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Sustainable Use of Biochar

From Basics to Advances

Edited by Hanuman Singh Jatav, Satish Kumar Singh and Bijay Singh

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Meet the editors

Dr. Hanuman Singh Jatav is an assistant professor in the Department of Soil Science and Agricultural Chemistry, Sri Karan Narendra Agriculture University, Jobner Jaipur (Rajasthan), India. He has more than seven years of teaching and 10 years of research experience in the fields of sewage sludge, biochar, micronutrients, heavy metals, and soil fertility management. He has published more than 120 papers including research papers, review papers, books, book chapters, and technical papers with various nationally and internationally reputed journals and publishers. Dr. Jatav was awarded a Certificate of Gratitude as an Ambassador by IUCN: International Union for Conservation of Nature, in Gland, Switzerland (https://www.iucn.org/) during the World Conservation Congress at the Global Youth Summit. He has been awarded an Outstanding Assistant Professor of the SKNAU, Jobner Jaipur (Rajasthan), India.

A soil chemist by training and one of the ten National Professors of the Indian Council of Agricultural Research from 2006 to 2012, Dr. Bijay Singh is presently associated with Punjab Agricultural University, Ludhiana (Punjab, India) as an INSA Honorary Scientist. He served as senior soil chemist from 1994 to 2006 and as INSA senior scientist from 2012 to 2017. Dr. Bijay Singh has contributed over 370 publications in high-impact-factor journals. He is a fellow of the Indian National Science Academy, National Academy of Agricultural Sciences, and Indian Society of Soil Science. He has received several national awards including the Rafi Ahmad Kidwai Memorial Prize of the Indian Council of Agricultural Research.

Dr. Satish Kumar Singh received his BSc (Ag) degree in 1985, MSc (Ag) degree in 1988, and his Ph.D. in soil science and agricultural chemistry in 1992 from Banaras Hindu University, Varanasi. He started his professional career in 1990 as a lecturer of agricultural chemistry at Amar Singh (PG) College, Lakhaoti, Bulandshahr. Subsequently, Dr. Singh joined as an associate professor in the year 1999 at G.B. Pant University of Agriculture and Technology, Pantnagar (Uttarakhand), and as a professor in 2006 at Banaras Hindu University, Varanasi. Dr. Singh is a fellow of the Indian Society of Soil Science, New Delhi, and the Indian Society of Agricultural Chemists, Allahabad. He has more than 270 publications to his credit including research papers, books, book chapters, laboratory manuals, technical bulletins, technical folders, and so on.

Contents

Preface

The present book *Sustainable Use of Biochar – From Basics to Advances* involves maximizing the benefits of biochar while minimizing potential drawbacks or unintended consequences. Here the book compiles some key principles and practices for the sustainable use of biochar that cover several aspects. The key points of the book are ecosystem management through biochar application techniques. It offers complete details of biomass sources that ensure biomass used for biochar production avoids using feedstocks that contribute to deforestation, land degradation, or depletion of natural resources. This book also covers efficient and environmentally friendly pyrolysis processes that minimize emissions and energy consumption. The book covers technologies such as slow pyrolysis or carbonization in closed systems that can reduce greenhouse gas emissions and energy requirements scientifically. It has detailed information on quality control standards for biochar production, including parameters such as particle size, porosity, stability, and nutrient content. High-quality biochar is more effective in improving soil health and carbon sequestration. The book gives in-depth information on soil biochar application judiciously based on soil type, crop requirements, and local conditions. It covers information on integrating biochar application with other sustainable agricultural practices such as organic farming, crop rotation, cover cropping, and water conservation measures. It provides information on long-term monitoring of the effects of biochar application on soil health, crop productivity, carbon sequestration, and environmental outcomes. It gives basic information on biochar application effects on changes in soil properties, microbial activity, nutrient cycling, and greenhouse gas emissions over time. The whole book has stepwise information. Integrating biochar with other sustainable agricultural practices and promoting knowledge sharing further enhances its sustainable use, fostering resilience and environmental stewardship in agriculture. Biochar applications can cope with global challenges of soil management, and ecological approaches restore the damaged ecosystem. The present sustainable use of biochar from basics to recent advances will be helpful to researchers, the scientific community, academics, business farmers, and policymakers to overcome various environmental concerns.

I, Dr. Hanuman Singh Jatav, as an author of the book "Sustainable Use of Biochar – From Basics to Advances," acknowledge my thanks to Prof. Balraj Singh, Honorable Vice-Chancellor, Sri Karan Narendra Agriculture University, Jobner-Jaipur, for the motivation to compile this work. I am deeply indebted to Dr. Amitava Rakshit, associate professor, Banaras Hindu University, Varanasi for his lifelong motivation to compile the work. I also extend my thanks to Prof. K.K. Sharma (HOD-SSAC) and Dr. S.K. Dadhich for their direct and indirect support.

It's very difficult to put into words my thanks to Prof. Harphool Singh (Dean-COA Fatehpur-Sikar) and Prof. S.R. Dhaka (Dean-COA Bharatpur) who not only supported but also guided me to compile this volume for the students. I would like to thank from bottom of my heart Dr. Sanjay K. Attar, Dr. Muddser A. Khan, Dr. K.C. Verma,

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I am also thankful to Prof. Bijay Singh (Co-Editor) for his critical guidance in choosing the book topic and Prof. Satish Kumar Singh (Co-Editor) for his regular efforts in compiling and critical evaluation of this book. I am also thankful to Dr. Vishnu D. Rajput for his continued support in checking the content to make it very worthy. I also extend my sincere gratitude to my family members who always support me in any difficult situation.

The graces of the Lord Hanuman Ji, Goddess Maa Sharde along with my parents Smt. Supedi Devi and Shri Babulal Jatav have always blessed me and given me the patience and power to overcome difficulties that came my way in the accomplishment of this endeavor. I cannot dare to say thanks but only pray to bless me always.

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Dedication

Dedicated to my parents, Late. Shri Babulal Jatav and Smt. Supedi Devi

For my parents, whose love, wisdom, and sacrifices have been the foundation of my journey.

This is dedicated to you with all my gratitude and affection.

Chapter 1

Approaches of Biochar in Ecosystem Management: Current Scenario and Future Perspectives

Ipsita Samal, Deepak Kumar Mahanta, Tanmaya Kumar Bhoi, J. Komal, Hanuman Singh Jatav, Surendra Singh Jatav and Eetela Sathyanarayana

Abstract

Agricultural crop growth and productivity are significantly influenced by a wide variety of biotic and abiotic factors. In order to address these shortcomings, substantial amounts of chemical fertilisers are administered to the land. The widespread use of chemical fertilisers has led to the degradation of ecosystems and various associated issues, including decreased nutritional quality of crops and the long-term decline in soil fertility. The excessive uses of fertilisers and pesticides have adverse implications for soil vitality, resulting in a substantial reduction in the biomass. Therefore, the use of biochar has been sustainable method and a potentially efficient strategy for improving soil quality and addressing the issue of heavy metal pollution in soil. Integrating biochar into the soil offers a significant chance to enhance soil quality and promote plant growth. The efficacy of biochar in enhancing nutrient cycles on agricultural lands is highlighted by its positive impact on plant growth and soil vitality, rendering it a practical instrument for mitigating nutrient deficiencies. The present chapter focuses on the utilisation of biochar and its impact on the soil microbial population, plant diseases, plant-parasitic nematodes, and insect pests and highlights the utility of biochar as an effective agent for plant protection.

Keywords: biochar, plant protection, sustainable method, plant diseases, insect pests

1. Introduction

The growth and productivity of agricultural crops are profoundly impacted by a diverse range of biotic and abiotic factors [1, 2]. To address these deficiencies, significant quantities of chemical fertilisers are applied to the soil [3, 4]. However, it is crucial to acknowledge that plants possess a restricted capacity to assimilate water-soluble nutrients. The remaining elements undergo a transformation process, leading to the creation of forms that are not soluble. Consequently, it becomes imperative to periodically administer fertilisers in order to ensure a consistent provision of essential nutrients for the purpose of facilitating plant growth [5]. The extensive utilisation of

chemical fertilisers has resulted in the deterioration of ecosystems and various related problems, such as diminished nutritional value of crops and the long-term reduction of soil fertility [6, 7]. Alongside fertilisers, pesticides provide a significant concern within the field of agriculture due to their noteworthy environmental consequences, which exert a major influence on the microbiological characteristics of soil. The overuse of fertilisers and pesticides, as well as their persistent presence in the soil, have negative consequences for soil health, leading to a significant decrease in the biomass of bacteria and fungus [8, 9]. Huang et al. [10] conducted a study to investigate the effects of prolonged exposure to inorganic fertilisers and/or organic manures on the structural diversity and dominant bacterial groups in agricultural soils. However, it is important to acknowledge that biofertilizers have the ability to enhance soil fertility by revitalising it. As a result, they present themselves as advantageous candidates for the promotion of sustainable agriculture and the mitigation of stress within agroecosystems. Moreover, the incorporation of organic soil supplements, particularly in the context of remediation, is occasionally justified based on their cost-effectiveness, often necessitating the implementation of alternative waste management methods (such as landfill deposition or cremation). In order to meet certain criteria, soil amendments must possess certain characteristics such as a strong capacity for binding, compatibility with the surrounding environment, and the lack of any detrimental impacts on soil structure, fertility, or the wider ecosystem [11]. The utilisation of biochar has been recognised as a sustainable approach and a potentially effective technique for enhancing soil quality and mitigating the problem of heavy metal contamination in soil [12]. When a bibliometric analysis was conducted using the key words "biochar", "insect pest management" and "plant protection", it was observed that the total number of publications was increased since 2001 (**Figure 1a**), while in agricultural, Veterinary and Food Sciences, Environmental Sciences and Biological Sciences related journals, this topic was focussed more (**Figure 1b**). Further, this topic was more focussed on diverse articles followed by book chapters and edited books (**Figure 1c**), while major articles were published in encyclopedia of the unsustainable development goals and the science of the total environment (**Figure 1d**), and bibliometric studies also indicated that biochars related to plant protection contributed higher to sustainable development goals (SDG) 2 (zero hunger) followed by SDG 13 (climate action) and SDG 15 (life on land) (**Figure 1e**). Thus, current study analyses the plant protection potential of biochar and its way forward in pest management.

A carbonaceous material with a sizable percentage of organic materials makes up biochar, an organic amendment. This chemical is produced as a byproduct of pyrolysis, a process that involves heating biomass to high temperatures and low oxygen levels. The process of pyrolysis, which includes the thermal breakdown of biomass materials like wood, dung, or leaves at high temperatures in an oxygen-poor atmosphere, produces biochar. The aforementioned procedure results in the production of biochar as the principal output, along with minor byproducts like as oil and gas. The extent of these remaining compounds is dependent on the specific processing parameters. Recent research have revealed that biochar, derived from the carbonisation of organic waste, possesses the potential to serve as a viable replacement material. The replacement of a certain element has consequences for the process of storing carbon in soil, as well as alterations to its physical, chemical, and biological characteristics [13]. The use of biochar shows promise in the production of renewable energy in agricultural regions, while also aligning with environmentally conscious principles. In a previous study, Verheijen et al. [14] observed that the use of biochar had discernible effects on the toxicity, transport, and destiny of specific heavy metals within soil.

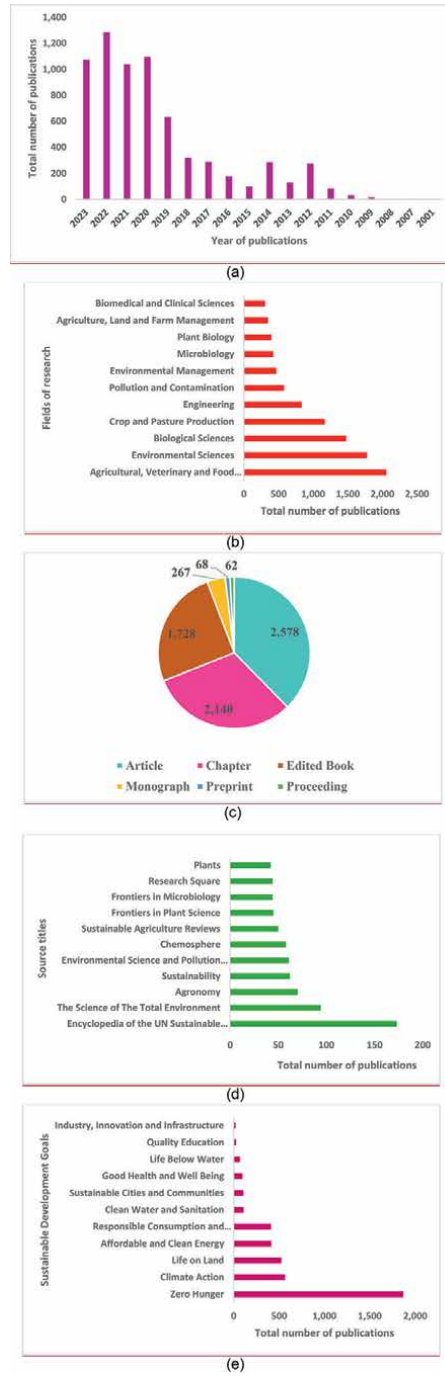

Figure 1.
This figure depicts the total number of publications were increased since 2001 (a), while, in agricultural, veterinary and food sciences, environmental sciences and biological sciences related journals this topic was focussed more (b). Further, this topic was more focussed in diverse articles followed by book chapters and edited books (c), while, major articles were published in encyclopedia of the un sustainable development goals and the science of the total environment (d) and bibliometric studies also indicated that, biochars related to plant protection contributed higher to sustainable development goals (SDG) 2 (zero hunger) followed by SDG 13 (climate action) and SDG 15 (life on land) (e).

The primary cause of this phenomenon was predominantly ascribed to the enhanced soil adsorption capacity aided by the presence of biochar. Several key elements can be ascribed to the enhanced soil properties and heightened nutrient uptake by plants in soils treated with biochar. The nutrient and ash composition, expansive surface area, porous structure, and microbe habitat function are among the aspects that encompass biochar [15]. The research conducted by Rawat et al. [16] demonstrated that the use of biochar led to a reduction in soil compaction, indicating its potential to effectively mitigate this issue. Significant attention has been devoted to evaluating the advantages of introducing rhizobacteria into soil. Nevertheless, it is crucial to recognise that the incorporation of biochar into the soil can also enhance the availability of nutrients, hence providing benefits to agricultural products. Bhanse et al. [17] emphasise the utilisation of plant growth-promoting microorganisms in conjunction with biochar as the optimal approach for boosting the development and output of French beans. The integration of biochar into the soil presents a considerable opportunity to improve soil quality and stimulate plant growth, so making a valuable contribution to the development of a sustainable agricultural paradigm. Extensive research has been conducted to investigate the viability of utilising biochar additions for soil reclamation [18] and for the promotion of sustainable agriculture practises that aim to achieve high crop yield while mitigating environmental damage. The potential of biochar to increase nutrient cycles on farms is underscored by its good influence on plant growth and soil health, making it a realistic tool for addressing nutrient deficits. As a result, there has been a significant focus on examining the advantageous impacts of using biochar amendments in relation to soil stability and the facilitation of plant growth.

2. Role of biochar in plant protection

What are the processes by which biochar regulates plant diseases? There have been a minimum of five proposed mechanisms, namely: (i) the induction of systemic resistance in host plants; (ii) the augmentation of beneficial microorganisms in terms of their abundance and/or efficacy; (iii) modifications to soil quality with respect to nutrient accessibility and abiotic factors; (iv) the direct fungitoxic effect of biochar; and (v) the adsorption of allelopathic and phytotoxic compounds (**Figure 2**). The phenomenon of induced resistance in plants has been suggested as a potential mechanism for the regulation of disease suppression [19, 20]. The application of biochar was carried out in this study in a specific geographic region that was physically segregated from the infection sites. This purposeful dissociation was carried out in order to successfully eradicate any alternative mechanisms that may otherwise aid in the suppression of illness. More empirical support for the idea of induced resistance was offered by Harel et al. [21] and Mehari et al. [22], who showed that both the induced systemic resistance (ISR) and systemic acquired resistance (SAR) pathways were involved. According to a study by Harel et al. [21], adding biochar to substrates used to grow strawberry plants caused several genes to be noticeably upregulated. The genes contained in this collection encode three pathogenic-related proteins (FaPR1, Faolp2, and Fra a3), a lipoxygenase producing gene (Falox), and a trans-acting factor (FaWRKY1) gene from the WRKY family. Mehari et al. [22] investigated the role of jasmonic acid (JA) in biochar-induced systemic resistance (ISR) in tomato plants by looking at the *S. lycopersicon-B. cinerea* pathosystem. Nevertheless, the data presented by Mehari et al. [22] and Harel et al. [21] was insufficient to determine whether the introduction of biochar altered the rhizosphere microbiome's composition and

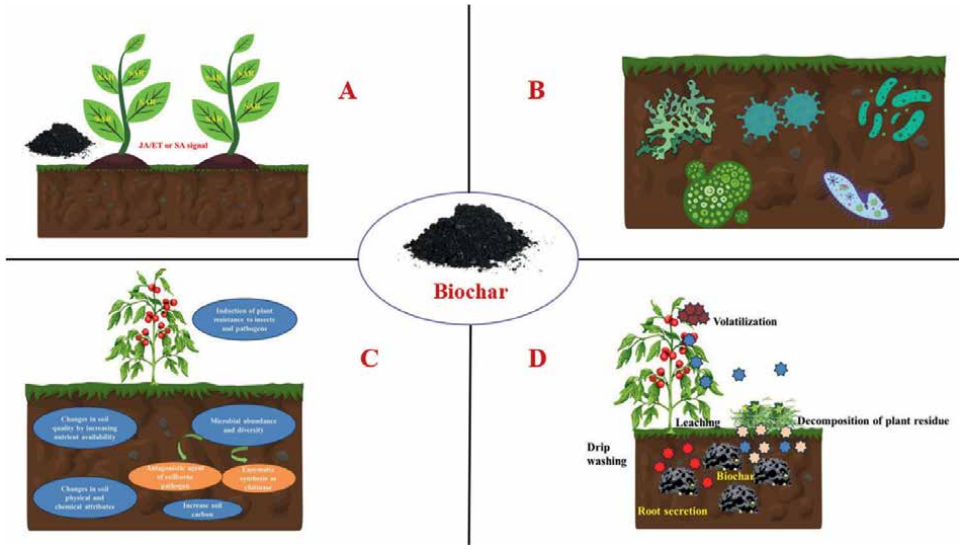

Figure 2.
Representing how biochar is affecting plant defence against biotic stresses, (A) induction of systemic resistance in host plants, (B) augmentation of beneficial microorganisms, (C) modifications to soil quality, including nutrient accessibility and abiotic factors and (D) adsorption of allelopathic and phytotoxic compounds.

functions or whether specific chemical compounds present in biochar directly caused induced plant resistance.

The second hypothesised mechanism for illness suppression involves the potential augmentation of beneficial bacteria' proliferation and/or functions through the incorporation of biochar. Subsequently, these bacteria confer protection to the plant by mitigating the risk of pathogenic assaults. There is a growing body of empirical evidence that substantiates the proposition that biochar exerts a beneficial influence on various facets of microbial activity. Liang et al. [23] demonstrated that the application of biochar results in a significant augmentation of microbial biomass. Furthermore, the study conducted by Warnock et al. [24] has demonstrated that the application of biochar has a positive impact on the colonisation of roots by mycorrhizal fungi. In addition, Graber et al. [25] and Kolton et al. [26] have both provided evidence supporting the notion that biochar facilitates the proliferation of microbes that stimulate plant growth. Positive effects have been found to be correlated with both physiological and nutritional factors. According to Lehmann et al. [27], the porous structure and substantial specific surface area of biochar create an environment that is favourable and safe for many microorganisms, such as mites, collembolan, protozoans, and nematodes. Based on the findings of Downie et al. [28], it has been observed through empirical research that the porous composition of biochar can effectively serve as a refuge for bacteria and mycorrhizal fungus, enabling them to evade predators. The assertion is additionally corroborated by the research conducted by Warnock [24]. In terms of its nutritional implications, biochar possesses the capacity to provide organic carbon that can facilitate the proliferation of saprophytic microbes. Nevertheless, it is crucial to acknowledge that the impact of this phenomenon is expected to be considerably less prominent in comparison to other organic additions, such as agricultural wastes and composts. The biochemical compatibility of biomass with microbial needs undergoes a substantial decrease throughout the pyrolysis process. The main reason for this phenomenon can be attributed to the

progressive exhaustion of carbon sources that can be easily broken down, coupled with the simultaneous accumulation of aromatic constituents that exhibit resistance to degradation [29]. Consequently, through the process of pyrolysis, biochar undergoes a conversion into an organic material that promotes agricultural productivity, albeit with restricted ability to facilitate microbial growth. The findings mentioned above collectively indicate that biochar has the potential to be a viable alternative to soil supplements such as agricultural wastes or composts. This is because biochar has the ability to enhance the functionality of beneficial microorganisms selectively, while also preventing the proliferation of pathogen populations and their detrimental impacts. Further investigation is required to explore this topic in greater depth, as the current body of research is limited in terms of establishing a clear link between changes in microbial communities resulting from biochar and the successful mitigation of diseases. This is despite the growing knowledge surrounding the influence of biochar on soil microbiomes, as highlighted by Lehmann et al. [27].

The third hypothetical mechanism posits that modifications in soil characteristics, specifically pertaining to nutrient accessibility and abiotic factors, have the potential to impact the overall dynamics of plant-pathogen interactions. In accordance with the findings of Gaskin et al. [30], the addition of biochar supplements generally enhances the concentrations of essential soil cations, including calcium (Ca^{2+}), magnesium (Mg^{2+}), and potassium (K^{+}). Furthermore, the study conducted by Yuan and Xu [31] revealed that the use of biochar amendments has a tendency to increase soil pH levels. Nevertheless, the impact of bioavailability on crucial plant nutrients, such as nitrogen and phosphorus, remains a subject of significant debate [32]. The biochar generated by the pyrolysis process frequently has an elevated carbon-to-nitrogen (C/N) ratio in comparison to the initial feedstocks. This is primarily attributed to the selective elimination of nitrogen in favour of organic carbon during the pyrolysis procedure. The C/N ratio of the biochar produced is determined by various factors, including the temperature used during the pyrolysis process and the initial characteristics of the biomass used. Schofield et al. [33] posited that the incorporation of organic materials characterised by a high carbon-to-nitrogen (C/N) ratio into soil leads to the augmentation of microbial activity. The heightened microbial activity that ensues consequently restricts the accessibility of mineral nitrogen, thereby impeding the saprophytic abilities of pathogens and hence inhibiting the progression of illness. Based on the aforementioned data, it may be deduced that biochar exhibits considerable potential in impacting the interactions between plants and pathogens. However, it is crucial to acknowledge that, based on current knowledge, there exists a dearth of definitive empirical data from research studies that definitively establish a causal relationship between the augmentation of soil nutrient levels or modifications in soil abiotic factors, such as the liming effect, through the utilisation of biochar, and the effective mitigation of diseases.

One plausible mechanism that may account for the decline in illnesses is the direct fungitoxic effect of biochar. Significant chemical transformations take place during the process of biomass pyrolysis, resulting in the degradation of O-alkyl carbons found in carbohydrates. Concurrently, there is a simultaneous generation of aliphatic and aromatic carbon compounds. In addition, it has been noted by Spokas et al. [34] that pyrolysis produces a diverse range of organic compounds that possess the capacity to demonstrate fungitoxic characteristics. However, studies investigating the precise fungitoxic properties of biochar have indicated that its ability to prevent fungal growth is often minimal or insignificant. An example of this may be seen in the study conducted by Jaiswal et al. [35], where it was observed that various forms of biochar

effectively inhibited the occurrence of damping-off disease caused by *Rhizoctonia solani* on *Phaseolus vulgaris*. However, numerous tests conducted in vitro and in vivo have consistently demonstrated that biochar has a limited or insignificant direct inhibitory effect on *R. solani*. The research group achieved similar results in their tests investigating the impact of *Medicago sativa* hay and wood biochars on hyphal growth in *Aspergillus niger, Fusarium oxysporum*, *Penicillium italicum*, and *Rhizoctonia solani*.

3. Biochar in plant-biotic interactions

3.1 Biochar in insect pest management

The existing body of research has extensively examined the positive impacts of biochar on soil in terms of chemical, physical, and microbiological enhancements. However, the potential indirect consequences of biochar on plant diseases and herbivorous insects in soils modified with biochar have not been thoroughly investigated (**Figure 3**). While several studies have demonstrated the potential of biochar applications in reducing infections caused by soilborne pathogens like *Fusarium* and *Ralstonia solanacearum* in crop plants, as well as mitigating foliar fungal infections such as *Botrytis cinerea* and minimising damage caused by pests like the broad mite (*Polyphagotarsonemus latus* Banks), there is a noticeable lack of research focusing on the effects of biochar on herbivorous insect pests. The utilisation of biochar in soil has attracted considerable attention owing to its capacity to mitigate greenhouse gas emissions, improve soil fertility, and enhance agricultural productivity. Nevertheless, there has been a relative lack of attention given to the impact of biochar additions on herbivorous insect pests. The objective of this study was to examine the potential impact of biochar supplementation on the developmental and reproductive outcomes of the rice brown planthopper (*Nilaparvata lugens*) while its feeding on rice plants.

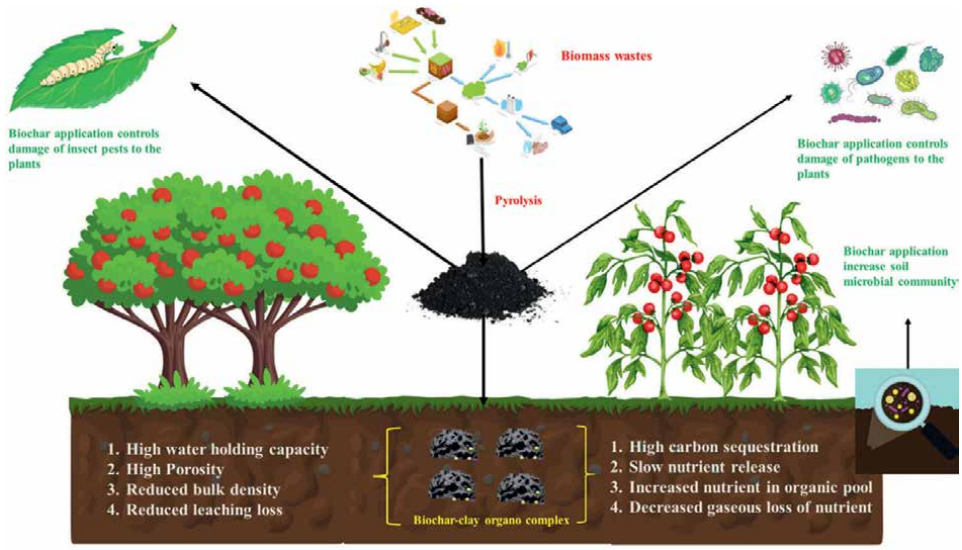

Figure 3.
Depicting biochar in plant-biotic interactions like insect pest management, controlling plant pathogens and increasing soil microbial community.

Biochar generated from wheat straw through the process of pyrolysis was utilised in the application of soils originating from a fallow rice field, with different amounts of application. Subsequently, the aforementioned treated soils were employed for the cultivation of rice seedlings within compact containers, hence facilitating the examination of the life cycle of *N. lugens*. The findings of the study indicate that the application of a substantial amount of biochar (200 g/kg of soil) resulted in a significant delay in the development of nymphs and a decrease in the survival rates of nymphs transitioning into adulthood. Moreover, the fertility of herbivores across their lifespan exhibited a decline with increasing rates of biochar application. Specifically, the number of eggs produced reduced from 256 eggs in the control circumstances to 69 eggs at the maximum biochar concentration. The egg-hatching rates exhibited a notable decrease when the biochar content reached its maximum value of 200 g/kg, in contrast to the lower biochar levels. The findings of this study indicate that the addition of biochar to rice fields, especially when applied at high rates, could potentially have negative consequences on the development and reproductive abilities of rice brown planthoppers [36]. In an independent investigation, considerable impacts on the population of *Cnaphalocrocis medinalis*, a prevalent pest, were identified as a result of incorporating biochar. In particular, the addition of biochar resulted in the prolongation of the larval development phase, an elevation in larval death rates, a drop in the final body weight of fully developed larvae, a reduction in the intake of rice leaves by the larvae, and a decrease in the adult lifespan of *C. medinalis*. It is important to note that, irrespective of the different levels of biochar treatment, there was no discernible variation in the ability to reproduce among the female insects that were able to mature. Another study that showed how biochar affected *C. medinalis* populations was a two-year study carried out in rice fields. In the first year, the use of charcoal resulted in a decrease in the *C. medinalis* population. The population level was discovered to be comparable to the control group in the second year, although. The study's results offer strong proof that using biochar can successfully impede *C. medinalis*'s development and proliferation, possibly resulting in a decrease in the species' population size [37]. A further investigation was undertaken to examine the efficacy of nanoscale and conventional biochar amendments in eliciting resistance in *Nicotiana benthamiana* when exposed to the pathogen *Phytophthora nicotianae* [38]. Nanoscale biochar derived from maize straw, produced at a temperature of 350°C and a concentration of 400 mg L^{-1}, with particle sizes of around 160 nm, exhibited a positive impact on enhancing the active immunity of *N. benthamiana* plants via the ethylene pathway. Consequently, this led to increased resistance against *P. nicotianae* infection. In the leaf assays conducted, the utilisation of nanoscale biochar led to a significant decrease of 9.26% in lesion sizes as compared to the control samples. The introduction of biochar nanoparticles resulted in a 20.3% rise in reactive oxygen species (ROS) through a mechanistic process, which in turn triggered the activation of the ethylene pathway. The activation event was subsequently accompanied by the manifestation of systemic acquired resistance (SAR), which was produced by the application of salicylic acid (SA). It is worth mentioning that the immune response caused by SA showed a significant rise of more than ten times in both treatments involving biochar nanoparticles. Furthermore, the biochar nanoparticles exhibited a much higher level of PR1-a expression compared to conventional biochar particles. In a conducted experiment, the utilisation of biochar nanoparticles as a delivery system for the plant elicitor 5-methoxyindole yielded notable enhancements in plant development. In particular, the use of biochar nanoparticles and 5-methoxyindole as seed treatments resulted in a significant enhancement of shoot length by more than 105.3%

and root length by 41.2% when compared to untreated control samples. This work highlights the effective application of exogenous nanoscale biochar amendment in stimulating plant defence responses and improving resistance against the prominent pathogen *P. nicotianae*. The aforementioned methodology exhibits significant potential as an innovative tactic for safeguarding plants in the realm of sustainable nano-enabled agriculture, as outlined by Kong et al. [39]. Biochar serves a multifaceted purpose beyond the mere sequestration of carbon in soil over an extended period. It possesses the capability to significantly enhance agricultural production. The white-backed plant hopper (WBPH) is well recognised as a significant agricultural pest due to its substantial consumption of rice, a staple crop that provides sustenance for a significant portion of the global population. The researchers initiated a scientific endeavour with two primary goals: Initially, embark on a quest to ascertain the optimal biochar formulation for two distinct rice cultivars, namely 'Cheongcheong' and 'Nagdong.' Furthermore, this study aimed to investigate the impact of several biochar blends on the growth and resistance of two rice cultivars, when exposed to the persistent threat of the white-backed planthopper (WBPH). Upon doing a comprehensive analysis of the ideal levels of biochar, the findings yielded remarkable insights. The use of biochar at a 10% ratio in the rooting media proved to be quite beneficial, as it substantially enhanced the physiological well-being of both rice cultivars. The introduction of these plants provided a significant boost in their vitality. Nevertheless, deviating excessively beyond this optimal range, such as by incorporating biochar levels of 1%, 2%, 3%, or 20%, resulted in adverse outcomes. The experience resembled indulging excessively in a desirable entity until its desirability diminished. Now, this is the point at which it becomes quite fascinating. The two rice cultivars, Cheongcheong and Nagdong, exhibited contrasting responses in the presence of the persistent white-backed planthopper (WBPH). Nagdong, when reinforced with biochar, exhibited robust growth and resilience in the presence of White-backed Planthopper (WBPH) infestation. In contrast, Cheongcheong exhibited a less favourable response when faced with comparable situations. The expansion of the entity was impeded, appearing to be overpowered by the very strength and adaptability it aimed to possess. It is noteworthy that the impact caused by WBPH was notably reduced in Nagdong and notably increased in Cheongcheong, in comparison to their respective untreated equivalents. The phenomenon of biochar's "priming effect" was shown to result in a significant increase in the production of jasmonic acid in response to the presence of white-backed planthopper (WBPH). In the context of Nagdong, this phenomenon can be interpreted as a catalyst for bolstering fortitude and facilitating personal development. However, in the region of Cheongcheong, there appeared to be an excessive abundance of favourable conditions, which resulted to an unfavourable reaction that hindered development and diminished the ability to withstand WBPH [40]. To determine how biochar amendments affected the English grain aphid *Sitobionavenae*'s ability to reproduce, more research was done. The study subsequently examined the expression of defence-related genes in wheat plants, specifically exploring the impact of biochar amendments and aphid feeding on this expression. When compared to the control group, the inclusion of biocahr modifications led to a decrease in aphid lifetime fertility by 9.09% and 20.23% for amendment levels of 3% and 5%, respectively. In addition, the utilisation of biocahr amendments resulted in a decrease in the aphid population by 18.68%, 21.69%, and 28.70% at amendment concentrations of 1.5%, 3%, and 5%, correspondingly. The silicon concentration in wheat plants increased significantly with the usage of biochar, surpassing a 40% elevation. Additionally, using biochar caused wheat plants to

activate four defence-related genes (AOS, LOX, PAL, and PR), extending the time that aphids may feed on the plant. Our study's findings indicate that adding biochar to soil may have a negative effect on aphids that infest wheat crops by reducing their ability to reproduce. The greater activation of plant defence mechanisms caused by applying biochar in response to aphid infestation is responsible for the reported negative effects [41].

