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Feasibility Study of Biodegradation of Gaseous Pollutants with Respect to Industrial Processes

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ABSTRACT

The present study examines the feasibility of biodegradation of gaseous pollutants with respect to industrial processes. There are several methods such as physical methods, chemical methods and biological methods for removing gaseous pollutants. Combustion of fossil fuels leads to the emission of various pollutants such as SO_2, CO_2, NOx, CO, VOC, etc. These pollutants are harmful to the environment and human health. NOx and SO_2 lead to acid rain and VOC compounds cause photochemical smog and ozone layer depletion and some VOCs are carcinogenic and cause gene mutations. Common methods for reducing gaseous pollutants are wet scrubbing using lime to remove SO_2, catalytic reduction for NOx and surface adsorption to remove VOCs. All these methods are complex chemical processes and produce wastes such as wastewater, catalysts and used surface adsorbents. The results of the present study showed that using microorganisms adapted to pollutant gas, these compounds are reduced or purified in the exhaust air from the chimneys of polluting industries. The pollutant is absorbed into the liquid phase, which contains active microorganisms. This group of adapted and specialized microorganisms decompose the pollutant and use the energy from the breakdown of these compounds for their own reproduction and metabolic interactions. The product of the microbial reaction is mainly CO_2, water and biomass, in this method the pollutant can be organic or inorganic.

Introduction

The emission of odorous substances is a common environmental problem and can cause serious problems in the vicinity of the emission point. Among these odorous substances, volatile organic compounds containing sulfur (VOSCs) produced by various industries occupy a large volume and can cause serious environmental problems that threaten public health. These compounds mainly include dimethyl sulfide, dimethyl disulfide, methane thiol, carbon disulfide, carbonyl sulfide and ethane thiol. The nature and concentration of molecules detected by olfactory cells vary from person to person and also under different environmental conditions such as temperature, pressure and humidity. Humans are able to detect very small amounts of odorous substances. It is estimated that only 10^8 or 10^9 molecules of odorous gases are sufficient in the nose to detect an odor [1].

This is while 1 microgram of ethyl mercaptan in air contains approximately 10^{16} molecules, i.e. 10^7 or 10^8 times the amount required for detection. Human sensitivity to odors makes it necessary to carefully control the amount of odorous waste if it is produced near human habitation [2].

Hydrogen sulfide (H_2S) is primarily produced in the atmosphere from the reduction of sulfate, which probably occurs in anaerobic organic environments. Other sources of H_2S production include the catabolism of heterotrophic organic sulfur compounds (proteins, glutathione), sulfur metabolism, and the chemical reduction of sulfate in seawater. Hydrogen sulfide has long been recognized as the dominant atmospheric component in the sulfur biogeochemical cycle.

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However, the identification of low but significant concentrations of dimethyl sulfide (Me_2S) in ocean water has strengthened the belief that the main sulfur gases in the sulfur biogeochemical cycle are organic compounds such as Me_2S, methane thiol (MeSH), dimethyl disulfide (Me_2S_2), carbon disulfide (CS_2), and carbonyl sulfide (COS). The main source of Me_2S is the marine environment, where Me_2S is formed by the enzymatic breakdown of dimethyl-beta-propiotin, which is produced by many species of algae. In terrestrial environments, Me_2S and MeSH are mainly produced during the degradation of sulfur compounds of biological origin, such as the amino acids methionine and cysteine, and their derivatives, S-methyl methionine and S-methyl cysteine [3].

The contribution of methylation reactions, reduction of dimethyl sulfoxide (DMSO) and lignin degradation to the production of Me_2S is still unknown. Kelly and Smith (1990) proposed that Me_2S formation is a result of the oxidation and dimerization of MeSH. About 30% of CO_2 in the atmosphere originates from the oxidation of CS_2, while the contributions from oceans and natural terrestrial sources are 28% and 24%, respectively. Human activities increase the natural levels of CS_2: The contribution from chemical industry waste is 58%. On the other hand, the contribution of oceans and natural land sources to the total CS_2 in the atmosphere is 34% and 4%.

Academic research in Iran is limited to the removal of hydrogen sulfide using bio filters. Shoja El-Sadati and Eliasi in 1999 investigated the removal of hydrogen sulfide using a bio filter consisting of mushroom and oyster shell waste inoculated with activated sludge. The bio filter was 14 cm in diameter and consisted of four 30 cm sections with a total volume of 18.46 liters. Initially, during the adaptation phase, this bio filter achieved a removal efficiency of 99% in less than three days with an inlet concentration of 110 ppm [4].

The group achieved a removal capacity of 1.22 $g/m^3/h$. The gas retention time at this removal capacity could not be determined from the data in the literature. If the hydrogen sulfide concentration is 135 ppm (the highest concentration used in this study) and the removal efficiency is 99%, the retention time should be 31 seconds, which, considering the subsequent work of this group, is unlikely to have achieved this low retention time in these experiments and not even a one-minute retention time in their subsequent experiments. Perhaps they achieved this removal capacity at higher concentrations and retention times.

In 2000, Shoja El-Sadati and Seifi, using a device similar to the previous one, obtained the bio filter efficiency at concentrations (20 ppm to 200 ppm) and different retention times (43,57,92 and 184 seconds). At a constant concentration and retention times of 92 and 184 seconds, they were able to find a removal efficiency of more than 95%.

This value is equivalent to a removal capacity of 7.78 $g/m^3/h$ in the best case. When they reduced the retention time to 57 and 43 seconds, they reported a removal efficiency of more than 85%. In this case, the best removal capacity is also 1.15 $g/m^3/h$. Bankdarpour, Heydarian and Taghinejad in 2008 used a trickling bed bio filter with an igneous rock bed to remove hydrogen sulfide from the gas phase [5].

This trickling bed bio filter was stabilized with Thiobacillus thioprosus bacteria and all experiments were performed with a counter-current flow arrangement. Experiments were performed over a range of concentrations (10 ppm to 90 ppm), residence times (9 seconds to 60 seconds) and different circulating flow rates (0.98 m/h and 1.95 m/h) and the removal efficiency and capacity were obtained.

The trickling bed bio filter was able to completely remove the pollutant at a concentration of 45 ppm and a residence time of 20 seconds, and even when the inlet concentration was increased to 75 ppm, the removal efficiency was higher than 96%. The effect of circulating flow on the efficiency of the bio filter was obtained at a constant concentration of 5 ppm and by varying the residence time. Increasing the circulating flow reduced the removal efficiency of hydrogen sulfide. The effect of varying the pollutant concentration was obtained at different column heights.

As the residence time decreased, the removal of hydrogen sulfide in the trickling bed bio filter became somewhat more uniform, but the effect of varying the inlet hydrogen sulfide concentration on its removal along the column was negligible. Almost all the removal was done in the first 85% of the useful height of the bio filter. This group investigated the change in the efficiency of the trickling bed bio filter after disturbances in the system. Although the efficiency of the system improved somewhat when it returned to normal, even after re-inoculation, the bio filter efficiency did not fully recover [6].

