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Kinetics of the removal of mono-chlorobenzene vapour from waste gases using a trickle bed air biofilter

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Abstract

The performance of a trickle bed air biofilter (TBAB) in the removal of mono-chlorobenzene (MCB) was evaluated in concentrations varying from 0.133 to 7.187 g m^{-3} and at empty bed residence time (EBRT) varying from 37.7 to 188.52 s. More than 90% removal efficiency in the trickle bed air biofilter was achieved for the inlet MCB concentration up to 1.069 g m^{-3} and EBRT less than 94.26 s. The trickle bed air biofilter was constructed with coal packing material, inoculated with a mixed consortium of activated sludge obtained from sewage treatment plant. The continuous performance of the removal of MCB in the trickle bed air biofilter was monitored for various gas concentrations, gas flow rates, and empty bed residence time. The experiment was conducted for a period of 75 days. The trickle bed air biofilter degrading MCB with an average elimination capacity of 80 g m^{-3} h^{-1} was obtained. The effect of starvation was also studied. After starvation period of 8 days, the degradation was low but recovered within a short period of time. Using macrokinetic determination method, the Michaelis–Menten kinetic constant K_m and maximum reaction rate, r_{max} evaluated as 0.121 g m^{-3} s^{-1} and 7.45 g m^{-3}, respectively.

Keywords: Mono-chlorobenzene; TBAB; EBRT; Biodegradation; Elimination capacity

1. Introduction

The Clean Air Act Amendments of 1990 list chlorobenzene as a hazardous air pollutant. Mono-chlorobenzene (MCB) is used in production of nitrochlorobenzene, adhesives, paints, paint removers, polishes, dyes, drugs, phenol, pesticides (like DDT), and aniline. Mono-chlorobenzene is a flammable liquid, its release to the ambient air may lead to an adverse environmental impact on air quality, thus endanger public health and welfare. There are number of removal technologies available to treat polluted stream. Among them, biological treatment method is an option, which uses the natural ability of microorganisms to remove the pollutants. It is cheaper, cost-effective and very efficient removal without any secondary air pollutants.

Among the biological waste gas treatment methods, biofiltration has attracted growing interest during the last few years. In Europe, biofiltration has been used successfully to control odours, and both organic and inorganic air pollutants that are toxic to humans, as well as volatile organic compounds from a variety of industrial sources. Biofiltration is a process for eliminating contaminants in air using microorganisms immobilized on a surface of solid support media. This technique has been applied successfully to control a number of air contaminants such as odors, volatile organic compounds (VOCs), and hazardous substances [1]. Biofilters, biotrickling filters and bioscrubbers may be generically referred to as organic perfusion columns. They consist of three phases in intimate contact: a solid organic phase, a liquid phase and a gas phase. All three may contain nutrients for degradation. The liquid phase (water or nutrient solution) and gas phase are passed through the solid organic medium, invoking the processes of adsorption and aerobic biological degradation of the nutrients in the liquid [2].

The process consists of a filter bed that usually comprised of natural organic media such as peat, compost, leaves, wood bark and/or soil. The bed moisture is kept at a constant level by humidification of influent air to maintain a biologically active

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Nomenclature

A_s	specific surface area packing material (m^{-1})
C_{gi}	inlet VOC concentration in the gas phase ($g\,m^{-3}$)
C_{go}	outlet VOC concentration in the gas phase ($g\,m^{-3}$)
C_g	MCB concentration in the gas phase ($g\,m^{-3}$)
d_p	diameter of the coal particle (m)
D_e	effective diffusion coefficient ($m^2\,s^{-1}$)
h	height of the packing material (m)
k_f	rate constant (s^{-1})
K_m	half saturation constant ($g\,m^{-3}$)
m	Henry's dimensionless coefficient
N_s	mass flux, biofilm ($g\,m^{-2}\,s^{-1}$)
Q	volumetric gas flow rate ($m^3\,h^{-1}$)
r	overall reaction rate ($g\,m^{-3}\,s^{-1}$)
r_{max}	maximum reaction rate ($g\,m^{-3}\,s^{-1}$)
t	time (s)
U_g	superficial velocity of the gas ($m\,s^{-1}$)
V	volume of the biotrickling filter (m^3)
x	distance in biofilm from the packing surface (m)
δ	biofilm thickness (m)
ϕ	Thiele modulus defined as $\phi = \delta\sqrt{(k_f/D_e)}$ dimensionless

layer surrounding the media, known as the "biofilm". VOC-containing air streams are transported to the air/biofilm interface, where they are absorbed into the biofilm and employed as carbon and/or energy sources by the microorganisms. Trickle bed air biofilter (TBAB) is a type of biofilter process that employs synthetic, inorganic media and receives liquid nutrients and buffers through a spray nozzle system on the top of the trickle bed air biofilter. Due to better control of pressure drop across the bed, pH, and nutrient feed, trickle bed air biofilters facilitate more consistent operation than do natural media biofilters. Furthermore, they do not suffer from the effects of aging as do natural media [3].

