations (Mudliar et al., 2010). A typical biofilter has been observed to operate with a removal efficiency in the range of 50-95% (Park et al., 2001).

The removal efficiencies of toluene were reported to exceed 80% with an inlet concentration less than 1,274 ppm using an agro waste based biofilter (Singh et al., 2006). Moreover, Jaber et al. (2016) studied the removal of \(\text{H}_2\text{S}\) using a biofilter under extremely acidic conditions. A maximum \(\text{H}_2\text{S}\) removal capacity of 24.7 \(\text{g m}^{-3} \text{h}^{-1}\) was reported for a 0.07 \(\text{m}^3\) reactor at an inlet flow rate of 4 \(\text{m}^3 \text{h}^{-1}\) with a removal efficiency of 78% up to 360 ppm (v/v).

### 2.2 Bioscrubber

A bioscrubber essentially consists of a two stage unit in which absorption occurs in one stage and biodegradation by suspended microbes occurs in the other stage. Bioscrubbers are usually used for the treatment of readily soluble VOCs in the waste air stream (alcohols, ketones, etc.) having a concentration less than 5 \(\text{g m}^{-3}\) (Kellener and Flauger, 1998). Reported removal efficiencies are in the range of 50-99% (Webster and Devinn, 1995). Bioscrubbers are stable enough to allow a better control of operating parameters. Also, they produce a lower pressure drop across the microbe suspension than other filter types. The major problem associated with bioscrubbers is the generation of excess sludge and liquid waste which over time reduces the efficiency of the process considerably.
2.3 Membrane Bioreactors
Membrane bioreactors have been investigated by various researchers for VOC and odor abatement (Mudliar et al., 2010; Kumar et al., 2008a, b; Shareef-deen and Singh, 2005; Ergas and McGrath, 1997). They were designed as an alternative to conventional bioreactors for waste gas treatment. In a membrane bioreactor, the mass transfer of VOCs from the gas phase to a microbially active liquid phase occurs through the microporous hydrophobic hollow fiber membranes. The selective permeation of the pollutant across the membrane occurs due to the concentration difference between the gas phase and the biofilm phase, which provides the driving force according to Henry’s law. The advantages of membrane bioreactors are the absence of moving parts, ease of scale-up, and the ability to vary the flow of gas and liquid independently without the problems of flooding, loading, or foaming. Disadvantages associated with membrane bioreactors are the high investment cost and possible clogging of the liquid channels due to the formation of excess biomass. Removal efficiencies have been reported in the range of 50-99% (Hartmans et al., 1992).

3. BIOTRICKLING FILTER

A schematic description of a typical biotrickling filter (BTF) is provided in Fig. 1 (Delhomenie and Heitz, 2005). For this filter, the gas percolates through a packed bed, which is continuously irrigated with an aqueous solution containing essential nutrients required by the bio-organisms. It was reported that neither a counter-current configuration for liquid and gaseous phases has any influence on the biodegradation performance (Cox and Deshusses, 1999). Microorganisms can grow as a biofilm on the packing material of the filter bed. The biodegradation takes place within the biofilm, as the target pollutants are absorbed on the aqueous film. The filtering material of BTF should facilitate the flow of both gas and liquid through the bed and the development of the microflora while resisting crushing and compaction (Giri et al., 2010).

The contact between the microorganisms and the pollutants occurs after the diffusion of the pollutant in the liquid film. Hence, the liquid flow rate and the recycling rate are recognized to be critical parameters for BTF operation. Removal efficiencies for trichloroethylene have been reported in the range of 50-90% (Govind and Bishop, 1994). Likewise, removal efficiencies of H2S were estimated in the range of 95-98% (Govind and Bishop, 1994). Removal efficiencies of ethylene have been reported in the range of 50-90% (Mudliar et al., 2010; Kumar et al., 2008a, b; Shareef-deen and Singh, 2005; Ergas and McGrath, 1997).

3.1 Biotrickling Filtration Capital Costs
Capital costs for biotrickling filters vary widely according to filter size and construction materials. The required size of the biotrickling filter is determined by such variables as air flow rate, the nature and concentration of the pollutant treated, and the required removal efficiency. The presence of corrosive gasses (e.g., H2S) or solvent vapors is the main factor in determining construction materials (polyethylene, fiberglass, perlite, etc.) (Popoola et al., 2013). The cost of operating a biotrickling filter is increased by the presence of dust or fine particles, excessively high or low temperatures, highly fluctuating pollutant concentrations, etc. Hence, before commencing reactor design and construction, a detailed analysis and characterization of the reactors input air stream needs careful consideration.

