ENCYCLOPEDIA OF

BIOPROCESS TECHNOLOGY:

FERMENTATION, BIOCATALYSIS,
AND
BIOSEPARATION

Michael C. Flickinger
University of Minnesota
St. Paul, Minnesota

Stephen W. Drew
Merck and Co., Inc.
Rahway, New Jersey

John Wiley & Sons, Inc.

BIOFILTERS

MARIO ZILLI
ATTILIO CONVERTI
Genoa University
Genoa, Italy

KEY WORDS

Air pollution control biotechnologies
Biofilter
Biofiltration
Biological methods
Degradation pathways
Microorganisms

OUTLINE

Introduction
Biological Technologies for Waste-Gas Treatment

INTRODUCTION

Biofiltration is the biological removal of air contaminants from waste gases by means of aerobic microorganisms that are immobilized on a porous solid support. In a biofilter, the waste-air stream is passed through a biologically active packing material; organic or inorganic air pollutants are degraded by microorganisms into harmless end products such as water, carbon dioxide, mineral salts, and new microbial mass. Clearly, biofiltration technology is limited to non-toxic gases with good biodegradability and water solubility, because these gases are used by microorganisms as their sole source of energy and carbon.

The idea of employing microorganisms to eliminate contaminants from waste gases is very old. Initially, biofiltration was used mainly for the treatment of odorous gases (such as hydrogen sulfide and ammonia) emanating from sewage-treatment plants. The first reports on actual applications of this technology, dating back to the late 1950s and the early 1960s, dealt with soil-bed installations in West Germany and the United States, where it was demonstrated that biodegradation, rather than sorption, was responsible for most of the odor abatement (1). After the first publication of Bohn's extensive investigation on soil-filter beds (2,3), several additional full-scale applications of biofilters in West Germany and the Netherlands followed. These biofilters were employed to control the odors from wastewater- and thermal-sludge-treatment plants,

composting facilities, rendering plants, carcass incinerators, and others (4–6). Jäger and Jager, who compared several methods for off-gas purification from the composting plant of Heidelberg, first demonstrated the economic convenience of biofiltration (7).

Process and technology development was attempted by Thistlethwayte et al., who utilized a trickling filter column packed with river gravel or glass balls and seeded with activated sludge. Waste air contaminated by H_2S, C_2H_5SH, $(C_2H_5)_2NH$, and C_4H_9CHO was purified, in countercurrent with a nutrient solution, with removal yields ranging from 40% to almost 100% (8). In addition, applications in which organic impurities were eliminated from waste gas, or air in beds inoculated with biologically active material, have been reported together with the first criteria for correct process design and operation (9).

A further development of this technology was marked by a biofilter installation with continuous regeneration of the bed, which was obtained by removing active material at the bottom of the reactor and recharging it after regeneration at the top of the reactor (9). A regeneration process by microbial, thermal, and chemical methods was also reported by Helmer (10), who used dry- and wet-soil filter beds to extract products of anaerobic fermentation of organic materials (e.g., NH_3, H_2S, sulfides, alcohols, and aldehydes). The importance of a sufficient moisture content in the filter bed and of a good distribution of the gas supplied to the biofilter was demonstrated for the first time (11).

Leson and Winer estimated that in 1991 the total number of biofilter and soil-bed installations used in the United States and Canada mainly for odor control was less than 50 (1). There was greater confidence in biofiltration in Germany and the Netherlands, where over 500 biofilters were operating in the same period. In the last two decades, the number of installations has significantly increased in Japan, growing from about 40 in the 1980s to 90 in the 1990s (12). On the contrary, in other European countries, including Italy, Switzerland, and Austria, the number of biofilter installations has been quite limited (1).

Although biofiltration has developed mainly with the primary aim of eliminating odorous compounds from exhaust gases, since the early 1980s this technology has been of increasing importance in air pollution control, and the number of applications of biological filters has strongly increased due to an increasing concern about environmental quality and more stringent air-emission quality standards.

The application field of biofiltration has widened thanks to better knowledge of the biodegradation process, in particular the control of the process conditions, as well as to improved biofilter construction and packing material composition (13,14). In particular, special packing materials having optimal structural properties prevent aging phenomena, decrease the pressure drop over the packed bed, and increase the long-term operational stability of the filter, as well as its microbial activity.

Furthermore, the considerable research carried out on the isolation, selection, and construction of strains or mixed cultures of microorganisms (mainly bacteria) has led to the extension of biodegradation to both anthropogenic (i.e., manmade) and xenobiotic compounds (chemi-

cals that do not occur naturally). Compounds that suffi-
ciently resemble structures of biological origin are
eliminated rather quickly, whereas biodegradation of xe-
nobiotics is very slow (recalcitrant compounds) or even im-
possible (persistent compounds). Nevertheless, the contin-
uous adaptation of microorganisms to new substrates is at
the basis of the increasing number of microbes capable of
degrading xenobiotics.

As a consequence, biofiltration is now a well-recognized
technology that is successfully applied to a wide range of
industrial facilities, waste-disposal and food-processing ac-
tivities to control odors, volatile organic compounds
(VOCs), and air toxics (both inorganic and organic), espe-
cially those that are readily biodegradable.

Biofiltration development is the result of the implemen-
tation of health and safety guidelines, enforced in many
countries to protect people and the environment, combined
with several advantages that this technique has demon-
strated over traditional chemical and physicochemical
treatment methods, such as water washing, chemical
scrubbing (ozonation and chlorination), activated carbon
adsorption, incineration, and catalytic oxidation.

Besides the overall low energy requirements (moderate
temperature and pressure conditions), easy maintenance,
simple control, and low capital and operating costs, it must
be stressed that biological filters do not transfer the pol-
lution problem to another environmental compartment, as
occurs with many other alternative control technologies.
Because the pollutants are converted into harmless oxi-
dation products (which are part of a natural cycle), they do
not create any secondary pollution and thus can be consid-
ered environmentally safe.

BIOLOGICAL TECHNOLOGIES FOR WASTE-GAS TREATMENT

Biological Systems

There are three basic types of biological reactor systems
used to treat waste gases: bioscrubbers, biotrickling filters,
and biofilters. These can be grouped into those employing
microorganisms dispersed freely throughout the liquid
phase (bioscrubbers), and those using microorganisms im-
mobilized on a packing or carrier material (biotrickling fil-
ters and biofilters). Moreover, in bioscrubbers and biotrick-
ling filters the water phase is continuously moving,
whereas in biofilters it is stationary.

A bioscrubber consists of a scrubber unit and a regen-
eration unit. In the scrubber (absorption column), water-
soluble gaseous pollutants are absorbed and partially ox-
idized in the liquid phase (the culture medium containing
the microorganisms), which is distributed from the top of
the unit. The contaminated water is subsequently trans-
ferred into an aerated stirred-tank reactor (regeneration
unit), similar to an activated-sludge unit, where the con-
taminants are fully biodegraded. The regenerated suspen-
sion is continuously recirculated to the top of the scrubber
section, thereby enhancing process efficiency (Fig. 1).

Biotrickling filters (Fig. 2) and biofilters (Fig. 3) are dif-
ferent from bioscrubbers in that gaseous pollutant absorp-
tion and biological degradation occur simultaneously in

the same compartment, resulting in more compact sys-
tems. The polluted air flows through a biologically active
bed, where microorganisms are attached in the form of a
biofilm. As the gas diffuses through the packed bed, the
pollutants are transferred to the biolayer and degraded.

As illustrated in Figure 3, in order to ensure optimal
operation of a biofilter, the inlet gas usually requires pre-
treatment processes such as: (1) particulate removal in or-
der to prevent possible clogging and/or sludge build up,
(2) load equalization in case the waste-gas concentration
is subject to strong fluctuations, (3) temperature control,
and (4) humidification.

In biological trickling filters the packed bed consists
only of inert materials (glass, ceramics, and plastics), while
the liquid phase, containing inorganic nutrients, flows
downward over the packaging material in countercurrent
with the contaminated gaseous stream and is continuously
recirculated through the bioreactor.

Bioscrubbers and biotrickling filters are applicable
mainly to the treatment of waste gases containing good or
moderately water-soluble compounds, whereas biofilters,
due to the large surface area available for mass transfer,
are also suited to treat poorly water-soluble compounds.
Moreover, due to their high reaction selectivity, biofilters
are particularly suitable for treating large volumes of air
containing easily biodegradable pollutants with relatively
low concentrations, typically below 1,000 ppm.

Compared with the other biological systems, biofilters
have the widest application because they are easy to op-
erate, simply structured, and imply low installation and
operating/maintenance costs. Also, the reliability of biofil-
ter operation is higher than that of bioscrubbers, where
the risk exists of washing away the active microorganisms.
Moreover, the presence of a large amount of packing ma-
terial with a buffering capacity diminishes the sensitivity
of biofilters to different kinds of fluctuations (15). Because
the major disadvantage is the difficult control of parame-
ters such as pH, temperature, and nutrient supply, biofil-
ters may be unsuitable for degrading halogenated com-
pounds (as acid metabolites are produced) and treating gas
streams containing high concentrations of VOCs, unless
long residence times or large bed volumes are applied (3).
Biotrickling filters and biofilters are currently utilized
mainly in compost-production plants, sewage-treatment
plants, and agriculture, whereas biofilters and bioscrub-
bers are preferred in industrial applications.

