Utilization of activated carbon derived from waste plastic for decontamination of polycyclic aromatic hydrocarbons laden wastewater

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ABSTRACT

Serious environmental deterioration caused by synthetic waste plastics, and the pollution of freshwater resources are the most alarming and marked challenges of the 21st century. Therefore, immense scientific efforts are being made towards the management of waste plastics and treatment of polluted water. The current study reports on the utilization of waste polyethylene terephthalate (wPET) and waste polystyrene (wPS) for fabrication of activated carbon (AC) and its application for the removal of hazardous polycyclic aromatic hydrocarbons (PAHs) pollutants from water. AC was prepared from wPET and wPS by carbonization under a N2 atmosphere followed by chemical activation with 1 M KOH and 1 M HCl. The AC was characterized by scanning electron microscopy, surface area analysis, and Fourier transform infrared spectroscopy. Adsorption of PAHs from aqueous solutions through AC was examined by batch adsorption tests. The optimum parameters for maximum adsorption of PAHs were found to be: initial PAHs concentration 40 ppm, 2 h contact time, pH 3, 5, and 7, 50 °C temperature and adsorbent dose of 0.8 g. Kinetic and isotherm models were applied to evaluate the adsorbent capacity for PAHs adsorption. The kinetic study shows that the adsorption of these PAHs onto AC follows pseudo-second-order kinetics. The experimental results demonstrated that the Langmuir isotherm model best fitted the data. The thermodynamic factors calculated such as entropy change (ΔS°), enthalpy change (ΔH°) and free energy change (ΔG°) show that the adsorption process is non-spontaneous and endothermic in nature. Results were also compared with the efficiencies of some commercial adsorbents used in practice. This examination revealed that the novel plastic-derived AC possesses a great potential for elimination and recovery of PAH elimination from industrial wastewater.

Key words: activated carbon, adsorption, carbonization, polycyclic aromatic hydrocarbons, waste plastic, water treatment

HIGHLIGHTS

- AC was prepared from wPET and wPS by carbonization followed by chemical activation.
- Adsorption of PAHs from aqueous solutions through AC was examined by batch adsorption tests.
- PAH adsorption from the aqueous solutions follows pseudo-second-order kinetics.
- The experimental data are best fitted to the Langmuir isotherm model.
- AC possesses a large potential for elimination and recovery of PAHs from wastewater.
INTRODUCTION

PAHs are recalcitrant and toxic organic pollutants, and consist of two or more fused benzene rings in linear, cluster or angular arrangements, and are associated with low water solubility (Johnsen & Karlson 2005). The PAHs are released from anthropogenic sources as well as from some natural sources. Compared with anthropogenic sources, the natural sources of PAHs in the environment are insignificant (Witt 1995; Charlesworth et al. 2002). Various natural sources include oil seepages, thermal geological reactions, volcanic eruptions, and prairie and natural forest fires (Wright & Welbourn 2002; Qamar et al. 2017; Hidayat et al. 2019). Among the different anthropogenic sources accountable for the PAH emissions, residential waste burning, agricultural processing, vehicular emissions, military operations, coal tar production, asphalt and shale oil production, coal conversion, metal working, petroleum processing, steel and iron foundry works, aluminum smelting and coke production and incomplete combustion are notable (Ravindra et al. 2008; Hall et al. 2009). PAHs are commonly used in lubricating materials, wood preserving, agricultural and pharmaceutical products, resins, dyes, pesticides, and the manufacture of pigments (Abdel-Shafy & Mansour 2016). PAHs are ubiquitous in tar deposits, coal and oil, and PAHs mostly in the aquatic system have originated from contaminated sediment, atmosphere deposition, and accidental leakage (USEPA 1994; Blanchard et al. 2004).

PAHs can be categorized into two major groups, including high-molecular-weight PAHs which contain six, five or four benzene rings, and low-molecular-weight PAHs containing three or two benzene rings (Wenzl et al. 2006; Waqas et al. 2014). Owing to their mutagenic and carcinogenic characteristics, high lipophilicity, persistence, and hazardous nature, PAHs are of great concern in the aquatic system (Sweetman & Jones 2000; Hidayat et al. 2019). Generally, the carcinogenicity of PAHs increases with rising molecular weight (Eisler 1987). PAHs can cause both long and transient wellbeing impacts, the long-term wellbeing impacts include symptoms of asthma, inhalation difficulties, lung abnormalities, damage to liver and kidney, cataracts and a suppressed immune system (Abdel-Shafy & Mansour 2016). The short-term health effects of PAHs in humans are not clear, in any case occupational exposure to PAHs in high levels may bring about transient impacts, for example diarrhea, eye irritation, vomiting, confusion and nausea (Abdel-Shafy & Mansour 2016). Various PAHs have been found to occur in plants as well as animal tissues (USEPA 2008), water bodies (Chang et al. 2004a, 2004b; Balati et al. 2015), sediments and soil (Vane et al. 2014; Wang et al. 2015) and in air (Liu et al. 2015), which signifies their presence in the environment. Since PAHs are non-biodegradable and persistent, they consequently enter the food chain via soil and move to the human body by dermal contact, inhalation, or ingestion (Abdel-Shafy & Mansour 2016). Once entering the
body, PAHs can go crosswise over cell membranes and are effectively adsorbed into cells, where the immune system changes PAHs to epoxide hydrolase and diolepoxides, reacting with DNA and preventing its production (Kaya et al. 2013).

The European Union and USEPA have listed 16 PAHs among the concerned major pollutants. The occurrence and fate of major PAHs in conventional activated sludge in a wastewater treatment plant have been extensively studied by researchers (Manoli & Samara 1999). Lower levels of various PAHs have been found to occur in wastewaters treated by various processes (Busetti et al. 2006). Other mechanisms must be considered, such as air-stripping, chemical degradation, advection, or volatilization, which help to remove PAHs, depending on the system operating conditions, process design as well as wastewater characteristics (Manoli & Samara 1999). Over time, different methods have been developed to cater for the removal of PAHs from contaminated water as well as soil to reduce the possible risk of PAHs to human health and the environment. Phytoremediation, biological, thermal, chemical, and physical processes (including pyrolysis, pyrolysis, ion exchange, oxidation, radio frequency heating, thermal desorption, and incineration methods) are the main treatment techniques for PAH contaminated water, sediment, and soil (Zhou et al. 2013; Hu et al. 2014; Li et al. 2014; Peng et al. 2014; Sun et al. 2014; Wang et al. 2014; Li et al. 2015). Although, most of these strategies have some disadvantages, for example, complicated operating procedures, high maintenance as well as investment costs. In addition, some of these treatment techniques result in the production of secondary by-products and some of them are mutagenic (e.g., haloacetic acid, trihalomethanes) and carcinogenic, which further increases the adverse effects to public health (Shih & Lederberg 1976; Comminellis et al. 2008; Oller et al. 2011).

