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# KINETIC PARAMETERS OF PHENOL BIODEGRADATION WITH DIFFERENT MICROORGANISMS: A REVIEW

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## ABSTRACT

**Most industries release wastewater contains high concentrations of phenol which is toxic and contaminating the environment. Biological treatments are preferable for the phenolic compounds treatment. This review aims to investigate the impacts of temperatures, pH, and concentration of substrate on the phenolic compounds biodegradation, to compare the kinetic parameters of different microorganisms, and to discuss the different between the kinetic parameters of aerobic and anaerobic treatment. The review showed that most of the phenol biodegrading bacteria are *P. Putida* species and mixed cultures but *P. Putida* has a better adaptation to phenol biodegradation. The values of $\mu_{max}$ and $K_s$ in anaerobic process are smaller than the values attained in aerobic process. The optimum temperature to acclimatize bacteria to the phenol substrate is 30 ºC while the optimum pH condition is between 6.5 and 7.5. As the phenol concentration was increased, there was an increase in the values of the *Ks* and even when concentration is low; phenols have a significant inhibitory impact on ($\mu$). The values of *Ki* for phenol degradation for *P. putida* species were higher than the values of *Ki* of mixed cultures. The highest *Ki* value for the phenol degrading among *P.Putida* species was 1185.8 mg/L and the highest *Ki* value among mixed cultures was 648.1 mg/L and the highest *Ki* value among the other species was 2434.7 mg/L.**

***KEYWARDS*:** *kinetic parameters, Phenol, different microorganisms, and biological treatment*.

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Symbols and abbreviations

| | | | |
|---|---|---|---|
| DO | Dissolved oxygen, mg/L | $\theta_c$ | Solid retention time, day |
| *dx* | Net growth of bacteria over interval time *dt*, mg/L | *P. putida* | *Pseudomonas putida* |
| EPA | Environmental Protection Agency | $Q_o$ | Influent flow rate, L/day |
| *k* | Maximum waste utilization rate per unit weight of bacteria, time$^{-1}$ | $S_e$ | Concentration of effluent substrate mg/L |
| $k_d$ | Decay cell coefficient, time$^{-1}$ | $S_o$ | Concentration of influent substrate, mg/L |
| *Ki* | Inhibition constant, mg/L | *V* | Volume of aeration tank, L |
| *K*s | Half saturation constant, mg/L | *X* | MLSS in aeration tank mg/L |
| MLSS | Mixed Liquor Suspended Solids, mg/L | *U* | Specific substrate utilization rate, time$^{-1}$ |
| $\mu_{max}$ | Maximum specific growth rate, time$^{-1}$ | *Y* | Yield coefficient, mg/mg |

## 1. INTRODUCTION

Phenols are considered as one of the most serious environmental pollutants released from many industries such as petrochemical plants, plastic plants, pulp and paper manufacturing plants, oil refineries, paint plants, tannery plants, pesticide plants, textile plants, and steel plants (Bajaj, et al., 2009; Edalatmanesh, et al., 2008; Vasiliadou, et al., 2008; Zhang et al., 2011). Phenol concentration 10 g/L may be present in industrial wastewater. Due to their toxic effects, as well as cytoplasmic coagulation and permeabilisation of cellular membranes, phenolic pollutants can damage sensitive cells and thus cause environmental problems and profound health (Bodalo et al., 2008). Phenols are carcinogenic and extremely toxic as well as can persist in the environment for a long time, owing to their capacity for bioaccumulation and stability. Several Phenolic compounds have been described to decline COD values as well as phosphorous and

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nitrogen uptake in numerous microorganisms (Kargi, et al., 2005; Li et al., 2010). Phenolic compounds are recorded by the Unites States Environmental Protection Agency (EPA) among 129 prior treatment pollutants (Keith & Telliard, 1979).

