Review of Biofiltration for Sewer Odour Abatement

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Executive Summary

Biofiltration is a biological technology that has been successfully applied in a range of industries to abate odours. Biofilters have seen significant uptake in Australia for sewer odour abatement applications (representing 44% of installed sewer odour abatement processes in an industry survey conducted by Sivret and Stuetz in 2010) and a preference exists for biological based odour abatement processes due to the perceived advantages associated with biological processes over conventional adsorption or chemical based odour abatement processes. However, the wastewater industry has limited independent and scientifically based information on how to assess and specify biofiltration technologies for the treatment of odorous air streams from both sewer networks and wastewater treatment facilities.

The purpose of this review was to carry out a critical assessment of available literature and to establish baseline knowledge on those factors that affect the selection, design, and performance of biofiltration technologies. The review identified key strengths, limitations, and knowledge gaps that are summarised in **Table E1**.

Table E1 Biofiltration review summary – strengths, limitations and knowledge gaps

Strengths	Limitations	Knowledge Gaps
- Relatively easy operation and start-up - Low operating costs - Effective reduction of low concentration VOCs and odorants - High removal efficiency for H_2S (>99%) - Significant removal of reduce sulfur compounds - No secondary waste streams	- Difficult to control operating conditions - Large footprint required - Reduced performance for fluctuating inlet gas component concentrations (particularly peak loads) - Clogging by particulate matter - Air channelling - Medium deterioration - Reduced efficiency at high odour concentrations - Potential for nutrient depletion - Experienced based design practices	- Biofilm thickness, growth and architecture - Biodegradation mechanisms / kinetics - Removal of VOCs found in sewers - Models and design practices - Full scale investigative work

While there have been significant advances in understanding biofilter design and operation, significant knowledge gaps remain. Development of an improved understanding of the microbiology and biomass distribution in biofilms (biofilm

thickness, growth and architecture) as well as improved understanding of biodegradation mechanisms/kinetics for specific contaminants of interest would lead to improved mechanistic design of biofilters for sewer networks and wastewater treatment plants that would provide improved odour abatement outcomes. Addressing these knowledge gaps would also support the improvement of biofilter models to support the design process as well as operating/diagnostic decisions.

Much of the reviewed literature has focused on laboratory and pilot scale processes operating under idealised conditions. Investigative work to consider conditions that are representative of those encountered at full scale, in particular the treatment of complex multi-contaminant mixtures at trace concentrations is necessary to provide information to provide better understanding of the biofiltration process and to provide a basis for improved design, operation and maintenance and to support the industry in critically assessing information provided by odour abatement process suppliers.

The program of experimental work in SP3 will contribute to addressing these knowledge gaps, in particular:

- identifying the key odour producing and treatment interfering compounds present in sewer networks to provide the industry with a more relevant list of compounds to use for design specifications and process performance evaluation;
- providing monitoring protocols to allow comprehensive abatement process performance assessment and support the diagnosis and solution of operational issues;
- evaluating the performance of full scale biofilters for the removal of odour and key odorants to provide the industry with data that can be used to identify the most appropriate odour abatement technology for a specific application; and
- using full scale odour abatement process performance data to identify the suitability of biofilters for abating specific odorants, supporting the industry in the selection and evaluation of odour abatement technologies.

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1.0 Introduction

Over the last two decades biological odour treatment has become a key component in the control of odorous emissions from composting facilities, wastewater treatment plants (WWTPs), sewer emissions, confined animal facilities, food industry and other sources of volatile organic compounds (VOCs) and odours. Biological processes are based on the action of microorganisms capable of converting VOCs and odorants into innocuous compounds such as CO_2, H_2O, and biomass at ambient pressure and temperature (van Groenestijn and Hesselink 1993; Kennes and Thalasso 1998; Converti 1999).

Biofiltration is the most commonly implemented odour abatement biotechnology. In biofilters, contaminated air passes through a bioreactor filled with a porous packed medium that supports microorganisms. Air contaminants are first transferred from the air to the water/biofilm phase surrounding the packing material in the bed. The contaminants are then aerobically degraded to various end products. Unlike biotrickling filters, biofilters do not have a recirculating liquid phase, and media itself provides the trace nutrients (such as organics).

Biofiltration has a long history, with the first successful applications and patents dating from the early 50s in the United States and Germany (Ergas and Cardenas-Gonzalez 2004; Alix and Williams 2006). The original primary application of biofilters was odour treatment. These biofilters were based on open-soil media systems, and were designed and set-up according to empirical considerations (Ergas and Cardenas-Gonzalez, 2004).

As environmental legislation became increasingly stringent in the 1970s and residential and urban areas encroached upon odour sources, the application of biofilters expanded significantly (Ergas and Cardenas-Gonzalez, 2004). Biofiltration was established as a cost-effective and environmentally friendly alternative to other physical and chemical methods for off-gases treatment, such as combustion or adsorption/absorption processes. Since the 1980s, biofiltration application at a full-scale has been extended to removal of volatile organic and inorganic compounds (VOCs and VICs, respectively) from industrial waste gases (Zhuang et al. 2001; Ergas and Cardenas-Gonzalez, 2004). For example, the use of biofilters as an odour reduction technique in livestock facilities was widely investigated in Germany during the early 1980s (Nicolai and Janni 2001).

Research developments since the late 1990s have focused on many of the key aspects influencing biofilter performance, in particular support media, bioreactor configuration, microbiological aspects, biomass control, transient conditions, treatment of mixtures and recalcitrant compounds, and robustness enhancement (Kennes and Veiga 2001; Ergas and Cardenas-Gonzalez 2004; Phillips 2009).

Despite these developments, the industry at large generally has limited independent and scientifically based information on how to assess and specify biofiltration technologies for the treatment of odorous air streams from both sewer networks and wastewater treatment plants. This review supports the improvement of the design, operation and monitoring of biofilter based odour abatement processes by providing a critical assessment of the literature to establish baseline knowledge on those factors that affect the selection, design, and performance of biofiltration technologies.

Fundamental process principles will be reviewed in **Section 2,** while existing process configurations and key design parameters will be discussed in **Section 3.** A review of available literature with regards to the operation and maintenance of biofilters will be presented in **Section 4**. Available process performance results and cost data are discussed in **Section 5**. Future directions and a summary are presented in **Section 6**.

2.0 General Process Principles – Biological Degradation

Biodegradation processes rely on both mass-transfer and biological oxidation mechanisms. In the first stage, contaminants are removed from the gas phase by absorption into the liquid phase or, less frequently, by direct transfer from the gas to the fungal or bacterial biofilm. Mass transfer is typically characterized by an overall mass transfer coefficient (K_La), obtained from a mass balance over the mixed liquid phase:

$$\frac{dC}{dt} = K_L a (C^* - C) \quad \text{Eq. 1}$$

Where C is the concentration in the bulk liquid (g/m^3), C* is the liquid equilibrium concentration (g/m^3), and K_La the overall transfer coefficient (1/s).

Contaminant absorption plays a major role in the biofiltration process, with the degree of absorption depending on the chemical characteristics of the specific contaminant (for example water solubility and molecular weight). Water solubility is related to the gas/water partition coefficient, commonly expressed by the Henry's Law constant (H_i). This constant is defined as:

$$H_i = C_{a,i} \,/\, P_{g,i} \quad\quad \text{Eq. 2}$$

Where $C_{a,i}$ is the concentration of a specific compound in the aqueous phase and $P_{g,i}$ is the partial pressure of the compound in the gas phase.

A compilation of the Henry's Law constants for a selection of compounds commonly found in odorous emissions is provided as **Table 1**. Those compounds with higher Henry´s Law constant (such as aldehydes and acids) are more hydrophobic and thus more difficult to transfer to the biofilm than those with lower values (such as hydrogen sulfide and mercaptans). Although some variability can be found in the value of the Henry´s Law constant depending on the source, the order of magnitude clearly shows the degree of hydrophobicity of each compound.

Table 1 Henry`s Law constants for a range of odorants associated with wastewater treatment processes (Sander 1999)

Substance	Compound	Henry`s Law Constant (M/atm)
Sulfurous	Hydrogen sulfide	0.001 – 0.1
	Methyl mercaptan	0.2 – 0.71
	Ethyl mercaptan	0.22-0.36
	Sulfur dioxide	1.1 – 1.5
	Dimethyl sulfide	0.08 – 0.71
	Diethyl sulfide	0.46 – 0.56
	Dimethyl disulfide	0.84 – 0.96
Nitrogenous	Ammonia	10 – 78
	Methylamine	36 – 140
	Ethylamine	35 – 100
	Dimethylamine	31 – 57
	Pyridine	90 – 110
Acids	Acetic	980 – 10000
	Butyric	1900 – 4700
Aldehydes and ketones	Formaldehyde	3000 – 14000
	Acetaldehyde	1.7 – 17
	Butyraldehyde	5.5 – 9.6
	Valeraldehyde	4.4 – 6.8
	Acetone	3.0 – 35
	Butanone	6.9 – 21

Contaminant absorption is followed by diffusion through the liquid phase from the gas-liquid interface to the liquid-biofilm interface, then adsorption onto the biofilm surface, diffusion into the biofilm, and finally degradation via biological reaction within biofilm (Li et al. 2002). A schematic representation of the biodegradation mechanism is presented in **Figure 1**.
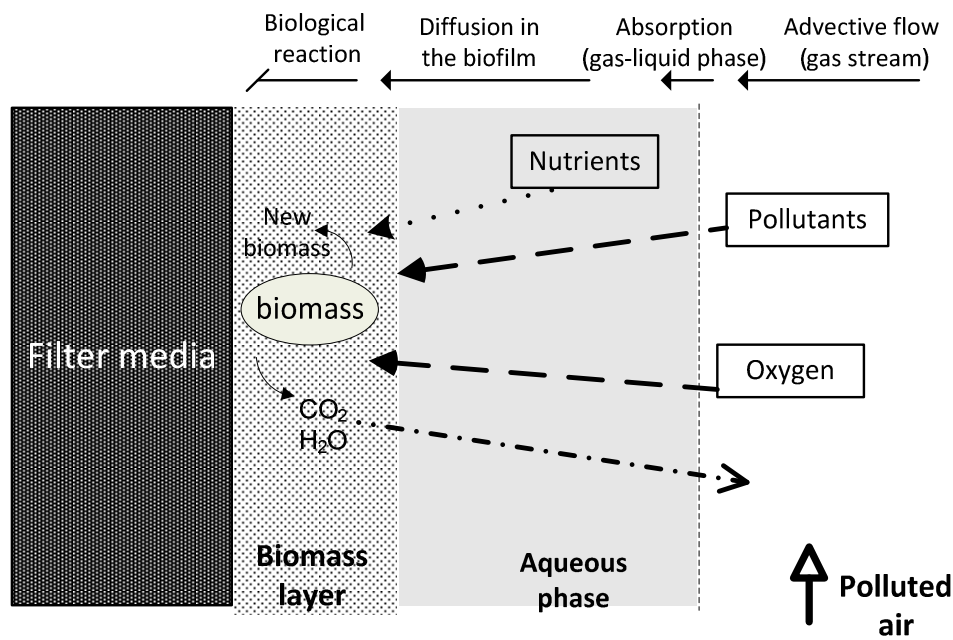
Figure 1 Biodegradation mechanisms in biofiltration processes

The gas/water equilibrium can be considered instantaneous since it is fast compared to adsorption and biodegradation, especially at empty bed residence times (EBRTs) over 1 min (McNevin and Barford 2000). Biological reaction rates can be zero or first order reactions, depending on the contaminant concentration. Zhuang et al. (2001) found that the reaction is of zero-order for H_2S concentrations above 400 ppm, first order for concentrations below 200 ppm, and fractional order for intermediate concentrations. Similar results were obtained by McNevin and Borford (2000), who observed that at high inlet concentrations, the biological reactions became zero order or independent of the concentration due to saturation kinetics.

