Article

Kinetic Biodegradation of Polycyclic Aromatic Hydrocarbons for Five Different Soils under Aerobic Conditions in Soil Slurry Reactors

Jeong Hyub Ha[†] and Suk Soon Choi*

Department of Integrated Environmental Systems, Pyeongtaek University, Pyeongtaek 17869, Republic of Korea
*Department of Biological and Environmental Engineering, Semyung University, Jecheon 27236, Republic of Korea

Abstract

In this study, soil slurry bioreactors were used to treat soils containing 16 polycyclic aromatic hydrocarbons (PAHs) for 35 days. Five different soil samples were taken from manufactured gas plant (MGP) and coal tar disposal sites. Soil properties, such as carbon content and particle distribution, were measured. These properties were significantly correlated with percent biodegradation and degradation rate. The cumulative amount of PAH degraded (P), degradation rate (K_m), and lag phase (λ) constants of PAHs in different MGP soils for 16 PAHs were successfully obtained from nonlinear regression analysis using the Gompertz equation, but only those of naphthalene, anthracene, acenaphthene, fluoranthene, chrysene, benzo[k]fluoranthene, benzo(a)pyrene, and benzo(g,h,i)perylene are presented in this study. A comparison between total non-carcinogenic and carcinogenic PAHs indicated higher maximum amounts of PAH degraded in the former than that in the latter owing to lower partition coefficients and higher water solubilities (S). The degradation rates of total non-carcinogenic compounds for all soils were more than four times higher than those of total carcinogenic compounds. Carcinogenic PAHs have the highest partitioning coefficients (K_{oc}), resulting in lower bioavailability as the molecular weight (MW) increases. Good linear relationships of K_m, λ, and P with the octanol-water partitioning coefficient (K_{ow}), MW, and S were used to estimate PAH remaining, lag time, and biodegradation rate for other PAHs.

Keywords: PAHs, MGP, Soil properties, Biodegradation, Soil slurry reactors, Nonlinear regression

1. Introduction

Coal tar waste is one of the primary byproducts of town gas produced at manufactured gas plants (MGPs). Improper disposal of coal tar from MGPs has resulted in the contamination of valuable land and aquifers. It is estimated that there are more than 5000 MGP sites in the United States that require remediation [1]. The primary constituents of coal tar waste are polycyclic aromatic hydrocarbons (PAHs), phenolic compounds, ammonia, and cyanide. Several treatment technologies, including biodegradation in soil slurry bioreactors, have been developed in the last decade [2–4]. The aerobic biodegradation of PAHs is well documented in the literature, and PAHs are reported to biodegrade via dioxygenase-, monooxygenase-, or peroxidase-catalyzed reactions [5, 6]. Low molecular weight PAHs containing two or three rings are readily degraded under aerobic conditions in the presence of appropriate organisms [7], while high molecular weight PAHs (four rings or more) degrade slowly and, in some situations, may be completely recalcitrant [8–10]. For comparison purposes, low molecular weight compounds, such as naphthalene and phenanthrene, degrade to

non-detectable limits within two–three weeks in soil slurry reactors [11,12], while it may take more than eight weeks in land farming systems [13,14].

Diverse kinetic bioremediation processes were performed under different experimental conditions [14–16]. The rates of naphthalene biodegradation in soil slurry reactors have been reported to be dependent on organic compound hydrophobicity, soil/water ratio, and soil organic carbon content [17]. Zero- and first-order rates, as well as Michaelis-Menten degradation rates were utilized to model the degradation of naphthalene in soil slurries [14], and an extended radial pore effective diffusion model was presented to describe the bioremediation of oil-contaminated soil in a slurry by dividing the soil into three fractions (clay, sand, and organic fractions), and first-order degradation kinetics [18]. Meanwhile, the relative degradation rates of 3- and 4-ring PAHs were compared with 5- and 6-ring PAHs at field and laboratory scales [9], and enhanced bioremediation of four-ring PAHs in marine sediment was examined under anoxic conditions [14].

