Review

Potential for Land Application of Biosolids-Derived Biochar in Australia: A Review

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Abstract: Thermal treatment in Australia is gaining interest due to legislative changes, waste reduction goals, and the need to address contaminants risks in biosolids used for agriculture. The resulting biochar product has the potential to be beneficially recycled as a soil amendment. On-farm management practices were reviewed to identify barriers that need to be overcome to increase recycling and examine the role of pyrolysis and gasification in effectively improving the quality and safety of biochar. Key findings revealed: (1) thermal treatment can effectively eliminate persistent organic pollutants, microplastics and pathogens, and (2) more than 90% of the total heavy metals content in biosolids become immobilized when these are converted to biochar, thus reducing their bioavailability following land application. While reported research on the short-term effects of biosolids-derived biochar suggested promising agronomic results, there is dearth of information on long-term effects. Other knowledge gaps include optimisation of land application rates, understanding of rate of breakdown and fate of contaminants in soil and water, heavy metal mobility in soil and bioaccumulation or transfer to the food chain. Improved understanding of nutrients and contaminants dynamics in soils receiving biosolids-derived biochar is a pre-requisite for their safe use in Australian agriculture, and therefore it is highlighted as priority area for future research.

Keywords: heavy metals; microplastics; organic pollutants; pyrolysis and gasification; sewage sludge; soil amendment; thermal treatments

1. Introduction

Biosolids are the solid end-product of urban wastewater treatment plants, consisting of sewage sludge treated to achieve safe environmental and health standards [1]. While these biosolids are rich in organic matter and contain agronomically significant concentrations of plant nutrients, they also contain contaminants, including organic chemicals, heavy metals, pathogens and microplastics, which cause concern due to the potential for long-term environmental and public health impacts [2, 3]. As the global population increases, biosolids production, which is proportional to population size, will also increase [4]. Annual sewage sludge production has been estimated at 10 million tons, 40 million tons, and 14 million tons in Europe, China, and the United States, respectively [5]. In 2021, Australia generated approximately 350,000 dry tons of biosolids [1]. These are growing concerns, given that biosolid production increased by 24% from 2010 to 2019 [6], and restrictions regarding the safe use of biosolids in Australia have increased with a trend toward diverting their reutilization as a source of carbon and nutrients in agriculture [7]. There is renewed interest both nationally and internationally in finding an alternative waste management strategy that models the principles of circular economy to recover carbon, nutrients, and energy from biosolids, while reducing the need for landfill disposal [8, 9].

Thermal processes including pyrolysis, gasification, and hydrothermal technology can be employed to sustainably process biosolids intended for land application [8]. The materials that result from these processes offer several advantages compared with biosolids, including: (i) reduction or improved control of odor, pathogens, organic and inorganic contaminants; (ii) mass reduction (range: 30 to 90%), which subsequently reduces handling, transport and storage costs; (iii) conversion of biosolids into higher-value products such as bio-oil, syngas and biochar [10]. These advantages should be perceived as opportunities to improve regulatory compliance, reduce existing costs, and generate additional revenue streams.

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This article critically reviews the potential of using biosolids-derived biochar as a soil amendment in Australian agriculture. While there have been reviews on biochar production from biosolids, including characterization, and evaluation of its effect on soil and crops [4, 11] (Figure 1), need remains for a specialized assessment on biosolids-derived biochar to understand its potential as a soil amendment in the Australian agricultural system. Initially, the current biosolids management practices and regulatory frameworks in Australia is analyzed to identify limitations associated with biosolids recycling to land. Furthermore, it also explores thermal treatment methods (i.e., pyrolysis and gasification) as potential biosolids management solutions. The review then evaluates the physicochemical properties and contaminant fate of biosolids-derived biochar to assess its potential for land application compared to biosolids. The aim of the review is to present biosolids-derived biochar as a promising soil amendment by highlighting the opportunities and challenges when applied to soil and taken up by plants.

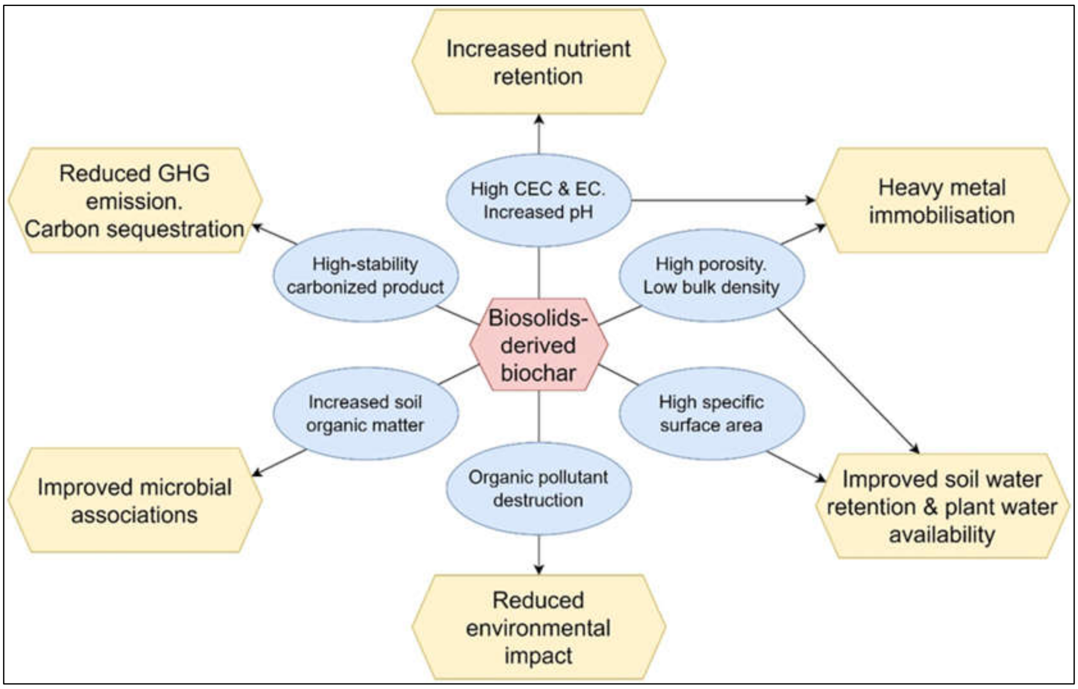

Figure 1: Relevant properties of biosolids-derived biochar that can improve soil properties.

2. Current biosolids management practices and regulatory framework in Australia

In 2019, Australia produced 2.3 million tons of wet biosolids with an average solid content of 16%, equating to 371,000 tons DBS [6]. The large majority (91%) of biosolids were applied to land. Of this 91%, approximately 67% of biosolids were used in agriculture, 16% in land rehabilitation, and 8% in landscaping. The remaining 9% went to landfill (4%), discharged into the ocean (1%), and 4% were used for other purposes [1, 6].

A national regulatory framework strictly controls the land application of biosolids [12], and state guidelines have been developed to ensure a high level of protection for both the environment and public health [13]. However, current guidelines for controlling nutrients, pathogens, and contaminants in land application of biosolids vary between states in Australia, as highlighted by McCabe et al. (2019) [14]. As a result, Victoria, Tasmania and Northern Territory [1] are faced with the problem of stockpiling biosolids that fail to meet the regulatory criteria [15]. Currently, there are no guidelines in Australia on the issue of microplastics present in biosolids.

3. Limitations with recycling biosolids to land

The concern around environmental health, food safety, and quality are due to heavy metals and metalloids, persistent organic pollutants (POPs), microplastics, and pathogens [16]. These contaminants hinder the land application of biosolids.

3.1. Heavy metals and metalloids

The risk of environmental availability or mobility (i.e. potential of metals when released in the environment, to move under natural forces to groundwater or to distance from the site of release) [17, 18]

and bioavailability (i.e. the ability a phenomenon strongly controlled by type of organism, type of exposure and metal availability) of heavy metals and metalloids in the soil is a primary concern for land application of biosolids [5]. Elements such as arsenic (As), copper (Cu), lead (Pb), zinc (Zn), and nickel (Ni) present in sewage sludge become concentrated during treatment and are present in biosolids [19]. Land application of these elements may result in uptake by plants and subsequent transfer to the food chain [20, 21] or environmental losses by processes such as leaching and runoff [2, 22].

The degree of these risks depends on both the concentration of heavy metals and metalloids in the soil amendment; and the environmental physicochemical properties in which they are released. The elemental concentrations vary depending on the location, wastewater source (commercial, domestic or industry), and sludge treatment process [15]. However, the most critical factors that affect the mobility and bioavailability of heavy metals are soil pH, soil permeability, soil organic matter content, and factors that affect microbial activity [23].