A simple relationship (Equation 1) was proposed to estimate the capital cost of a biotrickling filter based on bed volume (Deshusses and Cox, 1999). The costs include basic components (e.g., pumps and level switches) for a simple biotrickling filter constructed out of inexpensive materials. The cost estimated by Equation 1 has ±20% accuracy for reactor volumes 5 to 1000 m³. For more expensive materials such as stainless steel, the pre-exponent term (13,000) needs to be increased in equation 1.

Biotrickling Filter Capital Cost (USD) = 13,000 × Bed Volume⁰.⁷⁵⁷ (1)

The reactor volume can be determined from knowledge of the pollutant concentration, the intended removal efficiency, and the air flow rate. For the reactor capacities of 10, 100, and 1,000 m³, the unit cost decreases substantially with increasing reactor size, viz., 7,500, 4250, and 2400 (USD m⁻³), respectively. Equation 1 is then used to estimate the capital cost (Table 1). The final installed cost is somewhat vendor dependent. Other costs such as: land, site preparation, assessment, maintenance, operating, financing, taxes, insurance, other overheads, etc. are also needed to determine the final installation and operational costs.

3.2 Biotrickling Filtration Operating Costs
The operating cost estimation of a biotrickling filter should include: 1) Electricity for the blower and the recycle pump along with other electrical equipment, 2) cost of the water and nutrients, 3) maintenance, 4) costs associated with biomass growth control, 5) capital costs associated with amortization (Deshusses and Cox, 1999).

Electricity for the blower is often a major fraction of the total operating expenses. In contrast, water, nutrients, and chemicals needed for the control of moisture...
and pH are a relatively small fraction (10-30%) of the total operating costs. Inspection of spray nozzles for possible clogging is the most important task during maintenance, as it could lead to inadequate wetting of the bed. If the biotrickling filter is subject to clogging problems, the costs of controlling the biomass growth can be significant - up to half of the total operating costs (Deshusses and Cox, 1999). Note that there are various approaches to control biomass growth such as the use of ozone to curb biomass accumulation (Zhou et al., 2016) or by protozoan predation (Cox and Deshusses, 1999). However, those methods have not yet been reliably developed for the applications at the industrial scale. Careful evaluation of the various options is recommended. Since biotrickling filter operation is relatively inexpensive, capital cost amortization will be significant compared to other costs. Assuming the lifetime of a filter plant is 10-20 years, the amortization of capital costs represents on average 20 to 40% of the total treatment cost. This stresses the importance of careful selection of materials and proper sizing to minimize the actual capital costs.

4. APPLICATIONS

A widespread application of BTF has been for the treatment of VOC and odor. This is a significant development over the use of a conventional biofilter (BF) that has generally been limited to the elimination of odorous compounds and non-chlorinated volatile organic compounds. This is due to the permanent trickling mechanism in BTF, which ensures the continuous distribution of the nutrient solution. As a result, BTFs can favorably control the biological operating conditions (viz. pH). Also, BTFs are known to be capable of treating the acid degradation products of VOCs.

4.1 Application of BTFs for VOC Abatement

Lu et al. (2001) achieved removal efficiencies as high as 95% for a mixture of acetone (20 g m⁻³ h⁻¹) and methyl acetate (27 g m⁻³ h⁻¹) using a bench-scale biotrickling filter. The filter comprised of an acrylic cylinder packed with phanerochaete chrysosporium immobilized on glass beads. Clogging of the medium, the complex filter structure and operation of the biotrickling filter, were the only shortcomings reported.

