FIXED FILM BIOLOGICAL PROCESSES FOR CONTAMINATED GROUNDWATER CLEANING

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ABSTRACT

Traditional fixed film biological treatment processes include trickling filter (biological filter) and rotating biodiscs. while recently developed processes are fluidised bed and submerged aerobic filter. The most important future of fixed film biological reactors for wastewater treatment is the retention of the microbial population on inert surface of the support medium so that wash-out does not occur.

KEYWORDS

cleaning technology, biological treatment, biofilm

INTRODUCTION

Biological processes for treatment of water are in most cases preferable to other methods due to their low costs and capacity of destroying organic material. Contaminated groundwater may contain too low concentrations of biodegradable organic material to support the bacterial growth in conventional biological treatment reactors to such as trickling filter and activated sludge process.

Recent research studies have focused on fixed-film biological reactors which offer some encouraging potential of removing low concentrations organic material even to not detectable levels.

BACKGROUND CONSIDERATIONS

Efficiency of a biological treatment unit and its operational stability is function of the number of viable bacteria that can be retained in the treatment unit Hence immobilisation of microbial population, especially in biological treatment processes employing low growth rate microbes such as nitrification bacteria and methanogenic bacteria. Retention of the microbial population can be achieved by recirculation of separated biomass or by establishing biofilm on the surface of support material. Examples of these practices are the activated sludge processes, percolating biological filter and rotating biological contentors.

In wastewater treatment fixed-film bioreactors are studied under steady-state conditions in order that a certain amount of active microbial population is maintained in biofilm. Consequently there exists a minimum substrate concentration in treated effluent as shown by Rittmann and McCarty (1980). The sustaining substrate concentration, Ss, is expressed empirically as:

$$Ss = Ks\ (d/Yk - d))$$

in which

Ks = Monod half-velocity concentration
Y = True growth yield
k = maximum specific substrate utilisation rate
d = microbial decay coefficient

Stratton *et al* (1983) indicated that for aerobic system using simple compounds as substrate, typical values of Ss were in the range of 0.1 - 1.0 mg/l. When organic pollutants in water having concentrations lower than Ss are to be reduced through biotransformation and biodegradation to non detectable levels as in the case of groundwater remediation, the microbial film can not be maintain in steady-state. Instead, due to the net loss of biomass, the microbial population must be in unsteady-state. It is important to know if fixed film biological reactors are capable to treat low concentration organic pollutants normally encountered in contaminated groundwater. Moreover, the vulnerability and stability of a biosystem operating at unsteady-state must be addressed. Rittmann and McCarty (1981) demonstrated that a developed biofilm is capable of removing substrates to concentrations well below Ss.

FIXED-BED BIOFILM REACTORS

Laboratory experimental results presented by Rittmann and Brunner (1984) show that unsteady-state biofilm, originally cultivated with 3 mg/l of galactose, reduces the substrate from 0.3 to 0.04 mg/l in unsteady-state for a long period of operation. Neufeld *et al* (1994) have shown that an initially well developed biofilm in high concentration substrate is critical for the later removal of low concentrations (10-20 mg/l) of toluene. They also found that biotransformation rather than complete mineralisation of the tested compounds is the main mechanism of the removal.

Waste oil containing up to 13 mg/l of alkyl benzene and polynulear aromatic compounds of a storage lagoon was treated by Bouwer *et al* (1992) in laboratory scale aerobic and anaerobic upflow filters operating with hydraulic retention times of 5 hours and 2 days, respectively. Most of the compounds in the oil were removed at efficiency greater than 98%. C-14 tracer experiments showed that 60, 73 and 74% of the labelled carbon were converted to $14CO_2$ from mineralisation of toluene, naphthalene and 2-methyl-naphthalene, respectively, in the aerobic filters.

Catalysed by the enzyme, methane monooxygenase, produced by methanotrophic bacteria, chlorinated aliphatic hydrocarbons can be converted into still unconfirmed intermediate products which in turn are readily mineralised by co-existing heterotrophic bacteria. Employing laboratory biofilm reactors, Arvin (1991) showed that the aerobic biodegradation of trichloroethane with concentrations up to 1.2 mg/l and dichloroethane 4 mg/l by methanotrophic bacteria were first order reactions. It was suspected that relatively high concentration of copper in the experiment substrate hindered the degradation of the tested compounds.

Upflow biofilm fixed bed reactors were employed by Stucki *et al* (1992) to study the aerobic biodegradation of 1,2.dichloroethane (DCA) under simulated groundwater conditions, i.e. pH = 6.5-7.5, temperature = 10-20° C and conductivity = 300-1200μ S/cm. It was found that the degree of degradation of DCA reached 90% when the initial concentrations were in the range of 20-25 mg/l. It was pointed out that treating high concentration DCA in a groundwater with low pH buffering capacity may require attention on pH control due to the production of acid.

FLUIDISED BED BIOFILM REACTORS

Fluidised bed biofilm reactors have been studied intensively for denitrification of high nitrate content surface and ground waters. A typical fluidised bed biological denitrification process is shown in Figure 1 which must satisfy the following essential conditions:

- sufficient upflow velocity to maintain fluidisation of the biofilm support medium,
- continuous separation, cleaning and reintroduction of the support medium, and
- addition of carbon source and nutrients with continuous monitoring and control.

It must be emphasised that concerning the residual carbon and nutrients remaining in treated water, denitrification of drinking water is better targeted to lower the nitrate concentration to within the legal limit rather than achieving maximum removal. Carbon-limited denitrification practice may result in high residual nitrite concentration which must be taken into account in the downstream treatment such as chlorination.

Hall and Zabel (1984) reported that at an upflow velocity of 22 m/h and a capacity of treatment of 115 m^3/h, biological denitrification in a fluidised bed plant was successfully obtained with 70% efficiency and resulted in 5 mg N/l of residual nitrate. Methanol and phosphoric acid were added at 45 and 0.2-0.4 mg/l, respectively, as carbon source and nutrient.

Most of the published studies on biological denitrification employed heterotrophic denitrifying bacteria. Gros *et al* (1986) however presented a denitrification plant in Germany treating groundwater by fixed bed bioreactor containing hydrogen oxidising autotrophic bacteria using hydrogen as electron donor and carbon dioxide as carbon source. The treatment unit reduced 18 mg/l N/l of nitrate to less than 1 mg/l without leaving nitrite in the treated water in a calculated hydraulic retention time of merely 1 hour at a flow rate of 50 m^3/h. In the Netherlands, a fixed bed of limestone was employed by Driscoll *et al* (1978) to retain autotrophic denitrifying bacteria using sulphur and sulphide as the electron donor.

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