3.2. Persistent organic pollutants

Persistent organic pollutants (POPs), derived from synthetic organic compounds used in numerous industries, are present in wastewater and accumulate in biosolids [24]. Although primary and secondary treatments in WWTP result in the partial removal of organic pollutants (e.g., polyfluorinated alkyl substances (PFAS) [25] and triclosan [26], some may remain in residual concentrations in biosolids and include perfluorinated chemicals (PFOS, PFOA), polychlorinated biphenyls (PCB), polychlorinated alkanes (PCAs), polybrominated diphenyl ethers (PBDE), triclosan, polyaromatic hydrocarbons (PAH), polybrominated diphenyl ethers (PBDEs), dioxins, steroids and antibiotics [24]. The concentration of total PFOS, PFOA and total PCB detected in Australian samples of biosolids ranged from 0.021-0.386 mg·kg-1, 0.003-0.05 mg·kg-1, 0.27-0.77 mg·kg-1 and 0.02-0.41 mg·kg-1 respectively [25]. Consequently, the existence of POPs in land-applied biosolids results in ecosystem contamination with potential for bioaccumulation in plants and animals [26] and risk of human and animal toxicity [27].

To address the risk of environmental persistence, human and animal toxicity, and bioaccumulation of POPs in the food chain, the Australian government introduced strict concentration limits to restrict the land application of biosolids with high concentrations of POPs [28]. In Australia, the allowable limits of POPs in biosolids ranged from: PFOS 0.3-4.2 mg·kg-1; PFOA 0.05-33.6 mg·kg-1; total DDT 0.5-1 mg·kg-1; total PCBs 0.05-0.5 mg·kg-1 [12]. Although the disposal of biosolids in Australia complies with these limits, concerns remain regarding their bioavailability and mobility when applied to the soil [27]. More research is required to understand the bioavailability and mobility of heavy metals from biosolids when applied on land in the Australian context [28].

3.3. Microplastics

Microplastic particles range from 1 mm to 5 mm and can be detected in surface water, soil, sediment, and biota [29]. Microplastics commonly detected in biosolids are generally produced from polyethylene, polypropylene, polystyrene, polyvinylchloride, polyethylene terephthalate, and other polymers [30, 31]. These microplastics originate from the synthetic fibers of clothing and plastics used in personal care products which eventually enter WWTPs and can enter the environment via subsequent application of biosolids to land [32, 33].

Microplastic contamination of biosolids is widespread in Australia. For example, Okoffo et al., (2020) [34] collected biosolids samples from 82 WWTPs across Australia and reported that 99% of samples contained plastics at a concentration between 0.4 and 23.5 mg·kg-1 DBS. Okoffo et al., (2020) [34] further projected that around 4,700 Mt of plastics are released into the Australian environment through biosolids end-use, of which 3,800 Mt is released onto agricultural land.

Microplastics can persist in the environment for decades after their application. Although microplastics are not biodegradable, they are prone to photodegradation and thermo-oxidative degradation [32, 34]. The degradation of microplastics to nanoplastics is a concern for plants and animals [35]. At the nanoscale, plastics can pass through cell membranes and enter the food chain [36]. In general, microplastics and nanoplastics are capable of causing widespread physical and chemical impacts on soil physiochemistry, terrestrial food webs, growth inhibition in earthworms, lethal toxicity to fungi, mammalian lung inflammation and broad cytotoxicity [37].

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3.4. Pathogens

The transmission of infectious pathogens from biosolids to humans, animals or plants is a significant public health concern [19]. Biosolids contain pathogenic microorganisms, including viruses, bacteria, protozoa, and helminths [38]. The pathogen load depends on the treatment and stabilization processes used to produce the biosolids [19]. Moderate applications of biosolids can increase the diversity of the soil ecosystem, as the additional organic matter and nutrient inputs support the growth of microbial populations, leading to an increase in diversity [1, 39]. However, the impact of biosolids on soil microbial diversity is not always positive. For instance, a study conducted by Mossa et al. (2017) [40] found that the increasing application of biosolids resulted in a change in the soil microbial diversity. Soil samples collected from 17 maize fields showed that diversity decreased with increasing zinc (Zn) concentration in soils with more than 1000 mg kg-1 Zn. This indicates that above a certain level of accumulation of biosolids, the positive impact of organic matter on soil microorganisms is offset by the negative effect of high metal contamination [40].

Further inactivation of these pathogens depends on temperature, moisture content, pH, soil type texture and sunlight [41]. While viral and bacterial pathogens will die in 1-3 months, protozoan oocysts and helminth ova can survive in biosolids for up to a year [42]. Overall, the application of biosolids on soil can have a significant impact on soil microbial diversity and abundance, and its effects depend on the amount of biosolids applied, the level of metal contamination in the sewage sludge, and the soil type [39]. However, the lack of data makes it challenging to review viral and protozoan pathogens in biosolids and is worthy of further research [43].

4. Thermal treatment of biosolids

Several factors drive the international uptake of thermal treatment, including current market changes and policy developments, energy generation from waste, waste minimization, and reduced associated disposal costs [44, 45] (Figure 2). Pyrolysis and gasification are the two main thermal processes applied to the management of biosolids and provide two benefits. Firstly, the destruction of POPs [46], microplastics [47] and pathogens [3] and secondly, the technology requires reduced land footprint relative to other, more hazardous, waste management facilities (i.e., landfill or stockpiles) [8].

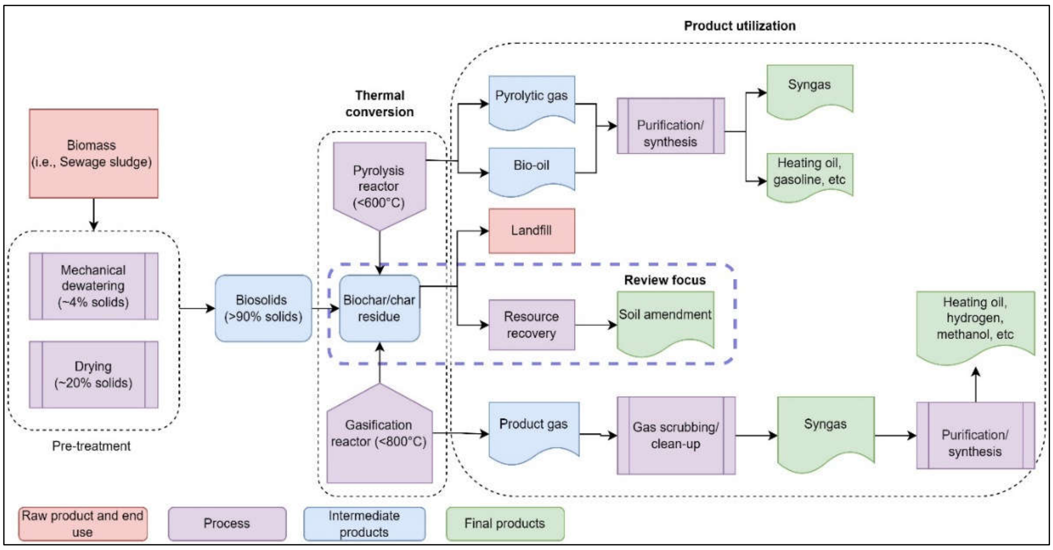

Figure 2: Schematic representation of thermal treatment of biosolids to produce biochar. The blue dotted area illustrates the focus of the literature review.

4.1. Pyrolysis

Pyrolysis involves heating organic materials in the absence of an oxidizing agent in a non-reactive environment (i.e., in the absence of oxygen). Contaminants including POPs, plastics and pathogens are destroyed during three major stages: (i) dehydration and removal of lightweight volatile compounds at 25-200°C; (ii) treatment of low and high molecular weight hydrocarbon complexes occurring at 200-600°C, and; (iii) decomposition of inorganics and formation of stable gases at >600°C [48, 49]. Typical processes require a vapor residence time ranging from 3-1500s [10]. The reaction produces the following products: bio-crude oil, solid biochar, and syngas (Figure 2), with the proportion of the

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products dependent on the pyrolysis method, reaction time, and quality of sewage sludge. Regarding biochar, as the process time and/or temperature increase, the biochar yield decreases [50].

4.2. Gasification

In contrast to pyrolysis, gasification takes place at a much higher temperature ranging from 800-1200°C (Figure 2) and a range of pressures (atmospheric to 35 bar) with controlled introduction of oxygen (~3%) to allow some combustion. Due to the partial combustion of the products of thermal treatment, gasification typically converts organic compounds to 15% biochar and 85% combustible gases which drive the process [51]. Similarly, as with pyrolysis, as process time and/or temperature increase, biochar yield decreases, and the biochar properties depend on the physicochemical properties of the feedstock biosolids. Currently, biochar generated from biosolids can be used for applications in landfill, agriculture, or in construction [11].