Due to the toxicity of phenolic compounds and their difficulty to degrade, many treatment methods have been employed to decrease the concentrations of phenolic compounds including ion exchange, adsorption, bioactive activated carbon, biodegradation, membrane processes, and fenton processes (El-Naas & Makhlouf, 2008; Juang & Wu, 2007; Marrot, et al., 2006; Vinod & Reddy, 2006). Mostly physicochemical methods have been the conventional methods but afterward they caused problems in the effluents for instance, if chlorination is used, phenol becomes chlorophenols (Marrot et al., 2006). Consequently, for the removal of phenolic compounds from wastewater, biological treatment has evidenced to be the most encouraging and cost effective method (El-Naas, et al., 2009). The most reported used type of bacteria for phenol biodegradation has always been the strain of *Pseudomonas putida. P. putida* has been used by many researchers in immobilized and free forms and with different applications of bioreactors (Juang & Wu, 2007; Tsai & Juang, 2006). Several organic substances are either toxic to the biological treatment or non-biodegradable, thus, it must be pretreated into less toxic compounds or biodegradable. Biological treatment is ineffective for the degradation of phenolic compounds at high concentrations (more than 1450 mg/L) due to its biorefractory nature (Edalatmanesh et al., 2008). Furthermore, when phenols concentrations exceed 400 mg/L, the nutrients removal reduces (Viero et al., 2008). Freire et al. (2001) obtained that the sequencing batch reactor was very efficient in removing ammonium, reaching level of 95 %. For phenol removal, on the other hand 65 % were reached. Fang & Zhou, (1997) indicated that phenol removal exceeded 80 % at 6 h cycle length of sequencing batch reactor when influent phenol concentration was 1050 mg/L.

Aerobic microorganisms are capable to use aromatic compounds as energy source and sole carbon. Even though anaerobic and aerobic microorganisms are capable to degrade phenolic compounds, but aerobic methods are favored (Ellis et al., 2006). Many of the phenol degradation kinetics from the experimental researches were concentrated on the initial phenol concentration when the aerobic processes used (Arutchelvan, et al., 2006; Chen, et al., 2004; Kumar, et al., 2005; Nuhoglu & Yalcin, 2005). Few refer to other environmental influences such as salinity and temperature (Peyton, et al., 2002; Polymenakou & Stephanou, 2005).

Many research works have been conducted to determine the kinetics of phenol biodegradation each with either a specific microorganism culture or different operational condition.To the best of my knowledge no single research work exists on the review of the kinetics of phenol biodegradation. Knowledge of the kinetics of biodegradation is important for the design of biodegradation facilities and to predict the behavior of the system of phenol biodegradation processes. Subsequently, This review article aims: firstly to compare the kinetic parameters of the phenol biodegradation by different microorganisms, secondly to discuss the different between the kinetic parameters of aerobic and anaerobic treatment, and thirdly to investigate the impacts of temperatures, pH, and substrate concentration on the phenolic compounds biodegradation.

## 2. BIOKINETIC PARAMETERS

Biodegradation plays a significant role in the outcome of synthetic organic chemicals in both engineering and natural environments. As a result, prediction of the fate of such chemicals requires quantification of their biodegradation rates in the form of kinetic coefficients that can be used in mass balance equations. A comprehensive biodegradation study emphasizes the analysis of kinetic parameters which describe microbial growth and substrate removal (Tabak & Grady, 1990).

Biokinetic coefficients ($k_d$, $K_s$, k, Y, and $\mu_{max}$) can be determined for activated sludge process to treat wastewater using the following five equations (Metcalf & Eddy, 2003; Mizzouri & Shaaban, 2013). Equations are based on a completely mixed and continuous flow system that operated with an aerobic treatment process.

$$\frac{1}{\theta_c} = Y.U - K_d \quad \text{.....(Eq. 1)}$$

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$$U = \frac{Q_o.(S_\circ - S_e)}{V.X} \quad \text{.....(Eq.2)}$$

$$\frac{1}{U} = \frac{1}{k} + \frac{K_S}{k}\frac{1}{S_o} \quad \text{.....(Eq. 3)}$$

$$k = \frac{\mu_{\max}}{Y} \quad \text{.....(Eq. 4)}$$

$$\frac{1}{\theta_c} = \frac{1}{X}.\frac{dx}{dt} \quad \text{.....(Eq.5)}$$

where $\Theta_c$ is the solid retention time, day; $V$ is the volume of aeration tank, L; $Q_o$ is the influent flow rate, L/day; $dx$ is the net growth of bacteria over interval time $dt$, mg/L; $X$ is the MLSS in aeration tank mg/L; $S_e$ is the concentration of effluent substrate mg/L; $S_o$ is the concentration of influent substrate, mg/L; $\mu_{max}$ is the maximum specific growth rate, time$^{-1}$; $k$ is the maximum waste utilization rate per unit weight of bacteria, time$^{-1}$; $K$s is the half saturation constant mg/L; $k_d$ is the decay cell coefficient, time$^{-1}$; $U$ is the specific substrate utilization rate, time$^{-1}$; and $Y$ is the yield coefficient, mg/mg.

$Y$ and $k_d$ can be determined from (1/ $\Theta_c$) versus ($U$) plot where the endogeneous decay rate, $k_d$, is intercept and the slope is $Y$.

