

Volatile Organic Compounds Removal Methods: A Review

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Received 2012-09-24, Revised 2012-10-04; Accepted 2012-10-17

ABSTRACT

Volatile Organic Compounds (VOCs) are among the most toxic chemicals which are detrimental to humans and environment. There is a significant need of fully satisfactory method for removal of VOCs. There are several methods including physical, chemical and biological treatments available to remove VOCs by either recovery or destruction. The aim of the present study is to summarize the available methods for VOC removal; trying to find a promising method among the available techniques. A wide range of VOCs can be treated biologically in which it offers advantages over more traditional processes including lower operating and capital costs and a smaller carbon footprint. However, due to a complex nature and diversity of VOCs it is hard to find a simple and promising method. Treatment still requires more research to solve the associate problems with available VOC elimination techniques.

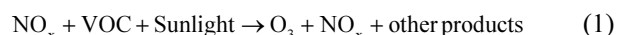
Keywords: Volatile Organic Compounds (VOCs), Removal Techniques, Bio-Treatment

1. INTRODUCTION

Volatile Organic Compounds (VOCs) are man-made and/or naturally occurring highly reactive hydrocarbons. World Health Organisation defined VOCs as any organic compound whose boiling point is in the range from (50-100°C) to (240-260°C), corresponding to having saturation vapour pressures greater than 102 kPa at 25°C (ISO16000-6, 1989). Many types of VOCs are toxic or even deadly to humans and can be detrimental to the environment. Therefore a multitude of definitions exist globally depending on the context frame used by different organisations such as United Nations Economic Commission for Europe (UNECE) and U.S. Environmental Protection Agency (EPA).

Natural origins of VOCs include wetlands, forests, oceans and volcanoes with the estimated global VOCs biogenic emission rate at about 1150Tg/yr (Guenther *et al.*, 1995). A majority of VOCs are created from anthropogenic activities consisting of manufacturing industries, petrochemical industries and vehicular emissions (EPA,

2012). Most VOCs are photo-chemically sensitive and when exposed to oxides of nitrogen and sunlight, would form ozone and other products as represented in Equation 1:



The reactions represented by Equation 1, involve VOCs oxidation with NO_x, hydroxyl catalysing some of the key reactions including other chemical compounds. Ozone formation is thus primarily driven by available nitrogen oxides and VOCs. Resulting ground level ozone formation and carcinogenic smog is the main cause of concern. As the wide range of VOCs implies a broad range of reaction rates, VOCs are capable of long range distribution and accumulation in components of environment. Efficiency of ozone formation varies with the ratio of nitrogen oxides to VOCs with higher NO_x -FQUOTE NO_x emissions resulting in reduced ozone production efficiency (Finlayson-Pitts and Pitts, 1986; 1999; Chanin, 1993). The isopleth plot in Fig. 1 illustrates the relation between O₃, NO_x and VOC with clear regimes of different O₃-NO_x-VOC-sensitivity.

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Dependencies vary on assumptions and conditions used during plot generation. When NO_x is limited, O_3 increases with increasing NO_x with relatively minimal response to increasing VOC. Thus VOC control could be less effective in reducing O_3 in NO_x sensitive areas e.g., rural areas and downwind from the source (Finlayson-Pitts and Pitts, 1986).

In the limited VOC regime to the left of the ridge line, lowering VOC concentrations result in lower peak ozone concentrations. Contradictory lowering NO_x at constant VOC concentrations predicts increased peak ozone concentrations to the ridge line. This is due to more hydroxyl radicals available to react with VOCs due to reduced NO_x -hydroxyl reactions. Only past the ridge line does the ozone concentration begin to decrease. Ozone concentrations are also lowered by proportionately decreasing both VOCs and NO_x concentrations simultaneously (Dodge, 1977; Finlayson-Pitts and Pitts, 1986). Thus, **Fig. 1** emphasises there is greater benefit in reducing VOC concentrations over NO_x for optimal ozone control strategies closer to the source of emission.

