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Potential Environmental Benefits from Blending Biosolids with Other Organic Amendments before Application to Land

Dharini Paramashivam, Nicholas M. Dickinson, Timothy J. Clough, Jacqui Horswell, and Brett H. Robinson*

Abstract

Biosolids disposal to landfill or through incineration is wasteful of a resource that is rich in organic matter and plant nutrients. Land application can improve soil fertility and enhance crop production but may result in excessive nitrate N (NO_3^-–N) leaching and residual contamination from pathogens, heavy metals, and xenobiotics. This paper evaluates evidence that these concerns can be reduced significantly by blending biosolids with organic materials to reduce the environmental impact of biosolids application to soils. It appears feasible to combine organic waste streams for use as a resource to build or amend degraded soils. Sawdust and partially pyrolyzed biochars provide an opportunity to reduce the environmental impact of biosolids application, with studies showing reductions of NO_3^-–N leaching of 40 to 80%. However, other organic amendments including lignite coal waste may result in excessive NO_3^-–N leaching. Field trials combining biosolids and biochars for rehabilitation of degraded forest and ecological restoration are recommended.

Core Ideas

- Landfilling or burning biosolids is an expensive waste of a valuable resource.
- High rates of biosolids restore degraded soil but cause excessive nitrate leaching.
- Combining biosolids with other biowastes can mitigate nitrate leaching.
- Dried, but not wet, wood waste effectively mitigates nitrate leaching.
- Partial pyrolysis of wood waste may provide energy-neutral drying.

We investigate the opportunity and challenges of blending biosolids (sewage sludge) with organic waste materials for land application to improve environmental and economic outcomes. There are numerous publications describing the effects of adding organic amendments such as biochar, wood waste, composts, and lignite to soils (Laghari et al., 2016). Similarly, the beneficial and detrimental effects of biosolids addition to soil are well described. However, there is only disparate information on the effects of mixing these amendments with biosolids, which have physicochemical properties that contrast sharply with most soils. This review seeks to determine whether such mixtures could alleviate some of the negative environmental outcomes associated with the land application biosolids.

Biosolids: Resource and Disposal

Biosolids are the end product of wastewater treatment (Jones-Lepp and Stevens, 2007), rich in organic matter and containing agronomically significant concentrations of plant nutrients (Evanylo, 2009). However, biosolids also contain heavy metals (Haynes et al., 2009), pathogens (Gary et al., 2011), organic and pharmaceutical residues (Shinbrot, 2012), and other xenobiotics including endocrine disruptors (Ramamoorthy et al., 1997; Blair et al., 2000; Liu et al., 2009). Most jurisdictions have regulations proscribing the land application of biosolids with excess contaminant concentrations (NZWWA, 2003).

Stockpiling biosolids can exacerbate the emission of greenhouse gases such as nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) (Majumder et al., 2014). Chemicals can leach through soils from stockpiled biosolids (Raghab et al., 2013). Treating biosolids as a waste requires disposal and incurs substantial costs for a product that could be a valuable resource if the environmental risks from residual contaminants could be avoided (Magesan and Wang, 2003).

The amount of biosolids produced globally is staggering. In 2008, biosolids production from 18 counties was estimated to be 18 million dry t yr^{-1} (LeBlanc et al., 2008), and China was reported to have doubled its sewage sludge production from 2005 to 2015 (GWI, 2015). Before landfilling and incineration, ocean disposal was the major disposal route; ~50% of the sewage produced worldwide was dumped into the sea (Bothner et al., 1994). Now, biosolids disposal differs between countries;

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*Corresponding author (brett.robinson@lincoln.ac.nz).