Chang and Lu [4] evaluated the performance of biotrickling filter treating isopropyl alcohol and acetone in a coal packed bed. More than 90% removal efficiencies were achieved with influent carbon loadings of isopropyl alcohol and acetone below 80 and 53 $g\,m^{-3}\,h^{-1}$, respectively. Mpanias and Baltzis [5] evaluated the performance of biotrickling filter treating monochlorobenzene in an Intalox saddle packed bed and 80–94.6% removal efficiency was achieved with an elimination capacity of 11–50 $g\,m^{-3}\,h^{-1}$. Peixoto and Mota [6] studied the performance of trickling filter treating toluene packed with raschig rings. They observed removal efficiency around 94% was achieved for inlet concentration less than 1.84 $mg\,L^{-1}$. Lu and Chang [7] studied the performance of biotrickling filter to treat trichloroethane in a coal packed bed. More than 95% removal efficiency was achieved. Lu et al. [8] studied the performance of biotrickling filter to treat acrylonitrile and styrene in a coal packed bed. More than 80% of removal efficiency was achieved using activated sludge as a microbial consortium.

Various authors studied the performance of trickle bed air biofilter and reported as an efficient process, and reliable process for the control of VOCs. The objective of the study was to investigate the performance of trickle bed air biofilter for varying influent MCB concentrations, gas flow rates and EBRTs and to find out the Michaelis–Menten kinetic constants for the biodegradation of MCB with mixed culture. Experimental results obtained herein provide useful information concerning the design criteria and operation of trickle bed air biofilter for the industrial operations.

2. Materials and methods

2.1. Organisms and culture medium

Activated sludge containing suspended solids (SS) of 3000 $mg\,L^{-1}$, and volatile suspended solids of 2100 $mg\,L^{-1}$ obtained from Khankal sewage treatment plant, Haridwar, India was used as the microbial seed in the trickle bed air biofilter. The suspended solids in activated sludge were allowed to settle for 5 h and the supernatant was discarded to obtain concentrated sludge. Then the sludge was acclimated to mono-chlorobenzene as the carbon source with a nutrient solution in a 250 mL flask. Flask was sealed with cotton stopper and stirrer at 200 rpm at room temperature. Enrichment cultures were developed by series of transfer at every 24 h interval for 7 days. The nutrient solution was continuously supplied to the microorganisms with a flow rate of 3–4 $mL\,min^{-1}$ and it was recycled. The nutrient feed contains inorganic salts, vital to the growth of attached microorganisms and $NaHCO_3$ as a pH buffer to maintain pH values of 7.0–7.7. The nutrient solution was changed twice a week. The composition of the nutrient solution is given in Table 1.

2.2. Experimental set-up

The experimental set-up of the trickle bed air biofilter used for MCB removal is shown in Fig. 1. It was made of Perspex tube and had a length of 100 cm and internal diameter of 5 cm. A 10 cm headspace was designed for the out let of treated gas and housing a nutrient spray nozzle, and a 10 cm bottom space was designed for the MCB waste gas inlet and leachate. There were two sampling ports P1 and P2 at 40 and 65 cm from the bottom respectively. The trickle bed air biofilter was filled with 1.57 L of

Table 1
Chemical composition of the nutrient solution

Chemical	Concentration
KH_2PO_4	1.19 $g\,L^{-1}$
$Na_2HPO_4{\cdot}12H_2O$	3.13 $g\,L^{-1}$
KNO_3	3.88 $g\,L^{-1}$
$(NH_4)_2SO_4$	2.58 $g\,L^{-1}$
$FeSO_4$	0.28 $g\,L^{-1}$
$MgSO_4{\cdot}7H_2O$	0.35 $g\,L^{-1}$
$NaHCO_3$	0.90 $g\,L^{-1}$
$MnSO_4$	1.52 $mg\,L^{-1}$
Na_2MoO_4	1.0 $mg\,L^{-1}$
$CaCl_2$	3.0 $mg\,L^{-1}$