Most volatile compounds and odorants are biodegradable to some extent by a wide number of microorganisms mainly bacteria or fungi, even the recalcitrant compounds (xenobiotics, chlorinated ethylenes, etc.), and complete degradation can be expected (Kennes et al. 2009). The makeup of the microbial community in the biofilm is dependent upon the contaminants present in the foul air and on the inlet stream characteristics (Kennes and Veiga 2001). Although bacteria communities are often found in properly operated biofilters, fungi can develop under conditions of extremely low water content, low pH or nutrient limitation, since they are more resistant to adverse environmental conditions. Fungi also show better performance in the degradation of hydrophobic compounds due to aerial hyphae that create

a large surface area in direct contact with the contaminated gas phase (van Groenestijn et al. 2001). The presence of fungi may lead to an increase in pressure drop as a result of filament proliferation, although this can be prevented by the use of large-pore materials or introducing fungi predator organisms such as mites (Kennes and Veiga 2001; van Groenestijn et al. 2001).

Biofilm thickness varies from tens of μm to more than 1 cm, with an average of less than 1 mm usually found (Mudliar et al. 2010), with the thickness typically being unevenly distributed in the biofilter, accumulating preferentially in the inlet of the reactor where carbonaceous nutrients brought by the gas have higher concentrations. As a result, most significant biodegradation activity is measured in the first third of the biofilter height (Kim and Sorial 2007; Cabrol et al. 2008).

In most waste gases, oxygen concentration is several orders of magnitude higher than the odorant concentration, thus oxygen limitation rarely occurs (Rappert and Müller 2005). This review will thus focus on the aerobic biodegradation pathways for key groups of odorous compounds (sulfur compounds, VOCs, and nitrogen compounds).

2.1 Sulfur Compounds

Hydrogen sulfide (H_2S) is typically the dominant sulfur odorant found in sewer emissions, although other sulfur odorants including mercaptans (such as methyl mercaptan and ethyl mercaptn) and sulfides (e.g. dimethylsulfide, dimethyldisulfide, and dimethyl trisulfide) are commonly identified. The transformation processes between the different sulfur oxidation forms is illustrated in the biological sulfur cycle in **Figure 2**.
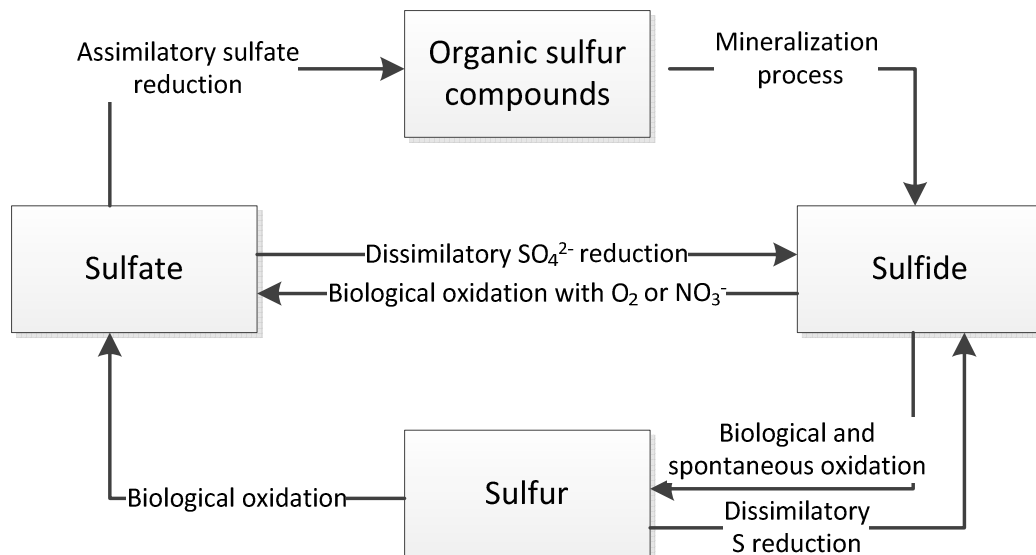
Figure 2 Biological sulfur cycle

The removal of H_2S can occur through two different biodegradation pathways (Janssen et al. 1999; Elias et al. 2002):

a) Under **aerobic conditions** biodegradation is carried out by chemolitoautotrophic bacteria using inorganic carbon (usually CO_2) as the carbon source. Sulfur oxidizing bacteria such as Thiobacillus have a high H_2S affinity. Depending on the amount of available oxygen, the sulfur compound will be degraded either to elemental S^o under oxygen-limited conditions, or to H_2SO_4 in the presence of excess oxygen.

$$2HS^- + O_2 \rightarrow 2S^o + 2OH^- \qquad \text{Eq. 3}$$
$$2HS^- + 4O_2 \rightarrow 2SO_4^{2-} + 2H^+ \qquad \text{Eq. 4}$$
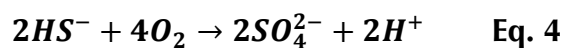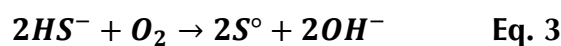
Incomplete oxidation and elemental sulfur formation will also occur if the bacteria are over-fed (high H_2S loadings).

b) Biodegradation under **anaerobic conditions** rarely occurs in biofilter biofilms since it requires anaerobic conditions together with radiant energy. H_2S is degraded by bacteria such as Chlorobioceae or Chromatiaceae into elemental sulfur according to the following reaction:

$$2nH_2S + nCO_2 \xrightarrow{h\vartheta} 2nS° + (CH_2O)_n + nH_2O \quad \text{Eq. 5}$$
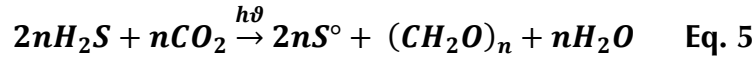
Degradation of other sulfur derived compounds, such as mercaptans or dimethyl sulfides, will eventually generate H_2SO_4 according to:

$$CH_3SH/(CH_3)_2S/(CH_3)_2S_2 + nO_2 \rightarrow mH_2SO_4 + CO_2 + H_2O \quad \text{Eq. 6}$$
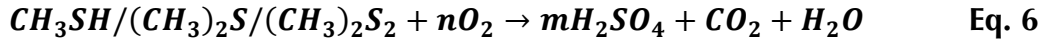
2.2 Volatile Organic Compounds (VOCs)

VOCs include hydrocarbons, aldehydes and ketones, alcohols and terpenes, which primarily consist of carbon and hydrogen, although oxygen atoms may also be present. Aerobic chemoheterotrophic bacteria are responsible for VOC biodegradation, using the VOCs as the carbon and energy sources for biomass growth and maintenance, and for the production of new biomass. Final products of VOCs biodegradation are H_2O, CO_2 and biomass (Lebrero et al. 2011).

$$VOC + x\,O_2 + nutrients \xrightarrow{microorganisms} y\,CO_2 + z\,H_2O + C_5H_7NO_2 \quad \text{Eq. 7}$$
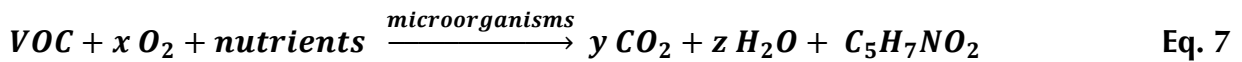
A specific group of VOCs also found in waste gas emissions are halogenated compounds, mostly chlorinated compounds. Besides being less biodegradable than VOCs, they produce hydrogen chloride as a by-product, which causes acidification of the medium.

2.3 Nitrogen Compounds

One odorous nitrogen compound of concern in many industries is ammonia. The most common biodegradation pathway of ammonia is the nitrification process in which it is converted to nitrate by nitrifying bacteria.

$$NH_{3(g)} \rightarrow NH_{3(l)} \rightarrow NH_3 + H^+ \rightarrow NH_4^+ \quad \text{Eq. 8}$$

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O \quad \text{Eq. 9}$$

$$2NO_2^- + O_2 \rightarrow 2NO_3^- \quad \text{Eq. 10}$$
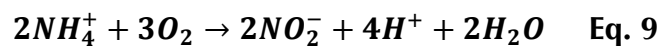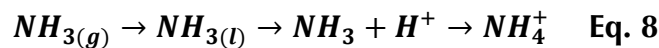
Further conversion of nitrate ions can occur in anoxic conditions by denitrifying bacteria:

$$2NO_3^- \rightarrow N_2 \qquad\qquad \textbf{Eq. 11}$$

Nitrification releases protons and decreases liquid pH. According to Shanchayan et al. (2006), about 88% of the total N leaves the biofilter as $N - NH^{4+}$ in the leachate, thus ammonia removal mechanisms is mainly scrubbing rather than nitrification (only 12% of the N was found as $N - NO_3^-$ and $N - NO_2^-$). This is due to the slow degradation kinetics.

Amines such as di- and tri-methyl amine (DMA and TMA) or di- and tri-ethyl amine are often found in gaseous emissions from chemical industries, food industries, and agricultural operations (Rappert and Müller 2005). The biodegradation pathways of these compounds are not very well known, although some authors suggest that, under aerobic conditions, the metabolism of DMA starts with conversion to methylamine (MTA) and formaldehyde (Rappert and Müller 2005):

$$TMA \rightarrow TMA \text{ N-oxide} \rightarrow DMA \rightarrow MTA \xrightarrow{O_2 + NADPH} NH_3 + formaldehyde \rightarrow further\ mineralization \quad \textbf{Eq. 12}$$
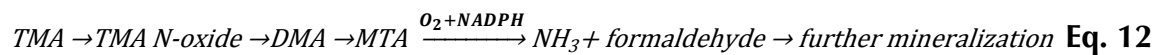
3.0 Process Configuration and Design

In the biofiltration process, the contaminated air passes in either downflow or upflow mode through a bioreactor filled with a porous packed medium that supports a thriving population of microorganisms. Downflow configuration is sometimes preferred since it reduces drying problems at the inlet of the reactor, which may appear as a result of the exothermic nature of biological reactions (Van Lith et al. 1990). There are two main structural configurations for biofilters: open biofilters and enclosed (partially or totally) biofilters (**Figure 3**). The design depends primarily upon space availability at the site, as well as costs and maintenance issues (Quigley et al. 2005).
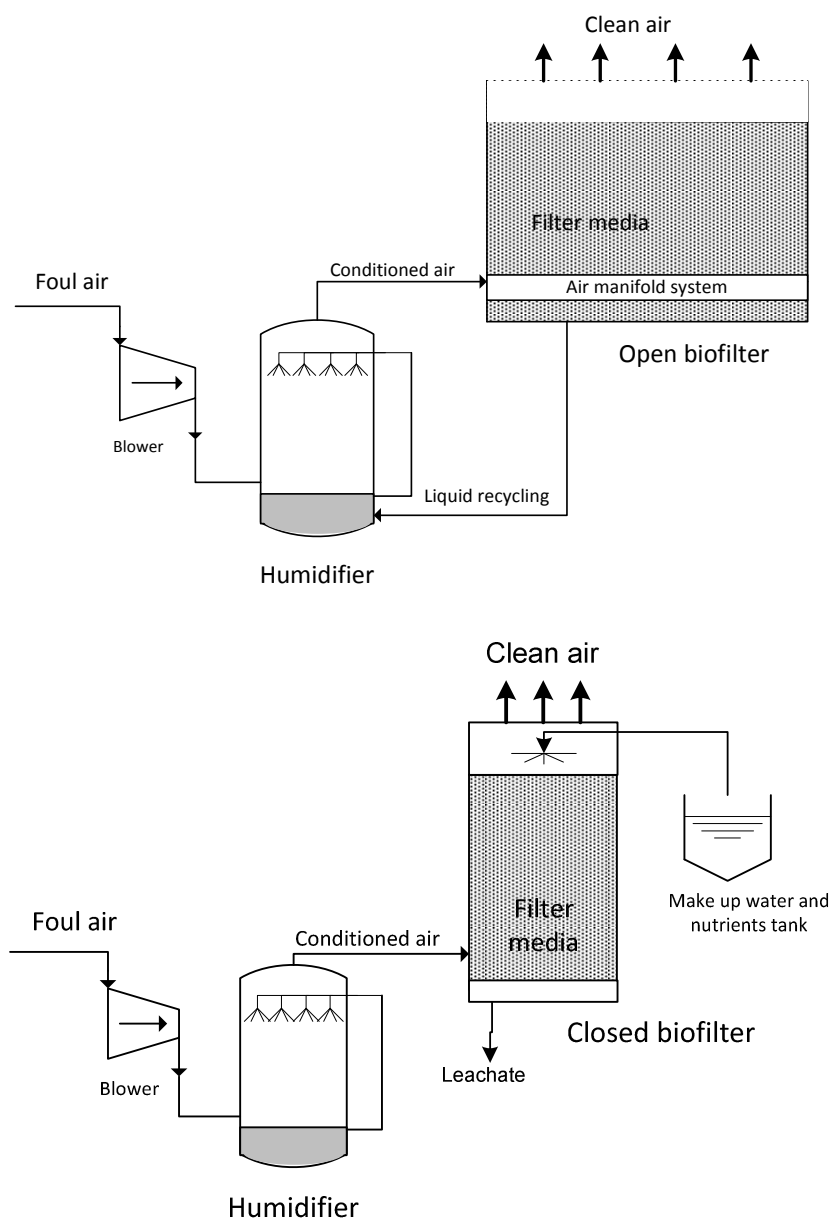
Figure 3 Schematic representation of biofilter configurations: open biofilter (top) and closed biofilter (bottom)