The biokinetic growth constants, $\mu_{max}$ and $Ks$ can be obtained from the Monod Eq. (6) for non-inhibitory wastewater.

$$\mu = \mu_{\max}\frac{S}{K_s + S} \quad \text{.....(Eq. 6)}$$

A μ versus $S$ plot can be used to find $\mu_{max}$ and $Ks$ ($When\ \mu = \mu_{max}/2\ (mg/L)$) as shown in Figure 1.

For the inhibitory wastewater Haldane model is preferred to describe the biodegradation as follows:

$$\mu = \frac{\mu_{\max} S}{K_s + S + \frac{S^2}{K_i}} \quad \text{.....(Eq. 7)}$$

Where $Ki$ is the inhibition constant.

There are some major factors that affects to the kinetic parameters values such as: the effects of pH, temperatures, inhibitor concentration, and initial substrate concentration. However, a number of researchers have investigated other parameters. For example, Park (1999) has stated that the greater the oxygen supply, the faster biodegradation takes place. As well as, Yavuz and Celebi (2004) have indicated that by using of magnetic field at optimum strength, the treatment efficiency of activated sludge was increased 40%. Besides, they specified that the kinetic coefficients of activated sludge for glucose removal have increased by 94, 41, and 36% for $\mu_{max}$, $Ks$, and $Y$ respectively after using of magnetic field at the optimum strength of 17.8 mT.

## 3. KINETICS OF PHENOL WITH DIFFERENT MICROORGANISM

The Monod equation (Eq.6) has been most frequently used to describe microbial growth kinetics in both pure and mixed culture systems on the one hand. The Haldane model (Eq.7) is desired in the case of phenolic compounds biodegradation on the other hand the biodegradation of self-inhibiting compounds is defined by Haldane kinetics (Bajaj et al., 2009; Juang & Tsai, 2006b; Monteiro et al., 2000; Li et al., 2010; Yan et al., 2005). The influence of a toxic compound on process practice is measured by inhibition parameter, $Ki$. It is important to mention that the Haldane Eq. simplifies to the Monod Eq. when $Ki$ is very large (indicating to lower biomass sensitivity to substrate inhibition). Therefore, small values of $Ki$ indicate that the effect of inhibition can be detected even at low concentrations of phenolic compounds (Marrot et al., 2006).

Marrot et al., (2006) have used a continuous regime to grow a mixed culture (biomass from wastewater treatment plant) using phenol as substrate. The results indicated that effluent with high phenolic compounds concentration (1.0g/L) can be treated by activated sludge with concentration of biomass around 10 g/L. However, Grady et al., (1996) have referred that the inhibition beyond 40 mg NO3-N /L and 300 mg phenol/L should always be taken into account in reactor design and Edalatmanesh, et al., (2008) have stated that there is a remarkable difference between the estimated and measured phenolic compounds concentrations in Haldane equation, once the initial phenolic compounds concentrations go above 100 mg/L. Many researches about the biodegradation of phenolic compounds by *P. putida* microorganism have been done because these microorganism have the

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highest degradative potential for phenols (Chung et al., 2003; Monteiro et al., 2000; Pazarlioğlu & Telefoncu, 2005). Nevertheless, Annadurai et al. (2002) have indicated that the mixed culture had a better capability than pure culture and *Pseudomonas putida* for phenol degradation.

Table 1 summarizes the Haldane kinetics for phenol biodegradation with pure culture using different process conditions obtained from earlier studies. *Pseudomonas putida* species were the most of the phenolic compounds biodegrading bacteria. Table 2 illustrates the Haldane kinetics for phenol biodegradation with mixed culture by different process conditions. In general, the $\mu_{max}$ values of mixed cultures (Table 2), *P. Putida* species, and other pure culture species (Table 1) were close each to other except Abohamed et al. (2004) and Arutchelvan et al. (2006) (Table 1) have found the specific growth rates very low in pure culture growth. As well as Suidan et al. (1988) and Firozjaee et al. (2011) (Table 2) have found the specific growth rates very low in mixed culture growth using anaerobic process. For *P. putida* species the *Ks* values increases with phenol concentration increasing. A small *Ks* value indicates that the $\mu_{max}$ could be reached quickly. The values of *Ki* for phenol degradation for *P. putida* species were higher than the values of *Ki* of mixed cultures. This is indicating that *P. Putida* species have a good adaptation to phenol biodegradation. The highest *Ki* value for the phenol degrading among *P.Putida* species was 1185.8 mg/L (Juang & Tsai, 2006a) and The highest *Ki* value for the phenol degrading among mixed cultures was 648.1 mg/L (Bajaj et al. 2009) and the highest *Ki* value for the phenol degrading among the other pure culture species was 2434.7 mg/L (Arutchelvan et al. 2006). Table 1 and Table 2 show a variation between kinetic coefficients. This variation may be due to many influences for example using different environmental factors, different adaptation processes and variation in microbial cultures.