Different Types of Biofilters

Among the different types of biofilters described in the lit-
erature for a variety of applications (16), open biofilters,
consisting of single beds of compost or porous soils, com-
monly 1-m deep, are used mainly for odor and VOC abate-
ment. Because they are in direct contact with the open air,
their performance is strongly influenced by weather con-
ditions (rain, frost, temperature fluctuations, etc.). The
simple design and low cost of these systems are counter-
balanced by difficult monitoring and control, as well as by
large space requirements.

All the other real-scale configurations are closed biofil-
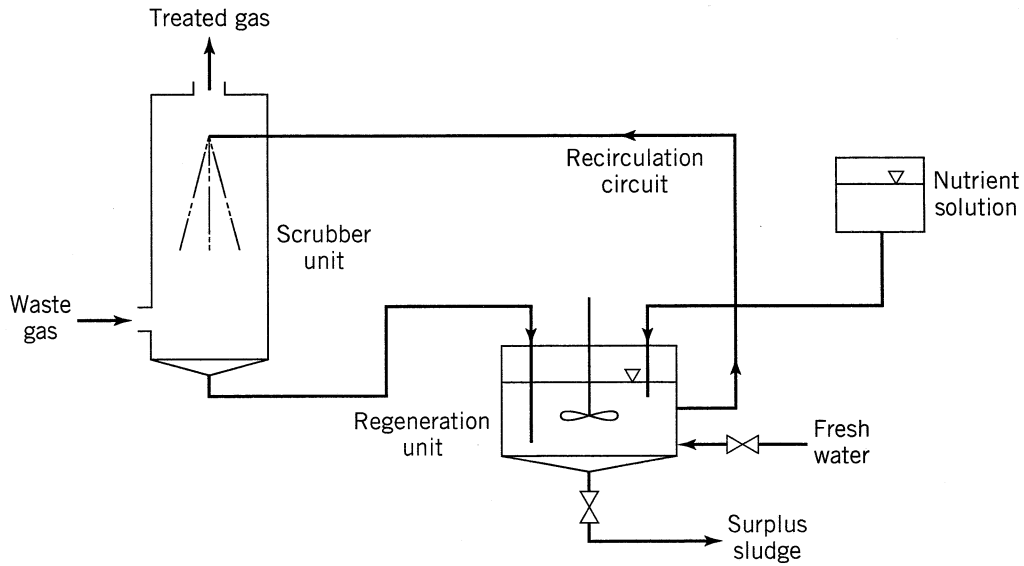ters containing mixtures of organic materials and bulking

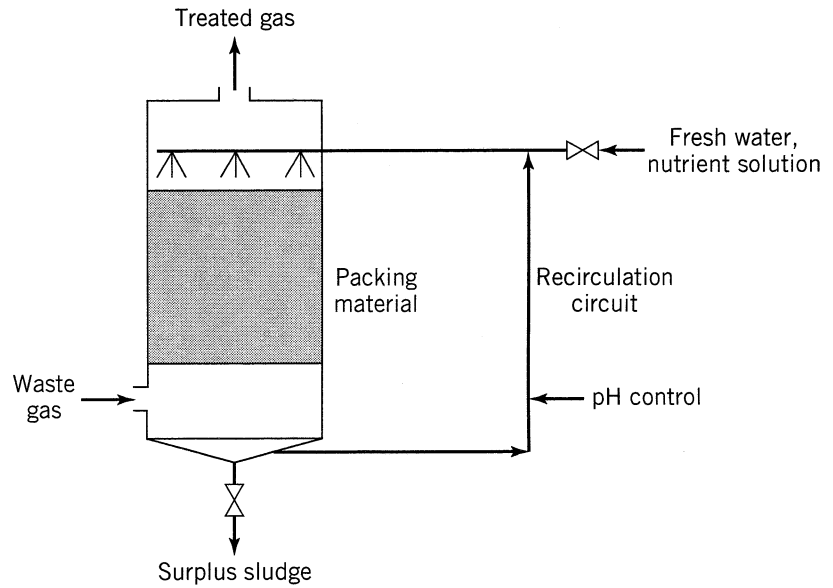**Figure 1.** Scheme of a bioscrubber system.

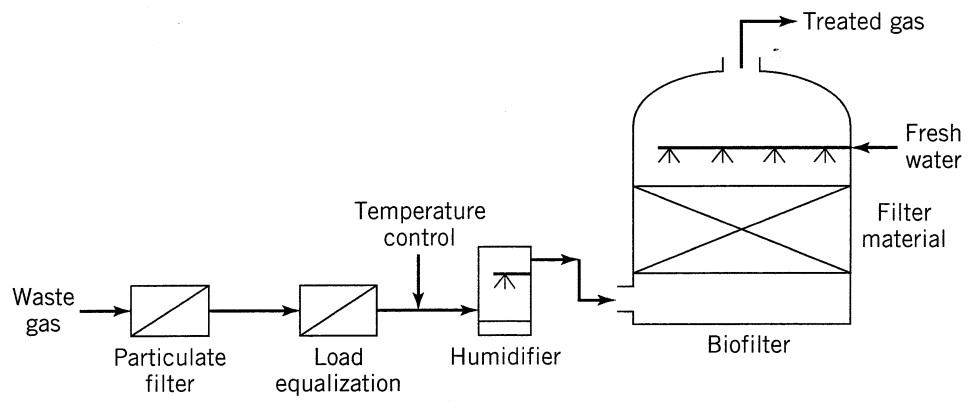**Figure 2.** Scheme of a biotrickling filter system.

Figure 3. Scheme of a closed single-layer biofilter system.

agents. Among these, simple single-layer systems, often accurately controlled and monitored, but scarcely flexible and requiring large space, can be used for VOC treatment. Multiple-layer systems, characterized by increased complexity and cost but less space requirements, consist of separately supported layers that allow optimal growth conditions for different microbial populations in each stage; a better control of the bed environment during the treatment of pollutant mixtures is also achieved.

More sophisticated and very promising configurations have been successfully tested on bench- or pilot-scale projects for the treatment of VOC mixtures, BTEX (benzene, toluene, ethyl benzene and xylene), and reduced-sulfur compounds, among which there are multistage, modular, and step-feed systems, that in general are characterized by better control and flexibility, but higher costs and operative complexity.

PROCESS PARAMETERS

Biofiltration is a very complex process that involves mass transfer and reaction and is also influenced by the fluid dynamic of the gaseous stream through the reactor. Furthermore, it is necessary to maintain an environment as close as possible to the optimal conditions for the microorganisms. Therefore, several important process parameters, such as filter material structure, moisture content, temperature, and pH, must be kept under strict control and within an optimum range.

Filter Material

The choice of filter material is of great importance in order to maintain the efficiency of a biofilter, in that it must (1) guarantee optimum environmental living conditions for the microorganisms; (2) constitute at the same time a nutritious reserve, a humidity reservoir, and a mechanical odorless support; and (3) provide the structural stability of the bed.

The choice of the material is strongly influenced by the need to minimize the overall volume necessary for the reactor, optimize the removal efficiency, keep energy consumption to a minimum, and minimize maintenance. In addition, the characteristics of the carrier material impact directly on the microbial growth and activity, thus in turn affecting biofilter performance. Biofilter beds have the advantage of immobilizing the microflora on the packing material, as a result of which these organisms, forming a biolayer, are not drained from the system, as is often the case in freely dispersed systems.

Small particles of natural organic materials, such as compost, peat, soil, or mixtures of these materials with bark, leaves, wood chips, heather branches, humus earths, or brushwood (less than 10 mm in diameter), are widely used as packing media in biofilters because they provide a high specific surface area (from 300 to 1,000 m^{-1}), favorable living conditions for the resident microbial population (ensured by high retention capacities of water and nutrients), and favorable immobilization for the microflora involved. In practice these packing materials have shown the common disadvantage of being strongly subject to aging phenomena, resulting in bed shrinkage. This phenomenon disturbs the homogeneous flow distribution of the gas and may cause a considerable increase in the pressure drop, a decrease in the specific area, and shorter filter lifetime. Moreover, the natural unhomogeneity of the structure of these natural materials prevents the uniform distribution of the gas flow, provoking short-circuiting channel development in the filter bed, with an increase in the flow resistance and a decrease in the biological degradation capacity of the bed.