Both pyrolysis and gasification of biosolids reduce volumes and masses, minimize the risk of pathogens, and reduce heavy metals and POPs [52]. However, the implementation of these technologies for large-scale application in WWTPs can be hindered by the high capital and operating cost [53, 54].

5. Biosolids-derived biochar

5.1. Physicochemical characteristics of biosolids-derived biochar

The physiochemical characteristics of biosolids-derived biochar are highly variable and depend on the composition of the input feedstock, the thermal treatment process, the temperature, and the residence time [54, 55]. Characteristics of particular interest include biochar yield; surface area; porosity; pH; electrical conductivity; concentrations of C, N & H; and N & P content. Figure 3 presents data related to the variation in BDB properties as a function of the temperature of pyrolysis/gasification. The data were compiled using UC Davis Biochar [56] and data from published peer-reviewed articles worldwide. The complete data sets used are presented in the supplementary material (Table S1).

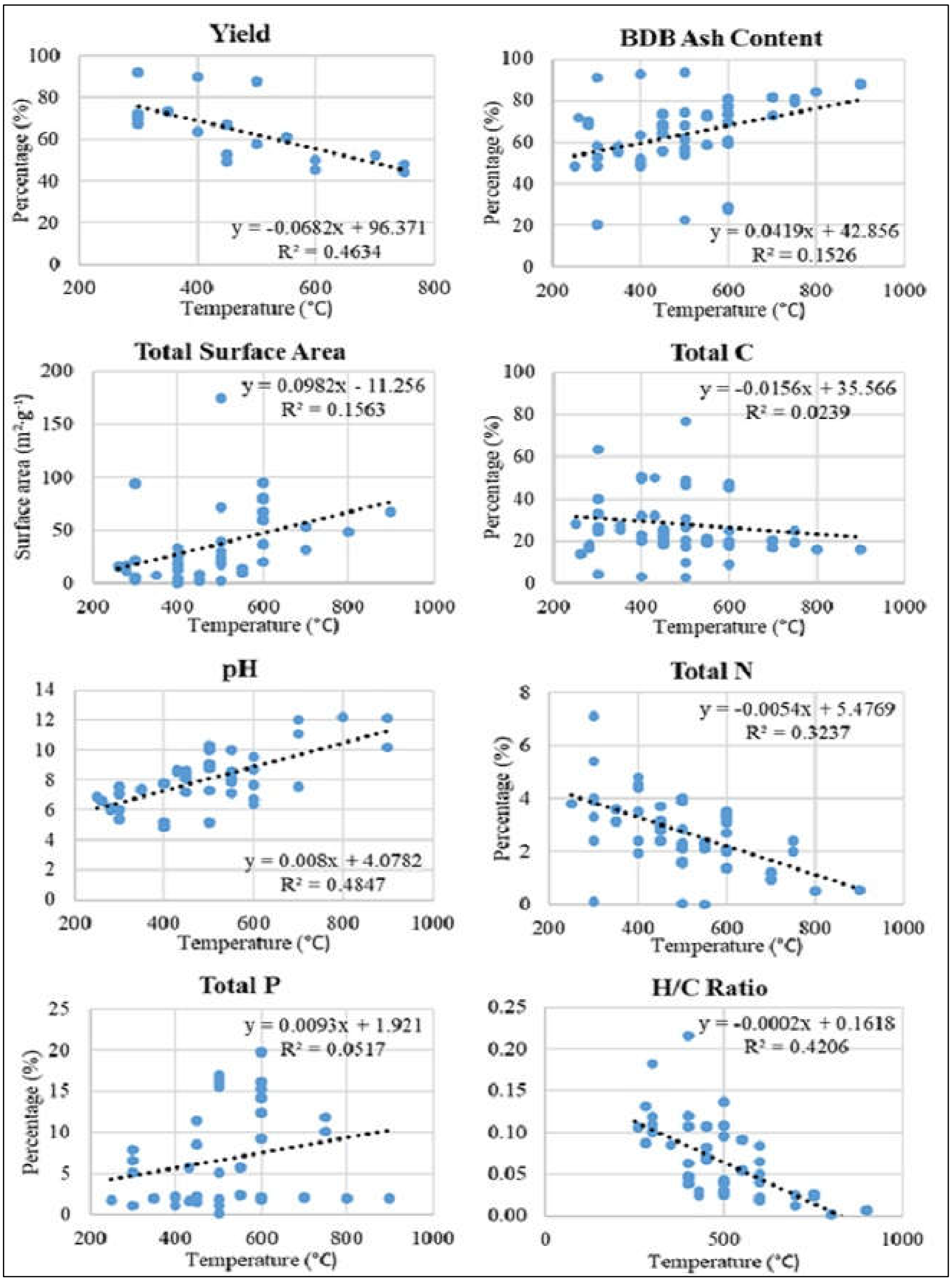

Figure 3: Change in the BDB properties as a function of temperature.

5.1.1. Biochar yield

While significant mass reduction of biosolids is achievable, the amount of biochar produced varies significantly depending on the production procedure and source properties [55, 57]. During thermal treatment, the high organic content of biosolids is transformed and fixed in stable carbon phase [58]. The decrease in yield is attributed to the volatilization of hydrocarbons and gasification of the carbonaceous compounds at high temperatures [55]. The relative ash content of biochar increases with pyrolysis residence time and temperature, which is expected as ash remains in the solid fraction while organic matter undergoes thermal decomposition [59-61]. Due to the elimination of volatiles, some of the nutrients and metals contained in feedstock biosolids become concentrated in biochar [62].

5.1.2. Surface area & porosity

Surface area and porosity play a crucial role in biochar applications, such as wastewater treatment and soil remediation. These properties are decisive to the quantity/quality of the available active sites in biochar and therefore enhance other biochar properties such as cation exchange capacity, water holding capacity, and adsorption capacity [63, 64]. The surface area and porosity of BDB are interlinked [65], and generally increases with process temperature due to three factors: 1) an increasing degree of aromatization and rearrangement in the chemical compounds [66]; 2) mass loss during thermal decomposition due to the liberation of water and volatile matter [67]; and 3) the volatilization of moisture content in biosolids could create micropores in biochar [68]. However, under extreme temperature, the surface area decreases and is likely due to the destruction of porous structure and development of deformation, cracking, or blockage of micropores in BDB [69, 70].

5.1.3. Electrical Conductivity and pH

The electrical conductivity (EC) and pH of biochar influence the mobility of macro- and micro-nutrients and heavy metals [87]. Electrical conductivity indicates the content of soluble salts. Biochar's high in ash content typically contains proportionally higher concentrations of salt ions. These salt ions act to reduce the exchangeable hydrogen and aluminum ions in the soil. Consequently, this has the effect of increasing the soil pH [87]. As treatment temperature increases, the EC of the material reduces dramatically, particularly with temperatures > 500°C [55, 59, 88]. Biochar EC correlates better with feedstock type than pyrolysis temperature because it is a function of ash content and elemental composition [89, 90].

Biochar pH influences the mobility of macro- and micro-nutrients, and heavy metals. In contrast with EC, resulting biochar pH increases with temperature from around pH 7 at 300°C to pH 10-12 at 900°C (Table 1, Figure 3) [55, 58, 91]. At temperatures higher than 550°C, cations such as Ca, K, Mg, Na and Si present in the biosolids will form carbonates and oxides leading to an increase in pH [92]. As pH increases, heavy metals become reduced and are present in residual phases or bound to carbonates, oxides, and organic matter [87].

5.1.4. H:C molar ratio

Biosolids-derived biochar is very stable. Estimates of the mean residence time of BDB in soil are in the order of 2000 years [93]. The molar H:C ratio is an indicator of this stability. More specifically, the ratio is an indicator of the degree of carbonization that can be used to characterize the degree of aromaticity (i.e. the degree to which aromatic rings are connected in two- and three-dimensional dimensions) of the biochar [65, 94]. This is indicated by a reduction of H relative to C, indicating increased aromatization and consequently increased chemical stability [94].

Consequently, biochar stability increases as the degree of aromatic condensation increases [95]. H and C concentration decreases significantly with increases in process temperature (Table 1). This occurs primarily due to the volatilization of elements as CO, CO_2, H2O, and hydrocarbons [19]. Additional losses of H occur due to the reduction of hydroxyl (OH-) functional groups, dehydration, and condensation in the thermal treatment processes [96].

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Table 1: Chemical analysis of biochar derived from biosolids at different temperatures. Results reported as average and (standard deviation).

