A Promising Solution for Food Waste: Preparing Activated Carbons for Phenol Removal from Water Streams

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ABSTRACT: Phenol and its derivatives are highly toxic chemicals and are widely used in various industrial applications. Therefore, the industrial wastewater streams must be treated to lower the concentration of phenol before discharge. At the same time, food waste has been a major environmental problem globally and the scientific community is eagerly seeking effective management solutions. The objective of this study was to understand the potential of utilizing food waste as a renewable and sustainable resource for the production of activated carbons for the removal of phenol from water streams. The food waste was pyrolyzed and physically activated by steam. The pyrolysis and activation conditions were optimized to obtain activated carbons with high surface area. The activated carbon with the highest surface area, 745 m² g⁻¹, was derived via activation at 950 °C for 1 h. A detailed characterization of the physicochemical and morphological properties of the activated carbons derived from food waste was performed and a comprehensive adsorption study was conducted to investigate the potential of using the activated carbons for phenol removal from water streams. The effects of pH, contact time, and initial concentration of phenol in water were studied and adsorption models were applied to experimental data to interpret the adsorption process. A remarkable phenol adsorption capacity of 568 mg g⁻¹ was achieved. The results indicated that the pseudo-second-order kinetic model was better over the pseudo-second-order kinetic model to describe the kinetics of adsorption. The intraparticle diffusion model showed multiple regions, suggesting that the intraparticle diffusion was not the sole rate-controlling step of adsorption. The Langmuir isotherm model was the best model out of Freundlich, Temkin, and Dubinin–Radushkevich models to describe the phenol adsorption on activated carbons derived from food waste. This study demonstrated that food waste could be utilized to produce activated carbon and it showed promising capacity on phenol removal.

INTRODUCTION

Phenol and its derivatives are widely used chemicals in industries, such as petroleum, pharmaceutical, leather, pesticide, paper, and plastic industries.¹⁻⁴ These chemicals are classified as priority pollutants because of their high toxicity and low biodegradability.⁵,⁶ Excessive exposure to phenolic compounds may cause negative effects on the brain, eyes, liver, skin, and other parts of humans.⁷ The discharge of wastewater containing phenolic compounds is also harmful to aquatic life, which may cause oxygen depletion in water.⁸ The United States (U.S.) Environmental Protection Agency (EPA) regulations set a restriction of less than 1 mg L⁻¹ on phenol concentration in wastewater.⁹ Therefore, the wastewater streams from industrial sides must be treated to lower the concentration of phenolic compounds prior to being discharged into the environment.⁴,¹⁰ The treatment approaches of wastewater containing phenolic compounds include physical, chemical, and biological techniques. Photocatalysis,¹ coagulation,¹² electrochemical¹³ and chemical oxidation,¹⁴ and adsorption⁵,¹⁵⁻¹⁷ are the most widely used techniques for removing phenolic compounds. Adsorption by activated carbons is considered the most favorable owing to their low cost, high efficiency, simplicity, and high availability.⁵,⁷,¹⁵

Activated carbon (AC) is a carbonaceous material that has a high surface area and large pore volume, which is widely used for the adsorption of pollutants. The resources to prepare AC can be categorized into two groups: (a) fossil-related resources, such as coal, peat, lignite, and petroleum residues, which are nonrenewable and not environmentally friendly,¹⁸ and (b) bioresources, such as agricultural waste and lignocellulosic materials.¹⁸ Although the carbon contents of fossil resources are higher compared to biomass, leading to a higher yield of AC, the overall cost of AC produced from biomass is lower due to the low feedstock price and ecological suitability.¹⁹

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Therefore, producing AC from bioresources is preferred considering its sustainability. There are two approaches to convert biomass into AC: (a) physical activation and (b) chemical activation. The physical activation is a two-step process: pyrolysis (or carbonization) followed by activation. In the first step, the feedstock is pyrolyzed under an inert atmosphere and turned into biochar.24-26 The pyrolysis temperature can vary from 400 to 900 °C.25-29 In the second step, the biochar derived from pyrolysis is activated at temperatures ranging from 500 to 900 °C in the presence of steam or carbon dioxide.25-29 The chemical activation is carried out with the assistance of chemical agents, such as ZnCl2,30-32 H2PO4,33-36 H2SO4,37 KOH,38 NaOH,39 and K2CO3.40 Among all these chemicals, ZnCl2 and H2PO4 are the most commonly used activation agents. The chemical activation is commonly performed at temperatures between 450 and 600 °C.18 The atmosphere for chemical activation can be either inert gas41 or air.35 After chemical activation, usually, a washing process is needed to remove the chemical agents from the desired products. Both physical activation and chemical activation have their own advantages and disadvantages. ACs produced via physical activation are cleaner than those produced by chemical activation and do not need a washing process. Furthermore, physical activation avoids using corrosive or environmentally unfriendly chemical agents.40 On the other side, chemical activation is commonly performed in one step at relatively low temperatures and results in AC with higher surface area and larger pore volume. Various biomass resources have been studied as feedstocks for the production of renewable AC, such as tomato waste,41 corncob,42 palm oil kernel shell,43 grape stalk,45 apple waste,46 coconut shell,47 and chestnut oak shell.48

Similar to the aforementioned biomass resources, food waste has a great potential to produce AC. The Food and Agriculture Organization of United Nations (FAO) has pointed out that nearly 1.3 billion tons of food is thrown away during production and consumption every year.49 Food waste is mainly composed of carbohydrates, lignin, proteins, organic acids, lipids, and ash.50 The traditional management of food waste includes feeding animals, composting, incineration, or landfilling.51 However, major efforts are implemented toward the utilization of food waste to produce bioenergy, biofuels, and other bio-products.52,53 The technologies of turning food waste to energy can be categorized into two main groups: biological and thermochemical.54 Biological technologies include anaerobic digestion and fermentation; the former technology produces biogas, while the latter is utilized to generate bioethanol. Thermochemical treatments involve incineration, pyrolysis, gasification, and hydrothermal carbonization. Incineration is utilized to generate heat and energy. However, this treatment can cause major environmental problems.55 Furthermore, food waste is not suitable for combustion due to its high moisture content, which results in low heat density.54 Hydrothermal carbonization can be applied to convert food waste into high-carbon and high-energy-density material, named hydrochar.56 However, intense energy requirements are a major barrier toward the commercialization of this process.57 Pyrolysis converts food waste into multiple products, such as syngas, biochar, and bio-oil, while gasification results mainly into syngas. Both of the aforementioned processes are considered appropriate for food waste utilization.54

Recently, renewable activated carbons derived from various biomass resources have been tested for the removal of phenols from aqueous solution. Black wattle bark waste was investigated by Lütke et al. for the production of activated carbons.38 The activation was performed using ZnCl2 as the activation agent and the maximum phenol removal capacity reached 98.6 mg g−1. Lv et al. prepared activated carbons from rice husk via a two-step activation process using KOH and EDTA-4Na as activation agents.59 The adsorption capacity for phenol reached 215.27 mg g−1. Hamadneh et al. used olive husk to prepare activated carbons using MgCl2,60 and the adsorption experiments revealed a maximum removal capacity of 43.86 mg g−1. Supong et al. used Tithonia diversifolia to prepare activated carbons with the assistance of KOH and showed a maximum phenol adsorption capacity of 42.61 mg g−1.61 Bark waste of Acacia mangium was activated using phosphoric acid and evaluated for phenol adsorption by Zhang et al.62 The maximum phenol adsorption capacity reached 96.92 mg g−1. Karri et al. have reported using coconut shell-based activated carbons for phenol removal in batch experiments4 and in a fluidized bed reactor10 that the removal efficiency was able to reach 96%. These studies have shown that renewable bioresources and wastes have a great potential as activated carbons and can be effectively employed for phenol removal.

The abundance and the need for proper food waste management justify the investigation of using this waste as a potential renewable resource to produce valuable bioproducts, such as AC. The objective of this study is to prepare high-surface-area AC from food waste and test its potential for phenol adsorption in aqueous solutions. We should note that the use of renewable biomass resources for the production of activated carbons is not a new concept. However, most of the studies related to the preparation of activated carbons from food waste were actually using a specific agricultural waste as a starting material (e.g., orange peels, olive stones, palm shells, coffee grounds, coconut shells, etc.).46,63-66 In this study, we use food waste derived from the dining halls located at the University of Connecticut (UConn); thus, this waste resource represents a complex mixture of various types of food. The generated activated carbons in this study were further applied for phenol adsorption from water streams. Lee et al. have studied the adsorption of phenol using biochar derived from food waste but not activated carbons.35 Hence, it is the first time that a food waste complex is used for the preparation of activated carbons and applied for phenol removal from water. This study suggested that the food waste complex has a great potential to be converted into valuable activated carbons that can be effectively used for purification purposes.