VOCs are not only outdoor pollutants as high concentrations have been recorded indoors as well. Indoor sources include solvents used in the production and maintenance of building materials, furnishings or equipment e.g., paint, carpets, plastics and photocopying machines. The National Health and Medical Research Council (NHMRC) interim national indoor air quality recommends a maximum hourly average total VOC level of $500 \mu\text{g m}^{-3}$, where each compound should not contribute more than 50% of the total (NHMRC, 2002). In a study investigating VOC emissions from office furniture, a range of major VOCs emitted over time was found to be in excess of the NHMRC recommendation. Formaldehyde was found to be emitted in high concentrations from furnishings using reconstituted wood-based panels e.g., particleboards, medium-density fibreboard (NHMRC, 2002). **Table 1** details the results for major VOCs emitted by new office furniture.

Indoor levels of VOC can be regulated by selecting low-emission materials for furnishing, cleaning and construction. This approach is impractical as many of the materials emitting VOCs are considered standard fixtures and attempting to replace them could prove costly. Furthermore, proper ventilation with outdoor air can help in managing air pollutants indoors by means of dilution. However with the increasing number of high-rise buildings developing in major cities, the problem of VOC pollution becomes both an indoors and outdoors issue that requires a permanent solution (Guo *et al.*, 2004; Alvarez-Hornos *et al.*, 2008).

Various traditional VOCs removal and elimination technologies exist. In the present study we summarize the available methods for VOCs removal; try to find a promising technique.

1.1. Common VOCs Removal Methods

There are physical, chemical and biological treatments available to remove VOCs from air by either recovery or destruction. The contaminated liquid can be aerated either via packed tower aeration or mechanical surface aeration with VOC emissions captured and treated. Traditional primary removal solutions for VOC-contaminated liquids are liquid-phase activated carbon adsorption or air stripping. For soil VOC contamination remediation, Soil Vapour Extraction (SVE) is used (EPA, 1991; Tchobanoglous and Kreith, 2002). These methods allow for the extraction of VOCs into another phase.

In liquid-phase activated carbon adsorption, the treated liquid is put in physical contact with activated carbon to allow dissolved organic contamination to bind to it. The activated carbon can be either regenerated or removed after treatment (DOE, 1994). Reactors commonly used for this process are the fixed bed and the pulsed/moving bed. When dealing with halogenated VOCs and pesticides, this carbon method has limited effectiveness. Economical and logistical issues would arise from disposing or decontaminating the spent carbon, hence carbon adsorption is applied more effectively for "polishing" post-treated liquid discharges with low VOC concentrations (EPA, 1990; 1993; 1995). Similarly VOCs in air emission can be treated with activated carbon by pumping it through activated carbon packed bed reactors. However, problems with spent carbon are the same as the liquid-phase carbon method.

Common permanent solutions to treat extracted VOC emissions would be through oxidation by thermal, internal combustion engine, catalytic or UV. Essentially VOCs are broken down into less harmful compounds such as carbon dioxide, water and hydrochloric gas.

Thermal oxidation units are generally single chambers with ceramic blanket refractory lining the oxidisers, equipped with a propane or natural gas burner and a stack. In the chamber, burner capacities range between 0.4 to 2 (mil BTUs)/hr with operating temperatures from 760 to 870°C and a maximum gas residence time of 1 sec. The internal combustion engine works in the same way; however, it is adapted to high VOC concentrations to allow the organic compounds to be used as fuel. Auxiliary fuel is only added to enhance oxidation. To reduce the need for auxiliary fuel, air to air heat exchangers can be used to transfer heat from the exhaust gases to the incoming feed. In treating halogenated VOCs, the exhaust stream would require a gas scrubber to control the acid vapour (Benitez, 1993). Oxidation is deemed inefficient for low VOC concentrations in the range of between 0.1 to 10g Mm^{-3} as continuous oxidation is difficult to maintain unless with more supporting fuel (Rao, 2007).