Technology	Sample[a], Temp°C	pH	Elemental Analysis (%)			Nutrient Composition ($g \cdot kg^{-1}$)					
			C	H	N	Ca	Fe	K	Mg	P	S
Pyrolysis[1]	BS 25	5.1	25.6	4.1	3.0	26.5 (19.4)	37.0 (22)	4.1 (3.3)	8.1 (9.9)	28.5 (6.8)	23.2 (24.8)
	BDB 300	5.9 (0.6)	23.1 (2.7)	2.7 (0.8)	3.0 (0.6)	31.24 (24)	44.01 (30.4)	4.17 (3.2)	10.18 (12.8)	32.89 (8.2)	23.23 (1.9)
	BDB 400	6 (1.3)	19.9 (0.4)	1	2.2 (0.3)	42.13 (19.7)	48.94 (35.5)	6.52 (3.5)	13.31 (13.4)	32.83 (8.7)	28.46 (26.5)
	BDB 500	7.1 (0.5)	15.3 (5.1)	0.9 (0.8)	1.0 (0.8)	40.41 (32.6)	54.72 (41.6)	5.12 (4.5)	13.19 (17.4)	41.83 (14.9)	24.43 (29.94)
	BDB 550	7	18.6 (12.5)	0.8 (0.2)	2.5 (0.5)	-	-	-	-	-	-
	BDB 600	8.7 (0.7)	-	-	-	24	41.7	13.3	7.86	45.1	-
	BDB 700	9.6 (2.0)	13.9 (5.6)	-	1.0 (0.3)	48.96 (21.7)	60.66 (43.3)	12.35 (6.0)	13.99 (12.7)	40.92 (7.8)	35.1 (37.7)
	BDB 900	11	5	-	0	71.82	33.37	9.83	29.06	40.65	9.69
Slow pyrolysis[2]	BS 25	7.1	25.6	4.5	4.5	42.4 (23.6)	30.4 (28.0)	5.1 (2.6)	9.3 (5.9)	38.7 (9.2)	20.9 (10.7)
	BDB 300	7.3 (0.2)	27.5 (4.7)	3.1 (0.3)	4.5. (0.9)	25.76 (28.7)	7.10 (2.9)	3.5 (2.6)	12.40 (7.4)	49.69 (21.6)	7.92 (3.0)
	BDB 400	7.3 (0.2)	22.2 (5.6)	1.9 (0.2)	3.6 (0.8)	7.43 (5)	-	2.17 (0.2)	9.10 (4)	42.03 (15.1)	6.07 (0.6)
	BDB 450	-	22.5 (4.1)	1.7 (0.1)	3.4 (0.5)	-	-	-	-	-	-
	BDB 500	7.4 (0.3)	22.2 (4.0)	1.2 (0.6)	2.8 (1.1)	56.47 (48.5)	63.8 (47.5)	7.59 (5.2)	13.56 (9)	56.73 (19.8)	19.73 (16.9)
	BDB 600	9.6 (1.6)	22.2 (3.9)	0.9 (0.3)	2.6 (0.9)	58.96 (42.5)	48.8 (50.5)	8.32 (4.9)	17.85 (13.5)	68.93 (2.9)	15.6 (13.9)
	BDB 700	12.5 (0.4)	22.5 (3.6)	0.5 (0.1)	2.3 (0.4)	93.05 (24.5)	51.93 (53.4)	11.98 (2.9)	20.42 (9.4)	83.63 (24.7)	24.08 (20)
Fast pyrolsis[3]	BS		43.40	6.99	5.66	27.1	8.5	5.9	6.0	23.9	10.1
	BDB 400	-	29.9	1.1 (0.6)	2.5 (1.4)	-	-	-	-	-	-
	BDB 500	8.8	19.7 (3.14)	1.1 (0.6)	2.5 (1.4)	73.2 (19.8)	28.8 (3.2)	13.2 (6.7)	17.2 (3.6)	46.6 (40.2)	-
	BDB 600	9.5	19.5 (1.6)	0.6 (0.6)	2.3 (1.3)	62.71	33.60	8.40	15.45	18.76	-
	BDB 700	11.1	16.9	0.2	1.0	64.37	35.32	9.30	16.36	20.35	-
	BDB 800	12.2	16.2	0.0	0.5	65.83	35.76	9.20	16.57	19.35	-
	BDB 900	12.2	15.9	0.1	0.5	69.56	37.20	8.60	17.52	20.23	-
Flash Pyrolysis[4]	BDB 350	7.7	20.5	2.4	8.2	17.07	0.4	13.52	9.88	24.12	-
	BDB 400	-	15.4	1.6	6.6	-	-	-	-	-	-
	BDB 450	-	12	1.2	5.9	-	-	-	-	-	-
	BDB 500	-	12.6	1.2	3.9	-	-	-	-	-	-

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	BDB 550	-	10.9	0.9	4	-	-	-	-	-	-
	BDB 650	-	10.3	0.7	0.7	-	-	-	-	-	-
	BDB 700	8.7	10	0.5	ND	5.35	ND	23.20	13.6	22.89	-
	BS	-	-	-	-	51	30	5	6	40	8
Two stage	BDB 850	-	5.8	-	0.1	14	7.5	15	17.0	11.2	20
LT-CFB[5, b]	BDB 750	-	7.2	-	0.6	13	8.1	15	17.0	11	10
Gasification[6]	BS	-	-	-	-	49.7	38.7	3	9.6	41.8	9.5
	BDB 700	12	22.3	0.77	1.9	11	8.8	7.6	24.5	10.2	-
	BDB 900	12	2.9	0.18	0.25	14.5	11.9	10.9	35.1	14.2	-

[a]BS – biosolids; BDB – biosolids-derived biochar; [b]LT-CFB - Low temperature circulating fluidized bed; ND – not detected.

[1](Hossain et al., 2011 [55]; de Figueiredo et al., 2020 [71]; de Figueiredo et al., 2019a [72]; de Figueiredo et al., 2019b [73]; Yuan et al., 2015 [74]; Vaughn et al., 2018 [75]; Yang et al., 2018 [76]) ; [2](Roberts et al., 2017 [77]; Lu et al., 2013 [78]; Zielińska & Oleszczuk, 2015[79]); [3](Chen et al., 2014 [80]; Barry et al., 2019 [81]; Zhou et al., 2018 [82]);[4](Piskorz et al., 1986 [83]; Uchimiya et al., 2015 [84]); [5](Thomsen et al., 2017b [85]; Hernandez et al., 2011 [86]).

5.1.5. Nitrogen, phosphorus, and other nutrients

Nitrogen, alongside phosphorus, is important for determining the fertilizer value of biosolids-derived biochar but experiences significant losses during thermal treatment (Table 1) [82]. Most nitrogen is lost due to volatilization of the different nitrogen groups (i.e., NH4-N or NO3-N) at low temperatures [50], and with temperatures above 600°C, nitrogen is gradually transformed into pyridine-like structures [80, 97]. Thomsen et al. (2017a) [98] operated numerous thermal technologies across a temperature range of 600-850°C, both with and without oxidation. Without oxidation, nitrogen content decreased from 3.7% in DBS to 2.2% in BDB at 600°C, 0.6% at 750°C, and 0.1% at 850°C. In contrast, the addition of oxidation at 600°C resulted in nitrogen content of 0.1% in BDB, which decreased further to 0% at subsequent temperatures. Consequently, a low process temperature without oxidation should be used if biochar with high nitrogen retention is sought [98].

Conversely, while there appears to be a loss of phosphorus during thermal treatment [55], total phosphorus concentration in biochar generally increases with process temperature (Table 1) [85]. Thomsen et al. (2017a) [98] measured an increase in total P from 4% in DBS to around 8% in BDB formed at 600 ° C and to 11% in BDB formed at 750 ° C. This increase could be due to the increased contact of Ca, Mg and P upon the transformation of organic matter in the biosolids, which would lead to the formation of insoluble Ca-P and Mg-P compounds [59]. However, while total P increases, the available fraction of phosphorus (Colwell P) decreases with increasing process temperature [55, 59]. This relatively unavailable P is expected to become available over time slowly [85].

There are several other agronomically essential nutrients contained within BDB. While the total nutrient concentrations of K, Ca, Mg and Fe typically increase with increasing temperature [55, 98], the total H:C ratio and sulfur decreases [99] (Figure 3 and Table S1 in supplementary materials).