**EXPERIMENTAL SECTION**

**Materials.** The feedstock, food waste, was obtained from the Dining Services Department at the University of Connecticut. The raw food waste was first washed five times with DI water to remove salt and soluble minerals. The washed food waste was then dried and crushed to small pieces. The crushed food waste was ground and sieved to a particle size between 180 and 355 μm and then used for pyrolysis and activation experiments. Phenol (99.0%) was purchased from Sigma-Aldrich. HCl (1 M) was purchased from Fisher Scientific. Ar was purchased from Airgas and used as a carrier gas for the preparation of biochar and AC.
**Preparation of Biochar and Activated Carbons.** ACs from food waste were produced via a two-step physical activation process. First, food waste was pyrolyzed to produce biochar. Typically, 3 g of washed and dried food waste was sandwiched by two pieces of quartz wool and placed at the center of a quartz tube. The quartz tube was then inserted into a vertical tube furnace. Ar was used to provide an inert atmosphere with a flow rate of 50 sccm. Ice-bathed methanol was used to absorb bio-oil that was generated during pyrolysis. Mass spectroscopy (Agilent 5975C) was used to measure the gas formed during pyrolysis. The pyrolysis was performed at a temperature range of 275 to 525 °C with a ramp rate of 10 °C min⁻¹. The residence time was varied from 30 to 120 min. After the pyrolysis of food waste, the derived biochar was ground and meshed to a particle size smaller than 300 μm. The biochar samples were labeled as “Biochar-pyrolysis temperature-residence time”. The biochar with the highest carbon content was selected to prepare AC via physical activation.

After the pyrolysis, biochar was activated in a horizontal tube furnace. The biochar was put into an alumina boat and placed at the center of the furnace. Ar was employed as a carrier gas to carry steam from a saturator. A controller was used to control the temperature of water in the saturator to keep the partial pressure of steam at approximately 50%. The activation temperature was varied from 750 to 950 °C, while the residence time was varied from 1 to 5 h. The flow rate of Ar was set to 50 sccm and the ramp rate of temperature was kept at 10 °C min⁻¹. Steam-activated ACs were labeled as “FWAC-activation temperature-residence time”.

**Characterization of Biochar and AC.** Surface areas and porosities of biochar and AC were determined by N₂ adsorption–desorption using a Micromeritics ASAP 2020C Sorption Analyzer. All materials were degassed for 12 h at 120 °C under vacuum. N₂ adsorption–desorption isotherms were then gathered at 77 K under a liquid nitrogen environment. Surface areas of samples were calculated using the Brunauer–Emmett–Teller (BET) method, while pore volumes were calculated using the single-point method below P/P₀ = 0.99.

A scanning electron microscope equipped with an energy-dispersive X-ray spectroscopy (EDX) detector was conducted to study the morphologies and regional element distributions of food waste, biochar, and AC. Scanning electron microscopy (SEM) was performed using an FEI Quanta FEG 250 scanning electron microscope operating at a potential of 10 kV.

Elemental composition of food waste, biochar, and AC was analyzed by elemental analysis and inductively coupled plasma optical emission spectroscopy (ICP-OES). Elemental analysis was applied to measure the content of carbon, hydrogen, nitrogen, and sulfur using an Elementar Vario Microcube analyzer. Noncombustible element concentrations (calcium, phosphorus, and sodium) were measured by ICP-OES using a Thermo Scientific iCAP 6500.

X-ray diffraction (XRD) patterns for biochar and AC were obtained using a Bruker D8 Advance powder diffraction spectrometer (CuKα radiation source). Chemical structures of biochar and AC were identified by ¹³C nuclear magnetic resonance (NMR) and Fourier transform infrared spectroscopy (FTIR). The solid-state magic angle spinning (MAS) ¹³C NMR spectra were acquired using a Bruker Advance III spectrometer. The diffuse reflectance FTIR (DRIFTS) spectra were collected on a Thermo Nicolet 6700 FTIR spectrometer with an MCT detector and a temperature-controlled Harrick Praying Mantis DRIFTS assembly. Samples were analyzed at 100 °C to exclude the effects of water, and all the samples were diluted in KBr. Temperature-programmed desorption (TPD) was conducted from 60 to 1000 °C at a heating rate of 10 °C/min under an Ar atmosphere using a tube furnace (Lindburg/Blue M) and the generated gases were analyzed by mass spectroscopy (Agilent 5975C); the results are shown in the Supporting Information.

**Phenol Adsorption Experiments.** Phenol adsorption experiments were performed using the prepared AC with the highest surface area. A phenol stock solution with a concentration of 1 g L⁻¹ was prepared by dissolving phenol crystals into DI water. The phenol solutions with lower concentration were prepared by diluting the stock solution to the desired concentration. The batch adsorption experiments were performed with different initial phenol concentrations ranging from 10 to 500 mg L⁻¹. For a typical experiment, 10 mg of dried AC was added to 50 mL of phenol solution. The duration of adsorption was varied to obtain kinetic data. After adsorption, the liquid samples were collected by filtration. All the experiments were repeated three times. Adsorption experiments were also conducted at different pH levels. All the experiments were performed under room temperature (25 °C) with a stirring rate of 200 rpm. The concentration of phenol in liquid samples was measured via ultraviolet–visible spectroscopy (UV–vis, Shimadzu UV-2600) using the peak height at 283 nm.

The adsorption capacity of AC at different times (qt) was calculated by eq 1

\[
q_t = \frac{V(C_0 - C_t)}{m}
\]  
(1)

where V (L) is the volume of phenol solution used for each experiment, C₀ (mg L⁻¹) is the initial concentration of phenol, C_t is the concentration of phenol at time t (mg L⁻¹), and m (g) is the amount of AC used for each experiment.

The adsorption capacity of AC at equilibrium (qe) was calculated by eq 2

\[
q_e = \frac{V(C_0 - C_e)}{m}
\]  
(2)

where C_e (mg L⁻¹) is the concentration of phenol at equilibrium.

Pseudo-first-order and pseudo-second-order models were employed to describe the adsorption kinetics in the study. The pseudo-first-order kinetic model can be expressed by eq 3

\[
q_t = q_e - \exp(-k_1t)
\]  
(3)

where \( q_t \) (mg g⁻¹) is the adsorption capacity at time t, \( q_e \) (mg g⁻¹) is the theoretical adsorption capacity at equilibrium, and \( k_1 \) (min⁻¹) is the rate constant of the pseudo-first-order kinetic model.

The pseudo-second-order kinetic model may be expressed by eq 4

\[
q_t = \frac{t}{1 + \frac{t}{q_e k_2}}
\]  
(4)

where \( k_2 \) (g mg⁻¹ min⁻¹) is the rate constant of the pseudo-second-order kinetic model.

To better understand the rate-determining step of the adsorption process, the intraparticle diffusion model was applied for the phenol adsorption process by FWAC.
Weber and Morris (1963) model has been widely used to describe intraparticle diffusion process and it is expressed as

$$q_t = k_{id} \sqrt{t}$$

where $q_t$ is the amount adsorbed at time $t$, and $k_{id}$ (mg g$^{-1}$ min$^{-1/2}$) is the intraparticle diffusion constant. The constant $k_{id}$ is derived by plotting $q_t$ vs $t^{1/2}$ and conducting a linear fitting.

The adsorption equilibrium isotherms were derived by adopting Langmuir and Freundlich isotherm models. The Langmuir isotherm model is commonly applied to model monolayer adsorption on homogeneous adsorbent surfaces, while the Freundlich isotherm model is used for heterogeneous sorption surfaces with nonuniform energy distribution. The general form of the Temkin isotherm model is expressed as

$$q = \frac{Q_0 K_L C_e}{1 + K_L C_e}$$

and as

$$q = K_F C_e^{1/n}$$

where $q_e$ (mg g$^{-1}$) is the adsorption capacity at equilibrium, $C_e$ (mg L$^{-1}$) is the concentration of phenol at equilibrium, $Q_0$ (mg g$^{-1}$) is the maximum adsorption amount of the monomolecular layer, and $K_L$ (L mg$^{-1}$) is the Langmuir constant related to adsorption energy. $K_F$ (mg g$^{-1}$) is the Freundlich constant and $1/n$ is a constant related to adsorption intensity.

In addition to Langmuir and Freundlich isotherm models, the Temkin and Dubinin–Radushkevich (D–R) isotherm models are also well-known models for the adsorption of activated carbons. The Temkin model assumes that the energy of adsorption decreases linearly with the coverage of sorbent surface due to adsorbent–adsorbate interactions. The general form of the Temkin isotherm model is expressed as

$$q = \frac{RT}{b_T \ln K_T C_e}$$

where $T$ is the temperature (298 K), $R$ is the universal gas constant (8.314 J mol$^{-1}$ K$^{-1}$), $b_T$ (J mol$^{-1}$) is the Temkin isotherm constant, and $K_T$ (L mg$^{-1}$) is the equilibrium binding constant.