Among all the treatment methods that have been developed, adsorption is one of the most relevant and promising methods to remove organic and inorganic micro-pollutants (Saucier et al. 2017; Umpierres et al. 2018; Mironyuk et al. 2019; Tatchuck et al. 2019). This process has many advantages: (i) probability of adsorbent regeneration and reuse; (ii) environmentally benign; (iii) low investment cost; (iv) toxicity removal; (v) vigorous continuous and batch processes; (vi) handling microlevels of pollutants; and (vii) simple to operate and to design (Torrellas et al. 2016, Bhatnagar & Anastopoulos 2017). Various adsorbents used for this purpose include AC (Dabek & Ozimina 2009), zeolite (Chang et al. 2004a, 2004b), bottom ash (Gupta et al. 2012), Curcuma caesia-based AC (Arora et al. 2020), Chenopodium album ash (Arora et al. 2019a), ash from the black turmeric rhizome (Asha et al. 2021), iron-based metal organic frameworks (Arora et al. 2019b), Fe-benzene dicarboxylic acid metal organic frameworks (Son et al. 2020), solid residues from agricultural activities such as pine bark and ash waste (Li et al. 2010; Pérez-Gregorio et al. 2010), rice husk (Hidayat et al. 2019), sugar cane bagasse, green coconut shells, chitin, and chitosan (Crisafulli et al. 2008), as well as other organic-free hydrophilic minerals such as talc, silica and alumina (Su et al. 2006; Müller et al. 2007) and numerous materials such as plant-derived material, sewage sludge, shells, petroleum coke and bituminous coal have been used for the production of AC (Awoyemi 2011; Gupta 2015) through carbonization followed by activation processes. AC is produced depending on its particular purpose in the form of granules, powder, or briquette. The more common form of AC in practice includes powder and granular form for PAH sorption (Paune et al. 1998; Ania et al. 2007; Valderrama et al. 2008). The development of a low-cost and more efficient adsorbent has led to continued research interests in this field.

A tremendous increase in the usage of plastic over the past five decades has been observed worldwide. Due to its inappropriate dumping and waste management, the increased production and application of plastic is threatening the environment. The unintentional as well as unwarranted disposal of conventional plastic in water reservoirs has contributed to the decline in the population of marine organisms (O’Brine & Thompson 2010). Due to its non-biodegradable and stable chemical structure, plastics stay longer in the environment, constituting a large part of the solid waste. Some techniques have been suggested for handling the growing plastic solid wastes, such as its conversion into liquid or gaseous fuels, recycling, incineration and land filling, etc. (Ilyas et al. 2018). All these techniques have been critically reviewed, which revealed that each technique has its own shortcomings, such as in land filling, the non-biodegradable materials dumped on the land cause soil pollution, recycling is an expensive and energy-requiring process, whereas incineration and conversion involve the generation of more hazardous emissions, such as CO₂ and persistent organic pollutants (POPs) etc. (Ilyas et al. 2018). However, a more practical and environmentally friendly approach is the utilization of plastic wastes in mitigating water pollution by utilization in carbonized form as adsorbents for the elimination of different toxic substances from wastewater (Ikram et al. 2020).

In the current study, AC was derived from waste polyethylene terephthalate (wPET-AC) and waste polystyrene (wPS-AC) by carbonization followed by its chemical activation, and was utilized for the adsorption of PAHs from water. This study aimed to investigate the potential of utilizing waste PET- and PS-derived AC for the removal of various PAHs from wastewater as a
low-cost adsorbent and mitigate the issue of plastic solid waste management. The characterization of AC, optimization of adsorption parameters, adsorption kinetics, and adsorption isotherms have also been discussed.

MATERIALS AND METHODS

Sample collection
Standard solutions of PAHs Mix-13 (USEPA 16) consisting of benzo(a)anthracene (BaA; 10 μg/L), benzo(a)pyrene (BaP; 10 μg/L), benzo(k)fluoranthene (BkF; 10 μg/L), benzo(b)fluoranthene (BbF; 10 g/L), dibenzo(ah)anthracene (D[ah]A; 10 μg/L), benzo(ghi)perylene (B[ghi]P; 10 μg/L), indeno(123-cd)pyrene (Ind; 10 μg/L), pyrene (Pyr; 10 μg/L), chrysene (Chr; 10 μg/L), fluoranthene (Flu; 10 μg/L), acenaphthene (Ace; 10 μg/L), acenaphthylene (Acy; 100 μg/L), anthracene (Ant; 10 μg/L), phenanthrene (Ph; 10 μg/L), fluorene (Fl; 10 μg/L) and naphthalene (Nap; 100 μg/L) purity >99% was purchased from the Supelco Company, USA. On a daily basis, the working standard solutions were prepared by diluting the PAHs Mix-13 standard solution in 2-propanol. Thus, to avert photochemical degradation, the sample extracts and standard solutions were stored in brown glass vials at 4 °C for further study.

Synthesis of AC from waste plastic by carbonization
Samples of wPET and wPS were collected from street waste bins. The wPET and wPS samples were washed, dried, and shredded in to small pieces. The waste plastic samples were converted into AC by carbonization in a specially designed steel reactor under an inert atmosphere at high temperature. About 5 g of wPET or wPS were taken in the steel reactor and inserted in the tube furnace, the nitrogen supply was connected to the reactor and heated to 600 °C for one hour, as shown in Figure 1. The AC was scrapped from the reactor and activated chemically by treatment with 1 M KOH and 1 M HCl alternately using a procedure reported in the literature (Lian et al. 2011). The solid carbon was finally ground to a fine powder and passed through a sieve with pore size 250 μm and then stored in glass vials for further study.

Characterization of AC
The AC samples synthesized from wPET and wPS were characterized by various instrumental analyses. The surface morphology of AC was studied by a SEM (JSM-5910, JEOL, Japan). The analysis of the functional groups of AC was performed using FTIR (Spectrum Two, S.No: 103385) and the specific surface area was analyzed using a pore size and surface area analyzer (NOVA2200e, Quantachrome, USA). The zeta potential of the wPET-AC and wPS-AC was obtained using a Zeta Meter 3.0 (Zeta Meter Inc.).

Adsorption experiments
Adsorption of PAHs from water samples over wPET-AC and wPS-AC was studied in batch mode adsorption experiments. About 100 ml of water sample containing PAHs were taken into a 500 ml Erlenmeyer flasks at 24 ± 2 °C, definite amounts

Figure 1 | Diagrammatic presentation of the AC preparation in the laboratory.
of AC adsorbent were added to it and stirred on a magnetic stirrer with 150 rpm speed until equilibrium was reached. Several variables were measured in the study, including dosage of AC (0.2, 0.4, 0.6, and 0.8 g/100 ml), contact time (0.25, 0.5, 1 and 2 h), PAHs concentrations (10, 20, 30, and 40 ppm) and pH of solutions (3, 5, 7, 9, and 11), pH was adjusted by adding 0.1 M HCl or NaOH to the solution as required.

Analyses and calculations

The PAHs concentration in the treated samples was examined by using a UV spectrometer (Mansouri et al. 2020). Adsorption efficiency at equilibrium \( (q_e) \) was calculated as the following (Chang et al. 2004a, 2004b):

\[
q_e = \frac{[(C_0 - C_i) \times v]}{m}
\]

where, \( C_0 \) is the initial concentration of PAHs (μg/L), \( C_i \) is the remaining concentration of PAHs at equilibrium in the solution (μg/L), \( V \) represents the volume of the solution (L) and \( M \) is the AC mass (g).