Bankdarpour and Abdehaq (2009) investigated the removal of hydrogen sulfide by a trickling bed bio filter with a polyurethane substrate and the immobilization of Thiobacillus thioprus and Thiobacillus thioprus. First, the possibility of immobilization of Thiobacillus thioprus on the polyurethane substrate was investigated and it was concluded that this bacterium does not have the ability to immobilize on this substrate. In the next step, the immobilization of Thiobacillus thioprus on the polyurethane substrate and the performance of the system under steady-state conditions were investigated, within a range of hydrogen sulfide inlet concentrations (25 ppm to 85 ppm) and gas retention times (9 to 45 seconds) [7].

The trickling bed bio filter completely removed a concentration of 25 ppm of the pollutant in a retention time of 45 seconds and had a removal efficiency of 94% by reducing the retention time to 27 seconds. The effect of liquid circulation on the performance of the bio filter was also investigated. The results of the experiments confirmed the effect of the contaminant inlet concentration and gas retention time on the effect of the liquid circulation flow intensity [8].

The results showed that at constant contaminant inlet concentrations and at different retention times, increasing the liquid circulation flow intensity will have a slight positive effect on the removal capacity and pollutant removal efficiency. However, this positive effect is noticeable at constant retention time and high contaminant inlet concentrations. In the last stage, the performance of the bio filter in unstable conditions was investigated by applying various periods of no-feed (removal of the pollutant from the gas flow, removal of the air flow and system shutdown). The results showed that the trickling bed bio filter has the ability to return to its initial state after 4, 8, 12, 16 and 24-hour no-feed periods and also after 1 and 2 days of complete shutdown, but after 5 days of complete shutdown it will no longer be able to remove pollutants [9].

Innovation

Extensive research is underway on the biological oxidation of alkyl sulfides. The innovations in this proposed research include the following:

- ✓ Biological removal of sulfides using a microbial system stabilized on a natural igneous rock retainer in a trickling bed bio filter with defined dimensions.
- ✓ Laboratory scale trickling bed bio filter deployment.
- ✓ Use of the Long-Holdin relationship to study growth kinetics.

Results

Three methods are considered in this section: Bio scrubbing, trickling bed bio filtration, and bio filtration. These methods were first developed in Europe and are now widely accepted and used in other industrialized countries such as the United States and Japan. These methods are briefly mentioned below.

When the velocity of the purified gas exiting the gas scrubber is high. Fine liquid droplets may be discharged as mist and vapor along with the gas from the top of the scrubber tower. In order to prevent this process, which reduces the efficiency of the system, interwoven mesh layers called demister pads or mesh pads are used. When installed at the top of the gas scrubber tower, they trap fine liquid droplets. Liquid distributors are used to continuously spray scrub liquid to wet the packing bed and contact the liquid with the gas flow, the number of which varies based on the height of the tower and the height of the packing bed. Packing media increase the mass transfer surface between the liquid and the gas by providing a wet bed for the gas flow. Packing media come in various forms. Each has its own characteristics such as contact surface, pressure drop, weight, corrosion resistance and price. The life of the packing's varies based on the possibility of corrosion, clogging and breakage. However, under ideal conditions, the life of the packing's is as long as the scrubber tower itself. Packing media is classified into two categories: Random and fixed. Random packing's are usually used in absorption towers. This type of packing was originally made of ceramic. However, due to their brittleness, their use decreased over time, and today, metal or plastic (polypropylene) packing's are mainly used. Metal packing's are not used for use in corrosive environments such as acidic gases. And plastic packing's are not suitable for use in towers with high gas temperatures [10].

Bio scrubbing

In this method, the pollutant is adsorbed in an aqueous phase and then subjected to biological treatment in a second stage in a bioreactor. The effluent from the reactor is returned to the absorption column. This method is suitable when the gaseous pollutants have high solubility in water. The main advantages of this method are as follows:

- ✓ Washing the reaction products out prevents their possible inhibitory effects.
- ✓ The biological process is easy to control by controlling the composition of the culture medium.
- ✓ The bio adaptation capacity of the microbial biomass is desirable with respect to the composition of the gas to be treated.

The main problem of this method is the need to dissolve the gaseous pollutants in the aqueous phase, which may cause gas transfer problems due to the short residence time of the gas phase in the absorption column. Therefore, this method is considered for gaseous pollutants with a Henry constant of less than 0.01 [29]. This is important considering that most of the pollutants of interest are volatile and have low solubility in water. This problem is probably one of the reasons why this method is less popular than bio filtration, although several successful examples of this method have been reported.

On the other hand, recent developments indicate that this method is of interest, as it seems that very high rates of biological desulfurization of gas flows (up to 2×10^6 m^3/h) are possible in bio scrubbers, and this method is one of the few examples of anaerobic treatment of gaseous wastes [10].

Trickling bed bio filtration

Waste gas treatment in trickling bed bio filters consists of a biological filter continuously fed with a liquid culture medium and a (usually synthetic) support on which the biofilm grows. The contaminated gas passes through the support either co- or anti-co-currently with the liquid mobile phase that provides nutrients for the microorganisms. The fresh culture medium entering the reactor may be mixed with water and returned to the system. The supports that are commonly used and reported in the literature include plastic or ceramic supports, celite, activated carbon, or a mixture of different materials. Trickling bed bio filters have similar advantages to bio scrubbers:

- ✓ Easy removal of reaction products by flushing out;
- ✓ Easy control of the biological process;
- ✓ Good capacity for accommodating the activated biomass [11].

The main problem with bio scrubbers is gas transport, which arises from the need to dissolve gaseous pollutants in the aqueous phase. This problem is less important here because trickling bed bio filters can be used effectively to treat compounds with a water/air partition coefficient of less than 0.1. Reducing the water flow rate (culture medium) can reduce the wetted area of the filter base, which is an estimate of the active area.

As a general rule, in a packed bed reactor, the wetted area is less than half of the total available specific surface area. Therefore, it is expected that the removal efficiency will increase with increasing the liquid flow rate, although this will also increase operating costs. On the other hand, recent studies have shown that better efficiency is achieved by reducing the liquid flow rate to the minimum microbial requirements. The liquid flow rate that allows for maximum removal capacity should be evaluated experimentally [12].

Another problem that may arise in trickling bed bio filters is excessive growth of biofilm at the base, which in turn reduces the free volume of the base and may cause an undesired increase in pressure drop. Biofilm growth can cause complete clogging of the filter bed. This highlights the importance of careful foundation design. Little information is available on the factors influencing clogging or effective ways to prevent it.

There are methods to limit fouling, including limiting biomass growth, frequent backwashing of the filter bed, or limiting the liquid flow rate. No increase in pressure drop or fouling was observed with liquid flow restriction over a 74-day operation period. However, reducing the liquid flow rate ultimately has a negative effect on microbial activity and reduces removal efficiency. Backwashing the bio filter bed with culture medium twice a week for one hour each time is an effective way to prevent excess biomass build-up. Biomass growth is limited by reducing the nutrient supply, although this may gradually reduce reactor efficiency. Carbon, hydrogen, or oxygen are not usually limiting unless the pollutant load is highly variable. Therefore, some amount of nitrogen, phosphorus, neutral salts, or metals is required to control the biomass. Biomass yield is a function of the nature of the nutrients available; for example, nitrate as a nitrogen source produces less biomass than ammonium [13].