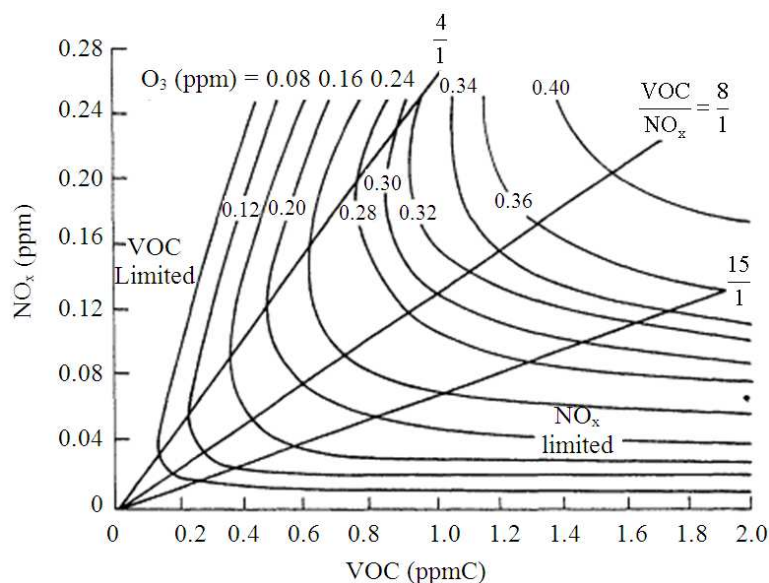


Fig. 1. General ozone isopleth used in EPA's, Empirical Kinetic Modelling Approach (EKMA) modelled against ratios found in cities (Dodge, 1977)

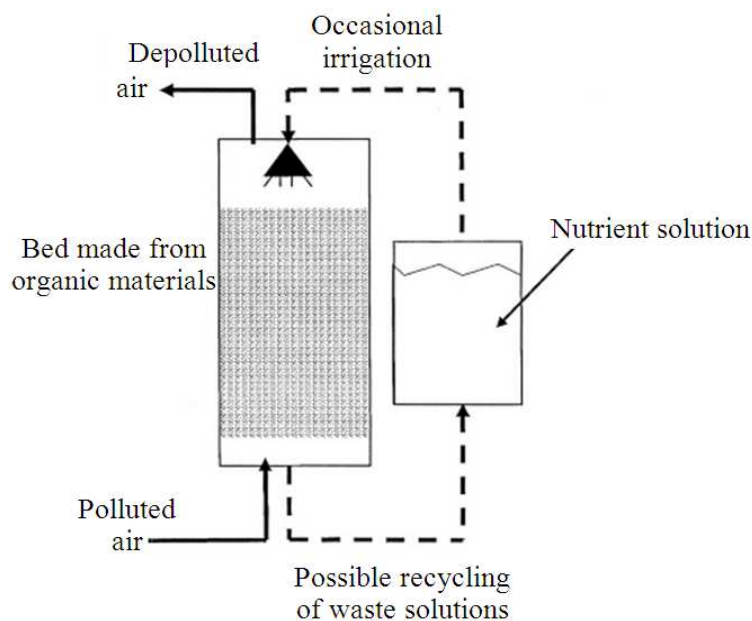


Fig. 2. Schematic diagram of a biofiltration unit (Delhomenie and Heitz, 2005)

Catalytic oxidation involves the addition of a catalyst to a thermal oxidiser to accelerate oxidation via the adsorption and reaction of oxygen and organic compounds on the catalyst surface. This lowers the reaction temperature to a range between 320 to 540°C compared to that required in a conventional thermal

oxidiser. The direct pre-heating of feed to the reaction temperature is required to initiate catalytic oxidation when passed through a solid catalyst bed. Catalysts may contain various metal oxides such as chromium oxide or nickel oxide and may even contain noble metals e.g., palladium.