3.2 Biochar effect on the soil microbial community

The planet's soil ecosystems are home to the widest variety of terrestrial communities, and soil microorganisms are largely responsible for this extraordinary diversity [42]. Therefore, it is imperative to understand how biochar affects the soil microbiota, as Ng and Cavagnaro [43] pointed out. Because of its porous structure, biochar has microsites that can host soil microorganisms and provide them with a fresh environment (**Figure 3**). Since little is known about how biochar affects different organisms selectively, its potential as a microbial habitat is yet unknown. Furthermore, it is unknown how other elements like food availability and predation affect microbial response to biochar [43]. It has been noted that adding biochar to several experiments causes a significant increase in microbial biomass. It has been observed that the microbial communities' composition and the activity of the enzymes are significantly altered by the ensuing increase in microbial biomass. A number of adjustments can be made to clarify the possible biogeochemical effects of biochar, including how it affects crop development, plant disease prevention, and nutrient cycling [27, 44]. One of the ways that biochar promotes microbial activity is through its porous nature. Furthermore, the abundance and availability of dangerous compounds are altered by biochar, changing abiotic variables like pH and giving some microbial communities an edge over others. Moreover, microbes can use biochar as a feasible energy source or as a way to obtain necessary mineral components [45]. There are several documented instances of interactions between soil, bacteria, and biochar that can have both positive and negative consequences. For instance, adding biochar made from wheat husks to temperate soils increased the diversity of microbes. Since the biochar's organic carbon utilisation and metabolic activity were determined to be negligible, the rise was mostly due to physicochemical factors [46]. In tomato plants, the introduction of biochar made from eucalyptus wood chips at a concentration of 1% (wt/wt) increased bacterial diversity and altered the plants' ability to metabolise nutrients. This was the conclusion of a study carried out by Kolton et al. [47]. Similar results were seen by Kolton et al. [26] when pepper plants were cultivated with citrus wood-derived biochar. By using biochar made from red spruce pellets and grapevine residues, Taskin et al. [48] report that the growth and enzyme activity of ligninolytic fungi living in the soil were enhanced. According to Wong et al. [49], the application of biochar derived from peanut shells and wheat straw to a recently constructed landfill cover topsoil resulted in an augmentation of soil bacterial community diversity. In contrast, certain biochar variants that had elevated levels of phosphorus, such as those derived from chicken sources, exhibited a notable decrease in the colonisation of roots by mycorrhizal fungi. However, it is worth noting that this reduction did not have any discernible impact on crop productivity, as indicated by Solaiman et al. [50]. Furthermore, the utilisation of rice straw-derived biochar was observed to elicit detrimental consequences on the model organism *Caenorhabditis elegans*. This was evident through the manifestation of neurotoxic attributes, which can be attributed to the activation of oxidative stress responses within the nervous system and the

identification of free radicals within the biochar [51]. An further advantageous consequence of biochar implementation is its capacity to mitigate the environmental repercussions associated with metal(loid) pollutants through interactions with microorganisms. The study conducted by Wang et al. [52] demonstrated that the utilisation of bamboo-derived biochar in soils contaminated with cadmium (Cd) resulted in a notable reduction in the uptake and accumulation of Cd in rice plants. This effect was attributed to the biochar's ability to modify the composition of the bacterial population in the rhizosphere. In a study conducted by Meng et al. [53], it was shown that the application of biochar derived from wheat straw in soils contaminated with herbicides resulted in an increase in microbial diversity and a subsequent improvement in the performance of wheat plants.

3.3 The use of biochar against plant pathogens

Research findings have provided intriguing revelations concerning the influence of biochar on plant diseases. According to Frenkel et al. [54], it has been observed that lower concentrations (≤1%) of biochar have the ability to suppress a range of disorders. Conversely, greater concentrations (>3%) tend to be ineffective, leading to a dose-response pattern that resembles an inverted U-shaped curve. In order to maintain a consistent and replicable impact of biochar on agricultural practises, it is advisable for biochar manufacturers to establish uniformity in the selection of feedstocks and concentrations, while also taking into account the potential implications on plant diseases [55]. The significance of this matter lies in its relevance, as the varied source and treatment of raw materials utilised in the production of biochar can result in variable outcomes with regards to disease suppression in agricultural systems (**Figure 3**). Biochar utilises various mechanisms to safeguard plants against diseases. These mechanisms encompass the facilitation of plant growth through nutrient provision, the augmentation of soil-microbial diversity, the adsorption of toxins generated by pathogens (such as extracellular enzymes and organic acids), the stimulation of antibiotic or fungitoxic compound production, the modification of root exudate chemistry, and the initiation of systemic plant defence mechanisms via chemical compounds acting as elicitors or microorganisms residing in microhabitats [56, 57]. Biochar has demonstrated efficient utilisation in combatting a diverse array of plant diseases, including those present in the air or soil, as well as many types of pests. Moreover, the research conducted by Lou et al. [58] has validated the growth-enhancing characteristics of biochar water-wash extracts, which contain a substantial amount of organic and inorganic chemicals. These findings indicate the need for additional investigation in this area. It is noteworthy that the advantageous impacts of biochar are occasionally more closely associated with its ability to enhance plant development rather than eliciting plant defence mechanisms. An experiment was conducted to investigate the effects of applying poplar woodchip biochar on the growth of *Arabidopsis thaliana*, a model plant, and lettuce, a crop plant. The results showed a notable enhancement in plant growth for both species, accompanied by the activation of genes associated with growth-promoting hormones such as auxins and brassinosteroids. These hormones are believed to play a role in mediating the observed growth effects. Nevertheless, the utilisation of global gene expression arrays and metabolomics techniques unveiled a significant decrease in the expression of multiple plant defence genes. These genes encompass those responsible for the synthesis of jasmonic acid, as well as genes encoding defensins and various classes of secondary metabolites that play a crucial role in safeguarding plants against insects and pathogens [59].

3.4 Biochar for the control of plant-pathogenic bacteria

The primary focus of utilising biochar as a mitigation technique for plant diseases caused by bacterial pathogens has been on tackling the specific issue of bacterial wilt disease, predominantly attributed to the pathogen *Ralstonia solanacearum*. The illness in question presents a substantial obstacle to the cultivation of vegetables on a worldwide level [60]. In light of the restricted accessibility of data, multiple sources were employed in the production of biochars with the aim of addressing bacterial concerns. The biochars utilised in this study were applied at a consistent rate of 2–3% (weight/weight). The results of the study indicated that these biochars were effective in reducing the incidence of bacterial wilt in tomato and tobacco crops. The protective effect of biochar is ascribed to multiple mechanisms, including the improvement of soil physicochemical qualities and the proliferation of bacteria and actinomycetes in the rhizosphere. The studies [61–63] also provide evidence for the combined impact of these processes on the decrease in swarming motility and root colonisation capacity of *Ralstonia solanacearum*.

3.5 Biochar for the control of plant-pathogenic fungi

Fungi are widely recognised as a prominent and highly deleterious category of plant pathogens [64], hence presenting a substantial agricultural risk. Soilborne infections, caused by the presence of *Fusarium* species, exert a substantial influence on a wide range of crops in different climatic zones [12]. As per the findings of Summerell [65], it has been established that Fusarium has the ability to produce mycotoxins in conserved plant-based goods. The aetiology of crown and root rot disease in asparagus can be ascribed to two fungal infections, specifically *Fusarium oxysporum f. sp. asparagi* and *Fusarium proliferatum*. Furthermore, the gravity of this ailment is augmented by the emission of allelopathic pollutants into the soil. The application of hardwood biochar resulted in a significant increase in the populations of antagonistic organisms, including Pseudomonas and arbuscular mycorrhizal fungi (AMFs) [66]. Furthermore, the latter have been recognised as agents that induce systemic resistance [12]. Additional research is necessary in order to have a comprehensive understanding of the fundamental mechanisms at play. Nevertheless, the results strongly suggest that biochars exert a substantial influence on nutrient availability and soil characteristics, as well as on the stimulation of soil microbial communities [66, 67]. The study conducted by Eo et al. [68] yielded similar results when investigating Panax ginseng plants cultivated on soil that was enriched with rice husk biochar. Similar effects were observed against *F. solani* and *Ilyonectriadestructans*, and these effects were ascribed to changes in the rhizosphere-microbial community. In a similar vein, the application of green waste biochar, along with compost, has been observed to yield advantageous effects in the reduction of *F. oxysporum f. sp. lycopersici* wilt. The aforementioned phenomenon was observed throughout a spectrum of intensities, ranging from 0% to 3% (w/w).According to Akhter et al. [69], the introduction of chlamydospores in tomato plants led to a notable augmentation in advantageous bacteria. These bacteria can exercise their influence through two mechanisms: direct antagonism or indirect activation of systemic resistance in the plant. The potential beneficial effects of biochar produced from beech wood chips or garden waste residues on the inhibition of *F. oxysporum f. sp. lycopersici* in tomato plants are likely due to the alteration of plant root exudates. These exudates may have a significant impact on the plant's response to stress caused by disease [70]. Furthermore, the use

of biochar alongside gaseous pesticides, which are generally referred to as fumigants, has promise for efficiently reducing fumigant emissions in the agricultural industry. The usage of two different types of biochar, obtained from a blend of hardwoods and *Eupatorium adenophorum*, in combination with dimethyl disulfide, is exemplified by an illustration. In the study conducted by Wang et al. [71], it was observed that the application of these chemicals at rates lower than 0.5% (by weight) did not have any detrimental effects on the control of *Fusarium spp.* Nevertheless, a significant reduction in the emission of dimethyl disulfide into the surrounding ecosystem was observed. The necrotrophic fungus *Rhizoctonia solani* is a soilborne pathogen that holds significant economic importance due to its ability to cause seedling illnesses in crop plants [12]. While the specific processes underlying the suppression of *R. solani* by biochar remain unclear, it has been determined that direct toxicity is not the primary factor. The utilisation of a 1% (wt/wt) biochar derived from eucalyptus wood and greenhouse wastes shown a notable decrease in disease occurrence among cucumber and bean plants, as observed in previous studies conducted by Jaiswal et al. [72, 73]. On the other hand, it was observed that the use of biochar derived from maple bark resulted in an increase in the incidence of *R. solani* damping-off disease across many plant species. This effect can be attributed to the existence of various organic chemicals within the biochar, which are believed to augment the pathogen's fundamental metabolic processes [74]. Hence, it should be noted that not all biochar sources have a favourable capacity for disease suppression.

Furthermore, biochar has been found to enhance the ability of plants to fight foliar plant pathogenic fungus. The primary mechanism observed for the phenomenon of induced resistance is the systemic activation of defence mechanisms, as described by Shirai and Eulgem [75]. *Botrytis cinerea* is a fungal pathogen with a broad host range, capable of inducing necrotic disease on numerous plant species spanning various taxonomic groups [76]. Grey mould, a commercially detrimental disease, is caused by *B. cinerea* in strawberries [76]. The application of biochar derived from holm oak in strawberry fields has been found to enhance the bacterial variety within the rhizosphere, as demonstrated by De Tender et al. [77]. This increased bacterial diversity has the potential to induce systemic resistance against pathogens, hence mitigating the incidence of diseases. Kolton et al. [47] reported comparable outcomes in the suppression of *B. cinerea* when biochar derived from eucalyptus wood chips was administered to tomato plants. This application resulted in heightened microbial diversity and metabolic activity in the rhizosphere, consequently bolstering the defence mechanisms of the tomato plants. In a study conducted by Poveda et al. [12], it was shown that the application of biochar derived from greenhouse waste on tomato plants resulted in the activation of both early- and late-acting defence mechanisms against *B. cinerea*. Several factors were observed in this study, including the upregulation of genes associated with jasmonic acid and ethylene responses, as well as a significant increase in the presence of active oxygen species like H_2O_2. These findings are of great importance in understanding the mechanisms behind resistance to *B. cinerea*, as reported by De Tender et al. [77]. In a study conducted by Al-Juboory et al. [78], it was observed that the application of biochar derived from citrus wood at concentrations ranging from 1% to 5% exhibited significant reduction in the incidence of two plant diseases, namely *B. cinerea* and *Leveillulataurica*, which is responsible for powdery mildew. This reduction in disease occurrence is likely attributed to the activation of systemic induced resistance elicitors. Consistent with this, the utilisation of biochar derived from pepper plant wastes demonstrated efficacy in combating various fungal pathogens, including

Colletotrichum acutatum and *Podosphaeraaphanis* [21] that are responsible for causing anthracnose and powdery mildew in strawberries, respectively.

3.6 Biochar for the control of plant-pathogenic oomycetes

Oomycetes exhibit characteristics that facilitate their effective infection and subsequent mortality of several plant species, including those of considerable economic importance as food and cash crops [79]. In parallel to the investigation of plant-pathogenic bacteria, there exists a restricted corpus of scholarly inquiry pertaining to the use of biochar for the purpose of oomycete management. The utilisation of biochar as a means of addressing plant disease caused by Phytophthora has been mostly focused on species that infect trees. Previous studies have shown evidence that the application of biochar generated from pine plant tissues at a concentration of 5% (vol/vol) can enhance the activation of plant defence systems through the induction of systemic resistance. The phenomenon described has been documented in *Quercus rubra* plants subjected to *Phytophthora cinnamomi*, as well as in *Acer rubrum* plants subjected to *Phytophthora cactorum*. As a result, the utilisation of biochar has been discovered to effectively reduce the advancement of diseases and alleviate physiological strain in these particular plant species [20]. In contrast, the utilisation of softwood biochar led to an increase in the colonisation of *Pythium ultimum* in the roots of sweet pepper, lettuce, basil, and geranium plants. Nevertheless, the investigation carried out by Gravel et al. did not reveal any observable adverse impacts on the root system or general growth of the plants [80].

3.7 Biochar for the control of plant-parasitic nematodes

Singh et al. [81] have reported that the occurrence of plant-parasitic nematodes (PPNs) has been linked to an average decline in agricultural productivity by roughly 12.3%. Biochar is widely recognised as an ecologically sustainable strategy for mitigating the impact of plant-parasitic nematodes (PPNs) by employing various techniques. For instance, modifications in the biodiversity of nematode populations residing in the soil have exhibited effectiveness in mitigating the impact of plant-parasitic nematodes (PPNs). The effect of incorporating biochar produced from wheat straw on the variety of soil nematodes was assessed in a microcosm experiment. According to Zhang et al. [82], the phenomenon resulted in a rise in the population of fungivorous nematodes and a decline in plant-parasitic nematodes (PPNs) from different genera such as *Coslenchus*, *Hirschmanniella*, *Rotylenchus*, and *Tylenchus*. The effectiveness of utilising biochar produced from burned log wood has been demonstrated in reducing the populations of *Pratylenchuscoffeae*, a migratory endoparasitic nematode that inflicts substantial harm to banana roots [83]. The decrease in nematode populations was seen at a concentration of 4% (wt/wt) [84]. The utilisation of biochar derived from poultry litter in the cultivation of grapevines led to a significant reduction in soil populations of plant-parasitic nematodes (PPNs), such as *Meloidogyne javanica*, *Tylenchulus semipenetrans*, *Pratylenchus spp., Helicotylenchus spp.,* and *Criconemoid spp*. The decrease in plant growth can be ascribed to the increased abundance of several species that have been found to be advantageous for plant development, as documented by Rahman et al. [85]. The activation of plant defence mechanisms is an effective strategy utilised to combat *Pratylenchus spp.* As an example, the application of biochar produced from coniferous wood and spelt husks at a concentration of 5% (volume/volume) led to an enhanced resistance to

Pratylenchus penetrans in carrot plants. The increased resistance seen can be ascribed to the activation of plant defence mechanisms. Nevertheless, it is crucial to acknowledge that the potential mechanism of direct toxicity or alteration of soil pH should not be overlooked [86]. Furthermore, the researchers successfully controlled the presence of the endoparasitic root-knot nematode *Meloidogyne graminicola* in rice agrosystems by introducing biochar generated from oak wood. This intervention involved the application of biochar at a concentration of 1.2% (wt/vol), as reported by Mondal et al. [87]. According to Huang et al. [88], it was discovered that the observed phenomena was linked to a concentrated accumulation of H_2O_2 in a specific area, as well as an upregulation of genes related to ethylene. This led to the alteration of plant defence genes. One intriguing application of biochar involves its effectiveness in reducing fumigant emissions while simultaneously controlling nematodes. In a study conducted by Cheng et al. [89], it was demonstrated that the use of a combination of 1,3-dichloropropene and chloropicrin, along with biochar derived from coconut shell, exhibited a sustained ability to manage plant-parasitic nematodes (PPNs) from various genera including *Pratylenchus*, *Meloidogyne*, *Tylenchorhynchus*, *Tylenchidae*, *Trichodorus*, and *Mesocriconema*. Moreover, this methodology additionally led to a decrease in the emission of detrimental atmospheric pollutants. Another example, as mentioned earlier, of managing *Fusarium spp*. is the application of biochar produced from hardwood and *Eupatorium adenophorum*. The aforementioned method has exhibited effectiveness in managing root-knot nematodes when administered at a concentration of 0.5% in relation to the soils weight. In addition, it is worth noting that the utilisation of this application rate has led to a significant reduction in fumigant emissions [90].

4. Conclusion

Knowledge of the role of biochar in plant protection can be highly beneficial in the context of advanced pest management for several reasons:

Increased availability of nutrients: enhancing the fertility and availability of nutrients in the soil through the use of biochar can lead to healthier plant development. Generally speaking, robust plants are better equipped to resist against pests and illnesses. You can contribute to the creation of an environment that is less favourable to pest infestations by implementing biochar into soil management techniques.

Advanced biological control: beneficial microorganisms in the soil, such particular bacteria and mycorrhizal fungi, can flourish when supported by biochar. Pathogenic organisms can be inhibited, consumed, or engaged in competition with these bacteria. Biochar aids in pest management indirectly by promoting a robust microbial population in the soil.

Reduction in chemical use: pesticides and fertilisers can be absorbed and retained by biochar, preventing them from leaking into the environment. This may result in less chemical pesticide being used, which is advantageous for the environment and may also help stop the emergence of pests that are resistant to pesticides.

Modification of pest behaviour: according to specific research, biochar may have an impact on how certain pests behave. Pests may find it more difficult to discover and infest crops if it interferes with the chemical cues that they utilise to locate their host plants.

Increased plant immunity: plants that have been exposed to biochar have been shown to develop systemic acquired resistance (SAR). Plants can repel a wide variety

of diseases thanks to a defence system called SAR. Plant resistance to pests and diseases may rise as a result.

Sustainable agriculture: eco-friendly and sustainable methods are critical in this day of sophisticated pest control. By enhancing soil quality, lowering the demand for chemical inputs, and lessening the detrimental effects of agriculture on the environment, biochar can contribute to sustainable agriculture. In addition, insect patterns are changing due to climate change, and crop pressure is rising. By boosting their general health and stress tolerance, biochar can help plants become more resilient to these shifting environmental circumstances.

Thus, knowing how biochar protects plants is important for advanced pest management because it provides a comprehensive strategy that enhances soil health, plant vitality, sustainability, and direct insect control. It is feasible to lessen the need for chemical pesticides, support healthier ecosystems, and improve agricultural systems' overall resistance to pests and other environmental problems by using biochar into pest management techniques.

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Chapter 2

Influence of Biochar on Soil Insect Dynamics and Infestation

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Abstract

Biochar, a carbonaceous material produced through pyrolysis of organic matter, has garnered attention for its potential to enhance soil fertility, structure, and overall health. However, its effects on soil-dwelling insects remain a subject of considerable interest and debate. This chapter critically examines the current state of knowledge regarding the interactions between biochar applications and soil-dwelling insects, encompassing diverse aspects such as alterations in insect community composition, population dynamics, behavioral changes, and potential mitigation of infestations. Through an exploration of empirical studies and theoretical frameworks, it aims to elucidate the intricate relationships between biochar amendments and soil insect ecology.

Keywords: biochar, soil insects, infestation, soil ecology, pyrolysis, soil health

1. Introduction

Soil health stands as the bedrock of terrestrial ecosystems, serving as a vital foundation for agricultural productivity, ecological stability, and the sustenance of diverse life forms [1]. Within this intricate web of soil dynamics, the role of soil-dwelling insects emerges as an integral component, orchestrating a delicate balance in the ecosystem [2–4]. In recent years, the integration of biochar as a soil amendment has garnered considerable attention for its potential to revolutionize soil health and agricultural sustainability [5, 6]. Biochar, derived from the pyrolysis of organic materials under controlled conditions, manifests as a stable carbonaceous product [7]. Its application as a soil amendment dates back centuries, with historical practices like terra preta showcasing its enduring benefits in enhancing soil fertility and structure [8–10]. The porous nature of biochar contributes to its exceptional capacity for water retention, nutrient adsorption, and microbial habitat provision [11]. Its introduction to soil systems holds promise not only for improving agricultural yields but also for mitigating environmental degradation by sequestering carbon [12]. Soil health constitutes a complex interplay of physical, chemical, and biological attributes that sustain life below ground [13, 14]. Soil-dwelling insects, encompassing a diverse

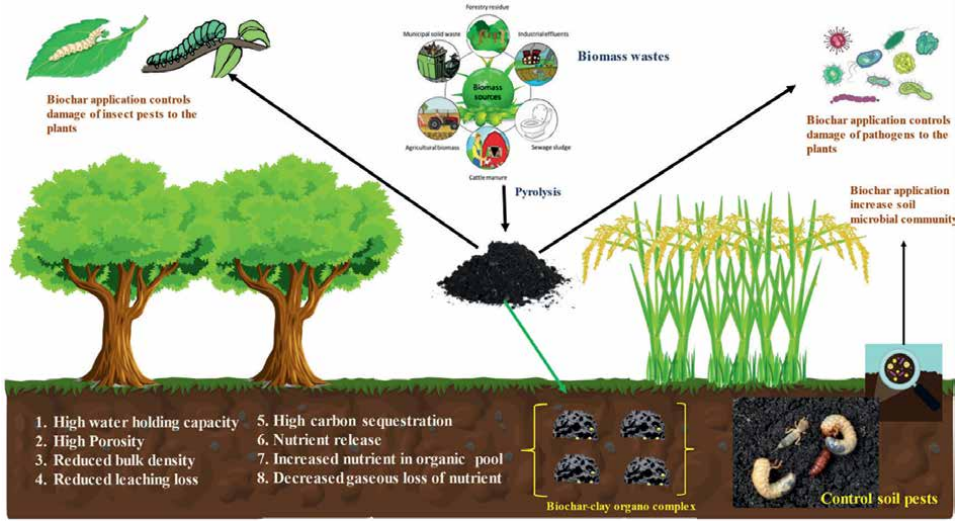

Figure 1.
Depicting biochar in plant-biotic interactions like soil insect pest management where the biomass waste by pyrolysis gets converted to biochar-clay organo complex in the soil resulting in controlling soil pests.

array of species, play pivotal roles in nutrient cycling, decomposition, soil aeration, and pest regulation [15]. Their interactions within the soil ecosystem contribute significantly to the overall health and functionality of terrestrial environments [16]. However, disturbances in soil health, induced by factors such as land-use changes, pollution, and climate variations, can disrupt the delicate equilibrium of soil-dwelling insect populations, potentially leading to ecological imbalances and decreased agricultural productivity [17–22]. This chapter aims to delve into the intricate nexus between biochar applications, soil health, and soil-dwelling insects. It seeks to comprehensively explore the influence of biochar on soil insect dynamics, community structure, and infestation patterns within soil ecosystems. Through a synthesis of empirical studies, theoretical frameworks, and practical insights, this chapter endeavors to unravel the multifaceted effects of biochar as a soil amendment on the intricate relationships between soil health and the myriad inhabitants dwelling beneath the soil surface (**Figure 1**).

2. Soil insect pests

Insect pests are almost a year-round problem in the whole universe. Pests can be above ground or below ground [23]. While it is somewhat easier to observe foliar insects, it is a daunting task to really know what is inside the soil before it is too late. Various soil insects such as wireworms, cutworms and white grubs among many others are regular occurrence in various fields [24]. It is always very important to take a shovel sample of soil in raised beds or open fields to retrieve sample of insect pests that may be infesting the soil. Soil provides a structure for a plant to anchor its roots and is a source of nutrition and water necessary for plant growth. Soil-inhabiting insects also utilize this substrate for part of or for their entire life. Although many insects are dependent on soil for food and shelter, only a few soil-borne insects such as weevils, ants, and termites are detrimental to the citrus tree.

3. Types of soil insect pests

Wireworms: These are the immature stage (larva) of the click beetle. Wireworms are cylindrical, about 1–1/2 inches long, brownish to yellow, and are rather hard-bodied. These insects eat seeds, cut into small shoots, and often bore into stems, roots, and tubers. They attack many vegetables including potatoes, onion, corn, carrots, peas, beans, and melons.

White grubs: These are cream-colored, C-shaped larvae with brown heads. They include the immature stage of European chafer, Japanese beetle, and May/June beetles. They stay in the soil and feed on the roots of corn, beans, peas, and other vegetables. They are most likely to damage plants in or near ground that was recently sod covered.

Cutworms (*Agrotis ipsilon*): These are the larval, or immature, stage of certain moths. They can often destroy a stand of plants in a garden. Cutworms are night feeders and are seldom seen during the day. These insects cut off small plants at or near the ground level and feed on the tender stem. Some types climb up the stem and feed on foliage. Many plants are attacked by cutworms, but they are especially damaging to corn, beans, tomatoes and peppers.

Seed corn maggots (*Delia platura*): These are the larvae of small flies. They develop in the soil and feed on seed and seedlings of corn, beans, peas, potatoes, cabbage, melons, and other crops. Cool wet springs and soil with a high concentration of organic matter favor the development of this pest [25].

Termites: Termites, often referred to as "silent destroyers," wield their devastating impact beneath the surface, wreaking havoc on soil health and structures. Their presence manifests through several damaging symptoms within the soil. One prominent sign of termite infestation is the formation of intricate tunnels and galleries snaking through the earth. These subterranean passages, constructed by termites as they forage for food and establish their colonies, weaken the soil structure, compromising its stability. Additionally, the accumulation of termite excrement, commonly known as "termite mounds," alters the soil composition, leading to nutrient imbalances and decreased fertility. As termites consume organic matter within the soil, they accelerate its degradation, impacting the natural processes that sustain its vitality. The cumulative effect of their activity disrupts the delicate balance of the ecosystem beneath the surface, posing a considerable threat to agricultural productivity and ecosystem health.

4. Definition and composition of biochar

Biochar, a carbon-rich, porous material derived from the thermochemical conversion of biomass under controlled conditions of low oxygen, holds immense promise as a versatile soil amendment. Its composition, production methods, and diverse applications across agricultural and ecological landscapes underline its potential in revolutionizing soil health and sustainability [26]. Biochar is characterized by its stable carbon structure, resulting from the pyrolysis or thermal decomposition of organic materials such as wood chips, agricultural residues, or other biomass. This process occurs in the absence or limited presence of oxygen, preventing complete combustion and yielding a carbonaceous residue. The resulting biochar typically contains varying percentages of carbon (ranging from 60% to 95%), along with hydrogen, oxygen, and small quantities of nitrogen and other minerals. Its high surface area, porosity, and stable carbon content distinguish biochar from other organic amendments, rendering it resistant to degradation and offering long-term benefits to soil systems.