A good biofilm structure is achieved by specifying the ionic strength. These observations indicate that it is important to find the optimal balance between biomass growth limitation or fouling and removal efficiency. At present, no general rule has been given for this purpose and it is necessary to determine the optimal operating conditions for each specific case experimentally.

Tripping bed bio filters are less commonly used than conventional bio filters. Bio filtration is cheaper and simpler than other biological methods. Tripping bed bio filtration and chemical scrubbing have similar application areas, and scrubbing is less expensive. Tripping bed bio filters facilitate continuous operation and provide more stable conditions due to the appropriate control of pressure drop, PH and nutrient content. In addition, these types of bio filters are preferred over other biological methods due to their molecular comminution capability, especially for high concentrations of acidic pollutants, including gas streams containing sulfur, chlorine or nitrogen compounds. Single volatile organic sulfur compounds such as methane thiol or dimethyl disulfide can be effectively degraded in trickling bed bio filters [14].

Bio filtration

The bio filter consists of a substrate, usually made of organic materials (peat, compost, sawdust) and used both as a base for the activated biomass and as a source of nutrients. Pollutants present in the polluted air are destroyed by the activated biomass as they pass through the filter bed. An important feature of the process is the absence of a mobile phase, as a result of which bio filters are suitable for the treatment of pollutants with low water solubility. Bio filtration is suitable for the treatment of pollutants with a water/air partition coefficient of less than 1 [15].

Numerous examples of successful industrial application of this method can be found in the literature, and today some industrial plants treat gas streams of up to 200,000 m^3/h with this method. Bio filters, trickling-bed bio filters and bio scrubbers consist of three phases in close contact: Solid phase, liquid phase and gas phase. All three phases may contain materials to be destroyed. In the removal of odorous compounds, the substances are transferred from the gas phase. The rate of adsorption or biological degradation of the substances may vary depending on the type of substrate or operating

conditions such as temperature and PH. Some of the biological transformations that are important in the bio filtration of odorous compounds, along with the common bacteria responsible for carrying out these transformations, are shown in table 1.

Table 1. Differentiating characteristics of the three main methods of biological treatment of gaseous pollutants

Active biomass	Carrier	Mobile phases	Reactor Design
Broadcast	-	Liquid and gas	Bio scrubber
Fixed	Synthetic	Liquid and gas	Tripping Bed Bio filter
Fixed	Organic/Synthetic	Gas	Bio filter

Kowal and colleagues showed that hydrogen sulfide removal in a bio filter occurs in two stages through three distinct phases:

- ✓ Adsorption in the liquid phase present in the bed.
- ✓ Adsorption on the solid surface and biodegradation.

Stage 1 is well interpreted by thermodynamic equilibrium and is very fast compared to stage 2, especially when the gas phase residence time in the bio filter is about minutes. Therefore, in a bio filter, the mass transfer resistance to adsorption on the solid surface or the biodegradation reaction can be limiting before equilibrium is reached. Biodegradation is usually expressed by the Monod equation, which makes the degradation rate dependent on the pollutant concentration:

- ✓ Rate= Maximum rate×(Cg/K+Cg)
- ✓ Cg: Pollutant concentration in the gas phase
- ✓ K: Saturation constant [16].

Depending on the pollutant concentration, the rate can be of the order of zero (at high concentrations) or of the order of one (at low concentrations) [15].

Parameters Affecting the Performance of Bio filters

Bio filter Substrate:

Different substrates may be used in bio filtration. To maximize the efficiency of a bio filter, it is important to select a suitable substrate for the immobilization of microorganisms. The characteristics that should be considered in selecting a suitable substrate are:

- ✓ High moisture retention capacity.
- ✓ Large surface area.
- ✓ High porosity.
- ✓ Low fouling rate.
- ✓ Low pressure drop.
- ✓ High strength and rigidity.
- ✓ Low cost.
- ✓ Low bulk density.
- ✓ Relative adsorption capacity of odorous gases [17].

From the perspective of microbial activity, characteristics i to iii are important, but from the perspective of bio filter construction and maintenance, characteristics iv to viii are important. Characteristic ix is significant when the concentration of odorous gases fluctuates. Various types of substrates, including natural, synthetic, and mixtures of both, have been used in bio filters.

Peat, soil, compost or their mixtures are the most common materials used as substrates for bio filters. However, these materials are not durable substrates and after a long period of use, several operational problems occur due to low porosity and adhesion of the substrate materials, which cause pressure drop and channelization of the flow [18].

These materials are subject to problems such as clogging and moisture loss during long-term operation. Maintenance of the substrate materials, which require replacement every 2-5 years, significantly increases operating costs. Replacement of the materials and replacement of new substrate requires a shutdown of the operation and the efficiency of the bio filter decreases until the new substrate is adapted and the efficiency reaches the previous level. Various types of synthetic substrates such as porous ceramics, granulated activated carbon, activated carbon fibers, polystyrene spheres and perlite have been used. However, these substrates are much more expensive than natural substrates. The use of inorganic substrates has gradually increased. The most important objectives of using mineral materials in bio filter beds are to limit pressure drop and avoid channeling of the flow. The possibility of using natural and porous igneous rock substrates in bio filtration has also been investigated and this substrate has been suggested as a suitable option for stabilizing microorganisms in bio filters [19].

Bio filter beds may be simple or multi-stage. In most cases, the height of the bed is between 0.5 and 1 m, but in some cases higher substrates have been used. Unlike mineral substrates, organic matrices provide all the nutrients necessary for microorganisms and do not require the supply of nutrients. However, there have been reports of nutrient deficiency and limitation in these substrates. To limit such risks, some researchers have proposed intermittent spraying of nutrients from above the bed.

Humidity

The moisture content of the substrate is another key parameter because the presence of water is necessary to ensure optimal microbial activity. If the moisture content of the substrate is too low, biological activity ceases. Furthermore, in a dry substrate, cracks open and channelization of the

flow occurs. Conversely, too high a moisture content causes the formation of anaerobic zones in the substrate, where the oxygen required for bio oxidation is depleted. For this reason, the capacity of soil substrates to remove odor is greatly reduced when the substrate becomes too wet [20].

Furthermore, the mass transfer of odorous compounds through a wet substrate is limited due to the presence of a thin film of water surrounding the substrate particles. Peat and compost have a good water retention capacity. In the case of peat, if the moisture content falls below 70%, microbial activity decreases, and if the moisture content rises above 85%, microbial activity increases. A moisture content below 30% usually causes biological activity to cease. Soil is much less moisture-resistant than compost due to the smaller pore size. However, soil is hydrophilic, while dry compost is hydrophobic, and rewetting it after drying is difficult due to its high porosity. The heat generated by biological activity in the bio filter may raise the temperature of the bed above the temperature of the incoming gas phase. Even if the gas entering the bio filter is saturated with water, it will become unsaturated because its temperature increases after contact with the bed. Drying of the bed is inevitable. This is partially compensated by biological degradation that produces water (biological oxidation of volatile organic compounds). Therefore, humidifying the air entering the bio filter or wetting the bed to replace the lost moisture is important [15].