Most of the information in the literature on soil slurry reactors concerns the biodegradation rate of one or two PAHs present in soils. Degradation kinetics of the US EPA 16 PAHs in a soil slurry reactor, however, are not well documented. The objectives of this study were to (1) estimate the kinetic parameters of 16 PAHs for the maximum degradation rate, lag time, and the maximum amount of PAH degraded using the nonlinear Gompertz equations for different coal-tar-contaminated soils in soil slurry reactors; (2) correlate degradation rates,

remaining PAH concentrations, and the lag time with various physical-chemical properties of PAHs and soils; and (3) investigate the controlling factors in soil slurry reactors.

2. Materials and Methods

2.1. Soil characteristics

Coal-tar-contaminated sites were obtained from several sites in Iowa, USA. Vandalia (EXC) soil was obtained from a former creek channel that was landfilled with MGP residuals. Vandalia (LTU) came from a land farming treatment unit containing a mixture of less contaminated soil and Vandalia (EXC) soil. Three other soil samples were obtained from former MGP sites (Charles City, Hampton, and Independence). Soil samples was homogenized by sieving through a 2-mm mesh sieve. Table 1 shows the physicochemical properties of the five soils. Soil was characterized using the particle size distribution test and the

USDA-modified soil texture triangle [19]. Soil organic carbon content was determined using the rapid dichromate oxidation procedure [20].

2.2. PAHs biodegradation of soil slurry reactor

Soil slurry biodegradation experiments were conducted in batch mode using 1-L glass reactors at a temperature of 21 ± 2 °C, and each reactor was loaded with approximately 150 g (dry weight basis) of the coal-tar contaminated soils, with 750 mL of distilled water and 20 mL of supernatant from a 2-L mother soil slurry reactor. The mother soil slurry reactor has been treating PAH-contaminated soil for several months by feeding approximately 300 g of Vandalia (EXC) soil in 1.5 L of water every 5 to 6 weeks. Nitrogen and phosphorus supplements were added in the form of ammonium nitrate and KH_2PO_4, respectively, in a C:N:P ratio of 100:10:2, where the carbon fraction was assumed to be the total carbon content of the 16 PAHs (see Figure 1) present in the soil. The reactors were mixed with a mixer (Model

Table 1. Physical-chemical Properties of Five Different MGP Soils

Parameters	Hampton, IA	Vandalia (EXC), IA	Charles City, IA	Vandalia (LTU), IA	Independence, IA
Oil texture [a]	Loam soil	Sandy loam	Sandy loam	Sandy clay loam	Sandy loam
Sand (wt%)	41	60	64	54	64
Silt (wt %)	35	26	18	24	25
Clay (wt %)	24	14	18	22	11
Organic carbon (wt %)	3.5	4.0	2.3	3.0	2.5
Soil moisture (wt %)	12.97	8.72	8.40	3.70	4.00
Soil pH	7.22	7.65	6.52	7.10	7.80

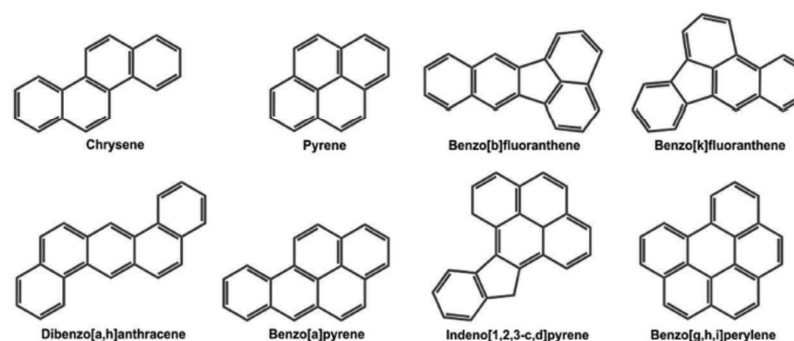[a] Based on USDA-modified soil texture classification
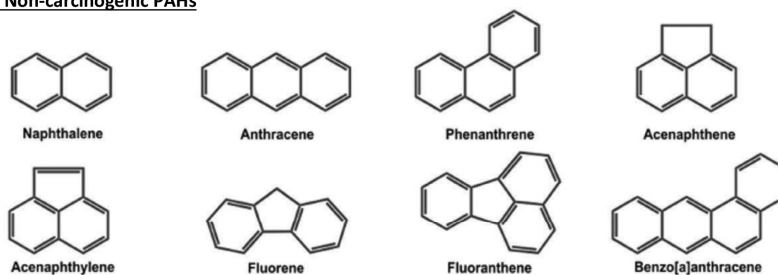
Figure 1. Molecular structure of the 16 polycyclic aromatic hydrocarbons (PAHs) selected priority pollutants by US EPA.

