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Progress in deployment of biomass-based activated carbon in point-of-use filters for removal of emerging contaminants from water: A review

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ABSTRACT

Emerging contaminants (ECs) are increasingly being detected in drinking water, posing numerous public health concerns. Granular activated carbon (GAC) offers good prospects for removing ECs at point-of-use (POU), enabling households to have access to safe water. This paper reviews the current advances in deployment of biomass-derived GACs (B-GACs) for removal of ECs at POU. It highlights key sources, pathways and impacts of ECs on public health. It also reveals preparation routes and performance aspects of B-GACs for POU water purification. Knowledge gaps on the subject matter were identified. The review revealed that shells of nuts, which are typically hard and of high density are the most investigated for valorization into GAC for POU water purification. To encourage wide application of B-GACs for POU water purification, there is need to tailor available soft, low-density biomass wastes and suit them as GAC precursors for removal of ECs at the point of use. Future studies need to focus on tailoring the production of B-GACs for removal of specific ECs from water. Additional future research insights include identification of re-generation options for B-GACs, options for removal of residual nanoparticles from the filtrate, lifecycle analysis and costing of various GAC-based POU water filters.

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1. Introduction

Point-of-use (POU) water filtration systems offer a sustainable solution to households without access to safe water by enabling them to effectively treat water at home (Freitas et al., 2022). The filtration systems are installed directly to any water sources at home and they include; bio-sand filters (Magdalera et al., 2020), ceramic water filters (Yang et al., 2020), pitchers or bottles (Doré et al., 2021), faucet-mounted filters (Mulhern et al., 2021), under-the-sink filters (Wu et al., 2021), membrane filters (Kim et al., 2018), wood filters (Siwila and Brink, 2019), bucket filters (Sobsey, 2002) and granular activated carbon (GAC) filters (Fernandes, 2017), among others. The systems are user-friendly, involve lower upfront costs, are easier to install, require less maintenance and are grid-independent (March et al., 2020; Pooi and Ng, 2018). Moreover, since POU systems treat water closer to the point of consumption, the risk of re-contamination during transport and storage is minimized (Mostafa et al., 2021). POU systems, however, are associated with lower installed capacities, high operational costs and poor performance towards removal of certain pollutants from water. For instance, Siwila & Brink (2019) found wood filters to be ineffective at removal of Pb, Cr and As with efficiencies as low as 79.9%, 21.9%, and 6.9%, respectively. POU filters have also been found to induce significant deterioration on the quality of tap water by functioning as traps and reservoirs for sludge, scale, rust, algae or slime deposits which promote microbial growth in household water distribution systems (Nriagu et al., 2018). Owing to these and other performance challenges, there is a need to develop robust POU water filter systems which are tailored to a wide range of source water quality. One way to achieve this is to deploy GAC in POU water filter systems, suiting them for removal of emerging contaminants (ECs) which otherwise escape removal during the treatment of water and wastewater (Chaffin et al., 2019; Iwar et al., 2022).

ECs are substances not commonly monitored in the environment, but have the potential to enter the environment and cause adverse ecological and/or human health effects even

at trace concentrations (Geissen et al., 2015; Teodosiu et al., 2018). In water, concentrations of ECs range from a few nanograms per litre (ngL^{-1}) to several micrograms per litre (μgL^{-1}) (Galindo-miranda et al., 2019). ECs may include pharmaceuticals and personal care products (PPCPs), endocrine disrupting chemicals (EDCs), natural organic matter (NOM), disinfection by-products (DBPs), polycyclic aromatic hydrocarbons (PAHs), pesticides, detergents, illicit drugs, flame-retardants, plasticizers, phthalates, microplastics, nanoparticles, gasoline additives, perfluoroalkylated substances, trace metals and viruses (Fawell and Ong, 2012). Essentially, removal of ECs from water by GAC occurs through; i) adsorption onto the GAC's external surface (Jusoh et al., 2007), ii) physical filtration of the particulate pollutant (Menya et al., 2018a), and iii) through biodegradation of the biodegradable component upon formation of an active biofilm on the GAC filter bed. However, unlike ECs such as caffeine, acetaminophen and ibuprofen, the removal of metals and other ECs like tetracycline and iopamidol is only possible via adsorption onto the GAC's external surface as they are non-biodegradable, and can therefore not be removed via biodegradation (Bilal et al., 2019; Joseph et al., 2019). The extent of pollutant removal depends on the adsorption capacity of the GAC, source water quality, competing species and operating conditions such as flow rate, filter bed depth, contact time, among others (Knezev, 2015). Compared to other POU filters such as membrane filters, GAC filters are more effective at removal of ECs from water (Snyder et al., 2007). This superior performance by GAC is due to its highly developed internal surface area, porosity, and consequently a large adsorption capacity (Menya et al., 2018a). Moreover, GAC is simple in design and easy to operate. In spite of the promising prospects for deployment of GAC in POU filter systems, commercial GACs are still expensive alongside being commonly derived from non-renewable materials such as coal, coke, peat, and petroleum-based residues (Gu et al., 2018; Vilén et al., 2022), which pollute the environment. Commercial GACs are also associated with high production and regeneration costs which limit their potential deployment in POU water filter systems. These numerous concerns have paved the way for wide research efforts towards valorization of the abundantly available biomass wastes into GAC for water and wastewater treatment (Jjagwe et al., 2021; Menya et al., 2018b, 2022).

Theoretically, any low-cost material with a high carbon content and low inorganics can be employed as a raw material for the production of low cost activated carbon (Ioannidou and Zabaniotou, 2007). Agricultural waste biomass such as rice straw (Wu et al., 2020), oil palm shell (Shimabuku et al., 2017), corn stover (Gale et al., 2021), banana peels (Shukla et al., 2020), sugarcane bagasse (Jiang et al., 2019), rice husks (Menya et al., 2022), coconut shells (Maneewong et al., 2022), araç á fruit husks (Lazarotto et al., 2022), eucalyptus tree bark (Yusuff et al., 2022), Nanche stone (Sanchez-Silva et al., 2022), Jacaranda seeds, guava seeds, grape pomace (Valdés-Rodríguez et al., 2022) and canola straw (Benis et al., 2022) are therefore potential raw materials for the production of GAC. The global potential for agricultural waste biomass increased from ≈ 3300 million tonnes in 2010 (Born et al., 2014) to ≈ 4000–12000 million tonnes in 2017 (Bindar et al., 2023). However, despite the increasing trend of waste generation, waste biomass still has little or no economic value. For instance, only 10% of waste biomass is utilized as feed, 6% as bedding material (Born et al., 2014), while ≈ 50% is left in the field for improving soil quality (World Bioenergy Association, 2019). The huge surplus of waste biomass without useful application, therefore, presents huge potential for converting the biomass into GAC, while at the same time offering a sustainable option for disposing of the waste biomass. Moreover, with careful selection of the waste biomass, coupled with pretreatment and activation under carefully controlled conditions, GACs with typically high adsorption capacity, considerable mechanical strength and low ash content can be obtained (Ioannidou and Zabaniotou, 2007; Zubrik et al., 2016).

Waste biomass derived-GACs (B-GACs) have already been deployed in POU filters either solely or in combination with other filter media materials. For instance, in a study by Ahmedna et al. (2004), GACs from shells of various nuts were filled in cartridges and deployed as POU water filters for removal of toxic metal ions from drinking water. In a study by Iwar et al. (2022), GACs from raffia palm shells alone were employed as adsorption filters for removal of fluoride from water. In a study by Rivera-Sánchez et al. (2020), GAC was employed as a post-filter for removal of waterborne bacteria which would have otherwise escaped removal by the ceramic-based POU water filter. Despite these and many other experimental works done concerning the preparation and application of biomass-based activated carbon in water and wastewater treatment, reviews on the deployment of B-GACs in POU filter systems for removal of ECs from water are yet to be done. This review, therefore, sheds light on key sources, pathways and impacts of ECs on human health, discusses the current advances in the preparation and performance of various biomass-based GACs for POU water purification, addresses the factors influencing performance of POU water filters and discusses performance challenges of POU water filters and how these can be averted through modification of GAC. The review also identifies knowledge gaps on the subject matter, informing the basis for future research. Overall, with this review, researchers will gain a broad picture of the state of research on deployment of biomass-based GAC for POU water purification, stimulating research interest to bridge the identified knowledge gaps.

2. Methodology

A search for scientific literature was done to find studies where biomass waste-derived GAC was employed in POU filters, either solely or in combination with other filter media materials for removal of ECs from water. Where possible, a comparison was made with POU filters that employed commercial GAC. Published articles relevant to the review were obtained by searching databases such as the Web of Science, PubMed, Scopus, and Google Scholar. A search was also made via publisher-specific search engines such as ScienceDirect, American Chemical Society, Taylor and Francis, Springer, Royal Society of Chemistry, and Wiley & Sons. Search terms included but not limited to point-of-use filter, household water filter, GAC, biomass-based GAC, water treatment, ECs, sources of ECs, DBPs, NOM, pathways and impacts of ECs. To include several of these keywords in a single search, the Boolean operator "AND" was employed. With the exception of research areas that had few publications, the search for published papers was limited to within the last ten years. A total of 212 references were selected, majority of which are journal articles within the fields of materials science, chemistry and engineering.

3. Number of published articles

The number of original research articles where B-GACs were deployed in POU filters to remove ECs from water was obtained through a thorough literature search. It was found that only a

Table 1 – Some of the potential ECs with their corresponding guideline values (WHO, 202 2).

Substance	Guideline value (μgL^{-1})	Brief description
Alachlor	20	Is a pre-emergence and post-emergence herbicide used to control annual grasses and many broad-leaved weeds in maize and other crops.
Aldicarb	10	A systemic pesticide used to control nematodes in soil and insects and mites on a variety of crops.
Aldrin and dieldrin (combined)	0.03	Are chlorinated pesticides used against soil-dwelling pests, for wood protection and, in the case of dieldrin, against insects of public health importance.
Chlorophenols, particularly 2,4,6-Trichlorophenol	200	Present in drinking-water as a result of the chlorination of phenols, as by-products of the reaction of hypochlorite with phenolic acids, as biocides or as degradation products of phenoxy herbicides.
Dichlorodiphenyltrichlorethane (DDT) and metabolites	1	Its use is restricted or banned in several countries. It is, however, still used in some countries for control of vectors that transmit yellow fever, sleeping sickness, typhus, malaria and other insect-transmitted diseases.
Atrazine	100	Is a selective systemic herbicide of the chlorotriazine class, used for the control of annual broadleaf and grassy weeds.
Chloroform	300	Is the most common trihalomethane (THM) and the principal disinfection by-product in chlorinated drinking-water
Bromoform	100	Is a brominated form of THM.

few publications were made on the subject matter in the last ten years. With only seven publications, 2020 had the highest number of publications, followed by 2022 and 2014, 2019 and 2013, 2017 and 2015, and 2016 and 2018. These findings reveal that less research attention has been directed to the deployment of B-GACs in POU filter systems. Instead, more attention has been directed to the deployment of commercial GACs in POU filters (Ahmedna et al., 2004; Bauer et al., 1996; Iwar et al., 2022; Mulhern et al., 2021) and waste biomass-derived ACs in batch adsorption studies (Lazarotto et al., 2022; Menya et al., 2022). Thus, to keep pace with the increasing public health concerns arising from detecting ECs in drinking water, more research attention needs to be directed towards finding low-cost, effective treatment methods, such as the deployment of B-GACs in POU filters.

4. Potential sources, pathways, permissible limits and impacts of ECs

ECs in drinking water originate from contamination of source waters by point and/or non-point sources where the former include wastewater treatment plants, households, landfill sites, fish farms, power stations and oil spillages, and others (Fawell and Ong, 2012; Mandaric et al., 2015). On the other hand, non-point sources include atmospheric deposition, sewer leakage, combined sewer overflow, terrestrial run-offs from roofs, pavements, roads and agricultural land (Farre et al., 2008; Tran et al., 2019). Out of the point and non-point sources, wastewater treatment plants are the major sources of ECs and are associated with low removal efficiencies (Reichert et al., 2019). Thus, ECs end up in source waters such as streams, rivers, lakes, reservoirs and eventually in drinking water supplies. Moreover, in some developing countries, sewage is still discharged into the environment without any form of treatment (Machado et al., 2016), increasing the risk of source water contamination by ECs. In Brazil, for instance, the concentration of ECs in drinking water was found to be higher than the median values compiled for drinking water samples around the world (Sodré et al., 2010). In different cities of China, the concentration of microplastics in tap water was found to be 440 particles per liter, with particles of size < 50 mm being the most predominant (Tong et al., 2020). Despite these, and other widespread public health concerns, there are still no enforceable drinking water standards for ECs. Instead, health-based guideline values have been derived to guide when there is reason for local concern. For instance, in the United States of America, multiple states have developed guideline values for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) to protect the public from drinking contaminated water (Cordner et al., 2019). However, for ECs such as carbaryl, bentazone and di(2-ethylhexyl)adipate (DEHA) whose concentrations are typically below those that would pose health concerns, no formal guideline values have been proposed (WHO, 2022). Guideline values have also not been derived for ECs such as cyanobacterial toxins, 2-chlorophenol and 2,4-dichlorophenol due to inadequate data available to permit derivation of the health-based guideline values (WHO, 2022). Table 1 highlights some of the potential ECs in drinking water with their corresponding guideline values.