Adsorption efficiency at a specific time \( (q_t) \) was calculated as the following (Crisafulli et al. 2008):

\[
q_t = \frac{[(C_0 - C_i) \times V]}{m}
\]

where, \( C_t \) is the liquid-phase concentration at that specific time \( t \) (μg/L). Percent removal efficiency \( (%MR) \) was calculated using the following relations (Björklund & Li 2015):

\[
%MR = \frac{[(C_0 - C_i) \times 100]}{C_0}
\]

The adsorption kinetics was investigated by applying pseudo-first-order (Ho & McKay 1998), pseudo-second-order (Chen et al. 2010), and intraparticle diffusion kinetic models (Weber & Morris 1963). Likewise, the adsorption mechanism was studied by applying Freundlich (Yao et al. 2014), and Langmuir (Lawrinenko 2014) isotherms to the adsorption data. The equations for these models are given in Table 1. All calculations were conducted using Excel 2010 software (Microsoft Office).

RESULTS AND DISCUSSION

Characterization of wPET-AC and wPS-AC

SEM analysis

The AC synthesized from wPET and wPS, i.e., wPET-AC and wPS-AC were characterized by SEM, FTIR, and SSA analysis. The SEM micrographs of wPET, wPET-AC, wPS and wPS-AC are given in Figure 2. In Figure 2(a) SEM images highlighting the porosity of wPET are shown. The surface features of wPET and the region of both external as well as internal structures and in addition the unfilled spaces between the particles can be seen (Ilyas et al. 2019; Ilyas et al. 2021). The SEM micrographs of wPET-AC (Figure 2(b)), show a stretched but uneven morphology, with some fine particles agglomerated on the surface. The surface also seems flaky, which indicates the highly porous structure of the AC. The micrograph of wPS displayed in Figure 2(c) shows a spongy structure with wide pores openings, formed from interconnected polymer films (Ilyas et al. 2019). The SEM micrograph of wPS-AC (Figure 2(d)) displays an irregular and rough morphology with numerous cavities and mounds on the surface. Some fine particles are agglomerated on the surface, these agglomerated particles may be spherical carbon black structures, whereas on the surface fissures and cracks are also visible. The SEM analysis suggests that wPET-AC exhibits better porosity than wPS-AC.

FTIR spectra

Figure 3 shows FTIR spectra of wPS, wPS-AC, wPET, and wPET-AC. The FTIR spectrum of wPS shows major peaks centered at around 3,000 and 2,800 cm\(^{-1}\) corresponding to aromatic C-H and methylene C-H bonds. Multiple peaks positioned at 1,650 and 1,500 cm\(^{-1}\) indicate aromatic C = C vibrations, and a peak around 1,300 cm\(^{-1}\) indicates methylene C-H vibration, all these configurations are in conformity with the structure of polystyrene (Herman et al. 2015; Subramani & Sepperumal
The FTIR spectrum of wPS-AC (Figure 3(a)), shows the major peaks positioned at 3,028 cm\(^{-1}\) (aromatic C-H, stretching), 1,599 cm\(^{-1}\) (aromatic C=C, stretching), 1,451 cm\(^{-1}\) (CH\(_2\) stretching), 1,017 cm\(^{-1}\) (C-O, stretching), 696 cm\(^{-1}\) (C-H, stretching) and 540 cm\(^{-1}\), which indicates that the resultant AC consists of aromatic functionalities. The characteristic infrared absorption bands of wPET (Figure 3(b)) at 1,696 cm\(^{-1}\) (C=O, stretch), 1,260 cm\(^{-1}\) (C=C, stretch), 1,092 cm\(^{-1}\) (C-O, stretching vibration), 1,010 cm\(^{-1}\) (C=O-C, stretching), 720 cm\(^{-1}\) (C-H, stretch) and 521 cm\(^{-1}\) (C-H, stretch) are clearly observed (Tantawy et al. 2014). The FTIR spectrum of wPET-AC (Figure 3(b)) showed that the wPET-AC had functional groups of wPET. The noticeable absorption peaks were at 3,414 cm\(^{-1}\) which might be due to stretching vibrations of –OH, 1,696 cm\(^{-1}\) (C=O, stretch), 1,260 cm\(^{-1}\) (C=C, stretch), 1,092 cm\(^{-1}\) (C-O, stretching vibration), 1,010 cm\(^{-1}\) (C=O-C, stretching), 720 cm\(^{-1}\) (C-H, stretch) and 521 cm\(^{-1}\) (C-H, stretch). These results suggest that the AC derived from wPET contains various oxy functionalities, which may have resulted from surface oxidation of AC.

### Surface area analysis

The surface characteristics such as total pore volume (V\(_t\)), average pore radius (R\(_p\)), BET surface area (S\(_{\text{BET}}\)), and BJH surface area (S\(_{\text{BJH}}\)) of the AC are shown in Table 2. Results show that wPET-AC exhibits S\(_{\text{BET}}\) of 65.35 m\(^2\) g\(^{-1}\), S\(_{\text{BJH}}\) of 11.02 m\(^2\) g\(^{-1}\), V\(_t\) of 0.009 cm\(^3\) g\(^{-1}\) and an R\(_p\) of 15.02 Å. In the case of wPS-AC, the S\(_{\text{BJH}}\) was found to be 14.68 m\(^2\) g\(^{-1}\), the S\(_{\text{BET}}\) 50.14 m\(^2\) g\(^{-1}\), with R\(_p\) of 14.09 Å and V\(_t\) of 0.010 cm\(^3\) g\(^{-1}\). It can be observed that the surface area of the wPS-AC is relatively smaller than that of wPET-AC, however, the surface areas of both the ACs are suitable for sufficient adsorption. The BET surface areas of the wPS-AC and wPET-AC were comparable with the literature reports, for example, the S\(_{\text{BET}}\) values obtained for bentonites by other researchers, which include 31.5 m\(^2\) g\(^{-1}\) reported by Xifang et al. (2007), 56 m\(^2\) g\(^{-1}\) obtained by Ding et al. (2009) and 34.1 m\(^2\) g\(^{-1}\) reported by Guerra et al. (2013), wPET-AC was found to have a higher surface area, which is desirable for efficient sorption.