5. Production methods and variations

Biochar production methods encompass a spectrum of techniques, each influencing the characteristics and properties of the final product. Traditional methods include pyrolysis in kilns, retorts, or pits, where organic materials undergo thermal decomposition at temperatures typically ranging from 300°C to 1000°C. Variations in pyrolysis conditions, such as heating rates, residence time, and temperature, significantly impact the physical and chemical properties of biochar [27]. Modern approaches, like gasification and hydrothermal carbonization, offer more controlled processes, allowing for the customization of biochar properties by adjusting operating parameters. Different feedstocks and production techniques yield biochars with diverse pore structures, surface areas, elemental compositions, and functionalities. For instance, pyrolyzing feedstocks like woody materials tend to produce biochars with higher carbon content and greater stability, while agricultural residues might result in biochars with varying nutrient contents and properties. These variations underscore the importance of tailoring biochar production to specific applications and desired soil outcomes [28].

6. Application techniques and considerations in agricultural and ecological settings

Biochar's versatility in application spans agricultural and ecological domains, offering a spectrum of benefits. In agricultural settings, incorporating biochar into soil enhances its physical structure, water retention capacity, and nutrient-holding capabilities [29]. Various application methods exist, including surface application, incorporation into soil via tilling or mixing, or integration into composts or organic fertilizers. The choice of application method depends on factors such as soil type, crop type, climate, and desired outcomes. For instance, mixing biochar into soils during land preparation facilitates its distribution and integration, while surface application suits established crops or erosion-prone areas. Ecologically, biochar finds use in habitat restoration, carbon sequestration, and remediation of contaminated soils [30]. Its porous structure serves as a refuge and substrate for beneficial microorganisms, promoting soil biodiversity and ecosystem resilience. Additionally, biochar's ability to sequester carbon aids in mitigating climate change by locking away carbon dioxide in stable forms, contributing to carbon-negative strategies. Considerations in biochar application involve dosage, feedstock selection, soil interactions, and potential impacts on soil pH and nutrient availability. Determining optimal application rates, accounting for regional variations, and monitoring long-term effects are crucial for maximizing biochar's benefits while minimizing any unintended consequences [31].

7. Soil insect dynamics and ecology

The complex world beneath our feet teems with an intricate tapestry of soil-dwelling insects, a diverse array of organisms that play pivotal roles in shaping soil ecosystems and influencing the health of terrestrial environments. These often-overlooked inhabitants encompass a broad spectrum of taxa, including beetles, ants, earthworms, mites, springtails, nematodes, and many others, each contributing distinctively to the intricate web of interactions within the soil matrix [32]. Their

presence and activities are integral to the functioning of soil ecosystems, orchestrating a myriad of ecological roles that profoundly influence nutrient cycling, decomposition processes, soil structure, and the broader dynamics of terrestrial environments. The diversity of soil-dwelling insects within these ecosystems is staggering, with an estimated one-quarter of all described animal species inhabiting soil habitats. This diverse assemblage fulfills an array of ecological roles, contributing significantly to nutrient cycling through their roles as decomposers, predators, herbivores, and symbiotic partners [33]. For instance, earthworms, among the most recognizable soil inhabitants, aid in organic matter breakdown and soil aeration, enhancing nutrient availability and soil structure. Ants and termites are key ecosystem engineers, influencing soil architecture and nutrient distribution through their burrowing and foraging activities. Meanwhile, predatory insects like ground beetles and rove beetles help regulate pest populations by preying upon other soil-dwelling organisms, thus exerting top-down control within these ecosystems. A multitude of factors influences the population dynamics and distribution patterns of soil insects. Abiotic factors such as soil moisture, temperature, pH, and texture, alongside biotic factors including plant root exudates, microbial communities, and interspecific interactions, collectively shape the abundance and diversity of soil-dwelling insects [34]. Soil physicochemical properties profoundly impact insect survival, reproduction, and movement, creating microhabitats that favor certain species over others. Additionally, the intricate relationships between soil-dwelling insects and aboveground organisms, such as plants and predators, contribute significantly to their population dynamics [35]. For instance, plant root exudates not only provide a food source for soil insects but also influence the composition and activity of soil microbial communities, indirectly affecting insect populations through complex trophic interactions. Interactions within soil ecosystems are characterized by intricate food webs and symbiotic relationships that govern energy flows and nutrient cycling [36]. Soil insects form intricate networks of interactions with microorganisms, plants, and other soil fauna, exerting cascading effects on ecosystem processes. For example, mycorrhizal associations between fungi and plant roots influence soil insect dynamics by altering resource availability and quality. Predatory interactions among soil insects regulate populations of other soil organisms, thereby influencing decomposition rates and nutrient cycling. Furthermore, soil insects play critical roles in the breakdown of organic matter, facilitating nutrient release and recycling within the ecosystem. Understanding the complexity of soil insect dynamics and their ecological roles is crucial for deciphering the intricate functioning of soil ecosystems [37]. These often inconspicuous organisms wield immense influence over soil health, nutrient cycling, and ecosystem stability. Thus, elucidating the factors shaping their populations and unraveling the web of interactions within soil ecosystems is pivotal for devising sustainable land management strategies, preserving biodiversity, and ensuring the resilience of terrestrial ecosystems in the face of global environmental changes.

8. Impact of biochar on soil insect community composition

8.1 Influence of biochar on soil insect populations

The impact of biochar on soil insect community composition represents a critical aspect of understanding how this soil amendment influences the intricate balance and dynamics within terrestrial ecosystems [38]. Soil-dwelling insects, comprising a

diverse range of taxa, play integral roles in nutrient cycling, decomposition, and ecosystem functioning. Biochar application can alter soil properties and, consequently, affect the abundance, diversity, and interactions among these vital organisms [37].

8.2 Response of different taxa

Biochar application exhibits variable effects on different soil-dwelling insect taxa. Studies across diverse ecosystems have demonstrated contrasting responses among various insect groups post-biochar application [39]. For instance, while some investigations report increases in the abundance of certain decomposer organisms such as earthworms and springtails, others document shifts in the relative abundance of predatory insects like beetles or ants. These responses often vary depending on biochar type, application rates, soil characteristics, and specific insect taxa [40]. The varied responses among soil insect groups highlight the complexity of biochar-soil interactions and the nuanced effects on different trophic levels within soil ecosystems. The altered conditions resulting from biochar application can influence resource availability, microhabitat preferences, and interactions among soil-dwelling insects, contributing to the observed changes in their abundance and distribution.

8.3 Altered relative abundance

Observations post-biochar application often reveal changes in the relative abundance of soil insect species [41]. Some studies indicate shifts in the dominance or proportional representation of specific insect taxa within soil communities. For instance, while certain studies report an increase in the abundance of earthworms or springtails following biochar incorporation, others note decreases in the abundance of certain beetle species or mites. These alterations in the relative abundance of soil-dwelling insects can be attributed to biochar-induced modifications in soil properties, such as changes in pH, nutrient availability, or microbial activity [42]. Such shifts might impact the competitive advantage of certain insect groups, influencing their population sizes within the soil ecosystem.

8.4 Shifts in diversity

Biochar application can elicit changes in the diversity indices and species richness of soil insect communities. While some studies report an increase in species richness and diversity metrics, others indicate no significant changes or even reductions in diversity following biochar incorporation [43]. The observed shifts in diversity indices might result from alterations in soil conditions that selectively favor certain insect taxa or impact their interactions within the ecosystem. Increased diversity might stem from improved habitat conditions, such as enhanced soil structure or nutrient availability, fostering a more favorable environment for a wider range of soil insects. Conversely, reductions in diversity might arise due to changes in competitive interactions, resource availability, or alterations in microenvironmental conditions induced by biochar amendments. Understanding these shifts in species richness and diversity indices is crucial in deciphering the ecological implications of biochar application on soil insect communities [44]. It underscores the intricate nature of biochar-soil insect interactions and the need for comprehensive assessments considering various environmental factors influencing community composition.

8.5 Modulation of population dynamics

The modulation of population dynamics among soil-dwelling insects in response to biochar application represents a crucial aspect of understanding how this soil amendment shapes the functioning and interactions within terrestrial ecosystems. Biochar, with its diverse physicochemical properties, can exert influences on the behavior, activities, and population sizes of various insect groups, impacting key ecological roles such as nutrient cycling, decomposition, and soil engineering [45].

8.6 Activity levels and foraging behavior

Biochar application can elicit alterations in the activity levels, movement patterns, and foraging behavior of soil-dwelling insects. Studies have documented changes in the behavior of various insect taxa following biochar incorporation into soils. For instance, increased activity levels and alterations in movement patterns among decomposer organisms like earthworms or springtails have been observed [46]. These changes are often linked to biochar-induced modifications in soil properties, such as enhanced soil moisture retention, improved nutrient availability, or changes in microhabitat conditions. Moreover, alterations in foraging behavior, feeding patterns, and resource utilization by soil insects post-biochar application have been noted. Certain insect groups might exhibit preferences for biochar-amended zones within the soil, influencing their feeding habits or colonization patterns. Such changes in behavior can potentially impact nutrient distribution, organic matter decomposition rates, and soil structure, thereby influencing ecosystem functioning [47].

8.7 Decomposer dynamics

Biochar amendments have demonstrated effects on decomposer populations, influencing their abundance and activity within soil ecosystems. Decomposers, such as earthworms, springtails, and other microarthropods, play pivotal roles in organic matter breakdown, nutrient mineralization, and soil organic carbon dynamics [48]. Studies suggest that biochar can enhance the abundance and activity of certain decomposer organisms, potentially accelerating organic matter decomposition rates and nutrient cycling processes. Increased populations or activities of decomposer organisms following biochar application can contribute to enhanced nutrient availability in soils. Through their roles in fragmenting and processing organic materials, these decomposers facilitate the release of nutrients essential for plant growth and ecosystem functioning. Biochar-induced improvements in soil conditions, such as increased microbial activity or changes in substrate quality, might favor decomposer communities, subsequently influencing nutrient turnover rates and soil fertility [49].

8.8 Engineering species response

Soil engineers, including earthworms, ants, termites, and other burrowing organisms, can respond to biochar applications by altering their abundance and activities within soil ecosystems. These organisms significantly influence soil structure, aeration, water infiltration, and nutrient distribution. Studies have indicated changes in the abundance or behavior of soil engineers post-biochar incorporation,

impacting soil physical properties and ecosystem processes [50]. For instance, increased earthworm activity following biochar application can contribute to improved soil structure and nutrient availability by enhancing soil aggregation and organic matter decomposition. Changes in the abundance of ants or termites might influence soil architecture through burrowing activities, affecting water infiltration and nutrient distribution patterns within the soil profile. Understanding the responses of soil engineers to biochar amendments is essential as their activities play critical roles in shaping soil properties and ecosystem functions. These changes in population dynamics and activities among soil engineers reflect the broader implications of biochar applications on soil structure, nutrient cycling, and ecosystem resilience [51].

8.9 Behavior modifications and trophic interactions

Behavior modifications and trophic interactions among soil-dwelling insects in response to biochar amendments represent critical aspects of understanding the nuanced influences on insect behavior, ecological interactions, and soil ecosystem functioning. Biochar-induced alterations in soil properties can intricately shape the foraging behavior, trophic relationships, and indirect effects on soil insects mediated through changes in soil microbial communities.

8.10 Foraging behavior changes

Biochar application can induce modifications in the foraging behavior and feeding patterns of soil-dwelling insects. Changes in soil properties resulting from biochar amendments, such as increased water retention, alterations in nutrient availability, or modifications in soil structure, can influence the resource utilization and feeding preferences of various insect taxa. Studies have documented shifts in the foraging behavior of soil insects, including alterations in their preferences for specific food sources or changes in feeding rates [52]. For instance, certain decomposer organisms might exhibit preferences for biochar-amended zones within the soil, potentially altering their feeding habits or substrate selection. Additionally, biochar-induced changes in nutrient availability or modifications in the quality of organic matter might influence the nutritional value of food resources for soil insects, thereby impacting their feeding behavior and resource utilization strategies [53].

8.11 Trophic interactions

Biochar amendments can influence predator-prey relationships and trophic interactions within soil ecosystems. Changes in the abundance, behavior, or distribution of predator and prey species in response to biochar application can significantly impact trophic dynamics and population regulation within soil insect communities. Studies suggest that biochar-induced alterations in soil conditions can affect the abundance or behavior of predatory insects or other predators within soil ecosystems. Changes in predator abundance or activity might influence the population sizes of prey species, potentially affecting community structure and trophic cascades. Additionally, alterations in prey populations due to biochar-induced modifications in their habitat or resources can influence the dynamics of predator-prey interactions, subsequently impacting the structure and stability of soil insect communities [54].

8.12 Microbial-driven effects

Biochar amendments can indirectly influence soil-dwelling insects through their effects on soil microbial communities. Biochar-induced modifications in soil properties, such as alterations in pH, nutrient availability, or microbial activity, can influence the composition and activity of soil microbial communities, subsequently impacting soil insects. Changes in microbial communities mediated by biochar can influence the availability of food resources, decomposition rates, or soil nutrient dynamics, indirectly affecting soil insect populations. For instance, alterations in microbial-driven processes, such as decomposition rates or organic matter breakdown, can influence the availability of substrates for soil insects, consequently shaping their abundance, behavior, or community composition [55].

9. Ecological interactions and feedback mechanisms

Ecological interactions and feedback mechanisms within soil ecosystems following biochar amendments represent a complex interplay between soil nutrient cycling, microbial dynamics, plant-insect interactions, and the broader soil food web. The application of biochar as a soil amendment exerts multifaceted influences on soil properties, subsequently impacting nutrient dynamics, microbial communities, and interactions among soil-dwelling organisms [56].

10. Biochar's impact on soil nutrient dynamics

Biochar amendments can significantly influence soil nutrient cycling by altering several key processes. Firstly, the porous structure of biochar provides a habitat for soil microorganisms, fostering microbial activity and affecting nutrient transformations. Microbes colonizing biochar pores engage in nutrient cycling processes, potentially enhancing nutrient retention or immobilization. Secondly, biochar's high cation exchange capacity can influence nutrient availability [57]. It can adsorb and release nutrients slowly over time, affecting their mobility and availability to plants and soil organisms. For instance, biochar can adsorb and retain cations such as potassium, calcium, and magnesium, preventing their leaching and improving their availability to plants and soil biota. Additionally, biochar can modify soil pH, albeit to varying degrees depending on feedstock and application rates. Changes in pH can influence nutrient availability, impacting the form and solubility of nutrients in soil. Alterations in soil pH resulting from biochar applications can indirectly affect nutrient uptake by plants and subsequently impact the quality and quantity of resources available to soil-dwelling insects [58].

11. Indirect effects of biochar on soil insects via nutrient cycling

The modifications in soil nutrient dynamics induced by biochar can have indirect effects on soil-dwelling insects. Soil insects, comprising a diverse range of taxa involved in decomposition, nutrient cycling, and predation, rely on soil resources for their survival and reproduction [43]. Changes in nutrient availability and cycling processes due to biochar amendments can influence the nutritional quality of food

resources available to soil insects. For instance, alterations in nutrient availability can impact the quality and quantity of organic matter and microorganisms available as food sources for soil insects [59]. Changes in the nutritional content of these resources may affect the growth, reproduction, and survival of soil-dwelling insects, subsequently influencing their population dynamics within soil ecosystems. Moreover, shifts in nutrient availability and cycling processes can indirectly impact plant health and growth. Soil-dwelling insects often interact with plants, either as herbivores, pollinators, or in symbiotic relationships. Changes in plant health resulting from biochar-induced alterations in nutrient availability may influence plant-insect interactions, subsequently impacting the dynamics of soil insect populations.

11.1 Interactions between soil microbes and plants in biochar-amended soils

Interactions between soil microbes, plants, and insects within biochar-amended soils represent a complex web of ecological relationships crucial for understanding soil ecosystem functioning, nutrient cycling, and the dynamics of soil-dwelling organisms. Biochar amendments can intricately influence the interactions among these key components, shaping soil microbial communities, plant health, root exudation patterns, and subsequently impacting soil insect populations [60]. Biochar amendments can alter the structure, composition, and activities of soil microbial communities. The porous structure and high surface area of biochar provide habitats for microbial colonization and influence microbial diversity, abundance, and functional traits. Studies indicate that biochar incorporation into soils can promote the proliferation of certain microbial taxa, alter microbial community structures, and affect microbial metabolic activities. The presence of biochar in soil can create microenvironments that support microbial growth, modify soil moisture, aeration, and nutrient availability. Changes in soil physicochemical properties resulting from biochar amendments, such as alterations in pH, organic carbon content, or nutrient availability, can selectively favor certain microbial groups, influencing their activities and functions within soil ecosystems [61].

Biochar-induced alterations in soil microbial communities can impact plant health, growth, and root exudation patterns. The interactions between biochar, soil microbes, and plant roots can lead to modifications in the quality and quantity of root exudates. Changes in the rhizosphere resulting from biochar applications can influence soil microbial communities associated with plant roots, affecting nutrient cycling, disease suppression, and plant growth promotion. Additionally, biochar amendments can influence plant responses to herbivory and modify plant secondary metabolite profiles [62]. These alterations in plant defenses and responses to biotic stressors can be attributed to changes in root exudates and the rhizosphere environment, subsequently affecting plant-insect interactions within soil ecosystems. The relationships between soil microbes and soil-dwelling insects in biochar-amended soils are multifaceted. Soil insects interact with microbial communities through their feeding activities, altering microbial communities and nutrient cycling processes. The activity of soil insects can influence the decomposition rates of organic matter and the release of nutrients through fragmentation and consumption, consequently affecting microbial communities and their functions [63]. Moreover, changes in microbial-driven processes, such as decomposition rates or nutrient turnover, can indirectly influence soil insect populations by altering the availability of food resources or habitat conditions. Interactions between soil insects and microbial communities can influence the structure and functioning of soil ecosystems, with implications for nutrient

cycling, organic matter decomposition, and ecosystem stability. Further research efforts should focus on unraveling the complexities of these interactions under varying environmental conditions, considering the context-dependent responses of soil microbes, plants, and insects to biochar amendments. Comprehensive assessments are crucial to elucidate the ecological implications of biochar applications and guide sustainable soil management practices for enhancing soil health and ecosystem functioning [64].

Biochar has shown promising potential as a tool for managing soil insect infestations, offering multifaceted mechanisms that contribute to mitigating infestation patterns within agricultural and natural ecosystems. Several theories and mechanisms underpin its ability to effectively control soil insect populations, thereby presenting practical implications for pest management strategies [65].

12. Biochar as a potential tool for managing soil insect infestations

The application of biochar has demonstrated notable effects in reducing soil insect populations and mitigating infestation patterns. Its properties, such as high surface area, porous structure, and physicochemical characteristics, play key roles in altering soil conditions and influencing insect populations. Biochar amendments have exhibited insecticidal properties, affecting insect behavior, development, and survival rates [66].

12.1 Mechanisms and theories behind biochar's ability to mitigate infestations

Several mechanisms contribute to biochar's efficacy in managing soil insect infestations. One mechanism involves the adsorption of compounds detrimental to insects, such as volatile organic compounds or toxic substances released during biochar production. These adsorptive properties may disrupt insect communication, impede feeding behavior, or deter oviposition, thereby reducing insect populations [67]. Furthermore, alterations in soil physicochemical properties induced by biochar, such as changes in pH, nutrient availability, and microbial activity, can influence insect survival and behavior. Shifts in soil pH can directly affect the survival of certain insect species, while changes in nutrient availability might impact insect nutrition, growth, and reproduction. Additionally, biochar's effects on soil microbial communities can indirectly influence soil insect populations by modifying their food resources or altering microbial-driven processes essential for insect survival [68].

12.2 Practical implications and applications in pest management

The practical implications of biochar in pest management strategies are significant. Incorporating biochar into agricultural soils offers a sustainable and eco-friendly approach to controlling soil insect infestations. Its use aligns with integrated pest management practices, minimizing the reliance on chemical pesticides and reducing environmental risks associated with conventional pest control methods. Moreover, biochar applications present opportunities for long-term pest management by improving soil health and fostering a balanced soil ecosystem. Enhancements in soil structure, nutrient retention, and microbial activity due to biochar amendments can create conditions unfavorable for certain insect pests, thereby contributing to sustained pest suppression [69]. Implementing biochar in pest management strategies requires considerations of various factors, including biochar type, application rates,

soil type, and specific pest species. Field trials and experimental studies tailored to specific agroecosystems are essential for elucidating the most effective biochar application methods and understanding the nuanced responses of different insect pests to biochar amendments [70].

13. Challenges in studying biochar-soil insect interaction

Studying the intricate interactions between biochar and soil-dwelling insects presents numerous challenges, highlighting limitations in current research while pointing toward essential future directions in understanding and applying biochar in pest management strategies [71].

One primary challenge lies in the complexity of soil ecosystems. Soil is a dynamic environment with intricate interactions between biotic and abiotic factors, making it challenging to isolate and understand the specific mechanisms through which biochar influences soil-dwelling insects [72]. The diverse responses exhibited by different insect taxa to biochar amendments add to this complexity, requiring comprehensive assessments and tailored approaches for various pest species. Moreover, the variability in biochar properties stemming from differences in feedstock, production methods, and pyrolysis conditions contributes to challenges in studying biochar-soil insect interactions. Understanding how these variations influence the effectiveness of biochar in mitigating pest infestations necessitates standardized protocols for biochar characterization and application across studies. Another challenge involves the timescale and persistence of biochar effects. Long-term studies are essential to discern the temporal dynamics of biochar-induced changes in soil properties and their sustained impacts on soil insect populations. Considering the slow release of nutrients and alterations in soil conditions due to biochar amendments, assessing its persistence and durability in affecting soil insect communities over time becomes crucial [73].

14. Limitations of current research and areas for improvement

Current research on biochar-soil insect interactions faces limitations in terms of the breadth and depth of investigations. Many studies focus on specific insect taxa or limited ecological contexts, providing a fragmented understanding of biochar's effects on soil insect communities. Broader assessments encompassing diverse insect groups, ecosystems, and environmental conditions are necessary to elucidate generalizable patterns and mechanisms [74]. Furthermore, the majority of existing studies predominantly explore short-term effects, often overlooking the long-term implications of biochar applications on soil insect populations and ecosystem dynamics. Longitudinal studies examining the persistence of biochar-induced changes and their cascading effects on soil insect communities are critical to ascertain sustained pest management outcomes. The methodologies employed in studying biochar-soil insect interactions also warrant improvements. Integrating interdisciplinary approaches that combine ecological, biochemical, and molecular techniques could offer comprehensive insights into the underlying mechanisms driving biochar's effects on soil insect populations. Additionally, employing advanced imaging technologies and molecular tools could enhance our understanding of the direct and indirect impacts of biochar on insect behavior, physiology, and community dynamics [75].

15. Future directions for research and practical applications

Future research endeavors should prioritize investigating context-specific responses of diverse soil insect taxa to biochar amendments. Understanding the differential responses among pest and beneficial insect species and their trophic interactions in various agricultural and ecological settings is crucial. This knowledge can guide the development of targeted biochar-based pest management strategies that minimize adverse impacts on beneficial insect populations. Long-term field trials across diverse ecosystems are imperative to assess the practical applicability and efficacy of biochar in managing soil insect infestations. Field studies spanning multiple growing seasons can offer valuable insights into the persistence of biochar effects on soil properties, insect populations, and crop performance, facilitating the development of sustainable pest management practices. Furthermore, translating research findings into practical applications necessitates collaborations between researchers, agricultural practitioners, and policymakers. Engaging stakeholders in knowledge dissemination, developing guidelines for biochar application, and integrating biochar-based pest management approaches into existing agricultural systems are essential steps toward fostering its adoption and implementation in real-world scenarios.

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Chapter 3

The Impact of Biochar Applications on Specific Soil Fertility Parameters

Mahamane Galadima Moustapha and Erdem Yilmaz

Abstract

Biochar technology is a rapidly growing field of interest within the scientific community due to its multifunctional functions. This study aims to assess the impact of the application of different types of biochar on specific soil fertility parameters. To conduct this study, four different types of plant residues (Vine (*Vitis vinifera* L.), Tomato (*Solanum lycopersicum* L.), Banana (*Musa*), and Carnation (*Dianthus caryophyllus*. L.)) were used to produce biochar through slow pyrolysis system at 300°C and 500°C. The experiment was designed in randomized complete block with five replications and nine treatments. The treatments included the Control, Vineyard biochar (300°C and 500°C), Tomato biochar (300°C and 500°C), Banana biochar (300°C and 500°C), and Carnation biochar (300°C and 500°C). The trial consisted of a total of 45 pots. Each pot contained 10 kg of soil and 80 g of biochar (equivalent to 20 tons ha^{-1}) strongly mixed and incubated for 300 days. At the end of the incubation period, the biochar treatments were found to improve specific soil fertility parameters (pH, EC, CEC, soil penetration resistance, and bulk density) compared to the control. The use of biochar as a soil enhancer proved to be an effective method for managing soil fertility. This research provides valuable insights into the potential benefits of biochar in sustainable agriculture.

Keywords: biochar, soil fertility, sustainable agriculture, agricultural residues management, agroecology

1. Introduction

The decline in fertility of agricultural land is primarily caused by a reduction in soil organic matter, excessive use of chemicals for nutrition and plant protection, and inadequate soil cultivation practices [1]. However, approximately 33% of agricultural land is estimated to be degraded due to erosion, salinization, compaction, acidification, and other chemical pollution factors [1]. To ensure the preservation and sustainable enhancement of soil fertility, it is crucial to implement innovative technologies for managing fertilization in a sustainable manner. Among these technologies, biochar has gained significant attention and sparked scientific debates. In the field of agriculture, biochar is utilized as soil enhancer to improve soil fertility parameters and boost crop productivity [2–9]. However, the impact of biochar application can vary significantly due to its intrinsic properties, which are heavily influenced by the type of biomass used and pyrolysis conditions [10–13]. Therefore, the aim of this

study is to assess the effectiveness of applying different types of biochar produced from various crop residues, pyrolyzed at two different temperatures rate (300°C and 500°C) on specific soil fertility parameters.

By evaluating the impact of these different biochar applications on soil fertility, we can gain valuable insights into their potential as sustainable fertilization management tools. This research will contribute to the development of more effective strategies for maintaining and enhancing soil fertility, ultimately benefiting agricultural practices and ensuring long-term sustainability.

2. Materials and methods

2.1 Location and characteristics of the experiment site

The research was carried out in a modern greenhouse located in Mediterranean climate zone of Akdeniz University, Antalya, Turkiye. The soil of this region belongs to Xerofluvent taxonomic class; summers are hot and dry; winters are warm and rainy. The annual average temperature is around 16.3°C, whereas the average annual precipitation is 725.9 mm, and most of the precipitation occurs in winter. The average of relative humidity is 63.2%.

2.2 Original soil collection and characteristics

The experimental soil was collected from the soil surface (0–20 cm) of Aksu field, located in the Agriculture Research and Application of Akdeniz University and analyzed in the laboratories of soil science and plant nutrition department of Agricultural Faculty of Akdeniz University of Antalya, Turkiye. The original soil had texture of clay loam soil and was composed of 10.88% sand, 42.4% silt, and 46.72% clay. Soil characterization results are presented in **Table 1**.

2.3 Biochar characteristics

Four different agricultural residues were used to produce biochar used in this study; it included Vineyard (VB), Tomatoes (TB), Banana (BB), and Carnation (CB). The biochar was produced by slow pyrolysis system at 300°C and 500°C for 12 h. Prior to use, the biochar were ground into a small particle size of less than 1 mm using a sieve. A sample of the biochar was taken for characterization. The following analysis were performed in order to figure out the characteristics of the biochar. Proximate analysis was performed following ASTM D3173-03, ASTM D3175-07, and ASTM D3174-02 methods for moisture, volatile matter, and ash, respectively. Fixed carbon content was determined by difference (ASTM D3172-07a). Elemental analysis was performed using the Elemental Analyzer (CHNS-932 LECO). pH 1:10 biochar:water ratio was used to determine the pH and EC [14]. The basic physical and chemical properties of pyrolyzed biochar are shown in **Table 2**.

2.4 Experimental design

The trial was designed in a factorial randomized complete block with nine treatments and five replications. The treatments consisted of biochar obtained from various residues pyrolyzed at 300°C and 500°C. They included Vineyard biochar

Soil properties	Results
pH (1:2.5)	7.7
EC dS m^{-1} (1:2.5)	1.96
Lime ($CaCO_3$) (%)	29.8
Sand (%)	10.88
Clay (%)	46.72
Silt (%)	42.40
Organic matter (%)	1.94
Total nitrogen (%)	0.119
Available P (kg P_2O_5 da–^{1})	2.62
Extractable K (kg K_2O da^{-1})	35.0
Extractable Ca (kg CaO da^{-1})	1407.4
Extractable Mg (kg MgO da^{-1})	74.4
Available Fe (ppm)	7.95
Available Mn (ppm)	5.43
Available Zn (ppm)	0.12
Available Cu (ppm)	1.70

Table 1.
Physicochemical properties of the original soil.

Parameters	Treatments							
	VB300	VB500	TB300	TB500	BB300	BB500	CB300	CB500
pH (1:10)	8.13	9.19	8.92	9.67	8.72	10.01	9.56	9.82
EC (dS m^{-1})	0.29	0.56	4.41	4.44	3.41	3.64	4.92	6.23
C (%)	63.57	54.75	60.00	51.62	51.75	47.69	54.00	49.93
H (%)	4.53	2.62	3.77	2.29	3.97	1.92	4.55	1.88
N (%)	1.37	0.86	1.78	1.39	0.67	1.04	3.22	2.49
O (%)	30.53	41.77	34.15	44.27	43.48	49.18	38.11	45.42
FC	58.01	73.03	40.42	46.51	41.42	54.35	33.05	40.82
Ash	5.39	8.93	23.82	30.62	17.65	25.55	23.92	32.48
VM	33.26	14.01	30.87	19.86	37.13	17.38	38.66	22.55

FC: Fix Carbon; VM: Volatile Matter; VB: Vineyard Biochar; TB: Tomato Biochar; BB: Banana Biochar; CB: Carnation Biochar.

Table 2.
Physicochemical properties of biochar.

(VB300, VB500), Tomato biochar (TB300, TB500), Banana biochar (BB300, BB500), and Carnation biochar (CB300, CB500). A total of 45 pots (four different types of agricultural residues × two pyrolysis temperature × five replications +five controls) constituted the trial. Each pot contained 10 kg of soil on which was added 80 g of biochar corresponding to the proportions of 20 tons ha^{-1}. Two (2) incubation periods

of five (5) months were observed. At the beginning of each incubation period, the biochar was thoroughly mixed to the soil in the pots and left to incubate. During incubation, the soil was regularly irrigated. The same protocol was repeated for the second incubation period. At the end of each incubation period (first and second), soil samples were collected and analyzed according to laboratory protocol. Soil fertility parameters such as pH, electrical conductivity (EC), cation exchange capacity (CEC), bulk density, and soil penetration resistance were measured.

2.5 Statistical analysis

The multivariate analysis was conducted on the data collected using SPSS V.17.0 (SPSS 2008). Duncan means of treatments were separated using Duncan Multiple Range Test (DMRT) where their significant differences ($p < 0.05$) were observed.