The effect of low dose ozonation was also investigated to prevent excess biomass accumulation and to maintain high removal efficiencies of toluene over extended BTF operation (Zhou et al., 2016). To optimize the biomass control strategy, the relative performance of five parallel BTFs was monitored at different ozone doses. The BTF was constructed from a Perspex pipe with a height of 0.95 m and an internal diameter of 9 cm. The active bed height was 48 cm with the bed volume of 3.1 L. The BTFs were packed with pelletized polyurethane foam (PUF) with a diameter of 10-15 mm. The pelletized PUF had an initial porosity of 91.0% with a specific surface area of 380 m² m⁻³. The ozone-free BTF performance declined after 150 days due to excess biomass accumulation, the buildup of extracellular polymeric substances (EPS) excreta, and a decline in metabolic activity of the biofilm. An optimized dose of ozone (e.g., 5-10 mg m⁻³, or 2.55-5.09 ppm) was sufficient to maintain stable operation (for 300 days) at

| Table 1. Estimated capital costs, footprint and treatment capacity of biotrickling filters of various sizesa. |
|----------------------------------------------------------|
| **Order** | Bed volume (m³) | Capital cost (Equation 1) (USD) | Approximate footprint (m²) | Approximate air flow rate (m³ h⁻¹) | Reactor unit cost (USD m⁻³) |
|-----------|----------------|-------------------------------|-----------------------------|-----------------------------------|-----------------------------|
| 1         | 5              | 45k                           | 1-2.5                       | 300-3,600                         | 9000                        |
| 2         | 10             | 75k                           | 2-5                         | 600-7,200                          | 7500                        |
| 3         | 20             | 125k                          | 4-10                        | 1,200-14,400                       | 6250                        |
| 4         | 50             | 250k                          | 10-25                       | 3,000-36,000                       | 5000                        |
| 5         | 100            | 425k                          | 20-50                       | 6,000-72,000                       | 4250                        |
| 6         | 200            | 720k                          | 40-100                      | 12,000-144,000                     | 3600                        |
| 7         | 500            | 1.4M                          | 100-250                     | 30,000-360,000                     | 2800                        |
| 8         | 1,000          | 2.4M                          | 200-500                     | 60,000-720,000                     | 2400                        |

aNot adjusted for inflation. 45% inflation from 1999 to 2016 (http://www.bls.gov/home.htm)
bEstimated using a 2-5 m bed height; to convert to sq. ft. multiply by 11.
cCalculated using EBRT of 5 s to 1 min; to convert to cfm. Multiply by 0.59.
a consistently high removal efficiency (>93%) at 400 mg m$^{-3}$ (106 ppm) toluene in the air; this prevented ex
cess biomass accumulation. On the other hand, ozone above 20 mg m$^{-3}$ (10 ppm) inhibited excessive biomass
growth to prevent poor BTF performance.

The biodegradation of toluene vapor was investigated by a lab-scale biofilter impregnated with *pseudomonas
putida* DK-1 (Park et al., 2001). Removal efficiencies in the range of 75-90% were observed for an inlet load-
ing ranging from 250-350 g h$^{-1}$ m$^{-3}$. They used a fiber
glass column with an inner diameter of 50 mm and
height of 200 mm. The pressure drop in the bed was
20-100 (Pa m$^{-1}$ of packing) and had limited impact on the
process efficiency. At the bottom of the reactor, a
perforated sieve plate was to support the medium. The
medium (wood chips) was impregnated with *pseudo-
onas putida* DK-1 and positioned over samples of
contaminated soil. The problems associated with the
filter were the difficulty in controlling the moisture and
pH in order to maintain the optimum environment
for the growth of degrading microorganisms and also
the relatively high rates of clogging and deterioration of
the medium. From these two studies. Biofilters have
more drawbacks compared to biotrickling filters. More-
over, biotrickling filters have been shown to reach
higher removal efficiencies for relatively low concen-
trations of VOCs and odorous compounds.

First ever reported laboratory scale BTF for the eli-
mination of nitrobenzene vapors was reported by Oh
and Bartha (1997). They used a stable microbial con-
sortium enriched by sewage sludge and immobilized
on dry perlite (the reactor occupied 59% (0.4 L) of the
total column volume (1.5L)). During the startup period
of four weeks, the inlet nitrobenzene concentration was
kept relatively low (<16 ppm) to avoid poisoning of
the culture, after which high and sustained nitrobenzene
elimination was observed with 80-90% degradation
for inlet concentrations ranging from 100 to 300 mg
m$^{-3}$ and an empty bed gas contact time of 21 seconds.
The resultant elimination capacity was of 50 g m$^{-3}$ h$^{-1}$
at a stream flow rate of 200 m$^{3}$ h$^{-1}$. This is a significant
removal rate that could lead to an economically viable
process. A nitrogen balance showed that 98% of the
removed nitrobenzene was converted into ammonia
while a small amount of nitrite was also produced.

