1. Introduction

Quaternary ammonium compounds (QACs) have been widely used as cationic surfactants for fabricating industrial products, such as antimicrobial soap, disinfectants, detergents, fabric softeners, preservatives, and pest control products since 1935 [1]. Among the 430 disinfectants listed by the US EPA, which included chlorine, H₂O₂, and QACs, QACs accounted for 50% of the reported disinfectants to date [2]. In addition, QACs have been recognized as capable of inactivating corona viruses.

Benzalkonium chlorides (BACs) are well-known QACs. BACs account for 95% of all QAC-based disinfectants reported to date [2]. BACs consist of a positively charged nitrogen atom linked to a long alkyl chain (typically 12–18 carbon atoms in length) on one side and a benzyl group on the other [3]. Among the BACs reported to date, benzyl dimethyl dodecyl ammonium chloride (BAC₁₂), benzyl dimethyl tetradecyl ammonium chloride (BAC₁₄), and benzyl dimethyl hexadecyl ammonium chloride (BAC₁₆) have been widely used because of their high bioavailability [4]. Also, BACs replaced well-known alternative disinfectants such as triclosan, triclocarban, benzethonium chloride, and chloroxylenol [5, 6].

Recent reports have indicated that the demand for BACs will increase during the severe acute respiratory syndrome-coronavirus-2 (SARS-CoV-2) pandemic [2] because BACs effectively inactivate the SARS-CoV-2 virus [7]. Therefore, high concentrations of BACs are using for laundry, sterilization, and hospital cleaning to prevent infections these days [8, 9]. The market report published before the SARS-CoV-2 pandemic indicated that the continuous growth rate of BACs is expected to reach 8.7% by 2025 [10]. However, BACs are known for their high toxicity values for the aquatic organisms. The EC₅₀ immobilization values of BAC for Daphnia magna was 16 μg L⁻¹ [11].

Previous studies reported that BACs can be frequently detected in the effluents of the wastewater treatment plants (WWTPs). The concentration of BACs in hospital wastewater effluents can range...
up to between 0.05 and 6.03 mg L\(^{-1}\) [12]. BACs in the influents and effluents of WWTPs also have been detected in the range of 25-300 and 2.1-4.1 μg L\(^{-1}\) respectively [13, 14]. As BACs usage has increased after the pandemic, it is predicted that the detection levels of BACs will be significantly increasing.

Municipal WWTPs can degrade up to 97% of BACs during the activated sludge process and reduce the content of BACs to parts per billion levels [13]. However, owing to their high toxicity, BACs are known to inhibit biological treatment processes at high concentrations such as parts per million levels (3 to 40 mg L\(^{-1}\)) [15, 16]. BACs are more toxic than the antimicrobials they replaced. A recent report has indicated that BACs exhibited acute lethal toxicity even at environmentally relevant levels (hundreds of μg L\(^{-1}\)) [17]. In addition, the predicted non-effective concentration of BACs has been reported to be 0.0014 μg L\(^{-1}\) [18]. Consequently, BACs pose high environmental risks even at low concentrations.

As chemical treatment options, advanced oxidation processes (AOPs) have been applied to treat BACs before the discharge of BACs containing wastewaters. Among AOPs, H\(_2\)O\(_2\)/O\(_3\), UV/C\(_\alpha\), O\(_3\)/H\(_2\)O\(_2\), and O\(_3\)/Cl\(_2\) reactions have been studied [19-22]. However, the specific facilities and additional oxidants are needed for the implementation of these AOPs in the fields. Instead, another option as a physical removal process, the adsorption process can be easily implemented and does not require specific facilities. Previous reports have described the adsorptive removal of environmentally relevant level BACs using municipal sludge and natural clays [23-26]. However, to our best knowledge, there are no studies on the adsorption removal containing BACs, especially BAC mixtures (BAC\(_{12}\), BAC\(_{14}\), and BAC\(_{16}\)) using commercially available adsorbents such as powered activate carbon (PAC).