Open biofilters are the least expensive, but most land intensive configuration. Open bed biofilters are also much more unstable than closed in-vessel systems. They are built usually about 1 m below the grade level and covered with plastic liner. A 3:1 slope is recommended (Gueissaz-Teufel 2004). Airflow distribution is carried out by a network of perforated plastic piping laid below the bed and surrounded by crushed stone (Gueissaz-Teufel 2004; Easter et al. 2005).

Closed (walled) biofilters have reduced footprint and improved aesthetic impact. Enclosing biofilters will further enhance abatement efficiency since it is less exposed to severe environmental conditions. The enclosure also allows for improved control of moisture, temperature and pH, as well as better dispersion of treated off-gas (Ergas and Cardenas-Gonzalez 2004). Closed biofilter configurations are preferable where reliable performance has to be maintained even under very hot, cold, wet or dry conditions. They usually have a slotted or grating type system for air distribution, with the concrete floor and walls forming a plenum for even air distribution and leachate collection (Adler 2001; Ergas and Cardenas-Gonzalez 2004; Easter et al. 2005).

Two stage (or two-media layer) biofilters is a configuration which separates the biofilter media into discrete zones, each of which is specialised for the abatement of a specific groups of contaminants. As a general rule, two stage biofilters are useful for high-strength airstreams or when high removal efficiencies are required (Chitwood and Devinny, 1999).

For combined H_2S and VOC removal, the first stage is conditioned for H_2S degradation (autotrophic bacteria and low pH) and the second for VOCs removal (heterotrophic bacteria that thrive at neutral pH). This allows for enhanced VOC removal efficiencies by avoiding inhibition of VOCs biodegradation due to sulfate accumulation, while using the same overall contact time (biofilter size) (Converse et al. 2003). Separation also allows for independent water control of each section, and prevents acidic leachate from trickling into the VOC section (Converse et al. 2003; Zappi et al. 2004). Chitwood and Devinny (1999) used a two-stage biofilter, with a first acid stage packed with lava rock and a second organic-stage. Although H_2S removal efficiency remained almost constant, VOCs elimination improved.

Two stage biofilters have also been applied to optimise the removal of sulfur odorants. Since the presence of H_2S and methyl mercaptan limits the degradation of dimethyl sulfides, Alix and Williams (2006) used a two stage biofilter to separate the degradation processes. This allowed for H_2S and methyl mercaptan removals in excess of 85% first stage, followed by DMS and DMDS removals up to 99.6% in the second stage (Alix and Williams 2006). Sorensen (2004) also presented a biofilter configuration based on the principle of H_2S and reduced sulfur compound treatment separation, in which the first stage consisted an acid resistant stone air plenum inoculated with Thiobacillus to remove approximately 60% of the H_2S in the inlet stream before entering the organic media (second stage). Complete removal of H_2S, mercaptans and dimethyl sulfide was achieved in the second stage (Sorensen 2004), and a longer life is expected in this configuration since an acid resistant support medium is used for the acidic stage.

3.1 Design Procedures

The main objective when designing a biofilter is to achieve the desired removal efficiencies while operating at maximum removal rates. Although information exists on the influence of specific parameters on biofilter performance, the process of designing and operating biofiltration units to control odorous and VOCs containing air emissions is not yet well developed and, as a result, the systems are prone to upsets of unknown origin (Li et al. 2002; Tchobanoglous et al. 2003). Current design practices are based on experience and knowledge of retention time and sulfur loading, but fundamental processes such as film transfer, diffusion or biological reaction are still poorly understood (Tchobanoglous et al. 2003). During biofilter design, it is common practice to design for highly hydrophilic contaminants at short EBRT, thus neglecting the mass transfer resistance from the gas phase to the biofilm, which may lead to insufficient mass transfer for hydrophobic compounds (McNevin and Barford 2000). If VOC removal is important, it is thus recommended that the design and operation be focused on VOCs removal as the rate-limiting parameter (Iranpour et al. 2005). If biofilm is transport limited, air phase EBRT must be longer than that suggested by considering the biological degradation rate alone (McNevin and Barford 2000).

Adler (2001) recommended several pre-testing steps before biofilter scale-up. Initially, flask tests might be performed to assess the biodegradability of the contaminats and the kinetics of

the microbial community. Bench-scale tests to determine the interaction between different contaminants, and between contaminants and filter media, are not commonly carried out because they require a synthetic contaminant stream containing the wide range of odorants commonly found in odorous emissions. A third stage, pilot testing, is recommended to establish the EBRT required to achieve the desired removal efficiency, as well as identify major incompatibilities and other design parameters (Adler 2001). Finally, prior to scale-up, it is recommended to characterize the target stream (flow rate, T, pH, particulate levels, contaminants and their concentrations), as well as consult regulations, and identify monitoring and leachate disposal methods (Adler 2001).

Models to predict biofilter behaviour are useful tools that will allow for anticipating the biofilter performance as well as reduce the need for extensive pilot and field testing prior to scale-up. Basic concepts used in reactor modelling are based on fundamental and general engineering equations, together with simplifying assumptions that present some limitations. The major remaining uncertainty in model development is the determination of appropriate constants (diffusion, Monod and maximum growth rate, etc.), as well as quantitative knowledge of the structural properties (biofilm distribution on the media surface, pores geometry, etc.) (Kennes and Veiga 2001; Li et al. 2002). Although some progress has been made in describing and understanding laboratory results and few models have been developed to support design and optimization, there is still no single model that has been accepted as a standard for biofiltration (Devinny and Ramesh 2005). Current models are not seen as being sufficiently reliable for application in biofilter design, and instead previous experience and pilot-testing are employed (Devinny and Ramesh 2005).

3.2 Design parameters

Several factors must be considered when designing biofiltration systems, such as the gas-biofilm contact time, filtration media, the moisture content, and temperature. The goal of considering these factors is to provide a suitable environment to sustain the microorganisms responsible for the biofiltration process, and ensuring efficient mass transfer. **Table 2** summarizes the typical range of values for the main biofilter design and operating parameters discussed in **Sections 3.2.1** to **3.2.5**.

Table 2 Typical range values of the main operation and design parameters

Parameter	Typical range
Empty bed residence time (EBRT)	15-60 s
Height	0.9-1.5 m
Superficial gas velocity	15-400
Volumetric gas flow rates	<200000 m³/h
Superficial gas velocity	15-400 m³/m²h
Area flow loading	50-600 m³/m²h
Volumetric loading rate	60-100 m³/m²h
Inlet gas humidity	90%-100% RH
Water content	40%-70% RH
Temperature	15-40ºC
Inlet gas concentration	0-1000 ppm

3.2.1 Sizing and Contact Time

The empty bed residence time (EBRT) represents the mean time that a volatile compound would theoretically spend in the biofilter if the bed is not packed. The actual residence time (τ) can be determined by multiplying the EBRT by the ratio between the void volume available to air and the total reactor or carrier volume (θ).

$$EBRT = \frac{V}{Q} \qquad \textbf{Eq. 13}$$

$$\tau = EBRT \times \theta \qquad \textbf{Eq. 14}$$

Where: V is the volume of the filter bed and Q is the gas flow rate.

Biofilter sizing must ensure sufficient contact time for odorous compounds to be transferred and biodegraded. Depending on the industrial application and the contaminants and concentrations to be treated, the EBRT can vary from several seconds to few minutes, generally ranging between 15 and 60 s (Adler 2001; Gueissaz-Teufel 2004; Iranpour et al. 2005). EBRTs vary between 30 and 120 s (nominal EBRT = 60 s) for in-ground open vessel biofilters, while they range 30 to 60 s for soil-based systems and organic-based closed biofilters (Easter et al. 2005; Van Harreveld 2007). Full-scale conventional biofilter operation is not seen as cost effective for EBRT exceeding one minute (Kennes and Veiga 2001).

Filter volumes range between 4.2 and 16.7 m^3 per 1000 m^3/h of air treated, with bed heights from 0.9 m up to 1.5 m to avoid excessively high pressure drops (Adler 2001). Biofilters can cope with volumetric gas flow rates of bulk air up to 200000 m^3/h, although higher values might be reached (Kennes and Thalasso 1998).

Superficial gas phase velocity (or surface loading rate, v_s) is the ratio between the gas flow rate (Q) and the sectional area of the carrier (A). The volumetric loading rate (L_v), which represents the ratio between the gas flow rate and the volume of the filter bed, is equivalent to v_s and also depends on the biofilter media and the configuration of the system:

$$v_s = \frac{Q}{A} \qquad \textbf{Eq. 15}$$

$$L_v = \frac{Q}{V} \qquad \textbf{Eq. 16}$$

Air surface loading rates are based on operational experience with full scale systems, with little published research data available. For soil-media biofilters, the v_s varies from 15 to 150 m^3/m^2h, with higher loadings for open vessel organic-media systems (35-90 m^3/m^2h) compared to enclosed vessels (150-400 m^3/m^2h) (Easter et al. 2005). According to McNevin and Barford, this value should vary from 30 to 200 m/h to provide at least 100 parts of oxygen for every part of oxidisable odorous gas (McNevin and Barford 2000). As v_s decreases, the EBRT increases, thus improving the biofilter removal efficiency per unit of bed height. However, the volume of bed required per unit of air flow required also increases, and consequently the capital and operating costs increase as well. According to (Van Harreveld 2007), area flow loadings are in the range of 50-600 m^3/m^2h for biofiltration systems, and between 100 and 200 m^3/m^2h in composting installations. Luo (2001) recorded higher v_s for biofilters with highly porous media, such as bark or peat moss, ranging from 35 to 180 m^3/m^2h, while lower values are used for compact media such as soil: 2-10 m^3/m^2h. Outlet gas odour concentrations below 1000 OUE/m^3 have never been recorded at v_s higher than 180 m^3/m^2h (Van Harreveld 2007). Maximum L_v are usually about 100 m^3/m^3h with complete contaminant removal. Loading rates up to 200 m^3/m^3h have also been treated, although the removal efficiency never reached 100% (Kennes et al. 2009).

3.2.2 Loading Rates and Compounds Treated

Biofiltration is best suited for treatment of moderate airflow rates with low concentration of contaminants, although high inlet concentrations (up to 1000 ppmv) and flow rates up to 170 000 m³/h have been treated (Adler 2001). Some authors recommend not exceeding inlet concentrations of 5-6 g/m³ to avoid inhibition problems (Kennes et al. 2009). Biofilters working at high substrate concentrations may present low removal efficiencies and accumulation of toxic intermediate biodegradation products can occur. Furthermore, high mass loadings will increase metabolic activity, producing additional heat and a subsequent moisture reduction in the bed, potentially impacting performance. The lack of a continuous liquid phase makes biofilters suitable for treating moderately hydrophobic and poorly water soluble compounds with Henry´s constant <1 (Kennes and Thalasso 1998).