## 4. KINETICS AND AEROBIC / ANAEROBIC TREATMENT

Degradation of phenol can be accomplished by aerobic and anaerobic microorganisms but the aerobic practices are preferred and the concentration of dissolved oxygen (DO) is a controlling aspect of the treatment method (Ellis et al., 2006; Etzensperger, et al., 1989). Melo et al., (2005) have used an aerobic rotating biological contactor for the phenol degradation. They have found as they increase the rotation speed, the phenol degradation rate is improved because the oxygen mass transfer coefficient increased.

Firozjaee et al. (2011) have used an anaerobic batch reactor to study the biodegradation of phenolic compounds in mixed culture. The $\mu_{max}$, $k_i$, and $K_s$ kinetic parameters for Haldane equation were 0.067 $h^{-1}$, 200 and 25.32 mg/L, respectively. Suidan et al. (1988) have reported the kinetic coefficients of phenol biodegradation with anaerobic process. The $\mu_{max}$, $K_s$, and $k_i$ coefficients were 0.027 $h^{-1}$, 0.03 and 363 mg/L, respectively. Table 2 illustrated a comparison between anaerobic kinetic coefficients with the kinetics achieved in aerobic studies.

The values of $\mu_{max}$ and $K_s$ are smaller than the values attained in aerobic process. As, the biodegradation in anaerobic process needs longer time, therefore, the biomass grows slowly and $\mu_{max}$ and $K_s$ values are smaller than the values reported for aerobic processes.

## 5. KINETIC PARAMETERS AND OPERATIONAL CONDITIONS

### 5.1 Kinetics and effects of temperature

Several environment impacts such as pH and temperature are significantly affecting to phenol degradation (Annadurai et al., 2002; Chung et al., 2003; Monteiro et al., 2000; Mordocco, et al., 1999; Pazarlioğlu & Telefoncu, 2005; Sa & Boaventura, 2001). A number of studies have found that high phenol removal efficiency can be accomplished near 30ºC. Nevertheless, the deviation outside the optimal range is sensitive to the rate of phenolic compound degradation (Chung et al., 2003; Mordocco et al., 1999). Sa & Boaventura, (2001) have stated that a deviation of 5ºC may origin a reduction in phenolic compounds biodegradation rate of almost 100% at the higher end and at least 50% at the lower end. This difference of efficiency removal at 30ºC is perhaps because at this temperature a higher level of metabolites produced. Martín et al. (2008) have referred that a noticeable delay in the uptake of phenol is produced when *P. putida* at 25 and 35ºC is cultured although optimum pH was retained. Biodegradation time was the longest at 35ºC (66 h). Besides, due to the mesophilic behavior of P. putida, at 35ºC kd was higher than at 25ºC. In most studies, the reported temperatures have

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usually been higher than 25°C. On the other hand, the temperature values of operating processes vary from 0 to 30°C in most wastewater treatment plants. Subsequently, data is limited about the effect of low temperatures on phenol biodegradation and biomass growth. (El Hajjouji et al., 2008; Shpiner et al., 2009).

Li et al.(2010) have stated that a psychrotroph, *Pseudomonas putida* LY1 bacterium can totally biodegrade 200 mg/L phenol with an optimal temperature value of 25ºC and a medium temperature ranged between 2.5 and 35 ºC. Chung et al. (2003) have studied the effects of temperature for immobilized and free cells on phenolic compounds biodegradation as presented in Figure 2. In both cases, the optimal temperature was equal to 30.8ºC. However the degradation rates were 19.4 and 28.1 mg/h for immobilized and free cell systems, respectively.

In summary, this review indicates that optimum temperature to acclimatize bacteria to the phenol substrate is 30ºC. However, many studies have reported the achievement of biodegradation of different concentrations of phenol in a medium temperature ranged between 10 and 37ºC according to the literature in Table 1 and Table 2.

**5.2 Kinetics and effects of pH**

The pH medium is a significant parameter in the successful of the biological treatment processes. Subsequently, researchers can use it for the indication of phenolic compounds biodegradation. A number of researchers reported that once the initial phenolic compounds concentration rises a slight reduction of phenol degradation is observed as pH variation increases (Monteiro et al., 2000; Pai et al., 1995).