In order to prevent these problems, inert materials, such as polystyrene spheres, lava particles, glass beads, porous clay, and ceramic, are usually added to the natural filling materials. This combination, improving the uniform distribution of the gas flow, lowers the head losses up to 100–150 mm of water gauge (6), thus contributing to reducing the power requirement necessary to convey the gas through the filter and, consequently, the operating costs. Moreover, the addition of porous materials (e.g., granular activated carbon) with high internal porosity and hydrophilic properties, increases the buffering capacity of the filter, (which is very favorable, particularly in the treatment of gas streams with strongly fluctuating pollutant concentrations). The selection of the most suitable material for a specific application depends on the nature of the pollutants and on the magnitude of the concentration fluctuations in the gaseous stream. In order to buffer the fluctuations in the concentration so that a constant supply of contaminants to the biofilter can be achieved, the adsorbent should have both good adsorbing power and reasonable desorbing properties. This combination provides the adsorbent with the capacity of adsorbing at high concentrations and desorbing at low concentrations and allows a significant reduction in the required filter volume (15).

Compared to soil, compost has the advantage of providing lower resistance to the gas flow and consequently contained pressure drop, which should not, in general, exceed 250 mm of water gauge (17). Peat is the material with highest water-retention capacity and constitutes the optimal substrate for the microorganisms; nevertheless, as it implies a higher pressure drop, it is often mixed with other materials in order to improve the structural stability.

Moisture Content

The good performance of a biofilter depends greatly on an optimal moisture content of the filter bed. Moisture, the most critical operational parameter, is, in fact, essential for the survival and activity of the microbial flora (which can absorb and degrade substances only in the aqueous phase) and contributes to the filter buffer capacity.

A lack of moisture causes cracking of the bed mainly at the gas inlet side, where the microbial activity is stronger due to the higher air-pollutant concentration. This results in (1) a considerable decrease of the microbial activity, (2) development of short-circuiting channels that cause bypass of the gas flow, and (3) contraction of the filter bed with consequent breakthroughs. On the contrary, an excess of water content promotes the development of anaerobic zones in the bed, due to occlusion of the pores, and provokes the formation of odorous volatile metabolic products, which are released in these zones and transported by

the gas flow. Moreover, due to the higher gas-flow resistance, these zones are bypassed by the gas flow, resulting in a decrease in the mean residence time of the gas and, hence, in the efficiency of the process, and causes a 10% increase in the power requirement (2). A moisture excess causes problems also in the oxygen transfer due to a reduction of the gas–water interface per unit biofilm volume, and in the drainage of the filter components, with production of low-pH and high-load leachate that requires disposal. The optimal moisture content of the filter bed should then range between 40 and 60% by weight, depending on the material. In order to preserve the microbial activity, it should exceed at least 40% (18).

Water consumption by a biofilter is generally low, depending on the temperature and the relative humidity of the input gas and, for open filters, on precipitation. Due to evaporation of the aqueous phase, freshwater must be continuously supplied to the system in order to ensure a degree of saturation of more than 95%, which is necessary for optimal living conditions. This is normally realized by humidifying the inlet gas stream in a water-spray scrubber upstream of the filter unit and by periodically spraying water onto the surface of the packing material by means of surface sprinklers, spray nozzles, and so on. This periodical water supply is necessary not only because it is not always possible to humidify the inlet waste gas up to the required level, but also because heat is produced by the microbial oxidation of the pollutants.

Moistening equipment must be operated in such a way that the moisture content stays within the indicated limits at any point of the medium. This operation must be realized in a very controlled way, otherwise the structure of the bed may be disturbed, which results in compaction of the filter bed in the long run and in increased gas-phase pressure drop.

pH

Maintaining an optimal pH in the filter material is a very important operational requirement for maximizing the biofiltration process because the microbial activity is strongly dependent on the pH. The effect of pH on biological systems is the indirect result of its action on the enzymatic reaction rate, which usually falls outside an optimal value. In biofiltration, as well as in most aerobic biological processes, the range of pH within which the biological systems can operate is typically between 5 and 9, with an optimum range of 6–8.

Because in biofilters the aqueous phase is stationary, problems of package acidification may arise during the degradation of pollutants such as sulfur- or nitrogen-containing compounds and halogenated hydrocarbons, which can generate acidic metabolites such as sulfuric, nitric, or hydrochloric acid. As a consequence of their accumulation, the microbial activity may decrease or even stop, seriously compromising the overall elimination capacity of the biofilter. These problems can be prevented by addition of buffering agents, such as lime, dolomite, limestone, marl, phosphates, or other water-insoluble alkaline materials, even if the neutralization products can sometimes reach inhibitory levels (13).

Besides the above adverse effect on the process, acidification of the biofilter, as well as the acidic nature of the waste gas, filter material, or leachate, can corrode the air distribution and leachate collection pipes. In such cases, the use of materials resistant to corrosion, together with a continuous pH monitoring, are suggested.

Temperature

Temperature remarkably influences the microbial growth through its effects on both bacterial enzyme activity and on various metabolic processes. Because the microorganisms generally applied in biofilters are mesophilic, temperature should be maintained between 20 and 40 °C, with an optimum of 35 °C for most aerobic microorganisms. Suboptimal temperatures can slow down the degradation capacity. The bed temperature should not rise above 45 °C, although the resident microflora can gradually adapt itself to temperatures as high as 50 °C (19). Microbial activity slows dramatically above 65 °C. The effect of low temperature on biofilters depends largely on the inlet gas temperature. In fact, although low bed temperatures, especially below 10 °C, decelerate biological oxidation, this effect can be partially counterbalanced by increased gas sorption by solids (3). However, the microbial activity will recover again upon preheating the influent exhaust gas. Fortunately, microorganisms have shown a tremendous ability to rapidly adapt themselves to drastic variations of temperature.

Reaction and diffusion rates typically increase with increasing temperature; nevertheless this effect is offset by a decrease of the solubility of the compounds to be removed, as well as a decrease in the physical adsorption capacity of the packing material at higher temperatures. Operation of the biofilter at high temperature shifts the microbial population towards thermophilic microorganisms.

In order to prevent the death of the microorganisms, it is often necessary to control the influent gas temperature by precooling through heat recovery, water injection, or wetting of the packing material, thus offering additional economic advantage and the possibility of raising the gas relative humidity. Moreover, the heat liberated by aerobic microbiological activity causes a significant increase in temperature within the filter material, which must be taken into consideration at the design phase, particularly in biofilters that treat waste gases containing high concentrations of volatile organic compounds. For good operation of the biofilter, it is obviously necessary to continuously monitor the filter-bed temperature.

PROCESS PRINCIPLES

Physical Model

In a biofilter, the microorganisms adhere to the surface of the porous solid material used as the support and grow, thus forming a biofilm. All models commonly used to describe the phenomena taking place in a biofilter under steady-state conditions are based on the principles of heterogeneous catalysis. According to the macrokinetic model

proposed by Ottengraf (20), the support particle is surrounded by a biofilm; both the pollutant and oxygen molecules dispersed in the gaseous phase continuously transfer to the active biofilm region. Due to the metabolic activity of aerobic microorganisms, a concentration gradient forms that is responsible for the continuous mass transfer of the pollutant from the gaseous to the liquid phase. At the same time, the products of the aerobic degradation (carbon dioxide, water, inorganic compounds, etc.) continuously counterdiffuse toward the gaseous phase, where they are removed by the upward flow and are finally released from the biofilter.

The metabolic activity is made possible by the continuous diffusion of inorganic nutrilites (mainly nitrogenous and phosphoric compounds) from the support to the biofilm. Nutrilite consumption is only partially counterbalanced by the mineralization of dead cells, therefore periodic addition or support renewal is necessary.

Because the small particle size and the low solubility of the organic pollutants in water make the mass-transfer resistance in the gas phase negligible, pollutant concentrations at the biolayer interface and in the gas-phase bulk can reasonably be assumed in equilibrium and related by Henry's law.

Degradation Kinetics

The degradation macrokinetics in biofilters were reviewed by Ottengraf (20). Assuming gas flow through the biofilter to be plug flow, and that degradation follows typical Monod kinetics law (21), two different situations may occur: first-order or zero-order kinetics, respectively. Several experimental works (13) demonstrated that zero-order kinetics are typically observed in biofilters for most of the volatile compounds, even at very low concentration levels (13,20,22).