Although ECs in drinking water are normally at trace concentrations, their long-term exposure to humans poses significant threat to public health (Galindo-miranda et al., 2019; Houtman, 2010). For instance, ECs can lead to infertility, endometriosis, reduction in sperm count, as well as increased incidences of breast, testicular and prostate cancer (Machado et al., 2016). ECs can also lead to bioaccumulation, biomagnification, toxicity, endocrine disruption, hormone disruption, DNA damage, foetal malfunction, as well as carcinogenic, mutagenic and teratogenic effects on humans and other organisms (Teodosiu et al., 2018). To avoid such scenarios, ECs in source waters need to be removed at drinking water treatment plants prior to drinking water supply (Borrull et al., 2020). Unfortunately, the conventional water treatment processes commonly employed at drinking water treatment plants can only partially remove ECs from water. For instance, coagulation by alum or ferric sulphate can remove < 20% of the EDCs present in water (Bolong et al., 2009). In Brazil, out of the eleven target ECs, six including stigmasterol, cholesterol, bisphenol A,

caffeine, estrone, and 17β-estradiol were found in tap water (Sodré et al., 2010). In Brazil and Chile, high levels of caffeine, nonylphenol and 17α-ethinylestradiol were detected in drinking water (Peña-guzmán et al., 2019). In Romania, pharmaceutical chemicals were detected in drinking water (Ilie et al., 2017). In Canada, acetaminophen, salicylic acid, caffeine, carbamazepine, ibuprofen, sulfamethoxazole and drospirenone were detected in drinking water (Pulicharla et al., 2021). Moreover, the presence of ECs in drinking water can also be attributed to the migration of ECs from pipes into water distribution systems and eventually into tap water. Such contaminants have included microplastics, bisphenol A, phthalates, perfluoroalkyl and polyfluoroalkyl, among others (Mohammadi et al., 2022). According to Wu et al. (2021), POU water purification is a suitable way for removal of ECs from tap water. This thus paves the way for deployment of biomass waste-derived activated carbons (ACs) in POU systems for removal of ECs from water.

5. Preparation and properties of GAC from biomass

GAC is a form of activated carbon with particle sizes typically ranging from 0.4 to 2.5 mm (Newcombe, 2006). It is regarded as one of the best available technologies for removal of ECs from water (Teodosiu et al., 2018). GAC is commonly produced via the physical and chemical activation methods as described under Sections 5.1 and 5.3, respectively.

5.1. *Physical activation*

In the physical activation method, biomass is first carbonized in an inert environment at temperatures ranging from 400 to 700 °C, producing biochar which is subsequently activated to produce activated carbon with a more well-developed pore structure than biochar (Menya et al., 2018a). The thermochemical processes for converting biomass into biochar and its activation into activated carbon are discussed below.

5.1.1. *Production of biochar*

The production of biochar is achieved through thermochemical conversion of biomass via processes such as pyrolysis, torrefaction, hydrothermal carbonization (HTC), flash carbonization and gasification. These production processes are described below.

Conventional pyrolysis. Biomass pyrolysis is a thermochemical conversion process in which biomass is degraded in an inert environment under temperatures of ˂700 °C, producing gases, tar and biochar (Menya et al., 2019). This process is the most commonly employed for obtaining biochar from biomass (Chatterjee et al., 2020). During pyrolysis, the lignocellulosic components undergo reaction processes like fragmentation, depolymerization and cross-linking, resulting into biochar (Yaashikaa et al., 2020). Temperature is the key parameter during pyrolysis. At high temperatures, the carbon content, surface area and pore volume of biochar are increased while the yield and oxygen content of biochar are reduced (Shen et al., 2017; Colantoni et al., 2016; Ahmad et al., 2013). At severe temperatures, however, the porous structure of biochar is destroyed resulting into cracking and/or blocking of micropores which inhibits porosity development (Xu et al., 2015). Pyrolysis temperature also affects the elemental composition and surface chemistry of biochar (Colantoni et al., 2016). For instance, at low pyrolysis temperature, the resultant biochar is rich in acidic surface groups while at high temperature, the biochar predominantly exhibits basic surface groups (Xu et al., 2015).

Based on operating conditions, pyrolysis can be classified as; slow, intermediate, fast and flash. These classifications are based on heating rate, pyrolysis temperature, residence time and/or particle diameter of biomass. For instance, based on heating rate, slow pyrolysis, intermediate, fast and flash pyrolysis involve heating biomass at a low heating rate ˂ 60 °Cmin^{-1}, moderate heating rate 60–600 °Cmin^{-1}, high heating rate 600–12,000 °Cmin^{-1} and a rapid heating rate ˃ 60,000 °Cmin^{-1}, respectively (Bindar et al., 2023). Due to differences in pyrolysis conditions, product yields vary with pyrolysis type. For instance, the yield of biochar resulting from slow or intermediate pyrolysis is usually ˃20%, while that from fast or flash pyrolysis is usually ˂15% only (Bindar et al., 2023). Out of the pyrolysis types, slow pyrolysis is the best option for obtaining high quantities of biochar (Jawale and Rupnar, 2022). However, conventional slow pyrolysis is limited in a way that electricity is mainly used to heat the walls of the pyrolysis reactor, which then transfers the heat to the material surface. This requires prolonged heating to reach the target temperature and can result into low heat transfer efficiency, high heat loss and long reaction time thus increasing the cost of biochar production (Quan et al., 2023). Such shortcomings can be reduced using microwave-assisted pyrolysis.

Microwave-assisted pyrolysis. Microwave-assisted pyrolysis is a promising alternative to conventional pyrolysis due to its fast heating rate, selective heating, volumetric and uniform heating, which accelerates reaction rates and increase energy efficiency (Li et al., 2016). During microwave-assisted pyrolysis, heat is generated by dipole rotation and ionic conduction which greatly increases the particle temperature that in turn uniformly distributes throughout the sample (Hesas et al., 2013). By operating at a lower temperature and with a more homogeneous and selective heating, microwave-assisted pyrolysis results in biochar of a superior quality as compared to conventional pyrolysis. Thus, biochar obtained from microwave-assisted pyrolysis exhibits higher specific surface area and pore volume than biochar obtained from conventional pyrolysis (Li et al., 2016).

Torrefaction. Torrefaction involves thermal treatment of biomass at temperatures ranging from 200 to 300 °C at atmospheric pressure in an inert environment (Niu et al., 2019). It is also referred to as low temperature pyrolysis, mild pyrolysis or thermal rectification (Acharya et al., 2015). Biomass is torrefied under dry or wet conditions. In dry torrefaction, biomass is heated in dry and non-oxidative (inert) or oxidative atmospheres. For non-oxidative torrefaction, biomass is heated at a rate of 50 °Cmin^{-1} for about 30–60 min (Acharya et al., 2015). In wet torrefaction, biomass is upgraded by water and dilute acid solutions at temperatures of 180–260 °C (Chen et al., 2021). Torrefaction can also be under steam where temperatures are kept below 260 °C with residence time of ≈ 10 min (Yaashikaa et al., 2020). Biochar produced during torrefaction has low adsorption capacity due to the partial oxidation of feedstock during the process (Wang and Wang, 2019).

Conventional hydrothermal carbonization. Hydrothermal carbonization is a wet process performed in a pressure

vessel at mild temperatures ranging from 180° to 220°C, and at pressures ranging from 10 to 15 bar (Yusuf et al., 2020). Hydrochar is the product derived from hydrothermal carbonization of biomass. It is produced through a series of mechanisms including hydrolysis, dehydration, decarboxylation, decarbonylation, condensation polymerization and aromatization (Xiao et al., 2022; Yaashikaa et al., 2020; Chen et al., 2020). HTC has numerous advantages over pyrolysis. For instance, the hydrochar obtained from HTC exhibits a higher fixed carbon content than the biochar obtained from pyrolysis (Shaheen et al., 2019). This is because lignin which contributes most to the fixed carbon content is mildly degraded within the low temperature range of HTC (Kabakcı and Baran, 2019). By operating at lower carbonization temperatures, coupled with the elimination of the need for drying of feedstock, HTC is associated with lower energy requirements than pyrolysis. Due to demineralization, HTC of biomass results in hydrochars with lower ash content than biochar from pyrolysis (Missaoui et al., 2017). Additionally, hydrochar is prevalent in oxygen functional groups, giving it a high cationic exchange capacity required of a good adsorbent (Tan et al., 2015). However, hydrochar exhibits lower carbon stability, fewer pores and smaller specific surface area than biochar (Amalina et al., 2022). These properties can be significantly enhanced upon activation of the hydrochar. For instance, when the corn stover-derived hydrochar was activated at 240 °C for 2 h, the resultant activated carbon exhibited a surface area of 1167 m^2g^{-1}, which was higher than 956 m^2g^{-1} for activated carbon obtained from the activation of biochar at 300 °C for 1 h (Gale et al., 2021). Moreover, GACs can be obtained from hydrochars without the need for binders, lowering the cost of production, avoiding pore blockage and environmental concerns that arise due to volatile organic compounds present in binders (Yu et al., 2020). For instance, hydrochars obtained from HTC of sawdust at 230 °C for 0.5 h, followed by granulation without a binder and activation at 700 °C for 1.5 h resulted in GAC with specific surface area of 641 m^2g^{-1} (Yu et al., 2020). Such surface area is within the range for many commercially available GACs.

Microwave-assisted HTC. HTC can also be achieved by microwave heating. However, the hydrochar derived from microwave assisted HTC exhibits inferior adsorptive properties as compared to the biochar derived from microwave-assisted pyrolysis. The properties can, however, be enhanced through use of catalysts, activation of the hydrochar and/or through increasing the reaction time and particle size of biomass (Nizamuddin et al., 2018).

Flash carbonization. Flash carbonization involves converting biomass feedstock into solid products at the pressure of up to 3 MPa within a temperature range of 300–650 °C and residence time ≈ 30 min (Wang and Wang, 2019). In the operation, the feedstock is initially packed into a bed reactor, followed by pressurizing the vessel to 1–3 bar using air, while heating its bottom using a flame (Amalina et al., 2022). The generated fire travels in an upward direction through the carbonization bed against the downward flow of air supplied to the process (Nartey and Zhao, 2014). Compared to hydrochar, the carbon content of the carbonaceous material obtained from flash carbonization is found to be relatively high (Hirst et al., 2018).

Gasification. Gasification is a thermochemical conversion process that converts carbonaceous materials like biomass into syngas, alongside by-products like tar and biochar (Benedetti et al., 2017). The process occurs at high temperatures ranging from 500° to 1400°C in the presence of a gasifying medium such as air, nitrogen, carbon dioxide, oxygen, steam or a combination of these (Wang and Wang, 2019). Out of these media, oxygen is not desired due to the exothermic reaction of carbon-oxygen, which is difficult to control (Benedetti et al., 2017). While the ultimate goal in the gasification process is to obtain high syngas yields, the generated biochar can be activated to produce activated carbon, increasing the overall efficiency of the gasification plant (Hernández et al., 2016). Depending on the feedstock type, gasification technology, temperature and gasifying medium, the specific surface area of biochar can range from 6.0 to 1027.0 m^2g^{-1} (Benedetti et al., 2017). In situations where the specific surface area is low, the resultant biochar needs to be activated prior to its deployment in POU water filters.

5.1.2. *Activation of biochar*

Biochar usually exhibits a very small adsorption capacity due to blockage of its pores by tarry products evolved during carbonization. These tarry products are removed by activation of the biochar at high temperatures (600–1000 °C) in the presence of suitable oxidizing gases such as steam, carbon dioxide (CO_2), air or their mixtures (Ioannidou and Zabaniotou, 2007). The diameter of pores in biochar is also enlarged during the activation process, alongside creation of new porosity. This provides the resultant activated carbon with a well-developed and readily accessible pore structure (Menya et al., 2018a). However, due to greater thermal degradation of the biochar, carbon is lost during activation, leading to low carbon yields. The Sections below highlight how the different biochar activation methods influence the properties of the resultant GAC.

Carbon dioxide activation. Carbon dioxide is among the most common oxidizing gases employed in the activation process of biochar. It is clean and easy to handle. Moreover, due to its relatively large size, it diffuses slowly into the pores of the biochar, leading to slow reaction rates that make the activation process easy to control. Activation using CO_2 favors the development of new fine pores, thus, compared to other oxidizing gases such as steam, activation using CO_2 results in activated carbon with a more well-developed microporous structure (Alvarez et al., 2015).

Steam activation. Activation using steam is one of the most common methods for activation of biochar. Compared to CO_2, steam is relatively small in size, allowing it to diffuse faster into the pores of the biochar and ultimately leading to fast reaction between the biochar and steam. Steam activation causes widening of existing microporosity quite quickly, leading to activated with a relatively wide pore-size distribution (Hagemann et al., 2018). In spite of these performance differences, the maximum BET surface area of activated carbons derived by steam and CO_2 activation are comparatively similar (Alvarez et al., 2015).