Table 1. Formaldehyde and VOC concentrations and emission factors from new office furniture (Brown, 1999)

VOC	Concentration ($\mu\text{g}/\text{m}^3$)		Emission Factor ($\mu\text{g}/\text{m}^2\cdot\text{h}$)	
	4 h	24 h	4 h	24 h
Formaldehyde	190	230	240	290
Freon 22	42	700	530	890
Methanol	4000	2700	5000	3400
Acetone	450	254	560	320
Butanal	290	220	360	280
Ethylacetate	660	580	830	730
Methyl isobutyl ketone	200	210	250	260
Hexanal	28	23	35	29
m- and p-Xylene	59	48	74	60
Cyclohexanone	55	54	69	68
Phenol	180	90	230	110
Total VOC	970	820	200	1030

Thermal oxidisers can be used as pre-treatment to catalytic units for high VOC concentrations (EPA, 1987; Benitez, 1993; Rao, 2007; Berenjjan and Khodiev, 2009; Berenjjan *et al.*, 2011).

UV oxidation method has been used to oxidise organic and explosive compounds in wastewater. Strong chemical oxidisers directly react to the contaminants with UV photolysis achieved through synergistic UV irradiation combined with ozone and/or hydrogen peroxide (Christman and Collins, 1990; EPA, 1990). Generally low pressure 65W lamps are used for ozone treatment systems and 15-60W lamps are used in hydrogen peroxide treatment. Final products of UV oxidation are carbon dioxide, water and salts. Although, able to treat a wide range of VOCs, certain contaminants may be volatilised rather than destroyed e.g., TCA. In such a case, post treatment by activated carbon adsorption may be required (Zappi *et al.*, 1992).

Bio-treatment of VOCs is a relatively less established VOC removal method; however, it offers possible advantages over more traditional processes with lower operating and capital costs and a smaller carbon footprint. This is due to lower energy requirements as microorganisms are used to metabolise organic compounds at ambient temperature instead being dependent on heat or radiation. Since 1923, gas bio-filtration has been applied to treat various VOC exhaust contaminants (Leson and Smith, 1997; Wiczorek, 2005). Biological methods include bio-filters, bio-trickling filters, bio-scrubber, suspended growth reactors and membrane bio-reactors (Doble and Kumar, 2005). Key parameters across bio-systems are identified as moisture content in the medium, temperature, pH and availability of essential and non-carbon nutrients (Leson and Winer, 1991; Swanson and Loehr, 1997; Sidkar and Irvine, 1998). As a wide range of VOCs are biodegradable, they can be treated biologically with

organisms matched to the type of VOC to be destroyed. Some easily biodegradable VOCs are esters, benzene, toluene and phenols. Due to future growth potentials for the use of biofiltration methods, these methods are detailed and expanded upon in the following sections.

1.2. Bio-Treatment

Bio-technologies effectively and economically treat contaminated air flows from 60 to 150,000m³/h with lower VOC concentrations than most other available technologies. This advantage enables it to be applied to smaller scale projects such as possibly retro-fitting building ventilations with a compact VOC treatment system. Existing bio-treatment of VOCs used in gas treatment revolves around the concept of bio-filtration. There are various other bio-filtration processes available such as Bio-Trickling Filters (BTFs), bio-scrubbers and newer technology e.g., bio-membrane bioreactors (Cox and Deshusses, 1998; Doble and Kumar, 2005). Bio-filtration is generally based on two principals: firstly, the transfer of pollutants from the air feed to the support medium. Secondly, the contaminants are bio-catalysed to biomass, carbon dioxide, water and other by-products (Miller and Allen, 2004). Schematic diagram of a biofiltration unit is shown in **Fig. 2**.

The traditional bio-filtration system is essentially a fixed-bed bioreactor where the bio-catalyst or microorganism is immobilised to an inert supporting medium to develop biofilms. Ideal packing bed materials have high void fraction, light weight, low pressure drop, hydrophilic and low bulk density. Occasional irrigation of nutrient solution onto the packing bed and humidity control maintains the biofilms. Polluted air is fed from the bottom through the porous biologically-active media, where pollutants diffuse into the aqueous-phase or are absorbed directly by the biofilm with treated air emitted

from the top. Unfortunately, VOCs containing sulphates, chlorides and nitrates have acidic by-products that can adversely affect the ecosystem and reduce the effectiveness of the bioreactor (Garner and Barton, 2002).