3.2.3 Filtration Media

Biofilter packing media (also known as carrier material, filter bed, support material or medium) has been widely reported as one of the key parameters in biofiltration performance, since the microorganisms responsible for contaminant biodegradation will grow in a biofilm on its surface and because it is responsible for the pressure drop and therefore energy consumption (Andres et al. 2006; Liu et al. 2009). Traditionally, organic media such as peat, compost, soil, bark, wood chips, sludge, and rice hulls has been used for biofilter packing, although inorganic materials are also frequently used. There is an increasing trend towards new proprietary media as well as mixtures of organic-inorganic materials (Easter et al. 2005).

Media selection involves a number of considerations, including the media material source, particle sizing, cross-sectional depth, surface loading, porosity, and desired service life. Desirable media properties include (Easter et al. 2004; Gaudin et al. 2008; Chen et al. 2009):

- Ability to retain microorganisms, providing a suitable environment for microorganisms to thrive and generating a clear leachate.
- High porosity (which in turn reduces the pressure drop across the bed), usually ranging from 0.4 to 0.9.
- High specific surface area for biomass attachment and gaseous contaminant

adsorption, ranging between 300 and 1000 m^2/m^3.

- Mechanical resistance, physical stability, high compressive and bearing strength and low bulk density. This will lead to better hydrodynamic properties and low bed compaction.

- Adequate moisture content and moisture retention capacity, freely drainage and release of excess moisture.

- Good pH buffering capacity for acidic end products to avoid pH fluctuations.

- For organic materials, a high and diverse mineral nutrients content to supply microorganisms with essential nutrients.

- Low cost and readily/locally available.

Organic media have been traditionally preferred over synthetic materials (Kennes et al. 2009). Some operational parameters such as the H_2S removal efficiency are essentially the same for organic and inorganic materials. The main advantages of organic media are the low cost, the abundant availability and the natural presence of a high concentration and diversity of microbial communities and nutrients. Their capacity for odour control has been widely proven and they are more resistant to drying out, being able to function at full capacity for 2-3 days without irrigation (Kennes et al. 2009). One of their major drawbacks is the decomposition of the bed with time, causing a deterioration of the bed structure and increasing compaction and head loss while decreasing biofilter performance over time (Kennes et al. 2009). Due to biodegradation of the filter bed or nutrient depletion, organic media lifespan is shorter compared to inorganic materials, ranging from 2 to 4 years (Alix and Williams 2006). The addition of bulking agents such as vermiculite, perlite or woodchips usually extends the life of organic media, and allows for better control of pressure drop (Kennes et al. 2009). Organic materials also present lower surface-to-volume ratios, thus low volumetric reaction rates and larger footprints are frequently required (Sorensen 2004; Alix and Williams 2006; Kennes et al. 2009).

Inert and inorganic materials are gaining in popularity due to their longer lifespan (and hence lower replacement frequency) when compared to organic media (Alix and Williams 2006). This increased lifespan, on the order of 10 years from some materials, is due to improved structural properties. They usually present low head losses due to larger interstices

between packing granules or pieces. Mass transfer of contaminants to the biofilm is also improved by the larger specific surface area (Kennes et al. 2009). Their composition is usually well defined and quite stable over time, and these materials can be manufactured to relatively uniform particle sizes and pore size distributions. Inorganic media replacement costs are typically lower than those for organic media (less than $18/m^3 vs. $71-103/m^3 (Sorensen 2004). Park et al. (2001) found that bed media replacement of a ceramic biofilter would cost about $514/m^3, with a lifetime of 10 years, while Adler (2001) estimated the costs of media replacement between $50 and $300/m^3. Examples of inert materials used as packing for biofilters are porous ceramics, lava rock, lime, activated and waste carbon, oyster shells or synthetic plastic materials. Inorganic materials are easily rewetted after drying compared to organic media due to their hydrophilic surfaces, however, irrigation is critical in inorganic packed-beds to avoid the bed drying-out (which can occur in under 24 hours) which would decrease performance. They also require an intermittent feeding of a nutrients solution, leading to a more complex irrigation system, which increases the operation and maintenance costs (Gueissaz-Teufel 2004; Sorensen 2004; Kennes et al. 2009).

Media selection is an application specific decision, where the composition and concentration of the contaminated air stream, the air flow rates to be treated, the pH, the availability of the packing material needs to be considered. The characteristics of the different packing materials will determine their suitability for a specific process, with readily-available materials typically being chosen.

Compost and soil have been the most widely used materials in biofilters. Soil-based media offers low shrinkage and high stability over years, although the high pressure drop across the bed leads to larger footprints. Compost has an advantage of reduced pressure drops, but it has a shorter lifespan due to the biodegradability of the material (Gueissaz-Teufel 2004). Biofilters based on compost media are also more sensitive to low moisture content, leading to crusting, cracking and air flow short-circuiting (Goldstein 1999; Gueissaz-Teufel 2004; Poulsen and Moldrup 2007).

Peat media offers a varied microflora and high specific surface areas. The primary limitation with peat based media is the extremely low buffering capacity that leads to pH being the

limiting factor when abating sulfur compounds (Kennes and Thalasso, 1998; Delhoménie and Heitz, 2005). Phillips (2009) established that the maximum concentration of H_2S that can be treated in a peat-biofilter is 50 mg/m^3.

Inorganic materials such as activated carbon usually show better performance than compost at shorter retention times and they have proven to be very effective for odour removal and spike loading buffering by combining adsorption and biodegradation (Webster et al. 1996; Jiang et al. 2008; Mudliar et al. 2010). The high investment costs and large amount of activated carbon required typically makes it economically unattractive as a biofilters media (Jiang et al. 2008). The utilisation of exhausted activated carbon instead of virgin activated carbon as a media serves to reduce the investment costs, with the carbon being partially regenerated through the biodegradation process in the biofilters, and the pre-deposited sulfur acting as a sulfur source for microbial growth (Jiang et al. 2008).

Other innovative/less commonly used inorganic packing materials have also been tested at laboratory scale. For example, Jeong et al. (2008) achieved H_2S removal efficiencies up to 99% in a biofilter packed with polypropylene fibrils, operating at an inlet concentration of 700 ppm and 10s of EBRT. The system also showed rapid recoveries from shock loads. Volcanic stone was also tested (Jeong et al. 2008), although lower removal efficiencies were obtained at higher EBRT (RE=95.8% at EBRT of 30-60s and inlet H_2S concentrations up to 600 ppm). Gaudin et al. (2008) mixed calcium carbonate (C source and buffer agent) with a binder organic and a nitrogen source (urea phosphate), obtaining a packing material (named UP20) capable of achieving NH_3 removal close to 100% and H_2S removals higher than 90% at H_2S inlet concentrations of 200 mg/m^3.

A summary of key characteristics of organic and inorganic packing materials reported in the surveyed literature is presented in **Table 3**. While the application of **proprietary media** is increasing, current information is generally limited to that supplied by the manufacturer and no comprehensive independent evaluations of proprietary media were identified in the reviewed literature.

Table 3 Main characteristics of organic and inorganic materials

Material	Microbial Diversity	Physical / Chemical Stability	Compaction	Pressure drop	Water content	Water Retention	pH Buffering Capacity	Loading Rate	Surface Area (m²/g)	Porosity	Lifespan	Reference
Soil Media	Medium	High	Low	High	_	58.8%	Good	Low EBRT: 2-3 min	Low	Medium 40-50%	10-30 years	Chitwood, Devinny et al. 1999; Goldstein 1999; McNevin and Barford 2000; Easter, C. et al. 2004; Gueissaz-Teufel 2004; Mudliar, Giri et al. 2010)
Compost	High	Low – degrades rapidly	High	Low	High 40%	Good 67-74%	Limited Neutral pH	Medium EBRT: 30-45 s	Large (1.4)	High 33-80%	Low up to 4 years	Goldstein 1999; McNevin and Barford 2000; Easter, C. et al. 2004; Poulsen and Moldrup 2007; Chen, Hoff et al. 2009; Liu, Li et al. 2009
Sewage Sludge Compost	High	Low - variable over time	_	_	_	High	_	_	Large	49%	_	Poulsen and Moldrup 2007
Peat	Medium	Low - variable over time	High	_	High 47-80%	Good 47%-69%	Limited	_	High (>200)	High	Low 2-4 years	McNevin and Barford 2000; Kennes and Veiga 2001; Gueissaz-Teufel 2004; Armeen 2006; Gaudin, Andres et al. 2008; Phillips 2009
Wood Chips	Medium	_	Low	Low (large particle size)	_	High 64.6%	Low	_	Low	Medium 38%	Medium 4 years	Chitwood, Devinny et al. 1999; Gueissaz-Teufel 2004; Sorensen H.W. 2004; Colon, Martinez-Blanco et al. 2009; Mudliar, Giri et al. 2010
Bark	Medium	_	Low	Low	Low 14%	High 56-58%	Acid material pH = 4-5	_	High	Medium 51-56%	_	Luo 2001; Gueissaz-Teufel 2004; Gaudin, Andres et al. 2008
Coconut Fibre	_	_	_	_	_	Low 4-5 times lower than inorganic materials	Intermediate 22 g SO_4^{2-}/kg	_	High (~0.75)	Low	_	Gabriel, Maestre et al. 2007
Mulch	Medium	_	_	Low	_	Good	_	_	_	_	_	Gueissaz-Teufel 2004
Porous Ceramic	Low	High	Low	Low	High 63-89%	Low	_	_	_	High 76-89%	Long	Hirai, Kamamoto et al. 2001; Park, Nam et al. 2001
Plastic Packing	Low	High	Low	Low	_	_	_	_	60-320 m²/m³	_	Long	Gueissaz-Teufel 2004
Sea Shells	Low	High	_	_	_	High	Good	_	_	High	_	Phillips 2009
Activated Carbon	Low	High	Low	Low	_	Good 48.8-56 wt%	_	High (buffer for fluctuating loadings)	Large 400-1200	18%	Long >5 years	Chitwood, Devinny et al. 1999; Armeen 2006; Duan, Koe et al. 2006; Jiang, Yan et al. 2008
Lava Rock	Low	High	Low	Low	_	15.5%	Good 10 mg/L alkalinity or 50-90 g SO_4^{2-} /kg	_	Large 1.4-3.5	High 25-49%	_	Chitwood, Devinny et al. 1999; Cho, Ryu et al. 2000

3.2.4 Moisture content

Many authors identify bed moisture content as the single most important parameter influencing biofilter performance (McNevin and Barford 2000; Kennes and Veiga 2001; Iranpour, Coxa et al. 2005). Optimal bed moisture content ranges from 40% to 70% on a weight basis (as expressed below), although it might depend on the packing material (Goldstein 1999; Gueissaz-Teufel 2004; Easter et al. 2005; Van Harreveld 2007, Webster et al. 1996).

$$\text{Water content (wet basis)} = \frac{\text{mass}_{\text{water}}}{\text{mass}_{\text{water}} + \text{mass}_{\text{dry carrier}}} \quad \textbf{Eq. 17}$$

Excessive moisture may inhibit mass transfer from the gas phase to the biofilm or to the adsorption surface of packing material, and has a significant influence on the compactability and air permeability of compost and most forms of organic media (Kennes and Thalasso, 1998). As the moisture level increases, the free air space drastically reduces with depth (up to 30% less of free air space within 1.5 m depth), which reduces oxygen availability and can lead to the formation of anaerobic zones (Kastner and Das 2005). The media also becomes denser and more compact with increasing moisture level, reducing the porosity and increasing the back pressure. Low moisture levels can lead to the media drying out (having detrimental effects on microorganism growth), and allowing short-circuiting within the bed will negatively impact abatement performance (Kennes and Thalasso, 1998).