### Table 1 | Adsorption models used in the study

| Kinetic models            | Equation                                                                 |
|---------------------------|---------------------------------------------------------------------------|
| Pseudo-first-order        | In (q\(_e\) - q\(_t\)) = ln q\(_e\) + k\(_{1t}\) where,                  |
|                           | - At equilibrium the quantity of PAHs adsorbed is presented by q\(_e\) (μg g\(^{-1}\)) |
|                           | - At time t the quantity of PAHs adsorbed is presented by q\(_t\), and     |
|                           | - The rate constant is given by k\(_1\) [1/h].                           |
|                           | t                                      |
|                           | q\(_t\)                                     |
|                           | q\(_e\)   + k\(_2\)q\(_e\)               |
|                           | where,                                     |
|                           | - K\(_2\) is the rate constant [g/(μg-h)] for second-order kinetics.     |
|                           | - At equilibrium the sorption capacity is presented by q\(_e\).           |
| Pseudo-second-order       | q\(_t\) = \frac{t}{q\(_e\)} + \frac{1}{K\(_2\)q\(_e\)}                   |
| Intraparticle diffusion   | qt = Kid(t)^0.5 + C                                                               |
|                           | where,                                     |
|                           | - Kid (h\(^{-1}\)) is rate of reaction for intraparticle diffusion process, and |
|                           | - C is the constant related to the thickness of boundary layer (μg g\(^{-1}\)). |
| Equilibrium models        |                                                                 |
| Freundlich                | In q\(_e\) = In K\(_f\) + \frac{1}{n}ln C\(_e\)                                      |
| Langmuir                  | C\(_e\) = \frac{1}{KQ\(_{\text{max}}\)} + \frac{C\(_0\)}{Q\(_{\text{max}}\)}    |
|                           | where,                                     |
|                           | - q\(_e\) represents the PAHs adsorbed concentration over the carbonaceous adsorbents (μg g\(^{-1}\)), |
|                           | - C\(_e\) is the PAHs equilibrium concentration (μg L\(^{-1}\)),            |
|                           | - While K\(_f\) and n represent the adsorption coefficient and constant for the model, respectively. |
The adsorption of various 16 PAHs was studied over AC derived from wPET and wPS from aqueous solution. Initially, the PAHs adsorptions over the AC examinations were carried out for 90 min contact time and adsorbent to solution ratio of 0.3 g/100 mL at a temperature of 24 ± 2 °C, the % adsorptions of 16 PAHs are given in Figure 4. Results indicate that the wPET-AC showed a greater tendency for the 16 PAHs adsorption than that of the wPS-AC. This is reliable with its larger average pore radius observed by BET analysis. In the case of wPET-AC the values of % adsorption of the 16 PAHs ranged from 53.80 to 92.44. The lowest value of 53.80% was noted for Nap while the highest value of 92.44% was noted for Pyr. Similarly, the adsorption values of these 16 PAHs ranged 44.3–82.3% for wPS-AC. The wPS-AC has shown the highest removal efficiency for B[ghi]P, while the lowest value for Nap from aqueous media. The data were compared with the previous work done by researchers for PAHs removal while utilizing AC. The adsorption test of B[ghi]P, Inp, BaA, BkF, BbF, Chr, BaP, Pyr, Fl, Ant and Ph was carried out by Amstaetter et al. (2012) using AC and achieved better adsorption efficiency for Ph (95%), Ant (97.5%), Flu (96.5%), Pyr (98%), BaP (88%), Chr (94.5%), BbF (86%), BkF (88%), BaA (94.5%), InP (73%), B[ghi]P (69%). Similarly, a high adsorption capacity using powder AC for PAHs (B[ghi]P (84%), Inp (85%), BaA (93%), BkF (92%), BbF (93%), Chr (95%), BaP (88%), Pyr (96%), Fl (95%), Ant (89%) and Ph (82%)) was attained from the contaminated soil (pore water) (Hale et al. 2012). Furthermore, Oleszczuk et al. (2012) studied the adsorption capacity of PAHs (B[ghi]P, BaP, Phr, Flu, Ph) from sewage sludge, which was 95% using AC. Hence, in the current study, the performance of the AC is also better. Further experiments were carried out under various conditions to investigate the most favorable adsorption factors.

Figure 2 | SEM images of (a) wPET, (b) wPET-AC, (c) wPS and (d) wPS-AC.
Effect of process parameters

After studying the efficiency of wPET-AC and wPS-AC, it was found that they showed better adsorption efficiency. In order to investigate the optimum conditions required for maximum adsorption of PAHs, the adsorption was investigated under different conditions of concentration of PAHs, pH, contact time, temperature, and adsorbent dose.

Effect of adsorbent dosage

The availability and accessibility of adsorption sites is represented by the quantity of adsorbent provided for adsorption (Abbas et al. 2013). The effect of the adsorbent amounts was investigated with a 100 mL solution of PAHs and varying amounts of each adsorbent tanning from 0.2 to 0.8 g. Figure 5(a) and 5(b) shows the PAHs adsorption at varying amounts of wPET-AC and wPS-AC. The increased amount of AC increases the availability of surface active sites for adsorption.

Table 2 | Surface area and pore dimensions of wPET-AC and wPS-AC.

| Adsorbent | BJH surface area (m<sup>2</sup> g<sup>−1</sup>) | BET surface area (m<sup>2</sup> g<sup>−1</sup>) | R<sub>p</sub> (Å) | V<sub>t</sub> (cm<sup>3</sup> g<sup>−1</sup>) |
|-----------|---------------------------------|---------------------------------|-------------|-----------------|
| wPET-AC   | 11.02                           | 65.35                           | 15.02       | 0.009           |
| wPS-AC    | 14.68                           | 50.14                           | 14.09       | 0.010           |

Figure 3 | FTIR spectra of (a) wPET and wPET-AC and (b) wPS and wPS-AC.
Figure 4 | % Adsorption efficiency of wPET-AC and wPS-AC for 16 PAHs adsorption.

Figure 5 | Effect of adsorbent dosage (a, b) and temperature (c, d) on the % adsorption of 16 PAHs over the wPET-AC and wPS/AC.
therefore a linear increase in the adsorption of PAHs is observed with raining AC dosage. The PAHs adsorption is not considerably affected by an extra increase in the adsorbent dose more than 0.8 g owing to the chance of aggregation, which restricts the number of active-site surfaces available for adsorption (Liu et al. 2014). It can be further observed from the data that, at low adsorbent dosage, the removal of all PAHs was lower. However, at optimum adsorbent dosage, the % removal of all PAHs was almost uniform. These results further show that the optimum dose of adsorbent in the current study is 0.8 g, which is higher than that reported by Awe et al. (2020) and Rad et al. (2014) and lower than those reported by Awoyemi (2011). However, this value was in agreement with Njoku et al. (2014).

**Effect of temperature**
Variation in temperature can affect the adsorption of PAHs onto the adsorbents from aqueous solution. The removal of PAHs from aqueous solution by wPET-AC and wPS-AC at different temperatures ranging from 20 to 50 °C is displayed in Figure 5(c) and 5(d), which shows that the removal of all PAHs increases as the temperature increased. This also suggests that the adsorption of PAHs over the wPET-AC and wPS-AC surfaces is endothermic. Various other researchers have reported the endothermic nature of the adsorption process using different adsorbents such as hazelnut shell activated carbon (Kobya 2004), coconut shell AC (Qiu et al. 2012), agricultural by-products (Chen et al. 2010) and hen feathers (Mittal et al. 2007).