The D–R isotherm model is based on the adsorption potential theory and assumes that the adsorption process is due to the pore-filling mechanism as opposed to layer-by-layer adsorption. The model can be expressed as

$$q = q_m \exp(-\beta \epsilon^2)$$

where $q_m$ (mg g$^{-1}$) is the maximum adsorption capacity, $\beta$ is the activity coefficient related to mean free adsorption energy (mol$^2$ kJ$^{-1}$), and $\epsilon$ is the Polanyi potential (kJ mol$^{-1}$), which is expressed as

$$\epsilon = RT \ln \left(1 + \frac{1}{C_e}\right)$$

### RESULTS AND DISCUSSION

**Yields of Pyrolysis and Activation.** A set of pyrolysis experiments were performed at different temperatures and residence times to reveal the effects of the operating conditions on the yield and carbon content of biochar from food waste. Figure S1 shows the biochar, sludge (the viscous and dark bio-oil left in the reactor), gas, and liquid yields as a function of pyrolysis temperature. Figure S2 shows the aforementioned yields as a function of pyrolysis residence time. The biochar yield decreased with increasing temperature of pyrolysis. A longer residence time also resulted in slightly lower biochar yields. Elemental analysis was conducted for all the produced biochars, and the results are listed in Table S1. The results show that upon increasing the pyrolysis temperature, the carbon content of biochar increased. A longer residence time showed a positive effect on the carbon content of biochar, which became stable after 60 min of pyrolysis. The carbon content of biochar was as high as 71.9% when food waste was pyrolyzed at 525 °C for 120 min. Therefore, despite the lower yields, these conditions were chosen to produce biochars as the precursor of AC from food waste.

To understand the effects of activation conditions on the surface area and porosity of the produced AC, the temperature and residence time were varied during activation. Thus, steam activation was conducted at a temperature range from 750 to 950 °C at a constant residence time of 3 h. From that set of experiments, the activation temperature that resulted in the AC with the highest surface area was selected. Keeping the activation temperature constant, further experiments were conducted where the residence time was varied from 1 to 5 h.

The yields of activation (based on the mass of biochar) are summarized in Figure S3. It was clear that higher temperatures (up to 950 °C) and longer residence times would result in lower AC yield. Hence, more carbon and other elements were lost during steam activation when a higher temperature and longer residence time were applied.

**Characterization Results.** $N_2$ sorption–desorption isotherms and pore size distributions of steam AC are shown in Figure 1. BET surface areas, micropore volumes, and total pore volumes are shown in Table 1. Before activation, the surface area ($S_{BET}$) of the precursor biochar was 10 m$^2$ g$^{-1}$. The micropore volume ($V_{mic}$) and total pore volume ($V_T$) were 0.004 and 0.016 cm$^3$ g$^{-1}$, respectively. After activation, $S_{BET}$, $V_{mic}$, and $V_T$ of all ACs were one magnitude higher than biochar. The AC with the highest surface area (745 m$^2$ g$^{-1}$)
were formed. Taking from 750 to 950 °C, the total pore volume, $V_T$, increased by increasing the activation temperature, which confirms its low surface area and porosity. After activation at 750 °C for 3 h, the particle showed more small pores that lead to the increase in surface area and pore volume. Activation at 950 °C for 1 h produced highly porous AC. Energy-dispersive X-ray spectroscopy (EDX) was performed for the aforementioned samples with a magnification of 1000× to reveal the regional elements. Mineral elements such as Na, P, and Ca were found in biochar and AC (Table S2).

Elemental analysis was performed for all of the produced AC and the biochar to understand the effects of activation on the changes in N, C, H, and S contents. ICP-OES analysis was also performed to reveal the changes in Ca, Na, and P. Assuming that Ca, Na, and P exist in the activated carbons in the form of CaO, Na, and PO$_4^{3-}$, the oxygen content in the corresponding minerals has been calculated. The elemental compositions of AC and biochar are displayed in Table 2. The washed and dried food waste had an initial carbon content of 48.8% and a relatively high hydrogen content of 7.4%. After pyrolysis, the carbon content of the generated biochar increased, while other elements (mainly oxygen) greatly dropped. AC produced at 950 °C showed a dramatic reduction of carbon content after activation; while a longer residence time resulted in an even more severe reduction of carbon. By studying the carbon contents and the total pore volumes of the produced AC, it can be concluded that, generally, the higher total pore volume would result in lower carbon content. This is probably attributed to the steam reforming of char to form carbon monoxide and hydrogen. Therefore, the higher total pore volume indicates that more carbon is lost during the activation, while the mineral contents are preserved. The ICP results shown in Table 2 provide the concentration of other elements. Before pyrolysis, there was a small amount of Ca, P, and Na in the food waste. Ca was probably from bones and/or milk, Na was from salt, and P might be attributed to meat, beans, and other ingredients. After pyrolysis, the total mass of solids decreased and the concentration of Ca, P, and Na in the char increased. After activation, because of the loss of carbon, the concentration of minerals increased even more. The greater the carbon loss during activation, the higher the Ca and P concentrations in the produced AC. Figure 3a exhibits the X-ray diffraction patterns of biochar and AC. The broad band at 20°–30°, which reaches a maximum at 23° and 26°, and the peak located at 43° is assigned to carbon. Specifically, the peaks at 23° and 26° correspond to the (0 0 2) graphitic plane, while the relatively small peak at 43° is related to the (1 0 0) graphite basal plane. Almost no carbon peaks were identified for AC produced at 950 °C, while sharp peaks at 28°, 31°, and 34.5° were detected, which can be attributed to tricalcium phosphate (Ca$_3$(PO$_4$)$_2$). The presence of tricalcium phosphate peaks should be attributed to the increased concentration of P and Ca after the activation.
process. When a longer residence time was employed, the peaks of calcium phosphate became sharper. This result is in agreement with the elemental analysis and the ICP results. In addition, the peak at 29.5°, which is only observed in the AC produced in the temperature range of 850 to 950 °C, might be assigned to the calcite phase of calcium carbonate.86,87

Figure 3b shows the FTIR results of biochar and AC. The spectra are only displayed in the range of 2000 to 650 cm⁻¹ because no significant bands were found at higher wave-numbers. The peaks at 1589, 1485, and 1406 cm⁻¹ are assigned to C=C bonds of aromatic rings.88−91 The most notable wide band for all ACs and the biochar is observed in the range of 1350 to 900 cm⁻¹, which might indicate the existence of C=O stretching vibrations of alcohols, phenols, acids, ethers, and esters.19,88,92 The intensity of this band increased after activation, suggesting the increased ratio of C=O bonds. However, this suggestion contradicts with the results of 13C NMR and TPD, which revealed no significant increment of C=O bonds after activation (Figures S4 and S5). Thus, we may conclude that this band is more likely attributed to the presence of phosphorus: the peaks located at 1120 and 1050 cm⁻¹ can be assigned to P=O in acid phosphate esters and to symmetrical vibration in a P=O=P chain.93−95 The peak at 980 cm⁻¹ for the AC produced at a temperature higher than 750 °C is attributed to P−O−P stretching31 due to the increased percentage of phosphorus after activation. The biochar and AC produced at 750 °C do not show this peak, possibly because of the lower phosphorus concentration, as

Table 2. Elemental Analysis and ICP Results of Food Waste, Biochar, and AC

| Sample                      | N (wt %) | C (wt %) | H (wt %) | S (wt %) | Others (by difference) | Ca (wt %) | Na (wt %) | P (wt %) | O (wt %) | O content calculated based on minerals
|-----------------------------|---------|----------|----------|----------|------------------------|-----------|-----------|----------|----------|---------------------------------------------|
| Food waste (washed and dried) | 5.7     | 48.8     | 7.4      | 0.8      | 37.3                   | 0.8       | 0.0       | 0.6      | 1.1      | 8875                                       |
| Biochar-525C-2H             | 6.5     | 71.9     | 1.6      | 0.1      | 19.9                   | 3.9       | 0.1       | 1.9      | 4.0      |                                             |
| FWAC-750C-3H               | 5.2     | 68.8     | 0.9      | 0.2      | 24.9                   | 5.2       | 0.2       | 2.7      | 5.6      |                                             |
| FWAC-850C-3H               | 2.5     | 62.6     | 0.5      | 0.3      | 34.1                   | 6.5       | 0.2       | 3.3      | 6.9      |                                             |
| FWAC-950C-3H               | 1.6     | 40.9     | 1.0      | 0.2      | 55.7                   | 13.6      | 0.3       | 7.8      | 15.6     |                                             |
| FWAC-950C-1H               | 1.9     | 48.0     | 1.0      | 0.2      | 48.9                   | 8.8       | 0.2       | 5.6      | 10.8     |                                             |
| FWAC-950C-5H               | 1.2     | 33.2     | 0.6      | 0.2      | 64.8                   | 14.5      | 0.4       | 9.0      | 17.6     |                                             |
revealed by ICP-OES results. The peak at 876 cm\(^{-1}\) is ascribed to the characteristic peak of asymmetric CO\(_3^{2-}\) deformation.\(^{86,87,96}\) This carbonate peak shows the highest intensity for the AC produced at 750 °C and shows lower intensities for the AC produced at higher temperature. This might indicate the formation of CO\(_3^{2-}\) (CaCO\(_3\)) during activation at 750 °C, which then decomposed at higher temperature, therefore causing the loss of carbon.