PM6015, EMI Incorporated, Clinton, CT) and aerated using an air stone diffuser at an airflow rate of 200 mL/min. The dissolved oxygen concentration was maintained above 3.0 mg/L and the pH of the slurry was between 6.5 and 8. A killed control with Vandalia (EXC) soil was also set up with 2% (w/w) sodium azide to inhibit microorganism growth and to monitor the loss of PAHs through abiotic processes, such as volatilization.

2.3. Analytical procedures

Soil slurries were removed from the soil slurry reactor at various time intervals and analyzed for 16 PAHs according to the extraction method [21]. A 10 mL slurry sample was placed in a 10 mL glass tube and centrifuged at 3,000 rpm for 30 min. The supernatant was discarded because preliminary tests showed that the aqueous phase contained negligible amounts of PAHs. Acetone (5 mL) was then added to the soil residue and the tube was shaken for 24 h. The suspension was centrifuged at 3,000 rpm for 30 min, and 5 μL of the supernatant was analyzed for the US EPA priority 16 PAHs by gas chromatography (GC) fitted with a flame ionization detector and equipped with a DB-624 capillary column from J&W Scientific. The injector and detector temperatures were 230 and 250 °C. The dry weight of the soil in the glass tube was determined by drying the soil sample at 105 °C for 24 h.

2.4. Nonlinear regression fitting to 16 PAHs

The nonlinear Gompertz equations [Equation (1)] were fitted to 16 PAH degradation data by nonlinear regression with the Marquardt–Levenberg algorithm [22]. The program automatically calculates starting values by searching for the steepest ascent of the curve (estimation of K_m) by intersecting this line with the x-axis (estimation of λ) and

by taking the final datum point as the estimation for the asymptote (P).

Since PAHs degrade exponentially, it is informative to plot the cumulative percentage of PAH degraded as a function of time. Three phases of the cumulative degradation curve can be described by three parameters: (1) the maximum degradation rate, K_m, is defined as the tangent at the inflection point; (2) the lag time, λ, is defined as the x-axis intercept of the tangent, and (3) the asymptote, P, is the maximum amount of PAH degraded.

$$y = p \cdot \exp\left\{-\exp\left[\left(\frac{Km \cdot e}{p(\lambda - t)}\right)\right] + 1\right\} \tag{1}$$

The Equation (2) is a search method to minimize the sum of the square errors (SSE) of the differences between the predicted and experimental values.

$$SSE = \sum (Q_{t \ exp} - Q_{t \ calc})^2 \tag{2}$$

3. Results and Discussion

3.1. Degradation of total cumulative PAHs, non-carcinogenic, and carcinogenic PAHs

The total cumulative 16 PAHs (T-PAHs), eight non-carcinogenic (N-PAHs), and eight carcinogenic PAHs (C-PAHs) degraded for five different soils are presented in Table 2. The experimental data were fitted using Gompertz equations, with dotted lines showing the model fitting results (Figure 2). In addition, the cumulative amount of PAH degraded (P), degradation rate (K_m), and lag phase (λ) constants of PAHs in different MGP soils for T-PAHs, N-PAHs, and C-PAHs were obtained from the nonlinear regression analysis; R^2 values are pre-

Table 2. Maximum Amount of P, K, and λ Constants of PAHs in Different MGP Soils

Compound	Soils	P (wt%)	K_m (day^{-1})	λ (day)	R^2
T-PAH[a]	Independence	82.23	0.11	0.00	0.973
	Charles City	91.97	0.11	0.00	0.981
	Vandalia (LTU)	80.91	0.11	0.00	0.983
	Hampton	81.75	0.09	0.81	0.975
	Vandalia (EXC)	85.08	0.07	1.12	0.970
N-PAH[b]	Independence	89.29	0.13	0.00	0.976
	Charles City	98.11	0.13	0.00	0.980
	Vandalia (LTU)	89.97	0.15	0.00	0.979
	Hampton	90.81	0.14	1.11	0.984
	Vandalia (EXC)	91.05	0.10	1.90	0.973
C-PAH[c]	Independence	52.33	0.03	0.72	0.985
	Charles City	50.57	0.05	8.70	0.987
	Vandalia (LTU)	54.57	0.05	1.91	0.971
	Hampton	40.14	0.04	8.47	0.969
	Vandalia (EXC)	55.31	0.03	7.42	0.959