**Effect of contact time**
The effects of contact time on PAHs adsorption onto wPET-AC and wPS-AC were investigated by carrying out adsorption experiments for different contact times, i.e., 0.25, 0.5, 1 and 2 hours. For all 16 PAHs, the adsorption on the AC increases up to 2 hr and becomes constant thereafter (Figure 6(a) and 6(b)). This pattern might be ascribed to a large number of active sites available on the surface of the adsorbent initially. After a contact time of 2 hr, the majority of active sites present on AC have been occupied and equilibrium is attained. It was observed that prolonged contact time had no effect on the PAHs adsorption. Thus, the optimum contact time was considered to be 2 hr. The absence of any particular trend in the adsorption of the studied PAHs is presumably due to the fact that different factors, for example, the nature of the adsorbent, chemical properties of the adsorbate, molecular size, solubility and hydrophobic character influence the adsorption process (Canzano et al. 2014). The adsorption characteristics, for example, surface functional groups, number of active sites, etc., also influence the adsorption process (Gupta & Gupta 2016). Previous studies have also reported similar observations for the adsorption of various PAHs on different adsorbents in that the adsorption of PAHs increases with increasing contact time and after equilibrium is attained (Awoyemi 2011; Gupta 2015).

**Effect of initial concentration of PAHs**
The adsorption of PAHs was studied as a function of initial concentration at varying initial PAH concentrations (10–40 ppm). Adsorption of PAHs was found to be increased when increasing the initial PAHs concentration (Figure 7(a) and 7(b)). This is because of the way that increasing of the initial PAHs concentration may result in the availability of PAHs to adsorb over the adsorbent surface active sites. Similar behavior has been observed in previous studies and arises from the effect of interactions between the initial PAHs concentration and adsorbent (Ping et al. 2006; Awoyemi 2011; Gupta 2015). Gong et al. (2007)
indicated that a high initial PAH concentration has positive influences, which may be due to the availability of a larger number of PAHs to interact with the active sites of the AC.

Effect of pH

pH is an important parameter that controls the PAHs uptake from aqueous solution. The influence of pH variation on the adsorption of PAHs using wPET-AC and wPS-AC as adsorbents from aqueous solution was studied at varying the pH (Figure 7(c) and 7(d)). The initial pH values of PAH mixture solutions were adjusted at 3, 5, 7, 9, and 11 with drops of diluted 0.1 M NaOH or HCl solution. The degree of adsorption limit is influenced by the pH of the solution due to the adsorbent surface charge distribution fluctuating the degree of adsorption, as indicated by the functional group of adsorbate (Putra et al. 2009). For example, Ph (89.44%), Fl (91.07%), BkF (88.67%), D[ah]A (92.47%), B[ghi]P (86.82%) adsorption over wPET-AC was recorded at pH 3.0 which is because the increment of positive charge on the surface of adsorbent at low pH resulted in a higher interaction between the PAH molecule and adsorbent surface. In agreement with this, an improved removal efficiency of Ph over soil comprising humic and fulvic acids, occurred at pH 3.0 (Ping et al. 2006). Similarly, the maximum adsorption of Nap (87.73%), Ace (84.00%), Acy (91%), Ant (91.63%), Flu (87.52%), Chr (89.49%) and InP (92.88%) occurs over the wPET-AC at pH 5.0. Results have shown that, under the optimum conditions, the maximum adsorption of BaA, Pyr, BaP, and BbF from aqueous medium attained over wPET-AC adsorbent was 90%, 87.27%, 86 and 88.23%, respectively at pH 7.0.

In the case of wPS-AC, the adsorption was Ph (55.33%), Fl (84.66%), BkF (86.23%), D[ah]A (85.09), B[ghi]P (86.54%) at pH 3.0, Nap (83.61%), Ace (79.40%), Acy (87.50%), Ant (89.64%), Flu (87.02%), Chr (82.30) and InP (99.25%) at pH 5.0 and BaA (84.70%), Pyr (84.10%), BaP (85.07%) and BbF (82.15%) at pH 7.0. At low pH values, due to the formation of positive charges on the surface of the adsorbent, higher adsorption occurs, and then the electrostatic interaction between the PAH and the surface charge of the adsorbent increases. Conversely, the positive charge on the surface of the adsorbent at higher pH is diminished and PAH molecules interact with OH⁻ ions for the adsorption on active sites, diminishing the adsorption.
efficiency of the adsorbent (Gupta 2015). As shown in Figure 7(c) and 7(d), the adsorptive efficiency of wPET-AC and wPS-AC decreased when the pH of the solution was increased from 7.0 to 11.0. This may be due to the increased number of surface functional groups. pH can change the chemical form, hydrophilicity, solubility, and protonation of PAHs in solution (Liao et al. 2008). These results are in agreement with the findings of Ping et al. (2006) and Balati et al. (2015). Furthermore, the zeta potential of the wPET-AC and wPS-AC decreased as the pH value was increased (as shown in Figure 8). The adsorption quantity decreased with increasing pH value due to the electrostatic repulsion force (Deng et al. 2013).

**Adsorption kinetics**

In order to investigate the mechanism of sorption and potential rate-controlling steps such as mass transport and chemical reaction processes, kinetic models have been used to test the experimental data. These kinetic models included the pseudo-first-order equation and the pseudo-second-order equation. Based on these results, the pseudo-second-order kinetics equation provided a better fit to the adsorption data compared to the pseudo-first-order equation. The reaction fits well to a pseudo-second-order model as it indicates a trend towards chemisorption. In this kind of adsorption, the chemical reaction seems significant in the rate-controlling step and the pseudo-second-order chemical reaction kinetics provides the best correlation of the experimental data and the adsorption’s mechanism is chemically rate controlling and because of this it is called chemisorption. According to Robati (2013), the removal from a solution is due to physico-chemical interactions between the two phases. The values of experimental adsorption capacity (qe,exp) were found to be in good agreement with the calculated qe values (qe,cal) using the pseudo-second-order model. The values of adsorption capacity, and correlation coefficient (R²) along with rate constants for pseudo-second-order kinetics models are given in Table 3. The adsorption data were also analyzed using an intra-particle diffusion kinetic model. The data from the Table (Table 4) about the regression factor and reaction rate are very accurate accordingly; however, the data manipulation via the pseudo-second-order equation is more reliable and thus shows the best fitting of data via the pseudo-second-order reaction rate.

**Isotherm study**

Adsorption mechanisms pathways, adsorbents’ surface properties and capacities of adsorbents which are basic parameters to design an effective adsorption system, could be determined with the help of adsorption isotherms. Adsorption isotherms explain how the contaminants, for example, PAHs interact with the adsorbent materials. Freundlich and Langmuir isotherm
models have been used to understand and determine the PAH adsorption mechanisms and adsorption capacity of the adsorbents. The equilibrium adsorption data were evaluated using the Freundlich and Langmuir isotherm models. The calculated Langmuir constants ($K_1$ and $q_m$), as well as the coefficients of correlation ($R^2$) for isotherms, are detailed in Table 5 for wPET-AC and wPS-AC.