Air activation. Biochar can also be activated using air as the oxidizing agent. However, due to the exothermic nature of the reactions between oxygen and biochar, coupled with difficulty to control the reactions, there is a risk of excessive burning of the biochar, leading to low carbon yield. Moreover, due to local

overheating of the biochar, the resultant activated carbon is usually not uniform (Menya et al., 2018a). Xiao et al. (2020) activated biochar using thermal air oxidation (TAO) at 400–500 °C without a regulated supply of activating gasses. This method is best-suited for small-scale production of GAC, for instance at household level, as it does not require specialized reactors. However, the resultant activated carbon exhibits a low surface area and an under developed pore structure (Xiao et al., 2018). Thus, to obtain activated carbons with a higher yield and superior adsorptive properties, steam or CO_2 are usually preferred as activating agents.

While comprehensive reviews on biochar activation have been given elsewhere (Jjagwe et al., 2021; Menya et al., 2018b), very few studies have directed B-GACs obtained from physical activation for POU water filtration (see Table 2). Moreover, the studies are typically centered on the production of GAC from nutshell-based waste biomass.

Nutshell-based wastes are preferred to other biomass types due to their requisite properties as GAC precursors, for instance, they exhibit high lignin content, low ash content, high density and mechanical strength (Menya et al., 2018b, 2018a). Consequently, the GACs obtained from these biomass wastes exhibit superior adsorptive properties and a high mechanical strength which allows them to withstand abrasion and attrition during backwashing (Jjagwe et al., 2021). However, to ensure a wide application of biomass waste-derived GACs for POU water purification, there is a need to tailor other abundantly available biomass wastes and suit them for production of GACs. Such biomass wastes may include rice husks and waste paper which are soft and of low-density. These wastes can be suited for GAC production through densification using suitable binders to produce hard, higher density granules or pellets prior to carbonization and activation (Pendyal, 1999).

5.2. *Chemical activation*

In the chemical activation method, both carbonization and activation usually occur in a single step process where chemical activating agents such as $ZnCl_2$, H_3PO_4, NaOH and KOH are employed at low temperatures (400–700 °C) to produce activated carbon. A single-step process, however, is only suitable when acidic activating agents are employed. In the case of metal alkaline activating agents such as NaOH and KOH, a two-step process is desired where char is generated in the first step followed by char activation in the second step. This ensures enough activating agent penetrates the resultant char, reacting with the carbon to generate abundant pores. Thus, activated carbons with superior adsorptive properties are obtained when metal alkaline activating agents are employed in a two-stage process than in a single-stage process. Essentially, upon impregnation of the activated carbon precursor, the chemical activating agents dehydrate the precursor, favoring the elimination of hydrogen and oxygen as water instead of carbon oxides (COx) and hydrocarbons. This inhibits the loss of volatile matter and tars, leaving behind more fixed carbon and a higher carbon yield than in physical activation. The adsorptive properties such as BET surface area and pore volume of chemically-derived activated carbon are also higher than those exhibited by the physically-derived activated carbon (Menya et al., 2022). In spite of these attractive attributes, chemical activation poses corrosion challenges, necessitating washing of the activated carbon to remove the residual chemicals. The recovery of residual chemicals, however, is often inefficient, causing these chemicals to leach into treated water. Table 3 summarizes the

Table 2 – Preparation and properties of physically-derived B-GACs from different biomass types for POU water purification.

Biomass type	Pretreatment	Oxidizing gas	Activation conditions	Adsorptive properties	Reference
Pecan nutshells	-	Steam	-	Carbon yield 18%; bulk density 0.4 g mL^{-1}; pH 7.8; ash 10.4%; BET surface area 724 m^2g^{-1}; micropores 71.4% and surface charge 0.23 H^+g^{-1}.	(Ahmedna et al., 2004)
Oil palm shells	-	-	-	BET surface area 497–575 m^2g^{-1}; micropore surface area 374–456 m^2g^{-1}; average pore diameter 1.16–1.19 nm; total pore volume 0.286–0.343 cm^3g^{-1}; micropore volume 0.268–0.319 cm^3g^{-1}	(Shimabuku et al., 2017)
English nutshells	-	Steam	-	Carbon yield 11%; bulk density 0.34 g mL^{-1}; pH 6.7; ash 4.3%; BET surface area 1060 m^2g^{-1}; micropores 81.5% and surface charge 0.25 H^+g^{-1}.	(Ahmedna et al., 2004)

Table 3 – Preparation and properties of chemically-derived B-GACs from different biomass types for POU water purification.

Biomass type	Pretreatment	Activating agent/ method	Activation conditions	Adsorptive properties	Reference
Raffia palm shells	The shells washed with clean water followed by drying in the sun for 7 days. Dried material shredded in a mortar and pestle to particle size 6 mm followed by sieving and drying.	H_3PO_4	Sieved and dried precursor impregnated with H_3PO_4 followed by carbonization to produce AC.	Bulk density 0.45 gcm^{-3}; pH 2.0; pH_{pzc} 2.1; BET surface area 456.10 m^2g^{-1}; total pore volume 0.45 cm^3g^{-1}; micropore volume 0.17 cm^3g^{-1}; average pore diameter 2.13 nm	(Iwar et al., 2022)
Date stones	Clean dry date stones milled to particle size 1.71 mm.	$ZnCl_2$	Impregnation ratio 1:2; carbonization at 700 °C for 3 h. Resultant AC lumps washed with HCl (3 wt%), followed by washing with distilled water until the wash water was free of chloride ions. AC dried overnight at 110 °C.	Bulk density 0.74 gcm^{-3}; total pore volume 0.456 cm^3g^{-1}; micropore volume 0.355 cm^3g^{-1}; bed porosity 0.427; average pore diameter 1.29 nm and BET surface area 951 m^2g^{-1}.	(Alhamed, 2009)
Date stones	Date stones washed with distilled water followed by drying overnight at 105 °C. Dried date stones grinded to obtain granular particles of diameter < 4 mm.	Mixture of $ZnCl_2$ (2 N) and H_3PO_4 (3 N)	Granular particles impregnated with a mixture of the chemical activating agents heated up to 110 °C for 1 h. Impregnated product subjected to heating at 600 C° for 2 h. Resultant AC dried overnight.	BET surface area 197.41 m^2g^{-1}, pH 6.77, uniformity coefficient 2.95, real density 0.4905 cm^3g^{-1}, and apparent density 0.37663 cm^3g^{-1}.	(Yamina et al., 2013)
Coconut shell	Biomass washed with distilled water and later dried at 110 °C for 24 h. Biomass refluxed with a 1 M NaOH solution for 1 h to reduce the ash content	$ZnCl_2$	Impregnation ratio 1:1–2:1; heating rate 10 $°Cmin^{-1}$; activation temp 600 °C; activation time 2 h and acid washing using 3 M HCl followed by oven drying for 24 h	-	(Suif et al., 2020)
Pecan nutshells	-	H_3PO_4	-	Carbon yield 32%; bulk density 0.51 g mL^{-1}; pH 3.2; ash 2.1%; BET surface area 682 m^2g^{-1}; micropores 92.7% and surface charge 2.43 H^+g^{-1}.	(Ahmedna et al., 2004)
Coconut shell	Coconut shell burnt in absence of oxygen to obtain charcoal, followed by grinding to obtain smaller charcoal particles.	$CaCl_2$	Charcoal particles impregnated with $CaCl_2$ for 24 h followed by heating in a kiln for 3 h at temps of 750–800 °C to obtain AC. Resultant AC washed with distilled water followed by oven-drying for 2 h at 105 °C. AC ground and sieved through a 2 mm sieve to obtain desired GAC.	GAC specific gravity 1.16; particle density 1.16 gcm^{-3}; porosity 43.10% and bulk density 0.66 gcm^{-3}	(Yusuf and Murtala, 2020)
Almond nutshells	-	H_3PO_4	-	Carbon yield 24%; bulk density 0.42 g mL^{-1}; pH 3.3; ash 7.9%; BET surface area 708 m^2g^{-1}; micropores 86.6% and surface charge 2.46 H^+g^{-1}.	(Ahmedna et al., 2004)
Coconut shells	Coconut shells scratched off to remove coconut meat, followed by washing with distilled water and drying at 110 °C for 24 h. Resultant coconut shell crushed and sieved to particle size 500–250 μm, followed by refluxing with NaOH (1 M) for 1 h to reduce the ash content in the sample.	Mixture of $ZnCl_2$ (2 N) and H_3PO_4 (3 N)	Impregnation ratio (wt/wt) range 1:1–2:1. Impregnated sample heated under N_2 flow at a heating rate of 10 °C min^{-1} from room temp to 600 °C and holding for 2 h following by cooling to room temp. Resultant AC washed with 3 M HCl followed by drying in oven for 24 h.	-	(Suif et al., 2020)

(continued on next page)

– Table 3 (Continued)

Biomass type	Pretreatment	Activating agent/ method	Activation conditions	Adsorptive properties	Reference
Almond, coconut, peanut and walnut shells	Biomass dried in an oven at 70 °C for > 3 days, followed by grinding and sieving to obtain shell particles between 0.23 and 0.43 mm which were briefly heated in a tube at 100 °C for 1 h prior to carbonizing under N_2 flow (200 $mLmin^{-1}$) heating at a rate of to 10°Cmin^{-1} to final temps (300–800 °C) and holding for 2 h. Furnace then allowed to cool to 25 °C.	Thermal air oxidation (TAO)	0.1 g of the produced charcoal particles placed in a cylindrically shaped aluminum foil tube and subsequently activated in a muffle furnace by TAO at temps of 400 and 500 °C.	BET surface area up to 850 m^2g^{-1}, pore width 0.8–0.9 nm	(Xiao et al., 2020)

Note: i) impregnation ratio is the mass of activating agent to the mass of dried biomass (wt/wt) and ii) *temp* implies temperature (°C)

preparation conditions and properties of chemically-derived B-GACs for POU water purification.

Similar to physically-derived B-GACs, few studies have deployed chemically-derived B-GACs for POU water purification. The studies are also typically centered on the production of GAC from nutshell-based waste biomass and in a few instances from date stones for reasons already cited in Section 5.1.

5.3. *Other properties of B-GACs*

Besides the adsorption properties indicated in Tables 2 and 3, B-GACs are also characterized based on elemental, mechanical and functional properties. These properties are discussed in the sections below.

5.3.1. *Elemental properties*

B-GACs contain elements such as carbon, hydrogen, oxygen, sulphur and nitrogen. These elements are contained in the starting material and remain as a result of imperfect carbonization, or become chemically bonded to the surface during activation (Clements, 2002). Though scarcely reported, the data available points to the fact that the elemental composition of B-GACs varies with biomass type and activation method, among others (see Table 4). Thus, to obtain B-GACs with desired elemental composition which is comparable to that of commercial GACs, biomass needs to be judiciously selected and activated under carefully controlled conditions. The elemental components can influence the adsorption capacity of GACs. For instance, with a high carbon content, GACs are capable of exhibiting a superior adsorption capacity (Chatterjee et al., 2020). Additionally, because ash blocks pores, GACs with a lower ash content are desired due to their superior adsorption capacity as compared to their counterparts with a higher ash content. Based on this, pecan shells-derived GAC with a low ash content of 2.1%db (Ahmedna et al., 2004) may exhibit a potentially higher adsorption capacity than the commercial GAC with ash content of 13.0%db (Jaria et al., 2019). However, it should be noted that there are many other factors that influence the adsorption performance of GAC as indicated in the earlier and subsequent Sections of this paper.

5.3.2. *Mechanical properties*

The mechanical properties of B-GAC include compressive strength, tensile strength, hardness and impact resistance index (Deiana et al., 2004; Philibert et al., 2019; Susilawati et al., 2019; Yu et al., 2020). These properties are critical for the successful deployment of B-GACs in POU water filtration. For instance, for purposes of handling, transportation, air scouring, backwashing and reactivation, the GACs need to have sufficient mechanical strength to withstand abrasion and attrition (Menya et al., 2018a). Otherwise, deployment of B-GAC which is not strong enough would cause operating losses and the formation of fine carbon particles in the adsorber, leading to a high pressure drop (Chemviron Carbon, 2021). The mechanical strength of GAC depends on a number of factors including feedstock type, activation method and conditions. For instance, Philibert et al. (2019) revealed that the mechanical strength of GACs as per their base material increased in the order: coconut > coal > lignite > wood. Despite these insights, there are still very few studies that evaluated the mechanical properties of B-GACs (see Table 5). This could be attributed to the fact that most previous studies are focused on batch adsorption experiments where the mechanical properties of activated carbons are not given attention.

Table 4 – Elemental composition of B-GACs.