Reaction Limitation. The process is reaction-limited above a critical pollutant concentration (C_{crit} in the off-gas (Fig. 4a). Due to there being no diffusion limitation, the wet biolayer is fully active (e.g., the biofilter works at its maximum elimination capacity), and pollutant elimination is limited by the reaction (Fig. 5a). Under these conditions, the degree of conversion (η) is given by the ratio between
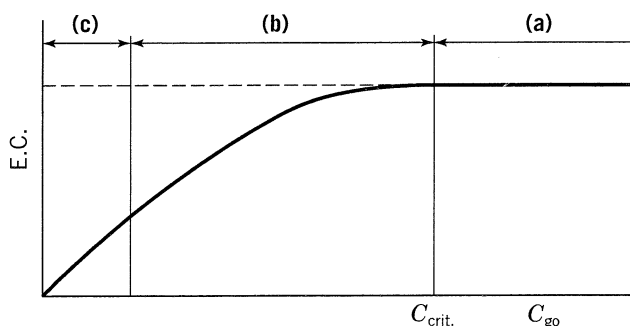

Figure 4. Dependence of the elimination capacity of a biofilter on the inlet off-gas concentration. (**a**) Reaction limitation; (**b**) Diffusion limitation; (**c**) 100% conversion.
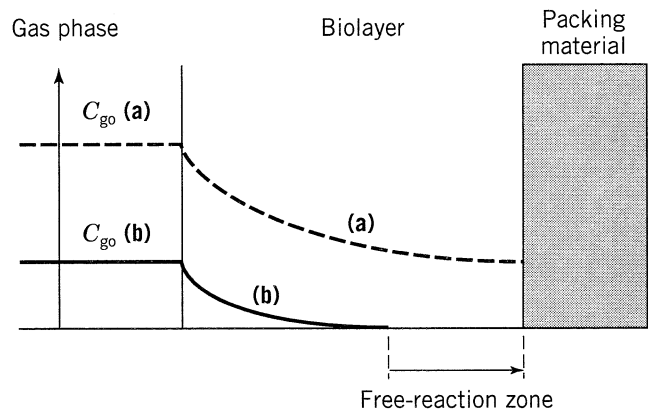

Figure 5. Biophysical substrate-penetration model.

the maximum elimination capacity, K_o which corresponds to an apparent zero-order kinetic constant, and the bed load per unit volume ($U_g C_{go}/H$) (20):

$$\eta = 1 - (C_{ge}/C_{go}) = HK_o/U_g C_{go}$$

So, at the macroscopic level, the pollutant concentration in the filter bed linearly decreases with the height, and the compound is completely consumed when $H\ K_o$ equals $U_g\ C_{go}$.

Diffusion Limitation. Below the critical concentration, diffusion limitation occurs and the rate of reaction tends to exceed that of diffusion (Fig. 4b). As a consequence, the depth of penetration becomes smaller than the layer thickness and the biolayer is not fully active (Fig. 5b). The pollutant is nearly completely consumed by the biofilm before having entirely crossed it, and the conversion rate, which decreases with decreasing pollutant concentration in the waste gas, is controlled by diffusion. Under these conditions, the elimination capacity of the biofilter is influenced by both the reaction and the diffusion rates and the compound is completely removed when $H\ K_o \geq 2\ U_g\ C_{go}$ (20).

The applicability of this model to multicomponent waste gases is limited by the theoretical complexity of the system because the various compounds are not biodegraded independently. Industrial application regarding mixture of pollutants requires pilot testing for accurate sizing of a full-scale system.

DESIGN AND MANAGEMENT CRITERIA

The main criteria for biofilter design and management are the degree of conversion or removal efficiency, the residence time, the superficial gas flow rate, the mass loading, and the elimination capacity (16).

Removal Efficiency

The removal efficiency (η), which provides information on the biofilter effectiveness, is given by the fraction of removed pollutant mass ($m_o - m_e$) with respect to total mass in the influent gas (m_o). The most common way of design-

ing and managing biofilters consists of selecting the desired removal efficiency for one or more pollutants and deriving from this all the other criteria. The best removal efficiencies (95–99%) are usually observed for aromatic compounds, such as benzene, toluene, and benzoic acid (23).

Residence Time

The residence time of a gaseous pollutant in a biofilter (τ), given by the ratio of the empty bed volume of the filter (V_e) to the waste-gas flow rate (Q_g), is a fundamental parameter that cannot assume values smaller than a critical level if satisfactory removal efficiencies are desired. However, due to the difficulty in estimating empty-bed volume, residence time is often referred to total-bed volume. According to the pollutant degradability, this parameter typically ranges between 15 and 60 s. For liquid pollutants it is necessary to take into account their partition coefficients between liquid and gaseous phases.

Superficial Gas Flow Rate

The superficial gas-flow rate (U_g), defined as the ratio of the gas flow rate (Q_g) to the total cross-sectional area (A), is a parameter increasing with flow rate and decreasing with residence time; it usually ranges, depending on pollutant type, from 50 to 200 $m^3 m^{-2} h^{-1}$. Since a superficial gas-flow rate increase obviously implies a reduction in removal efficiency, a maximum threshold of about 200 $m^3 m^{-2} h^{-1}$ should not be exceeded (24).

Mass Loading

The mass loading (L_v) is defined as the ratio of the gas-flow rate (Q_g) to the bed volume (V) multiplied by the inlet pollutant concentration in the off gas (C_{ge}); for VOCs it should not exceed values of 3,000 ÷ 5,000 $mg\ m^{-3}$. In a biofilter, load increase could be ensured by either increasing waste-gas concentration or decreasing residence time. Although increasing gas concentration should, in general, accelerate biodegradation, excess concentrations above a certain threshold can inhibit microbial activity, so that decreasing residence time should be preferred. In addition, since the removal efficiency is affected by an increase in mass loading, and because excess organic load may obstruct the biofilter with the possible formation of toxic and/or acidic intermediates (25), a value between 10 and 160 $g\ m^{-3}\ h^{-1}$ should be employed to achieve satisfactory removal yields.

Elimination Capacity

The elimination capacity (EC) is defined as the ratio of gas flow rate (Q_g) to the bed volume (V), multiplied by the difference between the pollutant concentration in the influent and the effluent waste gases ($C_{go} - C_{go}$), it indicates a measure of the biofilter ability in removing the pollutants. From this definition it is evident that, because removal efficiency is expected to be close to 100%, the elimination capacity also ranges between 10 and 160 $g\ m^{-3}\ h^{-1}$ (1). In addition, this parameter also is higher close to the inlet side, as happens for mass loading. The elimination capacity increases with mass loading and inlet pollutant concentration and decreases with residence time, the limiting value depending not only on the pollutant degradability, but also on the support material and the operating conditions. Beyond this value it is nearly constant (20).

APPLICATIONS OF BIOFILTERS

Bench-Scale Results

Although many experimental works on biofiltration, both at laboratory and full-scale size, have been published during the last two decades, a critical comparison of results is difficult due to various reasons, among which are the use of German and Dutch as the most common languages in this research field, the use of different criteria to express the performances, the utilization of different volatile organic and inorganic compounds, the different types of biofilters used, and so on. Nevertheless, the work of Ottengraf (20) can be taken as a useful reference basis to provide a complete overview on biofilter performance, as well as on the various effects influencing the process macrokinetics.

For most organic pollutants, the steady-state continuous elimination follows zero-order kinetics, so that pollutant concentration decreases with the column height. In the presence of mixtures of pollutants characterized by different biodegradability, a nearly complete removal of the most easily degradable compounds takes place at the bottom of the filter, whereas the most recalcitrant are mainly removed at the top. This suggests that different microbial populations could be responsible in the filter bed for the degradation of different organic components.

The path of Figure 4 explains another typical behavior of biofilter performances besides the one illustrated earlier. At low organic loads (corresponding to low concentrations for a given residence time), the elimination capacity linearly increases with this parameter, which means that nearly complete pollutant conversion is achieved. Over a critical load, the elimination capacity reaches a maximum value (corresponding to the apparent zero-order kinetic rate constant, K_o) and stays nearly constant. Maximum thresholds of 21, 27, 32, and 32 $g\ m^{-3}\ h^{-1}$, calculated for a mixture of toluene, ethyl acetate, butyl acetate, and butanol in a five-stage system simulating a biofiltration column with variable height (20), seem to reflect a direct dependence of this parameter on the pollutant degradability.

Pilot-Scale and Real-Scale Applications

Biofilters of different sizes (areas ranging from 10 to 2,000 m^2) are currently used in many industrial sectors, mainly in Europe, to remove a variety of mixtures of volatile organic and inorganic compounds from off-gases characterized by variable flow rates (1,000–150,000 $m^3 h^{-1}$) and low pollutant concentrations (usually less than 1,000 ppm). As one can see from Table 1, where the main pollutants removable by biofiltration are listed (3,13,16,26,27), a relatively variable degradability characterizes not only the different classes of pollutants, but also the various pollutants within the same class. While for easily biodegradable substances the microorganisms suited for their degradation are usually present in the natural packing materials, in-

Table 1. Classification of the Main Pollutants that can be Removed by Biofiltration, according to their Biodegradability

Excellent	Good	Minimum	No	Uncertain
Aliphatics	*Aliphatics*	*Alicyclics*	*Halogenated*	*Aliphatics*
Butadiene	Hexane	Cyclohexane	1,1,1-Trichloroethane	Acetylene
Aromatics	*Aromatics*	*Aliphatics*		*Nitrogenous*
Ethyl benzene, xylene Cresols	Benzene, styrene, toluene	Methane, pentane		Isocyanates
	Phenols	*Aromatics*		*Oxygenated*
Nitrogenous	*Halogenated*	PAH		Methyl methacrylate
Trimethylamine	Chlorophenols	*Halogenated*		
Oxygenated	*Nitrogenous*	Carbon tetrachloride		*Sulfuric*
Alcohols: butanol, ethanol, methanol	Amides	Dicloroethane		Isothiocyanates
Aldehydes: acetaldehyde, formaldehyde	Heterocycles: Pyridine	Dichloromethane		
Esters: Ethyl acetate	Isonitriles	Pentachlorophenol		
Ethers: Tetrahydrofurane	Nitriles: Acetonitrile	Perchloroethylene		
Ketones: Acetone		Trichloroethane		
Organic acids: Butyric acid		Trichloroethylene		
Sulfuric	*Oxygenated*	*Nitrogenous*		
Methyl mercaptan	Methylisobutylketone	Nitrocompounds		
Inorganics	*Sulfuric*	*Oxygenated*		
HCl, HF, H$_2$S, NH$_3$, NO$_x$ (except NO$_2$), PH$_3$, SiH$_4$, SO$_2$	Heterocycles: Thiophene Thiocyanates Thioethers: Dimethylsulphide	Dioxane		
		Sulfuric		
		Carbon disulfide		

Source: From Refs. 3, 13, 16, 26, and 27.

oculation and enrichment with cultivated organisms is often necessary for the slowly biodegradable ones.