### Table 3 | Pseudo-second-order kinetic parameters for adsorption of PAHs over wPET-AC and wPS-AC

| PAHs | wPET-AC | wPS-AC |
|------|---------|--------|
|      | $k_2$ (mg g$^{-1}$ min$^{-1}$) | $q_{e2}$ (mg g$^{-1}$) | $R^2$ | $k_2$ (mg g$^{-1}$ min$^{-1}$) | $q_{e2}$ (mg g$^{-1}$) | $R^2$ |
| Nap  | $5 \times 10^3$ | 12.987 | 0.973 | $1 \times 10^4$ | 10.870 | 0.999 |
| Ace  | $5 \times 10^3$ | 13.158 | 0.999 | $7 \times 10^4$ | 10.870 | 0.999 |
| Acy  | $5 \times 10^3$ | 13.699 | 0.965 | $2 \times 10^4$ | 11.364 | 1.000 |
| Ant  | $9 \times 10^3$ | 13.889 | 0.997 | $1 \times 10^4$ | 12.195 | 0.999 |
| Ph   | $7 \times 10^3$ | 12.821 | 0.996 | $9 \times 10^3$ | 11.628 | 0.998 |
| Fl   | $9 \times 10^3$ | 13.699 | 0.999 | $2 \times 10^4$ | 10.989 | 0.999 |
| Flu  | $8 \times 10^3$ | 12.048 | 0.998 | $1 \times 10^4$ | 11.765 | 0.999 |
| BaA  | $7 \times 10^3$ | 12.821 | 0.998 | $1 \times 10^4$ | 11.111 | 1.000 |
| Chr  | $7 \times 10^3$ | 12.346 | 0.998 | $7 \times 10^3$ | 11.628 | 0.997 |
| Pyr  | $1 \times 10^4$ | 11.765 | 0.999 | $4 \times 10^3$ | 12.346 | 0.979 |
| BaP  | $7 \times 10^3$ | 12.821 | 0.998 | $5 \times 10^3$ | 12.500 | 0.999 |
| BkF  | $8 \times 10^3$ | 12.346 | 0.998 | $5 \times 10^3$ | 12.346 | 0.978 |
| BbF  | $1 \times 10^3$ | 13.333 | 0.993 | $1 \times 10^4$ | 10.869 | 0.999 |
| D[ah]A | $6 \times 10^3$ | 15.152 | 0.998 | $9 \times 10^3$ | 11.628 | 0.999 |
| Bl[ghi]P | $1 \times 10^4$ | 11.905 | 0.999 | $4 \times 10^3$ | 10.309 | 0.995 |
| InP  | $8 \times 10^3$ | 13.333 | 0.999 | $6 \times 10^3$ | 13.158 | 0.999 |

### Table 4 | Intra-particle diffusion parameters for adsorption of PAHs on wPET-AC and wPS-AC

| PAHs | wPET-AC | wPS-AC |
|------|---------|--------|
|      | $C$ (mg g$^{-1}$) | $K_{id}$ (min$^{-1}$) | $R^2$ | $C$ (mg g$^{-1}$) | $K_{id}$ (min$^{-1}$) | $R^2$ |
| Nap  | 0.038   | 0.012  | 0.992 | 0.260 | 0.093 | 0.992 |
| Ace  | 0.365   | 0.124  | 0.897 | 0.334 | 0.139 | 0.969 |
| Acy  | 0.349   | 0.152  | 0.884 | 0.343 | 0.148 | 0.876 |
| Ant  | 0.169   | 0.108  | 0.828 | 0.23  | 0.188 | 0.872 |
| Ph   | 0.152   | 0.085  | 0.960 | 0.178 | 0.053 | 0.839 |
| Fl   | 0.202   | 0.052  | 0.976 | 0.199 | 0.055 | 0.947 |
| Flu  | 0.195   | 0.076  | 0.868 | 0.189 | 0.084 | 0.833 |
| BaA  | 0.042   | 0.016  | 0.974 | 0.038 | 0.204 | 0.879 |
| Chr  | 0.040   | 0.015  | 0.930 | 0.042 | 0.013 | 0.949 |
| Pyr  | 0.055   | 0.006  | 0.959 | 0.054 | 0.006 | 0.984 |
| BaP  | 0.027   | 0.011  | 0.974 | 0.028 | 0.009 | 0.973 |
| BkF  | 0.039   | 0.010  | 0.998 | 0.037 | 0.012 | 0.982 |
| BbF  | 0.055   | 0.008  | 0.999 | 0.052 | 0.012 | 0.873 |
| D[ah]A | 0.039  | 0.024  | 0.798 | 0.052 | 0.012 | 0.873 |
| Bl[ghi]P | 0.030 | 0.015  | 0.979 | 0.012 | 0.030 | 0.994 |
| InP  | 0.028   | 0.015  | 0.895 | 0.031 | 0.020 | 0.952 |
AC and wPS-AC. The results reveal that the adsorption data fitted better with Langmuir adsorption isotherms in the case of both wPET-AC and wPS-AC.

**Adsorption thermodynamics**

The investigations of thermodynamic parameters for adsorption studies are another essential factor. Thermodynamic parameters are used to decide that a process is physical or chemical, non-spontaneous or spontaneous, and exothermic or endothermic in nature. The thermodynamic parameters including $\Delta G^\circ$, $\Delta H^\circ$ and $\Delta S^\circ$ were calculated in batch mode experiments at various temperatures.

For the thermodynamic parameters calculation, Equations (1)–(3) were applied (Konicki et al. 2012):

\[
\Delta G^\circ = -RT \ln K_D
\]  
(1)

\[
\Delta H^\circ = R \frac{T_2 T_1}{T_2 - T_1} \ln \frac{k_2}{k_1}
\]  
(2)

\[
\Delta S^\circ = \frac{\Delta H^\circ - \Delta G^\circ}{T}
\]  
(3)

Results for thermodynamic parameters are detailed in Tables 6 and 7 for wPET-AC and wPS-AC, respectively. The results revealed that an increase in temperature enhances the adsorption of PAHs onto wPET-AC and wPS-AC. The adsorption phenomenon boosts up at higher temperatures due to the creation of more adsorption sites and intraparticle diffusion. The adsorption process is non-spontaneous and endothermic in nature, which was confirmed by the positive values of $\Delta G^\circ$, $\Delta S^\circ$ and $\Delta H^\circ$.

**Mechanisms of PAHs adsorption**

Generally, AC adsorbs pollutants from aqueous solutions, depending on the solution conditions, the adsorbent surface characteristics, and the nature of the adsorbate. Various mechanisms by which the carbon surface functional groups may affect the adsorption process from aqueous solution have been proposed: electron-donor–electron-acceptor interactions
Table 6 | Thermodynamic factors for the adsorption of PAHs onto wPET-AC