In this study, we investigated the adsorption mechanisms of the most widely used BACs, namely BAC\(_{12}\), BAC\(_{14}\), and BAC\(_{16}\) using commercially available PAC to tackle the increasing use of BACs during the COVID-19 pandemic. The effects of time, pH, and temperature on the adsorption kinetics of each BAC were examined along with the kinetic isotherm and thermodynamic analysis. Also, the competitive adsorption kinetics of binary and ternary BAC mixtures (BAC\(_{12}\), BAC\(_{14}\), and BAC\(_{16}\)) were performed to compare the relative removal of each BAC by PAC. In addition, Microtox toxicity test was conducted to ensure that BAC detoxification is achieved after the adsorption process.

2. Materials and Methods

2.1. Materials

BAC\(_{12}\) (benzyl dimethyl dodecyl ammonium chloride), BAC\(_{14}\) (benzyl dimethyl tetradecyl ammonium chloride), BAC\(_{16}\) (benzyl dimethyl hexadecyl ammonium chloride), sodium phosphate monobasic, sodium phosphate dibasic, phosphoric acid (H\(_3\)PO\(_4\)), ammonia solution, ammonium chloride, and acetic acid were purchased from Sigma-Aldrich (St. Louis, MO, USA). Subscript numbers in BAC represent the chain length of each BAC. All chemicals were of analytical grade and were used as received without further purification. Synthetic wastewater including buffer solutes and BACs was prepared using deionized water. Coconut shell-based PAC was provided by SK carbon (Yangsan, Korea). PAC was applied to adsorption experiments without further treatment. The physicochemical properties of BACs are summarized in Table S1 in the Supporting Information.

2.2. Adsorption Experiment

PAC was washed with distilled water three times to remove any impurities prior to the adsorption experiment. Thereafter, the washed PAC was filtered using a 0.45 μm membrane filter (ADVANTEC, Tokyo, Japan), followed by drying in an oven at 80°C for 12 h.

To examine the adsorption kinetics and isotherm, we conducted the adsorption experiment in a jar tester. 0.45 g of freshly prepared PAC was added to 400 mL of deionized water containing BACs in 0.01 M phosphate solution. To obtain accurate kinetic data and experimental tractability, we used the concentration of BACs in the range between 200 and 450 mg L\(^{-1}\), which is below the critical micelle concentration (CMC) of BAC (1.320 mg L\(^{-1}\)) [27]. It is because the concentration of BAC solution for laundry and sterilization (0.7-7%) is much higher than CMC [28, 29]. The pH was fixed to 7 using a 0.01 M phosphate buffer and the temperature was fixed to 25°C. To investigate the effect of pH, the pH of the solution was adjusted in the range of 4 to 8.5 using 0.01 M phosphate buffer and 0.01 M ammonia buffer solutions. The agitating speed was set at 50 rpm, and the temperature was remained at 4, 25, and 35°C to understand the mechanisms of BACs onto PAC by obtaining the thermodynamic parameters. The relative humidity during the experiments was maintained 50 ± 5% by using the isothermal-isohumidity air conditioning system.

To examine the competitive adsorption of BACs, we conducted experiments using three binary BAC combinations (BAC\(_{12}\) & BAC\(_{14}\), BAC\(_{12}\) & BAC\(_{16}\), and BAC\(_{14}\) & BAC\(_{16}\)) and a ternary BAC mixture (BAC\(_{12}\) & BAC\(_{14}\) & BAC\(_{16}\)). The concentration of each BAC in the binary and ternary mixtures was set at 100 mg L\(^{-1}\); therefore, the total concentration of BACs in the binary and ternary mixtures was 200 and 300 mg L\(^{-1}\), respectively.

2.3. Analysis

All samples were filtered prior to analysis to remove PAC using a 1 mL syringe (Korea vaccine, Ansan, Republic of Korea) and a 0.2 μm membrane filter (ADVANTEC, Tokyo, Japan).