D. Paramashivam, T.J. Clough, and B.H. Robinson, Dep. of Soil and Physical Sciences, Lincoln Univ., PO Box 84, Lincoln 7647, New Zealand; N.M. Dickinson, Dep. of Ecology, Lincoln Univ., PO Box 84, Lincoln 7647, New Zealand; J. Horswell, Environmental Science and Research, Kenepuru Science Centre, PO Box 50-348, Porirua 5240, New Zealand. Assigned to Associate Editor Minori Uchimiya.

for example, in New Zealand, biosolids are disposed of through landfilling (60%), ocean discharge (10%), application to agricultural land (10%) and forests (5%), and for land rehabilitation (10%) and composting or landscaping (5%) (ANZBP, 2014). Landfilling remains the most common form of biosolids disposal (O'Kelly, 2005), with many countries disposing of >50% this way (UNEP, 2016). Modern landfills are complex and costly facilities to build and operate; in New Zealand, disposal cost is approximately US$140 to 175 t^{-1}, excluding transport costs (WCC, 2008). Furthermore, landfills are major sources of emissions of greenhouse gases, particularly CH_4 and N_2O (Spokas et al., 2006; Bogner et al., 2008), which account for up to 5% of global emissions (Bogner et al., 2008; Miller-Robbie et al., 2015). Incineration of biosolids is similarly widely practiced (Werther and Ogada, 1999), accounting for 70% of biosolids disposal in Japan, 58% in the Netherlands, 34% in Germany, 30% in Canada, 15% in the United States, and 20% in France (LeBlanc et al., 2008). Incineration also releases potentially harmful chemicals such as dioxins, furans, NO_x, N_2O, SO_2, and hydrocarbons (Werther and Ogada, 1999).

Land application of biosolids offers the potential to recycle organic matter and nutrients (O'Connor et al., 2005), but pathogens and trace elements may present a risk to soil quality and human health (Sullivan et al., 2006; Singh and Agrawal, 2008). To meet regulatory conditions required for land application, biosolids often require treatment that includes some combination of aerobic digestion, anaerobic digestion, composting, alkaline stabilization, or thermal drying (Lu et al., 2012). All treatment systems reduce pathogen loads, volatile organic compounds, and other easily oxidized organic fractions, including some organic contaminants. Anaerobic systems reduce the organic matter content of biosolids more than aerobic systems (Marchaim, 2017). Biosolids may be dewatered through thermal drying or by using a press. Fresh biosolids usually have high concentrations of NH_4^+, which oxidizes to NO_3^- in aged biosolids (Ogilvie, 1998). Tables 1 and 2 give the physicochemical properties of biosolids resulting from different wastewater treatment processes.

Land application is also limited by agronomic loading rates of plant available nitrogen (N) in biosolids (Gilmour and Skinner, 1998). Plant available N is related to total inorganic N and a fraction of mineralization of organic N during biosolids decomposition (Gilmour and Skinner, 1998), although composition ratio changes with age and origin of the biosolids (Bernal et al., 1998; Rouch et al., 2011). Clearly, suitable amendment materials need to be low cost, readily available, and easy to transport and apply.

Potential of Biosolids to Improve Soil Fertility

Biosolids have been widely shown to improve several physicochemical properties of soil, including porosity, cation exchange capacity, conductivity, water- and nutrient-holding capacity, bulk density, aeration, and drainage (Haering et al., 2000; Stoffella et al., 2003; Michalk et al., 2004). It has also been proven that biosolids application improves carbon (C) sequestration in soils (Bolan et al., 2013). The main contributory constituents toward soil improvement are organic matter (>65%) and N (~5%) (Schmidt et al., 2001; Evanylo, 2009; Haynes et al., 2009).

Many countries already use biosolids as a supplement or replacement for inorganic fertilizer (Ozores-Hampton and Peach, 2002; Rajendram et al., 2011). Biosolids can also be used as a stabilized product to reuse as a fertilizer in wetland agriculture, where heavy metals and fecal bacteria indicators can be maintained under regulatory limits (Uggetti et al., 2012). Of course, there is always concern about biosolids application to agricultural land in relation to residual biological and chemical contaminants (Barnett and Russell, 2001; Magesan and Wang, 2003). For this reason, it has been suggested that it may be more efficacious to use biosolids for soil rehabilitation of degraded or disturbed soils, such as recovering plantation-forest soils (Paramashivam et al., 2016a) and for mine waste restorations (Haering et al., 2000).

