

Short Review

Bioreactor applications in waste treatment

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ABSTRACT

An overview of bioreactor applications in treatment of gaseous, liquid and solid wastes is presented with emphasis on newer technologies. Waste treatment is considered in a broad context including concentration by bioaccumulation, degradation to substances with reduced environmental impact and upgrading to such useful products as feeds, foods and fuels. Biofilters and bioscrubbers for gaseous pollutants, high-rate municipal and industrial wastewater treatment in airlift bioreactors, reactor-based soil bioremediation, artificial wetland filters for liquid effluents, and protein enrichment of agricultural solid residues are some of the technologies reviewed. The various treatment strategies are illustrated with examples. The developments discussed point to an increasing role for bioreactor based processes in waste treatment and reuse.

INTRODUCTION

The intensity of human activity, linked ultimately to population pressures, has unleashed such destructive forces which, if unchecked, have the potential to severely and irreversibly affect life on Earth. Excessive and irrational use of resources, destruction of entire ecosystems, forcing of animals and plants to extinction, generation of toxic (and huge amounts of not so toxic) wastes are all interrelated phenomena which remain largely uncontrolled in the guise of benefiting mankind. Undeniably, generation of gaseous, liquid and solid wastes is an unavoidable consequence of industrial, agricultural and domestic activities. Nevertheless, the environmental impact of human activity must be minimized to ensure sustainable quality of life and, eventually, for survival itself. While resource conservation, better utilization and balancing of the human population as a part of the overall ecosystem would ultimately have the greatest influence on sustainability of the planet, reduced generation, improved treatment and utilization of wastes will remain an essential component of an overall strategy for maintenance of environmental quality.

Many waste streams are amenable to biological treatment: either degrada-

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tion of harmful materials to ones with reduced environmental consequences, or, upgrading to useful products by means of natural, selected or engineered microorganisms and microbial enzymes. When degradation or upgrading are not feasible, microorganisms may be used to concentrate such pollutants as heavy metals in very dilute waste streams for subsequent disposal by other means. Biotreatment of wastes may be done in situ, in the place of occurrence of the waste (Halden, 1991; McDermott et al., 1989), as in bioremediation of contaminated soil in the field (Fouhy and Shanley, 1991). Alternatively, the contaminated material may be treated in bioreactors – as in treatment of most wastewaters – where better containment and superior environmental controls may allow faster, more complete and cost-effective treatment (Allsop et al., 1993; Choi et al., 1992; Eckenfelder et al., 1989). This overview looks at some newer bioreactor based technologies for treatment of gaseous, liquid and solid wastes.

TREATMENT PROCESSES

Gaseous effluents

Gaseous streams polluted with low loadings of biodegradable organics and inorganic odorous substances (e.g., hydrogen sulfide) can be treated in biofilters and bioscrubbers.

Biofilters. Biofilters are beds of soil or compost, about 1 m deep, with an underlying distribution system for the contaminated gas. As the contaminant-laden gas moves up through the moist bed, the pollutants are removed by sorption (Bohn, 1992) and oxidized by the microbial population immobilized in the bed. Gaseous effluents containing volatile organic compounds (VOCs) and such odorous substances as hydrogen sulfide are amenable to treatment in biofilters (Fouhy, 1992; Bohn, 1992). Typically, these units are suitable for gas streams with low dust loads, VOC loadings below ca. 1000 mg m^{-3} , and gas temperatures in the range of $10\text{--}43^\circ\text{C}$ (Fouhy, 1992). Good gas flow distribution and control of moisture content of the bed are necessary for effective operation. Humidity control is achieved by saturation of the inlet gas with water; occasional water sprays may also be employed. When contaminant loadings are high or variable, biofilters may be used as a polishing step following one of the more conventional treatment processes. Pollution removal efficiencies of 80–95% are attainable and have been demonstrated in several large installations (Fouhy, 1992). A new biofilter may take up to several months to become fully effective; however, once established, compost filters function virtually maintenance-free for several years (Fouhy, 1992) and soil based filters may be effective for decades (Bohn, 1992).