5.2. Contaminants in biosolid-derived biochar

5.2.1. Fate of Heavy Metals in biosolids-derived biochar

Heavy metals and metalloids contained within biosolids are either volatilized during thermal treatment or become concentrated in the biochar product [100]. Mercury, for example, has a low boiling point, and at temperatures above 500°C, almost all mercury can be volatilized during pyrolysis [76] (Table 1). Furthermore, Hossain et al. (2011) [55] observed enrichment of Pb, Ni and Cr in the biochar at temperatures of up to 500°C, followed by a decrease in concentration at 700°C, indicating partial loss of these metals at elevated temperatures. Metals that remain are commonly concentrated and converted to more stable to more chemically stable forms during thermal treatment [101]. Consequently, the focus has shifted to understanding the conversion of stabilized heavy metals into bioavailable forms and the subsequent mobility of heavy metals in a soil environment [82, 93].

High-temperature thermal treatment reduces the ability for heavy metals to leach from biochar into soils, and this phenomenon increases with temperature [66, 77, 87]. These BDB have high pH and CEC values (Table 1) along with more chemically stable heavy metal fractions that result in unfavorable conditions for leaching (Table 1) [102]. As a secondary effect of pH increasing with process temperature, heavy metal solubility decreases with increases in pH. Devi and Saroha (2014) [102] demonstrated that pH has a strong effect on water-soluble heavy metals, whereby the extractable rates of Pb, Zn and Cu decreased from 16%, 82% and 43% in sewage sludge to 1%, 2% and 2% in biochar, respectively, as pH increased from 3 to 7.

Consequently, heavy metal bioavailability is also typically reduced by thermal treatment and attributed to reductions in soil pH and the physical changes which both the heavy metals and biochar [103-105]. Yang et al. (2018) [76] pyrolyzed eight biosolids from four different wastewater treatment plants in southeast Melbourne, Australia. They produced biochar at two different temperatures (500 and 700°C) with residence times of 5 hours and a heating rate of 5°C min^{-1}. The concentrations of plant-available Cd, Cu, Pb, and Zn decreased by 93%, 84%, 98% and 86% respectively. In this case, treatment at 700°C was no more beneficial than 500 ° C. However, Yang et al. (2018) [76] declared that the DTPA method used to estimate plant-available heavy metal content extracts both readily exchangeable and more persistently bound heavy metals. Although the magnitude of reduction in plant-available heavy metals is large, these values may under-represent the benefit of thermal treatment.

Similarly, to Yang et al. (2018) [76], Hossain et al. (2011) [55] thermally treated biosolids from a Sydney (NSW, Australia) WWTP at 300, 400, 500 and 700°C with an unreported dwell time. Elements including Cu, Cd and Zn were extracted with DTPA to estimate their plant-available fractions. Copper initially experienced a decrease of at least 99% at a temperature of 300°C. However, when exposed to 400 and 500°C, Cu experienced a decrease of only 35% and 24%, respectively, before decreasing back to 99% at 700°C. Cadmium saw a similar effect at 400°C, displaying an increase in availability over the feedstock by 33%, while at all other temperatures, Cd was below the limit of detection, with an apparent decrease in the availability of at least 93%. By comparison, Zn followed a temperature-dependent reduction in plant-availability of 52%, 72%, 82% and 100% at 300, 400, 500 and 700°C, respectively [55]. Unfortunately, without a dwell time, it is difficult to compare results.

For international comparison, Lu et al. (2013) [78] pyrolyzed biosolids from 3 different wastewater treatment plants in China at 300, 400, and 500 degrees with a dwell time of 2 hours and a heating rate of 10°C·min^{-1}. Heavy metal bioavailability was in the range of 0–4%, 0–9%, 0–3%, 0–2%, 0–4% of total concentrations of Pb, Zn, Cu, Fe, and Mn, respectively (Table 2). DTPA-extractable heavy metals increased at higher treatment temperatures. Across the three WWTPs, a treatment temperature of 300°C resulted in an average reduction of plant-available extract by 99%, decreasing to 88% at 400°C and 89% at 500°C (Table 2).

The optimum temperature and dwell time appear to be somewhat feedstock specific. For example, both Yang et al. (2018) [76] and Lu et al. (2013) [78] produced no added benefit from additional treatment temperature (Table 1), while the results from Hossain et al. (2011) [55] indicate a higher treatment temperature is more effective at reducing heavy metal bioavailability in the biochar product. Therefore, independent feedstocks should be evaluated for optimum treatment temperature to maximize heavy metal immobilization while ensuring unnecessary energy expense.

Although there are competing results from various investigations, thermal treatment of biosolids can immobilize most of the heavy metals in the resulting biochar, and the expected environmental risk is low. However, data explaining the change in heavy metal and metalloid availability that occurs during thermal treatment is scarce [93]. Consequently, the detailed mechanism of how thermal treatment temperature influences the distribution and fraction transformation of heavy metals in sewage sludge still needs further investigation.

5.2.2. Fate of organic pollutants and microplastics in biosolids-derived biochar

Although biosolids are essential vectors for the transfer of POPs and microplastics to the environment, both can be destroyed by thermal treatment. Ross et al. (2016) [114] demonstrated that 2.5 minutes of pyrolysis at 500°C eliminates some common pollutants, including triclocarban and triclosan from the biochar product. At a temperature of 500°C, the removal rate of POPs, specifically dioxins (PCDD/PCDF), was 97% in sewage sludge [115]. Conversion of biosolids to biochar reduced PAH content by 95% [79]. Thermal degradation of PAH is further supported in Table 2. Thermal treatment is a promising technology for the decomposition of microplastics at higher temperatures [116]. Ni et al. (2020) [47] reported that the microplastic concentration in BDB decreased significantly from 550 to 960 particles per gram to 1.4-2.3 particle per gram with an increase in the pyrolysis temperature up to 500°C. According to Ni et al. (2020) [47], thermal treatment of biosolids at high temperatures (>450°C) can reduce microplastic concentration by 99%. A recent case study summarized evidence on this topic covering 20 studies and more than 100 different organic pollutants and concluded that pyrolysis reduces the concentration of organic contaminants with an efficacy of >95% to >96% in most cases [117].

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While pyrolysis has been demonstrated to be an effective method for removing organic contaminants, it is important to ensure the quality of biochar product meets the established guidelines. This may require an approval process that includes not only chemical analyses, but also bioassays to test the ecotoxicity to soil, water organisms and plants.

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Table 2: Heavy metals and organic pollutants in biosolids and biosolids-derived biochar and their allowable range according to guidelines.

Guidelines	**Sample**	**Temp °C**	**Total heavy metals ($mg\cdot kg^{-1}$ DBS.)[b]**								**Total PAHs $\mu g\cdot kg^{-1}$ d.b.**	**Reference**
			As	**Cd**	**Cr**	**Cu**	**Pb**	**Hg**	**Ni**	**Zn**		
AWA-Biosolid	-	-	20-30	1-20	100-600	100-2000	150-420	1-15	60-270	200-2500	-	Natural Resource Management Ministerial Council, 2004 [12]
IBI -Biochar	Category A	-	13	1.4	93	143	121	1	47	416	6000	International biochar initiative, 2015 [106]
	Category B		100	20	100	6,000	300	10	400	7400	300000	
EBC-Biochar	Premium	-	13	1	80	100	120	1	30	400	4000	Schmidt et al. (2013) [107]
	Basic		13	1.5	90	100	150	1	50	400	12000	
Technology												
Pyrolysis	BS	N/A	-	2.3-5.3	-	401-611	136-224	-	-	629-1238	-	Lu et al. (2013) [19]
	BDB	300	-	3.3-7.5	-	480-043	190-350	-	-	849-1909		
	BDB	400	-	3.8-9.8	-	549-1198	194-438	-	-	912-2104		
	BDB	500	-	4.3-8.9	-	565-1267	212-506	-	-	1014-2305		
Pyrolysis	BS	N/A	-	7.54	-	545	189	-	102	2398		Méndez et al. (2013) [88]
	BDB	400	-	9.67	-	632	239	-	129	2983	-	
	BDB	600	-	9.76	-	740	253	-	134	3922		
Gasification	BS	N/A	-	1.0-2.5	34-66	-	41	1.5	24	-	-	Thomsen et al. (2017b) [85]
	BDB	750	-	1.5-5.5	80-182	-	84-110	0.2	87-158	-		
Gasification	BS	-	-	0.93	80.8	580	78.27			402	-	Li et al. (2012) [108]
	BDB	350		1.5-1.6	218-227	851-900	114-121			597-623		
	BDB	400		1.5-1.7	228-247	886-922	120-125			612-637		
Gasification	BS	-	-	1	36 (7)	529 (8)	45	2	66(2)	423(10)	-	Hernandez et al. (2011) [86]
	BDB	700		ND	98 (1)	1159 (8)	88(1)	ND	122(1)	753 (5)		
	BDB	900		ND	104 (2)	1346 (6)	51(1)	ND	165(4)	757 (4)		
Pyrolysis	BDB	200	7.6-16.7	2-9.1	67.6-281	712-1000	28.4-60		65-635	1964-2940		Waqas et al. (2015) [109]
Pyrolysis	BDS	25	-	1.0	173	143	51.1		42	698	3339	Luo et al. (2014) [110]
	BDB	200		1.1	180	149	54.7		41.1	735	1644	
	BDB	500		1.4	233	193	67.9		55.1	887	70385	
	BDB	600		1.1	239	198	69.1		56.1	976	1241	
	BDB	700		0.7	247	202	74.2		55.2	986	179	
Pyrolysis	BS	25	-	3.6	-	487	167	-	-	922	-	Lu et al. (2013) [78]
	BDB	300		5.5		733	260			1417		
	BDB	500		6.5		841	506			1705		
Pyrolysis	BS	-	2.6	1.7	-	160	44	-	-	1200	3860	Waqas et al. (2014) [111]
	BDB	550	12	2.7		210	82			2080	900	
Pyrolysis	BS		2.3	1.5	-	171	53.8	-	-	1105	5780	Waqas et al. (2015) [109]
	BDB	550	11.9	2.3		237	71.9			1879	1701	
Pyrolysis	BS	Air	18	ND	20	165	42		23	703	-	Song et al. (2014) [60]