Problems Associated with Land Application

Any application of biosolids to land carries a risk of negative impacts on soil quality and human health. Most biosolids contain elevated concentrations of heavy metals (McBride et al., 1999) and organic contaminants, including polychlorinated biphenyls (PCBs), dioxins, polyaromatic hydrocarbons

Table 1. General properties of biosolids and sewage sludge at different stages of digestion.

	pH	EC†	Organic matter	CEC‡	Base saturation	Dry solids	NH_4^+	NO_3^-	References
		dS m^{-1}	%	$cmol_c\ kg^{-1}$	——— % ———		—— mg kg^{-1} ——		
Raw sludge	5.5–6.5	0.4	n/g§	53.5	n/g	6.2 (0.3)	n/g	n/g	(Ogilvie, 1998),(Sanchez-Monedero et al., 2004),(Rouch et al., 2011),
Aerobic sludge	7.6–8.2	2.2 (1.7)	74 (8.5)	101	n/g	2.3	208	526	(Ogilvie, 1998),(Magesan and Wang, 2003),(Rigby and Smith, 2013)
Anaerobic sludge	5.8–8.1	4	14	n/g	n/g	1.7 (0.2)	520	100	(Ogilvie, 1998), (Magesan and Wang, 2003), (Civeira and Lavado, 2008), (Rigby and Smith, 2013)
Digested dry sludge	6.4–7.3	5.5 (0.4)	53 (9.5)	39	n/g	94	4732	431	(Ogilvie, 1998), (Rouch et al., 2011), (Correa et al., 2006)
Aged biosolids (>3 yr)	4.4–4.5	2.4 (0.8)	n/g	n/g	n/g	65	208	1848	(Ogilvie, 1998),(Nash et al., 2011), (Mok et al., 2013),(Laidlaw et al., 2012)
Aged biosolids (>20 yr)	4.5	n/d	51.6	16.7 (0.7)	107 (2.3)	51 (2.2)	130 (7.3)	1352 (2.5)	(Paramashivam et al., 2016b)

† EC, electrical conductivity.

‡ CEC, cation exchange capacity.

§ n/g, not given by the authors; n/d, not determined.

Table 2. Biosolids and sewage sludge elemental composition (references in Table 1). Values in parenthesis represent the standard deviation of the mean.

	Raw sludge	Aerobic sludge	Anaerobic sludge	Digested dry sludge	Aged biosolids (>3 yr)	Kaikōura biosolids (>20 yr)
Total C, %	44 (0.05)	41 (1.8)	41 (4.5)	38 (1)	23	30 (0.5)
Total N, %	2.2 (0.1)	4.8 (2)	9.3 (4.2)	4.7 (0.9)	1.9	3.1 (0.06)
Organic N, %	2	3.6	n/g†	4.1	n/g	n/d
C/N	20 (1.1)	8.5 (2.7)	4.4	8.1 (1.3)	12.1	9.7 (0.02)
Al, mg kg^{-1}	3242 (962)	6522	11412	8618	21709 (333)	17351 (500)
As, mg kg^{-1}	2.2 (1.8)	6.8 (3.2)	16	0.5	22 (3.5)	n/d
Cd, mg kg^{-1}	0.9 (0.3)	2 (0.8)	3.7 (1.8)	3 (2.3)	20 (4)	2.3 (0.1)
Ca, mg kg^{-1}	15450 (8450)	21348	48294	4268	11953 (441)	9455 (534)
Cu, mg kg^{-1}	138 (9.5)	161 (113)	437 (106)	300 (105)	689 (105)	637 (39)
Cr, mg kg^{-1}	239 (206)	408 (396)	1022 (208)	114	776 (208)	34.2 (0.3)
Fe, mg kg^{-1}	2794 (274)	4253 (2573)	9882	28450 (1455)	16925 (2962)	8352 (221)
Pb, mg kg^{-1}	60 (0.0)	73 (42)	240 (15)	140 (60)	497 (112)	114 (5.5)
Mg, mg kg^{-1}	1995 (46)	n/g	15833 (1177)	3495 (455)	1746 (46)	2994 (55)
Mn, mg kg^{-1}	75 (29)	159 (66)	369	196 (1)	103 (17)	189 (2.2)
Hg, mg kg^{-1}	0.6 (0.3)	0.8	2.8	0.4	6.8 (0.8)	n/d
Ni, mg kg^{-1}	19.5 (11)	61	91	17.3 (1.8)	155 (29)	26.4 (1)
P, mg kg^{-1}	3100	1400	3710 (3290)	15444 (5729)	5737 (120)	3463 (248)
K, mg kg^{-1}	2814 (696)	4565	18206 (1180)	2887 (1152)	3171 (78)	3014 (173)
Se, mg kg^{-1}	1 (0.2)	3	5.3	2	n/g	n/d
Na, mg kg^{-1}	4959 (719)	3957	6647 (353)	2170 (1919)	1744 (59)	299 (12)
S, mg kg^{-1}	n/g	n/g	40000	n/g	12771 (4079)	6736(250)
Zn, mg kg^{-1}	325 (86)	617 (280)	1647 (47)	859 (323)	1611 (373)	1047 (69)