Biomass type/GAC type	Activating agent	Ash (%)	Elemental composition (%db)					Reference
			C	H	O	N	S	
Sawdust	$ZnCl_2$ & $FeCl_3$	0.10–0.43	50.08–72.24	5.29–6.54	21.36–42.67	0.29–0.48	-	(Yu et al., 2020)
Pecan shells	H_3PO_4	2.1	-	-	-	-	-	(Ahmedna et al., 2004)
Pecan shells	Steam	10.4	-	-	-	-	-	(Ahmedna et al., 2004)
Almond shells	H_3PO_4	7.9	-	-	-	-	-	(Ahmedna et al., 2004)
English walnut shells	Steam	4.3	-	-	-	-	-	(Ahmedna et al., 2004)
Almond shells	$ZnCl_2$	-	79.2	0.64	-	-	-	(Aygun et al., 2003)
Apricot stones	$ZnCl_2$	-	76.0	0.49	-	-	-	(Aygun et al., 2003)
Halzenut shells	$ZnCl_2$	-	72.0	1.8	-	-	-	(Aygun et al., 2003)
Walnut shells	$ZnCl_2$	-	75.3	0.71	-	-	-	(Aygun et al., 2003)
Paper mill sludge	KOH	9.0	81.2	1.9	13.1	3.0	0.80	(Jaria et al., 2019)
Commercial	-	13.0	92.4	0.75	6.0	0.75	0.05	(Jaria et al., 2019)
Commercial	-	<10	-	-	-	-	-	(Katsigiannis et al., 2015)

5.3.3. *Surface functional groups*

Fourier transform infra-red (FTIR) spectroscopy is normally used to analyze the functional groups present on the surface of GACs. The functional groups are bonded with heteroatoms on carbon surfaces, including oxygen, nitrogen, sulfur, phosphorus and halogens. Thus, based on heteroatoms bonded to the carbon surface, functional groups can be classified as oxygen-containing functional groups, nitrogen-containing functional groups and sulfur-containing functional groups (Yang et al., 2019). Some studies have found the FTIR spectra of B-GACs and that of commercial GACs to be similar (Xiao et al., 2020). The detected functional groups can be both acidic and basic. The acidic functional groups include lactones, phenols, carbonyls and carboxyls, which exhibit a high oxygen content and cation exchange capacity, while the basic functional groups include chromene and pyrone which exhibit a low oxygen content and high anion exchange capacity (Jjagwe et al., 2021). Functional groups, therefore, account for the surface chemistry of GACs, which contributes significantly to their adsorption performance (Xiao et al., 2020). For phenol adsorption, for instance, the presence of basic functional groups on the GAC's surface is desired, and vice versa (Aygun et al., 2003). Despite the critical role surface chemistry plays in the adsorption of ECs, very few studies evaluated the functional groups present on surfaces of B-GACs before and after use in POU water filtration (Xiao et al., 2020). Table 6 shows the functional groups commonly detected in B-GACs.

5.3.4. *Surface morphology*

Surface morphology of B-GACs has been examined using scanning electron microscopy (SEM). The surface morphologies resulting from SEM can then provide information about the pore size, shape and surface topography of GAC. For instance, for bamboo-derived activated carbon, the surface morphologies showed pores in micro meso sizes, which were generally circular and ellipse-shaped (Negara et al., 2019). The GACs derived from macadamia shells together with commercial GAC exhibited pore structures that were randomly distributed (Poinern et al., 2011). Moreover, due to a wide range of biomass types, preparation technologies and conditions, the surface morphologies of B-GACs vary widely. For instance, for walnut shells and jujube seeds-derived GACs, their microstructures varied in wall thickness, shape and pore size (Bae et al., 2014). The pores of activated carbon obtained from activation of bamboo-based char for 100 min were found to be more homogeneous than those obtained from activation for 50 and 150 min (Negara et al., 2019). Despite these insights, very few studies evaluated the surface morphology of B-GACs tailored for POU water filtration. Fig. 1 shows the surface morphologies of GAC derived from macadamia and oil palm shells in comparison with commercial GAC.

6. Application of B-GAC in POU systems

6.1. *Sole application of B-GAC*

GAC can be employed as a sole filter media material in POU water filters for removal of contaminants from water. It is most commonly incorporated in POU filter systems with the help of cartridges (Rozelle, 1987). Several studies have demonstrated successful deployment of B-GACs for POU water purification. In Brazil, for instance, coconut-based GAC was used as a POU filter for atrazine removal from drinking water (Fernandes, 2017). In a study by AWWA (2020), GAC was employed to remove per- and poly-fluoroalkyl substances (PFAS) from water. In the USA, GACs derived from almond, coconut, peanut and walnut shells showed promising prospects as POU filter media materials for removal of selected herbicides and natural estrogen from water (Xiao et al., 2020). The removal of these, and many other ECs by GAC is achieved via adsorption, physical filtration and biodegradation when an active biofilm forms on the GAC particles (Menya et al., 2018a; Xu et al., 2021). However, the removal efficiency by POU filters loaded with GAC alone can be potentially low due to rapid pore blockage by large molecules, leading to reduced sorption capacity of GAC. For instance, using a POU filter loaded with GAC alone, atrazine was removed from water with an efficiency as low as 57.2% (Fernandes, 2017). Due to this limitation, the application of GAC in POU filters is best suited for post-treatment of water when the large molecules have been sufficiently removed from water. For instance, after coagulation, a POU filter loaded with GAC alone can be applied to remove humic substances small enough to reach the adsorption sites of micropores (Menya et al., 2018a). Moreover, though beneficial, deployment of GAC as the sole filter medium in POU filter systems is associated with high capital costs (Crowe et al., 2022). Table 7 highlights some of the studies, where biomass waste-derived GACs alone were employed for POU water purification. Moreover, comparisons with POU filter systems loaded with commercial GAC have also been made.

6.2. *Application of B-GAC in combination with other filter media/types*

GAC can be employed in combination with other filter media materials for POU water purification. Such a combination

Table 5 – Mechanical properties of B-GACs in comparison with commercial GACs.

GAC source/ GAC type	Preparation conditions	Mechanical properties of B-GACs				Reference
		Compressive strength (MPa)	Tensile strength (MPa)	Hardness (%)	Impact resistance index	
Sawdust	HTC of sawdust at 170–290 °C for 0.5 h, followed by activation with $ZnCl_2$ and $FeCl_3$ for 1.5 h.	8.8	-	-	-	(Yu et al., 2020)
Eucalyptus wood	Biochar produced at 500 °C, mixed with a binder, steam activation of the mixture at 850 or 880 °C for 105 and 150 min	2.14–25.25 *	0.38–6.59 *	-	25–350	(Deiana et al., 2004)
Cocoa rind	-	0.002–0.005	-	-	-	(Susilawati et al., 2019)
Commercial	-	14090–22662 * *	-	50.1–99.6	-	(Philibert et al., 2019)
commercial	-	-	-	95.0	-	(Katsigiannis et al., 2015)

Note: i) * implies that the SI unit used is $kgcm^{-2}$ and ii) * * implies that the SI unit used is Nmm^{-1}.

leads to better filter performance than when GAC is employed alone (Xu et al., 2021). Some of the POU filters loaded with GAC and other filter media materials are discussed in the Sections below.

6.2.1. *GAC in combination with sand*

GAC can be used in combination with sand as filter media meterials for POU water purification. This is possible in two ways. One is to build a GAC filter-adsorber after the sand filter, while the other option is to retrofit a sand filter to a GAC-sand dual media filter (Feng et al., 2013). In either ways, due to the combined effect of adsorption and biodegradation, the removal of ECs from water can be enhanced (Li et al., 2018; Xu et al., 2021). Moreover, since slow sand filtration demands high-quality source water, employing GAC-sand dual media in a POU filter widens the range of source water quality treatable with success (Logsdon et al., 2002). The GAC plays an adsorptive function, compensating for the initial low treatment capacity of the sand filter (Zhao et al., 2019). However, at household level, slow sand filtration is less likely to be implementable, since its preferred filter designs and installations are often larger and suited for treating larger volumes of water than needed by individual households (Sobsey, 2002).

Besides slow sand filtration, GAC can also be employed in combination with biosand filter where water is treated by mainly the physical and biological processes (Sizirici et al., 2019). In the physical process, suspended particles are strained out as water flows from the top to the underdrain of the filter. On the other hand, the biological process involves formation of a biological layer (*or schmutzdecke*) at the sand-water interface, containing microbial communities which break down organic particles. Predatory bacteria which form majority of the microbial community in the biofilm prey on any microorganisms present in the water (Olupot et al., 2021; Sizirici et al., 2019). By incorporating the biosand filter with GAC, the efficacy of the resultant POU filter is enhanced since the filtration process benefits from a combination of physical filtration, biodegradation and adsorption (Roegner et al., 2014). Several studies have employed B-GAC in combination with biosand filter. For instance, a biosand filter, particularly P3 biosand bag filter was incorporated with GAC and biochar, where fluoride was removed from water with 100% removal efficiency (Fung et al., 2021). The superior performance of the resultant filter was attributed to the high surface area of GAC (≈739 m^2g^{-1}) and the hydroxyapatite groups in the biochar. In Philippines, a biosand filter was incorporated with coconut shell-based AC for water treatment, removing thermotolerant coliform and color with efficiencies as high as 100 and 92–95%, respectively (Magdalera et al., 2020). In Vietnam, a POU biosand filter with an added layer of mangrove activated charcoal was developed for removal of imidacloprid insecticide from water (Wyckoff, 2013). In Vietnam still, a packed bed of AC from mangrove was deployed as a pre-filter in series with a biosand filter, removing p-nitrophenol from water and achieving a removal efficiency of 96.81% (Safford and Lackey, 2014). In spite of the abundantly available biomass wastes, limited studies have employed B-GACs in combination with either slow sand/or biosand filters for POU water purification (see Table 8).

6.3. *GAC in combination with other filter types*

GAC has been employed in combination with wood filters to enhance the efficacy of the resultant POU water filter. For instance, when a wood filter was employed in combination with

Table 6 – Functional groups available on the surface of biomass-derived GACs (Bindar et al., 2023; Dwivedi et al., 2008; Jaria et al., 2019; Xiao et al., 2020; Zheng et al., 2018).

No.	Wave number (cm^{-1})	Vibration characteristics	Presence indication
1	3800–3600	O-H stretching	Alcohol, phenol
2	3357	O-H stretching	Hydroxy group, phenol, alcohols
3	3040–2880	C-H stretching	Alkene groups
4	2830	C═O	Carbonyl
5	2567–2163	C≡C stretching	Alkyne groups
6	1720–1699	C═C stretching, C═O stretching, and/or C═O stretching	Ketones, carbonyl, carboxylic acid
7	1587–1557	C═O stretching	Carboxylate
8	1560–1530	N-H and C-N bending	Secondary amide
9	1400–1370	O-H bending	Alcohols, phenols, esters, ethers
10	1240–1077	C-O stretching	Phenol
11	1100–1180	C-O stretching	Secondary alcohols
12	872–805	= C-H or = CH_2 bending	
13	747–745	O-H bending	Alcohols, phenols, esters, ethers

GAC, As, Cr and Cd were removed with efficiencies of 65.1%, 62.8% and 73.7% as compared to 6.9%, 21.9% and 0.0%, respectively when the wood filter was employed alone (Siwila and Brink, 2019). This enhanced removal efficiency was attributed to the combined effect of the wood filter and GAC materials, with the former allowing low flow rates and an empty bed contact time (EBCT) ˃20 min sufficient for adequate removal of pollutants by GAC (Siwila and Brink, 2019). By removing pollutants in drinking water that would otherwise not be possible with a wood filter alone, incorporation of GAC in POU filter systems presents good prospects for obtaining drinking water of desired quality, especially for households in resource-constrained settings and in areas where toxic metals are present in water. Besides the wood filter, GACs have also been employed in combination with other filter types including ceramic water filters (Rivera-Sánchez et al., 2020), faucet-mounted POU filters and pitcher filters (Doré et al., 2021). For instance, Rivera-Sánchez et al. (2020) coupled the silver-impregnated GAC with zeolite and a ceramic silver-impregnated pot filter (CSF) for removal of *Escherichia coli* and *Salmonella spp*., attaining a removal efficiency as high as 99%. The silver ions (Ag^{+}) have antimicrobial properties, which enable disinfection of the bacteria from water. Essentially, when water passes through the filter bed, the silver ions are released and come into contact with the bacteria. The bacteria are disinfected when the silver ions interact with the components of the bacterial membrane, damaging the membrane, deoxyribonucleic acid (DNA) and protein (Siritongsuk et al., 2016). The bacteria are also disinfected when silver ions catalyze the generation of destructive reactive oxygen species within the intracellular compartment of bacteria, causing damage to organic compounds within the cell (Akhigbe et al., 2016). Moreover, even without activation, biochar incorporated in ceramic filters was found to enhance the removal of water pollutants from water. For instance, a biochar-clay composite ceramic filter effectively removed dissolved organic carbon (DOC) from water, improving the colour of water (Chaukura et al., 2020). Bone char and AC derived from Cordial millenii (African drum) tree stem were also incorporated in a ceramic filter for removal of trace metals from water (Erhuanga et al., 2014). In spite of the promising prospects for utilizing biomass-derived ACs in combination with other filter types, there are still few studies that have been conducted (see Table 9). Therefore, more studies are needed to provide a basis for making evidence-based decisions on the application of biomass-derived ACs for POU water purification.

Fig. 2 illustrates some of the POU water filtration systems highlighted in Tables 7–9.

7. Mechanism of adsorption by B-GAC

A thorough understanding of the removal efficacy of ECs by B-GAC can be achieved through identifying the underlying adsorption mechanisms. The adsorption behavior of B-GAC depends on a number of factors such as nature of contaminant to be removed and GAC's adsorptive properties such as specific surface area, pore size distribution, surface functional groups and zeta potential. Depending on the surface forces/adsorption energy between the contaminant and B-GAC, the mechanism of adsorption can be physical or chemical in nature (Ouyang et al., 2020). Adsorption energy ˂– 30 $kJmol^{-1}$ indicates physisorption while energy ˃– 50 $kJmol^{-1}$ indicates a chemisorption (Khadhri et al., 2019). Physical adsorption implies that only weak interaction forces such as hydrogen bonding, van der Waals forces, ionic, electrostatic and dipole interactions are involved between the contaminant and GAC (Khadhri et al., 2019; Bojić et al., 2017). Physisorption is a reversible process mainly controlled by the textual properties of carbon materials, including pore structure and surface area (Quan et al., 2023). Chemisorption, on the other hand, is an irreversible exothermic reaction in which GAC's surface functional groups and alkali ions play a key role (Quan et al., 2023; Liu et al., 2021).