Also, noteworthy is the study by (Sun and Wood,
1997). They immobilized a pure culture of *Burkholde-
tia Cepacia* PR1$_{23}$, a Tn5transposon mutant of *B.
cepacia* G4 that constitutively expresses the trichloro-
ethylene (TCE) degrading enzyme and toluene ortho-
monooxygenase (TOM). Aerobic biodegradation of
TCE only occurs through co-metabolism with the addi-
tion of a growth substrate (usually toluene, methane,
propane, phenol or ammonia). This is required to in-
duce the expression of the appropriate TCE-degrading
enzyme. However, the bacterium strain *Burkholde-
tia Cepacia* PR1$_{23}$, expresses toluene ortho-monooxygenase
at a constant rate, regardless of physiological demand.
This circumvents the problem of competitive inhibi-
tion of TCE oxidation by the usual inducers during the
growth phase. They used glucose as a carbon and energy
source and observed TCE eliminations up to 200 times
higher than previously reported 90% TCE removal at
an inlet loading of 2.4-100 mg TCE L$^{-1}$ together with
95 mg toluene L$^{-1}$ (Guo et al., 2001). As observed pre-
viously in other bioreactors for TCE aerobic cometab-
olism, rapid inactivation of the TCE-degrading enzyme
by TCE breakdown products (e.g. TCE-epoxide) still
remains to be resolved (Guo et al., 2001; McFarland et
al., 1992).

In the past few years, much research has been done
to improve the BTF technology (Valero et al., 2017;
Zhou et al., 2016; Tsang et al., 2015). Several success-
ful conversions of full-scale chemical scrubbers to bio-
trickling filters have been demonstrated (Gabriel and
Deshusses, 2003; Kraakman, 2003, 2001).

### 4.2 Application of BTFs for Odor Control

There are over 16,000 publicly owned treatment
works (POTW) in the United States serving 75 percent
of the total population (U.S. DHS, 2016). The POTWs
treat 32 billion US gallons (120 gigalitres) of waste-
water every day (EPA, 2014). Emission of objection-
able odors from these facilities is a major problem.
POTW off-gases contain a wide range of odorous
compounds, air toxics and volatile organic compounds
(VOCs). These include volatile sulfur compounds,
ammonia, toluene, chloroform, dichloro-
methane, and trichloroethylene (Lewkowska et al.,
2016; Zhou et al., 2016). H$_2$S is the principal odorous
component that causes nuisances even at volume frac-
tions as low as 8 ppb (Smet et al., 1998). In addition to
its unpleasant odor, H$_2$S gas is highly toxic (Roth,
1993). Continuous exposure to low (15-50 ppm) con-
centrations will generally cause irritation to mucous
membranes, and may also cause headaches, dizziness,
and nausea. Higher concentrations (200-300 ppm) may
result in respiratory arrest leading to coma and uncon-
sciousness. Exposure for more than 30 minutes at concen-
trations greater than 700 ppm have been fatal
(MSDS, 1996). Concerns about the odor nuisance to
the surrounding communities as well as the implemen-
tation of more stringent regulations are forcing POTWs
to treat their off-gases.

Controlling H$_2$S is usually achieved by wet or chem-
ical scrubbers. Chemical scrubbing in a packed tower
is an established technique and is effective at gas con-
tact times as short as 1.3-2 s (Gabriel and Deshusses,
However, chemical scrubbing suffers from important drawbacks such as high operating costs, generation of halo-methanes that are known air toxics and can contribute to global warming (Wenhai et al., 2016), and the requirement for hazardous chemicals that pose serious health and safety concerns. Hence, more research is being done to convert chemical scrubbers into biotrickling units.