On the other hand, Table 2 provides a picture of the most significant industrial sectors where biofiltration is applied with success, with indication of the related pollutants (26).

Finally, from Table 3, where the results of the principal pilot and full-scale applications of biofilters are summarized, it is evident that conversion yields often higher than 90% are not uncommon in industrial practice (16,17,20,27–32).

MICROORGANISMS

Another factor strongly influencing biofiltration performance is the composition of the heterogeneous microflora, which primarily depends on the composition of the polluted gaseous stream. As a consequence of the progressive adaptation of microorganisms to the organic pollutants contained in the off-gas, which typically takes about 10 days (20), the population distribution shifts towards strains that naturally catabolize these pollutants.

Microbial Populations of Biofilters

Microbial population distribution along biofilters has recently been reviewed by Benedusi et al. (33). The purification of waste air containing a limited number of pollutants is effectively achieved by a population restricted to a few microbial species and generally requires, to shorten the start-up operation time, inoculation of the biofilter with pure cultures or activated sludges previously adapted to each pollutant. This practice is particularly suited to the degradation of very recalcitrant or complex organic pollutants, such as halogenated and aromatic hydrocarbons, which often require cometabolism with more easily degradable substances to accelerate the process (34,35). On the other hand, gaseous emissions simultaneously polluted by several compounds, such as those coming from water works, compost plants, and so on, require as inoculum a more heterogeneous population possessing a wider variety of metabolic pathways, which is available in the microorganisms naturally present in activated sludges and compost. Microbial loads of about 10^5, 10^4, and 10^3 colony-forming units per gram of support (CFU/g) have been reported for oligoheterotrophic, copioheterotrophic, and autotrophic metabolic types, respectively in heather-peat biofilters treating septic-tank emissions (33). Although Bohn (3) demonstrated that enriching compost-based biofilters with microorganisms is not advantageous in terms of removal yields, probably because of the unfavorable competition with the indigenous population, the addition of activated sludges to medium devoid of microorganisms

Table 2. Typical Pollutants Removed by Biofiltration, in Addition to Odor Control, in Different Industrial Applications

Type of Industry	Organics						Inorganics	
	Aliphatic	Aromatic	Oxygenated compounds	S-containing compounds	N-containing compounds	Halogenated compounds	H_2S	NH_3
Food and Agriculture Industry								
Aroma extraction		X						
Yeast drying			X		X			
Blood and fish-meal production			X	X	X		X	X
Bristle and feather drying			X	X	X			X
Fish roasting			X	X	X		X	
Fat processing			X	X	X		X	X
Gelatin production			X	X				
Coffee and cacao roasting		X	X	X	X			
Bone processing			X	X	X		X	X
Meat and fish curing	X	X	X		X			X
Animal feed production			X	X	X		X	X
Slaughterhouses			X	X	X		X	X
Tobacco processing		X	X		X			X
Livestock farming			X	X	X		X	X
Chemical Industry								
Foundries	X	X	X					X
Plastics processing	X	X	X		X			
Lacquer production and processing	X	X	X					
Adhesive production			X	X				
Oil and fat production	X	X		X		X		
Polyester manufacture				X	X			
Friction-linings production			X	X	X			X
Rendering plants			X	X	X	X	X	X
Glue production		X	X					
Treatment Plants								
Used oil processing	X	X		X		X		
Dump gas removal			X	X			X	
Sewage treatment and sludge drying	X	X	X	X	X	X	X	X
Composting and waste processing	X		X	X	X		X	X
Manure drying	X		X	X	X		X	X

Source: Elaboration from Ref. 26.

is a common practice (1). Finally, the growth and activity of the microorganisms are strongly influenced by the availability of oxygen and nutrients, the eventual presence of toxic substances in the exit gas, the degree of moisture, temperature, pH, and so on.

A great advantage of the heterogeneous population present in the biofilter is the excellent ability to survive long periods (up to two months) without activity, provided that periodic aeration of the bed is ensured, even if microbial activity seems to be hardly affected (36).

As revealed by plate counting in nutrient agar, most of the microorganisms growing in biofilters treating organic pollutants are heterotrophic eubacteria, actinomycetes, and fungi (1,37), which utilize the organic compounds contained in the gaseous streams as carbon and energy sources. Even when the presence in the off-gas of inorganic substances, like thiosulfate or hydrogen sulfide, should favor the growth of chemio-litoautotrophic microorganisms, such as *Thiobacillus* sp., by utilizing these substances as the energy source and carbon dioxide as the carbon source, the heterotrophic population becomes prevalent and is probably responsible for most of the degradation activity (33,38,39).

The bacteria more frequently detected in biofilters are soil bacteria, such as *Bacillus cereus* var. *mycoides* and several strains and species of actinomycetes belonging to the genus *Streptomyces*. Helmer (40), who did a microbiological characterization of compost-filter population, observed abundant growth of eubacteria (10^9 CFU/g), actinomycetes (2×10^6 CFU/g), and fungi (10^6 CFU/g) along the whole filter height, with higher density of fungi at the bottom. The highest bacterial densities are usually detectable near the inlet side, where substrates are more concentrated and more easily degradable, while different strains well-adapted to low substrate levels, as well as to more recalcitrant compounds, can be found at higher depth (37). The growth of several bacteria and fungi, whose cell densities decreased according to the sequence: *Actinomyces*

Table 3. Management Data of Representative Pilot and Industrial-Scale Applications of Biofilters

Applications	U_g (m h⁻¹)	Q_g (m³/h)	Pollutants	C_{go} (mg/m³)	τ (s)	N of stages	η (%)	Reference
Food industry								
Gelatin production		35,000	Odor		12–21	0.6–1	70–93	27
Cocoa and chocolate processing		10,000	Odor		22	2	99	27
	60		Odor	$4 \cdot 10^7$ o.u.		1	90	28
Fishmeal factory		40,000	Odor	230 (C)	20	1	50–90	27
Flavor and fragrance industry		25,400	Odor	10^5 o.u.	22	2	98	27
	550		Odor	$5 \cdot 10^4$ o.u.		3	98	28
Food processing industry		9,000	Odor (oil)	10^5 o.u.	20	2	93	27
	400		Odor	$2 \cdot 10^5$ o.u.		2	95	28
Meat rendering		80,000–120,000	Odor	$2 \cdot 10^4$ o.u.	60–90	1	99	16
Commercial bakery		90–200	VOC		25–60	2		
			Ethanol	1,000–4,000			80–100	
			Methane	700			10	16
			Aliphatics	200			10–30	
			Ethyl acetate				80–90	
Coffee industry	500		Odor	10^8 o.u.		2	90	28
Yeast production	100		Odor	7,500 o.u.		2	99	28
			Ethanol	500			99	
Petrochemical industry								
Gasoline tank		222	BTEX	3–70		2	75–90	16
Oil storage	100		H₂S	240		2	99	28
Tank cleaning			Benzene	30			50	
	100		Acrylonitrile	16–300		2	50–98	28
			Alcohols	60			80	
Petroleum processing			BTEX				>95	
		8.5–70	C2–C5		45–360	1	≤50	17
			Methane				≤5	
			>C5				40–90	
Other industries								
Tobacco industry		30,000	Odor		14	2	95	
			NH₃	1.5				27
			Nicotine	3.5				
Paint production		11,700	VOC	1,800	38	2	90	27
Pharmaceutical plant		75,000	VOC		108	3	80	27
Photo film production		140,000	VOC	400	30	2	75	27
Ceramics production		30,000	Ethanol		8	1	98	27
Metal foundry		40,000	Benzene	9	30	1	80	27
Industrial emission		17,000	Ethanol	1,000–4,000	60	2	80–90	16
Resin synthesis	65		Organics	25,000		2	40	28
			odor	10^6 o.u.			45	
Chemical industry		0.7–0.8	VOC	2,000–10,000		1	82	29
Printing industry	200		Esters, alcohols	1200		2	90	28
Treatment plants								
Wastewater treatment		10,000	Odor		29	2	90–95	
			H₂S	10				27
			Acetone	8				
		30,000	Odor			2		
			H₂S	4.7			96	30
			NH₃	0.15			45–60	
	200		H₂S	300		1	80	28
Petrochemical sewage treatment	100		Odor	$1.3\text{–}10^5$ o.u.		2	75	28
Composting		154,000–240,000	Odor		30–47	4	>90	31
		16,000–22,000	Odor	45–230 (C)	40–60	1	93–96	20
	100		Odor	$1.3 \cdot 10^4$ o.u.		2	95	28
Handicraft activities								
Wood paint and varnish	200		VOC	250		1	50	28
Metal paint and varnish	70		H₂S	4000 o.u.		2	90	28
Paint spraying			Styrene	60–600	15–60	1	95	32