3. Results and discussion

3.1 Effect of biochar application on soil pH

The application of various types of biochar produced at 300°C did not have a statistically significant impact on soil pH values during the initial incubation period. However, during the second period, the application of biochar did have a significant effect on soil pH ($p < 0.001$). Among the treatments, VB300 demonstrated the most effective leveling of soil pH, with 7.22 and 7.37 values compared to the control (7.44; 7.57) during the first and second incubation periods, respectively.

In contrast, the application of different types of biochar produced at 500°C did not significantly affect soil pH values during the first period. However, during the second period, the application of these biochar did have a significant effect on soil pH values ($p < 0.001$). Specifically, CB500 (7.25) and TB500 (7.45) exhibited the lowest pH values over both incubation periods compared to the control (7.44; 7.57).

Overall, these findings suggest that the application of biochar can have a notable impact on soil pH, with the specific effects varying depending on the type of biochar and the incubation period. The means of the pH results for each sample are presented in **Table 3**, and a boxplot of the pH results is presented in **Figure 1**.

After 300 days of incubation, it was found that the application of various biochar treatments produced at of 300°C and 500°C helped to stabilize the soil pH level compared to the control during the two incubation periods outside BB300, CB300, and CB500 [15]. The effect of biochar application on soil pH is strongly influenced by the type of biomass used and the pyrolysis process. There exists a close relationship between the pH of the biochar and the pH of the treated soil. The alkalinity of biochar plays a crucial role in regulating acidic soils [8]. Applying biochar to acidic soils would result in an increase in soil pH [16, 17]. Streubel et al. [18] demonstrated that the application of biochar derived from different herbaceous (pH 9.4) and woody (pH 7.4) plant sources led to pH changes in treated soils of 0.4–0.8 and 0.1–0.4 units, respectively.

3.2 Effect of biochar application on soil EC

The results of the application of various types of biochar produced at 300°C showed that the application of the different treatments did not have statistically

Treatment	Temperatures			
	300°C		500°C	
	I	II	I	II
Control	7.44	7.57AB	7.44	7.57AB
VB	7.22b	7.37aC	7.33	7.51BC
TB	7.25b	7.50aB	7.33	7.45BC
BB	7.45	7.53B	7.30b	7.49aBC
CB	7.39b	7.63aA	7.25b	7.60aA

Means with different small letters in a column are significantly different at the 5% level. Means with different capital letters in the row are significantly different at the 5% level. ns: not significantly different at the 5% level. Vineyard biochar (VB), Tomato biochar (TB), Banana biochar (BB), Carnation biochar (CB).
** Significantly different at the 5% level.*
**** Significantly different at the 0.01% level.*

Table 3.
Effect of biochar applications on soil pH.

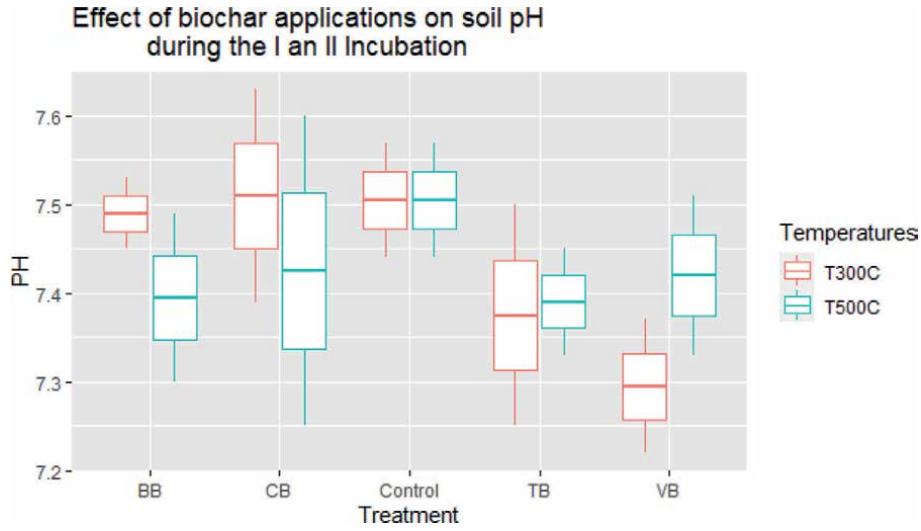

Figure 1.
Boxplot of pH measurements after biochar application. Upper and lower hinges correspond to 25th and 75th percentiles (1st and 3rd quartiles), respectively. Solid line corresponds to median. Whiskers extend to the highest and lowest values within 1.5 times the interquartile range.

significant effects on soil EC during the first incubation period. However, during the second application period, significant effects on soil EC ($p < 0.001$) were found. Among the biochar produced at 300°C, TB300 was the treatment that best improved soil EC relative to the control during the first and second incubation periods, respectively. Application of the different types of biochar produced at 500°C had no significant effect on soil EC during the first period. However, a statistically significant effect was observed on soil EC ($p < 0.001$) during the second incubation period. CB500 and TB500 were the treatments that best improved soil EC values over both incubation periods relative to the control. The means of EC results for each sample are presented in **Table 4**, and a boxplot of EC results is presented in **Figure 2**.

Treatment	Temperatures			
	300°C		500°C	
	I	II	I	II
Control	1.041	0.482aC	1.041	0.485B
VB	1.541a	0.518bBC	1.389a	0.488bB
TB	1.841a	0.868bA	1.623	0.893A
BB	1.022a	0.680bB	1.658	0.794A
CB	1.492	0.655aB	2.257a	0.783bA
LSD	0.3612 ns	0.0005***	0.2798 ns	0.0006**

Means with different lowercase letters in a column are significantly different at the 5% level. Means with different capital letters in the row are significantly different at the 5% level. ns: no significant difference at the 5% level. LSD: average of the five repetitions. Vineyard biochar (VB), Tomato biochar (TB), Banana biochar (BB), Carnation biochar (CB).
***Significantly different at the 1% level.*
****Significantly different at the 0.1% level.*

Table 4.
Effect of biochar applications on soil EC (dS m⁻¹).

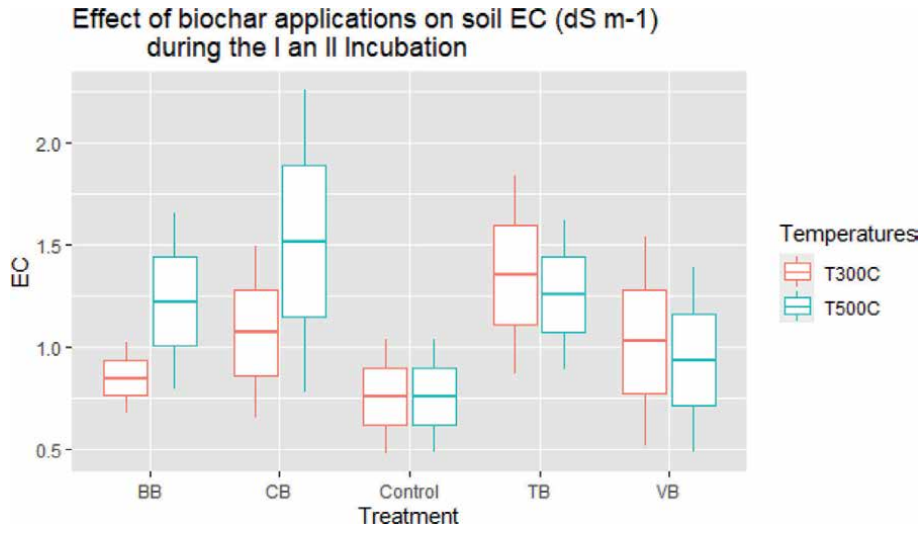

Figure 2.
Boxplot of EC measurements after biochar application. Upper and lower hinges correspond to 25th and 75th percentiles (1st and 3rd quartiles), respectively. Solid line corresponds to median. Whiskers extend to the highest and lowest values within 1.5 times the interquartile range.

At the end of the incubation period, although the soil EC values were low (less than 4 dS m^{-1}), the application of various types of biochar produced at different temperatures (300°C and 500°C) resulted in an increase on soil electrical conductivity values during the first and second incubation periods compared to the control. These values provide evidence that the soil is not saline [19]. This conclusion aligns with the similar results reported by [20].

3.3 Effect of biochar application on cation exchange capacity (CEC)

From the table, we observed the results obtained from the application of various types of biochar derived from different crop residues at 300°C temperature rate. These findings reveal that the application of these different treatments had a significant effect ($p < 0.01$) on soil CEC during the first incubation period. However, during the second incubation period, the treatments did not have a significant impact on soil CEC.

Among the biochar treatments produced at 300°C, TB300 and VB300 exhibited the most substantial enhancement in soil CEC compared to the control during the first and second incubation periods, respectively. On the other hand, the application of biochar produced at 500°C did not have a significant effect on soil CEC during the first period. However, during the first and second periods, biochar produced at 500°C demonstrated a significant effect at $p < 0.01$ and $p < 0.05$, respectively, on soil CEC. VB500 and CB500 were the treatments that displayed the greatest improvement in soil CEC values over both incubation periods compared to the control. In summary, the application of biochar derived from different agricultural residues pyrolyzed at different temperatures had varying effects on soil CEC. The treatments produced at 300°C showed significant improvements in soil CEC, while the treatments produced at 500°C had a significant impact on soil CEC during the second period. These findings highlight the potential of biochar application in enhancing soil properties and suggest that the choice of biochar type and pyrolysis process can influence its effectiveness in improving soil CEC. The means of CEC results for each sample are presented in **Table 5**, and a boxplot of CEC results is presented in **Figure 3**.

At the end of the incubation period, the application of biochar produced at different temperatures (300°C and 500°C) rate led to a noticeable increase in CEC values during the first and second incubation periods compared to the control (BB300, CB300, TB500, and BB500). Although these values remain relatively low (less than 20 meq 100 g^{-1}), this can be attributed to the type of clay present in the original soil.

Treatment	Temperatures			
	300°C		500°C	
	I	II	I	II
Control	7.49B	7.31	7.79C	7.31ABC
VB	9.75aA	7.90b	10.07aA	7.53bAB
TB	9.87aA	7.50b	8.40aBC	6.61bBC
BB	8.87aA	6.81b	8.99aB	6.20bC
CB	7.51B	7.10	8.90aB	8.04bA
LSD	0.0005***	0.4784 ns	0.0008***	0.0153*

Means with different lowercase letters in a column are significantly different at the 5% level. Means with different capital letters in the row are significantly different at the 5% level. ns: no significant difference at the 5% level. LSD: average of the five repetitions. Vineyard biochar (VB), Tomato biochar (TB), Banana biochar (BB), Carnation biochar (CB).
**Significantly different at the 5% level.*
****Significantly different at the 0.1% level.*

Table 5.
Effect of biochar applications on soil CEC (meq 100 g^{-1}).

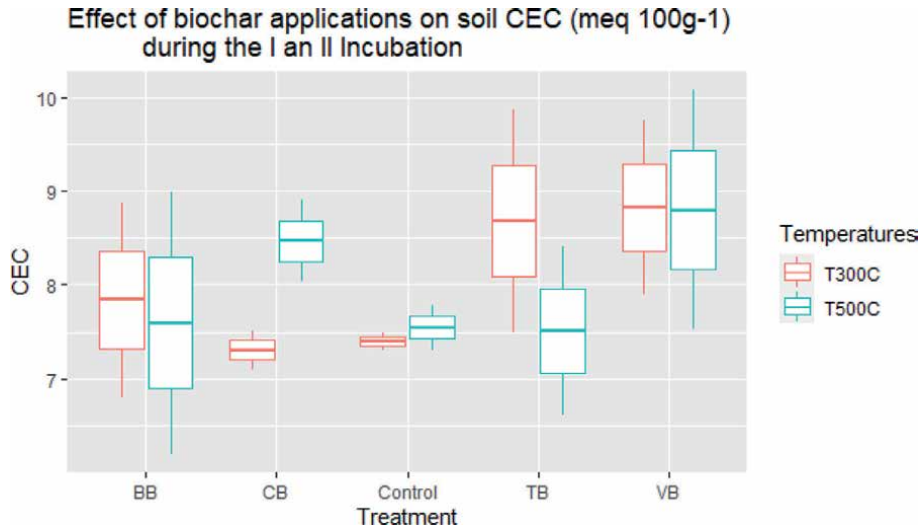

Figure 3.
Boxplot of CEC measurements after biochar application. Upper and lower hinges correspond to 25th and 75th percentiles (1st and 3rd quartiles), respectively. Solid line corresponds to median. Whiskers extend to the highest and lowest values within 1.5 times the interquartile range.

However, previous research by [21, 22] suggests that the specific surface area and pore structure of biochar can enhance soil CEC values when applied. In a study conducted by [23], it was found that the application of biochar derived from olive residues to a silt–clay–sand soil resulted in a 5% increase in CEC value. Similar studies in the literature, such as those by [3, 24, 25], have also reported significant increases in soil CEC following the application of biochar. Overall, these findings highlight the potential of biochar to improve soil CEC values, although further research is needed to fully understand the mechanisms behind this enhancement.

3.4 Effect of biochar application on bulk density

The application of various types of biochar, derived from different crop residues at 300°C, was found to have no significant impact on bulk density during the initial incubation period. However, during the subsequent incubation period, these treatments had a significant effect on bulk density. Among the biochar produced at 300°C, TB300 treatment was the most effective in reducing soil bulk density compared to the control, both in the first and second incubation periods. Furthermore, the biochar produced at 500°C showed a statistically significant effect, with p-values of less than 0.01 and 0.001% in the first and second incubation periods, respectively. Specifically, the TB500 and BB500 treatments exhibited the most substantial decrease in soil bulk density values in the first and second incubation periods, respectively, compared to the control. These findings emphasize the potential of biochar application, particularly at specific temperatures, to effectively reduce soil bulk density. This information is valuable for gaining insights into the impact of biochar on soil properties and can contribute to the development of sustainable agricultural practices. The means of the bilk density results for each sample are presented in **Table 6**, and a boxplot of the bulk density results is presented in **Figure 4**.

The application of various types of biochar derived from different crop residues at different temperatures rate (300°C and 500°C) led to a noticeable decrease in bulk

Treatment	Temperatures			
	300°C		500°C	
	I	II	I	II
Control	1.37b	1.45aA	1.37bA	1.45aA
VB	1.31	1.27B	1.32AB	1.29BC
TB	1.29	1.25B	1.26C	1.25C
BB	1.30	1.26B	1.30aBC	1.24bC
CB	1.34	1.29B	1.28 BC	1.33B
LSD	0.1202 ns	0.0009***	0.0028**	<0.001***

Means with different lowercase letters in a column are significantly different at the 5% threshold. Means with different capital letters in the lines are significantly different at the 5% level. ns: no significant difference at the 5% level. LSD: average of the five repetitions. Vineyard biochar (VB), Tomato biochar (TB), Banana biochar (BB), Carnation biochar (CB).
***Significantly different at the 1% level.*
****Significantly different at the 0.1% level.*

Table 6.
Effect of biochar applications on the bulk density (g cm^{-3}).

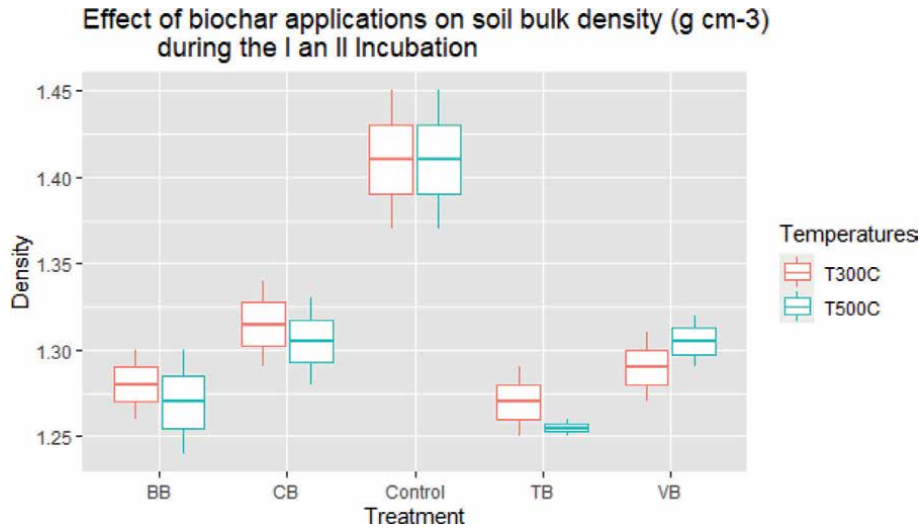

Figure 4.
Boxplot of bulk density measurements after biochar application. Upper and lower hinges correspond to 25th and 75th percentiles (1st and 3rd quartiles), respectively. Solid line corresponds to median. Whiskers extend to the highest and lowest values within 1.5 times the interquartile range.

density values during the first and second incubation periods compared to control. These findings align with the research conducted by [6, 26–28]. Moreover, similar results have been found in the literature studies [29–36].

3.5 Effects of biochar application on soil penetration resistance

From **Figure 5**, results showed a decrease in soil penetration resistance values after biochar application during the first and second incubation periods compared to the control. The application of these different treatments resulted in the soil

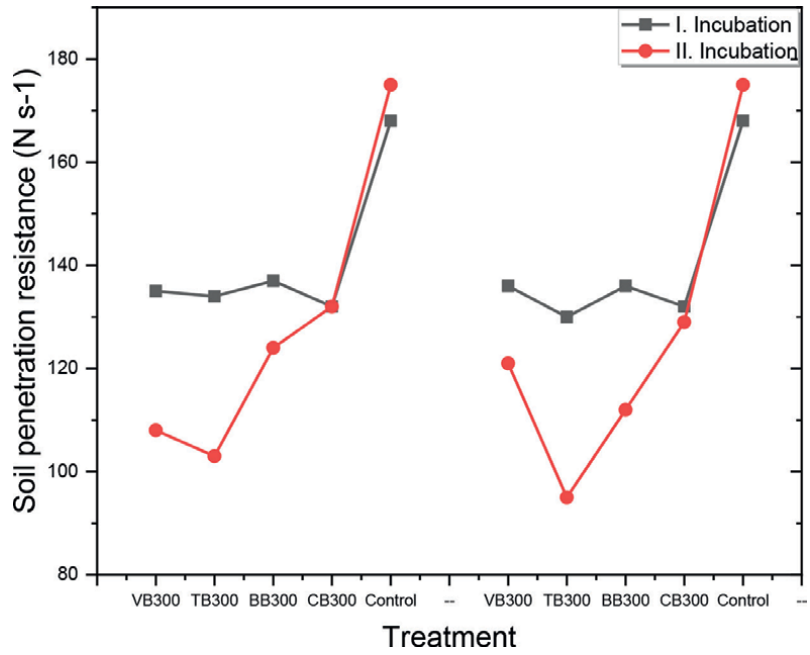

Figure 5.
Effects of biochar application on soil penetration resistance (N s^{-1}).

decompaction. Numerous studies have reported that applying biochar to soil causes a decrease in soil compaction [29, 31, 32, 37, 38]. Additional research conducted by [37, 38] also supports the thesis of notion that biochar application led to a decrease in soil compaction.

4. Conclusion

The study examined the impact of applying various types of biochar, derived from different agricultural residues through slow pyrolysis system at two different temperatures of pyrolysis, on specific soil fertility parameters. Results revealed a significant improvement in the fertility parameters of the treated soil. Firstly, the results demonstrated that the application of four biochar types VB, TB, BB, and CB, produced at two temperatures rate 300 and 500°C, effectively limed the soil pH levels in comparison with the control. Additionally, the electrical conductivity (EC) values also exhibited improvement as a result of the different treatments, surpassing those of the control. Moreover, the cation exchange capacity (CEC) values were significantly increased compared to the control. Furthermore, the application of these different treatments led to a reduction in the bulk density when compared to the control. Lastly, the various treatments applied successfully alleviated soil compaction by reducing the values of soil penetration resistance in comparison to the control soil. This study highlights the positive impact of applying different types of biochar, produced from various agricultural residues through slow pyrolysis system at different temperatures (300 and 500°C), on soil fertility parameters. The findings indicate improvements in soil pH levels, electrical conductivity, cation exchange capacity, bulk density, and soil decompaction. These results contribute to our understanding of

sustainable agricultural practices and the potential benefits of biochar application in enhancing soil fertility.

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Chapter 4

Perspective Chapter: The Role of Biochar in Soil Amelioration

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Abstract

Many techniques have been employed in restoring the health of physically, chemically and biologically degraded soils. Some of these techniques are expensive, time consuming and may involve soil excavation or chemical treatments with numerous washes in some cases. There is a novel technique that is cheap, can restore the properties of a degraded soil, mitigate climate change and sequestrate carbon in the soil. That technique is the biochar technology. In this review, we'll look at biochar technology as an ameliorant in improving impoverished soils. Biochar is a carbon-rich substance that is produced when biomass (feedstock) is subjected to a thermal decomposition process under limited oxygen called pyrolysis. Biochar can be used to ameliorate soil acidity and alkalinity depending on the feedstock. It has advantages such as increasing cation exchange capacity, soil carbon and nutrient in the soil. Biochar can be inoculated with specific organisms for pollutant breakdown and acts as a habitat for naturally occurring microbes; by binding pollutants in the soil through the process of bioaccumulation, sorption, electrostatic attraction and precipitation, it acts as a remediation agent. However, the feedstock, pyrolysis temperature, and heating period can all affect the properties of biochar and its biological processes.

Keywords: heavy metals, biochar, microbial diversity, soil carbon, remediation

1. Introduction

With the increasing loss of soil particles through erosion, nutrient loss through leaching and organic matter, maintaining agricultural output is a major concern on a global scale. The modern agroecosystem has a huge challenge to continuous production of food while conserving the soil fertility with the recent rise in population, which is approaching the historic number of 8 billion [1]. According to Kopittke et al. [2], about 99% of food consumed by man comes from the soil, which also serves a number of ecosystem services [3]. As a result, degradation of soil nutrients and productivity poses a danger to global food security [1]. To reclaim degraded soil, amendments are applied to the soil. One of the novel amendments that are recently used is biochar.

Biochar is a pyrolyzed material that is mass-produced by pyrolyzing organic material (waste from plants and animals) at elevated temperatures in an oxygen-free

atmosphere [4, 5]. The most prevalent approach to producing biochar for crop growth and environment is through the slow pyrolysis (350–700°C) of organic materials in the presence of limited oxygen. Alterations in temperature, raw material, moisture and pyrolysis time are major factors to be considered in the production of biochar. A wide range of organic raw materials such as agroforestry sources (wood, tree bark, wood shavings, roots and shells), livestock-related sources (beddings and animal waste products), agricultural waste-products (corn stalks, rice husk, root residues and forage crop), waste products from industries (carbon black and bagasse), agro-urban biosolids and materials that contains carbon can be used for the production of biochar [6]. The main components of biochar are cellulose and lignin, which are derived from plant biomass and have a high carbon concentration but low hydrogen and oxygen content. In recent years the usage of diverse sources of biochar has received numerous attentions, due to its function as soil amendment to minimize soil compaction and nutrient/water holding capacity and increase soil porosity due to attached functional groups, pore properties, high adsorption capacity and surface activity [7]. In general, biochar produced at pyrolysis of reduced temperature range of 100–300°C results to increase in the amount of acidic functional groups, adsorption capacity, yield and porosity tend to increase when applied in the soil, according to Sun et al. [8]; while at elevated temperatures range of 600–700°C, there is an increase in pH above 7, amount of major functional groups present and fixed carbon content, following applied biochar produced at that temperature range. Due to biochar's increased surface area, stable carbon content and high porosity, its application can provide numerous benefits in soil amelioration [6, 9]. The considerable porous carbon rich material with an increased specific surface area could be applied as soil amendments to absorb organic/inorganic contaminants to include pesticide residues, herbicides and heavy metals, thereby boosting the quality of the soil for environmental and agricultural purposes [10, 11].

2. Impact of biochar on soil physical properties

The effectiveness of applying biochar can be well illustrated on a coarse textured sandy soil, as they reduce bulk density, increase water retention, and improve the soil physical properties [6, 12]. In addition, the feedstock, the soil type used to make biochar determines it's efficient as a soil amendment. Improvement of soil aggregate stability after the usage of biochar to the soil as amendment is of a known significant effect [13, 14]. Improvement in aggregate stability and soil structure of soil in a one year (mustard red clover) and two years fallow period amended with wheat straw, vineyard pruning and wood chip biochar, (application rate 3% by soil weight and pH 8.3–9.7) was recorded. However, the result was more effective on soils with coarse textured compared to fine textured soil as stated by Burrell et al. [15]. In an upland red soil cultivation system for rapeseed and potatoes, wheat straw biochar (pH 10.35, application rate 2–40 t ha^{-1}) boosted soil aggregate stability and microaggregate content while also enhanced the yields of rapeseed and sweet potatoes [16]. Although the impact seems to be short-lived (2 years) and at higher rates of biochar application (>10 t/ha) and more effective in sandy soils with small size biochar [17], biochar amendment can improve soil bulk density and reduce soil compaction [18, 19]. With the addition of biochar, soil bulk density would decrease, which would lessen soil compaction. According to the soil textural class, the optimal soil bulk density varies for different crop production [20]. Low soil bulk density has been shown by Zhang

et al. [9] to enhance soil structure and make it easier for nutrients to be released and retained. The bulk density of the soil is decreased in part by biochar. After applying biochar from orchard pruning, Rombola et al. [21] research revealed that the soil bulk density fell from 1.44 0.10 to 1.38 0.06 g cm^3. Biochar generated from crop residues, had a more significant influence on increasing soil bulk density and lowering soil compaction [18, 22]. Khan et al. [23] also discovered that applying maize straw biochar decreased the soil bulk density in sandy loam.

Through larger pores and aggregate stability, the application of biochar has also been shown to improve soil water retention [24–26]. Crop productivity has been significantly impacted by soil water retention [25]. According to the research, adding biochar affects soil water retention. This effect may be brought on by the biochar's hydrophilic domains, high porosity, and high specific surface area. Oladele et al. [27] discovered that adding rice husk biochar at a rate of 12 t/ha improved soil water retention in sandy clay loam from 36.87 to 32.94%. According to Razzaghi et al. [25], biochar enhanced field capacity in coarse-textured and medium-textured soils by 51 and 13%, respectively. In addition, adding biochar increased the amount of water that was available to plants in coarse, medium, and fine-textured soils by 45, 21, and 14%, respectively. Furthermore, Hussaina et al. [28] found that a variety of variables, including the kind of raw materials, pyrolysis temperature, biochar particle size, soil type, and compaction condition, significantly influenced soil water retention.

3. Impact on soil chemical properties

Soils globally are mostly acidic and still experience the problem associated with acidity due to natural and anthropogenic activities [29]. Plant growth is dependent on many factors including the pH of the soil. An acidic soil affects the plant productivity and this is because in an acidity condition, aluminum fixes phosphorus making it less available for plant uptake [30]. Jeffery et al. [31] observed that the application of biochar significantly increased the pH of acidic soils such as alfisols, ultisols, and latosols [32]. Increasing the soil pH by biochar could be attributed to the following: (1) Functional groups such phenolic -COOH, OH, and alcoholic OH that are present on the surface of biochar and are capable of interacting with basic cations in soils based on varying reactivity levels to help raise soil pH [33]. (2) Biochar from different sources have different amount of ash which is alkaline. Ash has been reported to reduce soil acidity and increase the pH of the soil [34]. Biochar application directly to an acidic soil makes the alkaline components such as sodium oxides, hydroxides, potassium, carbonates, calcium and magnesium to be out in soluble forms into the soil. Thereafter, hydrogen ions and aluminum monomers present would react with these soluble substances to increase the soil pH [35, 36].

In order to determine cation exchange capacity (CEC), it is crucial to assess the adsorption capacity of soil and solid materials for cation exchange. In the process of reducing the nutrients leaching, CEC is also a relevant indicator of soil fertility in supplying sustainable nutrients for plants [37]. Increased soil cation exchange sites will result in an increase in soil CEC. The basic adsorption capacity of calcium, potassium, ammonium, and magnesium is higher in soils that have greater CEC, which enhances the use of nutrient in the soil and lowers nutrient loss [38]. Numerous research has investigated how biochar affects the CEC of soil. By applying maize straw biochar, Khan et al. [39] demonstrated that the cation exchange capacity increased from 12.9 to 15.6 cmol/kg, 17.4 cmol/kg, and 19.2 cmol kg with addition rates of 4 t/ha,

12 t/ha, and 36 t/ha, respectively. According to Hossain et al. [40], the addition of biochar resulted in a 20–40% increase in the total soil CEC and charge. The nutrients and alkaline cations in the soil will increase dramatically with even a tiny addition of charcoal. Additionally, according to Chintala et al. [41], both acidic and alkaline soils can benefit from biochar's improved soil CEC due to the availability of more anions on its surface.