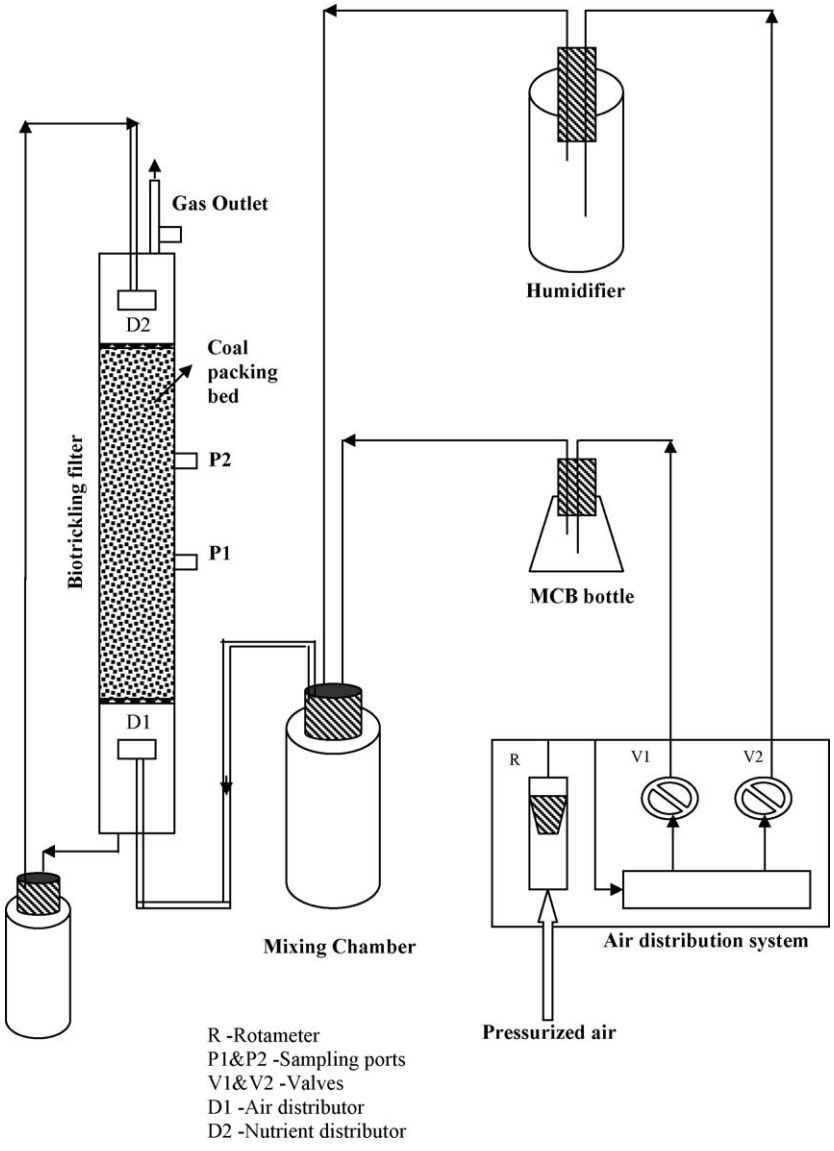

Fig. 1. Schematic diagram of the laboratory biotrickling filter system.

packing material consisting of coal particles with a specific gravity 2.514, equivalent-diameter of 1 cm and porosity of 66.1%. In order to support the filter bed and to ensure homogeneous radial distribution of input gas, an acrylic plastic mesh was installed at the bottom section. These supports were reinforced with other acrylic plate (fixed in pipe) in order to bear the weight of wet filter material. The packing material was inoculated with MCB acclimated mixed microorganisms. The trickle bed air biofilter assembly was placed within a temperature control box to control the temperature of MCB gas stream.

At first, compressed air was passed through a filter to remove particulate matter. After purification, the air stream was distributed to humidifier and mono-chlorobenzene bottle. Then, humidified air and MCB vapour were mixed in a mixing chamber and the mixture was passed through the packed bed. MCB concentration in the air stream was varied by adjusting the flow rates of the air stream passing through the water and the MCB solution. The trickle bed air biofilter was operated at various inlet MCB concentrations and gas flow rates. Samples were collected at regular interval from the inlet, outlet as well as from the various sampling ports using an airtight syringe and analyzed for residual mono-chlorobenzene.

2.3. Method of analysis

The MCB gas concentrations were analyzed by using a HP 5895A gas chromatograph equipped with a capillary column type HP 1, and a flame ionization detector (FID), connected to a computing integrator. The GC-FID was operated at an injection temperature of 190 °C, a detector temperature of 200 °C and an oven temperature of 190 °C. The fuel gas and carrier gas were hydrogen and nitrogen with a flow rate of 5 mL min^{-1}. The air samples were collected from the various sampling ports in a 5 mL airtight syringe. Air samples (50 μL) were then injected into the FID gas chromatograph for the analysis. Air samples with known mono-chlorobenzene were used for the calibra-

tion. The following parameters were determined according to Standard Method [9]: suspended solid, SS (2540-D), and VSS (2540-G). The pH values of nutrients solution and leachate were measured by a digital pH meter (NAINA NIG-333, India). Analyses of the coal, after 75 days was carried out by using a scanning electron microscope (Model LEO435VP, LEO Electron Microscopy Ltd., England).

3. Results and discussion

During the start-up, the trickle bed air biofilter was inoculated with the seed culture of mono-chlorobenzene acclimated inoculum. The experiment was conducted for various gas flow rates with EBRT of 188.52, 94.26, 125.68, 62.84, 47.13, and 37.7 s with varying inlet MCB concentrations. After seeding the trickle bed air biofilter, the inlet MCB gas concentration in the range of 0.133–1.67 g m^{-3} was maintained for EBRT of 188.52 s. The trickle bed air biofilter stabilized after a period of 16 days called as acclimation time required by the reactor from the start-up to attain maximum removal efficiency. The acclimation time is the start-up of trickle bed air biofilter period during which removal efficiencies steadily increase until they reach a sustained maximum value. This phenomenon occurs as microorganisms adapt enzymes and degradative pathways to metabolize the MCB [10]. Fig. 2 shows the variation of removal efficiency with MCB gas concentration and the empty bed residence time (EBRTs). It was observed that with the increase in the inlet MCB concentration there was a decrease in the removal efficiency. More than 90% of removal efficiency was achieved for the inlet MCB concentration less than 1.069 g m^{-3} and EBRT less than 94.26 s.

EBRT is a relative measure of gas residence time within the TBAB. Fig. 3 shows the removal efficiency as a function of EBRT. The removal efficiency increases with the increase in the residence time. The maximum removal efficiencies 94.35%, 92.64% and 95.20% were observed for the EBRTs 188.52, 125.68 and 94.26 s, respectively, when the inlet MCB concentration was maintained at around 1 g m^{-3}. The removal efficiency in

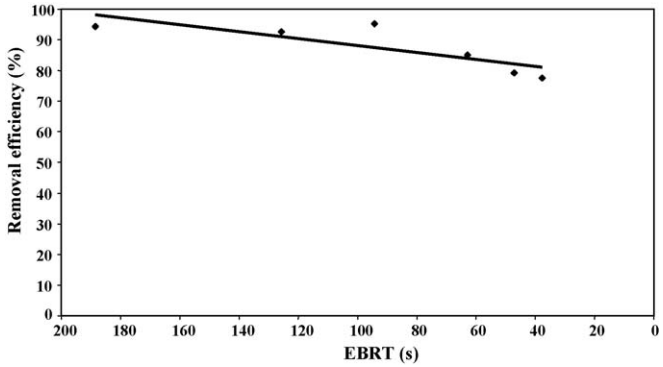

Fig. 3. Removal efficiency as a function of EBRT.