Experimentally, several authors have reported a decrease in biofilter performance with decreasing moisture content (Chitwood and Devinny 1999; Nicolai and Janni 2001; Zhuang et al. 2001; Converse et al. 2003; Gabriel et al. 2007; Chen et al. 2009). Chitwood and Devinny (1999) observed that H_2S removal efficiency decreased by 16% when the irrigation was stopped for 5 days and water content decreased from 16.4 to 12.5% w/w, while Converse et al. (2003) observed a temporary drop on H_2S removal by more than 4% after a 6-day interruption of water supply. Gabriel et al. (2007) reported that improper watering of packing material results in lower removal efficiencies and a low buffering capacity to face transient peaks. At high moisture contents (55%-65%), Nicolai and Janni (2001) obtained odour and H_2S removal efficiencies up to 78.8% and 90-95%, respectively, but when the moisture content was reduced (~27.6% humidity) the odour and H_2S removal efficiencies decreased to 42% and 3%, respectively. Similarly, for a biofilters treating odours from swine,

Chen et al. (2009) observed a decrease in H_2S and NH_3 removal performance from 84% and 67%, respectively (at a relative humidity of 60%), to less than 6% removal when the moisture content was reduced to 20%. Zhuang et al. (2001) also reported a severe loss of efficiency in their wood bark biofilter if water content decreased below 60%.

In order to prevent moisture being drawn out of the bed, the inlet air must be humidified (preferably close to saturation) prior to entering the biofilter. Pre-humidification of the inlet air can be performed by spray nozzles in the biofilter inlet air duct, in spray chambers ahead of the biofilter, in water-only packed towers before entering the biofilters, or by steam injection built into the biofilter (Kennes and Thalasso 1998). Spray towers also remove particulate matter, dust, etc. from the inlet air, preventing the packing bed from clogging, however, a complete saturation of inlet air (close to 100%) is required to completely compensate for water losses (Kennes and Thalasso 1998; Delhoménie and Heitz, 2005). Thus, besides air pre-humidification, biofiltration systems are also equipped with sprinkler systems (top-mounted irrigation or soaker hoses) to supply additional water directly to the surface of the biofilter bed.

Moisture can be controlled on-line with cost-efficient sensors (Sabo 2005). For example, Chen et al. (2009) used a soil moisture sensor. However, media moisture content is still difficult to control, and reliability on moisture readings is limited (Sabo 2005). Apart from on-line sensors, some techniques are available for estimating the water content: weight measurements of the whole filter bed with a load cell or laboratory analyses of media samples taken from the biofilter (Kennes and Veiga 2001). Moisture supply rates must take into account that the frequency of water sprinkling over the packing media needs to be increased as loading rates increase, mainly due to an increase in biological activity that generates more heat and therefore increases evaporation. A further consideration is that moisture content is not uniform throughout the bed, with lower values at the top section due to a higher metabolic activity related to the higher received contaminant concentration.

3.2.5 Temperature

Most biofilters utilise mesophilic microorganisms, with an ideal operating temperature (**Figure 4**) for the biofilter bed ranging from 15 to 40°C (McNevin and Barford 2000; Easter et al. 2005; Iranpour et al. 2005; Pagans et al. 2005). Below 15°C, biological activity slows down, while at high temperatures mass transfer and adsorption rates will decrease. Although temperature is usually not considered as the main operation parameter, a decrease of 10°C in the biofilter bed temperature can reduce its performance by more than 50%, while within the proper range, this same variation can double microbial growth rates (Kennes and Veiga 2001; Kennes et al. 2009). Although efficient biofiltration at temperatures as low as 10°C has been reported (Cho et al. 1992), enclosed insulated beds are recommended in colder climates.
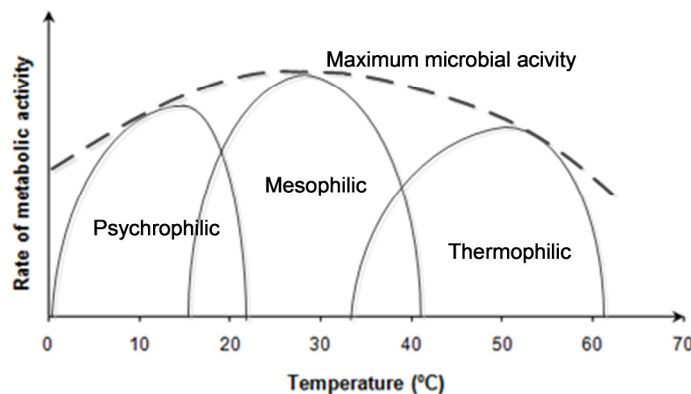
Figure 4 Effect of temperature on microbial activity in the biofilter (adapted from (Devinny et al. 1999; Kennes and Veiga 2001)).

High temperatures may be observed due to hot inlet gas streams or to metabolic activity, which can increase the biofilter temperature by 2-10°C, especially near the inlet where higher substrate concentrations are found (Morales et al. 1998). Above the mesophilic range, the microbial population will shift from mesophile to thermophile species (Van Harreveld 2007). Biofiltration at high temperatures accelerates the drying of the carrier material, providing challenges with regards to maintaining a homogeneous water distribution through the biofilters, and lowering the solubility of the contaminants being abated, as well as oxygen.

Cooling through dilution with cooler air, or the application of heat exchangers my improve

biofilter performance when treating high temperature inlet gas streams, but will significantly increase treatment costs. The impact of cooling on abatement performance was demonstrated by Phillips et al. (2003) for a biofilter treating composting exhaust gases. VOC and odour removal efficiencies were increased from initial values of 58% to over 94% when inlet gas temperature was reduced from the initial 55ºC to 35ºC (Phillips et al. 2003). Van Harreveld (2007) also reported that odour concentrations for the treated gas would not decrease below 1000 OU_E/m^3 at temperatures above 37ºC.

4.0   Operation and Maintenance

4.1   Media Conditioning and Inoculation

Biofilter media must be conditioned to ensure optimal moisture content prior to being distributed into the bed to avoid irregular airflow circulation through the media (Kennes and Veiga 2001). Furthermore, if the media is not sufficiently pre-moistened, it may not be able to retain sufficient water during operation to reach the high level of moisture required for optimal operation (Kennes and Veiga 2001). Smoke tests might be performed to ensure uniform material distribution and to assess for short-circuiting and air channelling prior to inoculation and start-up.

As a general rule, for organic media with readily biodegradable contaminants in the air stream being treated, it is usually not necessary to inoculate the biofilter media since the material will contain indigenous microorganisms capable of biodegrading the odorous compounds (Kennes et al. 2009). For recalcitrant or contaminants with low biodegradability, inoculation of specialized biocatalysts is usually required (Adler 2001; Kennes et al. 2009). Inoculation is also necessary when inert or synthetic packing materials are used.

The most common method of inoculating biofilters is through the application of activated sludge from a wastewater treatment plant, although a suspension of previously cultivated and acclimatized microorganisms (pure, mixed or genetically engineered cultures) can also be used (Kennes and Thalasso 1998). For biofilter inoculation with pure cultures, specialized bacterial inocula are often used, while fungi are also very popular for co-metabolic removal of semi-volatile complex aromatic substrates as well as for VOCs (Kennes et al. 2009). Although it is usually favourable to start-up the biofilter with a large and diverse microbial community, high removal efficiencies and elimination capacities have been achieved with single strains or defined consortia (Kennes and Veiga 2001).

The amount of inoculum employed differs widely depending on the process. Wabster et al. (1996) used 15 L/m^3 of activated sludge from the dissolved air flotation unit, and inoculated another biofilter with a suspension of Thiobacillus thioxidans from a sulfide-rich corrosive environment. Andres et al. (2006) inoculated using activated sludge at a rate of 125 g dry weight/m^3 bed. Similarly, Smet et al. (1996) inoculated their compost biofilter for DMS

treatment with ~120 g volatile solids (Hyphomicrobium MS3)/m³ reactor. Duan et al. (2006) inoculated the activated carbon with activated sludge by submerging and covering the carbon bed with the bacteria solution and blowing a synthetic waste gas with a low H_2S concentration into the column for a 1 week period. Despite the wide range of inoculation rates and techniques, it is believed that the initial biomass concentration only influences the time to reach the steady state, and not the final performance (Li et al. 2002).

4.2 Start up and microbial acclimation

Start-up activities are crucial for efficient operation. Immediately after start up, high removal efficiencies might be observed due to the dominance of the adsorption process. This first stage is followed by decreasing removal performance attributable to saturation of the adsorption capacity and to a lack of sufficiently microorganism community. The performance will then increase as biodegradation becomes the controlling step in the removal process (Kennes and Veiga 2001).

To allow the process to reach the point where biodegradation becomes the controlling step (and efficient abatement is achieved), the biofilter must be initially operated to allow microorganisms to fully adapt to the contaminants and biofilter conditions such as nutrients, pH, and temperature (an acclimation period). The acclimation period corresponds to the time required for microbes to synthesize the enzymes needed for contaminant biodegradation and the time needed to produce enough biomass to cope with the inlet contaminant concentrations. The time required by the microbial population to reach the steady state is unpredictable, ranging from few days to weeks or even months (Amirhor et al. 1997; Park et al. 2001; Andres et al. 2006; Duan et al. 2006). For an activated sludge inoculum in a wood bark biofilter, Andres et al. (2006) observed a 29 day acclimation period to reach steady state for ethanol removal, and nearly 10 weeks for less biodegradable compounds such as dicloroethane. A similar acclimation period (6 weeks) was observed by Amirhor et al (1997) for a wood bark biofilter treating H_2S and NH_3. Park et al. (2001) observed that one month was needed for a ceramic biofilter inoculated with activated sludge mixed liquor from a wastewater treatment plant to be effective. A much shorter acclimation period (6 days) was observed by Duan et al. (2006) for an activated carbon biofilter inoculated with a pre-adapted activated sludge. Thus, the acclimation period strongly

depends on the microorganisms inoculated to the bioreactor, the biodegradability of contaminants present in the waste gas stream and their concentration.

4.3 Key Operating Parameters

4.3.1 Pre-treatment

Pre-treatment of the odorous stream should be considered if the stream contains dust, particulates, or high concentrations of water soluble contaimants such as ammonia or condensable gases. The pre-treatment stage also allows for the adjustment of the inlet stream temperature and moisture content to promote improved operation. The presence of dust or particulate matter in the air stream can fill up the pore spaces faster than microorganisms can break them down, thus increasing pressure drop and shortening the media life (Adler 2001). Adler (2001) suggested a maximum concentration of particulate matter of 10 mg/m^3 to minimize bed clogging, which can be achieved by scrubbing or filtering of the inlet stream. Reduction of the concentration of specific contaminants, such as ammonia, can also lead to improved abatement performance by avoiding high, toxic concentrations in the biofilter. Amirhar et al. (1997) achieved a 10% increase in odour removal by ammonia pretreatment in a mist scrubber with sulfuric acid solution that removed more than 50% of the inlet NH_3. Shanchayan et al. (2006) reported high H_2S and total reduced sulfur compounds (TRS) removal efficiencies in a biofilter treating offgases from aerobic digestion when the inlet stream was pretreated in a wet scrubber operated with city water to reduce NH_3 concentrations, humidify the air, remove particulates and adjust the temperature of the off-gas.

4.3.2 Pressure Drop

Pressure drop is one of the main considerations for biofilter operation. Pressure drop across the biofilter bed initially depends on the characteristics of the packing material (such as porosity, bed voidage, bulk density, particle size), as well as on the depth of the medium and the gas surface loading rate. However, as the biofiltration process takes place, other parameters such as biofilm growth, excess of moisture content or clogging/compaction of the media will significantly contribute to pressure drop increase (Morgan-Sagastume et al. 2001;

Delhoménie & Heitz 2005). The increase in biomass thickness during operation will increase the pressure drop across the bed, Andres et al. (2006) observed an increase in the pressure drop across a biofilter from 0.04 to 1 cm H_2O after 83 days.