The adsorptive process is rarely purely physical or chemical, and in most cases a combination of these two mechanisms is involved (Alharbi et al., 2020; Ighalo et al., 2020). Such combined mechanisms involve; π-π dispersion, repulsive electrostatic interaction, donor-acceptor electrostatic interaction, formation of strong inner-sphere surface complexes, formation of hydrogen bonds with neighboring GAC oxygen group, ion exchange, van der Waal's forces and dipole-dipole forces (Ijagwe et al., 2021). Some of these mechanisms are illustrated in Fig. 3.

Electrostatic interaction involves electric forces of attraction and repulsion between contaminant and GAC (Ouyang et al., 2020). If electrostatic attraction is primarily involved in adsorption, then changes in solution pH significantly affect the adsorption capacity otherwise it ceases to play a key role (Salomón-Negrete et al., 2018; Tran et al., 2017). Hydrogen bonds prevail in presence of hydrogen donor/acceptor chemical groups on both the contaminant and GAC (Ouyang et al., 2020). The contribution of hydrogen bonding in adsorption can be identified from the FTIR spectra of the AC surface before and after adsorption. A decrease in the -OH peak intensity and its shift

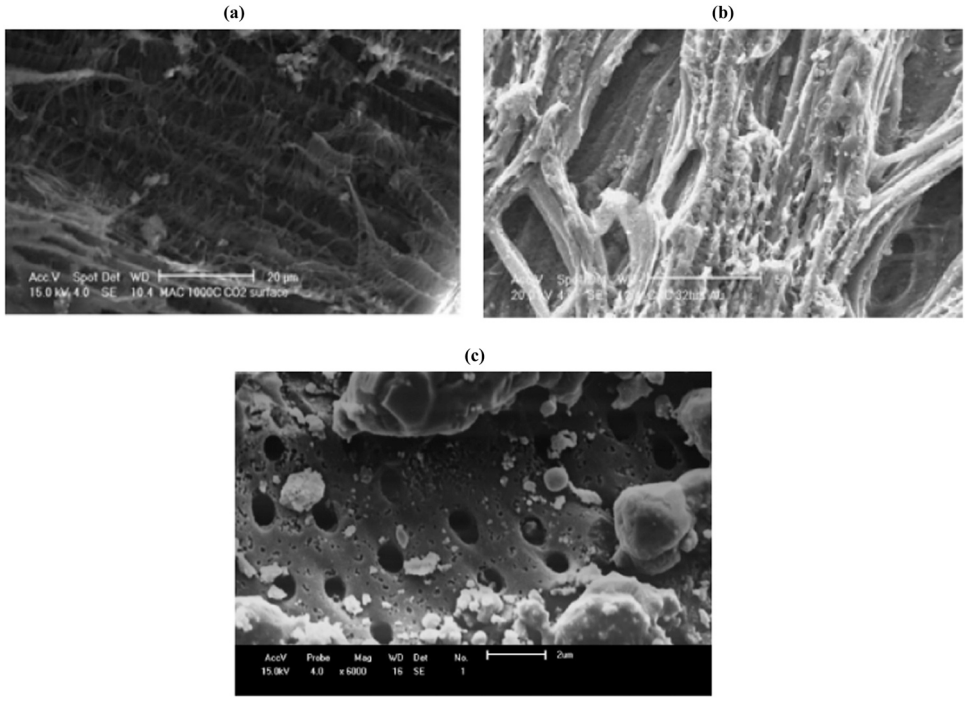

Fig. 1 – Surface morphology of (a) macadamia-based GAC 20 μm scale bar, (b) commercial GAC 20 μm scale bar (Poinern et al., 2011) and (c) oil palm shell-based GAC (Shimabuku et al., 2017).

towards higher wave numbers after adsorption indicates the presence of hydrogen bonding (Tran et al., 2017).

Pore filling occurs when there is an interaction between the contaminants already adsorbed on the surface of GAC and those that are free in the solution (Li et al., 2020). If an increase in surface area of GAC does not guarantee an increased adsorption capacity, then pore filling is likely not contributing to the adsorption mechanism (Sajjadi et al., 2018). A significant decrease in the pore volume and surface area of GAC after adsorption could indicate that pore filling was the dominant adsorption mechanism (Nguyen et al., 2020; Tran et al., 2017). Ion exchange occurs when the adsorbent adsorbs ions from the solution and releases equivalent ions back to the solution in order to maintain the solution electrical neutrality (Ewis et al., 2022). Π- Π interaction is a dispersive interaction through the delocalized π electrons of oxygen-free Lewis basic sites of GAC, and the free electrons of the contaminant in the aromatic rings (Chiang et al., 2020). The contribution of π- π interaction in adsorption is evident when there is a decreased intensity in the aromatic C=C bonds on the surface of GAC, and their increase in wavenumber after adsorption (Tran et al., 2017). Besides the FTIR, other advanced spectroscopy such as the field emission transmission electron microscopy, Raman spectroscopy, scanning electron microscopy, ultraviolet-visible spectroscopy, X-ray diffraction analysis, X-ray photoelectron spectroscopy can also indicate the different adsorption mechanisms as discussed in different studies (Hou et al., 2020; Yang et al., 2020; Benabid et al., 2019).

8. Factors influencing performance of POU filters

8.1. *Source and adsorptive properties of AC*

The performance of a POU filter is influenced by the source and adsorptive properties of AC. Essentially, upon activation of the biomass, the resultant AC exhibits pore sizes on the scale of micropores (< 2.0 nm), mesopores (2–50 nm) and/or macropores (> 50 nm), where the small and large pores are more suited for the adsorption of small and large molecular weight compounds, respectively (Chaffin et al., 2019). Moreover, the predominance of pore sizes varies with biomass type, activation conditions and method of activation (Menya et al., 2018a). For instance, at any burn-off, the micropore and mesopore volumes created in palm-shell-based ACs were found to be higher than those exhibited by coconut shell-based ACs (Daud and Ali, 2004). Elsewhere, ACs derived from coconut shells were found to be predominantly microporous, while those derived from wood were found to be both microporous and mesoporous (Chaffin et al., 2019; Samdin et al., 2015). Based on these reports, therefore, coconut shell-based ACs are suitable for the removal of organic micropollutants with small molecular size such as the fuel additive; methyl tertiary-butyl ether (MTBE) (Samdin et al., 2015). On the other hand, pollutants with large molecular sizes, including NOM and microcystins would be best suited for removal using wood-derived ACs (Chaffin et al., 2019). Besides the size and distribution of pores, the surface area and characteristics also determine the adsorptive properties of ACs. More specifically, surface area determines the amount of internal area available for access by the target contaminant (Menya et al., 2022). ACs with surface area as high as 1060 m^2g^{-1} have been obtained from the activation of biomass wastes for POU water purification (Ahmedna et al., 2004). With such insights from the literature, biomass from which GAC is to be derived should be selected judiciously for effective POU water purification.

8.2. *Competing species*

Pollutants are sometimes hindered from being effectively removed by GACs due to the presence of species that compete with target pollutants for adsorption sites. For instance, in POU filters employing GAC, organics and multiple synthetic organic chemicals (SOCs) are reported to be the competing species (EPA, 2006). For adsorption of microcystins,

Table 7 – Biomass waste-derived GAC employed for POU water purification.

GAC source/POU filter type	Adsorptive properties of GAC/description of POU filter	Adsorption/filtration conditions	Pollutant removed/removal efficiency (%)	Reference
Coconut shell	Iodine number 800 mgg^{-1}, pH 6–8, density 0.4–0.55 gcm^{-3}, moisture content 10 wt%.	Mass 90 g; flowrate 100 Lh^{-1}; temperature 20 °C; pH 8.1; and time 0–100 min	Atrazine removed by 57.2%.	(Fernandes, 2017)
Almond, coconut, peanut, and walnut shells	BET surface area up to 850 m^2g^{-1}, pore width 0.8–0.9 nm	Temp 22 ± 1 °C, pH 7.1–7.4, mixing rate 40 rotations per min for 7 days	Triazine (atrazine and prometon) and estriol	(Xiao et al., 2020)
Raffia palm shells	POU adsorption filters composed of AC-aluminum oxide composite	Column fed by gravity from a reservoir of capacity 60 L; internal column diameter/carbon depth ratio 0.2; volumetric flow rate 6.25 Lh^{-1} (0.11 $Lmin^{-1}$); linear flow rate 0.035 m min^{-1}.	Fluoride removal efficiency 55.88%; adsorption capacity 14.0 mgg^{-1}; treated volume at breakthrough 700 L and breakthrough time 112 h	(Iwar et al., 2022)
Oil palm shells	Filter preconditioned with 3 L of distilled water. Phage particles added in distilled water and homogenized at 100 rpm for 3 min followed by 20 min of mixing at 30 rpm. Solution containing virus pumped at 450 $mLmin^{-1}$ through GAC in the column	Household filter composed of GAC with S_{BET} = 497–575 m^2g^{-1}; $S_{mic}t$ = 374–456 m^2g^{-1}; D_p = 1.16–1.19 nm; V_t = 0.286–0.343 cm^3g^{-1} and V_{mic}= 0.268–0.319 cm^3g^{-1}.	Highest viral reductions > 5 log achieved, but after GAC was modified with a mixture of silver and copper oxide.	(Shimabuku et al., 2017)
Nutshell (almond, English walnut and pecan)	Water filtration cartridges filled with nutshell-based GACs	BET surface area 682–1060 m^2g^{-1}, micropores 71.4–92.7%, surface charge 0.23–2.46 mmol H^+g^{-1}, pH 3.2–7.8, ash content 2.1–10.4%, adsorbent mass in catridge 60–93 g	Removal efficiency Pb^{2+} ≈100%; Cu^{2+} 90–95% and Zn^{2+} 80–90%	(Ahmedna et al., 2004)
Coconut shell	Pitcher-style water purifier with cartridge filled with coconut shell-based AC	Adsorbent mass in cartridge 83.4 g	Microcystins removal efficiency ≤ 50%	(Chaffin et al., 2019)
Dates' stones	AC packed in a bed column suitable for POU water filtration. Column diameter was 1.0 m, column height 10.0 m and AC particle diameter 1.5 mm	Bulk density 0.74 gcm^{-3}, total pore volume 0.456 cm^3g^{-1}, micropore volume 0.355 cm^3g^{-1}, bed porosity 0.427, average diameter 1.29 nm, BET surface area 951 m^2g^{-1}.	Removal of phenol from drinking water	(Anisuzzaman et al., 2014)
Commercial	Water filtration cartridges filled with commercial AC	Adsorbent mass in cartridge 76–100 g	Removal efficiency Pb^{2+} ≈0.0018–0.0022 ppm g^{-1}; Cu^{2+} 0.0327–0.0435 ppm g^{-1} and Zn^{2+} 0.0596–0.0750 ppm g^{-1}	(Ahmedna et al., 2004)
Commercial	GAC-based POU filters	Column fed by gravity from a reservoir of capacity 60 L; internal column diameter/carbon depth ratio 0.2; volumetric flow rate 6.25 Lh^{-1} (0.11 $Lmin^{-1}$); linear flow rate 0.035 m min^{-1}.	Fluoride removal efficiency 54.47%; adsorption capacity 7.8 mgg^{-1}; treated volume at breakthrough 500 L and breakthrough time 80 h.	(Iwar et al., 2022)
-	Effective size 0.7 mm	Filtration rate 0.1–0.5 mh^{-1}, and bed depth 0.3–1.0 m and EBCT 6 min	-	(Bauer et al., 1996)
Under-sink AC block filter	POU composed of an extruded AC block without a prefilter membrane or fabric.	POU filter designed to treat the full flow of cold water at the tap, up to 5.67 $Lmin^{-1}$	Bacterial and viral microbial indicator organisms removed by 84%	(Mulhern et al., 2021)
-	POU GAC fixed bed adsorber with coarse and fine mesh GAC	Pb 150 ppb, pH 6.5; TDS < 100 ppm; hardness 10–30 ppm as $CaCO_3$ and alkalinity 10–30 ppm as $CaCO_3$.	Pb in drinking water reduced to levels < 15 ppb	(Taylor and Kuennen, 1994)
Commercial	POU filter composed of iron-impregnated GAC	Initial arsenic concentration 500 μgL^{-1}; filtration rate 0.23 mh^{-1}; filter bed volume 300 mL and contact time 21 min	High removal efficiencies of As(III) and As(V)	(Petrusevski et al., 2002)

Note: i) *Temp* denotes temperature (°C), ii) *EBCT* denotes empty bed contact time, iii) q_{max} is maximum adsorption capacity, iv) $1/n$ is defined as a measure of adsorption intensity or surface heterogeneity, v) K_F indicates the adsorption capacity of the adsorbent (mg^{1-n} L^n kg) and vi) R^2 is correlation coefficient.

Table 8 – Biomass waste-derived GAC employed in combination with slow sand and biosand filters.