Removal of contaminants in biofilters can be modelled as a first-order pro-

cess (Bohn, 1992). Hence, in an ideal plug flow bed, the residence time, t , necessary for reduction of a contaminant from an inlet concentration C_i to an outlet concentration C_o can be calculated using:

$$t = -\frac{1}{k} \ln \frac{C_o}{C_i} \quad (1)$$

when the removal rate constant, k , has been experimentally established. For a given residence time, the maximum gas flow rate, Q , through the bed is given as:

$$Q = \frac{Az\phi}{t} \quad (2)$$

where ϕ is the bed voidage, A is the cross-sectional area of the bed, and z is its depth. For 90% removal of alcohols and aldehydes, the residence time may range from 30 seconds to a few minutes (Bohn, 1992). More recalcitrant compounds such as trichloroethylene may require as much as 150 min for 90% reduction in concentration (Bohn, 1992). Apart from a consideration of removal kinetics, biofilter design requires consideration of such other factors as pressure drops and flow distribution characteristics.

Bioscrubbers. Conceptually similar to conventional gas scrubbers, bioscrubbers are employed when heavier contaminant loadings, less soluble contaminants or contaminant toxicity make biofilters unsatisfactory. A wider variety of contaminants can be treated in bioscrubbers than in biofilters (Fouhy, 1992). In the more common type of bioscrubbing scheme, activated sludge mixed in water is contacted with the gaseous effluent in a packed bed absorption tower. Contaminants transfer to the sludge-water slurry which is taken to holding or sedimentation tanks where most of the degradation takes place. Clarified liquor from the sedimentation tanks is recycled to the absorption column. A modification of the activated sludge bioscrubber is the 'Biosolv' process (Fouhy, 1992) which utilizes an emulsion of activated sludge, silicone oil and water for absorption. This scheme is particularly suitable for treating such compounds as aromatic hydrocarbons which have very low water solubilities. An alternative 'bioscrubbing' scheme combines conventional, non-biological, water scrubbing of the gas with treatment of the scrubber liquid in a trickle bed biological filter. Degradation of the contaminants occurs in the biofilms immobilized on the packing in the trickle filter. These filters have the usual features of the well-known wastewater treatment trickle beds.

Absorption of contaminant laden gases in activated sludge slurry in tall airlift bioreactors (Fig. 1) is a potentially better alternative to packed column scrubbing. When the gas stream is available at sufficient pressure, its injection at the bottom of a tall airlift reactor can be arranged to ensure complete

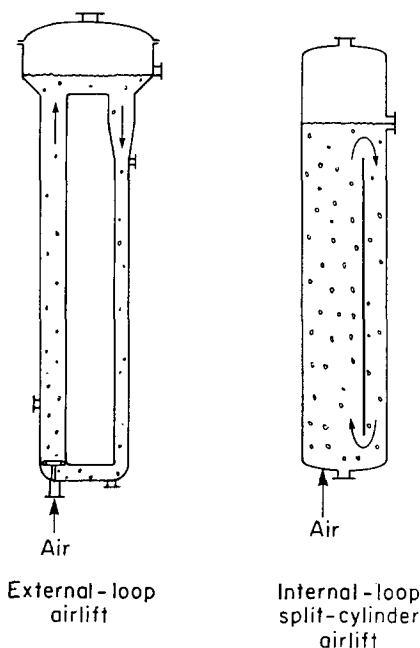


Fig. 1. External- and internal-loop airlift bioreactors. Gas is injected in the riser; the region with downflow of liquid (arrows) is the downcomer.

absorption of the gas and volatiles as a result of the high hydrostatic head in the reactor. Good absorption, combined with exceptional oxygen transfer capability of airlift systems (Chisti, 1989a), should allow rapid oxidation of the contaminants. Although well proven in treatment of municipal and industrial wastewaters (Chisti, 1989a,b; Opie, 1993; Redman, 1987; Varey, 1992), so far, the airlift bioreactors have not been applied to treating gaseous effluents as proposed here.