As soon as biochar is added to the soil, it starts to interact with the soil's chemistry; this interaction then advances through dissolution (1 month), the development of reactive surfaces (1–6 months), and degeneration (>6 months) [12]. Along with improving soil nutrient status, specifically covering a long period of time, soil's capacity to retain carbon and nitrogen may also minimize environmental degradation both now and in the future as well as its detrimental effects on animal and human health status [17]. Due to its surface properties and linked functional groups, biochar serves as a good adsorbent for nutrients and pollutants [42, 43]. As a result, biochar enhances nutrient retention in soil by boosting the soil's capacity to collect and hold onto nutrients and agro chemicals, thus reducing their vaporization and leaking into groundwater. Biochar has the potential to provide vital nutrients to the soil solution over time through weathering, decomposition, and other processes. Due to its longer carbon half-life—possibly spanning thousands of years and also because of its durability, biochar can have a long-lasting effect on soil quality and carbon sequestration [44]. Due to its potential for decreased nutrient losses and increased fertilizer efficiency [45–47], biochar has been a strategic tool for agricultural and environmental purposes [12, 48]. In comparison to non-biochar soil, its incorporation into deficient soil improves soil fertility, crop development, and production [49] as shown in **Figure 1**. However, the source and characteristics of the applied biochar, as well

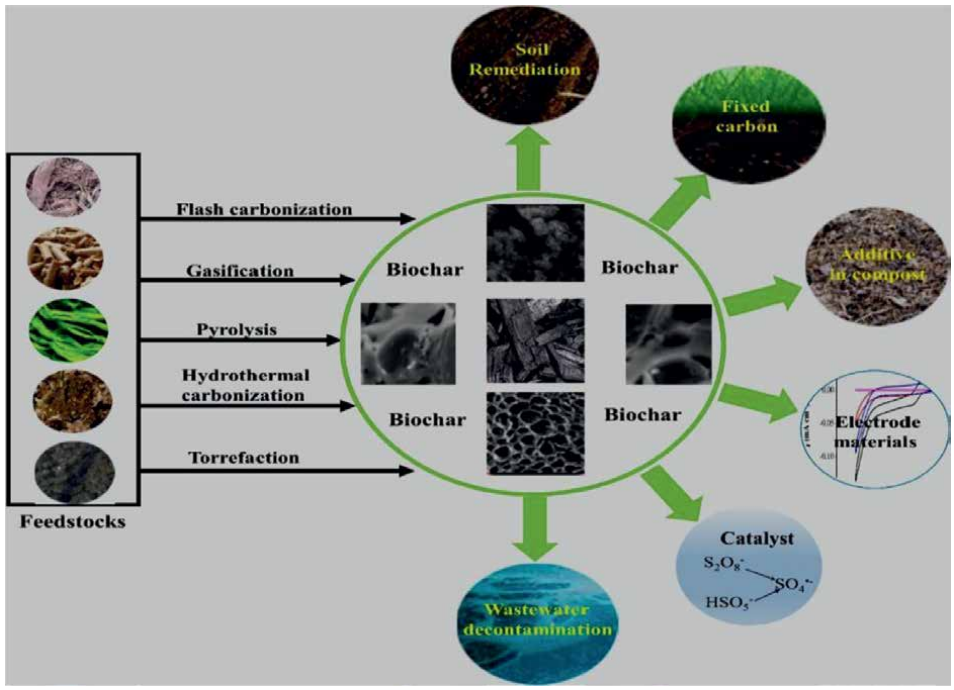

Figure 1.
Biochar production and its application in agriculture [50].

as soil characteristics, environmental plant responses, biochar-soil interactions, and its capacity to absorb nutrients, all play a significant role in these factors [47, 51]. Depending on its inherent nutritional value, capacity for nutrient and water retention, and ability to reduce moisture stress, it may have a notable influence on crop growth and development both directly and indirectly [52]. It is occasionally possible to forecast poor crop performance for the first 30 days following cultivation on acidic soil (pH = 5.5), which may be caused by transient N retention in N-deficient soil [12]. While minimizing nutrient loss, it efficiently transfers nutrients to the soil [53]. In general, biochar can be applied to boost agricultural output due to its chemistry and characteristics, especially in soils with low fertility and soil degradation, where it can be extremely helpful. It has boosted crop output by enhancing soil base saturation [54, 55], retaining nutrients in the soil column where roots are situated, and enhancing nutrient use efficiency [56, 57].

4. Impact on the biological properties of the soil

According to studies by Beheshti et al. [58], and Gorovtsov et al. [59], Due to its alkaline pH and structure, which improves soil porosity, aggregation, and water-holding capacity and promotes soil nutrient bioavailability and microbial growth, biochar has been revealed to improve important soil biological properties. As soon as biochar is applied to the soil, it can interact with soil biological elements like plant roots and root microbes, even on soils that are contaminated with microplastics and metal(loids) [45, 59, 60]. This is true even though biochar is frequently thought of as an appropriate soil treatment for long-term purposes. According to many studies [21, 61], it is a major role in boosting soil microbe development, an increase in soil respiration, soil biomass carbon, and microbial diversity. In fact, it has been claimed that biochar increases crop systemic resistance and even aids in the management of soil diseases by enhancing the biological quality of the soil [62]. According to Warnock et al. [63], biochar has also been linked to increased soil fertility and improved soil carbon sequestration, both of which have been shown to benefit the mycorrhizal community in the soil. Moreover, soil that has been treated with biochar can offer a better environment for the growth and performance of beneficial soil microorganisms. Rhizobia inoculation significantly increased micronutrient fertility (Fe, Zn, Mn, and Cu), soil organic matter (SOM), and soil quality in lentil plants [64]. Depending on the pyrolytic temperature variation and feedstock, these benefits could fluctuate [8, 65]. The alkaline pH and soil aggregating ability of biochar, which is currently acknowledged to improve the soil biological environment [12] are the primary factors. With improved nutrient accessibility for plants and advantageous microbial activity, biochar can be utilized as a soil treatment to improve soil quality [6, 66]. When biochar is put into the soil, it immediately has a major impact on clay minerals and soil organic matter (SOM), which serve as nutrient pool and exchange sites [11]. Applications of biochar have shown across a variety of soil types that soils with enriched microbial diversity and activity are frequently more resistant to management practices [24, 60, 61, 67]. The decomposition of organic substrates to form SOM, biogeochemical nutrient cycling and mineralization, soil structure and aggregation, disease suppression, regulation of hormones that promote plant growth, increased soil water holding and availability to plants, neutralization of toxic compounds, and many other critical processes all depend on soil microbes. By encouraging soil microorganisms,

biochar is essential for controlling soil health. The availability of nutrients needed by plants is increased by biological activity from soil microbes mediated with biochar, which reverses the fixation of nutrients through mineralization [58, 68]. Microbial diversity and community structure in soil are influenced by soil organic carbon, soil structure, and other biochemical soil variables. Recent advancements in technology have made biochar more effective for a range of environmental applications, including the treatment of soil pollution, targeted removal of metal(loid)s and contaminants from soil, solid waste management, and wastewater treatment [69]. This is because biochar can now be produced more efficiently and with the desired physical and chemical properties. The variety of microbial species found in soil provide a natural environment. Researchers can now delve deeply into the intricate mechanisms involved in establishing the interactions amongst microbes, soils, and plants [70] thanks to advancements in molecular studies. In soil, life can be found in three different domains, including prokaryotes, archaea, and fungi, animals, plants and protists. The addition of biochar changes the variety of soil microorganisms. Numerous studies supports adding biochar to soil to improve the microbial populations there, as seen in **Table 1**.

Feedstock	Type of soil	Impact of biochar	References
Chicken manure (500°C)	Sedimentary alfisol	Increase in activities of microorganisms in soil	Meier et al. [72]
Corn straw (500°C)	Sandy loam	Microbial biomass increased Increase in gram positive bacteria Increase in fungal population	Lu et al. [73]
Cotton straw (450°C)	Calcaric Fluvisol	Increase in enzyme activity, microbial activity, as well as microbial biomass	Liao et al. [74]
Date palm waste (300°C)	—	Sorption of heavy metals Increased the microbial biomass, soil respiration and soil organic matter	Al-Wabel et al. [75]
Gliricidia sepium wood (300°C, 900°C)	Serpentine soil	Increased overall soil enzymatic activity Sorption of heavy metals	Bandara et al. [76]
Gliricidia sepium wood (900°C)	—	Microbial biomass increased Increase in bacterial and fungal count Increase in plant growth-promoting bacteria population Absorption of heavy metals	Herath et al. [77]
Glucose	Forest and arable soil	Increase in gram positive bacteria	Steinbeiss et al. [67]
Maize straw (400°C)	Silt loam	Arbuscular mycorrhizal fungi/ saprotrophic fungi ratio affected	Luo et al. [78]
Oak pellet (550°C)	Clay Sandy loam	40–64% increase in phospholipid fatty acid biomass	Awad et al. [79]

Feedstock	Type of soil	Impact of biochar	References
Pinus radiata	Silt-loam	Beneficial for phosphate solubilizing bacteria and carbon degrading bacteria	Anderson et al. [80]
Pinus sp. Dairy + bull manure (500°C)	Fertile Mollisol	Enhances the microbial biomass and microbial activity	Kolb et al. [81]
Rice straw (550°C)	Clay loam	Improved soil organic carbon mineralization Slight increase in microbial biomass	Pan et al. [82]
Sawdust (550°C)	Sandy loam	Increase soil organic matter Increase in phospholipid fatty acid biomass	Gomez et al. [83]
Sugarcane bagasse (450°C)	Sandy loam	Reduction the bioavailability of heavy metals Increase in Actinomycetes population (280%)	Nie et al. [84]
Pine cone and vegetable waste (200°C)	Sandy loam	Absorbed the heavy metals and increased microbial abundance	Igalavithana et al. [85]
Willow leaves (470°C)	Flinty clay loam	Nitrogen cycling is affected Bacteria +28% Actinobacteria +62% Gram-negative bacteria +27%	Prayogo et al. [86]
Willow wood Swine manure (350°C)	Temperature sandy loam	Microbial biomass increased Increase in gram-positive and gram-negative bacteria	Ameloot et al. [87]
Yeast	Arable and forest soil	Nutrients levels enhanced Fungal population increased	Steinbeiss et al. [67]

Table 1.
Biochar to soil to improve the microbial populations [71].

5. Biochar in remediation of metal, metalloids and pollutants

In contrast to remediation techniques such leaching, soil washing or excavation of heavy metals; biochar stabilizes heavy metals in the soil, emanating in reduced solubility and bioavailability of elements [7, 88]. Metal(loid) soil contamination is a problem that affects the entire world [89, 90]. The use of biochar as a soil treatment to enhance soil quality is one of its relevant research and application purposes. It is particularly suitable for poor-quality soils due to its distinct characteristics, such as its porosity, structure, particle size, sizes, pH range, specific surface area and water-holding capacity. Maximizing the carbon content is typically thought to be the most important component because biochar is frequently employed as a soil amendment in the context of agro-ecosystem operations [12, 45, 61]. Its physical, chemical, biological, and biological-chemical interactions (cation exchange capacity, elemental composition, electrical conductivity, pH), as well as its density, specific surface area, size, stability, pore size and distribution, shape, persistence, and other characteristics are all factors that affect the quality of biochar and its efficacy for agricultural and

environmental applications [11, 68, 91]. By converting metals like Cd (II), Zn (II), and Pb (II) into oxide compounds, the formation of biochar at high temperatures (550–750°C) has demonstrated considerable benefits in phytoremediation [92]. However, the reactive surface area, pH, surface groups, ion-exchange capacity, and pore size distribution of biochar affect its capability for heavy metal adsorption [93]. One method used by biochar to lessen heavy metal contamination is to decrease the bioavailability of heavy metals in the soil [93]. Through diverse physical and chemical interactions and a strong adsorption mechanism, biochar serves to raise the pH of the soil and stabilize metals, potentially lowering the bioavailability and leachability of heavy metals in soils [89]. Because biochar has a higher specific surface area and a higher variable charge component, it has a better surface sorption capacity, which affects the soil's capacity to retain water, nutrients, and organic molecules [89, 94]. When biochar is applied to soil, the surface oxygenation of the material alters the O^{2-}-containing functional groups (hydroxyl, carboxyl, carbonyl groups, and phenol) on the large internal surface area of the material, causing an induced negative charge and increasing the soil pH and cation exchange capacity (CEC) [95]. Through the release of cations from biochar, it has been demonstrated that the reduction in soil acidity greatly modifies the soil's N ammonium and nitrate (NH_4^+ and NO_3) and AB-DTPA extractable phosphorus and potassium contents [41]. Due to surface complexation with different functional groups, inter-cyclical complexation with free hydroxyl of mineral oxides, and simple physical adsorption leading to surface precipitation, biochar can help fix metals through sorption by exchange of cations (calcium (Ca^{2+}), magnesium (Mg^{2+})). Additionally, metal(loid) stabilization may benefit from biochar that contains mineral components such phosphates and carbonates [96]. Currently, it has been revealed that soil-based biochar-nanocomposites or biochar-metal oxide modifications can further aid in the adsorption of pollutants. Concerns about its persistence in the environment, however, need to be more thoroughly examined [97].

Protons are added or removed from functional groups depending on the pH which gives biochar an electric charge. Tan et al. [98] and Xu et al. [99] claim that submerging biochar in a solution caused the surface of the biochar to become more negatively charged as the pH of the solution increased from 3 to 8. Through electrostatic interactions, these negatively charged functional groups bind to the cations of heavy metals. The fact that enhanced temperature biochar has a greater pH zero net charge (ZNC; the pH at which the biochar has zero net charge) explains why elevated temperature biochar has a lesser CEC [100]. This also implies that because the pH of a soil that has been altered with biochar is pH ZNC, there may be positive charge sites on the biochar that enable hazardous anionic metals to attach to the surface of the biochar [101]. Due to interactions, the electron-rich aromatic structure of biochar enables electrostatic attraction on metal cations with low electron density [102]. Cations connected to functional groups on the biochar surface can also interact with metals in soil solution through ion exchange or inner sphere interactions. Only when the pH and ionic strength of the soil solution are suitable for ion exchange can it occur. While high ionic strength solutions see reduced exchange as the level of competition in the solution increases, low pH solutions exhibit a reduction in ion exchange [101]. The ability of the biochar to adsorb metal cations via exchanging calcium, magnesium, potassium, and sodium equally increases as the CEC does [103]. According to research, applying biochar increases the availability of chromate (Cr) and arsenate (As), suggesting that combining phytoremediation with biochar application might be a good remediation strategy to effectively lower their mobility in soils.

Precipitation reaction is a crucial step in the detoxification of heavy metals using biochar. The majority of biochar is alkaline, which raises the pH of the soil solution and increases the density of hydroxyls on the surface of the biochar, which adsorbs aluminum or manganese [104]. This makes it possible for more hydroxyl-metal precipitates to form, which lowers the amount of heavy metals that can be found in soil solution. The mineral component of biochar, which includes CO_3^{2-}, PO_4^{3-}, SO_4^{2-}, and SO_3^{2-}, can also produce precipitates. By choosing the right feedstock and pyrolization temperature, sulphate and phosphate can be bonded by ligand exchange or precipitate on the biochar's surface [102, 105, 106]. Due to an increase in aromatic carbon and the loss of O-containing functional groups, which were replaced with phosphorus (p) and Sulfur (s); P sorption increases as biochar is formed at higher temperatures. Precipitation can produce more stable bonds than surface complexation or bonding when these anions and carbonates are combined with metals like mercury (S specific), Zinc, lead, copper or cadmium.

The functional groups and structural makeup of biochars all influence their ability to transport electrons. According to Chacón et al. [107], phenols are the functional groups that are essential to biochar's redox potential (reduction/oxidation). As was previously mentioned, the chemical structure of biochar changes to become highly aromatic as the pyrolysis temperature rises, increasing the _ electron availability. These enable speedy electron exchanges between the charcoal structure and the nearby metal [108]. Since mercury can get methylated if oxidation/reduction potential rises, the reducing potential may also be problematic [109]. Given that Iron (Fe) hydroxides are produced at redox conditions, this is related to Fe reduction [65]. This showed the possibility of a relationship between Iron and Mercury under long-theorized redox conditions [110].

Long-term experiments have shown that using biochar can successfully lower the bioavailability of heavy metals for uptake by plants. In rice fields, biochar treatments reduced cadmium (Cd) buildup in rice after one season, with the highest outcomes occurring at an application rate of 40 tons/ha over the course of three growing seasons, or two years [111]. One application of biochar resulted in an upward trend in accumulation throughout the course of the growing seasons, demonstrating that numerous treatments are likely required to retain healthy levels in crops [111]. Similar results were observed when 3 tons/ha of biochar were applied to fields growing Pak choi (Chinese cabbage), which decreased Cd and Pb concentrations below allowable levels for eating in their edible sections [84].

According to Wang et al. [65], the addition of 5% rice-straw biochar and 5% alkali-modified rice-straw biochar lowered the quantity of available Zn in soil by 28.96 and 36.86%, respectively. The addition of sphagnum biochar significantly decreased the mobility of heavy metals and the bioavailability of Pb, Cu, and Cd in contaminated soil, which was also reduced by 97.8, 100, and 77.2%, respectively [38]. This was accomplished through the coordination of metal electrons to C (double bond, length as m-dash) C (electron) bonds. The Cd buildup in brown rice decreased by 25.1 and 34.6%, respectively, when sulfur-modified biochar and sulfur–iron-modified biochar were introduced to the Cd-contaminated soil [112]. In general, biochar can alter the soil's adsorption capacity and solid–liquid ratio, which can then have an impact on the bioavailability of heavy metals. This may have an impact on the types and distribution of heavy metals in soil. For the reasons indicated above, modified biochar also influences the soil's pH, organic carbon content, and redox property, which further changes the bioavailability of heavy metals in the soil.

5.1 Reduced microorganism toxicity of contaminants

It is be possible to decrease the toxicity of soil contaminants on microbial communities by using biochar. The use of willow biochar produced at 700°C in heavy metal-contaminated soils could reduce the mortality of microorganisms [113, 114]. Through the immobilization of heavy metals such aluminum, cobalt, chromium, manganese, cadmium, and nickel on the pore of biochar [115, 116], it also improved the reproduction of *Folsomia candida* and reduced the leachate toxicity to *Vibrio fischeri*. This reduces soil contaminants and creates ideal circumstances for the development of both plants and microorganisms [117]. The application of rice straw biochar resulted in up to a 68% reduction in the levels cadmium, lead, of zinc, and copper. *Bradyrhizobium japonicum*, a bacterium that fixes nitrogen for plant growth, is no longer under heavy metal stress as a result [118]. Additionally, reducing the stress caused by the heavy metal on microorganisms enhances the synergy between biochar and soil microbial populations, which has a significant impact on the soil's fertility.

5.2 Changing microbiological habitats

By enhancing the physical characteristics of the soil, biochar indirectly alters the microbial habitats. It reduces bulk density, improves soil aeration, and regulates microbial movement. Biochar boosts food availability to the microbial cells and promotes water retention [119]. Additionally, it guards against dry-wet cycles that are harmful to microbial activity and occur in the natural ecosystem [38]. Additionally, biochar alters the acidity level of soil. In contrast to chemical factors like carbon, electrical conductivity and nitrogen concentration, a small adjustment in pH (0.2–0.3 units) can have a significant impact on the soil microbiota. Bacteria are more prevalent in soil when heavy metals (like aluminum) are reduced and the pH is raised [120]. The strength and direction of numerous biochemical processes in soils are reflected by soil enzyme activity, which is the closest agent of organic matter degradation and an indicator of intense microbial activity [121]. In terms of microbial activity and community structure in soil, biochar has been quite important [9]. To mineralize phosphorus from nucleic acids, phospholipids and other ester phosphates, plants have the ability to manufacture and release soil phosphatases [121]. After 7 weeks, the treatment with biochar significantly increased acid phosphatase activity by 27.85% when compared to the control therapy. Thus, size of the biochar particles boosted the soil phosphatase activity [122]. According to Pandey et al. [123], the addition of biochar boosted dehydrogenase activity by 7.4–39% and urease activity by 27% when compared to the control group. According to Jia et al. [124] and Pandey et al. [123], applying various types and amounts of biochar activated soil enzymes (such as urea, invertase, and dehydrogenase). According to Song et al. [125], the addition of biochar increased protease, urease, alkaline phosphatase, catalase, and sucrase by 8.4, 13.9, 81.3, 21.7, and 150.5%, respectively. Recent research by Liao et al. [74] demonstrated that the temperature variations of the pyrolysis process may have some control over how biochar affects the activities of soil nutrient acquisition of enzymes. Generally speaking, reduced temperature biochar aids to activate soil enzymes. The 500°C-derived high-temperature biochar's impact on soil enzyme activity, however, was minimal. Additionally, it was discovered that the properties of the soil and biochar only had a minor impact on how biochar affected enzyme activity. The characteristics of the biochar and the types of soil have a direct impact on the enzyme activity of the soil [74]. Therefore, the foundation of increasing soil enzyme activity is choosing the right biochar based on the qualities of the soil.

Therefore, using biochar in soil may be a viable way to improve nutrient usage effectiveness and, in turn, promote sustainable agricultural production. In order to increase the effectiveness of nutrient usage and ensure sustainable agricultural production, biochar may be a possible soil additive.

5.3 Future prospects of biochar in agro-environment

A lesser cost-effective, efficient way to increase the soil nutrient status, soil carbon sequestration soil quality, soil and water remediation and greenhouse gas reduction, biochar has significant promise in the agricultural and environmental industries. It is still essential to regard each biochar as distinct, characterize it, and test it for crop-soil management practices when using it in on crops because the positive depends on the type of biochar, soil conditions, management practices, etc. Technology advancements, notably in nanotechnology, have made it achievable to produce nano biochar with greatly increased specific surface area and reactive potential, which has significantly increased the carbonaceous material's effectiveness in the field over a very short period of time. The smaller size of biochar, dropped to a nanoscale, could offer significant potential for environmental remediation due to the tremendous increase in specific surface area for adsorption and chemical reactivity of pollutants in the soil as well as water systems. Further research is needed to determine whether this type of nano-biochar will be effective for soil carbon sequestration and greenhouse gas mitigation, which are frequently fueled by more refractory carbon in biochar. Over time, significant advancement has been made in the use of biochar as a low-cost, carbon-neutral means of reducing pollution in soil and water systems by harnessing its unique adsorption capacity. This promise could be further realized with the development of technology, such as nanotechnology, by comprehending how specially created biochar interacts with metal(loid)s, microplastics, organic pollutants, etc., from the water, soil and gaseous medium. It may be essential to create a specific methodology and contaminant capturing/filtering system before using universal biochar to treat wastewater from municipal trash, water sources, and landfills. Biochar can help reduce environmental dangers, consequently lowering threats to human health, as the manufacturing and usage of dangerous chemicals is projected to increase globally. By estimating the carbon sequestration capacity of various biochars and how it interacts with net greenhouse gas emissions, the potential for use of biochar as a soil supplement could be significantly boosted [11, 12]. In the future, a feasible alternative may involve activating biochar with fertilizers to improve nutrient use efficiency and utilizing its capacity to mitigate climate change in a coordinated manner.

6. Conclusion

Almost any carbon (cellulose-lignin) feedstock, such as byproducts from industrial waste, agriculture, livestock farming forestry etc., can be converted into the highly calcitrant carbonaceous material known as biochar through pyrolysis under oxygen-limited conditions. As a result, biochar is a potentially affordable, carbon resource with a wide range of environmental and agricultural applications. The most important applications for biochar are as a soil amendment, which improves soil's physical and biochemical properties, raises soil fertility, and increases soil production, especially over time. It may also increase water retention, soil aggregation, microbial activity and pH, thereby enhancing the general quality of the soil. Even yet,

there is significant diversity based on the type of biochar and soil-crop environment parameters, with the effect of biochar being obviously apparent in a soil of coarse-texture. Chemical fertilizers and organic sources like manure and compost can be supplemented with biochar; however, the best application technique to obtain the most synergistic benefit is still unknown. The unique adsorption rate and chemical properties of biochar, which can assist to capture and immobilize contaminants like organic pollutants, metal(loid)s and hazardous emerging contaminants like microplastics, have the potential to significantly improve the soil and water quality. In more recent times, biochar has been improved through activation with nutrients, metals, etc., resulting in a grade of biochar suitable for particular uses, even permitting the treatment of pollutants in hazardous sites and landfills. Long-term research studies, the synthesis of cross-site mechanistic experiments using different systems and types of biochar, the assessment of the life cycle of biochar in soil–plant environments, the quantification of biochar persistence and soil carbon sequestration potential, the validation of net greenhouse gas emissions spanning different biochar-management systems, the economic analysis of biochar amendment and application, and the potential integration of carbon markets should be put into cognizance. Due to the presence of the functional group, the increased pH from the mineral content, and the greater volatile matter, biochars have higher CEC. These attributes aid in the adsorption and precipitation of metals, which lowers their availability in soils. Biochar has the potential to be effective in the remediation of organic and heavy metal pollutants. It might also be used in conjunction with bioremediation methods to potentially minimize heavy metals through plants that accumulate heavy metals. Before biochar can be produced and modified at a price that is affordable to farmers all over the world, more needs to be done to lower costs. This is crucial because altering biochar can have a significant impact on its ability to bind or stabilize pollutants. Biochar manufacturing is a product that many farmers ought to take into consideration as a remediation choice because it may turn waste streams into a beneficial resource. Given that cost is the primary issue for the majority of farmers, producers of biochar should look at alternatives to cut costs if they earnestly desire to record significant sales of the product and distributed extensively.

Conflict of interest

The authors declare no conflict of interest.

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Chapter 5

Bio-Char as an Adsorbent for Wastewater Purification

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Abstract

This study was conducted to investigate the feasibility of use of olivepomace residues and citrus tree wood residues after burning at 550°(olivepomace charcoal and citrus charcoal) as alternative non-traditional substrates in vertical flow wetland systems (VFCWs) for removing inorganic pollutants and organic pathogens from municipal wastewater through secondary treatment for carbon emission reduction to combat climate change. The effectiveness of this treatment was examined by two pilot scale vertical flow constructed wetlands (VFCWs) systems with alternative substrates. Each system was designed from two operated units in series (two stages of treatment), each unit was manufactured with the same design and size. The difference between each system was the substrates used for treatment. The first system included olivepomace charcoal, while the second system included olivepomace charcoal with citrus charcoal. Both models were operated at the same time and under the same conditions. Both systems were tested with seven different hydraulic retention time (HRT) (12-24-36-48-96-144-192 hours). After conducting laboratory tests on wastewater samples after treatment for several biological, physical, and chemical tests, the results indicated that citrus charcoal and olive charcoal are effective alternative substrates in constructed wetland systems. The systemic way test results showed the lowest removal efficiency for TSS, BOD, COD, TP, TKN, and Fecal Coliform (95, 53, 44, 52, 40, and 66% at 12 hr), while the highest removal rates (97, 94, 94, 80, 69, and 98% at 12 hr), respectively in Model 1. And minimum removal rates for TSS, BOD, COD, TP, TKN and Fecal Coliform (95, 34, 22, 42, 48, and 50% at 12 hr), respectively, while the optimal removal (97, 98, 98, 71, 71, and 99% at 12 hr), respectively, in Model 2. This study proved that olive solid waste and tree wood residues are effective alternative substrates in removing pollutants from wastewater, which are inexpensive and environmentally friendly.

Keywords: charcoal, wastewater, purification, carbon emission, climate change

1. Introduction

Waste management is a global priority issue for the sustainability and preservation of natural resources and ecosystems. Recent studies are moving toward the management and treatment of organic food waste and their beneficial reuse, such as using them for the concurrent recovery of resource, water, and energy to facilitate civilizational sustainability and reduce environmental pollution to combat climate change [1].

In general, there are many environmental issues that need to be addressed and well managed to reduce the severity of environmental pollution, perhaps the most important of which is the issue of exacerbating the quantities of untreated wastewater, and these quantities increase with the increase in population numbers as a result of the increasing demand for various human activities such as drinking, domestic, agricultural, and industrial purposes. This leads to an increase in the amount of wastewater. Only 20% of wastewater overall gets satisfactory treatment and in low-income countries as low as 8% of the treatment capacity [2]. And the remainder is discharged without treatment or partially treatment, causing environmental damage and damage to public health. Every year, at least 1.8 million children under 5 years die from water-related diseases, according to a report published by the World Health Organization, due to unmanaged wastewater [3].

Moreover, there are increasing issues related to the aquatic environment, including water degradation, water shortage, and water pollution [4], and this issue has become a growing concern, in all countries, especially the countries of the Mediterranean region, which poses a threat to several sectors, the most important of which is the agricultural sector, which represents an important economic sector in many countries. Therefore, effective wastewater treatment and reuse for agricultural irrigation purposes can contribute to preserving potable water resources, overcoming the water shortage problem, and reducing the negative effects of wastewater associated with the release of liquid waste into the environment [5].

Wastewater treatment is both necessary and complex due to its complex composition [6]. Solids, organic matter, nitrogen, and phosphorus are general components of wastewater generated by urban communities [7, 8]. In addition, wastewater has pharmaceutical residues containing non-biodegradable toxic compounds and pathogenic microorganisms, which must be removed prior to final release to minimize environmental contamination and associated risks to human health [9, 10]. Historically, traditional centralized physico-chemical and biological technologies to sewage treatment have been used successfully for water pollution control in most countries [11, 12]. However, these wastewater treatment technologies such as membrane separation, membrane bioreactors, and activated sludge process are cost and not suitable for rural areas [13]. Furthermore, they are limited spread and insufficient when facing ever more stringent water and wastewater treatment standards [14]. Thus, selecting low-cost and efficient alternative technologies for wastewater treatment [15] is significant specially in developing regions [16].

Recently, the focus has been on developing systems and finding alternatives to reduce nutrient problems at its source of origin at low cost [17]. So the researchers have moved to using biological and physical remediation methods that are cost effective and cause no harm to the environment instead of chemical methods. Compared to conventional wastewater treatment technologies, constructed wetlands (CWs) are important alternative for decentralized wastewater treatment to solve water scarcity [18], in addition to its economical, environmentally friendly, and sustainable engineering systems due to their low cost, simple operation, and low maintenance [19–23]. CWs have used in worldwide purpose to improve water quality and reduce nutrients and pollutants in wastewater and maintain the diversity of the ecosystem [24, 25], without losing their ecological integrity or providing additional risks for global warming [4]. It is an energy efficient on-site technology that could be employed for wastewater polishing in urban areas with high population density due to small area requirement [26, 27].

This study combines the use of constructed wetlands as unconventional method and natural waste from solid waste as filler substrates instead of the usual substrates in constructed wetlands such as olive and plant wood waste to improve the efficiency of municipal wastewater treatment by integrating physical, chemical, and biological processes as an alternative to conventional treatment facility that requires high operating costs and energy.

Globally, olive oil industry is one of the most important industries in the agro-industrial sector in Mediterranean countries [28]. It is produced almost entirely in the Mediterranean region, especially in the State of Palestine, which covers most of its agricultural area with olive trees, approximately 57%, and relies mainly on industries based on olive oil [29]. Thus, the problem of environmental pollution from the waste of olive mills (OMW) increases in these producing regions, due to the increasing demand for olive oil all over the world.

Olive mills produce dark-colored wastes that contain liquid and solid wastes and a large amount of organic matter. It is composed of many complex materials that cannot be easily degraded. Production facilities are often associated with emissions of highly odorous volatile compounds. When their waste is dumped into the environment, it creates an odor nuisance, as it poses another threat to surface water and the environment when dumped without proper treatment and management, including changes in phytotoxicity, soil quality, natural water coloring, ground and surface water pollution that causes toxicity to aquatic animal life, and irritating odors, due to yeast treatment, paper, organic chemistry, winery, lignin, cellulose, hemicellulose, lipids, protein mill residues, and olive oil from olive pressing. It has a high chemical oxygen demand (COD >110 g/L), low pH and biological oxygen demand (BOD > 170 g/L), and a high organic toxicity load [28]. The main problem with olive mills waste disposal is to find an environmentally friendly and economically viable solution. In the biochemical treatment of wastewater from olive oil factories, capital and operating cost units must be installed with limited efficiencies due to high organic loads and chemical oxygen demand (COD) to biological oxygen demand (BOD) ratios. Since there are toxic organic materials that mostly come from the broken seeds, this waste is toxic to bacteria and direct biological treatment is not possible.

Therefore, olive oil-producing countries face a challenge to find feasible, economical, and sound ways to dispose of solid and liquid waste resulting from this industry. So, the solid waste from olive mills, especially the broken seeds in wastewater treatment, will be more useful and will add positive advantages to reduce the harmful effects and threats to the environment.