The concentration of BACs was determined using a high-performance liquid chromatography (HPLC) system equipped with a UV detector (Ultimate™ 3000, Dionex, Sunnyvale, CA, USA). The UV wavelength was set at 262 nm. A Luna® 5 μm C18 column (Phenomenex, 150 mm 4.6 mm, 100 Å) connected to a guard column (Phenomenex, 4 mm 3.0 mm) was used to separate the BACs. A mixture of acetonitrile and 0.2 M ammonium acetate (pH = 5, adjusted with H\(_3\)PO\(_4\)), ammonia solution, and acetic acid were purchased from Sigma-Aldrich (St. Louis, MO, USA). Subscript numbers in BAC represent the chain length of each BAC. All chemicals were of analytical grade and were used as received without further treatment.
A surface area analyzer (ASAP 2420, Micrometrics, Norcross, GA, USA) and the Brunauer-Emmett-Teller method were used to determine the surface area of PAC. The point of zero charge (pH_{pzc}) of PAC was evaluated using the pH drift method (Faria et al., 2004). The zeta potential of PAC was evaluated using a zeta analyzer at pH 7 (ELSZ-1000, Otsuka Electronics, Osaka, Japan). The physicochemical properties of PAC are summarized in Table S2. The surface area, pore-volume, and pore size of PAC were 945 m^2 g^{-1}, 0.52 cm^3 g^{-1}, and 21.9 Å (21.9 × 10^{-10} m), respectively. The zeta potential and pH_{pzc} of PAC were -26.49 mV at pH 7 and 6.48, respectively. This result indicates that the surface charge of PAC is negative at pH higher than 6.48.

2.4. Adsorption Isotherms

The Langmuir model represents the equilibrium distribution of ions between solid and liquid phases [30]. The Langmuir model is based on several assumptions; the adsorbed molecules do not interfere with each other, adsorption occurs via the same mechanism, and molecules adsorb onto the adsorbent outer surface as a homogeneous layer [25]. The equations for the Langmuir model can be expressed in Eq. (1) and (2);

\[ q_e = q_{\text{max}} \frac{K_L C_e}{1 + K_L C_e} \]  
\[ \frac{C_e}{q_e} = \frac{1}{q_{\text{max}}} + \frac{1}{K_L q_{\text{max}}} \times C_e \]  

where \( K_L \) (L mg^{-1}) is the Langmuir constant related to the adsorption capacity (mg g^{-1}) and \( q_{\text{max}} \) (mg g^{-1}) is the maximum adsorption capacity. \( K_L \) and \( q_{\text{max}} \) can be obtained from the slope and y-intercept, respectively, of the linear regression of \( C_e \) vs. \( C_e/q_e \).

The Freundlich isotherm model is used to describe adsorption on heterogeneous surfaces. The Freundlich model assumes that multilayer adsorption occurs rather than single-layer adsorption [31]. Heterogeneous adsorption occurs when the adsorbate enters the adsorbent and is subsequently adsorbed. Theoretically, the adsorbed amount increases continuously; therefore, an adsorption equilibrium cannot be attained [32]. The Freundlich model equation can be expressed as Eq. (3) and (4);

\[ q_e = K_F \times C_e^{1/n} \]  
\[ \log(q_e) = \log(K_F) + \frac{1}{n} \log(C_e) \]  

where \( K_F \) ((mg/g)/(mg/L)^{1/n}) is the Freundlich constant and \( n \) is the adsorption intensity. \( K_F \) and \( n \) can be obtained from the slope and y-intercept, respectively, of the linear regression of \( \log(C_e) \) vs. \( \log(q_e) \).

2.5. Acute Toxicity Test

The acute toxicity of the BACs was examined using a standard Microtox® bioassay by determining the changes in the luminescence of \textit{Vibrio fischeri}. The samples were adjusted using an osmotic adjusting solution (2% NaCl) to maintain an appropriate osmotic pressure for the luminescent bacteria. The pH of each sample was maintained at pH 7 using 10 mM phosphate buffer. The initial luminescence was determined using 0.5 mL of diluent and 10 μL of a solution containing \textit{V. fischeri}. We measured the change in the luminescence after injecting 0.5 mL of sample into the diluent containing \textit{V. fischeri} and waiting for 5 min.