† n/g, not given by the authors; n/d, not determined.

(PAHs), pesticides, herbicides, polybrominated diphenyl ethers (PBDEs), nonylphenol, linear alkyl sulfonate, and pharmaceuticals (LeBlanc et al., 2008). Particular concern has been raised about the presence of endocrine-disrupting compounds (Liu et al., 2009). Biosolids application can result in excessive N and P leaching, causing eutrophication of waters (Elliott et al., 2002; Paramashivam et al., 2016b), unpleasant odors (Rynk and Goldstein, 2003; Lu et al., 2012), and pathogens (Sullivan et al., 2006; Singh and Agrawal, 2008). Controlled application to appropriate soil depths in the correct season with consideration of crop harvest and animal grazing undoubtedly minimizes the impact of biosolids application (Smith, 1995); however, these forms of management are unlikely to provide a complete solution to concerns about real or perceived contamination issues.

Global standards exist for acceptable biosolids application rates to agricultural land (EPA-VA, 2004; NZWWA, 2003), but much higher amounts may be required to rebuild and rehabilitate forest or mine reclamation land (Sopper and Seaker, 1990). In one study, 20 yr of biosolids application to a plantation forest soil in Washington was shown to have enhanced tree growth without any apparent negative impact on either human health or the environment (Henry et al., 1994). Nevertheless, similar to concerns on agricultural land, excessive loading of biosolids to forest rehabilitation or mining land may lead to major concerns of nutrient leaching (Wang et al., 2003; Paramashivam et al., 2016b).

Organic Amendments

There is gathering evidence that a combination of biosolids with low-N organic materials offers an opportunity to reduce N leaching from soils amended with biosolids (Table 3). Coapplication of materials that sorb contaminants and mitigate negative impact on the environment obviously needs to provide convincing evidence that this can be realistically achieved (Miller-Robbie et al., 2015). Pyrolysis of organic waste such forestry, garden, and agricultural wastes produces biochar. Biochar shows promise to sorb xenobiotic organic contaminants (Spokas et al., 2009; Wang et al., 2010; Zhang et al., 2010); inorganic components including NH_3, N_2O, NO_3^-, and NH_4^+ (Taghizadeh-Toosi et al., 2011; Taghizadeh-Toosi et al., 2012; Paramashivam et al., 2016b); and metals (Uchimiya et al., 2010; Park et al., 2011). Knowles et al. (2011) and Gartler et al. (2013) demonstrated that biosolids–biochar mixtures resulted in the same or greater biomass production in a range of species compared with adding biosolids alone. This indicates that, although some biochars reduce N leaching from biosolids, they do not reduce N bioavailability to the point where it affects plant growth.