The biotrickling filter has gas flowing through a fixed bed with microorganisms immobilised through it. Ideal packing material is similar to the traditional set up. Nutrient solution is continuously irrigated from the top, collected at the bottom and recycled back up. Co-current or counter-current flow between the gaseous and aqueous phases was found to have no influence on the degradation performance (Cox and Deshusses, 1998). The biotrickling system allows for more efficient removal of soluble VOCs from the air stream as well as enabling feedback control for nutrient levels and pH. As there is a constant feed of nutrients to the system, excess biomass builds up quickly and can lead to performance loss and pressure drops from uneven biomass distribution (Cohen, 2001).

Bioscrubbers consist of an absorption tower and a bioreactor. In the absorption tower, gas phase contaminants are diffused into an aqueous solution via means of counter-current gas-liquid flows through inert packing. The washed gas is emitted from the top and the contaminated liquid pumped towards an aerated bioreactor. The microorganism or activated sludge in the reactor is suspended in a nutrient-rich media and residence time for treatment varies according to the type and concentration of VOCs in the feed. After completed degradation of contaminants, the medium is filtered and biomass left to sediment with portions being recycled through the bioscrubbing process again. An aqueous system with no high pressure drops allows for more evenly distributed temperature, nutrient and pH controls. System limitations are due to the narrow band of VOCs treatable. This range is limited by water solubility of contaminants thus only applying to organics with low Henry's coefficient (<0.01) at low concentrations ($<5\text{g/m}^3$). Compared to other bioreactors considered, the bioscrubber requiring two sub-units could be a space limitation; especially when multiple systems in series or parallel are required to process higher VOC volumes. Bioscrubbers, however, are not as well-utilised within the biotechnology market compared to biofilters and biotricklers (Swanson and Loehr, 1997). To treat less-soluble pollutants, silicon oil could be added to aqueous systems as an emulsifying agent to favour the VOC mass transfer from gas to liquid phases.

Finally, membrane bioreactors involve a membrane partitioning the liquid and gas phases. The nutrient and biomass growth is within the liquid side whilst the gaseous

contaminants are pressurised to induce diffusion over the partition into the aqueous solution. The membranes can be further treated to have various advantageous properties e.g., hydrophobic and microporous. Membrane bioreactors for VOC treatment still require more research to minimise fouling and high cost (Artiga *et al.*, 2005).

1.3. Operating Parameters

The operations of biofilters involve a series of steps beginning with the transfer of the pollutant air to the aqueous phase, followed by its adsorption onto the medium or absorption into the biofilm and then biodegradation of VOCs within the biofilm (Kumar *et al.*, 2011). The biodegradation of pollutants by biofilms in biofiltration system is a combination of physicochemical and biological phenomena. Basically these three mechanisms are responsible for the transfer and subsequent biodegradation within the bed (Doble and Kumar, 2005; Mathur *et al.*, 2006).

Governing control parameters across all bioreactors includes packing material, air flow rate, temperature (Darlington *et al.*, 2001), pH, humidity and non-carbon nutrients e.g., nitrogen, phosphate and potassium (VanLith *et al.*, 1997; Devinny *et al.*, 1999). However, as the type of microorganism used is dependent on the type of VOC to be destroyed, treating a wide range of VOCs in indoor air would result in a diverse ecology of mixed fungal and bacterial populations (Khammar *et al.*, 2005). This diversity creates complex interactions within the bioreactors which require research into the ecology of the biomass to optimise overall management of the ecosystem.