3. Results and Discussion

3.1. Adsorption Kinetics

The adsorption experimental results showed that, within the first 10 min of reaction, approximately 80% of the BACs in the aqueous solution was removed by PAC, and were almost completely removed within 120 min (Fig. 1). This result indicates that PAC is suitable for BACs removal in the aspect of time efficiency. In fact, agricultural soils only achieved 20–50% and 30–80% removal of BAC12 and BAC14 in 24 h, and biodegradation treatment took 19 days to eliminate 100% of BACs [33].

The adsorption kinetic results were fitted using two kinetic models [34, 35]. The pseudo-first order (PFO) and pseudo-second order (PSO) kinetic models can be applied in the adsorption kinetics as shown in Eq. (5) and (6), respectively;

\[ \ln(q_e - q_t) = \ln(q_e) - k_1 t \]  
\[ \frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{t}{q_e} \]  

where \( t \) (min) is the time, \( q_t \) (mg g^{-1}) is the amount of BAC adsorbed at time \( t \), which can be measured during the kinetic experiment, and \( k_1 \) (min^{-1}) and \( k_2 \) (g mg^{-1} min^{-1}) are the rate constants for the PFO and PSO kinetic models, respectively. The adsorption kinetics results were better fitted to the PSO kinetic model (\( R^2 > 0.99 \)) than the PFO kinetic model (\( R^2 > 0.88 \)) (Table 1).
the same amount of PAC (1.125 g L–1) was summarized in Table 2. The adsorption of BACs, and in other studies, the Langmuir model was also suitable for the homogenous distribution of active sites on the PAC surface [36].

Fitted to the Langmuir model. This result might be due to the 0.81 and 0.95, indicating that the adsorption kinetics were better those between the low-molecular-weight BACs and PAC. Ismail et al. [23] reported similar results for the simultaneous adsorption of dodecyl trimethyl ammonium chloride (C12TMA) and BAC16 using primary sludge; the adsorption capacity of C12TMA decreased up to 52.5%. However, the adsorption capacity of BAC16, which has a longer alkyl chain length and higher molecular weight than C12TMA, decreased slightly (< 4.3%).

### 3.2. Adsorption Isotherms

Adsorption isotherm experiments using BAC12, BAC14, and BAC16 were conducted in the concentration range of 200–450 mg L–1 using the same amount of PAC (1.125 g L–1) at pH 7 and 25°C. The equilibrium time was 120 min. The calculated isotherm model parameters, namely K0, qmax, Kf, and n values, and R2 values are summarized in Table 2. The R2 values for the Langmuir model were > 0.99 and those for the Freundlich model ranged between 0.81 and 0.95, indicating that the adsorption kinetics were better fitted to the Langmuir model. This result might be due to the homogenous distribution of active sites on the PAC surface [36]. In other studies, the Langmuir model was also suitable for the adsorption of BACs, and qmax varied according to the type of adsorbent used. When agricultural soil (BAC12: 6.3–35.7 mg g–1 and BAC14: 8.8–38.9 mg g–1) and sewage sludge (BAC12: 67 mg g–1 and BAC14: 73 mg g–1) were used as adsorbents for BAC removal, however, the qmax values obtained soil and sludge were lower than those reported herein [37, 38]. This is because PAC has more active sites that can react with adsorbents compared to soil and sludge.

### Table 2. Langmuir and Freundlich Model Parameters Obtained in This Study (pH = 7.0 ± 0.1, temperature = 25°C, contact time = 120 min, [PAC] = 1.2 g L–1)