the trickle bed air biofilter was mainly controlled by the transfer rate of the MCB gas in the biofilm and in the gas phase boundary layer, which is in turn controlled by the residence time. At low gas flow rate, the residence times were adequate; therefore, the removal efficiency was more. At higher gas flow rates, the removal efficiency was low due to shorter residence time, the MCB not able to reach the interface between the gas and biofilm for the degradation. From the experimental results, it was observed that during the start-up period, the removal was mostly due to the adsorptive capacity of the packing material as the initial biomass concentration is low and biodegradation cannot account for the complete removal of mono-chlorobenzene. After biofilm established in the packing material, the removal mechanism shifts toward the biodegradation. Similar observations were reported in the literatures [11,12].

Fig. 4 shows the variation of elimination capacity with inlet MCB gas concentration. Elimination capacity (EC) is defined as the amount of MCB removal in the trickle bed air biofilter per unit volume of the trickle bed air biofilter per unit time (g m^{-3} h^{-1}). It was found that average elimination capacity of 80 g m^{-3} h^{-1} was obtained. The maximum elimination capacity of 108.07 g m^{-3} h^{-1} was observed for a removal efficiency of 86.54%. The maximum elimination capacity of the biofilter is

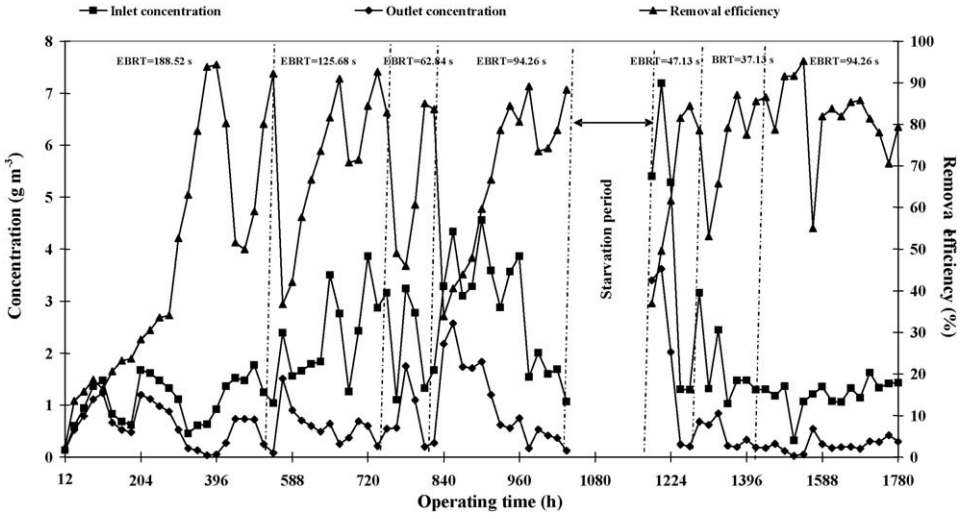

Fig. 2. Time-dependent MCB concentration and removal efficiency with operating time.

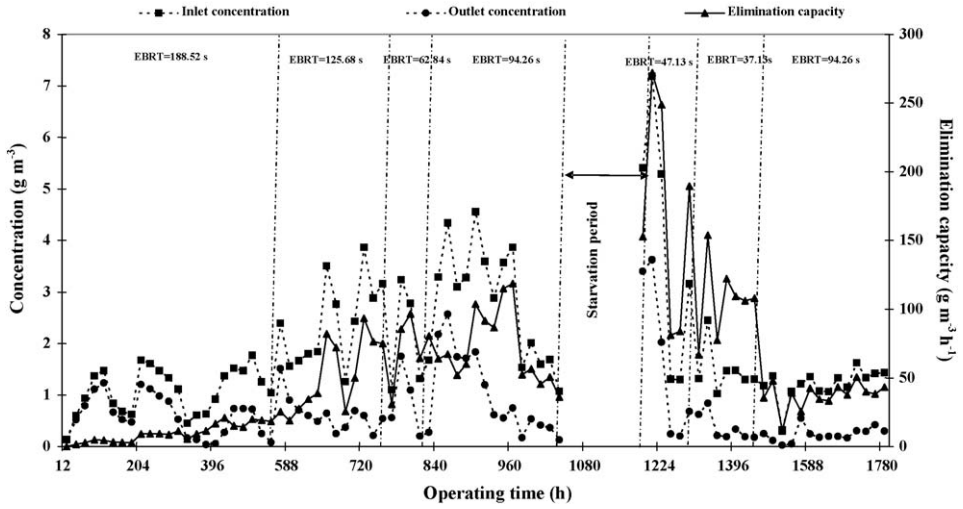

Fig. 4. Time-dependent of elimination capacity and MCB concentration with operating time.

the maximum mass-loading rate that the trickle bed air biofilter can tolerate without inhibiting its microbial population.