Porosity

Porosity is important because of its effect on the pressure drop of the gas passing through the bed. The porosity of organic beds is in the range of 40-50% for soil and 50-80% for compost. Peat is typically 90% porous, with one-fifth of its pores being less than 30μm in diameter. Hard materials such as wood chips, neutral substrates, and perlite can be used to increase the porosity of organic matter [15].

Pressure drop

The gas phase pressure drop in a bio filter increases with increasing flow rate and decreasing bed particle size. Young and Allen reported a linear relationship between the pressure drop and increasing gas velocity over a range of bed particle sizes. However, the pressure drop appears to increase exponentially with decreasing particle size, especially for particles less than 1 mm in size. Brennan and colleagues found that fibrous mixtures significantly increased the pressure drop compared to peat or granules. The pressure drop depends on the nature of the bio filter bed and its moisture content. Adhesion of the bed during long periods of use and by excessive wetting also increases the pressure drop. Proper selection and design of the base reduces the pressure drop. Williams and Miller, in their research on bio filters, considered the pressure drop produced by various beds unpredictable and proposed pilot scale testing for each type of bed [21]. Reducing biomass growth is also considered in controlling the pressure drop. Reducing biomass growth is a common method for reducing pressure drop, although it also affects removal efficiency [22]. It is important to monitor pressure drop to detect cracks in the bed. Any increase in pressure drop increases the operating costs of the bio filter because the inlet air must be supplied at a higher pressure to achieve the same flow rate.

Specific surface area of the bed

Specific surface area for soil and compost is in the range of 1–100 m^2/g. Peat has a larger surface area, resulting in a higher adsorption capacity. The larger surface area provides access for biological growth and adsorption, two mechanisms for reducing odorous substances from the gas phase. The adsorption capacity of the bed enables the bio filter to withstand fluctuations in loading without reducing the removal rate. Amounts greater than the average concentration of odorous compounds are adsorbed onto the bed surface. Microorganisms can meet their carbon and energy needs during periods of low loading from the compounds adsorbed onto the bed surface. Many organic bases used for bio filtration have a high capacity to adsorb odorous compounds [23].

PH

Biological metabolism is strongly dependent on PH. Many microorganisms grow only within a specific PH range. As a rule of thumb, most biological growth occurs near neutral PH and large deviations from it will result in reduced bio filter efficiency. In some cases, the natural characteristics of the gases being treated may have neutralizing effects. A notable exception is the case of sulfur-oxidizing bacteria, which grow well at low PH. Some researchers have achieved high removal efficiencies for volatile organic compounds under acidic conditions, generally as a result of fungal growth or the growth of extremophilic bacteria. Biological oxidation in most cases leads to an increase in acidity. This is due to the production of nitric acid as a product of ammonia degradation, sulfuric acid as a product of sulfide degradation, and carbon dioxide as a product of organic oxidation. Young and Wallen found that the H_2S removal efficiency of their bio filter decreased sharply at PH below 2.3 but was almost independent of PH at higher PHs. They identified the dominant active species as acidophilic bacteria that prefer an optimum PH of about 3. Brennan and colleagues also reported a decrease in PH from 5.6-7.0 to 4.8-6.3 after three weeks in bio filters exposed to a flow of hydrogen sulfide and methyl mercaptan.

Gas flow rate

Gas phase residence times in bio filters can vary from a few minutes for alcohols to 3 hours for 90% removal of trichloroethylene. Examples include a residence time of 15 seconds for removal of meat industry odorants and 23 seconds for removal of 99% of hydrogen sulfide. Maximum removal in a bio filter is achieved when the gas flow rate is as high as possible without increasing the maximum removal capacity of the bio filter. Another problem that arises is that the mass transfer of odorants from the gas phase to the biofilm is limiting. Young and Allen found that for residence times shorter than 23 seconds, the bio filter removal rate is affected by the resistance to the transfer of hydrogen sulfide from the gas phase to the biofilm. The apparent gas phase velocity in odorant removal bio filters is typically 30-200 m/h (50-300 cm/min) [24].

Pollutant concentration

A concentration load of 0-500 ppm is common in bio filters. When first-order kinetics are present in a bio filter, the removal rate is dependent on the concentration of odorous compounds in the inlet gas stream. This will continue until the inlet concentration is low enough that the biodegradation kinetics reach saturation (zero degree). Young and Wallen were able to remove more than 99% of hydrogen sulfide from the inlet gas in the concentration range of 5-2650 ppm, while the removal capacity was below its maximum. Williams and Miller suggested in their study that at least 100 parts of oxygen in the air stream should be provided for each part of oxidizable odorous gas. Since odorous substances are usually dilute, this condition is met.

Gas Phase Humidity

If the bio filter bed is not wet, the air containing odorous compounds must be pre-humidified before entering to prevent drying of the bed. Koch and colleagues explained in their study that the waste gases from the meat industry should be contacted with water to remove the grease and dust carried with them that cause clogging of the bio filter bed. In addition, the scrubber humidifies the air by over 99%, eliminating the need for additional wetting of the substrate to maintain its moisture content.

Liquid phase

The liquid phase flowing through a bio filter may serve several purposes. The liquid phase may maintain adequate liquid levels, dilute toxic metabolic products, provide additional nutrients for biological growth, and provide buffering agents to regulate PH. Dilution is critical in hydrogen sulfide bio filtration. Brennan and colleagues have emphasized in their research the importance of the water supply system for dilution and maintaining PH in the optimal range for bio filters that remove reduced sulfur compounds (RSC). Although sulfide-removing bacteria may grow well at low PH, other bacteria responsible for the degradation of various odorous compounds usually do not. Therefore, a high sulfide removal rate in a bio filter does not necessarily indicate a higher overall removal of odorous compounds. In a bio filter designed to remove hydrogen sulfide, Yang and Allen regularly rinsed the compost with distilled water to dilute the sulfuric acid that accumulated in the bed. This has increased the life of the bio filter. The apparent velocity required for the liquid phase is usually much lower than that for the gas phase. This is due to the slow drying rate of the bed and the slow biological growth.

Temperature

Low temperatures increase the adsorption of odorous compounds onto the biofilm but slow down microbial growth. High temperatures have the opposite effect. High efficiency for odorous compound removal is achieved in the temperature range of 25-40°C with an optimum value of about 37°C. The bed temperature is determined by the inlet gas temperature because the heat released by biological oxidation is negligible. Hydrogen sulfide oxidizing bacteria in the Young and Allen research bio filter showed greater activity in the temperature range of 25-50°C. Brennan and colleagues found in their study that the rate of hydrogen sulfide and methyl mercaptan removal in the bio filter decreased by more than 50% as the temperature decreased from 20-22°C to 9-12°C. Bio filtration can be effective at low temperatures. Pint and colleagues found that once the microbial population in the bio filter is established, odorant removal does not decrease at temperatures below 10°C. Even temperatures as low as 2°C have been reported for propane treatment. It should be emphasized that temporary biological inactivity does not necessarily mean a complete reduction in removal capacity, as adsorption may still occur. Therefore, bio filters can adapt to temporary temperature reductions even at temperatures below 0°C [25].