TPD studies were performed to qualify and quantify the oxygen functional groups of the biochar and the FWAC (sample FWAC-950C-1H), and the results are shown in the Supporting Information. TPD was performed for biochar and FWAC-950C-1H from 60 to 1000 °C at a heating rate of 10 °C/min under an Ar atmosphere. The CO and CO\(_2\) peaks desorbed at various temperatures shown in Figure S5 correspond to the different oxygen functional groups shown in Table S3. For both samples, the CO\(_2\) peaks appear at temperature ranges of 200–400 °C and 650–750 °C, which are attributed to carboxylic acids and lactones, respectively.\(^{97–99}\) The CO signal continued to increase as the temperature increased. The apparent peak at a temperature higher than 850 °C for both materials might be assigned to carboxylic acids and lactones, respectively.\(^{97–99}\) The shoulder from 600 to 800 °C is possibly attributed to phenol groups.\(^{97–99}\) The concentration of the assigned functional groups are shown in Table S3. Apparently, the concentration of oxygen functional groups in FWAC is lower compared to the biochar.

The \(^{13}\)C NMR results shown in the Supporting Information (Figure S4) indicated carbonyl/carboxyl groups, aromatic groups, and methoxyl groups, while the TPD results showed carboxylic acids, lactones, phenols, and carbonyl/quinone groups. Thus, \(^{13}\)C NMR and TPD techniques can complement each other for the detection of oxygen groups. Both characterization results showed that there is no significant difference on the type of oxygen functional groups of biochar and activated carbons. However, the concentration of the oxygen groups in the activated carbons was lower than in biochar.

The Effects of pH. From the characterization results, FWAC-950C-1H (below abbreviated as FWAC) showed the highest surface area; hence, it was selected to test the potential of pollutants (phenol) removal in the aqueous phase. The phenol adsorption experiments were first performed at various pH values (1.94 to 5.39). The solution with pH = 5.39 was prepared using only phenol and DI water, while lower pH was achieved by adding 0.01 M HCl solution. The adsorption experiments were performed with an initial phenol concentration of 30 mg L\(^{-1}\) for 24 h, and the results are shown in Figure 4. As the pH increased from 1.94 to 5.30, the phenol removal increased from 34 to 77 mg g\(^{-1}\). As the pH further increased to 5.39, the capacity slightly decreased. These results indicate that an environment with pH close to neutral is preferred for the removal of phenol from water using FWAC. A similar trend was also reported in the literature.\(^{5,7,100}\)

The Effects of Contact Time and Initial Phenol Concentration. To investigate the adsorption kinetics, the phenol adsorption experiments were conducted at various contact times (from 0.5 to 48 h) and initial phenol concentrations (10 to 50 mg L\(^{-1}\)). All the experiments were performed without adding additional chemicals. The results are shown in Figure 5. Apparently, the adsorption capacity increased as the contact time increased, and after a period of time, the adsorption reached equilibrium. The adsorption was rapid at the beginning of the experiments, and after 2 h, it approached equilibrium. A longer time was required to reach equilibrium as the initial concentration of phenol increased. The maximum phenol removal capability of FWAC at equilibrium increased with higher initial phenol concentration. The highest phenol adsorption capacity was 134 mg g\(^{-1}\), which was achieved with the initial phenol concentration of 50 mg L\(^{-1}\).

To better describe the phenol uptake rate during adsorption, pseudo-first-order and pseudo-second-order kinetic models were employed (Table 3). The models help predict the time required for reaching equilibrium and estimate the maximum adsorption capacity at equilibrium. The parameters were...
derived via nonlinear regression and the fit qualities were evaluated by $R^2$ and ARE. The values of calculated parameters $R^2$ and ARE are displayed in Table 4. The fitted pseudo-first-order and pseudo-second-order kinetic models are plotted in Figures 5 and 6, respectively. Based on the values of coefficient of determination values and average relative errors, it can be concluded that the pseudo-second-order kinetic model describes better the adsorption of phenol on FWAC. The calculated $q_e$ numbers by the pseudo-second-order kinetic model were also closer to the experimental data. Several other studies reported that the pseudo-second-order model describes better the phenol adsorption by activated carbons.1,6,59,101

The intraparticle diffusion model was applied to investigate the rate-controlling step of the adsorption process. The plot of $q_t$ vs $t^{1/2}$ is displayed in Figure 7 and the fitting parameters are shown in Table 4. From the figure, it is apparent that the overall profile did not follow a linear relationship; instead, two portions can be distinguished in the graph: a sharp first region and a low slope second region. As the profiles were not linear and the fitted models did not pass though the origin, it can be concluded that the intraparticle diffusion was not the sole rate-controlling step of the adsorption. The dual-stage behavior might be attributed to the multistep process where the first step involved the transportation of phenol molecules from bulk solution to the external surface of the adsorbent and the second stage was dominated by intraparticle diffusion.10,102

**Adsorption Isotherm Models.** To better understand the adsorption behavior of phenol adsorption on FWAC, several other studies reported that the pseudo-second-order model describes better the phenol adsorption by activated carbons.1,6,59,101

| Table 4. Fitting Parameters for Isotherm Models |
|-----------------------------------------------|
| $K_L$ (L mg$^{-1}$) | $Q_0$ (mg g$^{-1}$) | $R^2$ | ARE (%) |
|---------------------|---------------------|-------|---------|
| Langmuir isotherm   | 0.0071              | 760.76| 0.9942  | 7.6830  |
| Freundlich isotherm | 22.925              | 1.824 | 0.9851  | 19.6331 |
| Temkin isotherm     | 19.10               | 0.143 | 0.9537  | 37.9574 |
| D–R isotherm        | 524.41              | 673.243| 0.8864  | 47.8096 |

| Table 3. Kinetic Parameters Calculated from Pseudo-First-Order and Pseudo-Second-Order Kinetic Models and the Intraparticle Diffusion Model |
|-------------------------------------------------------------------------------------------------------------------------------------|
| parameters                                                                 | 10     | 20     | 30     | 40     | 50     |
| pseudo-first-order                                                              |        |        |        |        |        |
| $k_1$ (min$^{-1}$)                                                              | 0.0315 | 0.0119 | 0.0106 | 0.0097 | 0.0156 |
| $q_e$ (mg g$^{-1}$, calc)                                                       | 26.73  | 44.28  | 69.27  | 99.46  | 119.99 |
| $R^2$                                                                          | 0.9785 | 0.9865 | 0.9875 | 0.9793 | 0.9778 |
| ARE (%)                                                                       | 3.3615 | 6.0051 | 7.2607 | 10.9873| 11.4154|
| pseudo-second-order                                                            |        |        |        |        |        |
| $k_2$ (g mg$^{-1}$ min$^{-1}$)                                                  | 2.00 $\times$ 10$^{-3}$ | 3.50 $\times$ 10$^{-4}$ | 2.15 $\times$ 10$^{-4}$ | 1.34 $\times$ 10$^{-4}$ | 1.74 $\times$ 10$^{-4}$ |
| $q_e$ (mg g$^{-1}$, calc)                                                       | 28.12  | 48.39  | 75.15  | 108.50 | 130.64 |
| $R^2$                                                                          | 0.9785 | 0.9865 | 0.9875 | 0.9793 | 0.9778 |
| ARE (%)                                                                       | 3.3615 | 6.0051 | 7.2607 | 10.9873| 11.4154|
| intraparticle diffusion                                                        |        |        |        |        |        |
| $k_d$ (g mg$^{-1}$ min$^{-1/2}$)                                                | 0.155  | 0.393  | 0.599  | 1.173  | 1.417  |
| intercept (mg g$^{-1}$)                                                         | 21.720 | 30.264 | 48.829 | 63.468 | 81.657 |
| $R^2$                                                                          | 0.448  | 0.803  | 0.529  | 0.726  | 0.676  |
| $q_e$ (mg g$^{-1}$, exp)                                                       | 28.24 ± 0.57 | 48.05 ± 1.38 | 73.26 ± 1.54 | 108.80 ± 3.01 | 134.36 ± 7.19 |
| removal efficiency (%)                                                          | 56%    | 48%    | 49%    | 54%    | 54%    |

References:
1,6,59,101

**Figure 5.** Pseudo-first-order kinetic fitting of phenol adsorption at various initial concentrations.

**Figure 6.** Pseudo-second-order kinetic fitting of phenol adsorption at various initial concentrations.
adsorption isotherm models were studied. The initial concentration of phenol was increased up to 500 mg L$^{-1}$ to ensure the accuracy of model parameters. All the experiments lasted for 48 h to ensure the completion of adsorption. The Langmuir, Freundlich, Temkin, and D–R isotherm models were fitted to experimental data. The fitting parameters were derived via nonlinear regression and are shown in Table 4. Figure 8 shows the experimental data and fitted Langmuir and Freundlich isotherm models. The adsorption capacity at equilibrium increased as the initial concentration increased, which is consistent with the last section. With an initial concentration of 500 mg L$^{-1}$, the adsorption capacity reached a remarkable value of 568 mg g$^{-1}$. From Figure 8, it appears that the Langmuir model fits better the experimental data, which is supported by the higher $R^2$ value and the lower ARE.58,59,105 The fitted curves of Temkin and D–R models showed clear deviation from experimental data (Figure 9). The corresponding $R^2$ and ARE values also showed that these models were not suitable for describing phenol adsorption by FWAC. The results also indicate that the adsorption of phenol did not follow the pore-filling mechanism. Therefore, the Langmuir model might be the most appropriate model to describe the adsorption of phenol on FWAC. Lv et al.59, Kumar and Jena,1 and Yao et al.106 have also found that the Langmuir model was the most suitable model to describe phenol adsorption on activated carbons.