globisporus, Penicillium sp., *Cephalosporium* sp., *Mucor* sp., *Micromonospora albus, Micrococcus albus,* and *Ovularia* sp., was also reported for similar supports used for mixtures of halogenated and aromatic hydrocarbons (9). An opposite bacteria distribution has been reported by Benedusi et al. (33) along a pilot heather-peat filter used to remove hydrogen sulfide and ethyl mercaptan from septic-tank emissions. They observed maximum bacterial densities just near the biofilter outlet side, with oligo-heterotrophic population (2.4×10^8 CFU/g) predominating over all the others: copiochemioheterotrophs (5.7×10^6 CFU/g), sulfur-oxidizing chemioautotrophs (2.6×10^5 CFU/g), and nitrifying chemioautotrophs (7.2×10^5 CFU/g). Bacterial densities at biofilter bottom vary between 1% and 3% of these values, probably due to the hard pH decrease (up to 2.5–3.5) consequent to hydrogen sulfide oxidation (to H_2SO_4), as well as to the inhibiting effect of high concentrations of this pollutant on microbial activity. Other bacteria detected in biofilters are *Micromonospora vulgaris* and *Proteus vulgaris* (41).

Among the main soil fungal taxa able to degrade polysaccharides naturally present as symbionts or saprophites in heather or peat (*Geomyces pannorum* var. *pannorum, Penicillium citrinum, Oidiodendron griseum, Penicillium frequentans, Pestalotiopsis guepinii,* and *Trichoderma koningii*), only *G. pannorum* var. *pannorum* and *T. koningii* seem to be able to develop in the bed under regular and irregular moisture conditions, respectively (33). Other taxa, initially absent in the support material but normally living in polluted waters and soil, appear as a consequence of the air flowing and progressive acclimation, under conditions either of irregular (*Mucor* sp.) or regular moisture (*Mortierella bainieri, Drechslera australiensis,* and *Trichotecium roseum*). Other fungi detected in filter beds are *Circinella* sp., *Cephalotecium* sp., and *Stemphilium* sp. (41). Although the contribution of fungi in biofiltration is not so clear, it is likely that an important role is played in the degradation of organic nitrogen, sulfur, and potassium, such as takes place in the soil.

As far as the influence of the support structure on microbial density inside the bed is concerned, it was demonstrated that compost-based filters usually contain much higher population densities of these microorganisms than does soil (43). On the other hand, it seems that concentrations of bacteria cells and fungi spores in the treated emissions are only slightly higher than those observed in open air (43,44). Nevertheless, Liebe and Herbig advised against the risk of relevant fungi-spore emissions during biofilter maintenance and support-replacement operations (45).

A great number of other bacterial strains have been inoculated and tested in bench-scale biofilters to treat waste gases polluted by specific compounds, most of which appear to be particularly promising for future development of this technology (see "New Developments and Potentials").
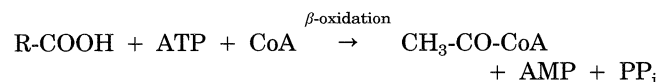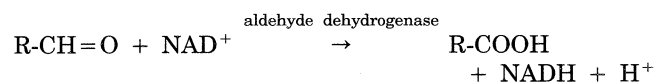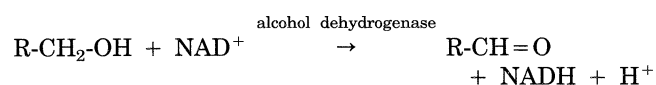
Metabolic Pathways for Organic Pollutant Degradation

Different metabolic pathways are employed by microorganisms to degrade a variety of organic pollutants.

Aliphatic Hydrocarbons. Several bacteria (mainly belonging to the genera *Nocardia, Pseudomonas,* and *Mycobacterium*), and some yeast and fungi, which are capable of utilizing saturated aliphatic hydrocarbons for their growth, are the main candidates for new applications in bioscrubbers and biofilters for the removal of such air pollutants. The oxidation of the terminal carbon to an alcohol, catalyzed by a monoxygenase, is responsible for the introduction in the hydrocarbon structure of only one of the atoms of molecular oxygen, which acts as a cosubstrate, while the other is reduced to water:

$$R\text{-}CH_3 + NADH + H^+ + O{:}O \xrightarrow{\text{monoxygenase}} R\text{-}CH_2\text{-}OH + NAD^+ + H_2O$$
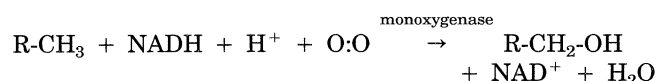
Further oxidations by NAD^+-dependent alcohol and aldehyde dehydrogenases lead to a carboxylic acid that can be decomposed to acetyl-CoA by the β-oxidation route (46):

$$R\text{-}CH_2\text{-}OH + NAD^+ \xrightarrow{\text{alcohol dehydrogenase}} R\text{-}CH{=}O + NADH + H^+$$

$$R\text{-}CH{=}O + NAD^+ \xrightarrow{\text{aldehyde dehydrogenase}} R\text{-}COOH + NADH + H^+$$

$$R\text{-}COOH + ATP + CoA \xrightarrow{\beta\text{-oxidation}} CH_3\text{-}CO\text{-}CoA + AMP + PP_i$$

On the other hand, polychlorinated hydrocarbons can be cometabolized in the presence of isoprene-degrading bacteria (47).

Aromatic Compounds. The aerobic degradation of aromatics is only possible if the aromatic ring undergoes an enzymatic cleavage. Benzene can be degraded by several microorganisms by two divergent pathways after preliminary dioxygenation to catechol: the so-called *ortho*-pathway, utilizing the intradiol cleavage catalyzed by the catechol-1,2-dioxygenase, and the so-called *meta*-pathway, following an extradiol cleavage. Mixtures of benzene, toluene, and xylene are preliminarily dioxygenated, decomposed mainly via the *meta*-pathway, and finally metabolized through the citric acid cycle (47).

The presence of an alkyl group in the benzene ring gives the microbe the opportunity to attack the compounds either on the side chain or the ring. In arenes with short side chains (up to seven carbons), both alternative routes are possible, mainly due to the widespread presence of transformable plasmids. In particular, the plasmid TOL (48), containing catabolic operons for both alkyl-group oxidation and ring cleavage via *meta*-pathway, gives several strains of *Pseudomonas* the ability to degrade toluene, *m*- and *p*-xylenes, and other monoalkylbenzenes (49,50). On the contrary, if the side chain is sufficiently long (more than seven carbons), its metabolization by initial side-chain attack via ω- and β-oxidation (51), provides the cells of different species with enough energy for growth, and then the ring cleavage is not necessary.

Of the dialkylbenzenes, *m*- and *p*-xylenes can be biode-

graded by certain strains of *Pseudomonas* containing the TOL plasmid by preliminary oxidation of one of the methyl groups to a carboxylic group and methylcatechol (52), followed by *meta*-cleavage. Members of *Nocardia* are able to cometabolize *m*- and *p*-xylenes via the *ortho*-pathway, and *o*-xylene via the *meta*-pathway in the presence of alkanes as carbon sources (53). More recently, it has also been demonstrated that *o*-xylene can be metabolized as the sole carbon and energy source, via 3,4-dimethylcatechol and subsequent *meta*-cleavage, by some strains of *Pseudomonas stutzeri* (54) and *Corynebacterium* (55).

There are reports of different bacteria (*Pseudomonas* sp., *Xanthobacter* sp., etc.) able to grow on styrene (56) and methylstyrenes (57) as sole carbon and energy sources, during which oxidation of the aromatic nucleus and subsequent *meta*-cleavage probably take place.

In several bacteria, after a preliminary monooxygenation to a catecholic structure, phenol is degraded (58) through the *meta*-pathway to directly give intermediates of the citric acid cycle (46). While for 4-chlorophenol degradation the chlorinated lactonic intermediate is directly dechlorinated before its introduction into the cycle, for 4-methylphenol a specific enzyme catalyzes the isomerization of the related intermediate into a more easily metabolizable lactone (47).

Biphenyl and Fused-Ring Hydrocarbons. Although the biodegradation of biphenyl, naphthalene, and polycyclic aromatic hydrocarbons (PAH) is a subject of less relevance for biofiltration applications given their solid state at ambient temperature, their high toxicity has brought lively interest in the actual possibility of purifying low contaminated off-gases by utilizing biofilters inoculated with liquid suspensions of strains specifically adapted.