[a] total 16 PAHs
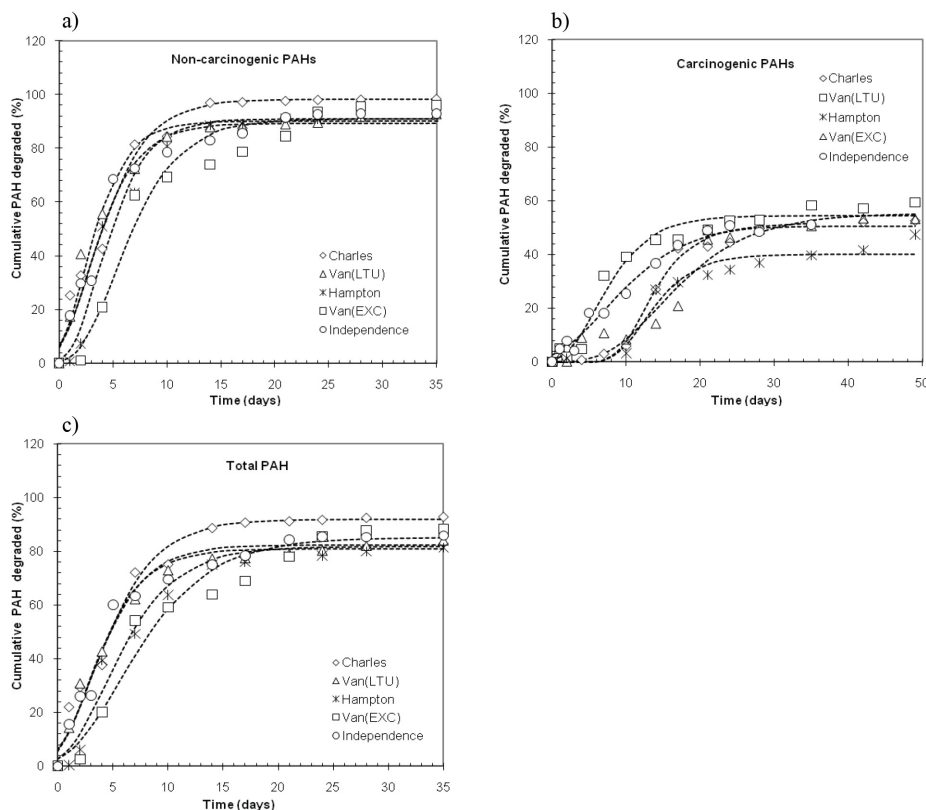[b] total eight non-carcinogenic PAHs
[c] total eight carcinogenic PAHs

Figure 2. Comparison of total cumulative a) non-carcinogenic, b) carcinogenic PAHs, and c)16 PAHs degraded for five different soils.

sented in Table 2 to evaluate the model and experimental data. The maximum amount of PAH degraded for T-PAHs was estimated to be 82, 92, 81, 82, and 85 % for soil samples from Independence, Charles City, Vandalia (LTU), Hampton, and Vandalia (EXC), respectively. First-order degradation rate constants for the T-PAHs were 0.11, 0.11, 0.11, 0.09, and 0.07 day^{-1} for soil samples from Independence, Charles City, Vandalia (LTU), Hampton, and Vandalia (EXC), respectively. Degradation rate constants were not statistically different for T-PAHs in soil samples with different properties from Independence, Charles City, and Vandalia (LTU), however, the degradation rate constants of T-PAHs in soils from Vandalia (EXC) and Independence were statistically different than other soils. In the case of Charles City soil, the cumulative amount of T-PAHs degraded was slightly higher than that in other soils. The lower clay and organic fractions in Charles City soil may result in T-PAHs being more readily available than other soils.