| Temperature (°C) | Nap | Acy | Ph | Flu | Chr | BaP | BbF | B[ghi]P |
|-----------------|-----|-----|----|-----|-----|-----|-----|--------|
|                 | ΔG° (kJ mol⁻¹) | ΔH° (kJ mol⁻¹) | ΔS° (kJ mol⁻¹) | ΔG° (kJ mol⁻¹) | ΔH° (kJ mol⁻¹) | ΔS° (kJ mol⁻¹) | ΔG° (kJ mol⁻¹) | ΔH° (kJ mol⁻¹) | ΔS° (kJ mol⁻¹) |
| 20              | 7.4333 | 16.9107 | 0.0313 | 7.7732 | 20.918 | 0.0437 | 0.0287 | 6.9060 | 14.7324 | 0.0258 |
| 30              | 7.0332 | 7.1001 | 6.7222 | 6.3993 | 6.4590 | 6.2807 | 6.1808 | 6.2807 | 6.4954 | 6.1808 |
| 40              | 6.9831 | 6.7371 | 6.3399 | 6.8631 | 6.3292 | 6.5037 | 6.4345 | 6.5037 | 6.4345 | 6.4954 |
| 50              | 6.3993 | 6.3399 | 6.1808 | 6.2807 | 6.3292 | 6.5037 | 6.4345 | 6.5037 | 6.4345 | 6.4954 |
| 20              | 7.1539 | 15.9961 | 0.0287 | 6.9060 | 14.7324 | 0.0258 | 7.0276 | 12.8202 | 0.0193 | 7.2546 | 14.7158 | 0.0248 |
| 30              | 7.1915 | 6.7371 | 6.3399 | 6.8631 | 6.3292 | 6.5037 | 6.4345 | 6.5037 | 6.4345 | 6.4954 |
| 40              | 6.8631 | 6.3399 | 6.1808 | 6.2807 | 6.3292 | 6.5037 | 6.4345 | 6.5037 | 6.4345 | 6.4954 |
| 50              | 6.2807 | 6.3292 | 6.1808 | 6.2807 | 6.3292 | 6.5037 | 6.4345 | 6.5037 | 6.4345 | 6.4954 |
| 20              | 7.0825 | 16.3666 | 0.0242 | 6.9640 | 14.6410 | 0.0256 | 7.0276 | 12.8202 | 0.0193 | 7.2546 | 14.7158 | 0.0248 |
| 30              | 6.7241 | 6.5037 | 6.4345 | 6.5687 | 6.6625 | 6.4954 | 6.4345 | 6.5687 | 6.6625 | 6.4954 |
| 40              | 6.5866 | 6.3292 | 6.1808 | 6.4195 | 6.4954 | 6.4954 | 6.4345 | 6.5687 | 6.6625 | 6.4954 |
| 50              | 6.3268 | 6.1808 | 6.4954 | 6.4195 | 6.4954 | 6.4954 | 6.4345 | 6.5687 | 6.6625 | 6.4954 |
| 20              | 7.0223 | 13.3523 | 0.0293 | 6.8024 | 10.3925 | 0.0122 | 7.0223 | 13.3523 | 0.0293 | 6.8024 | 10.3925 | 0.0122 |
| 30              | 6.7460 | 6.4536 | 6.4345 | 6.5344 | 6.4536 | 6.4536 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 40              | 6.5244 | 6.3454 | 6.3127 | 6.3985 | 6.4175 | 6.4175 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 50              | 6.3985 | 6.3127 | 6.3127 | 6.3985 | 6.4175 | 6.4175 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 20              | 7.0168 | 14.5079 | 0.0248 | 6.9111 | 12.6788 | 0.190 | 7.0168 | 14.5079 | 0.0248 | 6.9111 | 12.6788 | 0.190 |
| 30              | 6.8078 | 6.7665 | 6.4661 | 6.4006 | 6.4661 | 6.4661 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 40              | 6.4006 | 6.4661 | 6.4661 | 6.3326 | 6.3790 | 6.3790 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 50              | 6.3326 | 6.3790 | 6.3790 | 6.4006 | 6.4661 | 6.4661 | 6.4345 | 6.5344 | 6.4536 | 6.4536 |
| 20              | 7.3831 | 17.6264 | 0.0336 | 7.7726 | 23.1129 | 0.0511 | 7.3831 | 17.6264 | 0.0336 | 7.7726 | 23.1129 | 0.0511 |
| 30              | 6.4196 | 6.8912 | 6.5632 | 6.4894 | 6.8912 | 6.8912 | 6.5632 | 6.4894 | 6.8912 | 6.8912 |
| 40              | 6.4894 | 6.8912 | 6.8912 | 6.2715 | 6.1993 | 6.1993 | 6.5632 | 6.4894 | 6.8912 | 6.8912 |
| 50              | 6.2715 | 6.1993 | 6.1993 | 6.4894 | 6.5632 | 6.5632 | 6.5632 | 6.4894 | 6.8912 | 6.8912 |
| 20              | 6.8204 | 11.1740 | 0.0145 | 6.9904 | 14.9486 | 0.0264 | 6.8204 | 11.1740 | 0.0145 | 6.9904 | 14.9486 | 0.0264 |
| 30              | 6.5898 | 6.6549 | 6.3127 | 6.3803 | 6.6549 | 6.6549 | 6.3127 | 6.3803 | 6.6549 | 6.6549 |
| 40              | 6.3803 | 6.6549 | 6.3127 | 6.4167 | 6.2341 | 6.2341 | 6.3127 | 6.4167 | 6.2341 | 6.2341 |
| 50              | 6.4167 | 6.2341 | 6.2341 | 6.4167 | 6.2341 | 6.2341 | 6.3127 | 6.4167 | 6.2341 | 6.2341 |
Table 7 | Thermodynamic factors for the adsorption of PAHs onto wPS-AC

| wPS-AC | Nap | Ace |
|---|---|---|
| Temperature (°C) | \( \Delta G^o \) (kJ mol\(^{-1}\)) | \( \Delta H^o \) (kJ mol\(^{-1}\)) | \( \Delta S^o \) (kJ mol\(^{-1}\)) | \( \Delta G^o \) (kJ mol\(^{-1}\)) | \( \Delta H^o \) (kJ mol\(^{-1}\)) | \( \Delta S^o \) (kJ mol\(^{-1}\)) |
| 20 | 7.4333 | 15.7716 | 0.0276 | 7.7732 | 19.1471 | 0.0380 |
| 30 | 7.0332 | | | 7.1001 | | |
| 40 | 6.9831 | | | 6.7222 | | |
| 50 | 6.5279 | | | 6.6577 | | |
| Acy | | | | | | |
| 20 | 7.1539 | 14.8071 | 0.0248 | 6.9060 | 13.1943 | 0.0208 |
| 30 | 7.1915 | | | 6.7371 | | |
| 40 | 6.8631 | | | 6.3359 | | |
| 50 | 6.4141 | | | 6.3539 | | |
| Ph | | | | | | |
| 20 | 7.0825 | 13.0280 | 0.0198 | 6.9640 | 11.5808 | 0.0166 |
| 30 | 6.7241 | | | 6.5037 | | |
| 40 | 6.5866 | | | 6.3292 | | |
| 50 | 6.4769 | | | 6.4966 | | |
| Flu | | | | | | |
| 20 | 7.0276 | 12.7454 | 0.0191 | 7.2546 | 14.7158 | 0.0248 |
| 30 | 6.6506 | | | 6.8386 | | |
| 40 | 6.5687 | | | 6.6625 | | |
| 50 | 6.4278 | | | 6.4954 | | |
| Chr | | | | | | |
| 20 | 7.0223 | 11.8558 | 0.0162 | 6.8024 | 9.5362 | 0.0094 |
| 30 | 6.7460 | | | 6.4536 | | |
| 40 | 6.5344 | | | 6.4345 | | |
| 50 | 6.5675 | | | 6.5132 | | |
| BaP | | | | | | |
| 20 | 7.0168 | 13.1610 | 0.0204 | 6.9111 | 12.0389 | 0.170 |
| 30 | 6.8078 | | | 6.7665 | | |
| 40 | 6.4006 | | | 6.4661 | | |
| 50 | 6.4845 | | | 6.4506 | | |
| BbF | | | | | | |
| 20 | 7.3831 | 14.6493 | 0.0250 | 7.7726 | 20.5855 | 0.0430 |
| 30 | 6.4196 | | | 6.8912 | | |
| 40 | 6.4894 | | | 6.5632 | | |
| 50 | 6.5720 | | | 6.4839 | | |
| B[ghi]P | | | | | | |
| 20 | 6.8204 | 10.9495 | 0.0138 | 6.9904 | 13.7846 | 0.0227 |
| 30 | 6.5898 | | | 6.6549 | | |
| 40 | 6.3803 | | | 6.3127 | | |
| 50 | 6.4417 | | | 6.3648 | | |
Table 8 | Details of adsorption media and their PAHs removal efficiency