Apart some subtle differences, biphenyl is biodegraded through a pathway common to many different bacteria belonging to the genera *Pseudomonas* (59) and *Nocardia* (60), including a preliminary dioxygenation of the ring in the C2 and C3 positions to give a catechol-type intermediate, which is subsequently *meta*-cleaved and catabolyzed via 2-oxopenta-4-enoate and benzoate (60).

Naphthalene is degraded mainly by members of the genus *Pseudomonas* via dioxygenation to 1,2-dihydroxynaphthalene, followed by extradiol cleavage of the ring between C1 and C9, whereas PAHS, which are potential carcinogens, mutagens, and tetragens, are dioxygenated and cleaved in two different ways, according to their phenanthrenic or anthracenic structures (53). Less detail is available for higher molecular weight PAHs.

ECONOMIC CONSIDERATIONS

Comparing cost data quoted in the literature for biofiltration systems is very difficult because of the use of different currencies, the fluctuations of their reciprocal rates, the cost variability with time, the different units employed, and so on. Nevertheless, there is no doubt of biofilter economic convenience when compared with other traditional abatement techniques, due to the following advantages (6): minimum operational input, reduced production and disposal of by-products, easy start-up procedures, operational

stability at steady state, low operative temperature, low cost of materials, simple building technologies, simple control systems, and low power requirements.

From the comparison of different biofilters in use in Europe (1,37,61–63), total operating and maintenance costs ranging from \$0.2 to 0.5 per 1,000 m^3 of waste gas can be estimated, which includes depreciation and interest, water consumption, replacement of filter materials and personnel, wastewater treatment, and energy costs for compressors. Costs reductions showing a half of these amounts for United States installations do not include expenses for support replacement and reflect the lower cost of electricity (1).

Capital costs are often negatively influenced by several factors, among which high pollutant concentration in the waste gas, recalcitrant nature of the components, necessity of gas pretreatment, reduced waste-gas availability, lack of space near the polluting source, and transport. Since the transport costs are strongly dependent on the country and the nature of the site, a cost comparison is only possible if this factor is not considered. On these bases, capital costs for open single-bed filters are in the range between \$300 and \$1,000 per m^2 of filter material in Europe (64,65) and between \$600 and \$1,000 per m^2 in the United States (1). These costs would increase up to \$3,800–5,700 per m^2 if lack of space would force the build-up of the filter on the plant cover, because of the additional costs of personnel, piping, and compression (66), and up to \$1,000–5,000 per m^2 if particular meteorological situations (snow, frequent atmospheric precipitation, etc.) require an upper covering or enclosed systems (1). Multiple-bed systems seem to be twice as expensive as the single-bed ones.

The filling material may consist, according to the pollutants, of heather, peat, and earth, often mixed with other inert materials necessary to minimize head losses. Because of these variables and the market situation, costs may fluctuate remarkably from \$60 to \$700 per m^3 (65,66).

Taking odor abatement as a basis for comparison of different techniques, biofiltration is by far the most convenient process in terms of space availability, resulting in, as a function of waste-gas composition, total cost savings of 15–30%, 45–70%, 50–75%, and 80% with respect to bioscrubbing, chemical scrubbing, adsorption, and ozonation, respectively (17,65,67). The success of biofiltration appears to be the result of the very low incidence of running costs to total costs as compared to other techniques. Extending the above comparison also to incineration, it has been estimated that biofiltration allows total cost savings of 60–75%, and investment and operational cost savings of 40–75% and 70–80%, respectively (17,63).

NEW DEVELOPMENTS AND POTENTIALS

Pollutants' degradability often depends on their origin and source: although xenobiotics, which are characterized by unnatural structures, cannot be degraded (persistent compounds), or can be degraded only with difficulty (recalcitrant compounds), the biogenic ones are easily biodegradable. For this reason, much research effort is being spent on the isolation, selection, and construction of different mi-

croorganisms (27,69,70) able to grow on difficult carbon sources in pure cultures, at rates comparable with those usually encountered on common substrates (Table 4).

For these reasons, it is evident that future developments and new applications of this technology strongly depend on the actual possibility of utilizing these microbial strains in pure cultures or selected consortia under the peculiar conditions, as present in biofilters.

Table 4. Xenobiotic Compounds and Microorganisms Able to Degrade Them in Pure Cultures

Pollutant	Microorganism
Chlorinated hydrocarbons (27,68)	
Methyl chloride	*Hyphomicrobium* sp.
Dichloromethane	*Pseudomonas* DM1
	Methylobacter DM111
	Hyphomicrobium sp.
1,2-Dichloroethane	*Xanthobacter* GJ10
Vinylchloride	*Mycobacterium* L1
Epichlorohydrine	*Pseudomonas* AD1
Chlorobenzene	*Pseudomonas* WR 1306
1,2-Dichlorobenzene	*Pseudomonas* GJ 60
1,3-Dichlorobenzene	*Pseudomonas* sp.
1,4-Dichlorobenzene	*Alcaligenes* A175
Arenes (69)	
Benzene	*Pseudomonas* sp.
	Acinetobacter calcoaceticus RJE 74
Toluene	*Pseudomonas* sp.
	Pseudomonas putida
	Bacillus sp.
Monoalkylbenzenes	*Pseudomonas* sp. NCIB 10643
	Pseudomonas fluorescens
	Pseudomonas putida RE204
	Acinetobacter Iwoffi
m-,p-Xylene	*Pseudomonas* sp.
	Nocardia sp.
o-Xylene	*Pseudomonas stutzeri*
	Corynebacterium sp. C125
Styrene	*Xanthobacter* sp 124X
	Pseudomonas putida
α-Methylstyrene	*Pseudomonas* sp.
Biphenyl (69)	
	Pseudomonas sp. NCIB 10643
	Nocardia sp. NCIB 10503
	Pseudomonas cruciviae
	Pseudomonas putida
	Pseudomonas pseudoalcaligenes
	Pseudomonas aeruginosa
	Escherichia coli
(Fused-ring compounds) (69)	
Naphthalene	*Pseudomonas* sp.
	Pseudomonas fluorescens
	Pseudomonas putida
PAH	*Pseudomonas* sp.
	Pseudomonas paucimobilis
	Pseudomonas vesicularis
	Algaligenes denitrificans

CONCLUDING REMARKS

As pointed out here, biofiltration has become by now a well-recognized technology for the control of odors and the elimination of air contaminants from waste gases, and represents a new generation of air pollution control technologies.

Over the past two decades, thanks to the considerable progress made in the microbiological and process-engineering fields, biofiltration has gained the confidence of many of industries and has found increasingly widespread application. The suitability of this technique for the treatment of a wide range of inorganic and organic pollutants has been proven. Removal efficiencies as high as 90% or more are usually achieved for common pollutants, such as alcohols, ethers, aldehydes, ketones, and monoaromatic volatile chemicals.

The operating cost savings, the technological simplicity of the plants, the reduced maintenance and energy requirements, and the minimal generation of by-products that must be disposed of, make biofiltration, with respect to other air pollution control technologies, an attractive cost-effective alternative, particularly if applied to large volumes of gaseous-waste streams containing readily biodegradable contaminants in relatively low concentrations.

Although its application to different compounds has been demonstrated, considerable research is being undertaken to find appropriate strains able to degrade recalcitrant compounds, in particular, most of which are xenobiotics. In addition, new strategies should be developed to improve the control of essential operating parameters affecting biofilter performance and to optimize reactor design and performance, in order to widen the application field of biofilters.

BIBLIOGRAPHY

1. G. Leson and A.M. Winer, *J. Air Waste Manage. Assoc.* **41**, 1045–1054 (1991).
2. H.L. Bohn, *J. Air Pollut. Control. Assoc.* **25**, 953–955 (1975).
3. H.L. Bohn, *Chem. Eng. Progr.* **88**, 34–40 (1992).
4. W.H. Prokop and H.L. Bohn, *J. Air Pollut. Control. Assoc.* **35**, 1332–1338 (1985).
5. D.H. Kampbell, J.T. Wilson, H.W. Read, and T.T. Stocksdale, *J. Air Pollut. Control. Assoc.* **37**, 1236–1240 (1987).
6. F. Alfani, L. Cantarella, A. Gallifuoco, and M. Cantarella, *Acqua-Aria* **10**, 877–884 (1990).
7. B. Jäger and J. Jager, *Müll und Abfall* **2**, 48–54 (1978).
8. D.K.B. Thistlethwayte, B. Hardwick, and E.E. Goleb, *Chimie Ind.* **106**, 795–801 (1973).
9. M. Gust, H. Grochowski, and S. Schirz, *Staub-Reinh. Luft* **39**, 397–402 (1979).
10. R. Helmer, *Ges. Ing.* **94**, 21–30, (1974).
11. H. Hartmann, *Stuttg. Ber. Siedlungswasserwirtsch* **59**, 3–19 (1977).
12. S. Cernuschi and V. Torretta, *Ingegneria Ambientale* **25**, 248–264 (1996).
13. S.P.P. Ottengraf, J.J.P. Meesters, A.H.C. van den Oever, and H.R. Rozema. *Bioprocess Eng.* **1**, 61–69 (1986).
14. S.P.P. Ottengraf, A.H.C. van den Oever, and F.J.C.M. Kempenaars, in E.H. Houwink and R.R. van der Meer eds., *Inno-*

vations in Biotechnology, Elsevier, Amsterdam, 1984, pp. 157–167.