The effect of pollutant starvation in the trickle bed air biofilter was studied. The starvation period of 8 days was maintained without MCB, i.e. without carbon source for the microorganisms. During the starvation, the reactor lost its ability to degrade MCB, regardless of the mode of starvation. The microorganisms significantly decreased during starvation. Decrease of microorganisms may be due to biomass death and lysis, endogenous respiration of the process culture and shear by the liquid recycling. Fig. 5 shows the re-acclimation of the trickle bed air biofilter after starvation without MCB. It was observed that after starvation there was a sudden fall in the removal efficiency without air, which could be due to microbe's death and lysis. But it recovered quickly. It reached maximum removal efficiency of 84.46% for the concentration of 1.304 g m^{-3} and EBRT of 47.133 s. From the experiment it was found that the re-acclimation period was lesser than initial acclimation period. The re-acclimation period was 3 days.

Fig. 6 gives a clear representation of the elimination capacity and removal efficiency as a function of MCB loading rate. With the increase in the mass-loading rate, the elimination capacity also increases but the trend was opposite in the case of removal efficiency, it was decreased. From the observations, the elimination capacity varies with the gas flow rate and inlet concentration throughout the experiment. Thus, the elimination capacity and removal efficiency are important design parameters for trickle bed air biofilter operation. It was observed that for the maximum removal efficiency of 95.201% and for the inlet MCB concentration of 1.069 g m^{-3} with an EBRT of 94.26 s, the elimination capacity was 38.87 g m^{-3} h^{-1}.

The concentration profiles along the bed height were studied for the various gas flow rates in trickle bed air biofilter height. Fig. 7 shows the concentration profile of MCB gas stream along the bed. The MCB gas concentrations from 1.038 to 1.6 g m^{-3} were maintained for various EBRTs. The MCB concentration was decreased along the bed from the inlet to outlet (i.e. from bottom to top) due to biodegradation. The concentration profiles along the bed showed that for high EBRT the degradation was more in the trickle bed air biofilter, but it was reverse for the case of low EBRT. It was observed that the removal was more in the bottom of the filter. The MCB removal also depends on the oxygen concentration available to the microorganisms. The trend in the concentration profile of MCB changes with the changes in the concentration of oxygen in the bed [13].

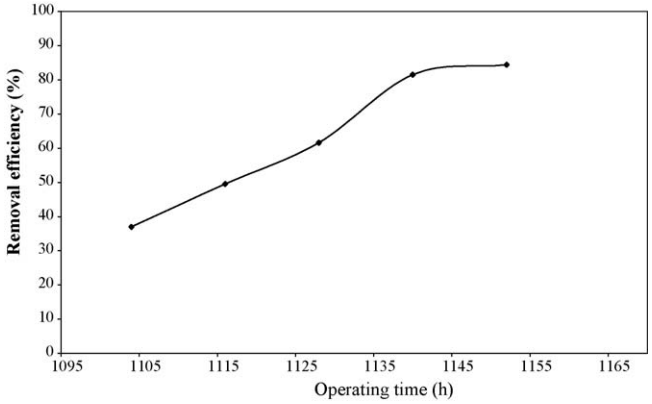

Fig. 5. Re-acclimation period after starvation.

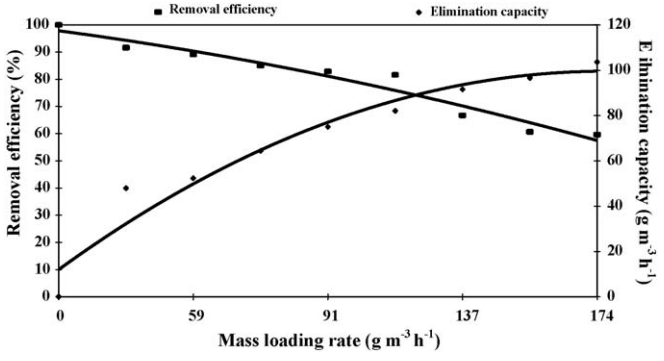

Fig. 6. Dependency of removal efficiency and elimination capacity on mass-loading rate.

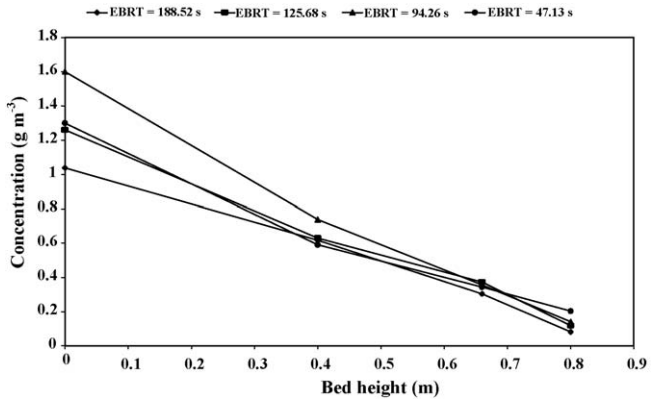

Fig. 7. Concentration profile of MCB concentration along the bed height.

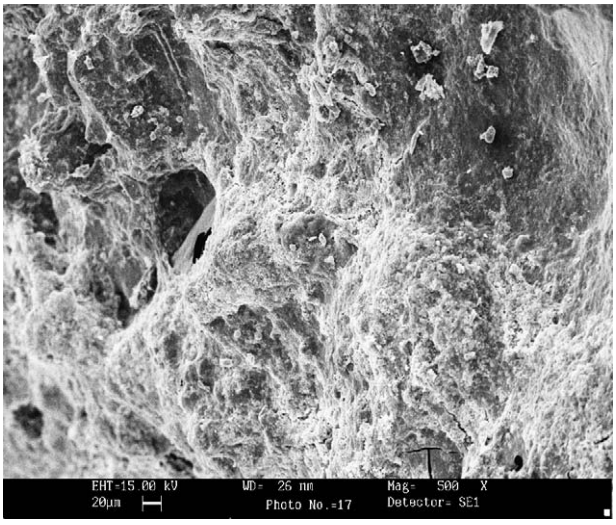

Fig. 8. Structure of biofilm on the coal packed material.