Gas phase pressure drop generally increases linearly with gas linear velocity until a certain flow rate value, while increasing exponentially with diminishing particle size, especially at sizes below 1 mm (McNevin and Barford 2000; Chen et al. 2009). As discussed previously, synthetic/inorganic media generally lead to lower pressure drops than organic media, thus they are usually mixed into other media types (such as wood chips, or compost) to reduce the pressure drop and optimize the flow characteristics. **Figure 5** illustrates the pressure drop in a compost-perlite biofilter of one meter height as a function of the percentage of compost in the mixture (Lebrero et al. 2011). Increasing the amount of inorganic material led to a reduction of pressure drop across the filter bed.
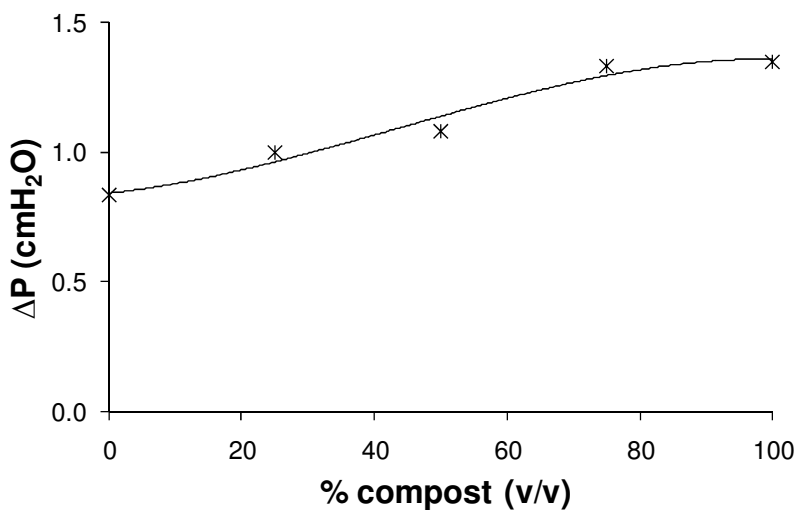
Figure 5 Effect of % in volume of compost on pressure drop in a compost-perlite biofilter (Lebrero et al. 2011)

The maximum economically-tolerable pressure drop (due to compaction and thus increase energy requirements) across a biofilter bed ranges between 6 and 8.4 cmH_2O/m bed (Liu, Li et al. 2009). Below this pressure drop range, no compaction problems are typically observed, and this range usually corresponds to packed bed heights less than 1.2 meters (Liu, Li et al. 2009), with higher bed heights leading to media compaction and correspondingly higher pressure drops. A lower pressure drop limit was suggested by Chen et al. (2009), who

recommended that biofilters run at pressure drops below 0.6 cm H_2O to maintain an acceptable level of ventilation fan efficiency. Differing recommendations have been provided with regards to what level of pressure drop signals the need for media replacement, with recommendations ranging between 20.3 and 33 cmH_2O/m media (Gueissaz-Teufel 2004; Elias et al. 2002; and Adler 2001).

Bed moisture content also influences the pressure drop, with an increase in moisture content increasing the pressure drop, particularly for less porous media (Luo 2001). When moisture content level increases over 60%, an increase in pressure drop up to 0.2-0.25 m H_2O might be expected (Kastner and Das 2005).

In addition to operating concerns, the pressure drop is directly linked to operating cost, since it determines the required energy to move air through the process (blower energy). According to Leson (1995), an increase in pressure drop from 4 to 25 cm H_2O will increase energy requirements from 7 to 27 kW.

4.3.3 pH

Microbial inhibition and packing deterioration can occur during the long term operation of biofilters due to sulfuric acid production from H_2S biodegradation. An increase in sulfate concentration decreases the pH, creating an excess in conductivity and suboptimal osmotic conditions for the biomass. Other produced acids, such as acetic acids, ethyl acetate or hydrochloric acid, also contribute to medium acidification (Van Harreveld 2007; Kennes et al. 2009). The optimal pH range for biofiltration has been reported to be around neutral values (6-8) for most VOC biodegrading bacteria, and especially for microorganisms degrading halogenated substrates, which are very sensitive to medium acidification (Van Harreveld 2007; Kennes et al. 2009). Other microorganisms, such as fungi or sulfur degrading bacteria, can tolerate very acidic or alkaline conditions. Some studies have demonstrated that, for such microorganisms, pH as low as 2 do not affect contaminant degradation (Webster et al. 1996). However, although at low pH values H_2S removal still remain high, the abatement performance in terms of odour removal tends to deteriorate, particularly if alkyl sulfides and mercaptans are present (Phillips 2009).

Control of pH is seen as a difficult and expensive task in biofilters, and frequently not economically feasible at field scale. Although periodic irrigation with water can contribute to pH control, it is usually inefficient to maintain pH at neutral values and chemicals must be continuously added to neutralise the acids generated (Duan et al. 2006). Furthermore, rinsing can leach out essential microbial nutrients from the system (Smet et al. 1996; Kennes and Veiga 2001). The addition of chemicals such as NaOH to neutralise the acids can adversely affect process performance due to the accumulation of high salt concentrations, which can have an inhibitory effect on the microorganisms in the biofilter.

Acidification of the media can also be avoided by addition of a buffer agent such as lime, $Ca(OH)_2$, dolomite or oyster shells to the biofilter media. This is, however, a labour intensive process that increases operating costs. Biofilter media can also be mixed with limestone, with the amount of limestone to be added based on the stoichiometric neutralization of the metabolite produced if the inlet concentration in the waste gas is known (for example $CaCO_3 + H_2SO_4 \rightarrow CaSO_4 + CO_2 + H_2O$) (Smet et al. 1996; Chitwood and Devinny 1999). Carbonate solubilization increases the pH and neutralizes the acidity generated by the abatement of sulfur compounds. Mixing a nutrient-rich carrier material with limestone powder is a less expensive alternative which has shown good results in biofiltration systems (Smet et al. 1996). One of the major drawbacks of limestone application is $CaSO_4$ coating, which can create clumps and lead to channelling and short circuiting in the bed (Chitwood and Devinny 1999). Furthermore, spent minerals and acid-degraded compost often forms small particles and can contribute to biofilter clogging. Typically the biofilter medium is replaced when the alkalinity is completely exhausted (Chitwood et al. 1999; Gueissaz-Teufel 2004).

An additional alternative to avoid the inhibitory effects of low pH on the removal of VOCs/MTBE/mercaptans is to separate the biofilter into two different sections with the operating conditions of each section optimised for the target compounds to be removed. Following this methodology, Converse et al. (2003) obtained removal efficiencies over 99% for H_2S and MTBE in a two-stage lab-scale biofilter.

4.3.4 Leachate Control and Drainage

Leachate produced during the biofiltration process must be contained, collected and disposed-of. A significant amount of the compounds trapped by the biofilter will be removed from the media in the drainage, which also contains sulfate salts (Na^+, Ca^{2+}, K^+, Mg^{2+}) from sulfate leaching, responsible for the conductivity in the percolate solution (Quigley et al. 2003; Easter et al. 2005). For biofilters located at wastewater treatment plants or sewer networks, leachate can be discharged back into the sewer or the headworks of the WWTP since it is a minor contribution to the influent stream. The collection and drainage piping have to be adequately sized to handle the maximum expected drainage load (Easter et al. 2005).

4.4 Maintenance and Long Term Operation

The most common reported sources of biofilter failure include failures in the moisture monitoring and control system, rapid degradation of the media or clogging by particulates or dust (which reduces the availability of adsorption/absorption sites), excessive variation of the inlet concentrations, accumulation of toxic and/or acidic by-products, excessive pressure drop, corrosion of ductwork or fan parts, overheating, and flooding of media (Adler 2001). Preventative actions recommended to improve biofilter operation and reduce the potential for process failure include (Wade 1999; Adler 2001; Alix and Williams 2006):

- Visual inspection and sampling for the media pH and moisture content at different bed heights.
- Listening for unusual sounds or equipment vibration, particularly from fans and blowers.
- Monitor inlet and outlet contaminant concentrations at the biofilter outlet.
- Monitor leachate pH and quality.
- Evaluate data and assess trends.

It is also recommended to check the pressure differential in the bed and its physical integrity, which will allow for detection of cracks and short-circuiting, and perform smoke tests at least twice a year. Occasionally, biofilter can be treated to prevent weed and deep root vegetation

growth (Wade 1999; Adler 2001; Alix and Williams 2006).

An important area of concern with biofilter operation is maintaining adequate treatment performance during peak loads, which is generally achieved over-designing the systems. After an overloading of the system, or when concentration peaks occur in the inlet stream, biofilters are able to recover their initial removal efficiencies in relatively short times, usually few hours (Zhuang et al. 2001; Jiang et al. 2008). An alternative approach to dealing with peak loads was demonstrated at the laboratory scale by Jiang et al. (2008) using a biofilter packed with exhausted carbon. The biofilter exhibited much higher removal efficiency in response to spike H_2S loadings due to the additional sulfur source for bacteria and the adsorption capacity of the activated carbon (Jiang et al. 2008). Packing materials with high adsorptive capacities will allow the biofilter to withstand fluctuations in loading rates, and the microorganisms can utilise the absorbed compounds to meet metabolic requirements during lean loading periods.

Nutrient depletion (such as nitrogen or phosphorous) during long-term operation can negatively impact biofilter performance. Several studies have demonstrated that the addition of nitrate solution or injection of ammonia gas can recover biofilter performance and increase VOCs elimination (Moe and Irvine 2001). Liu et al. (2009) found that phosphorous content of the biofilter media decreased by 27%, probably due to incorporation of insoluble phosphorous into biomass, thus supplementation of soluble phosphorous is also necessary to avoid nutrient limitation.

5.0 Process Performance

The majority of the reviewed studies on laboratory scale processes focused on removal of a single contaminant or a mixture of few selected compounds from a synthetic stream at high concentrations (in the range of g/m^3) and under constant operating conditions, circumstances rarely found in real waste air streams (Iranpour et al. 2005). Under these ideal conditions, biofilters have shown high removal efficiencies for odorous sulfur and amino-acid compounds, oxygenated aliphatics, aromatics and chlorinated compounds (Iranpour et al. 2005; Kennes et al. 2009).

The performance of full-scale biofilters in field applications can be affected by unsteady conditions (temperature, humidity, etc.) and discontinuous contaminant supply (system maintenance or breakdowns). Thus, although field experiences always report high H_2S removal efficiencies (usually between 90% and 100%), and observed odour reductions over 80%, removal performance of other odorous sulfur and volatile organic compounds varies from values as low as 20% up to 100%, being usually lower than 90% for easily biodegradable VOCs (Iranpour et al. 2005). This significant difference between laboratory and field performance is due to field loading patterns, the complexity of the emissions being treated, and contaminant concentrations that are orders of magnitude smaller than those employed in the laboratory.

5.1 Key Performance Parameters

5.1.1 Removal Efficiency

The removal efficiency (RE) is the ratio of the contaminant removed (difference between inlet and outlet contaminant concentrations, C_{in} and C_{out}, respectively) to the amount of contamiant fed to the biofilter, usually expressed as percentage:

$$RE = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \ (\%) \quad \textbf{Eq. 18}$$

The removal efficiency will depend on the biofilter configuration and operation, as well as on the contamiants to be removed. Usually, when mass loading rate increases, removal efficiency will gradually decrease (Kennes and Veiga 2001).

5.1.2 Substrate Mass Loading

The substrate mass loading rate (Sl) is the mass of contaminant fed to the process per unit time and unit volume of packing material.

$$S_l = \frac{Q}{V} C_{in} \ (g/m^3h) \ \ \textbf{Eq. 19}$$

Depending on the required removal efficiency, the amount of contaminant that can be treated in the biofilter can vary widely, which in turn depends on the characteristics of the waste gas, the nature of the contamiants and the mass transfer potential of the reactor (Kennes and Veiga 2001). The higher the desired removal efficiency, the lower the S_l. According to Kennes and Veiga (2001), complex substrate mixtures and poorly biodegradable or recalcitrant compounds will require low S_l unless low removal efficiencies are acceptable.