15. F.J. Weber and S. Hartmans, in A.J. Dragt and J. van Ham eds., *Biotechniques for Air Pollution Abatement and Odour Control Policies*, Elsevier, Amsterdam, 1992, pp. 125–130.

16. W.J. Swanson and R.C. Loeher, *J. Environ. Eng.* **123**, 538–546 (1997).

17. G. Leson and B.J. Smith, *J. Environ. Eng.* **123**, 556–563 (1997).

18. E. Eitner, in Verein Deutscher Ingenieure ed., *VDI Berichte 735: Biologische Abgasreinigung*, VDI Verlag, Düsseldorf, 1989, pp. 191–214.

19. H.L. Bohn, *86th Mtg. Air Waste Management Assoc.*, Denver, Col., June 14–18, 1993.

20. S.P.P. Ottengraf, in H.J. Rehm and G. Reed eds., *Biotechnology*, vol. 8, VCH, Weinheim, 1986, pp. 425–452.

21. J. Monod, *Ann. Rev. Microbiol.* **3**, 371–394 (1949).

22. S.P.P. Ottengraf and A. van den Oever, *Biotechnol. Bioeng.* **19**, 1411–1417 (1986).

23. R. Marsh, *Inst. Chem. Eng.* **3**, 13.1–13.14 (1992).

24. F. Sabo, U. Motz, and K. Fischer, *86th Mtg. Air Waste Management Assoc.*, Denver, Col., June 14–18, 1993.

25. J.S. Devinny and D.S. Hodge, *J. Air Waste Manage. Assoc.* **45**, 125–131 (1995).

26. Kommission Reinhaltung der Luft, in Verein Deutscher Ingenieure ed., *VDI 3477-Handbuch Reinhaltung der Luft: Biological Waste Gas / Waste Air Purification, Biofilters*, VDI Verlag, Düsseldorf, 1991, Band 6, pp. 1–29.

27. S.P.P. Ottengraf and R.M.M. Diks, in A.J. Dragt and J. van Ham eds., *Biotechniques for Air Pollution Abatement and Odour Control Policies*, Elsevier, Amsterdam, 1992, pp. 17–31.

28. A.J. Dragt and S.P.P. Ottengraf, *Proc. Conf. on Advanced Technologies and Biotechnology for the Pollution Prevention and Environmental Protection*, SIBA-ISER, Genoa, Italy, 1987, pp. 243–258.

29. M.A. Deshusses, *J. Environ. Eng.* **123**, 563–568 (1997).

30. M. Pescari and V. Pizzirani, *Microsymposium on Air Pollution Control*, Pavia, Italy, May 23, 1997.

31. I. Trentini and R. Vismara, *Biologi Italiani*, **28**, 16–26 (1998).

32. A.P. Togna and S. Frisch, *86th Mtg. Air Waste Management Assoc.*, Denver, Col., June 14–18, 1993.

33. L. Benedusi, G. Caretta, C. Collivignarelli, C. Dacarro, G. Del Frate, and V. Riganti, *Ingegneria Ambientale* **22**, 271–284 (1993).

34. S.J. Ergas, K. Kinney, M.E. Fuller, and K.M. Scow, *Biotechnol. Bioeng.* **44**, 1048–1054 (1994).

35. D.B. Janssen, R. Oldenhuis, and A.J. van den Wijngaard, in Verein Deutscher Ingenieure ed., *VDI Berichte 735: Biologische Abgasreinigung*, VDI Verlag, Düsseldorf, 1989, p. 25.

36. S.P.P. Ottengraf and A.H.C. van den Oever, *Biotechnol. Bioeng.* **25**, 3089–3102 (1983).

37. K.F. Kosky and C.R. Neff, *NWWA / API Conf. on Petroleum Hydrocarbons and Organic Chemicals in Ground Water*, Houston, Tex., November 9–11, 1988.

38. H. van Langenhove, E. Wuyts, and N. Schamp, *Water Res.* **20**, 1471–1476 (1986).

39. C. Kyeoung-Suk, H. Mitsuyo, and S. Makoto, *Appl. Environ. Microbiol.* **58**, 1183–1189 (1992).

40. R. Helmer, *Stuttgarter Berichte zur Siedlungswasserwirtschaft*, Band 49, Oldenbourg, München, 1972.

41. W. Ludwig, O. Fischer, and F.X. Kneer, *Chem. Ing. Tech.* **58**, 742 (1986).

42. M. Wainwright, *Trans. Br. Mycol. Soc.* **90**, 159–170 (1988).

43. S.P.P. Ottengraf and J.H.G. Konings, *Bioprocess Eng.* **7**, 89–96 (1991).

44. R.M.M. Diks and S.P.P. Ottengraf, in Verein Deutscher Ingenieure ed., *VDI Berichte 735: Biologische Abgasreinigung*, VDI Verlag, Düsseldorf, 1989, p. 7.

45. H.G. Liebe and V. Herbig, *Bericht II*, 1.1-50441-5/132, Umweltbundesamt, Berlin, May, 1986.

46. T.D. Brock, M.D. Madigan, J.M. Martinko, and J. Parker, *Microbiologia*, Città Studi Edizioni, Milan, 1995.

47. K.H. Engesser, in A.J. Dragt and J. Van Ham eds., *Biotechniques for Air Pollution Abatement and Odour Control Policies*, Elsevier Amsterdam, 1992, pp. 33–40.

48. R.C. Bayly and M.G. Barbour, in D.T. Gibson ed., *Microbial Degradation of Organic Compounds*, Marcel Dekker, New York, 1984, pp. 253–294.

49. S.F. Vecht, M.W. Platt, Z. Er-El, and I. Goldberg, *Appl. Microbiol. Biotechnol.* **27**, 587–592 (1988).

50. R.W. Eaton and K.N. Timmis, *J. Bacteriol.* **168**, 123–131 (1986).

51. O.O. Amund and I.J. Higgins, *Antonie van Leewenhoek* **51**, 45–56 (1985).

52. J.F. Davey and D.T. Gibson, *J. Bacteriol.* **119**, 923–929 (1974).

53. D.T. Gibson and V. Subramanian, in D.T. Gibson ed., *Microbial Degradation of Organic Compounds*, Marcel Dekker, New York, 1984, pp. 361–369.

54. G. Baggi, P. Barbieri, E. Galli, and S. Tollari, *Appl. Environ. Microbiol.* **53**, 2129–2132 (1987).

55. G. Schraa, B.M. Bethe, A.R.W. van Neerven, W.J.J. van den Tweels, E. van der Wende, and A.J.B. Zehnder, *Antonie van Leewenhoek* **53**, 159–170 (1987).

56. S. Hartmans, M.J. van der Werf, and J.A.M. de Bont, *Appl. Environ. Microbiol.* **56**, 1347–1351 (1990).

57. D.B. Dzhusupova, B.P. Baskunov, L.A. Golovleva, R.M. Alieva, and A.N. Ilyaletdinov, *Mikrobiologiya* **54**, 136–140 (1985).

58. M. Zilli, A. Converti, A. Lodi, M. Del Borghi, and G. Ferraiolo, *Biotechnol. Bioeng.* **41**, 693–699, (1993).

59. T. Omori, H. Ishigooka, and Y. Minoda, *Agric. Biol. Chem.* **50**, 1513–1518 (1986).

60. M.R. Smith and C. Ratledge, *Appl. Microbiol. Biotechnol.* **30**, 395–401 (1989).

61. J.A. Don, in Verein Deutscher Ingenieure ed., *VDI Berichte 561*, VDI Verlag, Düsseldorf, 1986, pp. 63–74.

62. B. Hippchen, *Mikrobiologische Untersuchungen zur Eliminierung Organischer Lösungmittel im Biofilter*, Stuttgarter Berichte zur Siedlungswasserwirtschaft, Band 94, Oldenbourg, München, 1985.

63. P.G. Maurer, *BMFT Forschungsbericht T*, 79–114 (1979).

64. L. Rodhe, L. Thyselius, and U. Berglung, Report N. 76 Jordbrukstekniska Intitutet, Uppsala, Sweden, (Joint JTI/AFRC Engineering, Silsoe, translation N. 1, New Series), 1988.

65. C.C. Pearson, V.R. Phillips, G. Green, and I.M. Scotford, in A.J. Dragt and J. van Ham eds., *Biotechniques for Air Pollution Abatement and Odour Control Policies*, Elsevier, Amsterdam, 1992, pp. 245–254.

66. V. Torretta, *Ingegneria Ambientale* **25**, 379–389 (1996).

67. J.L. Blitz and A.F. Elvidge, *Environ. Protect. Bull.* **16**, 3–6 (1992).

68. G. Stucki, R. Galli, H. Ebersold, and T. Leisinger, *Arch. Microbiol.* **130**, 366–371 (1981).

69. M.R. Smith, *Biodegradation* **1**, 191–206 (1990).