<table>
<thead>
<tr>
<th>Isotherm model</th>
<th>BAC</th>
<th>Parameters</th>
<th>R2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langmuir</td>
<td>BAC12</td>
<td>qmax = 188.68 mg g–1</td>
<td>K0 = 4.89 L mg–1</td>
</tr>
<tr>
<td></td>
<td>BAC14</td>
<td>qmax = 203.25 mg g–1</td>
<td>K0 = 6.97 L mg–1</td>
</tr>
<tr>
<td></td>
<td>BAC16</td>
<td>qmax = 206.19 mg g–1</td>
<td>K0 = 8.10 L mg–1</td>
</tr>
<tr>
<td>Freundlich</td>
<td>BAC12</td>
<td>n = 21.41</td>
<td>Kf = 144.84 (mg/g)/(mg/L)n</td>
</tr>
<tr>
<td></td>
<td>BAC14</td>
<td>n = 6.77</td>
<td>Kf = 119.84 (mg/g)/(mg/L)n</td>
</tr>
<tr>
<td></td>
<td>BAC16</td>
<td>n = 15.33</td>
<td>Kf = 139.35 (mg/g)/(mg/L)n</td>
</tr>
</tbody>
</table>

Interestingly, parameters such as K0 and qmax increased as the alkyl chain length increased. This result indicates that the amount of BAC16 adsorbed onto PAC was the highest, followed by the amounts of BAC14 and BAC12, implying that the effective adsorption process can occur as increasing the hydrophobicity of BAC molecules.

### 3.3. Competitive Adsorption Kinetics

In the water environment, more than two BACs can be simultaneously present in water [14, 18, 39, 40]. Therefore, it is important to examine the competitive adsorption of BACs. We conducted binary (BAC12 and BAC14, BAC12 and BAC16, and BAC14 and BAC16) and ternary (BAC12, BAC14, and BAC16) adsorption experiments (Figs. 2(a)–(d)).

The binary adsorption results showed that the adsorption removal rate of BACs with longer alkyl chains was higher than those of the BACs with shorter alkyl chains (Figs. 2(a)–(c)). For the binary adsorption of BAC12 and BAC14 by PAC for 120 min, the adsorption capacities of BAC12 and BAC14 were 64.0% and 9.2% lower than those achieved during the single adsorption of BAC12 (Fig. S1(a)) and BAC14 (Fig. S1(b)) by PAC, respectively. In addition, the binary experiment obtained using BAC12 and BAC16 revealed that the adsorption of BAC16 was more effective than that of BAC12; the adsorption capacities of BAC12 and BAC16 by PAC were 66.7 and 2.1% lower than those achieved during the single BAC12 (Fig. S1(c)) and single BAC16 (Fig. S1(d)) adsorption by PAC, respectively. Moreover, the binary experiment obtained using BAC14 and BAC16 indicated that the adsorption capacities of BAC14 and BAC16 were 48.3 and 24.3% lower than those achieved during the single BAC14 (Fig. S1(e)) and single BAC16 adsorption by PAC (Fig. S1(f)), respectively. The differences in adsorption capacity are summarized in Table S3.

For ternary adsorption experiment using three BACs showed the similar result (Fig. 2(d)). Fig. 2(d) showed that the removal of BAC16 is the highest, followed by BAC14, and BAC12. Fig. S2 also showed the reduction of adsorption capacity of each BAC between single and ternary conditions. The adsorption capacity of each BAC in ternary condition decreased compared to single BAC condition. BAC12 had the largest adsorption capacity reduction (76.0%), followed by BAC14 (52.0%), and BAC16(30.5%) at equilibrium. This result indicated that the van der Waals forces between the high-molecular-weight BACs and PAC were stronger than those between the low-molecular-weight BACs and PAC. Ismail et al. [23] reported similar results for the simultaneous adsorption of dodecyl trimethyl ammonium chloride (C12TMA) and BAC16 using primary sludge; the adsorption capacity of C12TMA decreased up to 52.5%. However, the adsorption capacity of BAC16, which has a longer alkyl chain length and higher molecular weight than C12TMA, decreased slightly (< 4.3%).

### 3.4. Effect of pH

Electrostatic interaction theory has been used to understand the adsorption of organic molecules, such as pharmaceuticals and personal care products by adsorbent [41, 42]. The electrostatic interaction between the adsorbent and contaminant can change upon changing the pH. A previous study reported that BACs adsorption
onto natural clays was controlled by ion exchange rather than van der Waals force at environmentally relevant concentration [39]. However, since BACs can be positively charged regardless of pH so that only the surface charge of PAC was changed with the pH [43] (Table S2). When the pH was lower than the pHpzc of PAC (6.48), PAC can be positively charged and when the pH was higher than the pHpzc of PAC, the adsorbent negatively charged. Therefore, electrostatic attraction between PAC surfaces and BACs can play an important role.