In this experiment, the wetland technique was used, built in an unconventional way, using natural substrates from the solid waste of olive presses. In this way, we can overcome several environmental problems in a smart, integrated, environmentally friendly way. Where the municipal wastewater is purified and treated using natural wastes from olive mills, and thus reducing the wastage of municipal wastewater, the problem of water scarcity will be overcome by providing an alternative source for irrigation and other purposes. Also, the quantities of waste from the olive mills will be disposed of properly instead of being exposed to the environment, which avoids damage to the soil, water, and air. Along with that, in this way built wetland technology is developed by the use of new natural substrates that are low in cost and operation. Thus, the desired goal of sustainability is achieved by simultaneously returning resources, water, and energy to reduce environmental pollution to combat climate change.

2. Materials and methods

2.1 Experimental setup

Two constructed wetland (VFCWs) models (namely model 1 and model 2) were built outdoors under prevailing environmental conditions, inside Sheikh Ajlin wastewater treatment plant (31°485,444′ N, 34°424,678′ E), Gaza, Palestine. This plant treats municipal wastewater using filtration basins as a first stage and aerobic, anaerobic treatment as a second stage. For the construction of the wetlands, work such as cleaning and extraction of the weeds from the site was initially carried out. Later, the excavation was executed, where drilling was carried out on several heights to put the wastewater tank and experiment basins consisting of two stages to transfer water by different levels through gravity without the need for operational energy.

2.2 Preliminary laboratory tests

Some analyses and preparations were carried out in the laboratory and on the field before the implementation of the experiment on the ground of Sheikh Ajleen Wastewater Treatment Plant, including: burning olivepomace and citrus plant wood waste at a temperature of 550° in isolation from oxygen, cracking citrus plant wood coal after burning to small sizes, substrate porosity calculation, sieve analysis for substrates, and substrate hydraulic calculation (sand, gravel, citrus plant wood coal, and olivepomace charcoal) to determine the suitability of the filter medium used in experiment models, where the size of the substrates was chosen from 1 mm to 40 mm with filter average porosity of 40%. The *Phragmites australis* plant was chosen to achieve the objective of the treatment and for growing them inside the Gaza Strip, 80 seedlings were transplanted and planted inside a basin at the Sheikh Ajleen plant before the wetlands were built and let to grow.

2.3 Constructed wetland setup

Two experimental models were built, each of which has two units (vertical flow sub-surface constructed wetlands). Each unit was made of strong wood covered on all sides with a layer of plastic liners mad of PVC with a thickness of 2.0 mm to prevent any leaks down, and this type of liner is available in local markets and its price is acceptable, which also has high resistance to the chemicals found in wastewater, and it is characterized by high flexibility when there are seasonal changes such as temperature changes [30]. All units had the same volume = 0.80 m × 0.40 m × 0.70 m, aspect ratio (2:1) with 45% average porosity, calculated in the laboratory using liquid saturating method and 1000–10,000 $m^2/m^2/d$ hydraulic conductivity, calculated in the laboratory using constant head method. Generally, the porosity should be used ranges from (0.4:1) to (3:1), which ensures high porosity and a high hydraulic conductivity to provide even influent liquid distribution and efficient infiltration, and facilitate and improve plant growth and collection of influent liquid.

In this experiment, the *Phragmites australis* plant was used in the eight basins. Their roots extend to depth of 0.7 m [31]. (Accordingly, the depth of each basin was 0.7 m, to take into account the growth of roots well and the occurrence of biological interactions between the liquid and the root.) In CWs, it is important achieve a slope in the lower layers of the bed to achieve a horizontal flow along the bed [32]. In general, the

preferable hydraulic gradient S < 10% of maximum potential gives the adaptability and the reserve capacity for future operational changes [30, 31]. For all basins in this experiment, the bed slope is designed with 5% slope as shown in **Table 1**.

The basins were placed with a slope to facilitate the flow of wastewater whereas transport pipes of 50 mm diameter were installed from the wastewater feeding tank to the other upper four VFSCWs. In each unit, two plastic vertical pipes were installed, 1 pipe used for every 4 inch to increase the amount of oxygen in the wetlands [33].

Experimental models were arranged to form two parallel model trains: models 1 and 2. Each model included VF sub-surface constructed wetland unit as the first stage (A), it represents the first stage of treatment, and another VF sub-surface constructed wetland unit as the second (B) stage, it represents the last stage of treatment, as shown in **Figure 1**.

The aim of this study was remove organic pollutants presented in post-treated wastewater by using CWs, so the kinetic model was used to estimate the concentration of BOD(Co) after treatment based on the general form of Eq. (1) [34].

$$\frac{C_e}{C_o} = e^{(-KTt)} \tag{1}$$

Where,

C_e = effluent BOD5 (mg/l), C_o = influent BOD5 (mg/l), K_T = temperature dependent rate constant (d^{-1}), t = hydraulic retention time (d).

$$K_T = K_{20}(\theta)^{(T-20°)} \tag{2}$$

Where,

K_{20} = temperature coefficient for rate constant, T = temperature of liquid in the wetland (°C).

K_{20} equals 1.104 (*d-1*), θ =Arrhenius coefficients used were 1.06 for BOD.

To calculate the surface area required the first-order BOD removal equation was used proposed by Kickuth [34], based on Eq. (3).

Parameter	Value	Unit
Total depth	0.70	m
Long-wide (aspect ratio)	2:1	
Long	0.80	m
Width	0.40	m
Surface area	0.32	m^2
Total volume	0.224	m^3
Slope	5	%
Depth of filter medium	0.60	m
Filter average porosity	0.45	%
Water depth in filter medium	0.55	m

Table 1.
Data of the design parameter to vertical constructed wetlands.

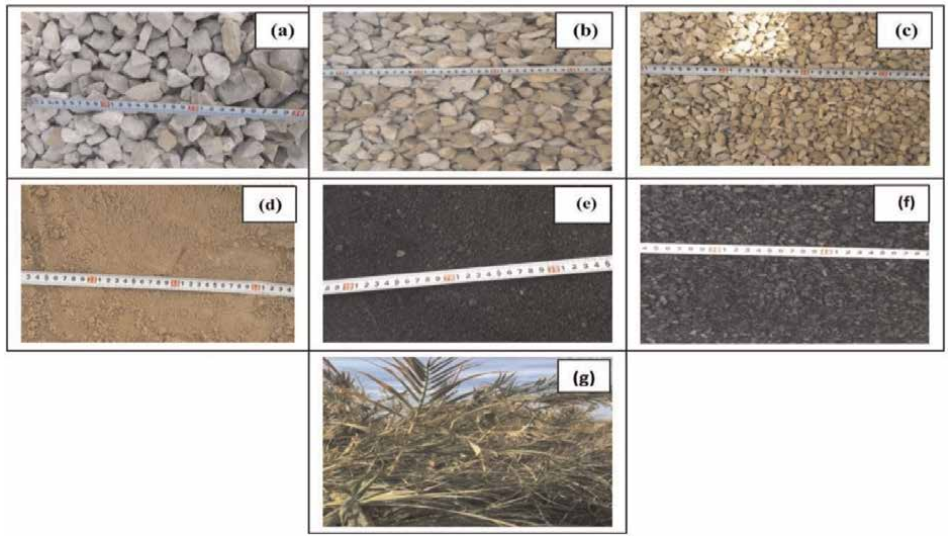

Figure 1.
Wetlands built to scale. (a) Coarse gravel, (b) medium gravel, (c) fine gravel, (d) loam sand, (e) citrus charcoal (f) olive pomace charcoal, (g) Phragmites australis.

$$As = (L)(W)\frac{Qd\,[\ln(\mathrm{Co}/\mathrm{Ce}])}{KBOD} \tag{3}$$

Where,

Ah = Surface area of bed (m^2), Qd = average daily flow rate of sewage (m^3/d), KBOD = rate constant (m/d), KBOD is determined from the expression Ktdn, where, KT = K20 (1.06) (T-20°), K20 = rate constant at 20°C (d-1), d = depth of water column (m), n = porosity of the substrate medium (percentage expressed as fraction).

Based upon Eq. (3), the required surface area (*As*) was 0.32 m^2 assuming the aspect ratio (L:W) is 2:1, so the length is 0.8 m and width is 0.4 m, the depth of the filtration media is 0.6 m, while the depth of water column is 0.55 m to maintain a level of 0.05 m under the filtration media as mentioned previously. Therefore, the total volume arithmetic equal is 0.224 m^3, and the total net volume arithmetic is 0.176 m^3, for each basin.

2.3.1 Filtration media

The types and sizes of the substrates were carefully chosen in this experiment to achieve the greatest degree of removal. Sandy loam, gravel, and coal were used, away from using the clay due to the low hydraulic conductivity, and working on the recommendations of previous studies such as unifying the size of the substrates, entry and exit areas using large filter media to prevent blockage, and accumulation ranging between 40 and 80 mm [35]. On the contrary, in the upper layer, a large size of media is not preferred used as it is not suitable for plant root propagation, but for the vegetable layer, it is preferred to use sand with gravel to filter and facilitate the plant growth process [36], and use gravel at the inlet and outlet to help prevent blockage [36]. This experimental basins were packed with biological and construction materials

that are gravel, sand, citrus charcoal, and olive pomace charcoal. Module 1 is the same as the module 2, but was replaced by the fourth layer with olive pomace charcoal substrate in module 1, and mix olivepomace charcoal and citrus charcoal substrate in module 2.

Coarse gravels (30–40 mm) were used as supporting layer at the bottom of basins, of 15 cm thick, as shown in **Figure 1(a)**, and this size was chosen because there was a valve in the bottom of each basin, which was used for controlling the flow in the module, so it was chosen larger than the entrance of valve (which was ½ inch), so the valve would not be clogged by small aggregates and facilitate effluent discharge. The second layer was 7.5 cm thick and filled with medium gravel (20 mm), as shown in **Figure 1(b)**, the third layer was 7.5 cm thick and filled with fine gravel (10 mm), as shown in **Figure 1(c)**, the fourth layer was 15 cm thick filled with olive pomace charcoal (1–19 mm) in the first model, as shown in **Figure 1(f)**; the second module used a mixture layer of citrus wood waste charcoal and olive pomace charcoal in equal proportions (1–19 mm), the fifth layer was 15 cm thick having a mixture of loam sand and small gravel at the ratio of 1:3 (aggregates: loam sand, respectively) as shown in **Figure 1(d)**, to support the growth of plants, and the upper part was empty of about 10 cm to allow aeration for the module and the roots of the plants. The common *Phragmites australis* plant was planted in all basins for the two models. Its roots extend to a depth of 0.7 m [4], so the depth of the substrate (0.7 m) was studied taking into consideration the greater root depth of the *Phragmites australis* plant. After planting the *Phragmites australis* plants in basins, they were fed with water for 20 weeks for growth and maturation. Substrates including sand, gravels, olivepomace charcoal, and citrus wood waste charcoal firstly were washed with tap water to remove ashes and then filled into the experience basins to a height of 60 cm. And in general each basin had an average porosity of 40% with an average working volume of 75 L, which is high porosity and a high hydraulic conductivity to provide efficient infiltration.

2.3.2 Inlet structure and outlet structure

Inlet structures play an important role in treatment wetlands by aiding effective flow distribution across the full width of wetland bed. In the four experimental basins, the main pipe of the feeding tank was used, which was 3 m length and 32 mm diameter. And four separate inlet pipes for each basin with the same length and diameter of 20 mm and 80 cm length were placed horizontally above the basins to allow the liquid to descend down through the unit layers evenly, each open pipe with equal openings and distance from each other at an equal distance to achieve distribution in all basin directions evenly and each pipe connected with a valve for open and close and controlling the flow liquid speed. While the function of the outlet in wetlands is to collect the effluent liquid after treatment toward the reuse of places or storage, there are several types, sizes, and location selected depending on the design [4].

In this experiment, four outlet pipes of the same shape and size were used with valve for each of them to close and open. The two inlet pipes of the second stage are the same as the outlets for the two basins of the first stage, the other four outlets were an exit for the second stage connecting with two pipes having each of length 2 m and 20 mm diameter with valve; and it was buried under the sand at distance of 2 m away from the module to discharge the wastewater after its treatment to the agricultural area inside the treatment plant.

2.4 Wastewater feeding and flow pattern

The CWs were feed with one type of wastewater pumped from the Gaza City wastewater treatment plant (municipal wastewater secondary treatment using aeration basins) for a total period of 10 months. The first 6 months allowed system stabilization; experimental analyses were conducted within the remaining 4 months.

All CWs in this study were designed as vertical down-flow subsurface flow constructed wetlands, each module had of two units (i.e., two-stage treatment) arranged on top of each other, and the layers of each unit arranged for vertical flow in the bed without a pump. The influent flows from inlet (wastewater feeding tank) toward the first two upper basins (i.e., first treatment stage) and then flows through the porous media under the surface of the beds in a vertical path via gravity (down-flow) until it reaches the outlet of second two lower basins (i.e., second stage and final treatment stage) to complete the treatment process, where it is collected. The flow was continuous at all stages for all trains in parallel, and the flow quantity was controlled using manual valves for each basin.

2.5 Process conditions

2.5.1 Hydraulic loading rate

To achieve the high removal efficiency for raw sweater within constructed wetlands, this requires long hydraulic retention with low hydraulic loading rate [37].

In this experiment, the hydraulic loading rate was determined based on the dimensions of the basins, where the design was for all basins similarly, according to the following Eq. (4):

$$HLR = Q \backslash A \tag{4}$$

Where,

HLR: hydraulic loading rate (m/d), Q: flow (m^3/d), A: surface area of the constructed wetland (m^2).

As mentioned earlier, more than one hydraulic retention time (HRT) was used to test the efficiency of the best, so the hydraulic loading rates were according to the HRT as shown in **Table 2**.

HRT (h)	Flow (m^3/d)	Hydraulic loading rate (m^3/m^2/day)
12	0.150	0.47
24	0.075	0.23
36	0.050	0.16
48	0.038	0.12
96	0.019	0.06
144	0.013	0.04
192	0.009	0.03

Table 2.
Flow rate and hydraulic loading rate for each basin at seven HRTs.

2.5.2 Hydraulic retention time

The mean actual HRT is the average time that the water remains in the CWs. One of the objectives of this study is to evaluate the effect of HRT on CWs contaminant removal efficiency under the same conditions. So seven HRTs are assumed and the samples are collected for each of hydraulic retention time (HRT). These were in the order of HRTs at 12, 24, 36, 48, 96, 144, and 192 h, to study the relationship between retention time and treatment efficiency. Between each HRT, the modules rested for 2 days to allow re-oxygenation of the substrate.

2.6 Sampling and analyses

Wastewater was collected from inlet and outlet across VF wetland models at different hydraulic retention times (HRTs) (12, 24, 36, 48, 144, and 192 h). The HRT was controlled using a digital timer regulator. The effect of hydraulic retention time and substrate bed was studied to find out the best system efficiency with HRT.

The temperature of the treated wastewater was (28 ± 2)°C during the study. The wastewater samples of VSSF – VSSF two systems, including influent and effluent from each system, were taken three times at HRT to monitor the system performance. In each analysis, three samples, two influent samples and one effluent sample, were collected separately.

The total samples for the influent and the effluent were 21 and 21 samples, from each system, respectively, and at each sampling time, triplicate samples were collected for analysis.

The constructed wetland sample units were collected in sterile plastic bottles. The volume sampled was 1 L for each sampling point. The sampling recipients were 1-L plastic bottles for standard parameters and sterilized 50-mL glass bottles for Fecal Coliforms. It was collected in glass bottles, and then mixed to form a composite sample and stored at 4°C. The most important challenge that was at the beginning of the experiment was to collect the substrates that were used, burned, and cracked to the required sizes when collecting samples manually.

Collected samples were analyzed immediately at the laboratory of Institute of Water and Environment and Chemistry Laboratory at Al-Azhar University Gaza and Department of Environmental and Sciences at the Islamic University Gaza.

The analyzed parameters included: PH, turbidity, electrical conductivity (EC), total dissolved solid (TDS), total suspended solid (TSS), dissolved oxygen (DO), ammonium nitrogen (NH4-N), phosphate (PO_4), nitrate nitrogen (NO_3-N), total nitrogen (TN), total phosphorus (TP), ammonia (NH_3), 5 days biochemical oxygen demand (BOD5), chemical oxygen demand (COD), chloride (Cl), calcium (Ca), magnesium (Mg), hardness, phosphate (PO_4), sodium (Na), potassium (K), total Kjeldahl nitrogen (TKN) and Fecal Coliform. The testing methodology was done according to the standard methods stated in the Examination of Water and Wastewater book edited by ANDREW D. EATON, LENORE S. CLESCERI, ARNOLD E GREENBERG, 1995 [38].

3. Results and discussion

3.1 Total suspended solid (TSS) removal

The total suspended solids (TSS) are the dry weight of suspended particles that are not dissolved, in a sample of water that can be trapped by a filter that is analyzed

using a filtration apparatus. Water quality parameter is used to assess the quality of a specimen of any type of water or water body, ocean water for example, or wastewater after treatment. TSS includes the organic matter (volatile) and inorganic matter (fixed) [39]. There are several mechanisms to remove suspended solids in the water including flocculation, interception, adhesion, straining, impaction, and filtration [40]. This test is a good measure of microbial contamination and turbidity to assess wastewater quality after treatment [41].

Table 3 shows the measured influent and effluent concentrations and removal efficiency for TSS. The concentration of suspended materials from the influent was few, due to feeding of the constructed wetland modules from the secondary treatment unit, where the wastewater underwent primary and secondary treatments (secondary treatment of municipal wastewater). After entering into constructed wetlands for treatment there was a noticeable decrease in concentration since the first hours in all modules. The removal rates in both systems were almost close to each other. The highest removal percentage of TSS for both systems was 97 and 96%, respectively, at 192 HRTs, and the lowest removal percentage was 95 and 95%, respectively, at 12 h as shown in **Figure 2**.

Mechanism of pollutant removal may change as a function depending on the concentration of pollutants entering, type of substrates, type of vegetation used, and wetland hydrology.

Physical, chemical, and biological factors such as: sedimentation, filtration, interception, settling, adsorption, and sagging all affect the bed layers of the systems and form the basis for the removal of suspended solids [42].

In this study, there was an effective removal of suspended solids in all modules. One of the essential mechanisms in the removal of suspended solids in these modules is the sedimentation and filtration of particulates, due to the arrangement of the layers, the type of substrates, and the flow velocity used in this experiment, this is consistent with the studies [43, 44].

Also, plants played a role in the removal of TSS in all modules. Where all the modules were planted along *Phragmites australis* plant, this plant needs temperature and a sufficient amount of oxygen to carry out the photosynthesis process for good

	HRT (hr)	**Average influent (mg\l)**	**Load In gm^{-2} d^{-1}**	**Average effluent (mg\l)**		**Removal efficiency (%)**		**Load out ggm^{-2} d^{-1}**	
Parameter				**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**
TSS	12	3.9	0.90	0.21	0.21	95	95	0.049	0.049
	24	3.9	0.45	0.17	0.2	96	95	0.020	0.023
	36	3.9	0.30	0.17	0.18	96	95	0.013	0.014
	48	4.3	0.26	0.16	0.2	96	95	0.010	0.012
	96	4.3	0.13	0.16	0.2	96	95	0.005	0.006
	144	4.5	0.09	0.15	0.2	97	96	0.003	0.004
	192	4.5		0.15	0.2	97	97	0.002	0.003

Table 3.
Measured TSS influent and effluent concentrations and input-output loading rate values for the (2 stages VF-CWs) module replicated seven times.

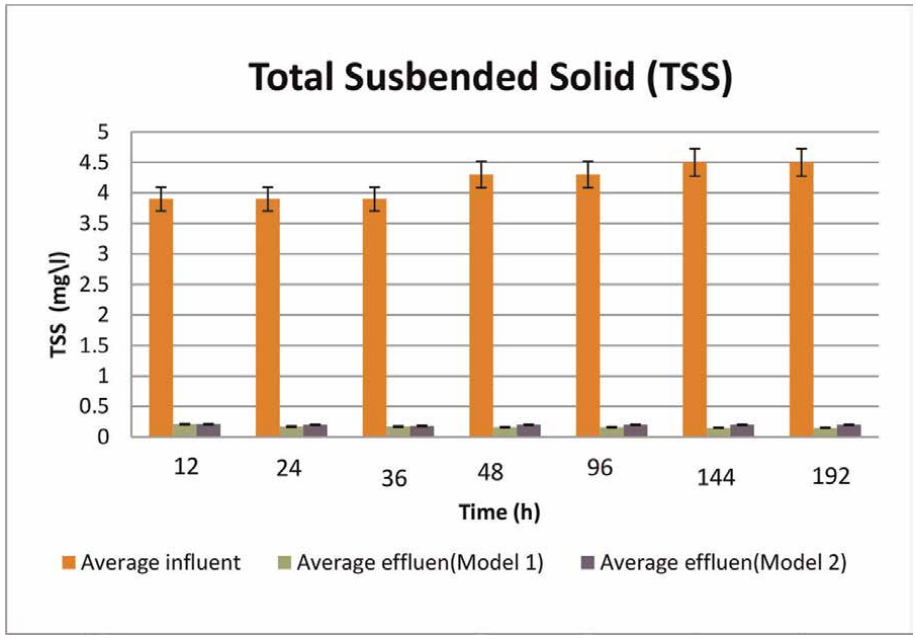

Figure 2.
Measured TSS influent and effluent concentrations during the operational period.

growth and growth of microorganisms. In subsequent steps, the organic matter suspended solids are removed in the wastewater [45] and refractory compounds [46]. As the network of bacteria formed around the roots and on substrate surfaces within the bed, it acts as a strong filtration agent for removal [42]. Another factor that affected the removal of TSS was hydraulic ration time, it was clear that with increasing contact time the removal of the both s modules increased, these results are consistent with it [42].

Generally, the results of this study were more efficient than previous study [47], which used inorganic types of substrates such as gravel and sand; this explains that olive pomace charcoal and citrus charcoal have a higher efficiency than gravel for removal of TSS, perhaps the reason for the increase in the surface of the substrate on the coal seams compared to the surfaces of inorganic materials, especially in the first model that contained one type of substrate, which gives more homogeneity that increases the surface. This means the type of substrate plays an important role in TSS removal, and these results are consistent with it [48].

It was noticed that the input loading values of TSS were decreasing with increase of HRT (**Figure 3**). However, the graph shows that all modules have the same behavior, there was continuous eliminated values of TSS with increasing HRT, the first model recorded the lowest elimination values, while the second module recorded the highest TSS elimination values at all HRTs, as for the third and fourth models, they were close to the results of the second models during the operating period.

3.2 Biological oxygen demand (BOD) removal

Biological oxygen demand (BOD) or another common term is biological oxygen requirement [49]. It measures quantity of oxygen required for microorganisms to oxidize or degrade organic waste present in water. The BOD test is also known as

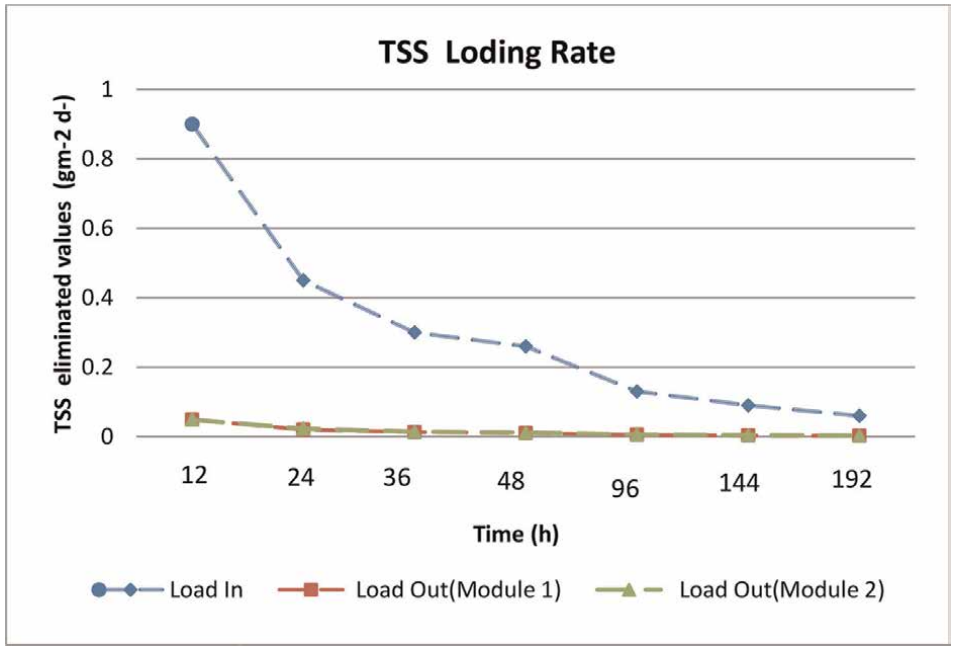

Figure 3.
The input-output loading rate values of TSS through the operational period of the system replicated seven times.

"BOD5" that measures dissolved oxygen during a five-day incubation in dark conditions at 20°C. This test is useful for measuring water and wastewater quality, the unit of measurement mg/l [50].

In this study, average influent concentration during the operation period was 249 mg/l. The results presented in **Table 4** revealed that the constructed wetland noticeably improved the effluent quality of BOD for all models.

	HRT (hr)	Average influent (mg\l)	Load in gm^{-2} d^{-1}mg\l	Average effluent (mg\l)		Removal efficiency (%)		Load out g gm^{-2} d^{-1}	
Parameter				**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**	**Model (2)**	**Model (2)**
BOD	12	275	48.9	131	182	53	34	30.6	42.7
	24	275	21.2	110	125	60	55	12.9	14.7
	36	275	10.2	65	113	76	59	5.1	8.8
	48	220	3.6	28	75	87	66	1.7	4.5
	96	220	0.6	24	27.5	89	88	0.7	0.8
	144	240	0.4	16	14.6	94	94	0.3	0.3
	192	240	0.3	15	5	94	98	0.2	0.07

Table 4.
Measured BOD influent and effluent concentrations a for the (two-stage VF-CWs) module that replicated seven times.

Where the highest effluent BOD concentration value after the treatment was 131 mg/l in the first model at 12 h, the lowest concentration value was 5 mg/l at 192 h in the second module, as shown in **Figure 4**.

The lowest removal rates of BOD were 53 and 34%, respectively, at 12 h, while the highest BOD removal rates were 94 and 98%, respectively, at 192 h. In the both modules, the process of removing pollutants continued increasing with increasing contact time, which gave varying results.

The availability of oxygen in wetlands is the main factor for the decomposition of organic matter in wastewater, where the availability of good porosity in addition to the roots of plants promotes the formation of micro-aerobic and anaerobic environments and thus accelerates the development of microbial communities [51].

The main source of ventilation was in the experimental models due to the high percentage of voids within the layers that are transported by the roots of plants. Where the average porosity was 40%, in all experimental models, all were planted with common reed plants, which promoted decomposition and removal of organic matter.

All models gave high results for removal of BOD, but at different rates. The removal process in all modules was effective with increasing the HRT. Perhaps the reason for the type of substrates used in both models is these results are consistent with it [48].

The curve shows the eliminated values of BOD for four repeated experiments using different types of substrates (**Figure 5**), where the input loading values were decreasing with increase of HRT. However, the graph shows that all modules have the same behavior, and there was continuous eliminated values of BOD with increasing hydraulic retention time. Where the second model achieved the highest eliminated values of BOD at 12 h, while the fourth model, which gave the least eliminated values at 12 h, then it achieved the highest eliminated values at 192 h. As for the second and third modules, it had almost the same behavior.

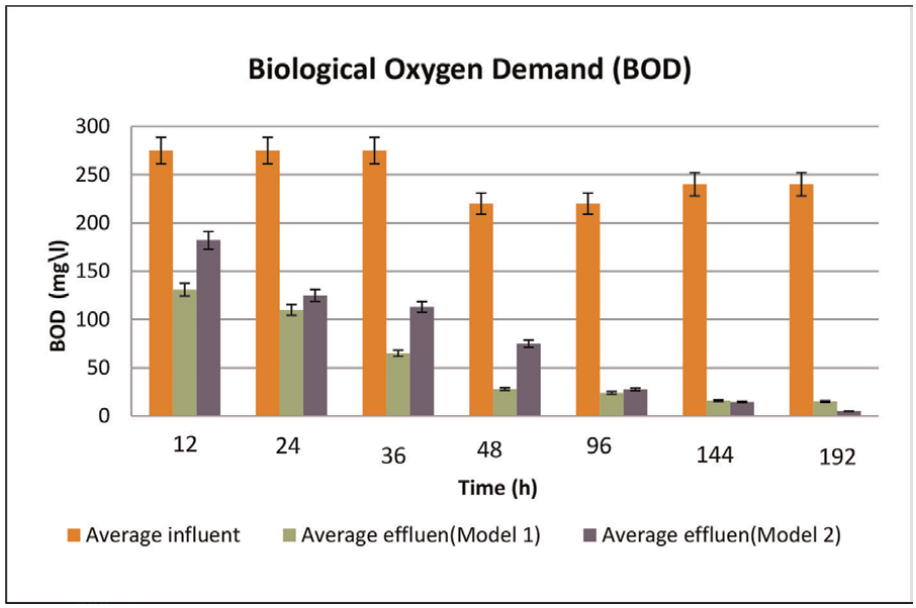

Figure 4.
Measured TSS influent and effluent concentrations during the operational period.

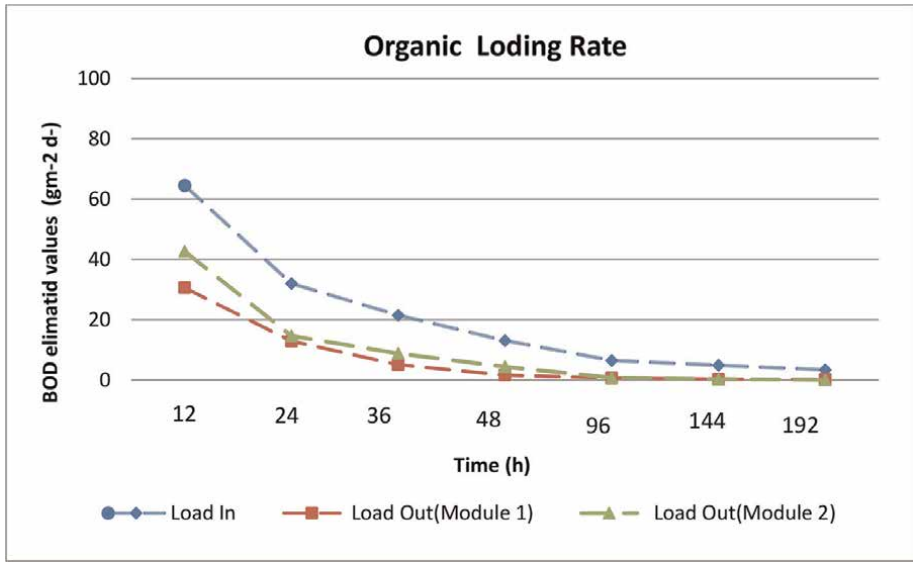

Figure 5.
The input-output loading rate values of TSS through the operational period of the system replicated seven times.