Microorganisms

Bacteria and fungi are the two main groups of microorganisms in bio filters. Conventional bio filters with a dominant bacterial population can rapidly and effectively degrade highly water-soluble compounds. However, bacterial bio filters face difficulties in removing hydrophobic compounds. These compounds are poorly adsorbed by the bacterial biofilm, and the operational stability of the bio filter is often impaired due to acidification and drying of the bio filter bed. To overcome these problems, fungal bio filters have been developed. Fungi can capture hydrophobic compounds faster than bacteria due to their mycelial structure, which provides a very large contact surface, and may be in

direct contact with the gas phase passing through the bioreactor. In addition, fungi are more resistant to dry and acidic conditions compared to bacteria. Recent research shows that fungi can degrade a variety of volatile organic compounds at a rate equal to or greater than bacterial systems. There are several reports of successful treatment of hydrophobic compounds from gaseous wastes in fungal bio filters. However, fungal growth in bio filters also presents problems. High pressure drops are reached more quickly in the presence of filamentous fungi, which ultimately leads to clogging and channelization of the flow in the bio filter, reducing efficiency. Recently, solutions such as adding materials to the bio filter bed have been tested to reduce this problem. Other methods such as backwashing, air blowing or the use of specific chemical compounds may be used to control the pressure drop. In addition, the range of substrates on which fungi grow is much smaller compared to bacteria, which may limit their application. Various types of microorganisms have been used to remove reduced sulfur compounds. However, further studies on the microbial ecology of bio filters are needed, especially to investigate the dynamics of microbial populations during biological treatment.

A biological adaptation period, usually of several weeks to several months, is usually required to achieve high efficiencies, especially for the treatment of more resistant compounds or compounds that are less soluble. The biological adaptation period is necessary to adapt to the new conditions resulting from a new medium. These conditions include nutrients, PH, temperature, and substrate characteristics. Microorganisms often improve their ability to remove odorants over the life of the bio filter due to their ability to grow [1].

Williams and Miller, in a study of operational bio filters, reported bio acclimation periods of more than 10 days. Longer periods have also been reported by Wang et al. They found that biomass accumulation was a slow process, taking 5 months for the biomass to reach equilibrium concentration. The substrate concentration was lower at the bottom of the bed, indicating that most of the biological degradation was occurring at the surface of the bed, where initial contact with the waste occurred. Furthermore, they found that the capacity of the bed to remove organic matter did not always increase with biomass concentration. Yang and Allen found that a bio acclimation period was necessary to achieve maximum removal efficiency for hydrogen sulfide. This period was required for the microbial population to reach optimum levels after exposure to hydrogen sulfide. The time required was found to be 2 weeks.

Since in the present study, the bacteria Ralstonia eutropha and the white filamentous fungus Phanerocote Chrysosporium are the microbial species under test to investigate the biodegradation of ethyl mercaptan, a brief description of these microorganisms is given in this section:

Bacteria Ralstonia eutropha

Bacteria Ralstonia eutropha is a microbe found in soil and water. This bacterium has high potential for use in wastewater treatment processes and is capable of using various aromatic compounds and chemical pollutants. Ralstonia eutropha is a gram-negative bacterium with a non-spore structure and forms white colonies. Many gram-negative bacteria are pathogenic, but this bacterium is not pathogenic. This bacterium is motile and facultative aerobic, and also has two flagella and two membranes and is rod-shaped. The optimum growth temperature for this microbe is 30 degrees Celsius.

White filamentous fungus Phanerocote Chrysosporium

White filamentous fungi are the most important lignin decomposers among microorganisms found in wood. After the discovery of lignin lytic enzymes of white filamentous fungi, the use of fungi and their non-specific enzymes for bioremediation was proposed. White filamentous fungi have a unique lignin degradative system that is suitable not only for removing lignin from wood but also for degrading various pollutants that have a similar structure to lignin. Lignin lytic enzymes oxidize various types of environmental pollutants such as polycyclic aromatic hydrocarbons, chlorophenols and aromatic dyes. The most abundant lignin lytic enzymes produced by these fungi are lignin peroxidases (LiP), manganese peroxidases (MnP) and phenol oxidases (laccases). In addition to the non-specificity of lignin lytic enzymes, the lignin lytic system of some white filamentous fungi is not induced by lignin or other related compounds. This feature allows the degradation of pollutants at relatively low concentrations [12].

This concentration may be lower than that required to induce the synthesis of degrading enzymes in other organisms. Extracellular lignin lytic enzymes initiate the oxidation of substrates in the extracellular environment. Since lignin is degraded non-specifically and through radical oxidation, lignin-degrading fungi have the ability to degrade a mixture of different pollutants. This feature is the greatest advantage of using white filamentous fungi in bioremediation. The number of compounds degraded by white filamentous fungi is increasing.

Lignin lytic enzymes produce cationic radicals as a result of single-electron oxidation. The cationic radicals catalyze reactions such as C-C bond cleavage or hydroxylation, which lead to the production of more hydrophilic products. These products are taken up by fungal cells and metabolized to carbon dioxide (CO_2) in the presence of a suitable carbon source. Despite much research on the oxidative mechanism of lignin lytic enzymes,

the mechanism of lignin degradation and oxidation of its related compounds is not fully understood. The reaction of lignin lytic enzymes is very complex and involves a variety of low molecular weight cofactors that act as reducing intermediates.

The phanerochaete Chrysosporium species is the most commonly used microorganism in lignin biodegradation research. The reasons for the popularity of this microorganism for research and its use for the decomposition of lignin and similar compounds are its rapid growth and rapid metabolism of lignin and similar compounds, its ability to grow optimally at a relatively high temperature of 40 degrees Celsius, its ability to produce conidia (asexual spores) and basidiospores (sexual spores), its ability to grow on specific chemical media, and the existence of considerable information and knowledge about its ecology, physiology, biochemistry, molecular biology, and genetics.

Lignin lytic enzymes

- ✓ **Two classes of extracellular oxidative enzymes are responsible for lignin degradation:** peroxidases and laccases (phenol oxidases) [16].
- ✓ **Three types of peroxidases are involved in lignin degradation by white filamentous fungi:** lignin peroxidase (LiP), manganese peroxidase (MnP), which was first identified in the phanerokeate Chrysosporium, and versatile peroxidase (VP), which was recently identified in Pleuritic and Bjerkandera species.