Annadurai et al. (2002) have reported that the pH medium considerably affects the phenolic compounds biodegradation; experiments with *Pseudomonas putida* indicated that it could not resist changing in pH efficiently. pH values affects significantly the decay rate of the substrate and as phenolic componds decomposed a significant reduction in pH occurred. Accordingly, as the pH value deviates from neutral circumstance, phenol degradation is declined. However, Marotta et al. (2012) have indicated that the efficiency of phenol removal increases significantly with increasing pH. Martín et al. (2008) have stated that at controlled pH (pH 5 and 7), the maximum specific growth rate for phenol substrates was low and biomass yield on the phenol substrate was decreased. As well as, *Ks* values for phenol at pH 7 were higher than at pH 5.

Chung et al. (2003) have studied the pH effects of immobilized and free cells on phenolic compounds biodegradation as presented in Figure 3. They have found two optimal pH values: 8.0 for free cells and 6.8 for immobilized cells. In conclusion, this review indicates that optimum pH condition to adapt bacteria to the phenol substrate is between 6.5 and 7.5 according to the literature in Table 1 and Table 2.

**5.3 Kinetic parameters and different substrate concentrations**

It is well known that at high phenol concentration the inhibition of biodegradation will increases either in mixed cultures or in pure cultures. (Kumar et al., 2005). This inhibition is related with the hydrophobic perturbation of the membrane of bacteria (Léonard & Lindley, 1999).

(Powell, 1967). Has indicated that the substrate concentration that is in contact with the biomass is a significant factor for the metabolic activity at any time. Okaygun (1991) stated that the higher the So/Xo ratios, the lower the enzymes orchestrated in phenol degradation would be induced due to increasing toxicity of phenol. Marrot, et al (2006) stated that as phenol concentration increases the phenol degradation rate rises. However, the degradation rate decreases with further rises in phenolic conpounds concentration because of the significant effects of substrate inhibition. Firozjaee, et al, (2011) have refered to the relationship between the concentrations of phenol and the specific growth rate of activated sludge as in Figure 4. Okaygun (1991) stated that as the substrate to biomass ratio increased there was an increase in observed Ks values.

It is renowned that the Haldane model simplifies to the Monod model when *Ki* is very large. Subsequently, low *Ki* values indicate that at low phenol concentration the inhibition effect can be observed. The observations are verified by other researchers. Such as, Jiang et al. (2005) (Yeast Candida tropicalis), Kumar et al. (2005) (P. Putida) (Table 1), and Marrot et al.(2006) (aerobic mixed culture)(Table 2) who have referred to an insufficiency of the kinetics of inhibition at higher substrate values. Nonetheless in the situation of an aerobic mixed culture this insufficiency looks like to be reached for phenolic compounds concentrations stronger than in other studies. Phenolic compounds had a significant inhibitory

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effect on (μ) even at low concentrations. Finally it is clear that at higher concentration of phenol, the poisoning factor will increase resulting in deactivation of microorganisms.

## 6. CONCLUSIONS

Wastewater with high concentrations of phenol is toxic to the environment. The most interesting method for treating this wastewater is the biological treatment. Kinetics of biodegradation is important for the design of biodegradation facilities hence kinetic parameters of phenol biodegradation was investigated in this review paper. The review indicated that *Pseudomonas Putida* species have the best adaptation to phenol biodegradation. However, mixed cultures and other pure cultures can also biodegrade phenolic compounds. Haldan equation is the preferred model to find the kinetic parameters for phenol biodegradation. Variation of temperature has clear effects to the kinetic parameters and the optimum temperature could be 30 ºC for phenol biodegradation while pH variation has a slight effect to biomass grows and phenolic compounds biodegradation is deteriorating as the pH values deviates from neutral conditions. As the phenol to biomass ratio increased there was an increase in observed *Ks* values and even low concentrations of phenolic compounds initiating a significant inhibitory effect on (μ) values. The values of $\mu_{max}$ and *Ks* in anaerobic process are smaller than the values attained in aerobic process. The values of *Ki* for phenol degradation for *P. putida* species were higher than the values of *Ki* of mixed cultures.