Gabriel and Deshusses (2003) demonstrated the conversion of a chemical scrubber located at the Orange County sewerage treatment facility in California. The converted bioscrubber was 9.75 m high, 1.82 m in i.d., and made of fiber-glass reinforced plastic with a nominal packing bed height of 2.8 m and bed volume of 7.3 m³. The water trickling flow rate was 4.5 m³ h⁻¹ and the foul air was fed to the reactor at atmospheric pressure with an average flow rate of 16,300 m³ h⁻¹. The researchers operated a laboratory pilot unit under conditions similar to those expected at a publicly owned treatment work and tested selected packing materials for their sustainability for biotrickling filtration. The conversion consisted of (i) replacing the existing packing, which had a low interfacial area and was not suitable for microorganism attachment; (ii) replacing the liquid recycle pump with a smaller one; (iii) disconnecting the chemical feeds; and (iv) modifying controls of the reactor. Before startup, the reactor was impregnated with 0.8 m³ of activated sludge from wastewater-treatment plant. The pH started to decline after three days of operation to reach pH 2, in seven days after startup. The decline in pH (from the production of H⁺ and sulfate from the oxidation of H₂S) was correlated with the increase in H₂S removal efficiency. Acclimation lasted for about ten days, after which H₂S removal was more than 99% for H₂S inlet volume fractions ranging from 5 to 25 ppm and remained high for the rest of the operation.

Vikromvarasiri and Pisutpaisal (2016), studied the removal of H₂S in a biotrickling system using a new bacterial strain of obligately chemolithoautotrophic, Halothiobacillus neapolitanus NTV01 (HTN). The biotrickling filter column was made from glass (0.475 m inner diameter and 0.72 m height). The column was packed with randomly structured packing media (GEA2H Water Technologies GmbH) to its working height of 0.282 m. The packing media had a random structure and made from high-density polyethylene (HDPE) 12 mm beads, surface 859 m² m⁻³, and density 150 kg m⁻³. A mixed gas stream containing CH₄, CO₂, and H₂S in the air was supplied to the inlet of the biotrickling filter; and [H₂S] varied between 0-255 ppm (v/v). The system could completely remove 45 ppm H₂S within 70 min of operation. The removal efficiency was in the range of 95-98%, for 225 ppm (v/v) H₂S.

The removal of H₂S in high-performance biotrickling filters was investigated by Kim and Deshusses (2005) using a differential biotrickling filter. The filter was designed to reach high gas velocities through a miniature packed bed. A small differential filter was used in this study. It was filled with a single cube (64 mL) of open-pore polyurethane foam packing. The biotrickling filter was operated in a counter-current mode. The air flow was circulated in a closed loop from an 85 L Tedlar bag to the differential biotrickling filter by a 0.2 horsepower blower up to a maximum linear velocity of 3 m s⁻¹. Pure H₂S was injected into the differential biotrickling filter system using a 20 mL syringe. Within a short time, gas velocity was varied to determine its effect on H₂S elimination capacity. The H₂S elimination capacity (35 to 125 g m⁻³ h⁻¹) was achieved at a liquid trickling velocity of 1.5 m h⁻¹ and inlet [H₂S] was between 50 and 65 ppm. Interestingly, the addition of dissolved sodium sulfide (1-3.5 mg L⁻¹) resulted in reduced H₂S gas degradation at pH 1.9-2.1. H₂S is utilized by the microbial population as an S source. Hence, the addition of sodium sulfide competes with H₂S gas for biodegradation as the only ionic species (from dissolved H₂S and Na₂S), i.e. SH⁻, are metabolized by the bacteria.

5. ADVANTAGES AND LIMITATIONS

Conventional waste air treatment technologies such as absorption, adsorption, chemical scrubbing, and oxidation are generally not cost effective for the treatment of VOCs in dilute waste air streams (Kamal et al., 2016). Moreover, hazardous by-products such as NOₓ, CO, dioxins, and furans are also generated whose treatment further adds to the operating cost. In the case of other conventional techniques like catalytic oxidation or adsorption, the development of specific adsorbents and the regeneration of catalysts adds to the total cost (Kamal et al., 2016). Also, these treatment techniques produce hazardous by-products also and require the addition of chemicals or fuels that may require further treatment or disposal, thereby creating additional environmental problems (Shareefdeen and Singh, 2005).

As compared to other biological removal techniques, biotrickling filters pose certain advantages over other methods. While they are cost effective (See Table 1) and have a low-pressure drop, but as is the case in other biofilters, biological operating parameters such as pH and moisture can be controlled by continuous trickling. Moreover, biotrickling filters are capable of treating acid degradation products of VOCs which
gives an edge over other methods (Lu et al., 2001).