A scanning electron micrograph (SEM) of microbial growth on peat was taken. The SEM shows the structure of biofilm formation on the surface of the coal. Fig. 8 shows the structure of the biofilm with high magnification. The formation of the biofilm on the surface of packing media not only depends on the packing media; it depends mostly on the nutrient provided. Packing materials with a good adsorbing capacity were shown to enhance biotrickling filter performance when biodegradable compounds exhibit low solubility in water [14]. From the SEM study, it was observed that coal was a good packing material and can be used as a good biological attachment medium for the biofiltration.

4. Determination of Michaelis–Menten kinetic constants

The kinetic constants of the biodegradation can be determined either microkinetically or macrokinetically. For microkinetic determination, the microorganisms were isolated from the trickling filter media and inoculated into shake flasks containing a nutrient solution, with the contaminant solution as the sole carbon source. The concentration of contaminant was continuously measured as a function of time to obtain the kinetic constants by plotting the relation of biodegradation rate and the contaminant concentration [15]. However, the microkinetic method of determination in biofiltration systems was still controversial because the kinetic behavior of biofiltration systems (a gas phase systems), and suspended cell systems (a liquid phase systems), may be not similar due to the different phases of biodegradation. In this study, macrokinetic determination was used following the suggestion of [16]. The same method of macrokinetic determination was used by the Krailas et al. [1] to determine the Michaelis–Menten constants. The kinetics of the system can be expressed by a Michaelis–Menten type relationship by assuming that oxygen limitation was not present in the system and the conversion was in the reaction-controlled regime (i.e. the biofilm was fully active). At steady state, the growth rate of microorganisms was balanced by its own decay rate, resulting in the biological equilibrium of the system. Hence, kinetic constants remained constant over the period of time considered.

The kinetic constants were determined using the plug flow model without dispersion at steady state Eq. (1):

$$\frac{\partial C_g}{\partial t} = -U_g \frac{\partial C_g}{\partial h} + r \tag{1}$$

where C_g is MCB concentration ($g\,m^{-3}$), U_g the superficial velocity ($m\,s^{-1}$), t the time interval (s), h the distance from the bed (m), and r is the overall reaction rate and it is defined by Eq. (2):

$$r = \frac{r_{max} C_g}{K_m + C_g} \tag{2}$$

where r_{max} is the maximum biodegradation rate per unit biofilter volume ($g\,m^{-3}\,s^{-1}$) and K_m is the saturation (Michaelis–Menten) constant ($g\,m^{-3}$) in the gas phase. At steady state, the accumulation term $\partial C_g/\partial t$ equals to zero. Eq. (1) was integrated under the following conditions:

$$C_g = C_{gi} \quad \text{at } h = 0 \tag{3}$$

and

$$C_g = C_{go} \quad \text{at } h = L \tag{4}$$

where C_{gi} and C_{go} are corresponding inlet and outlet MCB concentration ($g\,m^{-3}$) and L is the biofilter length (m). Solving Eqs. (1) and (2), Eq. (5) was obtained:

$$\frac{V/Q}{C_{gi} - C_{go}} = \frac{K_m}{r_{max}} \frac{1}{C_{ln}} + \frac{1}{r_m} \tag{5}$$

where C_{ln} is the log mean concentration $[(C_{gi} - C_{go})/\ln(C_{gi}/C_{go})]$, V the biofilter volume (m^3), and Q is the volumetric flow rate ($m^3\,s^{-1}$), r_{max} and K_m for the gas phase can be obtained by plotting $[(V/Q)/(C_{gi} - C_{go})]$ against $(1/C_{ln})$ with correlation coefficient of 0.9969. From Fig. 9, the r_{max} and K_m were calculated as 0.121 $g\,m^{-3}\,s^{-1}$ and 7.45 $g\,m^{-3}$, respectively.

5. Modeling of biotrickling filter

The theoretical description of the biofiltration systems has gained the interest of many researchers with the aim of better

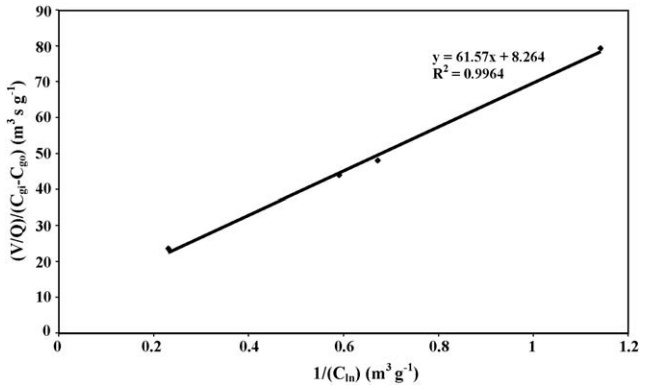

Fig. 9. Macrokinetic determination of Michaelis–Menten kinetic constants.

understanding of the process, optimizing the process and optimizing the design and operation of biotrickling filters. In this experiment, VOC-containing air stream is passed through a column filled with coal particles that are covered with a steady state biofilm. VOCs are transported to the air/biofilm interface, where they are adsorbed into the biofilm and employed as carbon and/or energy sources for the microorganisms. A mathematical model describing the transport and biological phenomena involved in the process of contaminant removal in biotrickling filters has been developed. The model predicts the concentration profile of the VOC in the gas phase, the biofilm and in the sorption liquid retained in the solid particles composing the filter bed at steady state. Fig. 10 depicts the concentration profile followed by the VOC. The Michaelis–Menten constant obtained using the macrokinetic determination was used in this model (Table 2).