GAC source	Adsorptive properties of GAC	Description of the POU filter with GAC	Removal efficiency (%)		Reference
			Slow sand filter alone	Slow sand filter with GAC	
Date stones	BET surface area 197.41 m^2g^{-1}, pH 6.77, uniformity coefficient 2.95, real density 0.4905 cm^3g^{-1}, and apparent density 0.377 cm^3g^{-1}.	Sand dune employed in combination with AC. Total thickness and height were 60 and 0–24 cm, respectively.	-	[a]COD and BOD_5 reduced by 80–95 and 78–94%, respectively.	(Yamina et al., 2013)
Coconut shell	-	POU filter consisting of gravel, sand and AC of layers	-	BOD_5 reduced by 81.88–94.38%.	(Suif et al., 2020)
-	-	Slow sand filter sandwiched with GAC; sand 10 cm, GAC 10–20 cm, and filtration rate 5–20 cmh^{-1}.	PPCPs removed by 51.9%.	Pharmaceutical and personal care products (PPCPs) removed by 98.2%.	(Li et al., 2018)
Commercial	Effective size 0.72 mm (size range 0.62–1.60 mm) and uniform coefficient 1.68	GAC loaded in the middle of sand bed to obtain a GAC sandwich	≤ 20% removal of antibiotics	> 90% removal of antibiotics	(Xu et al., 2021)
Commercial	-	Sand employed in combination with GAC to form a GAC-sand biofilter	-	49.1–94.4% removal of CEC	(Ma et al., 2018)
Commercial	-	Sand employed in combination with GAC to form a GAC-sand biofilter	-	-	(Paredes et al., 2016)
-	-	GAC bed depth 0.075–0.2 m, filtration rate 0.1–0.5 mh^{-1}, contact time 9–120 min, EBCT > 30 min	-	60% removal of total organic carbon (TOC) from water	(Bauer et al., 1996)
-	-	GAC: bed depth 1000 mm, Effective size 1.01 mm, and uniform coefficient 1.54 Sand: bed depth 400 mm, effective size 0.68 mm, and uniform coefficient 1.61 Gravel: bed depth 300 mm	-	-	(Feng et al., 2013)

Note: i) (a) implies the extent of efficiency enhancement due to deployment of sand dune in combinantion with AC as compared to sand dune alone

Table 9 – GAC employed in combination with other filter types.

Biomass derived/ commercial AC	Adsorptive properties/filtration conditions	Design description of POU filter	Pollutant removal	Country	Reference
Bamboo	-	POU system consisting of bamboo-based AC, membrane microfilter, and a UV-disinfection unit.	67–100% removal of TSS, 64–100% removal of turbidity, pH 6.87–7.74	Malaysia	(Kuok et al., 2022)
Coconut shell	GAC specific gravity 1.16; particle density 1.16 gcm^{-3}; porosity 43.10%; bulk density 0.66 gcm^{-3}	Ceramic filter cup in combination with GAC and a magnetic treatment unit	Removal of total coliform count and E. Coli bacteria from the water by 94.12% and 100%, respectively	Nigeria	(Yusuf and Murtala, 2020)
-	Impact strength of filter 46.74 kJm^{-2}, water absorption 18.6%, porosity 51% and flowrate 51 $mLmin^{-1}$.	A biochar-clay composite ceramic filter fabricated from clay and sawdust.	DOC effectively removed from water	Zimbabwe	(Chaukura et al., 2020)
Cordial millenii (African drum) tree stem	-	Ceramic filter incorporated with bone char and AC derived from Cordial millenii (African drum) tree stem.	Chemical contaminants effectively removed from water including fluoride, lead, and sulphate ions.	Nigeria	(Erhuanga et al., 2014)
-	-	Ceramic-silver impregnated pot filter with a post-filter composed of GAC and zeolite impregnated with colloidal silver	Escherichia coli and Salmonella spp. removed by 99%	Colombia	(Rivera-Sánchez et al., 2020)
-	-	POU water system consisting of a 5 µm sediment filter, carbon pre-filter, cellulose-acetate RO membrane, 1- µm sediment filter, a carbon post-filter and mineral filter	29.41–82.35% removal of total organic carbon	Iran	(Rezaeinia et al., 2018)
-	-	POU filter system equipped with two GAC cartridges and a boiler in sequence	Haloacetic acids removed with efficiency >60%, while haloacetaldehydes completely removed	China	(Wang et al., 2019)
Commercial	-	A five-stage water purification system comprising of a polypropylene cotton filter, a GAC filter, a compressed AC filter, ultra-filtration cartridge, and a post GAC filter	Antibiotics and antibiotic resistance genes removed with efficiencies of 91–92% and 0.46–2.43 log, respectively.	China	(Gu et al., 2021)
Commercial	pH 6.5–8.5	Faucet-mounted POU filters loaded with carbon blocks and natural zeolite.	99–99.9% removal of Pb	-	(Doré et al., 2021)
Commercial	pH 6.5–8.5	Pitcher filter loaded with ion exchange resin mixed with AC and fibrous matrix to hold material	98.5–99.3% removal of Pb	-	(Doré et al., 2021)
Commercial	pH 6.5–8.5	Pitcher filter loaded with AC, fibrous matrix to hold material, and an absorbent in filter media	99.5–99.6% removal of Pb	-	(Doré et al., 2021)
Commercial	-	POU filter system composed of a UV-LED/chlorine advanced oxidation process followed by GAC adsorption.	Bisphenol A, diclofenac and caffeine effectively removed from drinking water	Hong Kong	(Yin and Shang, 2020)

Note: (-) implies that the specific information was not provided

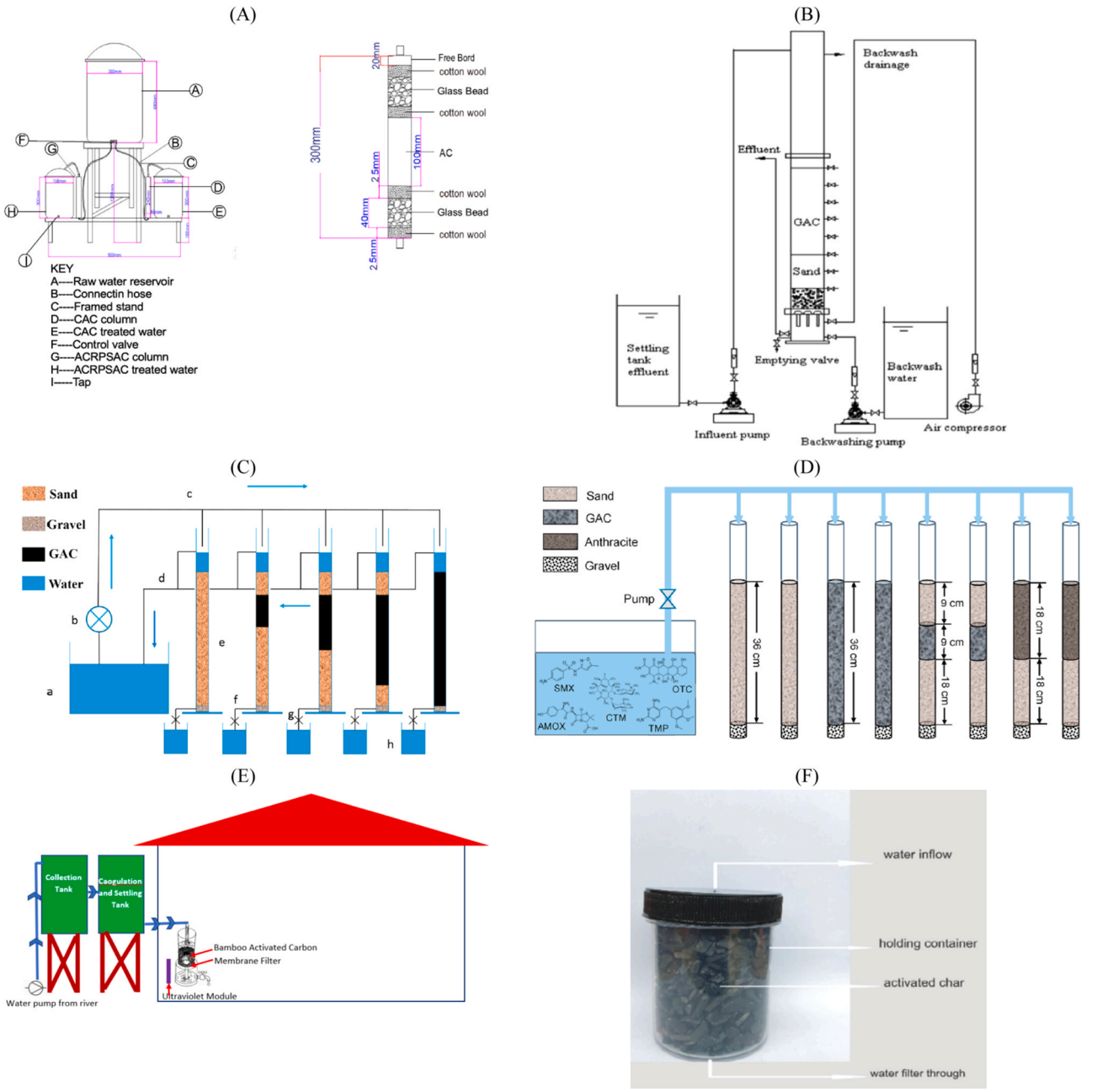

Fig. 2 – (A) Schematic of the POU adsorption system and filter details (Iwar et al., 2022); (B) schematic of the pilot-scale GAC–sand dual media filter (Feng et al., 2013); (C) schematic representation of the GAC sandwich SSF experiment. (a) influent tank; (b) peristaltic pump; (c) influent pipe; (d) overflow pipe; (e) filter; (f) effluent pipe; (g) effluent valve; (h) effluent bottle (Li et al., 2018); (D) schematic of biofilter composition. SMX, sulfamethoxazole; TMP: trimethoprim; AMOX: amoxicillin; OTC: oxytetracycline; CTM: clarithromycin (Xu et al., 2021); (E) schematic diagram of the envisioned treatment system for a household utilising river and rainwater sources (Kuok et al., 2022) and (F) GAC in the cartridge (Yusuf and Murtala, 2020).

residual chlorine is the competing species as it reacts with the active sites, leading to decreased GAC adsorption of microcystins (Roegner et al., 2014). For adsorption of per-and polyfluoroalkyl substances (PFAS), DOC is the competing species, hindering removal of PFAS from water (AWWA, 2020). For atrazine removal from water, chlorine and its by-products, as well as NOM, salts and metallic elements are the competing species (Fernandes, 2017). For NOM removal from water, the competing species are organic micropollutants (OMPs) which are ≈3–6 orders of magnitude smaller than NOM concentration (Menya et al., 2018a). Conversely, during OMPs removal from water, the competing species is NOM (Tröger et al., 2020). Aschermann et al. (2018) revealed that DOM affects desorption of OMPs in two ways; on the one hand, DOM blocks AC pores, resulting in decreased desorption. On the other hand, DOM displaces OMPs, resulting in increased desorption. Their studies further revealed that the effect of DOM on desorption of OMPs is also dependent on the pore structure of the AC, where the micro- and mesoporous AC exhibit lower extents of desorption. This tendency has to do with the blockage of AC pores by DOM, making it difficult for the previously adsorbed OMPs to leave the pores again. For the macroporous AC, pore blockage effects by DOM are minimal, since OMPs not only desorb again, but also get further displaced by adsorbing DOM responsible for enhanced desorption (Aschermann et al., 2018). Thus, when deploying activated carbon in POU filter systems, it is

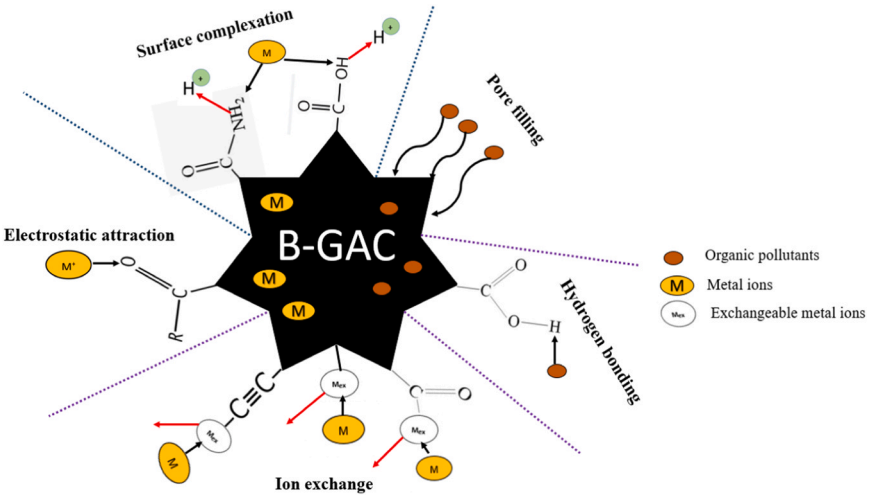

Fig. 3 – Adsorption mechanisms of B-GAC towards different metallic and/or organic pollutants.

important that the competing species are given due attention with necessary measures taken to remove them from water.