To investigate the effect of pH on the adsorption of BACs by PAC, we varied the pH of the reaction solution in the range of 4–10 using the buffer systems. The effect of pH on the \( q_e \) values of BACs is presented in Fig. 3, which illustrates that the \( q_e \) values of the BACs increased with increasing pH from 4 to 7 and finally it reached a plateau. At pH 4 and 5.5, which were lower than pHpzc of PAC, PAC is positively charged owing to the presence of H\(^+\) ions, thus PAC repelled the positively charged BACs, resulting in lowering \( q_e \) values. In contrast, at pH 7 and 8.5, the surface of PAC was negatively charged and \( q_e \) was higher than that of lower pH conditions due to the electrostatic interaction. Since PAC does not contain many functional groups prior to its chemical modification, we can assume that van der Waals interaction played a more important role in adsorption than chemical interactions.

Interestingly, while the \( q_e \) values of BAC14 and BAC16 did not change significantly with the pH, the change in \( q_e \) of BAC12 was more significant upon changing the pH (Fig. 3). This result implied that the electrostatic interaction is an important adsorption mechanism for BAC12, van der Waals interaction and hydrophobicity can play more important roles during the adsorption of BAC14 and BAC16 onto PAC.

3.5. Thermodynamic Analysis

Temperature is an important factor that can affect adsorption
kinetics. In this study, to investigate the temperature effect on the adsorption of BACs on PAC, adsorption experiments were performed at different temperatures (277 (4°C), 298(25°C), and 308 K(35°C)). Fig. 4 showed that, as the temperature increased from 277 to 308 K, the $q_{\text{max}}$ value of the BACs increased from 169.49–192.31 mg g$^{-1}$ to 232.56–264.62 mg g$^{-1}$. This result indicates that the adsorption of BACs on PAC was endothermic, implying that, as the temperature increased, BAC can move more freely and effectively attached to the active surface sites of PAC.

The thermodynamic parameters such as standard enthalpy change ($\Delta H^0$; kJ mol$^{-1}$), standard entropy change ($\Delta S^0$; J mol$^{-1}$·K$^{-1}$), and the resulting the standard Gibbs free energy change ($\Delta G^0$; kJ mol$^{-1}$), can be calculated using the temperature dependence of Langmuir equilibrium coefficient, as shown in Eq. (8) to (10).

$$\Delta G^0 = -RT \ln(K_L)$$  \hspace{1cm} (8)

$$\Delta G^0 = \Delta H^0 - T \Delta S^0$$  \hspace{1cm} (9)

$$\ln(K_L) = \frac{\Delta S^0}{R} - \frac{\Delta H^0}{RT},$$  \hspace{1cm} (10)

where $R$ is the universal gas constant (8.314 J mol$^{-1}$·K$^{-1}$), $T$ is the temperature (K), and $K_L$ is the Langmuir equilibrium coefficient (L mol$^{-1}$). $\Delta G^0$ was obtained using the experimental $K_L$ value. $\Delta S^0$ and $\Delta H^0$ were determined using the slope and y-intercept of the linear regression of $\ln (K_L)$ vs. $1/T$, respectively. The calculated values of the thermodynamic parameters are summarized in Table 3. The negative values of $\Delta H^0$ indicated that all adsorption reactions occur spontaneously. When the temperature increased from 277 to 308 K, $\Delta G^0$ was decreased significantly, indicating that the reaction proceeds more spontaneously at higher temperatures.