The biodegradation of the total cumulative eight N-PAHs and eight C-PAHs degraded for five different soils is shown in Figure 2. A comparison between the degradation of total N-PAHs and C-PAHs indicated that total N-PAHs showed a higher maximum amount of PAH degradation than total C-PAHs. There was a lag phase for five different soils on total C-PAHs, while a lag phase for total N-PAHs did not occur in soils from Hampton and Vandalia (EXC). The degradation rates of total N-PAHs were approximately four times higher than the total C-PAHs. For total C-PAHs in all soil samples, more than 50% of residual PAHs remained after 35 days (see Table 2). The high re-

sidual amount of total C-PAHs clearly illustrates the unavailable fraction of PAHs in the soil [12], which could be affected by bioavailability and biodegradability. The total C-PAHs have the highest partitioning coefficients, resulting in lower bioavailability as the molecular weight and number of rings increased.

3.2. Degradation of representative non-carcinogenic PAHs (2, 3, and 4-ring)

Table 3 shows kinetic parameters of representative non-carcinogenic PAHs of naphthalene anthracene, acenaphthene, and fluoranthene for all five soils, while representative data of the cumulative PAHs degraded for naphthalene and fluoranthene are presented in Figure 3. The biodegradation of naphthalene and other compounds of fluoranthene was not complete, with a residual fraction of about 0-5% and 4-20% remaining after 35 days, respectively. More than 95% of naphthalene was removed within four days for all four soils except Hampton soil, while a comparable amount of fluoranthene was removed after four days. This suggests that low molecular weight PAHs are susceptible to biodegradation in contaminated soil since PAHs with fewer benzene rings would have lower octanol-water partitioning coefficients (K_{ow}), higher water solubilities, and tend to be desorbed from organic matter in soils/sediments (i.e., more available as a carbon source for microorganisms). This result is consistent with high removal efficiencies resulting from the biodegradation of two- and three-ring PAHs in comparison to four- and five-ring PAHs showing little degradation [14].

Table 3. Maximum Amount of P, K_m, and λ Constants of Representative Non-carcinogenic PAHs in Different MGP Soils

Compound	Soils	P (wt%)	K_m (day^{-1})	λ (day)	R^2
Naphthalene	Independence	95.13	0.90	0.31	0.999
	Charles City	99.66	0.93	0.16	0.997
	Vandalia (LTU)	97.29	1.08	0.14	0.828
	Hampton	84.67	0.05	0.00	0.990
	Vandalia (EXC)	98.22	0.49	0.24	0.873
Anthracene	Independence	89.62	0.14	2.74	0.997
	Charles City	93.92	0.54	5.77	0.999
	Vandalia (LTU)	85.47	0.34	1.51	0.995
	Hampton	88.38	0.29	3.15	0.999
	Vandalia (EXC)	95.50	0.12	3.12	0.998
Acenaphthene	Independence	97.21	0.27	0.16	0.971
	Charles City	100.30	0.30	1.57	0.998
	Vandalia (LTU)	100.90	0.33	0.09	0.995
	Hampton	100.47	0.17	0.86	0.999
	Vandalia (EXC)	99.81	0.12	0	0.968
Fluoranthene	Independence	93.78	0.04	0.63	0.977
	Charles City	98.09	0.22	9.29	0.995
	Vandalia (LTU)	88.95	0.21	3.23	0.996
	Hampton	87.16	0.15	7.01	0.992
	Vandalia (EXC)	94.53	0.14	13.01	0.993
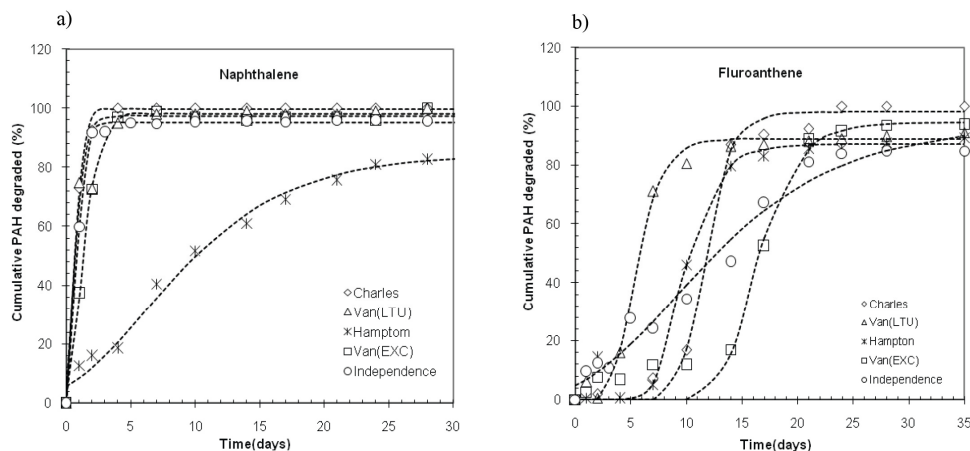
Figure 3. Representative a) naphthalene and b) fluoranthene of non-carcinogenic PAHs degradation for five different MGP soils.