5.1. Assumptions

Some assumptions are made to derive the governing model equations. These are discussed here below.

1. The external surface of the solid particle is completely covered with the liquid biofilm. The biolayer thickness is assumed to be uniform throughout the height of the filter bed.
2. Biodegradation of VOCs occurs only aerobically. The only substances affecting the rate of biodegradation are the VOCs and oxygen. The VOCs are assumed to contact the organisms comprising the biofilm by diffusion as characterized by Fick's law.
3. At steady state, the microorganisms are considered to be uniformly distributed throughout the biofilm and the bed as whole. The kinetics of the substrate reactions in the biolayer surrounding the coal particle follows the Monod type model [17].
4. The flow of the air stream through the filter bed is of a plug flow type and gas flow rates are sufficiently high to render axial dispersion negligible.
5. The coal particles are assumed to be spherical.
6. The pollutant concentration is assumed to decrease from the particle surface, where the concentration is in equilibrium with the bulk gas phase to a value of zero at some radial distance from the center of the particle. It is illustrated in Fig. 10.
7. The biofilter is isothermic. Ambient temperature prevails uniformly in the trickling filter [18].

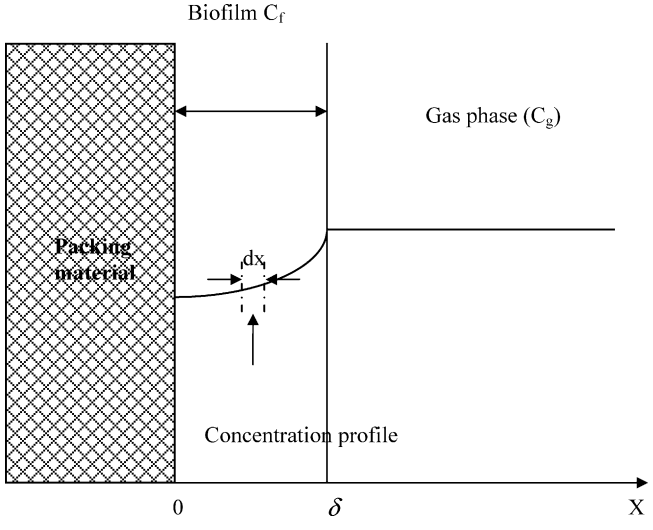

Fig. 10. Concentration profile of VOCs along the biofilm.

Table 2
Model parameters used in the model

Parameter	Units	Values	Reference
d_p	m	0.01	Present work
r_m	$g\,m^{-3}\,s^{-1}$	0.121	Present work
K_m	$g\,m^{-3}$	7.45	Present work
k_f	s^{-1}	0.0162	Present work
m	–	0.167	[13]
D_e	$m^2\,s^{-1}$	0.81×10^{-9}	[19]
δ	m	0.0001	[20]

5.2. Mass balance in the biofilm

Biodegradation rate is commonly described by the Monod model. The model is based on the two limiting cases, zero-order kinetics — for sufficiently high VOC concentration, and first-order kinetics — for low VOC concentration.

Monod kinetics:

$$-r_b = \frac{r_{max} C_f}{K_m + C_f} \tag{6}$$

Case 1: Zero-order kinetics:

$$-r_b = k_1, \quad \text{where } (C_f \gg K_m) \tag{7}$$

Case 2: First-order kinetics:

$$-r_b = k_1 C_f, \quad \text{where } (C_f \ll K_m) \tag{8}$$

where C_f is VOC concentration in the bulk fluid ($g\,m^{-3}$), r_{max} maximum specific reaction rate ($g\,m^{-3}\,s^{-1}$), K_m half saturation constant ($g\,m^{-3}$), and k_f is the biodegradation constant ($k_f = r_{max}/K_m$ for first-order, and $k_f = r_{max}$, for zero-order). In this work, the first-order kinetics is only considered.

When the substrate concentration is very low, i.e. for $C_f \ll K_m$, Eq. (6) reduces to a first-order rate. At steady state, the biofilm is assumed to attain a constant thickness, δ, and there is no accumulation of substrate within the biofilm. Based on the above assumptions, a mass balance in the biofilm of a coal particle (Fig. 10) is

$$-D_e A_s \left.\frac{dC_f}{dx}\right|_x + D_e A_s \left.\frac{dC_f}{dx}\right|_{x+\Delta x} - k_1 C_f A_s \Delta x = 0 \quad (9)$$

where D_e is diffusion coeffiecient in $m^2\,s^{-1}$, x axial distance in m, and A_s is biolayer surface area per volume of packing in m^{-1}. Dividing both sides by A_s and Δx, and taking the limit as approaches to zero, yields the following equation:

$$D_e \left.\frac{d^2C_f}{dx^2}\right|_x - k_f C_f = 0 \quad (10)$$

The boundary conditions, assuming reaction-limited kinetics are

$$\frac{dC_f}{dx} = 0 \quad \text{at } x = 0 \quad (11)$$

$$C_f = \frac{C_g}{m} \quad \text{at } x = \delta \quad (12)$$

where m is the distribution coefficient which is the ratio of (C_g/C_f) at equilibrium condition. The solution of Eqs. (9) to (12) are

$$\frac{C_f}{C_g/m} = \frac{\cosh(\phi(x/\delta))}{\cosh\phi} \quad (13)$$

where ϕ is 'Thiele modulus' and is defined as $\phi = \delta\sqrt{k_f/D_e}$. The steady state material balance for gas phase pollutant concentration (C_g) in a coal packed biotrickling filter having h meter depth, assuming negligible axial dispersion (Ottengraf et al., [15]) is

$$-U_g \frac{dC_g}{dh} = N_s A_s \quad (14)$$

where U_g is superficial gas velocity, $m\,s^{-1}$, dh is differential length in axial direction, and N_s is mass transport to biofilm given by Eq. (15):

$$N_s = -D_e \frac{dC_g}{dh} \quad (15)$$

N_s can be calculated by evaluating (dC_f/dx) at $x = \delta$ from Eq. (13) and then substituting in Eq. (15):

$$N_s = -D_e \frac{C_g}{\delta m}(\phi \tanh\phi) \quad (16)$$

The model Eq. (14) is solved numerically using MATLAB ODE suit. The equation is solved numerically by fourth-order Runge–Kutta algorithm. The theoretical results and experimental results were compared. From the theoretical results, it was observed that the first-order model prediction of MCB concentration profiles in the biotrickling filter was in very good agreement with the experimental results. Figs. 11 and 12 represents the concentration profile followed by the model and experimental results for the MCB concentrations of 1.4181 and 1.84 g m^{-3} at various sampling ports in the trickling filter. The predictive

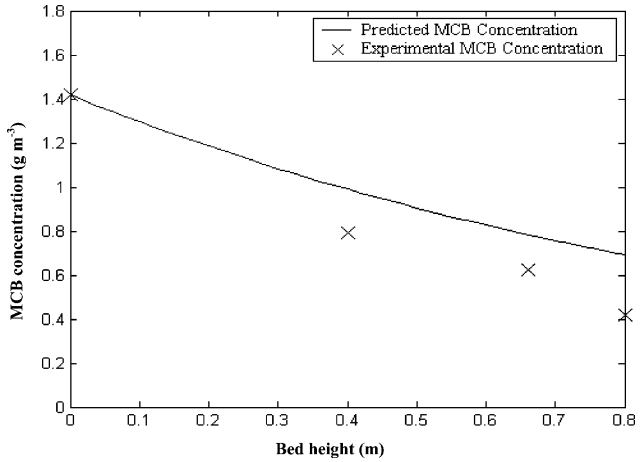

Fig. 11. Concentration profile along the bed height for the Inlet MCB concentration (1.4181 g m^{-3}).

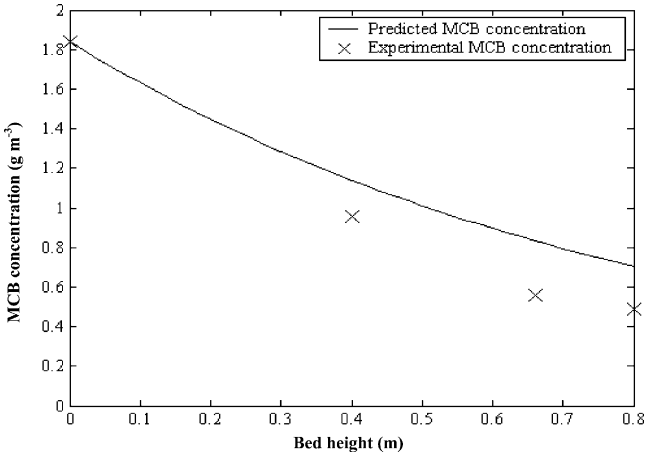

Fig. 12. Concentration profile along the bed height for the Inlet MCB concentration (1.84 g m^{-3}).

first-order concentration profiles of the developed model were the good predictors of the exponential character of experimental profiles. The deviations of experiment and predicted values were due to: (a) ignoring the some biochemical aspects such as self inhibition or cross inhibition at higher concentration, (b) the model predictions made does not include the adsorption of VOCs in the packing media, and (c) the microorganism would be more near the nutrient supply and near to the outlet, was not considered.

6. Conclusions

A laboratory trickle bed air biofilter with coal packing material was studied for the biofiltration of MCB. From the study, it is observed that coal is a good packing material and can be used as a biological attachment medium for the biofiltration. The performance of the trickle bed air biofilter for various gas flow rate, and inlet MCB concentration were studied. More than 90% of removal efficiency was achieved for the inlet MCB concentration less than 1.069 g m^{-3} and EBRT less than 94.26 s. Trickle bed air

biofilter overcame the starvation within 3 days of re-acclimation time. The Michaelis–Menten kinetic constants were determined by macrokinetic determination method. The r_{max} and K_m were calculated as $0.121\,g\,m^{-3}\,s^{-1}$ and $7.45\,g\,m^{-3}$, respectively. A mathematical model has been developed for the biotrickling filter. The theoretical and experimental results were compared. The model follows the first-order kinetics and we believe that it is a well representation of the biodegradation of MCB in the biotrickling filter.

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