Despite many advantages, biotrickling filters also have some disadvantages. With the continuous supply of nutrient, the biofilm accumulates which leads to clogging. The accumulating biofilm is one of the major obstacles faced at present for the proper functioning of biotrickling filters over extended periods. Clogging increases pressure drop across the reactor thereby decreases pollutant removal rate (Okkerse et al., 1999; Cox and Deshusses, 1999). Biomass growth can be controlled by reducing the overall rate of biomass accumulation by either reducing the specific growth rate or increasing death and lysis (Alonso et al., 1998). Other options include increasing predation, washing-out or periodically removing the excess biomass (Cox and Deshusses, 1999).

Long acclimation periods, complex construction and operation, and secondary waste streams are some of the other problems faced by the biotrickling filters. Moreover, the presence of microorganisms in the media has raised concern over their potential release into the treated air and resultant human exposure to pathogens (Ottengraf and Konings, 1991). Table 1 lists the estimated capital costs, footprint and treatment capacity of biotrickling filters of various sizes presents. Table 2 presents the typical characteristics of biotrickling filters. Table 3 presents the recent investigations on biological treatment techniques for VOCs and odorants. Table 4 presents a comparison between various VOC control techniques including cost estimation.

### 6. Future Research and Development Needs

Although much work has been done on biotrickling filters since 2005, almost all are primarily focused on the microbiology of pollutant-degrading microorganisms and the methods to control the biomass accumulation in the biofilter. The fundamental principles of biotrickling filters still need to be understood more clearly. Key questions to be addressed are mainly concerned with the complex ecology of biofilms. Future research work should concentrate on understanding the fundamentals of the degradation process through situ analysis using modern tools in biotechnology. This is essential to establish baseline information (presently not available) for logical reactor design and optimum process operation. In particular, studies are needed to understand the overall role of secondary processes (i.e. those processes not directly associated with the elimination of the primary pollutant) and how these can be controlled in practice. In the future, the ability to control the ecology of biofilms in BTFs may enable optimal and limited biomass growth, so that

### Table 2. Typical characteristics of biotrickling filters [Source: Deshusses and Cox (1999)].

| Order | Characteristics                                      | Values                                      |
|-------|------------------------------------------------------|--------------------------------------------|
| 1     | Biotrickling filter bed height                       | 1.5 m                                      |
| 2     | Biotrickling filter cross section area               | 1.3-0.0 m²                                 |
| 3     | Air flow treated                                    | 100-1,000,000 m³ h⁻¹                      |
| 4     | Packing void volume (a)                              |                                            |
|       | • Plastic rings, foam, random or structured packing  | 90-95%                                     |
|       | • Lava rock                                          | ~50%                                       |
| 5     | Empty bed gas retention time (b)                    | 2-60 s                                     |
| 6     | Pressure drop                                       | <100 Pa per m bed depth                   |
| 7     | Operating temperatures                              | 15-50°C                                    |
| 8     | Trickling rates (c)                                 | 0.01-10 m h⁻¹                              |
| 9     | Liquid dilution rate (d)                            | 0.1-2 day⁻¹                                |
| 10    | Usual pH of the recycle liquid                      | ~7                                         |
|       | • Removal of VOCs or compound difficult to degrade   |                                            |
|       | • Removal of H₂S                                    | 1-2                                        |
| 11    | Inorganic nutrient supply (N, P, K, traces)         | Usually 0.05 to 1 times the amount calculated using biodegradation stoichiometry |
| 12    | Inlet pollutant concentration                        |                                            |
|       | • VOCs                                               | 0.01-10 g m⁻³                             |
|       | • Odors                                              | 500-50,000 odor units                     |
| 13    | Typical pollutant removal efficiencies               | 60-99.9 + %                               |

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(a) Value at reactor startup; over time, biomass growth will decrease bed porosity, typically by 10-30%
(b) The empty bed gas retention time (EBRT) is defined as the bed volume / air flow
(c) Trickling flow rate / bed cross section area
(d) Liquid feed rate / recycle liquid volume
### Table 3. Recent investigations of biological treatment techniques for VOCs and odorants.