8.3. Filter design

The design of a GAC filter is commonly based on parameters such as empty bed contact time (EBCT), hydraulic loading rate (HLR) and bed depth. These parameters subsequently influence the performance of a given GAC filter. For instance, increasing EBCT increased haloacetic acids (HAA) removal in GAC columns, and vice versa (Wu and Xie, 2005). However, an optimum EBCT exists above which no improvement in removal efficiency is realized. DOM removal, for instance, increased up to an EBCT of ≈ 30 min, above which EBCT had little additional impact (Moona et al., 2021). For HLR, the higher the rate, the less time available for the filter to remove contaminants, and vice versa. A high HLR, however, ensures that solids are transported deeper into the filter bed, making use of the entire filter depth for solid storage and ultimately distributing headloss to all media layers (Ncube et al., 2016). Despite these insights, studies about the effect of HLR on ECs removal by POU GAC filters are still limited.

For bed depth, the deeper the filter, the higher the solids retention and removal efficiency of organic contaminants from water (Ncube et al., 2018). For instance, at increased bed depth, the removal of iron from water by GAC was enhanced (Rasul et al., 2015). Increase in bed depth is also associated with increased bed adsorption capacity due to the increased specific surface of the adsorbent which supplies more binding sites (Taty-costodes et al., 2005). In spite of these benefits, increasing the filter depth leads to increased head loss. Moreover, the benefit of increasing filter depth diminishes as the filter deepens, since due to stratification during backwashing, the media size increases with depth such that the bigger media grains exhibit less contact surface area than their smaller counterparts in the filter top (Ncube et al., 2018).

8.4. Filtration rate

Filtration rate is an important factor that influences the effectiveness of a POU filter. It is a key design consideration for filtration processes. Filtration rates typical of GAC filters could not be found in the literature, but for other filter types, particularly the slow sand filter, the filtration rates range from 0.1 to 0.3 mh^{-1} (Kubare and Haarhoff, 2010). Essentially, at slow filtration rates, POU water filters have been reported to be more effective at removal of organic contaminants from water than when operated at high filtration rates. For instance, pitcher-style purifiers with slow filtration rates were found to be more effective at removal of microcystins from water than their counterparts operated at higher filtration rates (Chaffin et al., 2019). A POU water filter consisting of gravel, sand and coconut shell-based AC operated at low filtration rates was also found to be more effective at removal of organic contaminants from water than when operated at high flow rates (Suif et al., 2020). However, compared to contact time, the filtration rate exhibits less influence on the performance of a POU water filter. For instance, during arsenic removal from water, contact time rather than filtration rate was the critical parameter for performance of the family filter (Petrusevski et al., 2001).

8.5. Contact time

Contact time is dependent on filter media depth and filtration rate, thus the optimization of either/or both of these parameters optimizes the contact time needed for optimal removal of contaminants from water (Siwila and Brink, 2020). The influence of contact time on the performance of a POU water filter has been reported in several studies. For instance, in a study by Rasul et al. (2015), longer contact times resulted in a higher removal efficiency of iron than at shorter contact times. A study by Chaffin et al. (2019) also revealed that longer contact times are necessary for effective removal of microcystins than at shorter contact times.

8.6. Particle size of filter media materials

The particle size of the filter media materials influences the performance of a POU water purification system. More specifically, the smaller the grain size of the filter media materials, the more the surface area available for contact as compared to the materials with larger grain size (Ncube et al., 2018). Moreover, media materials that are too coarse are associated with high initial filtration rates, leading to filtrate with a high turbidity (Kubare and Haarhoff, 2010). With these reports, organic contaminants including ECs can be effectively removed from water when POU water filters are loaded with fine grained filter media materials.

8.7. pH, zero point of charge and surface charge

The pH, point of zero charge (pH_{zpc}) and surface charge of carbon influence its adsorption performance when employed for POU water filtration. At lower pH, most organics are less soluble, and therefore readily adsorbed due to electrostatic attraction between the GAC's positively charged surface and the negatively charged organic contaminant. Conversely, at high pH values, both GAC's surface and the organic contaminants tend to be negatively charged, causing repulsive forces between the GAC surface and the organic contaminant (Elavarthi, 2021). For instance, as the pH was increased, the adsorption of humic acid decreased due to increased electrostatic repulsion between the negatively charged humic acid and the negatively charged carbon surface (Menya et al., 2022). In the case of lead, due to decreased solubility, the removal efficiency increases as pH increases within the pH range for drinking water (Taylor and Kuennen, 1994). Moreover, the extent of the effect of pH on the removal efficiency may vary with type of contaminant. For instance, while the removal of diclofenac from water was dependent on pH, the removal of caffeine was independent of pH (Onaga Medina et al., 2021). Based on pH and pH_{zpc} values, the surface charge of the carbon can be ascertained. More specifically at $pH < pH_{zpc}$, the carbon surface exhibits a net positive charge, while at $pH > pH_{zpc}$, the surface has a net negative charge (Bello et al., 2017). According to Roegner et al. (2014), the higher the pH_{zpc} of carbon, the more likely it will have a neutral or positive charge under typical water conditions (Roegner et al., 2014). Electrostatic repulsion occurs between the negatively charged carbon surface and the negatively charged adsorbate, and vice versa. For instance, a negatively charged carbon surface would be attracted to the positively charged pesticide residues and metal ions (Ahmedna et al., 2004). However, it should be noted that deployment of B-GAC in POU water filters can influence the final pH of treated water, affecting potability of drinking water. For instance, the pH of treated water resulting from filtration using H_3PO_4 activated coconut shell was found to be ≈ 5.78 (Samdin et al., 2015), which is below the permissible range of 6.5–8.5 for public drinking water supplies (WHO, 2007). The low pH of treated water may be attributed to the residual acid that was present in the activated carbon (Menya et al., 2022). The pH of treated water, however, could also be a function of influent water pH. For instance, the pH of piped water prior to filtration with coconut shell-based GACs was found to be 5.5 (Samdin et al., 2015). While consuming acidic water may not pose public health concerns, acidic water can be corrosive, dissolving copper and lead present in household plumbing (Benham and Ling, 2013). The dissolved metals in drinking water cause water to have a bitter or metallic taste, alongside posing public health concerns. On the other hand, water treated with B-GAC may also exhibit an alkaline pH level beyond the permissible range for drinking water supplies. This may be attributed to the inorganic ions that get leached out from B-GAC into the water (Farmer et al., 1996). The rise in pH may also be attributed to the adsorption of chlorides, sulfates, nitrates and bicarbonates from the influent water onto the carbon's surface. A high pH causes a bad taste, alongside causing corrosion of metal pipe materials. Therefore, to overcome these challenges, the acidic or basic water needs to be stabilized to permissible levels. For instance, in the case of acidic water pH, the B-GACs may be rinsed severally to remove the residual acids until a neutral pH is attained. On the other hand, the basic water pH could be corrected by acid-washing of B-GACs followed by rinsing severally until a neutral pH is attained.

8.8. Surface loading

Surface loading is another parameter that influences the performance of a POU water filter. It is defined as the concentration of an organic pollutant in solution per gram of carbon. It influences the filter performance in several ways. For instance, as the surface loading is increased, the adsorption capacity of carbon reduces, lowering performance of the resultant POU water filter. In addition, as the surface loading increases, the contact time required to reach equilibrium also increases (Kuennen et al., 1992).

9. Performance challenges and enhancement

9.1. Performance challenges

The incorporation of GAC in POU filters potentially enhances the removal of ECs as already indicated in Section 6. However, POU systems still remain ineffective at removal of certain contaminants from water. For instance, when a POU filter was incorporated with oil palm-derived GAC, the viral reduction from water was found to be as low as 0.32 log (Shimabuku et al., 2017). A study by Su et al. (2009) also found a GAC-based POU filter to be ineffective at removal of bacteria from water. This low removal efficiency by the POU filter was attributed to microbial colonization in the GAC filter media, increasing total bacteria in the filter effluent (Pan and Giammar, 2022). Nanoparticles are also challenging to remove from water, as they significantly pass through the POU filters (Doré et al., 2021). For instance, the size of Pb nanoparticles are usually fine enough to pass through the pores of carbon-based filters. Owing to these challenges, there is a need for development of robust POU systems capable of removing contaminants that are difficult to remove with GAC deployment in POU filters. The potential interventions against these challenges are discussed in Section 9.2.

9.2. Performance enhancement

9.2.1. Post disinfection

Source water with a high microbial load exacerbates the problem of microbial colonization of the GAC media, leading to a filtrate with substantial levels of bacteria which pose health risks to humans and biodiversity. Thus, when source water of a high microbial load is used, disinfection of the filtrate should be included as part of the POU water filtration strategy, and vice versa. Post disinfection ensures microbiological safety of the filtrate. Disinfection methods may include chlorination, ozonation and use of ultraviolet light (UV) (US EPA, 2006).

9.2.2. GAC modification

Several research efforts have also been made to modify GAC for enhanced removal of contaminants from water. One of such efforts include surface modification of GAC with strong oxidizing agents which lead to generation of more adsorption sites on the carbon's solid surface (Babel and Kurniawan, 2004). For instance, Jang et al. (2008) modified the GAC surface with hydrous ferric oxide, enhancing the removal of arsenic from water. Lee (2018) impregnated GAC with iron or titanium metal oxides, reducing As(V) and Cr(VI) from levels of common

occurrence to below regulatory levels. Besides oxidising agents, GAC has also been modified by incorporating nanoparticles to their surface mainly for removal of pathogens such as bacteria, viruses and protozoa from water. Silver is the most widely employed nanoparticle due to its low toxicity and high level of microbial inactivation in water (Amin et al., 2014). For instance, Rivera-Sánchez et al. (2020) impregnated GAC with silver nanoparticles which was subsequently employed in combination with a ceramic silver-impregnated pot filter and zeolite, achieving an overall removal efficiency of Escherichia coli and Salmonella spp. of up to 99%. Watson et al. (2016) also impregnated GAC with silver nanoparticles for enhanced removal of brominated DBPs from water.

Moreover, since only small quantities of nanoparticles are needed to generate a high nanoparticle concentration within the GAC and provided the nanoparticles are efficiently bonded to the GAC, only trace quantities of the nanoparticles get leached to the water, thus public health concerns are minimized (Rajshree et al., 2013). For instance, when copper nanoparticles were employed in a POU filter to deactivate bacteria, the amount of copper released in the effluent water was < 1 ppm, which was allowable in drinking water (Dankovich and Smith, 2014). When silver nanoparticles were incorporated in GAC for water disinfection, the silver concentration in the treated water was found to be 29.8 μgL^{-1}, which is much below the permissible limit (100 μgL^{-1}) (Watson et al., 2016). In a study by Rivera-Sánchez et al. (2020), the silver concentration in the effluent filtered using a ceramic-silver impregnated pot filter with a post-filter composed of GAC and zeolite impregnated with colloidal silver was found to be lower than the permissible level of silver in drinking water by the Environmental Protection Agency (EPA). In a study by Garbo & Świecicka (2012), the amount of silver in filtered water resulting from jug filtration systems containing cartridges filled with silver-modified AC was found not to pose long-term risks to public health (Garbo and Świecicka, 2012). On the contrary, a study by Fewtrell et al. (2017) pointed to the risk of nanoparticles escaping into the water, leading to public health concerns. In such situations, there is a need to limit the escape of the nanoparticles into water to permissible limits. For instance, a polishing unit may be necessary to remove the nanoparticles from water. Besides the risk of nanoparticles escaping into water, there is also a concern of increased cost of the POU filter following modification of the GAC, since, for instance, a polishing unit may be required to remove the nanoparticles present in the filtered water. Rivera-Sánchez et al. (2020) reported a 31% increase in the cost of a ceramic silver-impregnated pot filter system coupled with silver-impregnated GAC and zeolite as compared to the cost of a ceramic silver-impregnated pot filter alone. According to Prathna et al. (2018), enabling the reuse and regeneration of nanomaterials could compensate for the high cost of nanomaterials responsible for the increased cost of POU filters. Moreover, the reuse and regeneration could minimize potential environmental concerns that otherwise arise from the poor disposal of the spent nanomaterials.

10. Installation and cost analysis

10.1. *Installation*

To ensure that the POU GAC filters operate efficiently and have a longer lifespan, it is essential to install the filter correctly to prevent leaks, filter malfunctions, and maintenance difficulties. Proper installation entails installation of the filter in a manner that allows easy and quick servicing and monitoring of the filter. For instance, for a POU GAC filter at the kitchen tap, the treatment unit can be suitably installed under the sink (US EPA, 2006).

10.2. *Cost analysis*

POU GAC filter systems are low-cost alternatives as compared to central water treatment plants. Nonetheless, GAC filter systems need to be assessed to ascertain whether they will be economically viable for the end-user. This requires conducting a cost analysis of the filter system, with an understanding of the capital and operation and maintenance (O&M) costs involved and how these costs would be affected by different factors. For instance, capital costs may be affected by purchase cost, installation cost, number of taps to be treated and engineering analysis, among others. On the other hand, O&M costs may be affected by the frequency of monitoring and maintenance of the filter system, cost of disposing of spent cartridges, filter media materials and particulate pre-filters, among others (US EPA, 2006). In spite of these insights, studies concerning cost analysis of POU GAC filter systems are still limited.

11. Monitoring and maintenance

GAC filter media are prone to microbial colonization, with a possibility of having higher levels of bacteria in the filtrate than in the influent water (US EPA, 2006). Consequently, monitoring of a heterotrophic plate count is recommended after water filtration. This ensures that the microbiological safety of the water is not compromised. POU GAC filter systems can also be monitored to identify system components unable to provide adequate protection to customers. Components such as cartridges, filter media materials and particulate pre-filters can, therefore, need replacement to restore normal performance of the filter system. However, to avoid high operation and maintenance costs, the maintenance requirements of a POU GAC filter should be kept to a minimal (US EPA, 2006).