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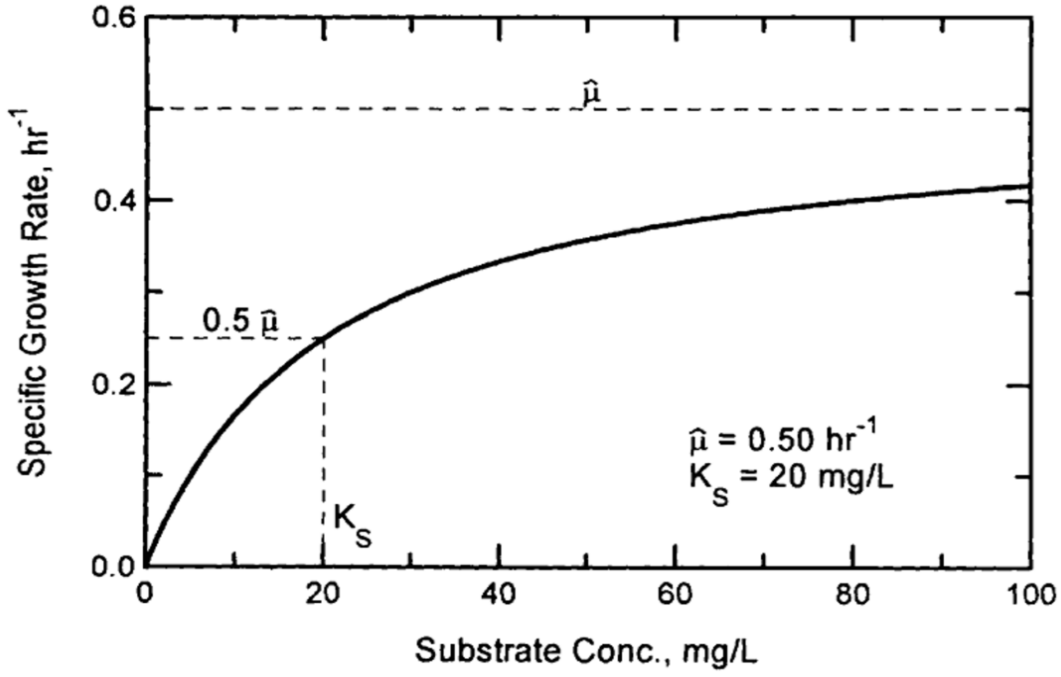


**Fig. (1):** Typical plot of the correlation between non-inhibitory substrate concentration and specific growth rate coefficient. The parameter values given were used to construct the curve with Monod Equation (Eq.6) (Grady et al., 2011).

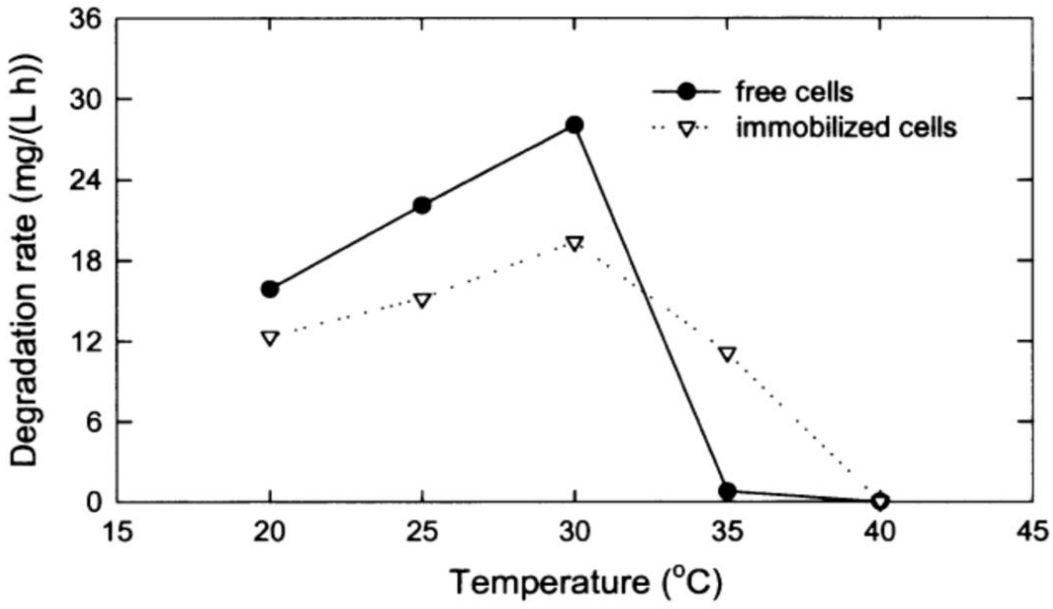


**Fig. (2)**: Relationship between phenolic compounds degradation and temperature values for immobilized and free cells (Chung, et al., 2003).