| Order | Scale    | Process type                        | Type bed                  | Bed volume | Pollutants treated and inlet gas concentration | Gas flow rate and empty bed residence time (EBRT) | Removal efficiency | Inoculation | Reference                          |
|-------|----------|-------------------------------------|---------------------------|------------|------------------------------------------------|--------------------------------------------------|-------------------|-------------|------------------------------------|
| 1     | Laboratory | Biotrickling                        | Polypropylene pall rings  | 2.8 L      | 12-100 ppm H₂S, 450-700 ppm CH₃OH, balance air | 7 L min⁻¹, 24 s                                    | >99% (H₂S), >95% (CH₃OH) | Yes (biomass from a H₂S-degrading biotrickling filter) | Jin et al., 2007 |
| 2     | Laboratory | Biotrickling                        | Polypropylene pall rings  | 10 L       | 170 ppm H₂S, 2.2 g m⁻³ toluene, balance air    | 1 m³ h⁻¹, 36 s                                    | 100% (H₂S), 25-75% (toluene) | Yes (biomass from a toluene-degrading biotrickling filter) | Cox and Deshusses, 2001 |
| 3     | Full      | Biofiltration                       | Polyurethane foam         | 1000 L     | 4.81-27.48 ppm H₂S, balance air               | 60 m³ h⁻¹, 60 s                                    | >90%              | Yes (culture from odor treatment bioreactor) | Li et al., 2013 |
| 4     | Laboratory | Biofiltration                       | Granular activated carbon | 1 L        | 100-4000 ppm H₂S, balance air                 | 15 L h⁻¹                                          | >98%              | Yes (culture from concentrated latex wastewater) | Rattanapan et al., 2009 |
| 5     | Bench     | Biofiltration                       | Compost and wood chips    | 12 L       | 0.31-1.44 ppm dimethyl sulfide (DMS)           | 1.5-2.5 L min⁻¹, 360 s                            | 71%               | Yes (seed culture of *B. sphericus* isolated from garden soil) | Giri and Pandey, 2013 |
| 6     | Laboratory | Biofiltration                       | Sugarcane bagasse         | 0.98 L     | 3.13 ppm benzene                              | Superficial velocity of 30.6 m h⁻¹                 | 100%              | Yes (culture of *Pseudomonas* sp. NCIMB 9688) | Sene et al., 2002 |
| 7     | Laboratory | Biotrickling                        | High-density polyethylene | 0.5 L      | 0-255 ppm H₂S                                  | 0.5 L min⁻¹, 60 s                                 | 95-98%            | Yes (sludge from a full-scale activated sludge system) | Vikromvarasiri and Pisapatpaisal, 2016 |
| 8     | Laboratory | Biofiltration                       | Compost and lava rock     | 4 L        | 296.88-857.66 ppm n-butanol                    | 4 L min⁻¹, 60 s                                   | >73%              | Eshraghi et al., 2016                |
| 9     | Full      | Bioscrubbing (absorption + aerobic biooxidation) + separate biofiltration | Alkaline solution for scrubbing, compost for biofiltration | Natural gas: 2000 ppm H₂S | 322,000 nm³ d⁻¹ | Indigenous | >99.8% (H₂S) | Benschop et al., 2002          |
reactor stability can be assured over a very long period (greater than 150 days) (Zhou et al., 2016). Additional research is needed for better understanding of the kinetic relationships for pollutant biodegradation. Particularly, understanding the biodegradation of mixtures of pollutants, the role and impact of oxygen and ancillary nutrients on the rate of biodegradation and the biomass yield, and the influences of various stresses, such as changing conditions and mass transfer limitations, is important. This, together with a number of pilot-scale operation and demonstration of techno-economic viability, would transfer this technology from lab to the field (Valero et al., 2017; Cox and Deshusses, 1999).

7. CONCLUSION

This review provides an overview of the biotrickling reactors being used for the treatment of various VOCs and odor laden waste gas streams, limitations of the already existing BTFs, future prospects, and avenues which can be explored for a better understanding and development. A summary of some of the other biological treatment methods has also been provided before describing the BTF in detail. Clearly, the design of the BTF still requires improvement, and demonstration of significantly better performance compared to existing designs. Further, developments of innovative combined bioreactor designs remain a high priority. Development in reactor design and development will require similar advances in understanding the fundamentals of bioprocesses so that a more logical, creative and focused approach to BTF design can be implemented.

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