12. Selection of POU filters

There are different types and designs of POU water filters that can be realized upon incorporation of biomass-derived ACs in water filters (see Section 6). Due to potential differences in type, amount and quality of filter media materials employed, the resultant POU filters also differ in cost and performance. Thus, due to these differences, there is a need for the judicious selection of POU water filters. According to Sobsey et al. (2008), POU filters are selected based on five sustainability criteria including; i) quantity of water produced, ii) ability to treat a range of water quality, iii) ease of operation and time to treat, iv) cost, and v) supply chain requirements. More specifically, a POU water filter capable of producing sufficient volumes of water for the entire household is desired. This reduces the need for a given household to rely on alternative source waters which may be contaminated. In achieving this criterion, however, user processing time should also be reasonable to reduce the need for users to supplement the treated water with untreated source waters. The POU water filter should also be tailored to a wide range of water quality such that regardless of source water quality, the filter is still capable of providing a given

household with water of high quality. On the side of supply chain requirements, POU water filters with limited/or no need for any type of supply chain should be selected, since costs that would otherwise be spent on supplying consumables are avoided. Compared to POU water filters loaded with commercial GACs, POU filters loaded with biomass-derived GACs are economically attractive, as they demand less carbon to achieve the same adsorption efficiency as their commercial counterparts (Ahmedna et al., 2004). Moreover, by regenerating and reusing the residual GACs from filtration, the economic viability of the POU water filters is further improved. For instance, residual GACs from a POU water filter was employed to remove hexavalent chromium (Cr(VI)) from contaminated-water (Sangkarak et al., 2020). Other considerations for selecting a POU filter include the ease with which the filter can be operated and maintained (Venkatesha et al., 2020).

13. Restoration of adsorption capacity

GACs are exhausted when their adsorption capacities get typically low. In such a scenario, the GACs can be backwashed, regenerated, recycled, incinerated or landfilled. In this Section, we highlight the backwashing and regeneration as processes for restoration of GAC's adsorption capacity.

13.1. *Backwashing*

GAC filters may require backwashing at the beginning (or initial backwashing) and from time to time during operation (or operational backwashing). The former is carried out on the virgin or reactivated GAC to remove air, fine particles and to segregate the bed while operational backwashing is required to overcome pressure drop due to build-up of particles across the GAC filter bed (Chemviron Carbon, 2021). The frequency of backwashing depends on the influent turbidity and the throughput of the GAC filter adsorber. The backwashing process can, however, significantly change the properties of GAC, affecting the performance of the GAC filter (Clements and Haarhoff, 2004). The process also results in generation of liquid waste, which would need to be treated prior to discharge to the environment (US EPA, 2006).

13.2. *Regeneration*

GACs are exhausted when their adsorption capacities get typically low. In such a scenario, the spent GACs can be backwashed, regenerated, recycled, incinerated or landfilled. Regeneration is the most widely adopted due to its cost effectiveness and environmental-friendliness as compared to the other options (Clements, 2002). Regeneration involves desorbing accumulated adsorbates and restoring the original porous structure with minimal or no damage to the carbon. Moreover, the regenerated carbon could be disposed of safely to the environment. However, the ease at which the spent GAC is regenerated depends on whether the adsorption is physical or chemical. Physical adsorption requires heating of the GAC to cause a shift of equilibrium from adsorption to desorption, while chemical adsorption requires an energy supply greater than the adsorptive force required to break the strong ionic or covalent bonds (Ani et al., 2020).

Regeneration of spent GACs is mainly achieved through thermal and chemical regeneration methods (Clements and Haarhoff, 2004; Larasati et al., 2020). Thermal regeneration involves heating to provide the energy required for removal of the adsorbate from the GAC's surface. The process involves dewatering, drying, pyrolysis, oxidating, reactivating and quenching (Mishra, 2021). Several researchers have employed the thermal process to regenerate the GACs. For instance, Márquez et al. (2022) regenerated the spent carbon by heating at temperatures $\leq 350\,^{\circ}C$ for 1 h using an oxidizing atmosphere. The regenerated carbon exhibited a specific surface area and micropore volume of $\approx 475\,m^2g^{-1}$ and $0.264\,cm^3g^{-1}$, respectively, which were close to $406\,m^2g^{-1}$ and $0.229\,cm^3g^{-1}$ for pristine carbon. Egirani et al. (2021) regenerated the spent carbon by treating it with 1 M nitric acid, followed by heating at 500 °C for 24 h. Consequently, the generated carbon exhibited a removal efficiency of 90% while its adsorption capacity regained up to 60%. Despite these promising results, thermal regeneration has some drawbacks. For instance, the process is linked to loss of adsorption capabilities due to changes in GAC's pore structure, needs high energy requirements and pollutes air with the emitted gases. Some adsorbates are also incompatible with the high-temperature operations that are typical of thermal regeneration (Sufnarski, 1999).

For chemical regeneration, alkali or acids with oxidizing capacity are applied to the spent carbon along with organic solvents that dissolve the adsorbates and restore the properties of the spent carbon (Mishra, 2021). For instance, Newcombe & Drikas (1993) chemically regenerated spent GAC from an operating water treatment plant in Australia. The researchers achieved high regeneration efficiencies for adsorption of DOC. Larasati et al. (2020) evaluated four types of regenerant solutions in batch tests for their ability to desorb five target contaminants from GAC. The researchers found $NaOH/CH_3CH_2OH$ to be more effective in desorbing the target contaminants from the spent GAC than the individual solutions. Chemical regeneration has several advantages over thermal regeneration. For instance, unlike thermal regeneration, chemical regeneration is not associated with burn-off responsible for carbon loss. Chemical regeneration can also be done in situ, avoiding losses that arise due to pumping, transport and repacking of the regenerated carbon. The disadvantages, however, include the high cost of reagents and environmental pollution that could arise from chemical use (Ani et al., 2020). Other methods that can be employed to regenerate GACs include wet air oxidation, electrochemical regeneration, microwave regeneration and biological regeneration (Mishra, 2021). Despite the research progress made thus far, studies focused on the regeneration of B-GACs deployed in POU water filters are yet to be done.

14. Recycling

Though backwashing and regeneration procedures can extend the service life of a GAC filter, the filter will eventually become saturated with contaminants. To divert the spent filter from reaching the landfill and consequently minimize wastage, recycling of the saturated GAC filter is necessary. This can be achieved through downcycling of the GAC filter, especially when dealing with contaminated low-cost adsorbents. The choice of method depends on the type of contaminants, the level of contamination and the intended use of the recycled carbon. An example of such an approach is the work of Wurzer et al. (2023) who explored the recycling of EC-saturated GACs using thermochemical treatment. The approach resulted in a high carbon retention in biochar,

making it a viable candidate for environmentally sound recycling of carbon adsorbents saturated with ECs. Another approach has been pursued by Li et al. (2017), where the electrochemical energy storage potential of GAC downcycled from water filters was demonstrated. Through thermochemical treatment, supercapacitor electrodes were created with excellent energy densities of up to 32.9 Whkg $^{-1}$. In a study by Chen et al. (2012), the spent activated carbon were water-washed and subsequently added into organic sludge to form organic sludge-derived fuel. This exhibited improved fuel properties in terms of carbon content and heating value. These, and many other studies show great promise for recycling of spent GAC filters.

15. Life cycle assessment

B-GAC exhibits huge potential for deployment in POU water filtration due to its effectiveness in removal of ECs from water. However, its production and application pose significant emissions to the environment. Through a life cycle assessment (LCA), several studies have quantified the environmental impacts of GAC production from waste biomass. For instance, Loya-González et al. (2019) conducted an LCA on the production of activated carbon from corn pericarp. Amin et al. (2022) used LCA to quantify the environmental impacts associated with the production process of activated carbon from olive trees. Their study revealed that chemical activation using H_3PO_4 exhibited the greatest ecological impacts, while the total energy demand and global warming potential were found to be 70.521 MJkg^{-1} and 5.210 kg CO_{2eq}, respectively. Vilén et al. (2022) conducted a cradle-to-gate LCA on coconut shell, wood, coal, peat and reactivated coal GACs. Their study revealed that coal-based GAC exhibited the highest impact, while coconut-based GAC and reactivated coal GACs exhibited the lowest impacts. These, and many other studies point to the fact that the choice of raw material and production method significantly influence the impact of GAC production on the environment. However, despite the numerous studies conducted to quantify the environmental impacts due to various waste biomass-derived GACs, most LCAs are limited to the basis of cradle-to gate. LCAs focusing on cradle-to gate-to grave are, therefore, needed to provide adequate information on whether, on a life cycle basis, B-GACs do not cause environmental impacts which otherwise outweigh the benefits of deploying B-GACs in POU filters.

16. Future prospects

Abundant quantities of agricultural residues and biomass wastes are available in developing countries which can be leveraged to produce GAC for POU water purification. Unfortunately, the production of GAC from biomass wastes for POU water purification has been limited to shells of nuts such as English, almond, coconut, peanut, pecan and walnut shells. Though these GAC precursors exhibit superior properties as compared to the soft, low-density biomass wastes such as rice husks, there is a need to tailor these biomass wastes and suit them as GAC precursors. This could help encourage the wide application of biomass waste-derived GACs for POU water purification. Therefore, future studies need to focus on producing GAC from soft, low-density biomass wastes, particularly through densification to produce a hard, higher density briquette or pellet which can be subsequently carbonized and activated.

The application of biomass-derived GAC in POU water filters for removal of ECs has been demonstrated in various studies (Anisuzzaman et al., 2014; Erhuanga et al., 2014; Fernandes, 2017; Iwar et al., 2022; Kuok et al., 2022; Shimabuku et al., 2017; Suif et al., 2020; Xiao et al., 2020; Yamina et al., 2013; Yusuf and Murtala, 2020) where the biomass-derived GACs have been deployed either as a sole filter media material or in combination with other filter media materials. The findings reported in the literature show promising prospects for utilizing the biomass-derived GACs for removal of ECs at the point-of-use. However, the studies are still few and with limited reports on the mechanisms by which the biomass-derived GAC-based POU water filters get rid of specific ECs from water. Thus, to facilitate more understanding of the filtration process and ultimately optimize the removal efficiency of ECs, future studies should focus on tailoring biomass-derived GACs for removal of specific ECs from water. Moreover, for improved economic viability of deploying GAC for POU water purification, possibilities of regenerating the biomass-derived GAC could also be explored in future studies.

Research has shown that deployment of GAC for POU water purification enhances the removal of ECs from water (see Section 5). However, the resultant POU filters are ineffective at removal of some of the contaminants from water such as bacteria and virus. This has prompted research efforts aimed at suiting the GAC for removal of such contaminants, including surface modification of GAC using oxidizing agents and nanoparticles (Jang et al., 2008; Lee, 2018; Rivera-Sánchez et al., 2020; Watson et al., 2016). Moreover, since small quantities of nanoparticles are needed for this process, only trace quantities of nanoparticles may leach to the water, minimizing public health concerns (Rajshree et al., 2013). In the case that nanoparticles in the filtrate are at concentrations beyond permissible limits, then a polishing unit may be required to remove the residual nanoparticles from the filtrate. Moreover, reuse and regeneration of nanoparticles could further offer promising prospects for improving the economic viability of POU water purification while at the same time offsetting the need to dispose of the nanoparticles into the environment. Lastly, future studies should also consider life cycle analysis and costing of various GAC-based POU water filters to facilitate decision making on the potential technologies which may ultimately encourage their wider uptake and adoption, especially in resource-constrained settings.

17. Conclusion

A review of progress in the deployment of biomass-based activated carbon for removal of emerging contaminants at the point of use has been made. The review highlights the key sources, pathways and impacts of emerging contaminants on human health, discusses the current progress in the preparation and performance of biomass-based GACs for POU water purification, discusses the factors influencing performance of POU water filters, discusses performance challenges of POU water filters and how these could be averted through modification of GAC. This review revealed that the production of GAC from biomass wastes for POU water purification is limited to shells of nuts, particularly the English, almond, coconut, peanut, pecan and walnut shells

which are hard, high density biomass wastes. However, to encourage the wide application of biomass waste-derived GACs for POU water purification, there is a need to tailor the soft, low-density biomass wastes and suit them as GAC precursors for POU water purification. This can be achieved through densification to produce a hard, higher density briquette or pellet which can be subsequently carbonized and activated. The review also reveals the limited reports on the by which biomass-derived GAC-based POU water filters get rid of specific ECs from water. Thus, to facilitate more understanding of the filtration process and ultimately optimize the removal efficiency of ECs, future studies need to focus on tailoring biomass-derived GACs for removal of specific ECs from water. Moreover, for improved economic viability of deploying GAC for POU water purification, possibilities of regenerating the biomass-derived GAC should also be explored in future studies. In an attempt to incorporate nanoparticles on GAC's surface and enhance its removal efficiency of contaminants such as bacteria and virus, keen consideration should be taken to ensure residual nanoparticles do not escape to the filtrate, posing public health concerns. Overall, the incorporation of GAC in POU filter systems offers good prospects for utilizing the biomass-derived GACs for removal of ECs at the point of use.

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The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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