**Comparison of AC Produced by Food Waste and Other Biomass Resources.** The phenol removal capacity is affected by various factors (such as pH, initial concentration, dosage, and temperature). Thus, a direct comparison between the adsorption capacity of sorbents in this study and others in the literature is difficult. In Table 5, we present the adsorption capacity of our FWAC, as well as activated carbons derived from other resources, while giving experimental conditions. Although the experimental conditions vary, the table can provide a rough estimation on the performance of the sorbents. It is observed that the surface area of AC derived from different biomass can vary significantly. For example, ACs from rice husk show a much higher surface area than ACs from other resources. The FWAC in this study showed a moderately high surface area. The phenol adsorption experiments are commonly performed at room temperature (25–30 °C) and initial phenol concentrations lower than 500 mg L$^{-1}$. The sorbent dosage is commonly between 1 and 2 g L$^{-1}$. This study uses less AC than most of references because the yield of FWAC was low and a limited amount was available for adsorption experiments.

The maximum adsorption capacity of FWAC in this study is very high compared to the literature; one potential reason is the high mesoporosity of the FWAC used for adsorption experiments that might enhance the transport of phenol within the AC particles and can be beneficial for adsorption. To verify the effects of mesoporosity of AC, the sample FWAC-850C-3H (Table 1), which showed a similar micropore volume but lower mesopore volume compared to FWAC-950C-1H, was tested for phenol adsorption. The experiments were conducted with an initial phenol concentration of 50 mg L$^{-1}$ and lasted for 48 h. The adsorption capacity of FWAC-850C-3H was found to be 109.12 ± 4.58 mg g$^{-1}$, which was lower than FWAC-950C-1H (134.36 ± 7.19 mg g$^{-1}$). The lower surface oxygen group concentration of FWAC-950C-1H as shown by TPD results, compared to the surface oxygen groups of other activated carbons reported in the literature,74,99,107 might be another possible explanation for the high phenol removal capacity of FWAC. Based on the literature, surface oxygen groups are adverse to adsorption of organic compounds from...
order and pseudo-second-order kinetic models and an intraparticle diffusion model were employed to study the kinetics of adsorption. Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models were applied to describe the adsorption behavior of FWAC. It was found that the adsorption of phenol on FWAC follows a pseudo-second-order kinetic model and the isotherm was more accurately represented by the Langmuir model. The pseudo-second-order kinetic model showed $R^2$ values >0.99 for most of initial concentrations studied in this work. The Langmuir model also showed an $R^2$ value of 0.9942 and suggested $Q_e$ as 760.76 mg g$^{-1}$. The results of intraparticle diffusion model fitting show that the intraparticle diffusion was not the sole rate-controlling step of the adsorption. The highest phenol adsorption capacity was 568 mg g$^{-1}$, achieved with an initial phenol concentration of 500 mg L$^{-1}$. The proposed FWAC showed a remarkable potential for phenol removal, comparable with other biomass-derived activated carbons reported in the literature. However, we notice that the yield of high-surface-area activated carbon from food waste was low (around 10 wt %), which means that a balance between yield and surface area has to be made when activated carbons from food waste are prepared. Overall, the results of this study demonstrate that the utilization of food waste to produce renewable and sustainable AC can be an excellent food waste management solution.

## CONCLUSIONS

In this study, activated carbons were prepared from food waste and evaluated for phenol adsorption in aqueous solutions. The ACs were produced via two-step activation: pyrolysis and activation. The pyrolysis and activation conditions were varied to study their effects on the properties of the produced AC. The results demonstrated that the activation temperature and residence time greatly affect the surface area and pore volume of the produced AC. Elemental, FTIR, and XRD analyses indicated that the ACs were mainly composed of carbon, while other elements (such as Ca, P, etc.) were also found in their structure due to the complex nature and origin of food waste. The FWAC with the highest surface area (745 m$^2$ g$^{-1}$) was derived after physical (steam) activation at 950 °C for 1 h of the corresponding biochar and was tested for phenol adsorption from aqueous solutions. The effects of pH, contact time, and initial concentration of phenol on the adsorption capacity of FWAC were investigated. A neutral solution was found to be beneficial for phenol adsorption. Pseudo-first-

| starting material                        | maximum adsorption capacity (mg g$^{-1}$) | initial concentration (mg L$^{-1}$) | $T$ (°C) | adsorbent dosage (g L$^{-1}$) | BET surface area (m$^2$ g$^{-1}$) | reference |
|------------------------------------------|------------------------------------------|------------------------------------|----------|-----------------------------|----------------------------------|-----------|
| black wattle bark waste                  | 98.6                                     | 500                                | 55       | 1                           | 414                              | 58        |
| corn husk                                | 7.8                                      | 30                                 | 25       | 2                           | 114                              |           |
| oil-palm shell                           | 168                                      | 200                                | 30       | 0.17                        | 898                              | 115       |
| rice husk                                | 194.24                                   | 500                                | 25       | 2                           | 2087                             | 59        |
| Borassus flabellifer fruit husk           | 5.94                                     | 10                                 | 1.6      | 116                         |                                  |           |
| tea residue                              | 320                                      | 1400                               | 30       | 1                           | 819                              | 117       |
| rice husk                                | 201                                      | 500                                | 25       | 2                           | 2138                             | 118       |
| food waste                               | 568                                      | 500                                | 25       | 0.2                         | 745                              | this work |

| maximum adsorption capacity (mg g$^{-1}$) | initial concentration (mg L$^{-1}$) | $T$ (°C) | adsorbent dosage (g L$^{-1}$) | BET surface area (m$^2$ g$^{-1}$) | reference |
|------------------------------------------|------------------------------------|----------|-----------------------------|----------------------------------|-----------|
| black wattle bark waste                  | 98.6                               | 500      | 55                          | 1                                | 414       | 58        |
| corn husk                                | 7.8                                | 30       | 25                          | 2                                | 114       |           |
| oil-palm shell                           | 168                                | 200      | 30                          | 0.17                             | 898       | 115       |
| rice husk                                | 194.24                             | 500      | 25                          | 2                                | 2087      | 59        |
| Borassus flabellifer fruit husk           | 5.94                               | 10       | 1.6                        | 116                              | 117       |
| tea residue                              | 320                                | 1400     | 30                          | 1                                | 819       | 118       |
| rice husk                                | 201                                | 500      | 25                          | 2                                | 2138      | 11     |
| food waste                               | 568                                | 500      | 25                          | 0.2                             | 745       | this work |

### ASSOCIATED CONTENT

#### Supporting Information

The Supporting Information is available free of charge at [https://pubs.acs.org/doi/10.1021/acsomega.0c06029](https://pubs.acs.org/doi/10.1021/acsomega.0c06029).

Product yields of pyrolysis as a function of temperature, product yields of pyrolysis as a function of residence time, biochar-to-AC yield under different activation temperatures and residence times, elemental analysis of biochars derived under different conditions, EDX results for biochar and AC, $^{13}$C NMR spectrum of biochar and AC, and TPD results for biochar and AC (PDF)

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Notes
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■ REFERENCES