	BDB	400	9.4	3.2	60.7	357	83		77.1	1478		
	BDB	500	14	3.2	61	334	92.6		68.4	1704		
Pyrolysis	BDB	550	9.3	3.7	74.1	222	27		34.5	1102	-	Khan et al. (2013a) [112]
Pyrolysis	BS	-	-	-	-	-	-	-	-	-	2950	Khan et al. (2013b) [113]
	BDB	500									4350	
Pyrolysis	BS	-	-	-	-	-	-	-	-	-	8625-13333	Tomczyk et al. (2020b) [68]
	BDB	500									612-766	
Technology	**Sample**[a]	**Temp**	**Available heavy metals (mg·kg⁻¹ DBS**[b]**)**									**Reference**
		°C	**As**	**Cd**	**Cr**	**Cu**	**Pb**	**Hg**	**Ni**	**Zn**		
Pyrolysis	BS	25	-	7.80	9	700	309	-	135	3565		Luo et al. (2014) [110]
	BDB	300		0.45	11	45.5	48		20.5	280		
	BDB	500		2.30	9	205	27.5		25	385		
	BDB	600		5.90	8.5	295	67		37	635		
	BDB	700		10.5	8	365	115		46.5	970		
Pyrolysis	BS	25	-	1.8	-	139	34.9	-	-	586.6		Lu et al. (2013) [78]
	BDB	300		ND		1.7	ND			4.5		
	BDB	500		ND		0.4	6.5			50.8		
Pyrolysis	BS	-	1.1	1.1	-	37	8.2	-	-	371		Waqas et al. (2014) [111]
	BDB	550	0.04	0.2		3.4	2.5			66		
Pyrolysis	BS	-	1.07	1.03	-	35.3	9.02	-	-	387		Waqas et al. (2015) [109]
	BDB	550	0.05	0.17		4.35	3.41			56.7		
Pyrolysis	SS	Air	-	-	-	-	-	-	-	-		Song et al. (2014) [60]
	BDB	400	0.9	ND	0.2	0.3	0.5		0.3	7.9		
	BDB	500	0.6	ND	ND	0.2	0.6		ND	1.8		
Pyrolysis	BDB	550	0.04	0.26	1.24	6.5	2.13		2.26	127		Khan et al. (2013b) [113]
Gasification	BS	-	-	0.62	1.26	22.63	2.74	-	-	112		Li et al. (2012) [108]
	BDB	350		0.03-0.12	1-3.91	0.42-1.17	0.58-1.13			7.67-17.19		
	BDB	400		0.01-0.24	1.2-7.51	0.37-0.97	0.59-1.40			9.05-12.25		
Gasification	BS	-	-	-	8.89	16.3	-	-	3.44	-		Hernandez et al. (2011) [86]
	BDB	700			0.06	0.49			0.04			
	BDB	900			0.04	2.08			<0.01			

[a]BS – biosolids; BDB – Biosolids- derived biochar; [b]DBS – dry biosolids; N/A- not applicable; ND – not detected

6. Use of biosolids-derived biochar as a soil amendment

The current understanding of the agricultural effects of biosolids-derived biochar in Australian agricultural soil is limited and is primarily based on few biomass feedstock materials. Furthermore, commercial biochar in Australia is marketed with only limited (or without) analytical data for the biochar [118]. For the land application of biochar, it is vital to know the composition of the biochar and, consequently, the properties of soils used [119]. Thus, international experiences do not necessarily apply to Australian soils. Consequently, research and development must be undertaken to integrate information on Australian soils into management decisions.

There are no legislative standards available in Australia that prescribe limits for the concentrations of heavy metals in biochar intended for soil application. Regulations and standards for composts and biosolids in Australia are based upon an assessment of the total concentration of metals in the material, without any consideration of their mobility in soil and bioavailability. Consequently, inappropriate regulation may limit the use of these nutrient-rich bioresources [93]. Voluntary biochar quality standards exist in Europe i.e., European Biochar Certificate [107], and in the USA i.e., International Biochar Initiative, and they aim to guarantee the quality of a product. These voluntary schemes define biochar as a material produced by the thermal treatment of biomass under low oxygen conditions, and consequently both these guidelines allow the use of biosolids as feedstocks for biochar production under defined regulation [106]. Importantly, according to these guidelines, organic contaminant and heavy metal concentrations are the major determinants of the end-use of the biochar [106, 107]. However, there are currently no legislative guidelines for the application of biochar in Australia; however, the International Biochar Initiative [106] and European Biochar Certificate [107] guidelines provide a valuable reference to understand if BDB is suitable for land application in terms of heavy metals and total persistent organic pollutants [11].

6.1. Soil effects

Biochar applied to soil can be used for locking carbon in soil, heavy metal immobilization, greenhouse gas reduction and soil water retention [118, 120, 121].

6.1.1. Soil acidity and nutrient leaching

Naturally, high pH and CEC values for BDB can reduce soil acidity, limit nutrient leaching, and heavy metal release in soil. Hossain et al. (2011) [55] demonstrated that by manipulating the temperature of pyrolysis, it is possible to create a range of BDB products with pH values targeted for application in acidic or in alkaline soils. Additionally, the highly negative surface-charge density of biochar enables the retention of cationic nutrients via ion exchange, whereas the relatively extensive surface area, internal porosity, and polarizability facilitate the sorption of anionic nutrients via covalent bonds [122]. Therefore, BDB could adsorb heavy metals and organic contaminants such as pesticides and herbicides from the environment [11].

6.1.2. Soil hydrology

Biosolids-derived biochar has both a high specific surface area and porosity, which could represent an improvement in soils nutrient status and physical properties such as water retention and hydraulic conductivity [123]. The bulk density of biochar is lower than that of mineral soils [124], suggesting that the application of biochar can alter soil hydrology and further increase soil porosity, which can result in long-term impacts on soil aggregation [121, 125]. Méndez et al. (2013) [88] applied the BDB obtained at 600°C at 8% (w/w) application rate and observed increases in soil field capacity from 23% to 29%, and available water increased from 10 to 16%.

Typically, high biochar application rates are necessary to improve soil physical properties, such as water holding capacity or bulk density, i.e., >40 $t \cdot ha^{-1}$ [126]. In specific cases, however, lower biochar application rates (~10 $t \cdot ha^{-1}$) have been shown to improve physical soil properties [127, 128]. There is a lack of research regarding the appropriate level of biochar application for different soil types [118, 129].

6.1.3. Greenhouse gas emissions

Organic materials, such as sewage sludge, added to the soil result in N2O emissions that are sometimes far greater than equivalent amounts of chemical fertilizer [130, 131]. Van Zwieten et al. (2010) [131] demonstrated that if biosolids are processed via slow pyrolysis, they do not pose the same greenhouse gas risk as untreated organic material. Biosolids-derived biochar were effective in reducing overall emissions of N_2O compared with the control soil. The control soil that received an equivalent 165 kg N (in the form of urea) released 15% of this N as N_2O, while amendment of the soil with 5% BDB resulted in only 2% of the N being converted into N_2O (i.e., an 84% decrease). Grutzmacher et al. (2018) [132] conducted an incubation experiment in which they applied a range of biochar from different feedstocks to the soil and investigated the potential of biochar to reduce fertilizer induced N_2O emissions. When ammonium nitrate was co-applied with biochar, the smallest emission was observed in soil amended with BDB, which reduced the N_2O emission by 87% [132].