In gas adsorption systems, $\Delta H^0$ is usually negative (exothermic reaction). However, in liquid adsorption systems, $\Delta H^0$ can be either negative or positive [44]. The positive values of $\Delta H^0$ in this study indicate that the adsorption of BACs by PAC was endothermic. Furthermore, the $\Delta H^0$ values of the BACs obtained in this study (1,000–3,571 kJ mol$^{-1}$) indicated that adsorption is governed by physisorption, such as van der Waals forces (i.e., ~20 kJ mol$^{-1}$), and not chemisorption (i.e., 80–200 kJ mol$^{-1}$) [25].

Next, positive values of $\Delta S^0$ obtained in this study imply that the increasing randomness at the solid–liquid interface during the adsorption of BACs on PAC [45]. When BACs are added to water, the water molecules surrounding the BACs are detached and their degree of freedom increases [46]. Another reason for the increased randomness is that the PAC surface becomes more congested because large molecules are adsorbed on its surface [48]; BAC12, BAC14, and BAC16 have similar tendencies. These results imply that $\Delta S^0$ would increase when large molecules, such as BACs, are adsorbed on PAC. It is also reported that the removal reactions of heavy metals, dyes, and micropollutants using PAC were endothermic and presented positive $\Delta S^0$ values [47–49].

### 3.6. Detoxification of BACs

Finally, to assess the detoxification of the treated solution after the adsorption process, we conducted acute toxicity tests using a Microtox test. While 200 mg L$^{-1}$ of BACs were almost completely adsorbed by 1.2 g L$^{-1}$ of PAC (< 98%) within 120 min, up to 69% of detoxification was achieved after the adsorption reaction (Fig. S3). One-way of analysis variance (ANOVA) have conducted by
using R program (R Core Team, Vienna, Austria) to compare the toxicity responses before and after the BACs adsorption onto PAC. We observed statistical significance before and after the adsorption (p < 0.05). Even though the remaining concentration of BAC12, BAC14, and BAC16 in the treated solutions were 2.2, 1.4, and 0.2 mg L⁻¹, we observed remaining toxicity for V. fischeri. The reported 5 min EC₅₀ values of V. fischeri for BAC12, BAC14, and BAC16 were 0.19, 0.38, and 0.92 mg L⁻¹, respectively [4]. This result implies that, if we treat high concentrations of BACs in water, additional treatment or dilution strategy to reduce the BAC level is required to reduce the remaining acute toxicity.

The BAC adsorption mechanism by PAC was summarized in Fig. 5. The adsorption mechanism of BACs can be explained by physisorption mediated with van der Waals forces and hydrophobic interactions, and by the endothermic and spontaneous reactions, along with the reduction of the toxicity.

4. Conclusions

Kinetic studies revealed that high concentration of BACs was effectively adsorbed onto PAC and the PSO kinetic reaction model was more suitable to describe the adsorption process. Competitive BAC adsorption indicated that BACs with longer alkyl chains showed higher adsorption removal than BACs with shorter alkyl chains. The adsorption equilibrium results suggested that the adsorption of BACs on PAC fitted the Langmuir model (𝑅² > 0.99) better than the Freundlich model. The effect of pH was assessed, and the optimum adsorption pH was determined to be 7. The thermodynamic analysis demonstrated that the adsorption of BACs on PAC was spontaneous and endothermic. The adsorption reaction was governed by physisorption owing to the low Δ𝐻₀ value (< 20 kJ mol⁻¹). Since Δ𝐻₀ and Δ𝑆₀ were positive, the adsorption was an endothermic reaction. Finally, we observed 70% of BACs detoxification in the presence of PAC using V. fischeri.

In this study, we obtained the insight in which the adsorption process with PAC can be an effective option for treating BACs with the reduction of toxicity in real wastewater treatment facilities especially even high concentration levels. However, further study for process optimization is required to apply our findings with real wastewater containing BACs.

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Author Contributions

T.K.K. (postdoctor) conducted the experiments and wrote the manuscript. W.S.C. (master graduate) conducted the adsorption experiments. T.Y.K. (Ph.D. student) conducted the adsorption experiments. S.H.C. (Researcher) concepted and supervised the adsorption experiments. Y.S.H. (Researcher) concepted, supervised the toxicity experiments. K.D.Z. (Professor) concepted, supervised, wrote and revised the manuscript.

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