(1) Kumar, A.; Jena, H. M. Removal of Methylene Blue and Phenol onto Prepared Activated Carbon from Fox Nutshell by Chemical Activation in Batch and Fixed-Bed Column. J. Cleaner Prod. 2016, 137, 1246–1259.
(2) Ahmed, M. J.; Theydan, S. K. Physical and Chemical Characteristics of Activated Carbon Prepared by Pyrolysis of Chemically Treated Date Stones and Its Ability to Adsorb Organics. Powder Technol. 2012, 229, 237–245.
(3) Srivastava, V. C.; Swamy, M. M.; Mall, I. D.; Prasad, B.; Mishra, I. M. Adsorptive Removal of Phenol by Bagasse Fly Ash and Activated Carbon: Equilibrium, Kinetics and Thermodynamics. Colloids Surf., A 2006, 272, 89–104.
(4) Karri, R. R.; Sahu, J. N.; Jayakumar, N. S. Optimal Isotherm Parameters for Phenol Adsorption from Aqueous Solutions onto Coconut Shell Based Activated Carbon: Error Analysis of Linear and Non-Linear Methods. J. Taiwan Inst. Chem. Eng. 2017, 80, 472–487.
(5) Rodrigues, L. A.; da Silva, M. L. C. P.; Alves-Mendes, M. O.; dos Reis Coutinho, A.; Thim, G. P. Phenol Removal from Aqueous Solution by Activated Carbon Produced from Avocado Kernel Seeds. Chem. Eng. J. 2011, 174, 49–57.
(6) Huiruddin, M. N.; Mubarak, N. M.; Khalid, M.; Abdullah, E. C.; Walvekar, R.; Karri, R. R. Magnetic Palm Kernel Biochar Potential Route for Phenol Removal from Wastewater. Environ. Sci. Pollut. Res. 2019, 26, 35183–35197.
(7) Kilic, M.; Ayapdin-Varol, E.; Pütünn, A. E. Adsorptive Removal of Phenol from Aqueous Solutions on Activated Carbon Prepared from Tobacco Residues: Equilibrium, Kinetics and Thermodynamics. J. Hazard. Mater. 2011, 189, 397–403.
(8) Han, Y.; Boateng, A. A.; Qi, P. X.; Lima, I. M.; Chang, J. Heavy Metal and Phenol Adsorptive Properties of Biochars from Pyrolyzed Switchgrass and Woody Biomass in Correlation with Surface Properties. J. Environ. Manage. 2013, 118, 196–204.
(9) Liu, X.; Han, Y.; Cheng, Y.; Xu, G. Microwave-Assisted Ammonia Modification of Activated Carbon for Effective Removal of Phenol from Wastewater: DFT and Experiment Study. Appl. Surf. Sci. 2020, 518, 146258.
(10) Karri, R. R.; Jayakumar, N. S.; Sahu, J. N. Modelling of Fluidised-Bed Reactor by Differential Evolution Optimization for Phenol Removal Using Coconut Shells Based Activated Carbon. J. Mol. Liq. 2017, 231, 249–262.
(11) Ahmed, S.; Rasul, M. G.; Martens, W. N.; Brown, R.; Hashim, M. A. Heterogeneous Photocatalytic Degradation of Phenols in Wastewater: A Review on Current Status and Developments. Desalination 2010, 261, 3–18.
(12) Jeong, H.; Lee, J.; Ju, Y. M.; Lee, S. M. Using Electro-Coagulation Treatment to Remove Phenolic Compounds and Furan Derivatives in Hydrolysates Resulting from Pilot-Scale Supercritical Water Hydrolysis of Mongolian Oak. Renewable Energy 2019, 138, 971–979.
(13) Zhang, M.; Zhang, Z.; Liu, S.; Peng, Y.; Chen, J.; Yoo, K. S. Ultrasound-Assisted Electrochemical Treatment for Phenolic Waste- Water. Ultrason. Sonochem. 2020, 65, 105058.
(14) Zhang, H.; Zhang, X.; Ding, L. Partial Oxidation of Phenol in Supercritical Water with NaOH and H₂O₂: Hydrogen Production and Polymer Formation. Sci. Total Environ. 2020, 722, 137985.
(15) Chiang, C.-H.; Chen, J.; Lin, J.-H. Preparation of Pore-Size Tunable Activated Carbon Derived from Waste Coffee Grounds for High Adsorption Capacities of Organic Dyes. J. Environ. Chem. Eng. 2020, 8, 103929.
(16) Lee, K. X.; Valla, J. A. Adsorptive Desulfurization of Liquid Hydrocarbons Using Zeolite-Based Sorbents: A Comprehensive Review. React. Chem. Eng. 2019, 4, 1357–1386.
(17) Dight, L. B.; Garcia-Martinez, J.; Valla, I.; Johnson, M. M. United States Patent US8,524,625B2, 2013, 2 (12).
(18) Yahya, M. A.; Al-Qodah, Z.; Ngah, C. W. Z. Agricultural Bio-Waste Materials as Potential Sustainable Precursors Used for Activated Carbon Production: A Review. Renewable Sustainable Energy Rev. 2015, 46, 218–235.
(19) Zubrik, A.; Matik, M.; Hredzák, S.; Lováš, M.; Danková, Z.; Kovačová, M.; Briancín, J. Preparation of Chemically Activated Carbon from Waste Biomass by Single-Stage and Two-Stage Pyrolysis. J. Cleaner Prod. 2017, 143, 643–653.
(20) Gamliel, D. P.; Cho, H. J.; Fan, W.; Valla, J. A. On the Effectiveness of Tailored Mesoporous MFI Zeolites for Biomass Catalytic Fast Pyrolysis. Appl. Catal., A 2016, 522, 109–119.
(21) Gamliel, D. P.; Bollas, G. M.; Valla, J. A. Two-Stage Catalytic Fast Hydropyrolysis of Biomass for the Production of Drop-in Biofuel. Fuel 2018, 216, 160–170.
(22) Gamliel, D. P.; Bailie, B. P.; Augustine, E.; Hall, J.; Bollas, G. M.; Valla, J. A. Nickel Impregnated Mesoporous USY Zeolites for Hydrodeoxygenation of Anisole. Microporous Mesoporous Mater. 2018, 261, 18–28.
(23) Yu, L.; Farinmade, A.; Ajumobi, O.; Su, Y.; John, V. T.; Valla, J. A. MCM-41/ZSM-5 Composite Particles for the Catalytic Fast Pyrolysis of Biomass. Appl. Catal., A 2020, 602, 117727.
(24) Farinmade, A.; Ajumobi, O.; Yu, L.; Su, Y.; He, J.; Valla, J. A.; John, V. A. One-Step Facile Encapsulation of Zeolite Microcrytallites in Ordered Mesoporous Microspheres. Ind. Eng. Chem. Res. 2020, 59, 13923–13931.
(25) Bouchelta, C.; Medjram, M. S.; Bertrand, O.; Bellat, J.-P. Preparation and Characterization of Activated Carbon from Date Stones by Physical Activation with Steam. J. Anal. Appl. Pyrolysis 2008, 82, 70–77.
(26) Cagnon, B.; Py, X.; Guillo, A.; Stocklki, F. The Effect of the Carbonization/Activation Procedure on the Microporous Texture of the Subsequent Chars and Active Carbons. Microporous Mesoporous Mater. 2003, 57, 273–282.
(27) Daud, W. M. A. W.; Ali, W. S. W.; Sulaiman, M. Z. The Effects of Carbonization Temperature on Pore Development in Palm-Shell-Based Activated Carbon. Carbon 2000, 38, 1925–1932.
(28) González, J. F.; Román, S.; Encinar, J. M.; Martínez, G. Pyrolysis of Various Biomass Residues and Char Utilization for the Production of Activated Carbons. J. Anal. Appl. Pyrolysis 2009, 85, 134–141.
(29) Ahmadpour, A.; Do, D. D. The Preparation of Active Carbons from Coal by Chemical and Physical Activation. Carbon 1996, 34, 471–479.
(30) Açıkylıdzı, M.; Gürses, A.; Karaca, S. Preparation and Characterization of Activated Carbon from Plant Wastes with Chemical Activation. Microporous Mesoporous Mater. 2014, 198, 45–49.
(31) Danesh, M.; Hashim, R.; Ibrahim, M. N. M.; Sulaiman, O. Effect of Acidic Activating Agents on Surface Area and Surface Functional Groups of Activated Carbons Produced from Acacia Mangium Wood. J. Anal. Appl. Pyrolysis 2013, 104, 418–425.

(32) Deng, H.; Yang, L.; Tao, G.; Dai, J. Preparation and Characterization of Activated Carbon from Cotton Stalk by Microwave Assisted Chemical Activation-Application in Methylene Blue Adsorption from Aqueous Solution. J. Hazard. Mater. 2009, 166, 1514–1521.

(33) El-Hendawy, A. N. A.; Samra, S. E.; Girgis, B. S. Adsorption Characteristics of Activated Carbons Obtained from Corncobs. Colloids Surf., A 2001, 180, 209–221.

(34) Ahmad, A. A.; Hameed, B. H. Fixed-Bed Adsorption of Reactive Azo Dye onto Granular Activated Carbon Prepared from Waste. J. Hazard. Mater. 2010, 175, 298–303.

(35) Ma, X.; Ouyang, F. Adsorption Properties of Biomass-Based Activated Carbon Prepared with Spent Coffee Grounds and Pomelo Shell for Zn (II) Disposal from the Aqueous Environment Using Sulfuric Acid Activation and Their Use on Phenolic Compounds from Aqueous Solution Using MgCl₂-Activated Activated Carbon. Vitrochim. Technol. 2008, 99, 6214–6222.

(36) Lozano-Castelló, D.; Lillo-Ródenas, M. A.; Cazorla-Amorós, D.; Linareś-Solano, A. Preparation of Activated Carbons from Spanish Anthracite: I. Activation by KOH. Carbon N. Y. 2001, 39, 741–749.

(37) Tseng, R.-L. Physical and Chemical Properties and Adsorption Type of Activated Carbon Prepared from Plum Kernels by NaOH Activation. J. Hazard. Mater. 2007, 147, 1020–1027.