3.3 Chemical oxygen demand (COD) removal

Chemical oxygen demand (COD) is a measure quantity for the amount of oxygen required to oxidize particulate organic matter that is soluble in water [4].

The influent average concentration during the operating period was between 415 and 437 mg/l. Where the highest concentration effluent of COD value was 245 mg\l in the first system at 12 h, the lowest COD value was 10 mg/l at 192 h in the second system, as shown in **Table 5**.

In both models, continuous removal of COD was observed, but the second system achieved slightly higher efficiency than the first system after 96 h, on subsequent increase in the hydraulic retention time, as shown in **Figure 6**. The minimum removal rates in systems 1 and 2 were 44 and 22%, respectively, at 12 HRTs, while the

	HRT (hr)	Average influent (mg\l)	Average effluent (mg\l)		Removal efficiency (%)	
Parameter			**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**
COD	12	437	245	340	44	22
	24	437	205	232	53	50
	36	437	123	211	72	52
	48	415	51	140	88	66
	96	415	44	52	89	88
	144	430	29	26	93	94
	192	430	28	10	94	98

Table 5.
Measured COD influent and effluent concentrations a for the two-stage VF-CWs module replicated seven times.

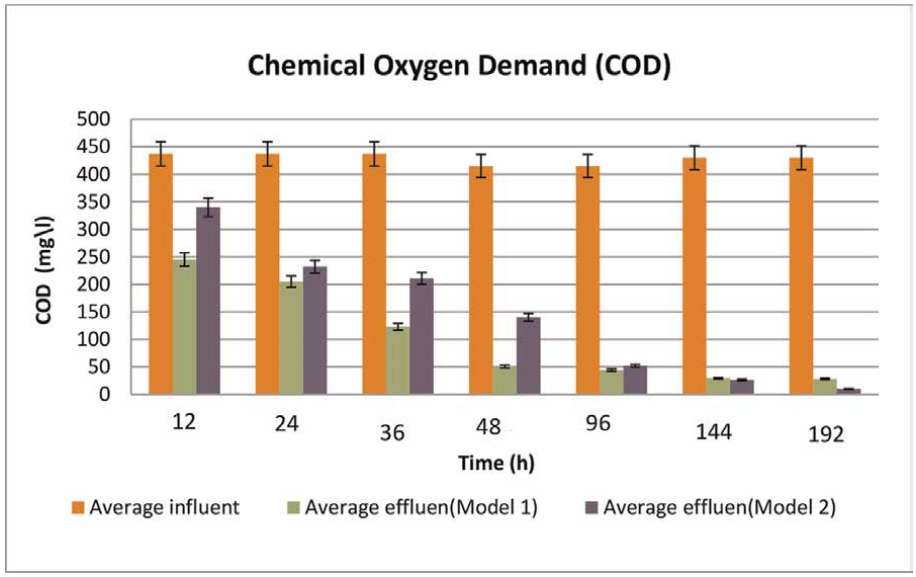

Figure 6.
Measured TSS influent and effluent concentrations during the operational period.

maximum removal rate for both systems reached 94 and 98%, respectively, at 192 HRTs.

The reason for that may be due to the size of the heterogeneous substrates and the interaction between olivepomace charcoal and citrus charcoal. Oxygen-consuming conditions encouraged the development of aerobic microorganisms and helped the degradation of organic matters [52].

Based on the previous results, it is clear that there is a relationship between the biological oxygen demand and chemical oxygen demand represented by the following equation: COD ≈1.8 BOD.

This type of wetland with organic materials gave higher removal efficiency of COD and BOD compared to another study, which used organic and inorganic materials as substrates: sugarcane bagasse, biochar, coal, and oyster shell; rushed mortar, recycled bricks, gravel, and sand [22].

3.4 Total phosphorus (TP) removal

Table 6 shows the removal percentages of TP at different hydraulic retention times. Generally, phosphorus in CWs is removed by plant uptake, microbial immobilization, substrate adsorption and accretions of wetland soils.

In this experiment, the removal rate was nearly halved in both regimens after 12 h (adsorption period) and the removal continued to increase when the HRT increases. Where the minimum removal rates for models 1 and 2 were 52 and 42%, respectively, at 12 HRTs and the maximum removal rate of both systems reached 80 and 71%, respectively, at 192 HRTs, as shown in **Figure 7**. This indicates that the HRT is a crucial parameter for the effective P removal [53]. These results are more efficient than previous study [54], which used inorganic materials such as maifanite, steel slag, bamboo charcoal, and limestone. This indicates that olivepomace charcoal substrates

	HRT (hr)	Average influent (mg\l)	Average effluent (mg\l)		Removal efficiency (%)	
Parameter			Model (1)	Model (2)	Model (1)	Model (2)
TP	12	13	6.3	7.6	52	42
	24	13	6.6	6	49	54
	36	13	5	5.5	62	58
	48	14.7	4.3	5.7	71	61
	96	14.7	3	4.2	80	71
	144	13	2.8	4	79	70
	192	13	2.6	3.8	80	71

Table 6.
Measured TP influent and effluent concentrations a for the two-stage VF-CWs module replicated seven times.

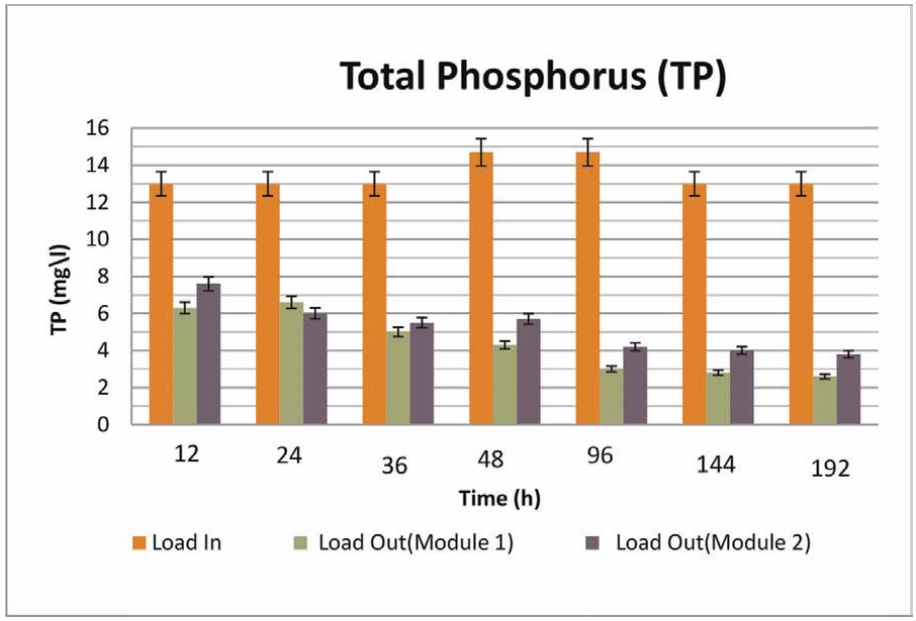

Figure 7.
Measured TP influent and effluent concentrations during the operational period.

are suitable for removing phosphorous, and it is best to use as a layer alone without mixing with other substrates to give higher efficiency.

3.5 Total Kjeldahl nitrogen (TKN) removal

The total Kjeldahl nitrogen (TKN) is defined as the sum of ammonia (NH_4 and NH_3) and organic nitrogen (Org-N) [55]. In wastewater, the presence of organic nitrogen is due to deamination reactions during the metabolism of organic matter [56].

High TKN concentrations were detected in the influent throughout the experimental during the operating period; the average influent concentration was 216 mg/l.

Table 7 shows the results of total Kjeldahl nitrogen (TKN) analyses for all models during the operating period. It is clear that the lowest removal rates for the both models were 40 and 48%, respectively, at 12 of HRT, while the highest removal rates for the modules were 69 and 70%, respectively, at 192 of HRT. The results showed that the TKN removal rates gradually increased with increasing contact time for all models without exception.

The minimum value were in model 2 (concentration 62 mg/l - removal efficiency 71% at HRT 192 h) and the maximum value were in model 1 (concentration 137 mg/l - removal efficiency 40% at HRT 12 h) as shown in **Figure 8**. Hence, it is possible to

	HRT (hr)	Average influent (mg\l)	Load in g gm^{-2} d^{-1} gm^{-2} d^{-1}	Average effluent (mg\l)		Removal efficiency (%)		Load out g gm^{-2} d^{-1}	
Parameter				**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**
TKN	12	230	53.9	137	120	40	48	32.1	28.1
	24	230	26.9	120	109	48	53	14.1	12.8
	36	230	17.9	112	93	51	60	8.8	7.3
	48	200	11.9	96	74	52	63	5.7	4.4
	96	200	5.9	84	66	58	67	2.5	1.9
	144	210	4.3	78	64	63	70	1.6	1.3
	192	210	2.9	65	62	69	71	0.9	0.8

Table 7.
Measured TKN influent and effluent concentrations and input-output loading rate values for the two-stages VF-CWs module replicated seven times.

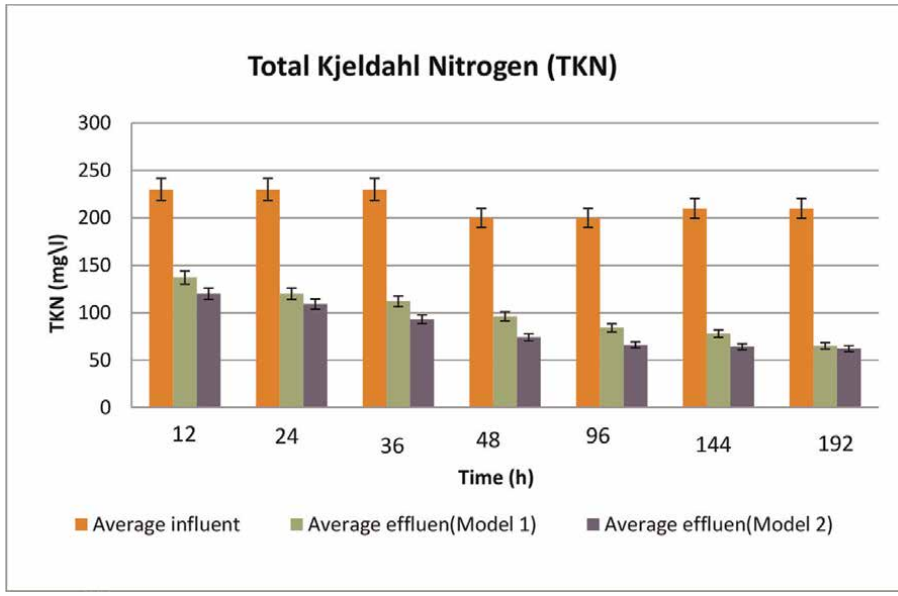

Figure 8.
Measured TKN influent and effluent concentrations during the operational period.

notice that, HRT and material are factors that possibly influence TKN removal in CWs.

In general, the process of removing nitrogenous compounds depends on the aerobic and anaerobic conditions provided by layers and plant roots [57]. Where the plant contains biomass in the lower area of "stems and roots" and the upper region "leaves", where it can be grown either subsurface horizontally or vertically and create enormous networks with the bed particles and soil; and thus form a suitable surface to absorb nutrients and ions [45, 47]. Also, increased nitrification and denitrification result from plant respiration provides aerobic conditions and develops microorganisms at the root [58].

The removal rates in wetlands vary from plant to other, wherein the study [59], four types of plants were used to remove nutrients: *Typhalatifolia, Phragmites australis, Scirpus (Bulrush), and Alismaplantago*, the results indicated that the systems constructed with *Typhalatifolia*, *Phragmites australis* plant, which gave a higher efficiency in removing nitrogen and phosphorous compared to other plants used. In this experiment, *Phragmites australis* plant was used, as the results showed that almost half of concentrations were removal in all modules. To obtain better removal rates, continuous ventilation unit must be provided with the plant in modules, in this experiment, the porosity of the layers was the main factor for ventilation without the presence of a continuous ventilation unit, so the lack of an essential continuous source of aeration in the modules gave these results because the limited oxygen reduces the removal efficiency, especially in the early stages of operation. With the passage of time and plant growth, the efficiency of TKN removal will improve [60].

Other factors contributed to the removal of TKN, including the temperature. The temperature of wastewater in this experiment was moderate between 27 and 30°C, as the temperature affects the rate of nitrification, which is able to promote plant growth, release of root secretions, and microbial activity [56]. Thus, the rate of nutrient uptake and the rate of denitrification are accelerated, as nitrification reaches a maximum rate at temperatures between 30 and 35°C [61]. Also, the organic materials used played a role in the removal of TKN, where the presence of ammonia and organic carbon leads to the adsorption of microorganisms and the decomposition of the pollutants in addition to changing oxidation conditions due to the diversity and abundance of the microbial community. As it helps remaining carbon source, it can be used as an electron donor in the denitrification process. These results are consistent with the study [48].

In addition, in all models there was an increase in removal with increase of HRT, as shown in **Figure 9**. This corresponds to some studies that indicated HRT is one of the factors affecting nitrogen removal. Moreover, the author studies found a decrease tendency on the TKN removal during 1 year and a half of operation, this is a consequence of the vegetative cycle of the plants, since their absorption capacity decreases as the plants age [58]. It was observed in this experiment that there was a slight discrepancy in the removal rates between the first and second systems, where the second system, which contained organic substrates mix of olivepomace and citrus charcoal, gave slightly higher results than the first system that contained citrus charcoal only. The explanation may be due to interaction between the different substrates.

3.6 Fecal Coliform (CFU/100 ml) removal

A pathogen in the oldest and broadest sense is any organism that can produce disease. A pathogen may also be referred to as an infectious agent, or simply a germ.

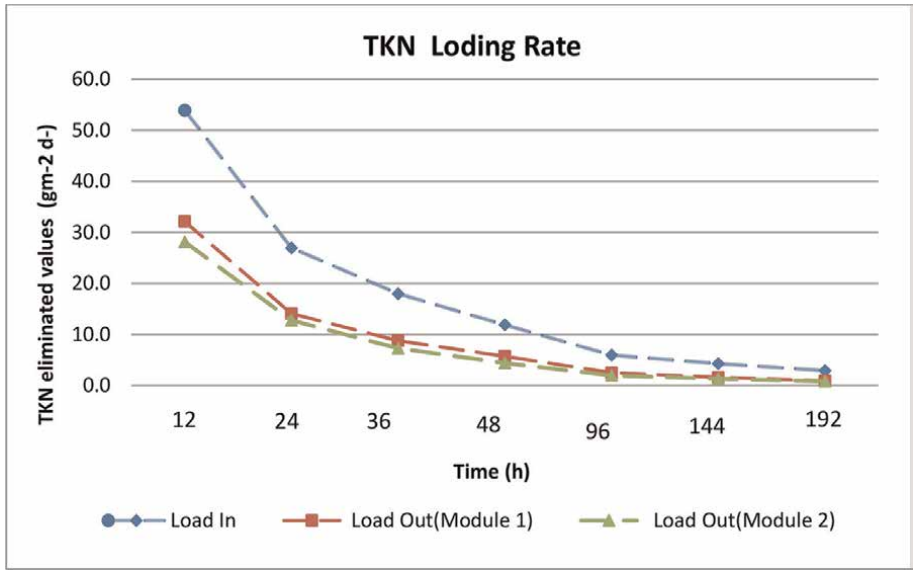

Figure 9.
The input-output loading rate value of TSS through the operational period of the system replicated seven times.

Typically, the term is used to describe an infectious microorganism or agent, such as a virus, bacterium, protozoan, prion, viroid, or fungus humans [62]. The presence of pathogens such as bacteria and viruses in treated wastewater is of concern to public health and the environment if not disposed it. Pathogens are associated with many human diseases when a person comes into contact with water contaminated with bacteria, viruses, and other pathogens [4]. These pathogens are natural factors everywhere including humans, animals, and wastewater. Wastewater is the main source of these pathogens and an important source of transport to humans [63]. Where Fecal Coliform are considered one of the serious pathogens in wastewater and their presence as a function of pollution [60]. Previous studies have shown that hybrid or integrated systems have a higher efficiency in removing pathogens from single-phase systems. Some have suggested that wetlands be combined with chemicals or physical substances to improve performance so that when effluent is formed treatment is suitable for irrigation. In general, wetlands have proven their ability to remove pathogens significantly is well recognized and that pathogen removal in CW is greatly influenced by several factors including: temperature sunlight/UV radiation, pH, hydraulic loading, retention time, texture and porosity, water depth, flow rates, vegetation, and substrate media [63]. Some studies have shown that there is no effect of changing seasons on the process of removing pathogens, while other studies have shown higher removal of pathogens in warmer season and in the dry season in equatorial regions [63]. In this experiment, there was effective removal of Fecal Coliform in both models at (28 ± 2)°C, which means suitable conditions for removal. In addition, the removal of pathogens in CWs is greatly influenced by hydraulic retention time (HRT) [64]. This is in agreement with the results of this experiment, the models achieved an increased removal of Fecal Coliform with an increase of HRT.

In this experiment, there was effective removal of Fecal Coliform in both systems, which means suitable conditions for removal. The results of the experiment achieved high removal with increased HRT for all models. The highest level removal of Fecal

Coliform reaches around (99%, module 2–192 hr), from 81,264 to 700 Cfu/100 ml, while the lowest removal level of Fecal Coliform reaches around (16%, module1–12 hr) from 81,264 to 67,980 Cfu/100 ml. The highest removal rates were as follows: (98, 99%, respectively) after 192 hours, while the lowest removal rates are as follows (66, 50%, respectively) after 12 hours for the bath modules, as shown in **Table 8**.

The results also gave clear differences and higher efficiency in removal compared to other studies that used types of gravel and inorganic substrates, and this is due to olive residues and organic plant wood residues that were used in this study. This explains the role of substrates used in the treatment, especially module 1 that may be for the homogeneity of the material and this is consistent with [64]. In general, this

	HRT (hr)	Average influent (CFU/100 ml)	Average effluent (CFU/100 ml)		Removal efficiency (%)	
Parameter			**Model (1)**	**Model (2)**	**Model (1)**	**Model (2)**
Fecal Coliform	12	81,264	28,000	40,900	66	50
	24	81,264	14,500	20,000	82	70
	36	81,264	10,000	11,900	88	85
	48	81,264	8150	4550	90	94
	96	81,264	4184	2200	95	97
	144	81,264	2000	1000	98	99
	192	81,264	1500	700	98	99

Table 8.
Measured fecal coliform influent and effluent concentrations and input-output loading rates values for the two-stages VF-CWs module replicated seven times.

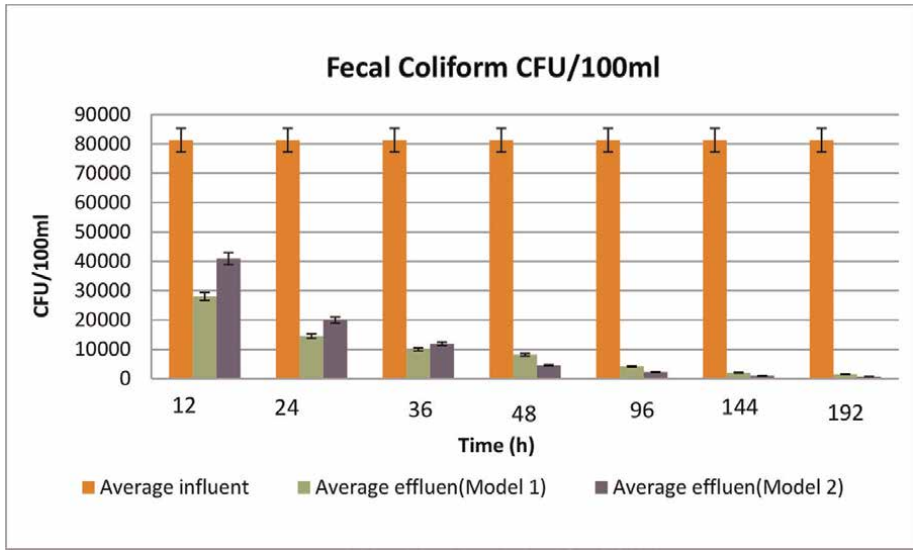

Figure 10.
Measured fecal coliform influent and effluent concentrations during the operational period.

wetlands have proven their ability to remove pathogens significantly and treated water is suitable compared to the Palestinian standards for the re-use of treated wastewater (Ministry of Environmental Affairs) (**Figure 10**) [65].

4. Reuse potential and future perspective

By referring to the results of this study according to the tests carried out after the treatment process for the two constructed systems, and compared to the Palestinian standards for the re-use of treated wastewater, it is found that there is a compatibility between the results each of BOD, COD, TSS, Fecal Coliform that occurred in this study which is suitable for irrigation [64].

5. Conclusions

These types of CWs proved to be effective removal for pollutants and the effluent meets the Palestinian standards for wastewater reuse. The pollutant (COD, BOD, TKN, and Fecal Coliform) adsorption capacity of olivepomace charcoals is faster than citrus charcoal with olivepomace charcoal, but citrus charcoal with olivepomace charcoal gives more efficient removal with increasing HRT. Furthermore, removal capacity of olivepomace charcoal for TSS and P is faster and more efficient with increasing HRT than citrus charcoal with olivepomace charcoal. Using solid plant wood residues and solid olivepomace residues after burning is considered a good alternative substrate in CWs that are inexpensive and environmentally friendly, and gave better results compared to other organic substrates that have been used in CWs.

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Chapter 6

Perspective Chapter: Agronomic Properties of Biochar from Slow Pyrolysis of Human Waste

Hannah Larissa Nicholas, Aisling Devine, Iain Robertson and Ian Mabbett

Abstract

The treatment and safe disposal of sanitation waste is imperative to human health and the environment. In developed countries, the emphasis is on recovering phosphorus from municipal sewage sludge (SS) and the reduction of landfill. Whilst in developing countries, the focus is on long-term mechanisms to treat fecal sludge (FS) generated from non-sewered sanitation facilities. This chapter summarizes the thermal treatment of FS and SS via slow pyrolysis, and the resultant characterization of FS and SS-derived biochar with the aim of utilization by agriculture. In general, FS and SS biochars have high pH, ash content and macronutrient concentrations, with a low surface area and carbon content. The concentration of potentially toxic elements is a key difference between FS and SS biochars with FS biochars containing lower concentrations of these harmful metals. Assessing the properties of these biochars is challenging because of the different methods involved in the processing of raw sludge. The slow pyrolysis of FS and SS to produce biochar can play a pivotal role in a circular economy through the recovery and re-use of waste. Waste-derived biochar provides an opportunity to utilize an integrated systems-based approach to improve soil health, increase crop yield, and improve water retention.

Keywords: sewage sludge biochar, fecal sludge biochar, soil, crop, agronomic, properties

1. Introduction

In developed countries there are sewer systems and wastewater treatment plants that transport and safely treat sewage sludge, however, dramatic population growth, as well as stringent requirements for the treatment of sewage effluent have resulted in a steady increase in the volume of sewage sludge produced [1]. Conventionally the methods for disposing of treated sewage sludge include three main routes: reuse (land application), incineration or landfilling [2]. However, these options are becoming less desirable due to the accumulation of potentially toxic metals and pathogens in sludge which effect its use in agriculture [3]. The impact of EU Directive 2018/851/EC resulted in a ban on landfilling, limited land application of sewage sludge and a focus on sustainable material management and a transition to a circular economy [4].

In developing nations, the emphasis is on sustainable and longer-term solutions to treat fecal sludge from non-sewered onsite sanitation systems. Goal 6 of the UNs 17 Sustainable Development Goals is to "ensure availability and sustainable management of water and sanitation for all" [5].

In low and middle income countries there has been an increase, since 2000, in the proportion of the population reliant on "unimproved" sanitation systems [6]. Globally a total of 3.4 billion are still reliant on onsite sanitation amenities such as composting toilets, septic tanks and pit latrines, [7], with open defecation still practiced by approximately 494 million people globally [8]. Worldwide 2.1–2.6 billion people use onsite sanitation systems that produce large quantities of fecal sludge [9]. Untreated fecal sludge from these facilities is generally discarded straight into the local environment, reused on agricultural land, or disposed of within the household compound [10, 11].

In developing countries, the management, to date, of treatment and disposal of fecal sludge from onsite sanitation facilities has been poor. This had led to pollution of water courses, groundwater and soils [12], negative public and environmental health outcomes and has ultimately resulted in reduced social and economic development [13, 14].

Recently the method of pyrolysis as a solution in treating both sewage and fecal sludge has been a focus of research. Pyrolysis is a thermochemical method where biomass is heated to temperatures of 350–1000°C in an oxygen-free environment [15]. This process effectively and quickly kills pathogens within the sludge [16] and produces a carbon rich product, biochar, which can be used as a soil amendment [17]. The characteristics of biochar are dependent on the composition of the original feedstock, the highest treatment temperature during pyrolysis, the hold time, and the heating rate [11].

The use of biochar as a soil amendment emerged from studies on Amazonian Black Earth (*Terra Preta*). *Terra Preta* (Portuguese for 'black earth') refers to a specific type of dark, incredibly fertile soil found in the amazon basic that was discovered to contain much higher nutrient levels and organic carbon levels than the neighboring soils [18, 19]. The advantages of treating soil with biochar include an increase in carbon levels, [19], an increase in the cation exchange capacity (CEC) [19], an increase in the water holding capacity of the soil [20, 21], and a decrease of acidity especially in acidic soil [22], as well as the reduction and immobilization of toxic metals [23]. The treatment of soil with biochar also leads to carbon sequestration due to the recalcitrant nature of biochar and therefore is an important tool in achieving net zero targets [24].

The characteristics and effects of sewage sludge derived biochars, on soil fertility, and crop yield has been the major focus of attention [25–33], with little research in comparison conducted on fecal sludge derived biochars [34–36].

The end-use of biochar is dependent on biochar properties which in turn depend primarily on two factors: the original feedstock and the high treatment temperature (HTT) used during pyrolysis [37]. Differences in properties of the original feedstock arise from the large variation of processes used to treat the sludges, the sludge holding times and the type of storage facilities used.

There are similarities and differences between the types of human waste discussed in this chapter. The characteristics of each type of waste can vary significantly, depending on several factors outlined below. In general human-waste is a complex heterogeneous mixture which can contain microorganisms, water, oils, nutrients, inorganic material and can be rich in organic matter.

Sewage sludge characteristics can vary with time, type of wastewater treatment facility, the operational method and the sources of the sewage. Wastewater treatment plants receive discharges from industry as well as residential areas. The high concentrations of metals and organic compounds found in sewage sludge can vary greatly depending on nearby industrial activities [38, 39].

Fecal sludge quantities and characteristics can vary greatly depending on several important factors including location, climate, age of the sludge, type of sludge collection and the types of onsite sanitation facilities [40]. These onsite sanitation technologies include septic tanks, aqua privies, pit latrines (including ventilated improved pit latrines VIPs), public ablution blocks and dry toilets. Another difficulty in quantifying FS is that in cities different types of these facilities can be found side-by-side.

2. Composition of human waste

Untreated primary sludge can have pH values ranging from 5.0 to 8.0, with digested primary sludge in the range pH 6.5–7.5 [41]. The range of pH of fecal sludge has been reported between 6.55 and 9.34 [42]. A difference in pH of FS between peri-urban areas and rural areas has been reported with a mean pH of 6.7 in rural areas and 7.3 in peri-urban areas in the Ashanti Region of Ghana [43].

The total solids present in FS comprises of organic and inorganic matter. FS total solids concentration has been measured at 12,000–35,000 mg/l [44] and volatile solids measured at between 0.45 and 4.3 g VS/g ash [45]. Total dry solids of untreated primary sludge and digested primary sludge have been reported at 5–9% and 2–5% respectively [41]. Total solids of liquid, dewatered, dried, or compost biosolids have been reported at 2–12%, 12–30%, and 50% TS, respectively [46].

Nitrogen in FS is found in various forms; as nitrate, nitrite, organic forms (amino acids), and ammonium with ammonium in fecal sludge originating from the urine component [47]. Faecal sludge from septic tanks has been found to contain ammoniacal-nitrogen concentrations at 150–1200 mg/l [44], <1000mg/l and 2, –5000 mg/l reported in studies from Ghana, Thailand and Philippines [48]. Typical municipal sewage from tropical countries has been reported to contain ammoniacal-nitrogen levels of 30–70 mg/l [48]. Nitrate concentrations in fecal sludge have been measured at 0.2–21 mg N/L [49]. A considerable proportion of nitrogen (N) in sewage sludge is organically bound and not immediately available for plant uptake [50]. Dewatered anaerobically stabilized primary sewage sludge has a reported nitrate-N content of 0.253 mg/g ± 0.015 dried sludge [51].

Total phosphorus levels found in FS is high, it is usually present in phosphate form (e.g., H_3PO_4/PO_4-P) or in the organic phosphate form found in plant tissue [52]. Phosphorus levels in sludge from VIPs were found to be 3.4 times higher than sludge from septic tanks. The content of phosphorus in SS has been reported at 20.1–28.4 g/kg^{-3} [53] with phosphorus in sludge mainly present in an inorganic form [54, 55].

Potentially toxic metals that are found in human waste include zinc, cadmium, chromium, nickel, copper, lead and mercury. Potentially toxic metals in sewage sludge originate largely from industrial wastewater entering the sewer system as well as runoff from business effluents and traffic emissions carried via stormwater into the sewer system [56, 57]. Fecal sludge contains lower levels of potentially toxic metals; a recent study reported that FS from pit latrines contained lower concentrations of these metals compared to wastewater sludge [58] and levels in fecal sludge ash have been reported to be below the thresholds for land disposal [59].

3. Pyrolysis

3.1 Pre-treatment of sludge for pyrolysis

Sewage sludge needs to be dewatered and dried before pyrolysis can occur. Sewage sludge is transported through a waterborne sewer system, so it contains a higher liquid content than fecal sludge. Total solids (TS) in biosolids increase from 2 to 12%, in liquid biosolids to 12–30% in dewatered sludge and finally to 50% in dried biosolids [46]. A pelletizing process is sometimes used after the drying step to produce dried pellets of SS which is safer for handling. Dewatering fecal sludge is usually achieved using drying beds [60]. FS total solids concentration have been measured at a range of 12–35 g/l [44], and 20–50 g/l [61, 62].

The source, and drying methods of sewage and fecal sludge can vary considerably. For fecal sludge the sources can range from septic tanks [35], septage drying areas [34], vacuum trucks [63], to latrine waste from ventilated and improved pit latrine (VIP) toilets [64]. The treatment processes at each wastewater treatment plant from where sewage sludge is collected is not always described in the literature but can vary from conventional biological trickling filtration systems [26], to anaerobic digestion and belt-filter-press dewatering systems [1].

3.2 Slow pyrolysis

In this chapter we focus on the most common method of producing biochar: slow pyrolysis. Slow pyrolysis is defined by slow heating rates between 1 and 30°C min°1 [65] with high treatment temperatures of 400–900°C in the absence of oxygen. Slow pyrolysis is often deemed the most practical process for agronomic biochar production [66].