| Materials                                         | PAHs removal efficiency (%)                                                                 | Reference                        |
|---------------------------------------------------|-----------------------------------------------------------------------------------------------|----------------------------------|
| Combined permanganate (Mn(VII))/bisulfite advanced oxidation process | Pyr, Chr, BaA, BbF, BkF, BaP, and D[ah]A (85–100%)                                          | Fang et al. (2021)               |
| Clinoptilolite                                    | NaP (100)                                                                                     | Puszkarewicz & Kaleta (2020)     |
| Graphene wool                                     | Ph (99.9), Pyr (99.1)                                                                          | Adeola & Forbes (2019)           |
| Stearoyl grafted cellulose                        | Ph (97.61), Pyr (96.94)                                                                        | Kim et al. (2018)                |
| Grafted cellulose                                 | 16 PAHs (94 and 98)                                                                           | Euvrard et al. (2014)            |
| Activated carbon bed                              | NaP (98.1), Ace (98.2), Acy (98.4), Ant (98.9), Ph (100), Fl (94.7)                           | Smol et al. (2014)               |
| Activated carbon coconut shells and anthracite based | Ph (95), Ant (97.5), Flu (96.5), Pyr (98), BaP (88), Cry (94.5), BbF (86), BkF (88), BaA (94.5), InP (73), B[ghi]P (69) | Amstaetter et al. (2012)         |
| Light weight expanded clay aggregate              | Ph (92.61), Flu (93.91), Pyr (94.15)                                                           | Nkansah et al. (2012)            |
| Activated carbon                                  | Nap (97), Fl (98)                                                                              | Awoyemi (2011)                   |
| Coke-derived porous carbon                        | NaP, Fl, Ph, Pyr, Flu (>99 for all PAHs)                                                        | Yuan et al. (2010)               |
| Wood ashes                                        | Fl, Pyr, Cry, BaA, BbF, BkF, BaP, D[ah]A, InP, B[ghi]P (>99)                                   | Pérez-Gregorio et al. (2010)     |
| Inorgano-organo-bentonite                         | Ph (99)                                                                                       | Ma & Zhu (2006)                  |
| Wood char                                         | Pyr, Ph, BaA (≥60)                                                                             | Wang et al. (2006)               |
| wPS-AC                                            | Nap (83.61), Ace (79.4), Acy (87.5), Ant (89.64), Ph (85.33), Fl (84.66), Flu (87.02), BaA (84.7), Chr (82.3), Pyr (84.1), Bap (85.07), BkF (86.23), BbF (82.15), D[ah]A (85.09), B[ghi]P (86.54), InP (89.25) | This article                     |
| wPET-AC                                           | Nap (88.16), Ace (85.9), Acy (92.74), Ant (96.22), Ph (90.48), Fl (96.31), Flu (87.02), BaA (91.10), Chr (88.38), Pyr (87.46), BaP (90.07), BkF (88.23), BbF (92.15), D[ah]A (95.09), B[ghi]P (87.54), InP (94.25) | This article                     |
(Mattson et al. 1969), hydrogen bonding (Coughlin & Ezra 1968; Mahajan et al. 1980; Yuan et al. 2010), and π–π dispersive interactions (Coughlin & Ezra 1968). Lyklema (1995) reported that the PAHs adsorption from aqueous solution is basically an exchange process. Therefore, the adsorption process of organic compounds on the surface of AC has been described as a complex interaction between electrostatic and dispersion interactions, especially in the case of weak organic electrolytes. However, since PAHs are not considered to be electrolytic, the intermediate hydrogen bonding, as well as van der Waals dispersive interactions, have become the main mechanisms proposed in the literature (Yuan et al. 2010). The mechanism of adsorption of PAH by AC is given in the reaction scheme in Figure 9.

Comparison of adsorption potential of different adsorbents

The comparative adsorption efficiency of the various adsorption media reported in the literature and the AC, i.e., wPET-AC and wPS-AC for PAHs adsorption are detailed in Table 8. The efficiency of wPET-AC was probably because of the larger surface area (65.35 m² g⁻¹ for wPET-AC compared to 50.14 m² g⁻¹ for wPS-AC) as well as hydrophobicity which advances higher adsorption capacity than the wPS-AC utilized in our research. The decision for adsorbent for water treatment applications relies upon many factors, for example, reusability, flexibility, availability of material, nontoxicity, efficiency, etc. wPET-AC shows high potential for water treatment applications with a removal efficiency >85% for all PAHs at the same time.

Desorption and reusability

In the adsorption processes, desorption/regeneration of adsorbents is one of the essential aspects, as it controls the economy of water treatment technology (Ali 2012). For the recovery and effective regeneration of adsorbents, acids (such as H₂SO₄, HCOOH, HCl, CH₃COOH, and HNO₃), buffer solutions (such as bicarbonate and phosphate), salts (such as NH₄NO₃, CaCl₂,2H₂O, KNO₃, (NH₄)₂SO₄, C₆H₅Na₃O₇·2H₂O, KNO₃, NaCl and KCl), deionized water as well as chelating agents have been used in various studies (Lata et al. 2015). In this study, HCl solution was used for the desorption experiment. This acid gave satisfactory desorption results. The regeneration studies were carried out in batches using 100 mL of standard solutions indicating that wPET-AC and wPS-AC had the potential for reusability (Figure 10).
CONCLUSIONS

In the present work, the efficiency of wPET-AC and wPS-AC for adsorption of 16 PAHs from aqueous medium was examined. The effects of operating variables such as pH, contact time, temperature, adsorbent dose, and initial PAHs concentration were investigated. An increase in adsorbent dose, initial PAHs concentration, and contact time improved the adsorption efficiency. The adsorption process after a contact time of 2 hr reached equilibrium. Kinetics studies showed that the adsorption system followed a pseudo-second-order equation. The study revealed that the Langmuir equation represented the best fit for the experimental data. Thermodynamic results indicated that the adsorption of PAHs onto wPET-AC and wPS-AC is endothermic in nature. These results indicated that wPET-AC can be used as a suitable and efficient adsorbent for the removal of PAHs from wastewater. This study will benefit the entire community through minimizing the environmental pollution caused by solid plastic waste and as well as removing the toxic PAHs from wastewater. Thus, the study serves to develop a route for better reuse of waste plastic materials, thereby minimizing the environmental degradation due to waste plastics.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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