(38) Adinata, D.; Wan Daud, W. M. A.; Aroua, M. K. Preparation and Characterization of Activated Carbon from Palm Shell by Chemical Activation with K₂CO₃. Bioresour. Technol. 2007, 98, 145–149.

(39) Saygılı, H.; Güzel, F. High Surface Area Mesoporous Activated Carbon from Tomato Processing Solid Waste by Zinc Chloride Activation: Process Optimization, Characterization and Dyes Adsorption. J. Cleaner Prod. 2016, 113, 995–1004.

(40) Sych, N. V.; Trofymenko, S. I.; Pudubnaya, O. I.; Tsyba, M. M.; Saygılı, H.; Ilyumchik, D. O.; Puziy, A. M. Porous Structure and Surface Chemistry of Phosphoric Acid Activated Carbon from Corncob. Appl. Surf. Sci. 2012, 261, 75–82.

(41) Karri, R. R.; Sahu, J. N. Modeling and Optimization by Particle Swarm Embedded Neural Network for Adsorption of Zinc (II) by Palm Kernel Shell Based Activated Carbon from Aqueous Environment. J. Environ. Manage. 2018, 206, 178–191.

(42) Karri, R. R.; Sahu, J. N. Process Optimization and Adsorption Modeling Using Activated Carbon Derived from Palm Oil Kernel Shell for Zn (II) Disposal from the Aqueous Environment Using Differential Evolution Embedded Neural Network. J. Mol. Liq. 2018, 265, 592–602.

(43) Ozdemir, I.; Sahin, M.; Orhan, R.; Erdem, M. Preparation and Characterization of Activated Carbon from Grape Stalk by Zinc Chloride Activation. Fuel Process. Technol. 2014, 125, 200–206.

(44) Hesse, R. H.; Arimi-Niya, A.; Daan, W. M. A.; Sahu, J. N. Preparation and Characterization of Activated Carbon from Apple Waste by Microwave-Assisted Phosphoric Acid Activation: Application in Methenylene Blue Adsorption. BioResources 2013, 8, 2950–2966.

(45) Hu, Z.; Srivivasan, M. P. Mesoporous High-Surface-Area Activated Carbon. Microporous Mesoporous Mater. 2001, 43, 267–275.

(46) Ad Niazi, L.; Lashanizadegan, A.; Shariflārd, H.; Chestnut O.K. Shells Activated Carbon: Preparation, Characterization and Application for Cr (VI) Removal from Dilute Aqueous Solutions. J. Cleaner Prod. 2018, 185, 554–561.

(47) Food and Agriculture Organization. Towards the Future We Want: End Hunger and Make the Transition to Sustainable Agricultural and Food Systems. FAO: 2012, 1–42.

(48) Uşkun Kiran, E.; Trzciński, A. P.; Ng, W. J.; Liu, Y. Bioconversion of Food Waste to Energy: A Review. Fuel 2014, 134, 389–399.

(49) Girotto, F.; Alibardi, L.; Cossu, R. Food Waste Generation and Industrial Uses: A Review. Waste Manage. 2015, 45, 32–41.

(50) Cheng, F.; Tomsett, G. A.; Murphy, C. M.; Maag, A. R.; Caraballo, N.; Bailey, M.; Hemingway, J. J.; Romo, C. I.; Paulsen, A. D.; Yelvington, P. E.; et al. Synergistic Effects of Inexpensive Mixed Metal Oxides for Catalytic Hydrothermal Liquefaction of Food Wastes. ACS Sustainable Chem. Eng. 2020, 8, 6877–6886.

(51) Ebbikade, E.; Athaley, A.; Fisher, B.; Yang, K.; Wu, C.; Ierapetritou, M. G.; Vlachos, D. G. The Future Is Garbage: Repurposing of Food Waste to an Integrated Biorefinery. ACS Sustainable Chem. Eng. 2020, 8, 8124–8136.

(52) Pham, T. P. T.; Kaushik, R.; Parshetti, G. K.; Mahmood, R.; Balasubramanian, R. Food Waste-to-Energy Conversion Technologies: Current Status and Future Directions. Waste Manage. 2015, 38, 399–408.

(53) Katami, T.; Yasuhara, A.; Shibamoto, T. Formation of Dioxins from Incineration of Foods Found in Domestic Garbage. Environ. Sci. Technol. 2004, 38, 1062–1065.

(54) Brown, A. B.; Tomsett, G. A.; Partoubour, P.; Deskins, N. A.; Timko, M. T. Hydrochar Structural Determination from Raman-Free Analysis. Carbon 2020, 167, 378–387.

(55) Li, L.; Diederick, R.; Flora, J. R. V.; Berge, N. D. Hydrothermal Carbonization of Food Waste and Associated Packaging Materials for Energy Source Generation. Waste Manage. 2013, 33, 2478–2492.

(56) Lütke, S. F.; Iagni, A. V.; Pegoraro, L.;otto, G. L.; Pinto, L. A. C.; Badal, T. R. S., Jr. Preparation of Activated Carbon from Black Wattle Bark Waste and Its Application for Phenol Adsorption. J. Environ. Chem. Eng. 2019, 7, 103396.

(57) Lu, S.; Li, C.; Mi, J.; Meng, H. A Functional Activated Carbon for Efficient Adsorption of Phenol Derived from Pyrolysis of Rice Husk, KOH-Activation and EDTA-4Na-Modification. Appl. Surf. Sci. 2020, 510, 145425.

(58) Hamadneh, I.; Abu-Zarayk, R. A.; Al-Dujaili, A. H. Removal of Phenolic Compounds from Aqueous Solution Using MgCl₂-Impregnated Activated Carbons Derived from Olive Husk: The Effect of Chemical Structures. Water Sci. Technol. 2020, 81, 2351–2367.

(59) Supong, A.; Bhomick, P. C.; Karmaker, R.; Ezung, S. L.; Jamir, L.; Sinha, U. B.; Sinha, D. Experimental and Theoretical Insight into the Adsorption of Phenol and 2,4-Dinitrophenol onto Tithonia Diversifolia Activated Carbon. Appl. Surf. Sci. 2020, 529, 147046.

(60) Zhang, F.; Zhang, S.; Chen, L.; Liu, Z.; Qin, J. Utilization of Bark Waste of Acacia Mangium: The Preparation of Activated Carbon and Adsorption of Phenolic Wastewater. Ind. Crops Prod. 2021, 160, 113157.

(61) Shehzad, A.; Bashir, M. J. K.; Sethupathi, S.; Lim, J.-W. An Overview of Heavily Polluted Landfill Leachate Treatment Using Food Waste as an Alternative and Renewable Source of Activated Carbon. Process Saf. Environ. Prot. 2015, 98, 309–318.

(62) Kosheleva, R. I.; Mitropoulos, A. C.; Kyzas, G. Z. Synthesis of Activated Carbon from Food Waste. Environ. Chem. Lett. 2019, 17, 429–438.

(63) Pollard, Z. A.; Goldfarb, J. L. Valorization of Cherry Pits: Great Lakes Agro-Industrial Waste to Mediate Great Lakes Water Quality. Environ. Pollut. 2021, 270, 116073.

(64) Santos-Clotet, E.; Cabrera-Codony, A.; Ruiz, B.; Fuente, E.; Martin, M. J. Sewage Biogas Efficient Purification by Means of Lignocellulosic Waste-Based Activated Carbons. Bioresour. Technol. 2019, 285, 207–215.

(65) Lee, C.-G.; Hong, S.-H.; Hong, S.-G.; Choi, J.-W.; Park, S.-J. Production of Biochar from Food Waste and Its Application for Phenol Removal from Aqueous Solution. Water, Air, Soil Pollut. 2019, 230, 1.

(66) Long, C.; Lu, J.; Li, A.; Hu, D.; Liu, F.; Zhang, Q. Adsorption of Naphthalene onto the Carbon Adsorbent from Waste Ion Exchange
Resin: Equilibrium and Kinetic Characteristics. J. Hazard. Mater. 2008, 150, 656–661.

(69) Hernández-Abreu, A. B.; Álvarez-Torrellas, S.; Águeda, V. I.; Larriba, M.; Delgado, J. A.; Calvo, P. A.; García, J. Enhanced Removal of the Endocrine Disruptor Compound Bisphenol A by Adsorption onto Green-Carbon Materials. Effect of Real Effluents on the Adsorption Process. J. Environ. Manage. 2020, 266, 110604.

(70) Tang, Y.; Zhao, Y.; Lin, T.; Li, Y.; Zhou, R.; Peng, Y. Adsorption Performance and Mechanism of Methylene Blue by H2PO4-, Modified Corn Stalks. J. Environ. Chem. Eng. 2019, 7, 103398.

(71) Anbia, M.; Moradi, S. E. Adsorption of Naphthalene-Derived Compounds from Water by Chemically Oxidized Nanoporous Carbon. Chem. Eng. J. 2009, 148, 452–458.

(72) Boparai, H. K.; Joseph, M.; O’Carroll, D. M. Kinetics and Thermodynamics of Cadmium Ion Removal by Adsorption onto Nano Zerovalent Iron Particles. J. Hazard. Mater. 2011, 186, 458–465.