4. Properties of biochar

Biochar characteristics and yield are related to the composition of the raw sludge and the pyrolysis process such as the high temperature treatment (HTT) used, and heating rate [67]. HTT is the main factor influencing final biochar characteristics [65, 68].

The pyrolysis of fecal sludge can be conducted using lab-scale technology such as tube furnaces [35], and muffle furnaces [64] as well as large-scale pyrolysis reactors [11, 69]. There are markedly different pyrolysis conditions reported with HTT ranging from 300°C [16] to 750°C [69] as well as a range of holding times from 10 min [63] up to 120 min [64].

Sewage sludge pyrolysis conditions range from self-made stainless-steel reactors heated in a muffle furnace [70], fixed bed laboratory pyrolyzers [31], mechanically fluidized reactors [71], drum pyrolysis reactors fired using coal, biomass gasifiction units [72] to laboratory-scale muffle furnaces [1]. Similar to fecal sludge the pyrolysis conditions of sewage sludge also vary considerably with high treatment temperatures from 200°C [32] to as high as 700°C [32, 53, 73] with the range of residence times from 10 min [63] up to 120 min [64]. Residence times also vary considerably from as short as 15 min [72] to as long as 360 min [26].

4.1 Biochar pH

Both sewage sludge and fecal sludge-biochars are generally alkaline with greater HTTs resulting in biochars with higher (more alkaline) pH values [16, 73]. Examples of pH for SS biochars are presented in **Table 1** and FS biochars in **Table 2**. The alkalinity of biochar results from the increase in alkali salts and salts of alkaline metals such as calcium, and magnesium during pyrolysis [75]. The effect of biochar in altering soil pH is one of several ways in which biochar increases crop yield. Approximately 30% of land cover globally is comprised of acidic soil which leads to diminished crop yield, and also impacts the kind of crops that can be grown. Maize, a cereal corn widely cultivated globally, is negatively affected by acid soil [76], in fact out of all the soils worldwide that are suited to arable agriculture, a large proportion, up to 50%, are acidic [77].

Biochars typically exhibit neutral to alkaline pH values so can increase the pH of soil by reducing the acidity of acidic soil or increasing the alkalinity of neutral/alkaline soil. This is termed the "liming effect" and contributes to improved plant growth and crop yield especially in acidic soils. It is one of several pivotal mechanisms that contribute to the increased plant growth and yield upon biochar addition [78].

The liming effect improves various soil-plant interactions including:

- Increased phosphorus bioavailability and calcium and magnesium bioavailability
- The reduction in the available concentration of aluminum, a metal toxic to plant growth [79]
- Enhanced nitrogen fixation in legumes
- Increased microbial activity [80]

Generally, biochars pyrolyzed at temperatures ≥500° tend to be alkaline whereas sewage sludge derived biochars pyrolyzed at relatively lower temperatures of 300–400°C are more acidic [25, 32, 73]. There are differing results from the effect of FS and SS biochars on the pH of soils [32].

Sewage sludge biochars have increased soil pH, available nutrient concentration and shown a decrease in the bioavailable forms of As, Cr, Co, Ni and Pb [28]. Fecal sludge derived biochar has resulted in an increase of soil pH and cation exchange capacity (CEC) of soil [35]. However, another study showed that sewage sludge biochar treatment resulted in a decrease in soil pH despite the alkalinity of the biochar applied to the soil [32]. Soil pH also affects phosphorus adsorption and bioavailability with this process more evident in acidic soils due to the liming effect of biochar contributing to an increase in phosphorus bioavailability [81]. The availability of nutrients within the biochar itself is also positively affected by soil pH, with an increase in the release of $HxPO_4$ and NH^{4+} from biochar associated with decreasing pH [82, 83].

4.2 Ash

The concentration of ash in biochar is generally higher than in raw sludge and increasing HTT during pyrolysis results in increased ash content of biochar [84]. The original feedstock has a significant effect on ash content, for example, biochar from hazelnut produced an ash content of only 1.2% compared to poultry litter biochar with an ash content of 51.2% with both pyrolyzed at 350°C [85].

Pyrolysis temperature (°C)	pH	Ash content (%)	SBET surface area (m^2g^{-1})	CEC (cmol(+) kg^{-1})	Reference
BC200	6.54	68.62	—	—	[32]
BC300	7.20	70.14	—	—	
BC500	8.70	79.00	—	—	
BC700	11.15	85.75	—	—	
BC-1300	6.89 ± 0.08	69.2 ± 1.3	24.9	—	[72]
BC-2300	7.06 ± 0.04	61.4 ± 1.4	11.9	—	
BC-3300	7.18 ± 0.04	76.1 ± 1.5	2.2	—	
BCKN500	7.13	73.56	31.8	—	[53]
BCKN600	11.03	77.77	24	—	
BCKN700	12.23	79.08	54.1	—	
BCKZ500	7.08	68.09	16.3	—	
BCKZ600	11.45	70.27	9	—	
BCKZ700	12.38	74.28	29.9	—	
BCCM500	7.17	68.98	34.2	—	
BCCM600	11.33	70.22	16.4	—	
BCCM700	12.44	71.99	9.2	—	
BCSI500	7.25	64.1	35.7	—	
BCSI600	8.05	63.86	19.2	—	
BCSI700	13.1	67.98	18.1	—	
BC300	7.2–7.5	—	4.0–6.7	—	[74]
BC400	7.1–7.5	—	8.7–17.7	—	
BC500	7.6–7.7	—	10.2–26.5	—	
BC600	8.1–8.5	—	6.3–18.2	—	
BC300	6.0	—	4	—	[1]
BC500	—	—	18	—	
BC300	5.32	52.8	—	—	[73]
BC400	4.87	63.3	—	—	
BC500	7.27	68.2	—	—	
BC600	12.00	72.5	—	—	
Sludge Biochar 300–500	8.54 ± 0.08	—	—	140 ± 0 .4	[26]
Slow pyrolysis 300	—	38.3	—	—	[71]
Slow pyrolysis 400	—	44.0	—	—	
Slow pyrolysis 500	—	50.4	—	—	
300	6.0 ($CaCl_2$ method)	—	—	—	[25]

Pyrolysis temperature (°C)	pH	Ash content (%)	SBET surface area (m^2g^{-1})	CEC (cmol(+) kg^{-1})	Reference
450	8.6	—	—	—	[31]
500	8.9	61.4	71.6	—	[70]

Table 1.
pH, ash, surface area and CEC of various sewage sludge biochars.

Pyrolysis temperature (°C)	pH	Ash content %	SBET surface area ($m^2\ g^{-1}$)	CEC cmol (+) kg-1	Reference
600	10.4	—	690.8*	—	[35]
450	8.23	—	3.36	23.2	[34]
N-BC 500–700	10.5 ± 0.5	45.6 ± 4.2	—	—	[69]
W-BC 500–700	10.8 ± 1.2	60.8 ± 5.5	—	—	
350 (10 min)	9.1	54.5	—	9.8	[63]
350 (20 min)	9.2 ± 0.02	57.2 ± 1.8	—	13 ± 0.7	
350 (40 min)	9.3	57.5	—	9.8	
450 (10 min)	9.7	65.6	—	22.9	
450 (20 min)	9.7 ± 0.02	66.9 ± 1	—	23.2 ± 0.9	
450 (40 min)	9.7	66.2	—	23.5	
600 (10 min)	11.0	68.1	—	24.6	
600 (20 min)	11.1 ± 0.01	72. ± 0.9	—	26 ± 1.7	
600 (40 min)	11.2	73.8	—	27.7	
BC-300	7.3 ± 0.1	26.3 ± 0.8	—	—	[16]
BC-400	7.5 ± 0.1	31.3 ± 0.9	—	—	
BC-500	10.3 ± 0.2	45.5 ± 1.2	—	—	
BC-600	10.7 ± 0.2	58.8 ± 0.6	—	—	
BC-700	11.1 ± 0.2	62.5 ± 0.4	—	—	
350	6.94	84.60	7.5	5.09	[64]
550	7.02	90.23	23.7	4.91	
650	7.14	92.97	25.7	5.65	
WAI_BC 550–750°C	11.81 ± 0.01	62.3 ± 0.32	3.52 ± 0.78	90.0 ± 6.5	[11]
NSP_BC 550–750°C	11.82 ± 0.01	67.0 ± 2.68	3.69 ± 0.36	41.9 ± 2.2	
WGL_BC 550–750°C	12.45 ± 0.01	88.3 ± 0.21	12.07 ± 4.12	129.3 ± 2.3	

**Biochar was milled to pass through a 74 μm sieve and demineralized with HCl (2 mol/L).*

Table 2.
pH, ash, surface area and CEC of fecal sludge biochars—hold times are in brackets.

The initial feedstock of human waste tends to be high in ash. Sewage sludges can contain high concentrations of calcium (5.1–7.4%), silica (19–58%), iron (5.2–6.8%) and phosphorus (3.4–4.9%) [53] and ash content of fecal sludge can be as high as 17.0% compared to sawdust measuring only 0.8% [16]. Ash content of SS biochars have been measured at 52.8% at 300°C HTT and 63.3% at 400°C HTT [73]. Biochars produced from mixed urine and feces samples, similar in ratio to sludge from on-site sanitation systems, had ash content of 50.1% at 450°C HTT and 56.3% at a HTT of 650°C [86]. Recently a study comparing mixed urine and feces (MUF) biochar and source-separated feces (SFF) biochar found that MUF biochar exhibited higher ash contents which they associated with greater quantity of inorganic salts in urine [86]. The higher ash content in fecal sludge biochars compared to sewage sludge biochars are due to digestion of the sludge during holding in onsite sanitation amenities [63], the ingress of sand and grit due to poorly lined containment structures [87] and the adhesion of sand to fecal sludge from the surface of drying beds [88]. Biochar may benefit from high ash contents if its end-use is as a soil amendment as the minerals found in ash such as calcium, magnesium, potassium, are essential plant nutrients.

The high ash content of SS and FS biochars is connected to the alkalinity of these biochars. Increasing pyrolysis temperatures leads to an increase in alkalinity due to an increase in ash in biochars derived from sludge feedstocks [16, 73].

Previously the ash content of a FS biochar was reported to play a role in the increase in plant height, below ground biomass and yield of tomatoes grown in acidic soil [36].

4.3 Surface area and porosity

The porous structure of biochar strongly resembles the cellular structure of the original feedstock [84]. In the case of fecal and sewage sludge biochars the cellular porous structures arise from undigested fibrous vegetable matter (**Figure 1**).

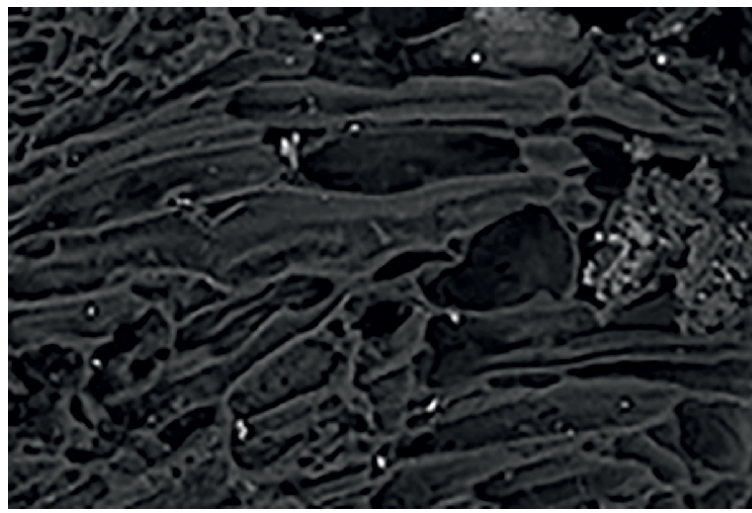


Figure 1.
SEM micrograph of fecal sludge biochar [11].

The addition of biochar to soil can greatly improve soils water retention. A study in 2002 showed that *Terra preta* exhibited 18% greater water retention compared to neighboring soils that contained very little charcoal [89]. The porous structure of biochar results in greater water holding capacity of soil [21] and increases water availability [90–92].

The BET surface area of biochars is increased with increasing HTT during pyrolysis, as at higher temperatures there is an increase in volatile matter released.

Municipal sludge pyrolyzed at temperatures between 500 and 900°C produced a greater biochar yield and greater microporous network within the biochar at increasing HTTs [93]. Biochar produced from sewage sludge-derived fertilizer was shown to mainly consist of mesopores with some microporous structure present [94].

Surface area measured by N_2 is generally quite low for SS derived biochars, values have been reported ranging from 2.2 $m^2\ g^{-1}$ [72] to 54.1 $m^2\ g^{-1}$ (**Table 1**) [53]. Research has shown that sewage sludge biochars have low surface areas due to high ash content [1, 94, 95]. Surface areas are reduced due to the high ash content present which fills or blocks access to the micropores within the biochar [66]. Surface areas of fecal sludge biochars have been measured at 3.7 $m^2\ g^{-1}$ and 25.7 $m^2\ g^{-1}$ (**Table 2**) [34, 64]. It has been reported that a greater surface area of fecal sludge biochar (690.8 $m^2\ g^{-1}$) can be attained by washing with 2 M HCl acid [35].

Despite comparatively low surface areas, sewage sludge biochar addition has been shown to significantly increase available water of soil [96] and increased sunflower production under the Mediterranean climate without additional water irrigation [97].

4.4 Cation exchange capacity (CEC)

CEC relates to the ability of soil or biochar to adsorb positive ions in exchangeable forms and plays a key role in nutrient leaching and retention in soils [98]. The CEC is largely determined by the oxygen containing functional groups present on the surface of biochar such as C=O groups [99]. These functional groups allow the adsorption of nutrients in the form of cationic such as K^+, Ca^{2+}, and NH^{4+}.

Biochar addition to soil has been shown to increase CEC and pH [19, 35] and limit nutrient leaching, [100] and improve nutrient retention [66].

CEC values of FS sludge biochar has been reported at 23.2 cmol(+) kg^{-1} for biochar pyrolyzed at 450°C [34]. FS biochar CEC values range from 23.2 cmol(+) kg^{-1} for biochar pyrolyzed at 450°C [34] to 129 cmol(+) kg^{-1} for biochar pyrolyzed at between 550 and 750°C [11].

A decrease in CEC values for biochars derived from fecal sludge and sewage sludge with increasing pyrolysis temperature from 350°C to 550°C has been reported [64]. However, both biochars exhibited an increase in CEC values at pyrolysis temperatures of 650°C. Another study conflicts these findings as they recorded increasing CEC values of fecal sludge chars with increasing HTT up to 600°C [63]. Cation exchange capacities of sewage and fecal sludge biochar are sparsely recorded in the literature therefore it is difficult to conclude what impact feedstock and HTT has on sludge-derived biochar CEC values.

The effect of sewage sludge biochars on soil CEC has been reported with application of sewage sludge biochar pyrolyzed at 300°C resulting in increased soil CEC one year after its application and increased maize grain yield [101]. Wastewater sludge biochar has also been found to increase soil CEC by up to 40% [102].

Examples of CEC values for SS and FS biochars are given in **Tables 1** and **2**.

4.5 Elemental Microanalysis (C, H, N and O)

FS and SS-derived biochars generally have low total C concentrations (11–40%) compared with biochars from lignocellulosic feedstocks [103]. It is the ash content of the original feedstock that influences the biochar ash content and therefore carbon content of the resulting biochar. Fecal and sewage sludges are naturally high in ash therefore produce biochars with high ash and low carbon content. Sewage sludge biochars have reported carbon contents of 21.6–26.2% with a low percentage of H also reported (3.8–5.1%) [53]. The carbon content of fecal sludge biochars have been measured at 21.1–23.8% [11].

Carbon is concentrated within the biochar during the thermochemical process with an increase in carbon content relative to the feedstock commonly reported, however with sewage sludge biochar many studies have reported a decrease in carbon content in biochar relative to the original feedstock [1, 28]. C and N content as well as ash content are reduced with an increase in HTT signifying that as more ash is relatively accumulated, carbon and nitrogen content is reduced. Soils amended with sewage sludge biochar have increased total nitrogen, and organic carbon [28].

Nitrogen in fecal sludge is found in mainly organic form [104] and is volatilized at temperatures of 200°C [80]. Therefore, the nitrogen content of FS and SS biochars can be very low. The total nitrogen content of biochars can vary markedly across a large range [105]. Nitrogen content of wastewater sludge biochar and sewage sludge biochar generally increases with decreasing pyrolysis temperature [73, 106]. Application of SS biochar has increased N uptake and enhanced N use efficiency two years after addition of the biochar, indicating its potential as an alternative to fertilizer [107].

4.6 Potentially toxic metals

The high variability of potentially toxic metal (PTM) levels both in sewage and fecal sludge affects the PTM content in the resultant biochar. The thermochemical process does, however, constrain these metals in immobile and stable forms. The entrapment of potentially toxic metals within the biochar reduces the risk of plant uptake of these metals allowing the potential use of SS biochar as a soil amendment without negatively impacting soil-plant systems.

The levels of potentially toxic metals generally increase as HTT increases [74, 108], however there are conflicting reports on the impact that increasing HTT has on these metals in sludge biochar. The general trend does seem to be an increase in metal concentration with an increase in pyrolysis temperature with some noticeable exceptions at higher temperatures. It has been reported for sludge biochar that toxic metal levels peaked at 450°C and decreased at higher temperatures of 500–550°C [109]. Others have reported a decrease in all PTM concentrations of sludge biochar pyrolyzed at 700°C except for cadmium [73]. Potentially toxic metal levels in FS-biochar conforms to the general trend with an increase in PTM concentrations reported with increasing HTT [63].

Biochar pyrolyzed at higher temperatures can have beneficial qualities for use as a soil amendment including higher pH values and greater surface areas. Consideration needs to be paid to ensure that the higher temperatures do not increase PTM concentration in biochars to greater than the recommended guidelines for PTMs in soils.

Potentially toxic metals in most SS and FS derived biochars are below International Biochar Initiative (IBI) accepted upper thresholds [110].

The leaching of potentially toxic metals from SS and FS biochar has also been investigated and it was found that in contrast to the original sludge, biochar contains significantly less total concentrations of these metal as well as less soluble and extractable fractions [111]. Toxic metal mobility of biochar from fecal sludge co-treated with agricultural waste has been shown to be markedly reduced compared with metal mobility of original feedstock [69]. There are conflicting reports on the effect of HTT on extractable fractions of potentially toxic metals. DTPA-extractable concentrations of these metals have been reported to decrease with increasing HTT (from 300 to 700°C) in wastewater sludge biochar [73] and in a separate study extractable potentially toxic metal concentrations in SS biochar increased with increased HTT (from 300 to 500°C) [74].

4.7 Phosphorus

There is an abundance of mineral nutrients found in sewage and fecal sludges such as potassium, ammonium, nitrate, and trace elements. SS and FS are rich in phosphate which is of particular importance as phosphorus is a finite resource and an essential plant limiting nutrient [112]. Concentrations of phosphorus on a dry weight basis range from <0.1 to 14% in sewage sludge [113]. The resultant concentration of phosphorus in biochar is greater compared to the original feedstock as other elements such as carbon, hydrogen, and oxygen are volatilized at the high pyrolysis temperatures [25]. In general, an increase in HTT results in increased phosphorus concentration within the biochar. Confirming the general trend, phosphorus concentrations in sewage sludge biochar increased from 5.6% at a HTT of 250°C to 12.8% at a HTT of 800°C [114].

Phosphorus within sewage sludge is found mainly in the inorganic fraction and is significantly affected by volatilization losses at HTT greater than 700°C [115]. This trend has been observed in SS biochar [53, 74] and FS biochar [16, 63], however decreases in phosphorus concentrations in FS biochar have been reported at HTTs of 700°C [16]. The reduction in phosphorus content at higher pyrolysis temperatures may be due to different sludge types containing different forms of phosphorus. Chemical and biological treatment process of sewage and fecal sludges can also affect the forms of phosphorus present and the degree to which phosphorus is volatilized at high HTTs >700°C [116]. There are recorded increases in total phosphorus levels in FS biochar with increasing HTT from 3.2% at 350°C to 3.9% at 600°C [63] and 5.4% at 300°C to 8.1 wt.% at 600°C. The latter study did however, report a slight decrease in P concentration at 700°C [16].

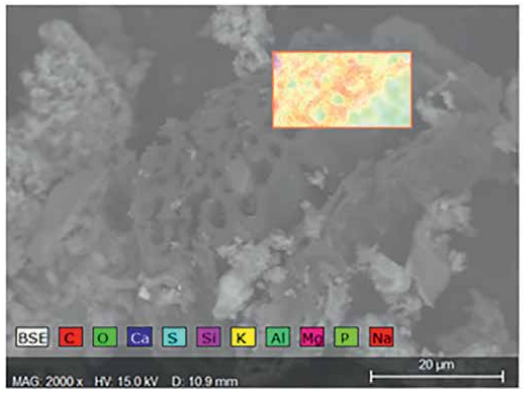
The bioavailable phosphorus within biochars is generally less than the total P content with an increase in HTT resulting in increased available phosphorus compared to the raw sludge feedstock [31]. One of the most important effects of SS biochar application to soil is the increase in bioavailable P content [71].

Biochar-treated soils have higher organic bioavailable phosphorus concentrations relative to soil without biochar treatment but these mechanisms concerning the release of nutrients from biochar are not fully understood. The potential mechanisms include the supply of nutrients from the biochar itself, the liming effect of biochar (especially alkaline biochar) which increases the plant-available phosphorus levels [117, 118] and enhanced nutrient retention capacity [119].

Bioavailable P within sewage sludge biochars has been shown to decrease with increasing HTT [32, 73], however analysis of FS biochar has revealed the opposite trend with an increase in available phosphorus with increasing HTTs from 350°C [63].

4.8 Macronutrient concentrations (Ca, Mg and K)

Sewage and fecal sludge biochars contain large amounts of macro-nutrients such as calcium, potassium, and magnesium. Thermochemical treatment of sludge increases these elements concentrations in biochar compared to raw sludge. Increases in Ca, K, and Mg have also been reported with increasing HTT. These metallic elements cannot be volatilized at the pyrolysis temperatures, so these are concentrated within the biochar as C, H, and O are gradually lost at higher HTTs [120]. Large concentrations of Ca, Mg and K in SS and FS biochar has been reported (**Figure 2**) [31, 34, 69, 72, 73]. Both SS and FS biochars have been reported to contain increasing Ca, Mg, and K concentrations with increasing HTT [53, 74] [16]. The treatment process of sewage sludge can impact the concentration of certain elements; it was noted that a relatively high proportion of calcium present in sludge biochar was caused by to the addition of CaO during the sludge conditioning process [74]. Soil amended with SS biochar has recorded greater levels of potassium [121] and calcium and magnesium [25]. In the latter study an increase in Mg concentration was recorded in leaves of radish plants grown in the soil, however there was no effect of the biochar on calcium concentration in radish leaves despite the increased calcium levels recorded in the soil [25]. Other studies have found SS biochar application to soil, has not affected the levels of exchangeable calcium and magnesium ions in the soil over an average of 5 years [122]. The conflicted results of these studies highlight the many factors that influence the concentration levels of macronutrients in soils. These factors include the treatment of the raw feedstock, the HTT used during pyrolysis and the rate of biochar application (**Tables 3**–**5**).



Element	Wt. %
C	70.55
O	21.00
Ca	3.24
S	1.62
Si	1.29
K	0.73
Al	0.63
Mg	0.50
P	0.27
Na	0.17

Figure 2.
SEM-EDX map for all elements distribution across the area highlighted in image and associated energy dispersive X-ray (EDX) quantification of fecal sludge biochar with calcium the most abundant metal [11].

Pyrolysis temperature	K	Mg	Ca	Reference
BC-1300	4.32 ± 0.14	12.4 ± 0.3	42 ± 3	[72]
BC-2300	3.54 ± 0.31	9.6 ± 0.3	65 ± 2	
BC-3300	3.92 ± 0.28	5.9 ± 0.3	34 ± 2	
BCKN500[1]	0.92 ± 0.08	0.94 ± 0.09	8.27 ± 0.51	[53]
BCKN600[1]	1.01 ± 0.08	1.08 ± 0.09	9.18 ± 0.56	
BCKN700[1]	1.09 ± 0.08	1.13 ± 0.10	9.71 ± 0.59	
BCKZ500[1]	1.4 ± 0.11	1.47 ± 0.12	6.75 ± 0.43	
BCKZ600[1]	1.55 ± 0.12	1.65 ± 0.14	6.02 ± 0.38	
BCKZ700[1]	1.64 ± 0.12	1.78 ± 0.14	7.42 ± 0.46	
BCCMI500[1]	1.25 ± 0.10	1.13 ± 0.10	12 ± 0.70	
BCCMI600[1]	1.34 ± 0.11	1.25 ± 0.10	11.4 ± 0.61	
BCCMI700[1]	1.34 ± 0.11	1.27 ± 0.10	12 ± 0.70	
BCSI500[1]	1.06 ± 0.08	3.29 ± 0.23	10.2 ± 0.61	
BCSI600[1]	1.12 ± 0.09	2.57 ± 0.19	10. 8 ± 0.64	
BCSI700[1]	1.12 ± 0.10	2.44 ± 0.18	11.9 ± 0.70	
DTS300	2.1	8.2	8.1	[74]
DTS400	2.4	8.4	8.4	
DTS500	2.4	8.2	8.8	
DTS600	2.8	9.3	6.7	
LD300	1.6	11	11.6	
LD400	2	13.4	11.9	
LD500	2.2	12.5	12.2	
LD600	2.6	14.5	14.6	
XL300	1.8	5.4	1.8	
XL400	2.1	5.5	2	
XL500	2.2	5.9	2.1	
XL600	2.3	3	2.3	
BC300[1]	—	0.35 ± 0.01	3.47 ± 0.15	[73]
BC400[1]	—	0.43 ± 0.01	4.17 ± 0.02	
BC500[1]	—	0.46 ± 0.01	4.62 ± 0.12	
BC600[1]	—	0.54 ± 0.01	5.35 ± 0.10	
Sludge biochar 300–500	3.0 ± 0.4	35.9 ± 3.9	19.9 ± 0.7	[26]
300	0.16	1.8	9.7[3]	[25]
450	13.8	—	—	[16]
500	5.25	6.45[2]	65.7[2]	[70]

[1]*Values in percentages %.*
[2]*mg kg^{-1}.*
[3]*cmol kg^{-}1.*

Table 3.
Macronutrient (Ca, Mg, K) concentrations in sewage sludge biochar. Values in g/kg unless otherwise stated.

Pyrolysis temperature (°C)	K	Mg	Ca	Reference
450	28.9	—	32.8	[34]
N-BC 500–700	8.1 ± 0.8	7.8 ± 0.7	56.4 ± 3.9	[69]
W_BC 500–700	11.7 ± 1.9	9.6 ± 1.7	89.4 ± 11.5	
BC-300	1.9 ± 0.9[1]	—	—	[16]
BC-400	2.1 ± 0.9[1]	—	—	
BC-500	2.8 ± 0.3[1]	—	—	
BC-600	2.7 ± 0.9[1]	—	—	
BC-700	2.6 ± 0.6[1]	—	—	

[1]= *wt%.*

Table 4.
Macronutrient concentrations (Ca, Mg, K) in fecal sludge biochars. Values in g/kg unless otherwise stated.

Pyrolysis temperature	NH_4^+-N (mg/kg)	NO_3^--N (mg/kg)	Reference
BC200	533.51	0.10	[32]
BC300	119.28	1.97	
BC500	21.41	2.77	
BC700	17.72	2.72	
300	1175	<0.2	[73]
400	142.5	<0.2	
500	25	0.24	
600	1.34	0.32	
300	431.9	17.5	[25]

Table 5.
NH_4^+-N and NO_3^--N concentrations of sewage sludge biochars.

5. Conclusion

In this chapter, the focus is on assessing the similarities between fecal sludge and sewage sludge biochars, with the observed distinctions arising from variances in the transportation and treatment of the raw material. Sewage having traveled through a sewered sanitation system, receives discharges from industry as well as residential areas and thus, generally exhibits higher levels of potentially toxic metals compared to fecal sludge which is contained in onsite sanitation systems. The variances in potentially toxic metal concentrations between fecal sludge and sewage sludge may not be significant since these metals are confined within the biochar in inert and stable forms. However, higher temperatures during pyrolysis can lead to elevated concentrations of potentially toxic metals. To prevent exceeding recommended guidelines for these metals in soils, it is advisable to employ lower pyrolysis temperatures when producing sewage sludge biochar. The similarities between fecal sludge (FS) and sewage sludge (SS) biochars include high pH, ash content, and nutrient composition. These properties suggest their potential to enhance soil fertility and improve crop yields, particularly in acidic and nutrient-deficient soil conditions.

Assessing the properties of these biochars is challenging because of the different processes used in collection, storage, and transportation of the raw sludge. Properties of fecal sludge vary depending on location, season, climate, sanitation technology, and sludge age. These aspects combined contribute to the difficulty in being able to characterize, in general terms, fecal sludge and therefore the properties of fecal sludge biochar. It is suggested that characteristics of large-scale produced FS biochar should be examined on a case-by-case basis considering the factors described.

This chapter emphasizes the importance of the physical and chemical properties.

of sludge biochars and also the physical and chemical properties of the soil to which the biochar is added.

Future research should concentrate on short-term and long-term field studies of sludge biochar application to acidic, low nutrient soils. Long-term field trials are needed to determine the duration of the reported positive liming effects of sludge biochars and the long-term effects of repeated biochar applications on potentially toxic metal content in soils.

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Conflict of interest

The authors declare no conflict of interest.

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The present book *Sustainable Use of Biochar - From Basics to Advances* compiles detailed information on a thoughtful approach that maximizes biochar's benefits while minimizing potential environmental and social impacts. To achieve this, it's crucial to start with sustainable sourcing of biomass, ensuring it comes from renewable and responsibly managed sources. During the pyrolysis process, adopting efficient technologies that reduce emissions and energy consumption contributes to sustainability. Quality control measures, such as monitoring biochar's properties like particle size and nutrient content, ensure its effectiveness in soil improvement and carbon sequestration. Application rates should be tailored to soil types and crop needs, supported by long-term monitoring to assess biochar's impact on soil health, greenhouse gas emissions, and crop productivity. Integrating biochar with other sustainable agricultural practices and promoting knowledge sharing further enhances its sustainable use, fostering resilience and environmental stewardship in agriculture. This book has information on sustainable agricultural practices, and promoting knowledge sharing further enhances its sustainable use, fostering resilience and environmental stewardship in agriculture. Biochar applications are able to cope with global challenges in soil management, and ecological approaches can restore the damaged ecosystem. The present book will be helpful to researchers, the scientific community, academics, business farmers, and policymakers to overcome various environmental concerns.

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