Meanwhile, more than 80% of fluoranthene was removed within 7, 13, and 14 days for Vandalia (LTU), Charles City, and Hampton, respectively, but more than 20 days were required to achieve a similar removal percentage for Vandalia (EXC) soil containing the highest organic carbon content. The higher organic carbon content may result in the PAHs being less available than other soils [11].
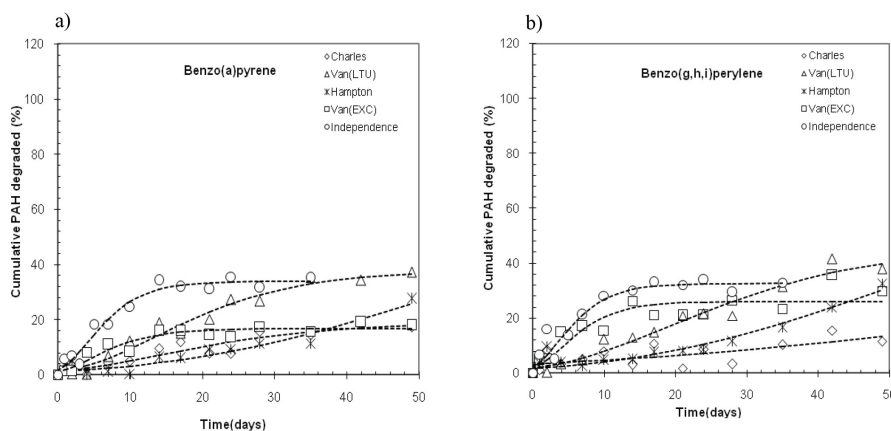
3.3. Degradation of representative carcinogenic PAHs (4, 5, and 6-ring)

Table 4 shows kinetic parameters of representative carcinogenic PAHs, including chrysene, benzo[k]fluoranthene, benzo(a)pyrene, and benzo(g,h,i)perylene for all five soils. Representative data of the cumulative PAHs degraded for benzo(a)pyrene and benzo(g,h,i)perylene for all five soils are presented in Figure 4. Since 4, 5, and 6-ring compounds have higher partition coefficients and lower water solubilities than 2 and 3-ring compounds, bioavailability may be a major factor that may limit their degradation. Benzo(a)pyrene and benzo(g,h,i)perylene for all soils showed differences in the extent of removal and the time needed to achieve similar removal percentages. While organic matter content has been shown to affect degradation rates of low molecular weight PAHs (*e.g.*, naphthalene and anthracene), this, however, has not been observed for benzo(a)pyrene and benzo(g,h,i)perylene [6,11].

Table 4. Maximum Amount of P, K$_m$, and λ Constants of Representative Carcinogenic PAHs in Different MGP Soils

Compound	Soils	P (wt%)	K$_m$ (day^{-1})	λ (day)	R^2
Chrysene	Independence	83.33	0.04	1.97	0.978
	Charles City	85.51	0.11	9.24	0.996
	Vandalia (LTU)	80.55	0.16	3.71	0.986
	Hampton	74.06	0.09	7.83	0.995
	Vandalia (EXC)	82.80	0.11	14.04	0.985
Benzo[k]fluoranthene	Independence	35.56	0.03	0.33	0.962
	Charles City	40.45	0.01	3.53	0.957
	Vandalia (LTU)	65.99	0.03	4.22	0.991
	Hampton	24.33	0.01	7.42	0.915
	Vandalia (EXC)	42.09	0.01	3.05	0.957
Benzo(a)pyrene	Independence	33.89	0.03	0.36	0.961
	Charles City	19.29	0.01	0	0.881
	Vandalia (LTU)	38.11	0.01	2.72	0.971
	Hampton	-	0.01	27.82	0.933
	Vandalia (EXC)	16.72	0.01	0	0.959
Benzo(g,h,i)perylene	Independence	32.55	0.03	0	0.921
	Charles City	18.11	0.01	0.76	0.615
	Vandalia (LTU)	46.55	0.01	3.25	0.963
	Hampton	-	0.01	16.56	0.923
	Vandalia (EXC)	25.99	0.02	0	0.823

Figure 4. Representative a) benzo(a)pyrene and b) benzo(g,h,i)perylene carcinogenic PAHs degradation for five different MGP soils.