Some fungi possess both classes, while others possess one or two of the enzymes. Lignin lytic enzymes of white filamentous fungi are not only directly involved in the degradation of lignin in natural lignocellulosic materials, but also in the degradation of many hard-to-degrade compounds such as dyes. Lignin lytic enzymes are required for lignin degradation, while for mineralization, they are usually accompanied by other processes in which additional enzymes are involved. The physiology of lignin lytic enzyme production by white filamentous fungi for delignification or degradation of resistant contaminants has been extensively studied. In summary, lignin lytic enzymes are produced by white filamentous fungi during secondary metabolism. Since lignin oxidation does not provide any net energy to the fungus, the synthesis and secretion of these enzymes is often induced by nutrient limitation (carbon or nitrogen source). In most species, peroxidases and laccases are expressed as different isoforms. Usually, more than one isoform of lignin lytic enzymes is produced by different species under different conditions. Both classes of lignin lytic enzymes are glycosylated, which may increase the stability of these enzymes. Peroxidases are heme-containing enzymes. They require the presence of hydrogen peroxide for the oxidation of lignin and its related compounds. Their molecular weight and isoelectric points vary from 35 to 47kDa and 2.8 to 4.5kDa. Peroxidases are of two types based on the very different range of substrates they act on. One type is manganese peroxidase (MnP), for which Mn^{+2} is the best substrate. The other type is lignin peroxidase (LiP). Lignin peroxidase oxidizes phenolic and non-phenolic aromatic compounds. Laccases are copper-containing phenol oxidases. These enzymes oxidize phenols and aromatic amines. In addition to hydrogen peroxide, these enzymes use molecular oxygen as an oxidizing agent and reduce it to water via four electrons. Laccases have molecular weights of 50-300kDa and have acidic isoelectric points.

Kinetics of oxidation of environmental pollutants by lignin lytic enzymes

Catalytic cycle of peroxidases

Lignin lytic peroxidases have a characteristic catalytic cycle that is also characteristic of other peroxidases. The primary enzyme (which contains iron compounds) is first oxidized by hydrogen peroxide, resulting in a two-electron oxidizing form of the enzyme (compound I). During the oxidation of the ferric enzyme by hydrogen peroxide, one electron is taken from Fe^{+3} and one electron from the porphyrin ring, producing Fe^{+4} and a porphyrin cation radical. Compound I is reduced to the primary form in two steps via the intermediate compound Fe^{+4} (compound II) in the presence of a suitable substrate. The first two reactions are rapid, while the reaction of compound II is usually ten times slower. Therefore, k_3 is very effective in the rate of enzyme turnover (kcat). The reaction of compound II from LiP and MnP has a hyperbolic dependence on substrate concentration. This is consistent with the initial equilibrium binding step of the substrate occurring rapidly and before electron transfer. Another reaction also occurs in the presence of excess hydrogen peroxide or substrate. Compound II can react with hydrogen peroxide to form a less active form of peroxidase (compound III).

Manganese peroxidase reactions: The most common peroxidase produced by almost all white filamentous fungi is manganese peroxidase. This enzyme is an iron-containing glycoprotein with a molecular weight in the range of 32 to 62.5kDa and is produced in various isoforms. Manganese peroxidase is unique among peroxidases due to its affinity for substrates complexed with Mn(II). Compound I derived from MnP is capable of oxidizing various types of phenols such as glycol and 2,6-dimethoxyphenol, but with lower efficiency than Mn(II).

The reduction of compound II exclusively requires Mn(II) as a substrate. During the enzyme cycle, two molecules of Mn(II) are oxidized to Mn(II). Mn(II) binds to organic acids such as oxalate (a metabolite produced by white filamentous fungi). The Mn(III)-oxalate complex is very stable and is capable of oxidizing a variety of phenolic compounds that have a lower reduction potential than the Mn(II)-oxalate complex [16]. Therefore, MnP is capable of depolymerizing and oxidizing lignin and persistent pollutants such as textile dyes.

Lignin peroxidase reactions

Lignin peroxidase catalyzes the oxidation of non-phenolic aromatic moieties of lignin and similar compounds. LiP has a molecular weight in the range of 38-47kDa. LiP catalyzes the oxidation of lignin side chains and related compounds by removing an electron and forming active radicals. The role of LiP in delignification is to convert the lignin fractions produced by the initial action of MnP. This enzyme is not essential for the attack on lignin, and many highly active white filamentous fungi do not produce this enzyme. LiP has been used to mineralize a variety of resistant aromatic compounds such as dyes. 2-Chloro-4,1-dimethoxybenzene, a natural metabolite of white filamentous fungi, has been reported as a redox-mediator in the oxidation by LiP. The range of substrates on which LiP is effective is very different from that of MnP. The compound that is used as a substrate for this enzyme is determined by two factors: The size of the molecule and its reduction potential. Research shows that LiP is capable of oxidizing molecules larger than nonaromatic or diaromatic compounds. Lignin peroxidase has a higher reduction potential than other peroxidases, which makes it a better oxidizer for polyromantic hydrocarbons (PAH). However, not all polyromantic hydrocarbons are degraded by LiP.

Polyromantic hydrocarbons with a reduction potential of 7.7 eV or higher cannot be oxidized by LiP. Furthermore, the oxidation potential of compounds I and II is different. Compound I is able to oxidize compounds with higher reduction potential. The addition of a secondary substrate can facilitate the oxidation of the more difficultly degradable primary substrate by acting as a reducing agent for compound II. Tyne and Koduri (1994) were the first to make this observation and explain the role of secondary substrates [13]. A suitable secondary substrate is veratyryl alcohol [13]. White filamentous fungi produce veratyryl alcohol (4,3-dimethoxybenzyl alcohol), a secondary metabolite. Its role in lignin degradation is unknown. Veratyryl alcohol is an ideal substrate for the synthesis of I and II. The addition of veratyryl alcohol enhances the oxidation of many different substrates such as lignin, aromatic dyes, anisyl alcohol (4-methoxybenzyl alcohol), guaiacol, pentachlorophenol and benzopyrene. Harvey (1986) proposed the role of veratyryl alcohol as an intermediate in the lignin lytic system [8]. The radical cation of veratyryl alcohol provides the oxidizing power of the active sites to different targets. The role of veratyryl alcohol as an intermediate in the oxidation of various compounds has been demonstrated by enzyme kinetic studies.

Methoxymadelic acid is very well oxidized by LiP in the presence of veratyryl alcohol. Increasing the amount of veratyryl alcohol increased the rate of the initial oxidation of the acid, while the oxidation of veratyryl alcohol to veratyryl aldehyde was inhibited until the acid was removed from the reaction mixture. Similar observations were obtained when 4-methoxymadelic acid was replaced by guaiacol, chlorpromazine, or pentachlorophenol. According to these observations, veratyryl alcohol acts as an intermediate. LiP also oxidizes anisyl alcohol, which has a higher reduction potential than veratyryl alcohol. When veratyryl alcohol is added to the system, the formation of anisaldehyde occurs much more rapidly. Tyne and Koduri (1994) showed that the role of veratyryl alcohol in this case is not as an intermediate compound. Unsteady-state kinetic experiments showed that anisyl alcohol is a relatively good substrate for compound I but is not oxidized by compound II. When small amounts of veratyryl alcohol are added to the system, the formation of anisaldehyde increases, but increasing the amount of veratyryl alcohol leads to less formation of anisaldehyde. If veratyryl alcohol acts as an intermediate, increasing its amount should lead to an increase in the amount of anisaldehyde produced. Veratyryl alcohol can react with compound II. As a result, small amounts of veratyryl alcohol prevent the enzyme from remaining in the oxidized form of compound II. When large amounts of veratyryl alcohol are added, both veratyryl alcohol and anisyl alcohol compete for compound I, leading to less formation of anisaldehyde. Apart from its role as an intermediate, veratyryl alcohol could be useful in other pathways.