5.1.3 Elimination Capacity

Elimination capacity is the amount of contaminant that the biofilter is able to remove per unit volume of carrier material and unit time. The maximum elimination capacity (EC) is often specific to the process, and is affected by bed material, porosity, inlet contaminant concentration and the biomass concentration.

$$EC = \frac{Q}{V}(C_{in} - C_{out}) = \frac{RE \times Q}{V} C_{in} = RE \times LR \ (g/m^3h) \ \ \ \textbf{Eq. 20}$$

High elimination capacities are usually desired to reduce the size of the biofilter and correspondingly the investment costs.

5.2 H₂S and Sulfur Compound Removal

Biofilters have a demonstrated ability to efficiently remove H_2S, with typical reported REs between 90 and 100% (Easter et al. 2005). For a biofilter packed with activated carbon, Duan et al. (2006) recorded H_2S REs greater than 97% at 2 s EBRT with inlet H_2S

concentrations up to 30 ppmv, while Jones and Bañuelos (2000) reported 99.9% H_2S RE in biofilters packed with compost and wood chips treating inlet streams with H_2S concentrations ranging from 50 to 400 ppm. However, the removal efficiencies for reduced sulfur compounds are usually lower and more variable than those obtained for H_2S (Easter et al. 2005). For example, Converse et al. (2003) recorded average H_2S REs of 99.3% at inlet concentrations ranging from 10 to 50 ppm, and between 91-94% for methyl mercaptan. However, REs <40% were observed for dimethyl sulfide and carbon disulfide (Converse et al. 2003). Easter et al. (2005) compiled data for 6 soil-based and 11 compost-based biofilters. The results indicated that while both systems were able to efficiently remove H_2S (REs >99%), there were much lower REs for other sulfur compounds, 84% for mercaptans and 37-60% for DMS. In another study, Easter et al. (2004) compiled information from 24 biofilters with different packing media. Similar H_2S REs (98%, 99% and 90%) were obtained for soil-based, organic-based and inert media biofilters respectively (Easter et al. 2004). Biofilters packed with inorganic materials performed worse for removal of reduced sulfur compounds, with very low and variable REs, while organic-media biofilters achieved 90% RE for methyl mercaptan, and REs of 70% and 85% were observed for carbonyl sulfide and mercaptans (Easter et al. 2004).

Since the degradation of reduced sulfur compounds is not limited by their water solubility and partition coefficient, the observed low removal efficiencies appear to be associated with microorganism metabolism (Quigley et al. 2003; Rappert and Müller 2005).

5.3 Ammonia Removal

Ammonia is one of the primary odorants in odour emission from composting, with highly variable concentrations that increase during the summer due to higher temperatures, up to 700 mg NH_3/m^3 (Park et al. 2002; Pagans et al. 2005). One of the primary challenges for ammonia removal is the slow growth rates exhibited by nitrifying bacteria. To address this issue, it has been recommended that the media for biofilters used to removing NH_3 be based on are compost made from sewage sludge, animal manure, industrial wastes from food processing, or source separated household waste; all materials that contain relatively high numbers of nitrifying bacteria (Poulsen and Moldrup 2007). The use of pine bark as packing material is also recommended since it provides good pH conditions for ammonia removing

microorganisms (Gaudin et al. 2008).

Ammonia removal from composting odorous emissions has been extensively reviewed, usually showing high REs (over 95%) on a wide range of support materials, both organic and inorganic (Pagans et al. 2005; Colon et al. 2009). The biological activity of NH_3 degrading microorganisms can be inhibited by high inlet NH_3 loading rates or dry zones in the bed. Accumulation of free NH_3 and nitrification products will reduce the nitratation rate (conversion NO_2^- to NO_3^-) more severely than the nitritation (conversion of NH_3 to NO_2^-) rate, and the accumulation of NO_2^- causes inhibits NH_4^+ conversion (Poulsen and Jensen 2007). Van Harreveld (2007) reported complete nitrification inhibition in the filter at concentrations above 4 g $NH_4^+ - N$/kg wet material, and above 6 g NO_3^-/L microbial activity is reduced (Van Harreveld 2007). Pagans et al. (2005) observed REs in the range 98.8-99.4% for inlet NH_3 concentrations up to 8500 mg/m³ at EBRT of 86s, and reported microbial activity inhibition for NH_3 concentrations higher than 2000 mg/m³, and a critical load of 27 000 mg NH_3/m³h. Gabriel et al. (2007) observed inhibition of ammonia removal at inlet concentrations over 120 ppm. Typical NH_3 concentrations in exhaust gases from the composting processes are usually below the inhibition limit (Colon et al. 2009).

Another important parameter influencing NH_3 removal is the water content of the bed material (Poulsen and Jensen 2007). Poulsen et al. (2007) recorded near zero REs at low water contents near air-dry conditions, while achieving REs greater than 90% at high water contents. This improvement is due to enhanced transfer of NH_3 from the air phase into the water and subsequently into the biofilm at higher moisture contents.

5.4 VOC Removal

While control of emissions containing VOCs and hazardous air pollutants is feasible, typical REs of VOCs and hazardous air pollutants (HAPs) are lower than those observed for other odorous sulfur compounds such as H_2S (Iranpour et al. 2005; Kennes et al. 2009). Biofiltration is a cost effective technology for treatment of off-gases with low soluble-VOC concentrations (typically below 3 g/m³) and commonly found inlet concentrations of VOCs in waste gases are in the ppb level (Webster et al. 1996; van Groenestijn et al. 2001;

Iranpour et al. 2005). The effectiveness of biofiltration for the elimination of hydrophobic compounds such as aromatics, alkenes and alkanes is limited due to the low concentration gradients available for mass transfer.

Removal of VOCs in biofilters has primarily been investigated at lab-scale, with synthetic single-contaminant streams, and focusing on industrial emissions which typically have much higher VOC concentrations than those found in odour emissions from biological sources (such as sewers, wastewater, agriculture, and livestock). Reported REs in pilot scale plants for VOCs vary widely from 20-95%, and complete removal of aromatics requires an EBRT greater than 3 min (Liu et al., 2009; McNevin and Barford, 2000; Chitwood and Devinny, 1999). Liu et al. (2009) measured high REs for alkylated benzenes (ranging from 82% to >90%), low molecular weight (MW) alkanes (85%-91%), and sulfur compounds (>99%), while the measured REs of terpenes and high MW alkanes were < 64%, and negative REs were obtained for some compounds such as β-pinene.

The use of fungi based biofilters has been can investigated as a solution for abating hydrophobic VOCs, since fungi have a higher affinity for these compounds, as well as a greater resistance to dry conditions and low pH values (Kennes et al. 2009). Van Groenestijn et al. (2001) observed REs >99% in a biofilter treating toluene with a fungal community, and similar results were obtained by Webster et al. (1996), with removals of aromatic hydrocarbons, aldehydes and ketones greater than 80%. However, despite the higher resistance of fungi to adverse environmental conditions, they usually lead to higher pressure drops in the biofilter bed due to their filamentous nature (Kennes et al. 2009).

The abatement of chlorinated hydrocarbons is seen as a particular challenge to biofiltration processes, Converse et al. (2003) observed nil removal of chlorinated hydrocarbons (chloroform, trichloroethylene or dichlorobenzene), and concluded that removal of these substances may require either co-metabolism or anaerobic conditions, which are difficult to control in biofilters. Application of fungi based biofilters for the removal of chlorinated hydrocarbons demonstrated similar limitations, with very low REs for chlorinated hydrocarbons such as chloroform, vinyl chloride and trichloroethylene (Webster et al. 1996). Improved removal of chlorinated hydrocarbons has been observed in activated carbon-

packed biofilters, with REs up to 98%, which is likely due to adsorption into the carbon and the formation of anaerobic zones inside the carbon particles where reductive dehalogenation occurs (Webster et al. 1996).

5.5 Odour Removal

Biofilters are generally effective at removing odours, with typical REs greater than 90% as indicated by several studies surveying larger numbers of processes (McNevin and Barford, 2000; Easter et al. 2004; Easter et al. 2005). In an evaluation of 17 full-scale organic-based biofilters, Easter et al. (2005) measured an average odour RE of 95%. In an earlier study of a wider range of biofilter types (24 process sample size), Easter et al. (2004) determined average odour REs of 95%, 98% and 67% for soil-based, organic-based and inert media biofilters, respectively. A study of 123 biofilters by Van Harreveld (2007) demonstrated that biofilters can treat a wide range of raw gas odour concentrations (up to several hundreds of thousands of OU_E/m^3) with consistent REs and outlet concentrations below 10 000 OU_E/m^3. For the biofilters tested, 4% achieved outlet odour concentrations under 500 OU_E/m^3, 14% under 1000 OU_E/m^3 and 52% under 2500 OU_E/m^3 (Van Harreveld 2007).

A summary of specific biofilter applications (laboratory, pilot and full scale) presented in the reviewed literature is provided as **Table 4**. Emphasis is placed on available results with regards to contaminant removal performance, specific operating parameter values, and process costs. It should be noted that unless otherwise indicated, all costs are presented as USD.

Table 4 Removal performance, operation parameters and costs of industrial, pilot and laboratory-scale biofilters

Application	Scale	Source Type	Source Characterization	RE	Treated flow rate (m³/h)	Packing media	EBRT (s)	Comments	Cost Capital	Cost Operation	References
Odour control at a large metropolitan WWTP	Industrial	H_2S	100 ppm >280 ppm	99.8% 93%	10704	Wood bark	56		_	_	(Zhuang, Keener et al. 2001)
Odours from industrial and domestic wastewater treatment	Industrial	H_2S MeSH DMS DMDS	0.933 ppm 0.94 ppm 21.2 ppm 1.48 ppm	97.9% 98.3% 79.9% 99.7%	4800	Organic media	_	Overall removal in a two stage biofilter	$39.23 /(m³/h)	_	(Alix and Williams 2006)
Wet well exhaust air	Industrial scale	H_2S	5-50 ppm	No perceptible odors	1700	_	60	Wall biofilter. <1 hour maintenance/ week	$88.3 /(m³/h)	_	(Gueissaz-Teufel 2004)
Wastewater collection system exhaust air	Industrial scale	H_2S	0.1-12 ppm	99.9%	8495	_	60	In-ground biofilter	$16.5 /(m³/h)	_	(Gueissaz-Teufel 2004)
Autothermal thermophilic aerobic digestion offgas	Industrial scale (258 m³)	NH_3 H_2S MeSH DMS DMDS	250-420 ppm 4-20 ppm 1.5-4.8 ppm 0-0.25 ppm 0-0.25 ppm	75-86% 95-99% 99.8-99% 75.3-94.8% 86.7-95.7%	5160	Coarse chipped and compost root wood	180	Open BF NH3 pretreatment in a water-wet scrubber	_	_	(Shanchayan, Parker et al. 2006)
Off-gas from headworks, grit chamber and grit classifier from a WWTP	Industrial scale (85 m³)	H_2S MeOH/acetone Other VOCs	0.43 ppm	98.5-99,6% ~100% 56.6%	13600	Wood chips + lava rock	16-30	Two stage biofilter	¬_	_	(Chitwood, Devinny et al. 1999)
Composting plant off gases	Industrial scale (74-164 m³)	NH_3 VOCs	0.08-1.4 g C/m³ 0.004-0.142 g NH_3/m³	89-100%/41-74% 71-74%/42-65%	3950-15800	Wood chips New/reused from composting	25-98	Study conducted during material replacement	_	_	(Colon, Martinez-Blanco et al. 2009)
Foul gas from a municipal solid waste treatment facility	Industrial scale	NH_3	13-80 ppm 125 ppm peaks	50-100%	345000	Coconut fibre	_	Low REs due to dried zones and short-circuiting	_	_	(Gabriel, Maestre et al. 2007)
Odors from asphalt processing plant	Industrial scale (44 m³)	H_2S Hydrocarbons	16-30000 _	40%-100% Negligible	425	Wooden materials (proprietary mixture)	366	Reduce odour complaints. Media bed pH=2	_	_	(Blake, Cook et al. 1999)