Liquids and slurries

Municipal and industrial wastewaters and concentrated slurries of biodegradable organics have traditionally been treated predominantly by aerobic or anaerobic biological processes (Eckenfelder et al., 1972; Eckenfelder et al., 1989; Hill et al., 1991; Winter, 1984). Activated sludge reactors with microbial populations suspended in the wastewater, immobilized biofilm reactors such as the rotating disc contactor and the trickle filter, or anaerobic digestion tanks are well established bioreactors in waste treatment. These systems are not discussed further, except when used in novel applications.

A recent trend in biological treatment of liquid effluents by the activated sludge method is toward *process intensification* by greater application of low-volume-high-rate airlift and deep shaft bioreactor technologies (Chisti, 1989a;

Chisti and Moo-Young, 1993; Opie, 1993; Redman, 1987; Vary, 1992). These reactors are being employed as stand alone treatment units, as well as being used to extend the performance of the older, conventional, plants. Airlift reactors are pneumatically agitated by air injection into the riser (Fig. 1). Up-flow of air and wastewater occurs in the riser, most of the gas leaves the liquid in the head region of the reactor (Chisti and Moo-Young, 1993) and gas-free wastewater recirculates through the downcomer. In deep shaft systems, once the reactor has been started in the airlift mode, most of the gas injection is transferred from the riser to part way down the downcomer. The downcomer gas injection point is so arranged that the overall gas content in the downcomer remain lower than in the riser; hence, the lower density riser fluid moves up the riser and the denser liquid in the downcomer flows downward, maintaining a circulatory pattern. Highly turbulent flows, combined with good oxygen absorption characteristics of these reactors, create conditions which allow rapid biological oxidation of pollutants.

Yet other developments have occurred in biological *phosphate removal* from wastewater. Conventional activated sludge wastewater treatment processes typically remove <40% phosphate from the water. The treated effluent can cause eutrophication problems in receiving waters. To alleviate this situation, chemical additives (e.g., calcium, iron or aluminum salts) are sometimes used to remove phosphates by precipitation. As an alternative, biological phosphate removal technology has been developed to achieve up to 90% removal of phosphate in activated sludge plants incorporating the advanced phosphate removal steps. The literature in this area has been reviewed by Yeoman et al. (1986). For phosphate removal, activated sludge from the final clarification stage is contacted with the incoming wastewater in an anaerobic first stage followed by aerobic stages in a plug flow type bioreactor configuration (Yeoman et al., 1986). A combination of biological and chemical mechanisms removes the phosphate without any additives. The design of the treatment process, its operation and the characteristics of the raw wastewater are important, interdependent, considerations in attaining consistent phosphate removal performance in these processes.

In other related developments, methods of utilizing microalgae for wastewater treatment (including nitrate and phosphate removal) have been reviewed (de la Noue and de Pauw, 1988) and design and operational considerations for anaerobic immobilized film wastewater treatment plants have been described (Hall, 1987).

Another relatively new technology employs *artificial wetland*, or 'reed bed' bioreactor systems for reduction of biochemical oxygen demand (BOD_5) and total suspended solids (TSS) in municipal and industrial wastewaters. Aquatic plants such as bulrush, cattails, common reed, water hyacinth, swamp potato and duck potato rooted in rock and gravel media beds flooded with wastewater flowing through the bed and root zone, make-up the wetland filters. The

media and plant roots provide a filter mechanism for suspended solids removal, in addition to supporting microbial films of pollutants degrading microorganisms. Often, wetland filters are employed to upgrade effluent already treated in conventional facultative lagoons, which are quite capable of BOD₅ reduction to acceptable levels, but may not meet TSS criteria because of the natural occurrence of algae (Zachritz and Fuller, 1993). The rate of BOD removal is the limiting factor in the treatment performance of wetlands. First-order removal kinetics apply for plug flow of effluent through the bed. Zachritz and Fuller (1993) cite several design guidelines which recently became available for these systems.