Discussion

Biological degradation is the most common and widespread technique used in the treatment of colored wastewater. The advantages of this method are low cost, inorganic and non-toxic product. This process is carried out aerobically (in the presence of oxygen) or anaerobically (in the absence of oxygen) or a combination of aerobic and anaerobic. Many microorganisms such as bacteria, fungi and algae have the ability to remove pollutants. However, due to the need for a large surface area for interaction, varying sensitivity to various chemicals and toxicity, their application is limited. In this process, the metabolic ability of microorganisms is used to destroy hazardous pollutants. The goal of this process is to convert organic pollutants into

harmless metabolic products or to mineralize pollutants into carbon dioxide and water. Potentially, biological degradation is used to remove any compound that microorganisms can trap or absorb. Various hydrocarbons, chlorinated hydrocarbons, aromatic nitrogen compounds, organophosphorus compounds, cyanides can be removed through biological degradation. It is clear that microbial-based treatments of environmental pollutants and industrial wastewaters are economically very cost-effective compared to other conventional methods such as adsorption, oxidation processes [22].

A study of previous research shows that the basis of the presented models is the model of Ettenberg et al., which was presented in 1983, in which the biological degradation process follows the Monod kinetics. This degradation becomes first-order kinetics at low concentrations and zero-order kinetics at high concentrations. The model of Sharafuddin et al. was presented in 1993 to investigate the behavior of methanol vapor removal. In this model, while considering the effect of oxygen gas, it is assumed that the entire load is covered by biofilm. However, in the model of Sharafuddin et al. (1994), it is assumed that part of the influent is covered by biofilm and the other part is exposed to gas phase contact.

The model of Amanollah et al. in 1999 takes into account more facts about the bio filtration process and their assumptions are as follows: the existence of surface adsorption from the biological layer to the filler and the existence of a biodegradation term within the adsorbent, which is assumed to be of degree n, and the surface adsorption reaction was considered isothermal, and the pollutant concentration within the filler has a linear relationship with the pollutant concentration within the gas phase, and the mass transfer rate into the filler follows the linear driving force method.

Bakouriz et al. (2005) presented a model for ammonia removal by bio filtration, which is based on the mathematical equations of mass balance for 4 phases (gas - liquid - biofilm and solid phases). The model of Chamil et al. (2005) used the concept of linear driving force to approximate the interphase transfer of pollutants, in which the surface adsorption of the model pollutant to the filler is taken into account and the biodegradation of the pollutant inside the filler is obtained using the modified Monod equation.

Reducing PM2.5 productions in large cities requires investment. The main solution to reduce PM2.5 is lower fuel consumption and increased energy efficiency in all sectors. If you achieve the implementation of such a policy in a focused and continuous manner in the country's programs, it can be said that the problem of air pollution in metropolitan areas will be resolved in the future, otherwise we will witness more emergency and dangerous days in the air of metropolitan areas in the coming years. Perhaps the only thing people can do is reduce the amount of activity in emergency conditions, and to completely solve the problem, the government must implement comprehensive planning and strict monitoring plans. For example, substandard buildings should not be built in big cities, substandard vehicles should not circulate in big cities, and polluting industries should be moved to locations further away from the city.

Most mechanical air filters are effective at trapping larger airborne particles, such as dust, pollen, and cockroach allergens, some molds, and animal dander. However, because these particles settle quickly, air filters are not very good at removing them completely from indoor areas. Although human activities such as walking and vacuuming can recirculate particles back into the air, many larger particles settle before they can be removed by the air filter. Consumers can choose an air filter based on how effectively an air filter removes particles from the air stream passing through it. According to the ASHRAE methodology, filter efficiency is measured by the Minimum Efficiency Reporting Value (MERV) for air filters installed in HVAC ducts. MERV values (range 1 as the lowest to 20 as the highest) allow comparisons between air filters made by different companies [23].

Flat or panel air filters with MERV ratings between 1 and 4 are commonly used in HVAC systems and some residential fireplaces. In most cases, these filters are used to protect HVAC equipment from the buildup of undesirable materials on surfaces such as fan motors and heating and cooling coils, and are not intended to directly improve indoor air quality. These filters are poorly effective at removing light particles. They are also moderately effective at removing larger particles, provided the particles remain suspended in the air and pass through the filter. They are not very effective at removing smaller particles found in the home, including viruses, bacteria, spores, mold, significant amounts of dog and cat dander, and small amounts of allergens.

Medium-efficiency filters with MERV ratings between 5 and 13 are reasonably effective at removing small to large airborne particles. Filters with MERV ratings between 7 and 13 are about as effective as HEPA filters at controlling many particles in indoor air. Medium-efficiency air filters are typically less expensive than HEPA filters. Additionally, due to the lower airflow resistance of these filters, air conditioning fans operate quieter and have higher airflow rates than HEPA filters. Higher efficiency filters with MERV ratings of 14 to 16, sometimes incorrectly referred to as HEPA filters, are similar in appearance to true HEPA filters with MERV ratings of 17 to 20. True HEPA filters are not typically installed in residential air conditioning systems. This is because installing a

HEPA filter in an existing air conditioning system would likely require professional modification of the system. A typical residential air handling unit and its associated air ductwork would not be able to accommodate such filters due to their physical size and increased airflow resistance [24].

In some residential air conditioning systems, the fan or motor may not have enough capacity to accommodate higher efficiency filters. Therefore, the air conditioning manufacturer's information should be reviewed before upgrading filters to assess the feasibility of using more effective filters. In homes that are specifically built and have high efficiency, it may be possible to use a true HEPA filter installed in a properly designed air conditioning system.

There is no standard method for measuring the effectiveness of electronic air purification systems. While these systems remove small particles, they may be ineffective at removing large particles. Electronic air purifiers can produce ozone, which can be a lung irritant. The amount of ozone produced varies among models. Electronic air purifiers may also produce very fine particles as the ozone reacts with chemicals in the building. These chemicals can come from products such as household cleaners, air fresheners, certain types of paint, hardwood floors, or carpets. Very fine particles may be associated with health problems in some sensitive groups.

Although there is no standard measurement for measuring the effectiveness of gas-phase air filters, ASHRAE is currently developing a standard method for selecting gas-phase filters for installation in HVAC systems. Gas-phase filters are much less commonly used in homes than particulate air filters. The useful life of gas-phase filters can be short due to rapid saturation of the filter material and the need for filter replacement. There is also concern that when these filters become full, the trapped contaminants are released back into the air. Ultimately, it seems unlikely that a properly designed and constructed gas-phase purification system can be incorporated into an HVAC system or portable air purifier [25].

There is no standard method for measuring the effectiveness of UVGI purifiers. Common household UVGI purifiers have little effect on killing bacteria and mold. Killing some viruses and most molds requires the use of UV radiation at levels higher than those observed in a typical home. In addition, dead mold spores can still cause allergic reactions, so UVGI purifiers may not be effective in reducing allergy and asthma symptoms. There is also no standard method for measuring the effectiveness of PCO purifiers. The use of PCO purifiers in homes is limited by the ineffectiveness of the catalysts used in removing gaseous pollutants from indoor air. Some PCO purifiers not only fail to completely remove pollutants, but also produce new indoor pollutants that may irritate the eyes, throat, and nose [26].