The maximum amounts of PAHs degraded (P) for chrysene were estimated to be 83, 86, 81, 74, and 83 %, whereas those for benzo[k]fluoranthene were estimated to be 36, 41, 66, 24, and 42 % in soils from Independence, Charles City, Vandalia (LTU), Hampton, and Vandalia (EXC), respectively (see Table 4). As shown in Table 4, the degradation rates of chrysene for soils were found to be up to 10-times faster than for benzo[k]fluoranthene. The percentages of benzo(a)pyrene and benzo(g,h,i)perylene degraded after 35 days were approximately 34, 19, 38, and 17% and 32, 18, 47, and 26% in soils from Independence, Charles City, Vandalia (LTU), and Vandalia (EXC), respectively (Table 4). The percentage values for benzo(a)pyrene and benzo(g,h,i)perylene in Hampton soil are not shown in Table 4 because of

poor fitting results. The kinetic model results in Table 4 indicate that increases in ring size and molecular weight appear to be more closely related to a decrease in biodegradation rate and an increase in the percentage of PAH remaining. This observation is consistent with results that show high molecular weight PAHs are highly recalcitrant compounds and are difficult to degrade [4,13].

3.4. Correlations of R, λ, and K$_m$ with K$_{ow}$, MW, and S

For five different soils, correlations using the regression results for % PAH remaining (R), biodegradation rate (K$_m$), and lag time (λ) with water solubility (S), molecular weight (MW), octanol-water partitioning coefficient (K$_{ow}$) are presented in Table 5. Table 5 illustrates

Table 5. Correlation of R (% PAH remaining), K$_m$ (100 x biodegradation rate, the percent improvement), and λ (lag time) with K$_{ow}$, MW (molecular weight), and S (solubility) in Different MGP Soils

	Log (K$_m$) versus Log (K$_{ow}$, MW, S)	R^2	Log (R) versus Log (K$_{ow}$, MW, S)	R^2	Log (λ) versus Log (K$_{ow}$, MW, S)	R^2
Independence	Log K$_m$ = -1.88 Log K$_{ow}$ + 6.91	0.72	Log R = 1.91 Log K$_{ow}$ + 2.82	0.74	Log λ = -0.45 Log K$_{ow}$ + 5.12	0.04
	Log K$_m$ = -0.18 Log MW + 2.49	0.80	Log R = 0.18 Log MW + 2.10	0.77	Log λ = -0.03 Log MW + 2.31	0.02
	Log K$_m$ = 2.64 Log S − 3.70	0.76	Log R = -2.58 Log S + 1.92	0.72	Log λ = 0.54 Log S − 1.19	0.03
Charles City	Log K$_m$ = -1.11 Log K$_{ow}$ + 6.36	0.40	Log R = 0.75 Log K$_{ow}$ + 4.54	0.57	Log λ = -0.26 Log K$_{ow}$ + 5.07	0.03
	Log K$_m$ = -0.11 Log MW + 2.43	0.45	Log R = 0.07 Log MW + 2.26	0.65	Log λ = -0.01 Log MW + 2.31	0.01
	Log K$_m$ = 1.59 Log S − 2.97	0.44	Log R = -1.09 Log S − 0.35	0.65	Log λ = 0.16 Log S − 1.08	0.01
Vandalia (LTU)	Log K$_m$ = -1.42 Log K$_{ow}$ + 6.64	0.76	Log R = 0.69 Log K$_{ow}$ + 4.43	0.32	Log λ = 0.25 Log K$_{ow}$ + 5.18	0.02
	Log K$_m$ = -0.13 Log MW + 2.45	0.78	Log R = 0.07 Log MW + 2.25	0.35	Log λ = 0.03 Log MW + 2.32	0.05
	Log K$_m$ = 1.88 Log S − 3.18	0.70	Log R = -0.99 Log S − 0.20	0.35	Log λ = - 0.60 Log S − 1.29	0.07
Hampton	Log K$_m$ = -1.12 Log K$_{ow}$ + 6.02	0.40	Log R = 0.67 Log K$_{ow}$ + 4.16	0.26	Log λ = 0.95 Log K$_{ow}$ + 4.39	0.59
	Log K$_m$ = -0.09 Log MW + 2.38	0.30	Log R = 0.06 Log MW + 2.22	0.28	Log λ = 0.10 Log MW + 2.24	0.75
	Log K$_m$ = 1.37 Log S − 2.25	0.32	Log R = -0.99 Log S + 0.18	0.30	Log λ = -1.36 Log S − 0.17	0.64
Vandalia (EXC)	Log K$_m$ = -1.34 Log K$_{ow}$ + 6.46	0.45	Log R = 0.99 Log K$_{ow}$ + 4.14	0.50	Log λ = -0.26 Log K$_{ow}$ + 5.19	0.04
	Log K$_m$ = -0.13 Log MW + 2.44	0.46	Log R = 0.09 Log MW + 2.22	0.50	Log λ = -0.01 Log MW + 2.31	0.01
	Log K$_m$ = 1.71 Log S − 2.89	0.39	Log R = -1.37 Log S + 0.19	0.51	Log λ = 0.17 Log S − 1.26	0.01