1.4. Packing/Filter Bed

As the filter bed physically supports the entire biosystem, some of the criteria required to achieve optimal conditions are high-specific surface area which is excellent for microbial activity, decent water retention capacity and high porosity. Basic traditional organic materials used for packing are peats, composts and soils because of their stability, low cost and effectiveness (Doble and Kumar, 2005). Modern approaches enable beds to have evenly-distributed rigid structures preventing bed compaction. Furthermore, biocatalysts can be immobilised within a porous structure (polymer beads) to allow for easier maintenance and preservation of specific microorganisms (Lu *et al.*, 2002). Pressure drop of the filter bed is an important biofiltration parameter which drastically effects on operating cost. Filter beds including small particles have high specific

surface area which in turn make them suitable for microbial activity and increase their resistance to air flow (Kumar *et al.*, 2011).

1.5. Flow Rate

In general, overall efficiency of biofilters is related to two main microscopic processes. The first process is transfer rate of the VOCs from gas phase to biofilm and the second one is biodegradation rate of VOCs (Delhomenie and Heitz, 2005). The air flow rate is an important parameter which can affect both of these processes. In fact, by increasing the flow rate, the rates of VOCs transfer to the biofilm and degradations decrease. This can be explained by the fact that in extremely high air flow rates, contact times between microorganisms and gases are too short and the biodegradation reaction cannot be completed (Cox and Deshusses, 1997). The Empty Bed Residence Time (EBRT) is defined as a time that parcel of air will remain in the biofilter and calculated as the empty bed filter volume divide by the air flow rate. Several studies have shown that to improve biofiltration performance and better VOCs removal efficiency, the EBRT should be greater than the time required for diffusion process (Doble and Kumar, 2005; Kumar *et al.*, 2011). It has been reported that by increasing the size of particles in the filter beds, the resistance of filter beds decrease towards gas flow (Clark *et al.*, 2004).

1.6. Temperature

Operating temperature is one important factor that affects biofilter performance. The microorganisms that most effectively degrade VOC compounds in biofilters are mesothermic and their optimum activity temperatures are between 30 and 40°C (Clark *et al.*, 2004). In fact, the proper temperature increases the rate of biofilm development and biomass accumulation (Yang *et al.*, 2010). Deeb and Alvarez-Cohen (1999) indicated that in the optimum range of operating temperature, the degradation performance on biofilter can increase 2-fold by 10°C increase in temperature. This can be explained by the fact that by rising the temperature, VOCs and O₂ solubilise in water decrease and diffusion transfer increases. Due to exothermic nature of the biodegradation reactions, the variation of temperature in the filter bed is a consequence of microbial activity (Hwang *et al.*, 2002).

1.7. pH

For most bioreactors, pH has a significant impact on biofiltration efficiency. The pH of the biofiltration

medium can have direct effect on microorganisms and microbial enzymes and also indirect influence on the availability of required nutrients. In fact, depending on the type of microorganism the pH has significant effect on the biofiltration efficiency (Clark *et al.*, 2004; Delhomenie and Heitz, 2005). Fungi are able to grow under both neutral as well as acidic conditions and are metabolically active over a wide pH range between approximately 2 and 7. Whereas bacteria are usually considered to be less tolerant to pH fluctuations and require a near neutral environment for their activity (Kennes and Veiga, 2004). Due to neutrophilic behaviour of the most of microorganisms in biofilters, maximum degradation of VOCs would usually be achieved at pH around 7 (Kumar *et al.*, 2001). Veiga *et al.* (1999) studied the effect of pH on alkyl benzene degradation (between pH 3.5 and 7.0) and found that alkyl benzene degradation increases with pH increase. Lu *et al.* (2002) reported the maximum degradation for BTEX between pH values of 7.5 and 8.0. However, the ideal pH of the biofilter medium depend on the pollutant being treated and the characteristics of the microbial ecosystem.