6.1.4. Soil nutrients, soil organic matter, and soil carbon

Pyrolysis makes biosolids very stable against chemical and biological degradation, and biosolids-derived biochar in the soil can store carbon in the form of stable structures for centuries. de Figueiredo et al. (2019a) [72] evaluated the effects of applying BDB in combination with mineral fertilizer on soil organic carbon fractions (SOC). They demonstrated that the increase of organic C in the soil promoted by biochar varies with the pyrolysis temperature employed [51]. The biochar produced under lower pyrolysis temperature (300°C) affected the more labile fractions of soil organic matter (SOM), whereas the biochar produced under higher pyrolysis temperature (500°C) influenced the more stable fractions of SOM [72]. These differences among biochar greatly influence their mineralization rates, nutrient release, and C accumulation in the soil [133]. Considering the importance of equilibrating the supply of C in both labile and stable forms of SOM, the biochar produced at 300°C pyrolysis temperature presents great potential to be used for agro-environmental purposes [72]. Additionally, BDB is beneficial for the soil microbiota. Carbonized organic matter represents energy for microorganisms that inhabit the soil [134], and its application to the soil increases soil microbial activity [135, 136]. Furthermore, the high surface area and porosity increase microbial activity by promoting optimal growth conditions [137].

Compared to biochar derived from plant residues, BDB generally contains a higher level of nutrients [138]. Additionally, the high porosity increases the surface area in the structure of the material. It facilitates the adsorption of both hydrophilic and hydrophobic molecules [62], which subsequently improves nutrient retention [60]. In one of the first studies in Australia, Bridle and Pritchard (2004) [139] investigated the effect of BDB on N and P recovery in an incubation experiment over eight weeks. Water-soluble N was retained in the biochar. Biosolids-derived biochar did not initially increase soil mineral N levels, as observed with land application of biosolids, although soil bicarbonate-extractable P levels gradually increased. This study demonstrated that nitrate and ammonium concentrations did not increase in soil within 56 days after application, suggesting that land application can minimize the risk of nitrogen leaching [139].

Biochar also provides a source of P for plant growth and could have applications on soils as a slow-release form of P [139]. Biosolids-derived biochar can be utilized as a reservoir of P for soils, and that a certain fraction of this P is in a suitable form available for plant uptake [59, 140].

6.2. Crop effects

6.2.1. Crop yield

All the above-mentioned soil impacts play an important role in promoting crop yield (Table 3). Sousa and Figueiredo (2016) [141] reported enrichment of nutrients in soil treated with BDB, especially P, available N and exchangeable cations (Ca and Mg). This enriched soil promoted the development of radish plants with increased plant height, above-ground dry weight and number of leaves at different rates of BDB application. Furthermore, Hossain et al. (2010) [142] studied the use of BDB on the production of cherry tomato and found the addition of biochar (10 $t \cdot ha^{-1}$) increased the average dry weight of shoot production from 62 to 74 $g \cdot plant^{-1}$, and increased yield by 64%.

The interaction between soil and BDB can alter over a long period of time. An extensive literature search revealed limited investigations that demonstrated long-term impacts of BDB on soil and crop yield (Table 3). Faria et al. (2018) [93] conducted a two-year field experiment which resulted in increased soil fertility, mainly P, Mg, Cu, and Zn, and an increase in CEC, while soil K was not affected. Increased soil fertility resulted in greater crop yield, especially in the second cropping season. Figueiredo et al. (2020) [143] investigated the direct (first and second cropping season) and residual (third and fourth cropping season) effects of BDB on soil P fractions, P uptake and corn grain yield. Positive effects of the trial were observed on corn yield and P content in soil. BDB also maintained a high soil P content for two

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years without re-application, indicating that BDB can behave as a slow-release P-fertilizer [143]. Given that there are limited long-term studies, it is challenging to assess the long-term effect of BDB when applied to land. Despite the increasing research effort in recent years in this area, a sound understanding of the relationship between desired biochar characteristics and production conditions and feedstock is still lacking. Further work is needed, especially to identify which combination of feedstock and treatment conditions would provide the most appropriate properties for biochar as a soil amendment [65].

6.2.2. Bioavailability and bioaccumulation of pollutants

The main limitation in using biosolids and BDB as a soil amendment is the presence of heavy metals and PAH (Table 2). To cause a toxic effect, heavy metals must dissolve in soil solution, be taken up by organisms, and transported to cells where a toxic effect can occur [147]. Through conversion of biosolids to biochar, it is possible to decrease PAH concentrations (Table 2) and the bioavailability of heavy metals (Table 4). Waqas et al. (2014) [111] conducted research on contaminated soil from farmland near an iron refinery plant in Fujian Province, China, in which the researchers applied both biosolids and BDB. The conversion of biosolids to biochar significantly decreased the concentration of PAH and available heavy metal concentration (Table 2). Additionally, the application of BDB to soil was much more effective in reducing the availability of PAHs and heavy metals than biosolids, and therefore reducing pollutant transfer from soil to water and subsequently to plants. Consistent with these observations, plants with biochar application were less prone to PAH accumulation. Studies that involved growing lettuce [113], tomatoes [109], and cucumber [111] with biosolids and BDB, revealed that PAH concentration in plant biomass was lower in the biochar trials (Table 3).

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Table 3: Effect of biosolids-derived biochar on soil physicochemical characteristics, crop yield and heavy metal bioaccumulation. Thermal treatment process used to biochar from biosolids was pyrolysis.

Temp °C	Plant species	Soil fertility	Agronomic performance		Reference
			Crop yield	Heavy metals bioaccumulation	
300	Radish	Increased soil base saturation, CEC, available P, Ca, and Mg, except K. Soil pH was not affected.	Increased plant height, yields, and above-ground dry weight.	-	Sousa & Figueiredo (2016) [141]
450	Wheat	Increased soil CEC, K, and available P.	Increased plant height, biomass, and grain yield.	-	Rehman et al. (2018) [46]
500	Rice	Increased pH, EC, total N, C and available P and K. Availability of heavy metals in the soil was reduced.	Increased shoot biomass, grain yields, and above-ground dry weight.	Reduced bioaccumulation of As, Co, Cr, Cu, Ni and Pb in rice grains, stems, and leaves.	Khan et al. (2013a) [112]
400-550	Garlic	-	Increased average plant height, plant biomass (stem and leaves) and garlic yield when compared with control.	No heavy metal accumulation was found in stem and leaves. Although, higher Zn and Cu content was found in roots and bulbs compared to the control.	Song et al. (2014) [60]
550	Coolatai grass	-	Increased grass yield was observed, specifically when biosolids-derived biochar was combined with chemical fertilizer.	-	Hossain et al. (2015a) [144]
550	Cherry tomatoes	-	Increased plant height and fruit yield.	Heavy metals concentrations in the fruits were lower in the biochar treatment than the biosolids treatment.	Hossain et al. (2015b) [145]
550	Cucumber	-	Increased plant biomass and fruit yields	Reduced bioaccumulation of As, Cu, Cd, Zn and Pb in the fruit when compared to the biosolids treatment.	Waqas et al. (2014) [111]
200-700	Turf grass	Increased soil organic carbon, total N, available P and K, decreased soil pH.	Increased above-ground dry matter and total N, P and K content.	Reduced bioaccumulation of heavy metals was observed in above-ground biomass	Yue et al. (2017) [146]

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In a Mediterranean context, Mendez et al. (2012) [147] evaluated the effects of biochar from pyrolyzed sewage sludge applied on agricultural soil. The evaluated properties included heavy metal solubility and bioavailability in BDB-treated soils compared to those treated with raw sewage sludge. The risk of leaching of Cu, Ni and Zn were lower in the soil treated with BDB than in the sewage sludge treatment [147]. Biochar amended samples also reduced the availability of Ni, Zn, Cd and Pb in plants compared to amended samples of sewage sludge (Table 3, Table 4).