Conclusion

Microorganism selection-catalysis of biological decontamination is carried out using an enzyme complex of oxidoreductases in a selected cellular system. In this regard, the presence of ethyl mercaptan pollutant and the biological system in the aqueous-solid phase is considered. The importance of cytochrome P450 should also be considered in this regard. The selection of "Pure culture" fungi or bacteria is carried out considering the microbial system's possession of the above-mentioned enzymatic-cellular positions. In the present study, the position of Chrysosporium phanerocoete with the manganese peroxidase and lignin peroxidase enzymatic systems as well as the presence of cytochrome P450 cellular system was considered and the selection of Chrysosporium phanerocoete was carried out based on these documents. On the other hand, the attention to the gram-negative bacterium Ralstonia eutropha was given by considering the presence of several oxidoreductase enzymatic systems and the possibility of selecting this microorganism was made on this basis. Bio adaptation of the selected microorganism is carried out by introducing ethyl mercaptan in several concentrations. The performance of the microorganism is evaluated by determining the capacity to destroy the cell system under test as the response variable. The study of the growth kinetics of the microorganism by considering the growth-limiting substrate - the substrate itself has an inhibitory role on growth - for example, the Monod relationship in its expanded form is proposed to consider substrate inhibition. In addition, other relationships such as the Holden and Long relationships can also be considered in this category.

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References

[1] Yang, Y., Huang, S., Zhang, Y., & Xu, F. (**2013**). Field applications for NOx removal from flue gas in a biotrickling filter by Chelatococcus daeguensis TAD1. *Aerosol and Air Quality Research, 13*(6), 1824-1831.

[2] Samimi, A. Zarinabadi, S. (**2012**), The Comparison of Increasing Method for Petroleum Pits Output (Fluids Dynamic), *International Journal of Basic and Applied science*, 435-439

[3] Yoshikawa, M., Zhang, M., & Toyota, K. (**2017**). Biodegradation of Volatile Organic Compounds and Their Effects on Biodegradability under Co-Existing Conditions. *Microbes and Environments*, *32*(3), 188-200.

[4] Samimi, A. (**2013**), Micro-organisms of cooling tower problems and how to manage them, *International Journal of Basic and Applied science*, 705

[5] Lenis, A., Ramírez, M., González-Cortés, J. J., Ooms, K., & Pinnekamp, J. (**2023**). Implementation of a pilot-scale biotrickling filtration process for biogas desulfurization under anoxic conditions using agricultural digestate as trickling liquid. *Bioengineering*, *10*(2), Article 160.

[6] Sundari, U. Y., Qubra, T., Chiarawipa, R., & Defri, I. (**2024**). A review of parameters, performance models, and applications of biofiltration technology in reducing H_2S compounds in the air. *Agroindustrial Technology Journal*, *7*(3), 46-64.

[7] Swanson, W. J., & Loehr, R. C. (**1997**). Biofiltration: Fundamentals, design and operations principles and applications. *Journal of Environmental Engineering*, *123*(6), 538-546.

[8] Fernández, M., Ramírez, M., Pérez, R. M., Gómez Montes de Oca, J. M., & Cantero Moreno, D. (**2013**). Hydrogen sulphide removal from biogas by anoxic biotrickling filter packed with Pall rings. *Chemical Engineering Journal*, *225*, 456-463.

[9] Wierzbińska, M. (**2023**). Deodorisation of industrial gases using a biofiltration plant. *Journal of Ecological Engineering*, *24*(10), 47-56.

[10] Samimi, A. Zarinabadi, S. (**2012**), The Comparison of Increasing Method for Petroleum Pits Output (Fluids Dynamic), *International Journal of Basic and Applied science*, 435-439

[11] Samimi, A. Zarinabadi, S. Setoudeh, M. Safavian, A. (**2012**), Review Applications to Prevent Corrosion Reducing Gas Pipe Line, *International Journal of Basic and Applied science*, 423-428

[12] Hedayati, A. Almasinia, B. Samimi, A. (**2014**), Optimize pictures of industrial radiography in corrosion and sediment recognizing in oil or gas transmit pipe lines, *International Journal of Chemistry*, 5, 20-29

[13] Leson, G., & Winer, A. M. (1991). Biofiltration: An innovative air pollution control technology for VOC emissions. *Journal of the Air & Waste Management Association*, *41*(8), 1045–1054.

[14] Swanson, W. J., & Loehr, R. C. (1997). Biofiltration: Fundamentals, design and operations principles, and applications. *Journal of Environmental Engineering*, *123*(6), 538–546.

[15] Kennes, C., Thalasso, F., & Veiga, M. C. (2002). Fungal biofilters for the treatment of waste gases containing volatile organic compounds. *Journal of the Air & Waste Management Association*, *52*(9), 944–954.

[16] Shareefdeen, Z., Baltzis, B. C., Oh, Y. S., & Bartha, R. (1993). Biofiltration of methanol vapor. *Biotechnology and Bioengineering*, *41*(5), 512–524.

[17] Rene, E. R., & Veiga, M. C. (2009). Biodegradation of mixtures of BTEX vapors using a compost biofilter. *Journal of Hazardous Materials*, *168*(1), 282–288.

[18] Jorio, H., Kiared, K., Brzezinski, R., Leroux, A., & Heitz, M. (2000). Biofiltration of air contaminated with toluene on compost filter media. *Environmental Science & Technology*, *34*(9), 1764–1771.

[19] Chung, Y. C., Huang, C., & Tseng, C. P. (1996). Biological elimination of hydrogen sulfide and ammonia from waste gases by biofilter packed with immobilized heterotrophic bacteria. *Chemosphere*, *33*(12), 2447–2455.

[20] Malhautier, L., Khammar, N., Bayle, S., & Fanlo, J. L. (2005). Biofiltration of volatile organic compounds. *Applied Microbiology and Biotechnology*, *68*(4), 398–404.

[21] Ramirez, M., Gómez, J. M., & Cantero, D. (2012). Biological removal of hydrogen sulfide from air: A review. *Biochemical Engineering Journal*, *64*, 19–29.

[22] Yang, Y., Huang, S., & Xu, F. (2013). Field applications for NOx removal from flue gas in a biotrickling filter. *Aerosol and Air Quality Research*, *13*(6), 1824–1831.

[23] Mudliar, S., et al. (2010). Bioreactors for treatment of VOCs and odorous emissions. *Journal of Environmental Management*, *91*(5), 1039–1054.

[24] Rene, E. R., Montes, M., & Veiga, M. C. (2010). Biotrickling filters for VOC abatement: A review. *Environmental Technology*, *31*(8–9), 841–857.

[25] Diks, R. M., & Ottengraf, S. P. P. (1991). Verification studies of a simplified model for the removal of dichloromethane from waste gases in a biological trickling filter. *Bioprocess Engineering*, *6*(3), 131–140.

[26] Rene, E. R., & Veiga, M. C. (2009). Performance of biofilters and biotrickling filters for VOCs. *Journal of Chemical Technology & Biotechnology*, *84*(10), 1421–1432.