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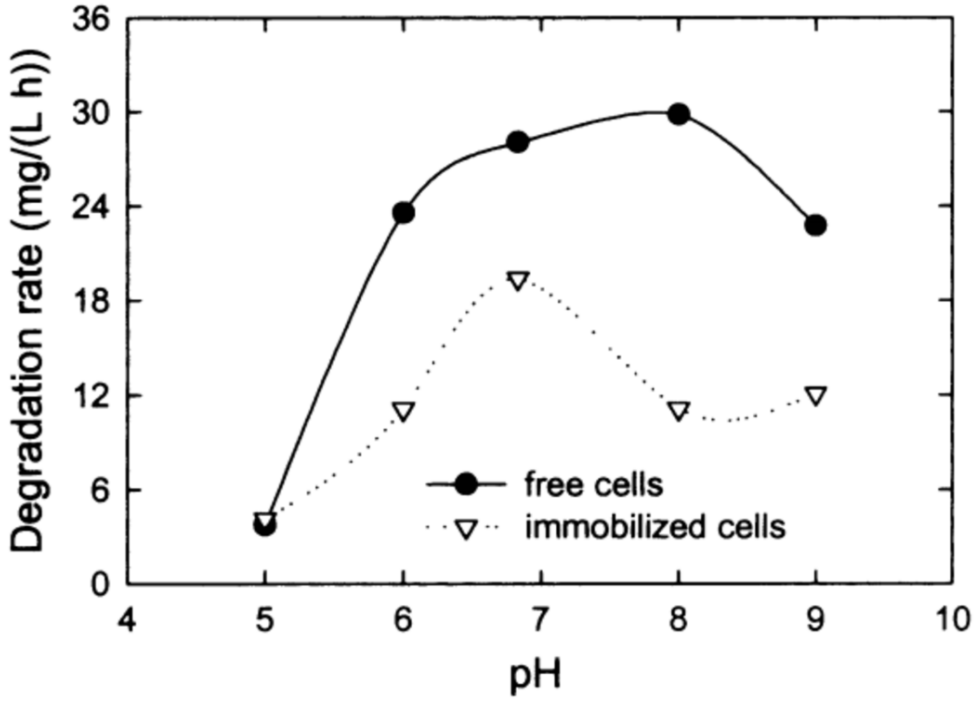


**Fig. (3):** Relationship between phenolic compounds degradation and pH values for immobilized and free cells (Chung, et al., 2003).

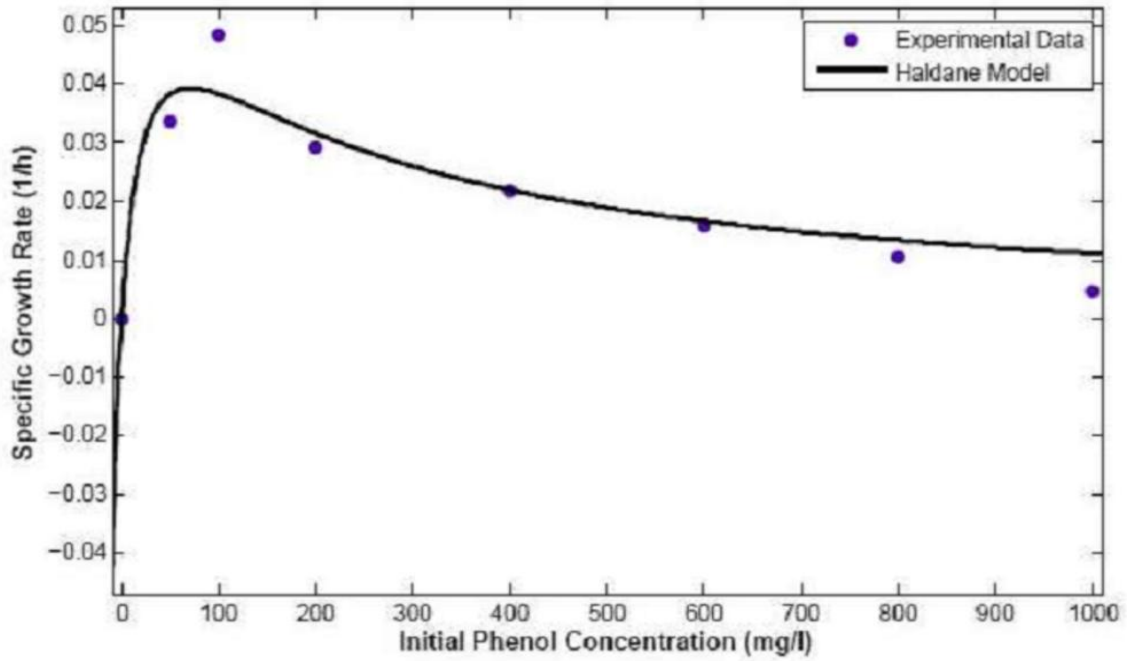


**Fig. (4):** Relationship between initial phenol concentration and specific growth rate due to Haldane's growth equation (Firozjaee, et al, 2011)