Table 4 Removal performance, operation parameters and costs of industrial, pilot and laboratory-scale biofilters (Continued)

Application	Scale	Source Type	Source Characterization	RE	Treated flow rate (m³/h)	Packing media	EBRT (s)	Comments	Cost Capital	Cost Operation	References
Exhaust air from organic waste composting facility	Field scale (12 m³)	H₂S NH₃	3.3-5.5 ppm 8-90 ppm	95%-98% odour removal	6000-9000	Ceramic (fly ash and diatomite)	7.2	Closed biofilter Inoculated with AS Wet scrubber pre-treatment	$9.7 /(m³/h)	$0.9 /(m³/h) per year	(Park, Nam et al. 2001)
Waste gases from a dairy industry	Field scale (75 m³)	H2S MeSH DMS+DMDS NH₃ VOC	5 mg/m³ 1 mg/m³ 1 mg/m³ 3 mg/m³ 5 mg/m³	60%-99%	4000	Coconut fibre/Fibrous peat	40	No inoculation, with one month for acclumation	–	–	(Kennes 2001)
Odours from solids processing and biosolids composting facility	Pilot scale (19 m³)	H₂S NH₃ MeSH	0.01-3 ppm 1-2 ppm <1.2 ppm	84%-98% - -	49270-83250	Wood barks and compost	38-64		$40.4 - $68.3 /(m³/h)	–	(Arts 2004)
Odours from municipal solid waste treatment plants	Pilot scale (0.18 m³)	VOCs (overall) Alkylated benz. Alkanes S-compounds Terpenes	0.68-87.2 ppmv	20%-95% >90% 10%-85% >99% 0%-64%	10-20	Mature compost, no inoculation	32.5-65	Operated for 140 days	—	— Estimated full-scale	(Liu, Li et al. 2009)
Odours derived from a lift-station	Pilot scale (8.4 m³)	H₂S DMS	1-12 ppm 3-15 ppm	>90%	360	Two media layer, no media information	67-102	Inoculated with returned activated sludge	–	–	(Zappi, Dufreche et al. 2004)
Odours from headworks building of a WWTP	Pilot scale (0.53 m³)	H₂S MeSH S-compounds Aromatic VOCs Chlorinated VOCs	10-50 ppm 150-165 ppb 2-52 ppb 0.5-150 ppb 4-140 ppb	99.3% 91-94% 0-36% 56-79% 0-75% Odour RE: >99%	20.4-42.5	Compost, perlite and oyster shells	42	Two-stage system (separated VOCs and H2S removal sections)	–	–	(Converse, Schroeder et al. 2003)
Foul gas from a municipal solid waste treatment facility	Pilot scale	NH₃	90-300 ppm	100%	0.63	Coconut fibre	19-36	Removal inhibition at CNH3>120 ppm right after start up	–	–	(Gabriel, Maestre et al. 2007)

Table 4 Removal performance, operation parameters and costs of industrial, pilot and laboratory-scale biofilters (Continued)

| Application | Scale | Source | | RE | Treated flow rate (m³/h) | Packing media | EBRT (s) | Comments | Cost | | References |
		Type	Characterization						Capital	Operation	
Foul gases from a swine facility	Pilot scale (0.11 m³ x 18 BF units)	H_2S NH_3	–	90%-95% 81% Odour RE 78.8%	81.5	Compost + wood chips	5	Compost content must be >30% wt. High influence of moisture	–	–	(Nicolai and Janni 2001)
Odorous emission from a swine facility	Pilot scale (0.12 m³)	H_2S NH_3 VFA Phenolics/indolics	1.5-6.3 ppm - - -	92.5%-100% 61.3%-79.8% 97.6%-99.3% 100% Odour RE: 76-90%	61.5-136	Wood chips	4-7.3	Important RE reduction at low moisture content	–	–	(Chen, Hoff et al. 2009)
Odours from animal rendering process	Pilot scale (0.28 m³)	143100-890000 OU/m³		86.6%-96.6% 95%-99.8%	1.2-9.9	Sand Wood bark	32-456	3 years operation	–	–	(Luo 2001)
	Lab scale	H_2S VOCs and chlorinated VOCs	1-10 ppmv 1-75 ppbv	100% 50-90% 0-45%	42-168	1. Granular activated carbon 2. Yard waste compost	17-70		–	–	(Webster, Devinny et al. 1996)
VOCs treatment	Lab scale	Synthetic mixture: Ethanol MEK Dicholomethane Toluene	(mg/m³) 0.39 0.09 0.11-0.22 0.078	60-80% 60-80% 20-50% 100%	20	Wood bark	37-46	Competitive/ toxic effects of dichloromethane addition	–	–	(Andres, Dumont et al. 2006)
H2S treatment	Lab scale	H_2S	10-125 ppmv	94%-97%	34.2-240	Pellet activated carbon	2	Inoculation with acclimated activated sludge	–	–	(Duan, Koe et al. 2006)
H2S treatment	Lab scale	H_2S	20-160 ppm	94-99%	0.05-0.23	Exhausted activated carbon	4-20	Inoculation with return AS previously enriched	–	–	(Jiang, Yan et al. 2008)

Table 4 Removal performance, operation parameters and costs of industrial, pilot and laboratory-scale biofilters (Continued)

Application	Scale	Source		RE	Treated flow rate (m³/h)	Packing media	EBRT (s)	Comments	Cost		References
		Type	Characterization						Capital	Operation	
H2S treatment	Lab scale	H$_2$S	<700 ppm 700-900 ppm	99.99% 89%-97.5%	0.1-0.2	Lava rock	12-18	Inoculation with Thiobacillus Thiooxidans	–	–	(Cho, Ryu et al. 2000)
VOCs treatment	Lab scale	Toluene	400-700 mg m-3	>99%	–	Perlite // lava stone	–	Perlite/fungi system: increase costs	5-8 €/(m3h-1)	0.5-1 €/1000 m3gas treated	(van Groenestijn, van Heiningen et al. 2001)

TRS: total reduced sulfur compounds

5.6 Costs

Biofiltration costs are highly application-specific and depend on the flow rate, the concentration and biodegradability of the contaminants, the desired RE, the reactor design and type of media, the level of monitoring and control, and the materials of construction. More specifically, capital costs are dependent on the complexity of the biofilter internals, its configuration, and the degree of sophistication of the control and maintenance system (Adler 2001; Estrada et al. 2010). Operating costs include electricity, water and steam usage, direct labour, maintenance materials and media replacement (Adler 2001). Due to the reticence of manufacturers to provide cost data with regards to their process, very little cost data (capital, operating, or maintenance) was obtained in the reviewed literature.

The major contribution to the investment costs is the price of the filter bed. According to Kennes and Veiga (2001), it may vary from less than 125 $/m^3 for organic carriers (compost 50-60 $/m^3, heather 125-150 $/m^3, wood bark 75-93 $/m^3) to more than 1200 $/m^3 for some synthetic or non-natural media. The economy of scale are of particular concern with biofiltration processes, Haecker et al. (2004) demonstrated an inverse relationship between flowrate and the cost per m^3/h treated gas. Using a 20-year present worth analysis for biofilters treating different inlet H$_2$S concentrations at different air flow rates, Haecker et al. (2004) concluded that biofiltration becomes feasible only when airflow rates exceeded 425 m^3/h, with total costs of $294.3/(m^3/h), $127/(m^3/h), $100/(m^3/h) and $77.8/(m^3/h) at flow rates of 170 m^3/h, 425 m^3/h, 850 m^3/h and 1530 m^3/h, respectively.

The literature has identified biofilters as a low cost treatment option when compared to physical and chemical methods, with the primary advantage of not requiring the use of chemicals which in addition to the acquisition costs, also incur shipping, storage and pumping costs (Estrada et al. 2010). Available estimates of the relative operating and maintenance costs of biofilters relative to chemical treatment systems are highly variable, ranging from being 95% to 50% lower cost than the equivalent chemical based treatment system (Wade, 1999; Amirhor et al., 1997; Alix and Williams 2006). When compared to carbon filters, biofilters are still the less

expensive technology, with 40% lower operating costs (Alix and Williams 2006). A summary of available cost data obtained from reviewed literature is provided as **Table 5**.

Table 5 Literature data of capital and operation& maintenance costs in biofilters as a function of the air flow treated

Capital (construction) $/(m³/h)	Operation and maintenance $/(m³/h)	Flow rate treated (m³/h)	Reference
Design and build: 11.03	-	6800 - 27000	(Wade 1999)
Pre-fabricated units: 29.5 – 117.7 In-ground BF: 23.5 – 70.6 Walled BF: 29.4 – 88.3	-	-	(Gueissaz-Teufel 2004)
-	8800	4725	(Alix and Williams 2006)
Closed biofilter: 9.7	0.9 /year	6000	(Park, Nam et al. 2001)
5.7-11.1 12.7-13.1 23.0	1.2-2.7 /year 3-3.1 /year 4.7 /year	169900	(Adler 2001)
21.2	4.8 /year	8500	(Kraakman 2001)

6.0 Future Directions

A review of biofiltration as an odour abatement technique was conducted. Through this review, strengths, limitations and existing knowledge gaps were identified and assessed, with a summary provided as **Table 6**.

Table 6 Current status of biofiltration as an odour abatement technique

Strengths	Limitations	Knowledge Gaps
• High gas/liquid surface area • Relatively easy operation and start-up • Low operating costs • Reduction of low concentration VOCs and odorants • High potential removal efficiency for H_2S (>99%) and significant removal of reduce sulfur compounds • No secondary waste streams	• Difficult to control operating conditions • Large footprint required • Reduced performance when faced with fluctuating inlet gas component concentrations (particularly peak loads) • Clogging by particulate matter and air channelling • Medium deterioration • Reduced efficiency for high odour concentrations • Limited removal of chlorinated hydrocarbons • Potential for nutrient depletion • Experienced based design practices	• Biofilm thickness, growth and architecture • Biodegradation mechanisms / kinetics • Removal of VOCs found in sewers • Models and design practices • Full scale investigative work

While there have been significant advances in understanding biofilters design, operation and maintenance, significant knowledge gaps remain. Additional research is needed to develop an improved understanding of the microbiology and biomass distribution in biofilms, and little information exists about biofilm thickness, growth and architecture. Knowledge gaps also exist with regards to biodegradation mechanisms, in particular the adsorption processes in the packing material, and understanding specific contaminant the biodegradation pathways and reaction kinetics.

While several models have been developed in recent decades, the lack of reliable mass and heat transfer correlations and diffusion coefficients for biofilms hinder the development of models suitable for application in the design of full scale biofilters treating complex contaminant loadings. Much of the reviewed literature has focused on laboratory and pilot scale processes operating under idealised conditions. The needs exists for investigative work to consider conditions more representative of those encountered at full scale, in particular the treatment of complex multi-contaminant mixtures at trace concentrations.

Biofiltration success is closely linked with accurate characterization of waste gas streams, control of process conditions for optimization of biological activity, control of biomass overgrowth, and improvement of the physical-chemical properties of the filter media. A greater knowledge of these design and operation aspects, together with development of modelling and engineering features, will help to better understand the biofiltration process and to provide a basis for improved design, operation and maintenance.

SP3 will focus on addressing several of the knowledge gaps (**Table 6**) associated with the application of biofilters for odour abatement from sewer systems, in particular:

- identifying the key odour producing and treatment interfering compounds present in sewer networks to provide the industry with a more relevant list of compounds to use for design specifications and process performance evaluation;
- providing monitoring protocols to allow comprehensive abatement process performance assessment and support the diagnosis and solution of operational issues;
- evaluating the performance of full scale biofilters for the removal of odour and key odorants to provide the industry with data that can be used to identify the most appropriate odour abatement technology for a specific application; and
- using full scale odour abatement process performance data to identify the suitability of biofilters for abating specific odorants, supporting the industry in the selection and evaluation of odour abatement technologies.

7.0   References

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