Another area for bioreactor application is in *soil bioremediation*. A particularly attractive treatment method for soils contaminated with hydrocarbons and other organics is soil washing (Fouhy and Shanley, 1991; McDermott et al., 1989). Washing with high-pressure jets of hot water, with or without added surfactants, is already practised commercially in Europe in Weert at a plant operated by BSN Bodemsanering Nederland. An alternative washing technique utilizes solid-liquid contactors such as mechanically agitated tanks to rub the soil particles against each other in presence of wash solutions. This 'agitation scrubbing' procedure has been tested on pilot scale (10 t h^{-1}) at the Soil Recycling Demonstration Plant operated by the Toronto Harbour Commission and SNC Inc. in Toronto, Canada. These examples of large-scale soil washing operations are by no means the only ones (Chowdhury, 1992; Pheiffer et al., 1990). While the ex-situ bioremediation of hydrocarbon contaminated soils using washing followed by treatment of the wash water has been proven (Pheiffer et al., 1990), problems remain. The washing processes clean only the coarser particles ($\geq 0.063 \text{ mm}$ particle size); smaller particles or fines which retain the bulk of the original contamination are not cleansed. At present, the fines are disposed of in secure landfills. As a result, soil washing technologies are economically applicable only to soils with less than 20–30% fines by volume. The applicability of soil washing operations can be extended to clayey soils ($> 30\%$ fines) by incorporating a fines treatment step based on airlift bioreactors. Airlift reactors have proven economics of operation in large scale wastewater treatment applications (Chisti, 1989a). Sufficiently high axial liquid velocities, capable of suspending the fines, can be developed in these reactors (Chisti and Moo-Young, 1993). In a conceptual fines treatment process an aqueous slurry of the fines could be aerated and pneumatically agitated in airlift devices containing mixed populations of contaminant(s)-degrading microorganisms. Depending on the hydrodynamic conditions, the microbes may grow either as a film on the soil particles, or, they may remain suspended in the liquid phase. While much design information exists for airlift reactors in wastewater treatment, cell culture and fermentation applications (Chisti, 1989a; Chisti and Moo-Young, 1991), little is known of the hydrodynamic and transport behaviour of soil slurries in those

systems. We are now addressing questions regarding the effects of solids-loading on circulation of the slurry, the effects of particle size and density on hydrodynamics, and solid-liquid and gas-liquid mass transfer characteristics, with a view to demonstrating the feasibility of airlift bioreactors for decontamination of soil fines.

Solid wastes

Biodegradable solid wastes – agricultural and forestry residues, remains from food and feed processing, vegetable and animal component in domestic refuse, etc. – have traditionally been composted or landfilled. With increasing expense of landfilling, composting is again in prominence. Established techniques such as heap composting are being supplemented by bioreactors based techniques utilizing such designs as rotating drum composters. Smaller amounts of such residues as wheat bran are often used in other non-compost solid-state fermentations to produce relatively high value products such as enzymes.

Anaerobic biogas digesters, treating predominantly solid wastes, have historically been used for sewage sludge stabilization in wastewater treatment plants (Winter, 1984). Digesters designed specifically for methane generation from animal manures are now fairly common, particularly in China and India. Anaerobic treatment bioreactors are receiving renewed attention for treatment of concentrated solid wastes, including petrochemical and other industrial wastes. Methane production from landfills has been the subject of some attention and a few landfills have been expressly designed with gas production in mind; however, this technology is expected to remain of limited use.

Some of the solid-state fermentation methods and bioreactor systems are now being adapted for such novel applications as bioremediation of soils contaminated with hydrocarbon and other pollutants (Fouhy and Shanley, 1991). Rotating drum type fermenters have a demonstrated potential in this area.

Waste treatment approaches which fully utilize the waste by conversion to higher value products without generating further waste are particularly desirable. One such scheme for food and fodder production is illustrated here.