While bioaccumulation of heavy metals in plants grown in BDB is a potentially concerning pathway for them to enter the food chain, the bioavailability of heavy metals represents low risk. Jin et al. (2016) [95] and Lu et al. (2016) [148] reported that although carbonization leads to the enrichment of heavy metals in the matrix of BDB, they exist mostly in oxidizable and residual forms. This results in a significantly reduced bioavailability of these pollutants and presents a very low ecological risk [95]. Hossain et al. (2010) [142] investigated the effect of BDB on cherry tomatoes and concluded that, while heavy metals were taken up by the plant, there was no significant bioaccumulation in the fruit (Table 4). In contrast, an experiment conducted by Song et al. (2014) [60] reported the accumulation of heavy metals, mostly Ni, in garlic tissues in soil amended with BDB. It should be noted that this study used high application rates of BDB (50%), which is unrealistic from an agronomic point of view. However, this does indicate that plants undertake preferential storage of heavy metals in different tissues. More research is required to understand the specifics of preferential heavy metal storage in edible crops. Furthermore, interactions between biochar, soil, microbes, and plant roots are known to occur within a short period of time after application to the soil [121]. But the extent, rates, and implications of these interactions are still far from understood, and this knowledge is needed for an effective evaluation of the use of biochar as a soil amendment [44, 89].

Despite increasing the concentration of total heavy metals in relation to the raw material, pyrolysis reduces the bioavailability of metals [3, 72]. Due to the reduced metal leaching resulting from immobilization during thermal treatment, BDB is generally understood to be safe, and hence, several researchers recommend establishing limit values in Australian regulations on leachability of metals instead of total metal concentrations [76, 77]. For example, in an Australian study by Hossain et al. (2010) [142], 10 t·ha-1 of BDB was used, which were over maximum concentrations allowed by the Australian food standards. Although total metal concentrations in the soil exceeded the guidelines, tomatoes grown in this environment did not result in the accumulation of potentially toxic concentrations of heavy metals (Table 3, Table 4).

Table 4: Heavy metal accumulation in plants. All treatments were applied as % w/w basis and are represented as mg·kg^{-1}.

Plants	Treatments	As	Cd	Cr	Cu	Ni	Pb	Zn	References
Rice grain	Control	0.45	0.4	ND	20	ND	0.95	54	Khan et al. (2014) [149]
	5 % BDB	0.19	0.32	ND	17	ND	0.6	44	
	10% BDB	0.17	0.28	ND	16	ND	0.5	41	
Tomato	Control	0.35	0.26	ND	2.8	ND	0.5	85	Hossain et al. (2010) [142]
	2% BDB	0.17	2.6	ND	4	ND	0.25	20	
	5% BDB	0.16	2.5	ND	2	ND	0.2	12	
	10% BDB	0.12	2	ND	1.2	ND	0.17	8	
Rice grain	Control	0.14	0.02	0.3	4.8	0.68	0.35	8	Khan et al. (2013a) [112]
	5% BDB	0.05	0.12	0.21	4.7	0.55	0.1	26	
	10% BDB	0.04	0.13	0.17	4.6	0.49	0.05	28	
Turnip	2% BDB	0.12	0.11	ND	3.2	ND	0.22	48	Khan et al. (2015) [150]
	5% BDB	0.11	0.1	ND	1.9	ND	0.19	36	
Turf grass	Control	0.14	0	0.19	0.25	ND	0.18	0.59	Yue et al. (2017) [146]
	1% BDB	0.08	0.02	0.08	0.12	ND	0.2	0.23	
	5% BDB	0.03	0	0.04	0.1	ND	0.05	0.11	
	10% BDB	0.07	0	0.06	0.14	ND	0.14	0.18	
	20% BDB	0.06	0	0.05	0.1	ND	0.08	0.11	
	50% BDB	0.05	0	0.04	0.1	ND	0.05	0.05	

[a]BS – biosolids, BDB – Biosolids- derived biochar; ND – not detected

7. Conclusions and future research needs

Options for the beneficial use of biosolids in Australia are centered on application to agricultural land. The presence of contaminants such as heavy metals, persistent organic pollutants, microplastics and pathogens are of concern, and represent a risk to the environment, human and animal health. It is anticipated that measures implemented towards achieving low- or neutral-carbon economy, assisted by technological advances for the treatment of sewage sludge (e.g., improved removal of contaminants and energy recovery from treatment processes), coupled with volatility of fertilizer and energy markets, will stimulate increased uptake of biosolids in Australian agriculture. Increased recycling of biosolids and biosolids-derived products to land may go some way to reduce the reliance on synthetic and mineral fertilizers and help improve the carbon balance of arable land. The use of biosolids is also leaning towards nutrient extraction and power generation, as witnessed, for example, in some European Union countries and the United States.

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This review brought together scientific evidence showing that thermal treatment (e.g., pyrolysis and gasification) of biosolids can be employed to reduce pathogens, microplastics, and organic pollutants load, and decrease the bioavailability of heavy metals maintaining them within environmentally and agronomically safe levels. Where biosolids or biochar are used, on-farm implementation of best (or recommended) management practices for crops, soil and applied nutrients must always be exercised to control such risks. While research into the short-term effects (e.g., <10 years) of biosolids-derived biochar on crop, soil and environment appears to support their use in agriculture, the longer-term effects are less known. Therefore, longer-term studies are required for better understanding the viability of using biosolids-derived biochar (BDB) as a safe soil amendment. The nutrient and contaminant dynamics in soils receiving BDB, and the inherent risk of transferring these contaminants to the food chain need to be determined together with measures to mitigate such risks. Key research gaps identified by this review are summarised below:

1. Explore the potential for cost-effective thermal technology to treat biosolids, including alternatives for recovering energy for electricity generation and conversion of biosolids to biochar.
2. Thermal treatment appears to be effective at eliminating persistent organic pollutants, microplastics, and pathogenic contaminants from biosolids. However, the efficacy of thermal treatment in reducing (or avoiding) soil contamination from these sources is not well documented. This information is critical for supporting the safe use of biosolid-derived biochar as a soil amendment and for removing concerns associated with recycling.
3. There is potential to customize biochar products to suit specific users' needs (e.g., soil and crop type, farm application method), which will require understanding of the relationship between the desired biochar characteristics and the production conditions and feedstock. The optimal combination of feedstock and treatment conditions to match specific crop and soil requirements needs to be determined. Optimizations of the physical and mechanical properties of biosolids-derived biochar will facilitate field application with standard fertilizer applicators, improving field delivery efficiency and logistics, and their acceptability by farmers.

A comprehensive analysis of the strengths, weakness, opportunities, and threats associated with the conversion of biosolids to biochar in the Australian market is discussed in figure 4. The circular economy approach and closing the waste-loop gap are identified as opportunities. However, challenges such as long-term studies, understanding nutrient and contaminant dynamics, and cost of equipment for the thermal treatment are recognized as weaknesses.

Strengths	Weakness
• Reduction in waste volume, which can decrease disposal costs and alleviate pressures on landfills. • Thermal treatment of biosolids can produce energy, which can be used to power the treatment process. • Biosolids-derived biochar has the potential to improve soil health and fertility, leading to increased crop yields and reduced need for synthetic fertilisers. • The production of biochar from biosolids can contribute to the greenhouse gas emissions by sequestering carbon in the soil.	• Concerns about potential contaminates in biosolids may limit public acceptance of the use of biochar in agriculture. • Thermal treatment facilities may require significant capital requirement. • The lack of long-term studies on the effects of biosolids-derived biochar on soil health and contaminant transfer may limit the adoption of the technology.
Opportunities	**Threats**
• The global shift towards a low-carbon economy and increasing demand for sustainable products may create new markets for biosolids-derived biochar. • The ability of biochar to sequester carbon can be used to generate carbon credits, providing additional revenue streams. • The development of custom biochar products tailored to specific crops and soil types may create niche markets and increase demand. • Research into the long-term effects of biosolids-derived biochar on soil health and contaminant transfer can help to address concerns and increase acceptance.	• Fluctuations in the prices of fossil fuels may affect the competitiveness of biochar as the energy source. • Regulatory barriers or lack of clear guidelines for the safe use of biosolids-derived biochar may hinder market growth. • Competition from other waste-to-energy technlogies and alternative fertilisers may limit the market for biosolids-derived biochar • Lack of awareness or education about the benefit of biosolids-derived biochar may limit market uptake.

Figure 4: SWOT analysis of biosolids-biochar conversation on the Australian market.

Abbreviations. BDB: Biosolids-derived biochar; CEC: Cation exchange capacity; DBS: Dry biosolids; WWTP: Wastewater treatment plant; POPs: Persistent organic pollutants; PFOS, PFOA: Perfluorinated group of chemicals; PCBs: Polychlorinated biphenyls; PCAs: Polychlorinated alkanes; PBDEs: Polybrominated diphenyl ethers; PAHs: Polyaromatic hydrocarbons; PBDEs: Polybrominated diphenyl ethers.

Supplementary Materials: The following supporting information attached here: **Table S1:** Variation in BDB properties as a function in pyrolysis/gasification temperature. The data were compiled using UC Davis Biochar Database (http://biochar.ucdavis.edu/) and data from published peer-reviewed articles from around the world.

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