(73) Sahin, R.; Tapadia, K. Comparison of Linear and Non-Linear Models for the Adsorption of Fluoride onto Geo-Material: Limonite. Water Sci. Technol. 2015, 72, 2262–2269.

(74) Li, B.; Yang, Y.; Wu, H.; Zhang, C.; Zheng, W.; Sun, D. Adsorptive Removal and Mechanism of Monocyclic Aromatics by Activated Carbons from Water: Effects of Structure and Surface Chemistry. Colloids Surf, A 2020, 605, 125346.

(75) Hu, Q.; Zhang, Z. Application of Dubinin–Radushkevich Isotherm Model at the Solid/Solution Interface: A Theoretical Analysis. J. Mol. Liq. 2019, 277, 646–648.

(76) Bonilla-Petriciolet, A.; Mendoza-Castillo, D. I.; Reynel-Avila, H. E. Adsorption Processes for Water Treatment and Purification; Springer: 2017, DOI: 10.1007/978-3-319-58136-1.

(77) Dehghani, M. H.; Karri, R. R.; Yeganeh, Z. T.; Mahvi, A. H.; Nourmoradi, H.; Sarali, M.; Zarei, A.; Sillánpää, M. Statistical Modelling of Endocrine Disrupting Compounds Adsorption onto Activated Carbon Prepared from Wood Using CDD-RSM and DE Hybrid Evolutionary Optimization Framework: Comparison of Linear vs Non-Linear Isotherm and Kinetic Parameters. J. Mol. Liq. 2020, 302, 112526.

(78) Karri, R. R.; Sahu, J. N.; Meikap, B. C. Improving Efficacy of Cr (VI) Adsorption Process on Sustainable Adsorbent Derived from Waste Biomass (Sugarcane Bagasse) with Help of Ant Colony Optimization. Ind. Crop. Prod. 2020, 143, 111927.

(79) Dehghani, M. H.; Karri, R. R.; Yeganeh, Z. T.; Mahvi, A. H.; Nourmoradi, H.; Sarali, M.; Zarei, A.; Sillánpää, M. Statistical Modelling of Endocrine Disrupting Compounds Adsorption onto Activated Carbon Prepared from Wood Using CDD-RSM and DE Hybrid Evolutionary Optimization Framework: Comparison of Linear vs Non-Linear Isotherm and Kinetic Parameters. J. Mol. Liq. 2020, 302, 112526.

(80) Li, W.; Yang, K.; Peng, J.; Zhang, L.; Guo, S.; Xie, H. Effects of Carbonization Temperatures on Characteristics of Porosity in Coconut Shell Char and Activated Carbons Derived from Carbonized Coconut Shell Charls. Ind. Crops Prod. 2008, 28, 190–198.

(81) Asadullah, M.; Rahman, M. A.; Motin, M. A.; Sultan, M. B. Adsorption Studies on Activated Carbon Derived from Steam Activation of Jute Stuck Char. J. Surf. Sci. Technol. 2007, 23, 73.

(82) Sivachidambaram, M.; Vijaya, J. J.; Kennedy, L. J.; Jothiramalingam, R.; Al-Lohedan, H. A.; Munusamy, M. A.; Elanthamilan, E.; Merlin, J. P. Preparation and Characterization of Activated Carbon Derived from the Borassus Flabellifer Flower as an Electrode Material for Supercapacitor Applications. New J. Chem. 2017, 41, 3939–3949.

(83) Wang, G.; Pan, C.; Wang, L.; Dong, Q.; Yu, C.; Zhao, Z.; Qu, J. Activated Carbon Nanofiber Webs Made by Electrospinning for Capacitive Deionization. Electrochem. Acta 2012, 69, 65–70.

(84) Zhao, J.; Yang, L.; Li, F.; Yu, R.; Jia, C. Structural Evolution in the Graphitization Process of Activated Carbon by High-Pressure Sintering. Carbon 2009, 47, 744–751.

(85) Miculescu, F.; Stan, G. E.; Ciocan, L. T.; Miculescu, M.; Berbecaru, A.; Antoniac, I. Cortical Bone as Resource for Producing Biomimetic Materials for Clinical Use. Dig. J. Nanomater. Bionanotechnol. 2012, 7, 1667–1677.
Textural and Surface Properties. *Microporous Mesoporous Mater.* 2008, 111, 276–284.

(103) Wang, S.; Nam, H.; Nam, H. Preparation of Activated Carbon from Peanut Shell with KOH Activation and Its Application for H2S Adsorption in Confined Space. *J. Environ. Chem. Eng.* 2020, 8, 103683.

(104) Cheung, W. H.; Szeto, Y. S.; McKay, G. Intraparticle Diffusion Processes during Acid Dye Adsorption onto Chitosan. *Bioresour. Technol.* 2007, 98, 2897–2904.

(105) Córtes, L. N.; Druzian, S. P.; Streit, A. F. M.; Godinho, M.; Perondi, D.; Collazzo, G. C.; Oliveira, M. L. S.; Cadaval, T. R. S., J.; Dotto, G. L. Biochars from Animal Wastes as Alternative Materials to Treat Colored Effluents Containing Basic Red 9. *J. Environ. Chem. Eng.* 2019, 7, 10346.

(106) Yao, P.; Cen, J.; Fang, M.; Wang, T.; Wang, Q. A Study on the Preparation of Pitch-Based High-Strength Columnar Activated Carbon and Mechanism of Phenol Adsorption from Aqueous Solution. *RSC Adv.* 2018, 8, 17558–17568.

(107) Moreno-Castilla, C.; López-Ramón, M. V.; Carrasco-Marín, F. Changes in Surface Chemistry of Activated Carbons by Wet Oxidation. *Carbon* 2000, 38, 1995–2001.

(108) Partlan, E.; Ren, Y.; Apul, O. G.; Ladner, D. A.; Karanfil, T. Adsorption Kinetics of Synthetic Organic Contaminants onto Superfine Powdered Activated Carbon. *Chemosphere* 2020, 253, 126628.

(109) Moreno-Castilla, C. Adsorption of Organic Molecules from Aqueous Solutions on Carbon Materials. *Carbon* 2004, 42, 83–94.

(110) Li, J.; Liang, N.; Jin, X.; Zhou, D.; Li, H.; Wu, M.; Pan, B. The Role of Ash Content on Bisphenol A Sorption to Biochars Derived from Different Agricultural Wastes. *Chemosphere* 2017, 171, 66–73.

(111) Wang, C.; Chen, W.; Yang, L.; Wei, R.; Ni, J.; Yang, Y. Insights into the Roles of the Morphological Carbon Structure and Ash in the Sorption of Aromatic Compounds to Wood-Derived Biochars. *Sci. Total Environ.* 2019, 693, 133455.

(112) Tan, X.; Zhu, S.; Show, P. L.; Qi, H.; Ho, S.-H. Sorption of Ionized Dyes on High-Salinity Microalgal Residue Derived Biochar: Electron Acceptor-Donor and Metal-Organic Bridging Mechanisms. *J. Hazard. Mater.* 2020, 393, 122435.

(113) Zhao, J.; Zhou, D.; Zhang, J.; Li, F.; Chu, G.; Wu, M.; Pan, B.; Steinberg, C. E. W. The Contrasting Role of Minerals in Biochars in Bisphenol A and Sulfamethoxazole Sorption. *Chemosphere* 2021, 264, 128490.

(114) Mishra, S.; Yadav, S. S.; Rawat, S.; Singh, J.; Koduru, J. R. Corn Husk Derived Magnetized Activated Carbon for the Removal of Phenol and Para-Nitrophenol from Aqueous Solution: Interaction Mechanism, Insights on Adsorbent Characteristics, and Isothermal, Kinetic and Thermodynamic Properties. *J. Environ. Manage.* 2019, 246, 362–373.

(115) Lua, A. C. A Detailed Study of Pyrolysis Conditions on the Production of Steam-Activated Carbon Derived from Oil-Palm Shell and Its Application in Phenol Adsorption. *Biomass Convers. Biorefin.* 2020, 10, 523–533.

(116) Sathy Priya, D.; Sureshkumar, M. V. Synthesis of Borassus Flabellifer Fruit Husk Activated Carbon Filter for Phenol Removal from Wastewater. *Int. J. Environ. Sci. Technol.* 2020, 17, 829–842.

(117) Tao, J.; Huo, P.; Fu, Z.; Zhang, J.; Yang, Z.; Zhang, D. Characterization and Phenol Adsorption Performance of Activated Carbon Prepared from Tea Residue by NaOH Activation. *Environ. Technol.* 2019, 40, 171–181.

(118) Fu, Y.; Shen, Y.; Zhang, Z.; Ge, X.; Chen, M. Activated Biochars Derived from Rice Husk via One- and Two-Step KOH-Catalyzed Pyrolysis for Phenol Adsorption. *Sci. Total Environ.* 2019, 646, 1567–1577.