Foods and feeds by protein enrichment. Lignocellulosic residues (e.g., straw, corn stover, sugarcane bagasse) from agriculture and silviculture represent a solid waste disposal problem which can be abated by reuse of this resource. Much of the cellulosic residues originates from plants used traditionally in food and feed production. Because they come from acceptable food sources, the residues could potentially be upgraded to food by improvements in digestibility, nutritive value and palatability. Although many processes have been described for converting various types of cellulosic solid substrates to protein-

rich products for food and fodder, most are based on cellulolytic organisms of doubtful food safety. Moo-Young et al. (1992) have developed a new process for food grade mycoprotein. The process is an extension of a recent invention for converting cereal-grain bran residues into proteinaceous products (Moo-Young et al., 1990). The process is based on the filamentous fungus *Neurospora sitophila* which has a long history of use as food in oriental preparations such as *ontjom* (Hesseltine and Wang, 1967; Steinkraus, 1986; Wood and Yong, 1975). Additionally, *N. sitophila* has a processing advantage as being one of the faster growing microfungi. With a maximum specific growth rate of 0.40 h^{-1} it has a doubling time which is shorter than that of some bacteria (Solomons, 1975). By comparison, the maximum specific growth rates of other common industrial fungi are half (e.g., for *Aspergillus niger*) or even less than a third (e.g., for *Penicillium chrysogenum*) than that of *N. sitophila* (Solomons, 1975).

Conceptually, the *N. sitophila* mycoprotein production process consists of the following steps:

- size reduction of the cellulosic residue by milling or grinding;
- treatment of the residue with alkali, acid and/or steam to increase the accessibility of the cellulose in the particles;
- fermentation of the residue with *N. sitophila* either in submerged or surface culture;
- solid-liquid separation and dehydration of the product for direct use as fodder; and
- blending, possible nucleic acid reduction, texturizing and flavouring operations for human food applications.

The size reduction, solid-liquid separation and dehydration steps use well known chemical engineering operations discussed elsewhere (Chisti and Moo-Young, 1991). The alkali pretreatment step (which, depending on the cellulosic material type, may not be needed) has also been described (Moo-Young et al., 1992). Blending, texturizing, colouring and flavouring operations enhance the palatability and the organoleptic properties of the product. These operations are in common use in the food processing industry, and have been developed also for the fungal protein food 'Quorn' being marketed in the United Kingdom (Steinkraus, 1986). Nucleic acid reduction is also used in Quorn manufacture (Steinkraus, 1986). The reduction of nucleic acids may be required if the product is used for human food, but it is not necessary for fodder product because in animals uric acid, formed by breakdown of nucleic acids, is readily excreted by conversion to allantoin. The RNA reduction techniques have been reviewed before (Solomons, 1975; Sinskey and Tannenbaum, 1975).

For the *N. sitophila* mycoprotein process, the optimal conditions for cellulose utilization and protein production were found to be 35–37°C tempera-

ture, pH 5.5 and agitator tip speed not exceeding 2.35 m s^{-1} in a 75-L stirred tank fermenter (Moo-Young et al., 1992). Up to ca. 90% utilization of cellulose could be achieved within 40 hours. The cellulolytic performance of *N. sitophila* compared favourably with that of *Chaetomium cellulolyticum* which is well known for its ability to degrade cellulose. The *N. sitophila* raw protein product had a pleasant almond smell. Almond or minced-meat flavour occurs also in *ontjom* produced by fermentation of peanut press cake by *N. sitophila* (Hesseltine and Wang, 1967). The average composition of the fungus was (% w/w): 45% crude protein, 40% carbohydrates, 10% fats, 5% minerals, vitamins, etc. The amino acid composition of the fungal protein compared favourably with that of fodder yeast (*Candida utilis*), soyabean meal and the FAO reference protein.

The process has been successfully scaled-up to 1300-L pilot plant (Moo-Young and Chisti, 1988). Because *N. sitophila* fermentations are susceptible to mechanical damage in high speed Rushton disc turbine agitated fermenters (Moo-Young et al., 1992), the pilot bioreactor used slower running axial flow 'Prochem' hydrofoil-type impellers (Chisti and Moo-Young, 1991) placed inside a draft tube (Fig. 2) to enhance axial flow of the highly viscous, non-Newtonian fermentation broth (Moo-Young and Chisti, 1988). This arrangement improves the bulk mixing in the fermenter. Air is sparged in the annulus between the draft tube and the walls of the fermenter. This airlift-stirred tank hybrid bioreactor (Fig. 2) has proven superior to either a basic airlift or a stirred tank. Although the airlift configuration without mechanical agitation could be used (Moo-Young et al., 1987), the bulk mixing of the broth was not as effective as in the hybrid device. Unlike many viscous fer-