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**Table (1)**: Kinetic parameters of phenol biodegradation with pure culture microorganisms.

| Pure culture | Process conditions | | | Kinetic coefficients | | | References |
|---|---|---|---|---|---|---|---|
| | Max. phenol, mg/ L | Temp., °C | pH | $\mu_{max}$, 1/h | *Ks*, mg /L | *Ki*, mg/L | |
| Pseudomonas putida LY1 | 800 | 25 | 7.1-7.3 | 0.217 | 24.4 | 121.7 | Li Y. et al (2010) |
| Pseudomonas putida MTCC1194 | 1000 | 29.9 | 7.1 | 0.305 | 36.33 | 129 | Kumar et al. (2005) |
| Pseudomonas putidaCCRC 14365 | 600 | 30 | 6.8 | 0.330 | 13.9 | 669 | Chung et al. (2003) |
| Pseudomonas putidaCCRC 14365 | 400 | 30 | 7 | 0.245 | 12.1 | 1185.8 | Juang and Tsai (2006) |
| Pseudomonas putida Q5 | 200 | 10 | 7 | 0.119 | 5.27 | 377 | Kotturi et al.(1991) |
| Pseudomonas putida F1 ATCC 700007 | 50 | 30 | 7 | 0.051 | 18 | 430 | Abohamed et al.(2004) |
| Pseudomonas putida DSM 548 | 100 | 26 | 6.3-6.8 | 0.436 | 6.19 | 54.1 | Monteiro et al. (2000) |
| Pseudomonas putida | 800 | | | 0.9 | 6.93 | 284.3 | (Wang & Loh, 1999) |
| Yeast Candida tropicalis | 2000 | 30 | 6 | 0.48 | 11.7 | 207.9 | Jiang et al. (2005) |
| Bacillus brevis | 1750 | 34 | 8 | 0.026–0.078 | 2.2–29.3 | 868.0–2434.7 | Arutchelvan et al. (2006) |
| Alcaligenes faecalis | 1400 | 30 | 7.2 | 0.150 | 2.22 | 245.4 | Bai et al. (2007) |
| Ewingella americana | 1000 | 37 | 7.5 | 0.290 | 5.16 | 1033.7 | Khleifat (2006) |
| Acinetobacter | 350 | 30 | -- | 0.83 | 1.5 | 250 | (Hao, et al. (2002) |
| Trichosporon cutaneum R57 | 500 | 28-30 | 7.6 | 0.42 | 110 | 380 | Alexievaa et al. (2004) |

**Table (2)**: Kinetic parameters of phenol biodegradation with mixed culture microorganisms.

| Mixed culture | Process conditions | | | Kinetic coefficients | | | References |
|---|---|---|---|---|---|---|---|
| | Max. phenol, mg/ L | Temp., °C | pH | $\mu_{max}$, 1/h | *Ks*, mg /L | *Ki*, mg/L | |
| Aerobic | 2500 | Ambient | 6.5 | 0.438 | 29.5 | 72.4 | Marrot et al.(2006) |
| Aerobic | 1450 | 25 | -- | 0.143 | 87-45 | 107 | Nuhoglu and Yalchin (2005) |
| Aerobic | 800 | 21 | -- | 0.25 | 300 | 450 | Vázquez-Rodríguez et al. (2006) |
| Aerobic | 800 | 27 | 7 | 0.308 | 44.92 | 525 | Saravanan et al.(2008) |
| Aerobic | 659 | 25 | 7.2 | 0.309 | 74.6 | 648.1 | Bajaj et al. (2009) |
| Aerobic | 40 | 15 | -- | 0.258 | 3.9 | 121.7 | Buitron et al.(1998) |
| Aerobic | 500 | -- | -- | 0.542 | 36.2 | 145 | Kumaran & Paruchuri (1997) |
| Anaerobic | 10000 | 35 | -- | 0.027 | 0.03 | 363 | Suidan et al (1988) |
| Anaerobic | 1000 | 26 | 7 | 0.067 | 2524 | 200 | Firozjaee et al (2011) |