1.8. Moisture

The moisture content of the filter bed plays an important role in biofilter performance because microorganisms need water to attain their metabolic activity (Kumar *et al.*, 2011). Availability of excess water hinders the transfer of oxygen and hydrophobic pollutants to the biofilm, leading to the development of anaerobic zones within the bed. It also leads to the reduction of the specific surface available for gas/biofilm exchanges, which in turn, causes bed compaction and increasing the pressure drop. Too low bed moisture content leads to bed desiccation and gas flow channeling, which particularly affects the microflora (Delhomenie and Heitz, 2005). The support of microbial populations sufficient to reduce VOCs requires that moisture levels in the filter medium be maintained between 40% and 70% (w.b.) (Clark *et al.*, 2004). The main factors affecting the bed moisture content are gas flow rate through the bed, water holding capacity of the filtering materials, reaction exothermicity and inlet gas relative humidity (Devinny *et al.*, 1999).

1.9. Microorganism

Microorganisms including bacteria and fungi are used in VOCs biodegradation methods. Researchers have indicated that heterotrophic bacteria and fungi are the main microorganisms used in elimination of VOCs (Delhomenie and Heitz, 2005). However, the main

advantage of fomenting fungal rather than bacterial growth for the biofiltration of hydrophobic pollutants is their ability to degrade the substrates under extreme environmental conditions regarding pH, low water content and limited nutrient concentrations (Kennes and Veiga, 2004). Generally, heterotrophic microbial strains used in VOC bioreactors, can bio-catalyse VOCs via two pathways: (1) consuming organic compounds in the course of catabolic pathway for energy or (2) using VOCs as a carbon source for anabolic processes. Species of *Pseudomonas*, *Candida*, *Mycobacterium*, *Alcaligenes*, *Exophiala*, *Acetobacter*, *Fusarium*, *Cladosporium*, *Rhodococcus*, *Aspergillus* and *mucor* are some of microorganisms which identified and used for degradation of VOCs by biofiltration (Marek *et al.*, 1999; Diehl *et al.*, 2000; Christen *et al.*, 2002; Qi *et al.*, 2002; Woertz *et al.*, 2002). The filter bed inoculation depends on both the nature of the filtering materials and the VOC biodegradability level. Generally, a biofilter contains between 10^6 and 10^{10} cfu of bacteria and actinomycetes per gram of bed, respectively. It can also contains 10^3 to 10^6 spores of fungi per gram of bed (Delhomenie and Heitz, 2005).

1.10. Biomass

Biofilm is a complex aggregation of microorganisms (aerobic, anaerobic and facultative type bacteria, fungi, algae and protozoa) which attach themselves on the surface of the packing media and forms a biological film or slim layer with a viscous slimy structure (Kumar *et al.*, 2011). There are three main biological processes that occur in the biofiltration systems, namely: attachment of microorganisms, growth of microorganisms and decay and detachment of microorganisms (Yang *et al.*, 2010).

Assuming the biofilm is built on water, Henry's Law represented in Equation 2 roughly estimates maximum concentrations of aqueous pollutant (C_{aq}^*) available to be bio-catalysed:

$$C_{aq}^* = \frac{P_i}{H_i} \quad (2)$$

where, p_i represents the partial pressure of a specific pollutant in gaseous phase and H_i is Henry's constant coefficient of the pollutant. This model of estimation, however, does not in fact express the transport and reaction processes of the biomass itself. Water-insoluble compounds such as α -pinene have been

found to be significantly removed by biofilm (VanGroenestijn and Liu, 2002). Upon comparison, the α -pinene partitioning of biofilm containing biomass and organics was also found to be greater than biofilm in water (Miller and Allen, 2004).

As the dominant carbon source, the VOC gas supply for the biotrickling filter creates a gradient of high specific biomass growth rates starting from the feed inlet and tapering off towards the outlet (Song and Kinney, 1999). In a gas-phase biofilter, biomass accumulation rate is equal to the biofilm growth rate minus the biofilm decay rate resulting from hydraulic scouring or detachment. Thus, allowing for general modelling of biofilm thickness over time to be calculated from bacterial growth and decay as shown in Equation 3 and 4 (Alonso *et al.*, 1997; 1998):

$$\frac{\partial L_f}{\partial t} = (r_d D_w \frac{\partial C_f}{\partial x} |_{x=L}) \frac{Y}{X_f} - L_f b \quad (3)$$

$$b = b_s + b_d \quad (4)$$

Where:

L_f = Biofilm thickness

r_d = Ratio between VOC diffusivities in biofilm and water

D_w = Contaminant diffusivity in water

C_f = VOC concentration in biofilm

Y = Yield coefficient

X_f = Film bacterial density

b = Specific shear/decay coefficient

b_d = Specific decay coefficient

b_s = Specific shear rate

Other factors such as oxygen availability, carbon source and biofilm activity all interact to affect the biofilm decay rate. Factors contributing to shear force could result from calculating in fluid flow or particle attrition as well. This general model of prediction would enable biomass controls to be set in place to prevent a drop in efficiency from excess biomass. There are five methods of control including physical, chemical, biological, improved biofilter designs and improved operation modes. Most physical treatments involve bed stirring or backwashing to drain excess accumulated biomass. Chemical controls can be put on the carbon and nutrient sources to starve the biofilm. Periodical starvation does not affect removal performance in biotrickle reactors as microorganisms degrade their own extracellular polymers for energy (Zhang and Bishop, 2003). Chemical washing of the biofilter to control

biomass could have a detrimental effect on performance and requires time to reacclimatise. It must be noted that to achieve higher bio-conversion efficiency, the bio-filter should be exposed to low VOC concentrations initially to allow culture adaption to utilising VOCs as a carbon source. Following which, the organic loading can be increased over time.

1.11. Overview of Using Biological Treatments

As previously detailed, the four most common types of biological treatment units are biofilter, biotrickling filter, membrane bioreactor and bioscrubber. These systems have differences in their complexity, process design, equipment dimensions and working parameters, but they have unique advantages which make them proper techniques to remove wide range of compounds. Biofilters have advantages such as low initial investment and subsequently minimized operating cost, degradation a wide range of components, easy to operate and maintain, no production of unnecessary waste streams and low pressure drop (Yang *et al.*, 2010). Biotrickling filters have less operating and capital constraints, less relation time/high volume through put and capability to treat acid degradation product of VOCs (Cox and Deshusses, 1998). In membrane bioreactors there are no moving parts, process scale up is more easy and flow of gas and liquid can be varied independently, without the problems of flooding, loading or foaming (Artiga *et al.*, 2005). Bioscrubbers need relatively smaller space requirements and able to deal with high flow rates and severe fluctuations. Other advantages of using this system can be mentioned as operational stability, better control of operating parameters and relatively low pressure drop (Swanson and Loehr, 1997).

Applications of biological systems to remove VOCs have some limitations. For example, limitations of using biofilters are less treatment efficiency at high concentrations of pollutants, extremely large size of bioreactor challenges space constraints to require close control of operating conditions and limited life of packing and clogging of the medium due to particulate medium (Delhomenie and Heitz, 2005). Accumulation of excess biomass in the filter bed, requirement of design for fluctuating concentration, complexity in construct and operation and secondary waste stream are among the disadvantages of using biotrickling filters (Cox and Deshusses, 1997; Doble and Kumar, 2005). Limitations of application of membrane bioreactors to removal VOCs are high construction costs, long-term operational stability and possible clogging of the liquid channels due the formation of excess biomass (Doble and Kumar,

2005). Some disadvantages of using bioscrubbers are treatment of only water soluble compounds, generation of liquid waste, need of extra air supply, need of excess sludge disposal and operation/maintenance complications (Kumar *et al.*, 2011).

2. CONCLUSION

Based on the present study, biological methods (biofilter, biotrickling filter, membrane bioreactor and bioscrubber) can be used as potential methods for VOCs removal as compared to the other available elimination techniques. However, use of biological systems to remove VOCs still have limitations and challenges. These systems do not response well to sudden loading stresses and can fail. Due to detrimental nature of VOCs to humans and environment, more practical and more efficient VOC removal techniques are in demand. A combination of existing treatments can be considered as another approach which may help to increase VOCs treatment effectiveness at higher concentrations, whilst reducing costs e.g., biofiltration as a post-treatment to adsorption.

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