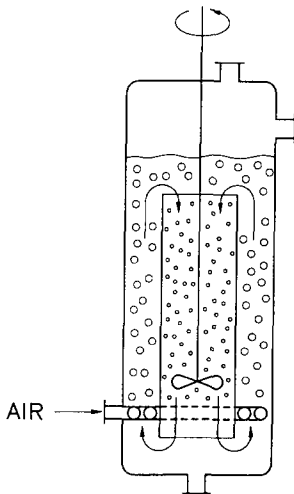


Fig. 2. Hybrid agitated airlift bioreactor for low impeller speed bulk mixing of viscous mycoprotein fermentation broths.

mentations which have been successfully performed in airlift bioreactors (Chisti, 1989a), *N. sitophila* broths containing cellulosics particulates are more viscous and pseudoplastic. For example, *C. cellulolyticum* broths are rheologically easier to handle than those of *N. sitophila* (Moo-Young et al., 1987). Other bioreactor scale-up considerations such as gas-liquid mass transfer and gas holdup effects in those broths have been reported on previously (Chisti and Moo-Young, 1988; Moo-Young et al., 1987). Other general aspects of fermentation plant design which apply in different degrees to food and feed plants have been detailed elsewhere (Chisti, 1992a,b; Chisti and Moo-Young, 1991). Unlike most waste treatment processes which use *mixed* microbial populations under non-sterile conditions, processes for upgrading wastes to foods, feeds and other products are often carried out aseptically.

The economics of fungal protein manufacture were discussed in detail by Moo-Young et al. (1979; 1986) for a production processes based on *C. cellulolyticum*. Those economic analyses are equally valid for the *N. sitophila* based process because of the similarities between the two production schemes: same cellulosic substrates, media supplements, and pretreatment operations; identical downstream processing of the product and similar growth and protein production characteristics of the two fungi.

For conversion of Kraft paper pulpmill clarifier sludge (95% cellulose) to mycoprotein, Moo-Young et al. (1986) determined that a minimum processing capacity of 6.5 t per day of sludge was required to break-even. A 96% conversion of the cellulose to a product containing 38% protein was assumed with soymeal-based protein being the reference selling price. The major contributors to production cost were the utilities (at 37.2% of total cost), the nutrients (at 36.4% of total cost) and the equipment depreciation (at 18.1% of total cost).

In specialty foods industry the production cost considerations are of lesser importance than in feeds manufacture. For example, the Quorn mycoprotein is commercially produced using hydrolysed starch – a more expensive substrate than cellulosic residue – for fungal cultivation (Steinkraus, 1986). Hence, the *N. sitophila* protein process is expected to be economically viable in specialty foods markets. In addition to producing foods and feeds, industrial cellulases may be produced from lignocellulosic wastes using *N. sitophila* (Oguntimein et al., 1992) and other microfungi.

CONCLUSION

Bioconversion of wastes to harmless substances or higher value products already has a significant role in environmental pollution control and improved resource utilization. Both in situ and bioreactor based treatment processes are experiencing rapid development and increasing deployment in practical applications. The current infancy of the many of these bioprocesses,

combined with continuing advances in biochemical engineering, microbiology, biochemistry and genetics of the waste converting organisms, guarantee yet better future performances and improved process economics which will further enhance the scope of these waste conversion technologies.

ABBREVIATIONS

A	Cross-sectional area of the bed
BOD_5	Biochemical oxygen demand (5 days)
C_i	Inlet concentration of contaminant
C_o	Outlet concentration of contaminant
FAO	Food and Agriculture Organization of the United Nations
k	Removal rate constant
Q	Volume flow rate of gas
RNA	Ribonucleic acid
TSS	Total suspended solids
t	Residence time
VOC	Volatile organic compounds
z	Depth of the bed
φ	Bed voidage

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