that the octanol-water partitioning coefficients, molecular weight, and water solubility of 16 PAHs were strong predictors for % PAH remaining and biodegradation rates of 16 PAHs in the case of low soil organic carbon and clay contents. In contrast, as soil organic carbon and clay contents increased, the relationships between the octanol-water partitioning coefficients, molecular weight, and water solubility, the % PAH remaining, and biodegradation rates of 16 PAHs were poor because of isotherm nonlinearities and hysteresis. For example, the linear relationship of K$_{ow}$, S, and MW with R and K$_m$ in the case of Vandalia (EXC) and Hampton showed low R^2 values, while the linear relationship between K$_m$ and K$_{ow}$ for Vandalia (LTU) and Independence soils yielded high R^2 values. In addition, as the PAH ring size increased, the solid/water distribution ratio increased, leading to small fractions of PAHs remaining in the water phase, and the biodegradation rate and cumulative amount of PAH degradation also decreased. As shown in Table 5, the linear relationship of K$_m$ and R with K$_{ow}$, MW, and S obtained may be very useful in estimating the PAHs remaining and biodegradation rate for other PAHs when soil organic carbon and clay contents are low [6,21]. Meanwhile, the relationships of the octanol-water partitioning coefficients, molecular weight, and water solubility of 16 PAHs with lag time (λ) indicate low R^2 values and no correlations (Table 5).

4. Conclusions

The cumulative amount of PAH degraded (P), degradation rate (K$_m$), and lag phase (λ) constants of PAHs in different MGP soils for total PAH, N-PAH, and C-PAH were successfully obtained from the non-linear regression analysis. Comparison of the degradation between total non-carcinogenic and carcinogenic PAHs indicated that total non-carcinogenic PAHs displayed a higher maximum amount of PAH degradation than total carcinogenic PAHs due to lower partition coefficients and higher water solubility. Meanwhile, the total carcinogenic PAHs showed a longer lag time (λ) than the total non-carcinogenic PAHs. The degradation rates of total non-carcinogenic compounds for all soils were found to be approximately four times faster than those of the total carcinogenic compounds. Although the compositions of the polluted soil are usually very complicated and heterogeneously composed of unknown organic and inorganic chemical components, the three parameters obtained from nonlinear regression analysis may be useful for characterizing 16 PAHs in different soils. In addition, the linear relationships of S, MW, and K$_{ow}$ with K$_m$, λ, and % PAH remaining were obtained to predict the biodegradation rate and PAH degradation of other PAHs. Finally, the kinetic parameters obtained from these nonlinear regression results may be useful for designing reactor systems and selecting technologies to remediate other PAHs.

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Authors

Jeong Hyub Ha; Ph.D., Professor, Department of Integrated Environ-mental Systems, Pyeongtaek University, Pyeongtaek 17869, Republic of Korea; jhha@ptu.ac.kr

Suk Soon Choi; Ph.D., Professor, Department of Biological and Environmental Engineering, Semyung University, Jecheon 27236, Republic of Korea; sschoi@semyung.ac.kr