Sawdust has been used effectively to adsorb contaminated dyes (Shukla et al., 2002), heavy metals (Handreck, 1990; Trolove and Reid, 2003; Esperschütz et al., 2016) and inorganic and organic contaminants from water (Robertson and Cherry, 1995; Bugbee, 1999; Schipper and Vojvodic-Vukovic, 2000; Kim et al., 2003; Harmayani and Anwar, 2012; Israel et al., 2014). Used as a bulking agent with biosolids, sawdust can remove pathogens, which can be explained, for example, by the toxicity of tannin compounds to pathogens (Banegas et al., 2007). Pine sawdust has been used to treat groundwater; one study showed that removal of NO_3^-–N from groundwater via denitrification was at a rate of 0.8 to 12.8 ng N cm^{-3} h^{-1} (Schipper and Vojvodic-Vukovic, 2000), where the sawdust provided a C source for denitrifying bacteria in a low-oxygen environment.

Lignite is a waste from the coal industry that is ineffective for energy production due to its high moisture content (Simmler et al., 2013). Budaeva et al. (2006) reported that NH_4^+ sorption from wastewater by a type of lignite was up to 23.2 g NH_4^+ kg^{-1}.

Table 3. Outcomes of experiments combining biosolids with organic materials. Ratios are given by mass unless otherwise stated.

Name and details of the material	Mixing ratio	Experiment	Finding of the study (mixture of biosolids + material compared with control)	References
Pine biochar 350°C (*Pinus radiata*)	Biochar 102 t ha^{-1} eq. Biosolids 600 and 1200 kg N ha^{-1} eq.	Lysimeter in the field	50% reduction in NO_3^- leaching	(Knowles et al., 2011)
(i) Pine biochar 350°C (*Pinus radiata*)	(i) 1:1 biochar:biosolids	(i) Laboratory column	80% reduction in NH_4^+–N	(Paramashivam et al., 2016b)
(ii) Dry pine sawdust (wood waste)	(ii) 2:5 sawdust:biosolids	(ii) Laboratory column	Eliminated NH_4^+–N and NO_3^-–N was reduced >40%	(Paramashivam et al., 2016b)
Pine biochar 350°C (*Pinus radiata*)	Soil containing 20% biochar and 10% biosolids by volume	Greenhouse pot experiment	Beetroot (*Beta vulgaris*) Zn (dry weight) uptake increased 178 mg kg^{-1} and Cd, Cu, Pb were below the WHO's guideline values.	(Gartler et al., 2013)
Sawdust	2:1 sawdust:sawdust. 1250 kg N ha^{-1} eq.	Greenhouse pot experiment	(i) N, P, Cu, Mn and Zn uptake increased in Italian ryegrass compared with control (ii) Cd uptake was reduced by 50% compared with biosolids alone treatment.	(Esperschütz et al., 2016)
Sawdust	Various combinations of sawdust with aerobic and anaerobic sludge	Incubation experiment	Suitable composting ratio was identified as 1:1 by volume with aerobic sludge and 1:3 for anaerobic sludge.	(Banegas et al., 2007)
Sawdust	Sawdust: biosolids mixtures with the C/N adjusted to 20:1. Mixture added at 1, 3, 7, and 14 t ha^{-1}	Mine tailings reclamation and leaching experiments with lysimeters	Sawdust mixed with highest sewage sludge rate reduced NO_3^-–N by 100 mg kg^{-1} compared with no sawdust addition.	(Daniels et al., 2001)
Lignite (mine waste)	2:5 lignite:biosolids	Laboratory column	66% reduction in NH_4^+–N, but no effect on NO_3^-–N leaching	(Paramashivam et al., 2016a)
Lignite	1, 3.4, and 7.1 parts of lignite to 3.4 parts of biosolids	Greenhouse pot experiment: *Lolium perenne* ryegrass were grown in each treatment	Cd uptake was reduced by 30% in ryegrass at 1% lignite application rate	(Simmler et al., 2013)

After several years of amendment with lignite, soil had a higher C content (35%), slightly higher N content (33%), and a higher C/N ratio than control soil. Lignite mixed with biosolids has also been shown to reduce heavy metal uptake by plants (Simmler et al., 2013) and inorganic N leaching from soil (Paramashivam et al., 2016a).

Addressing Nitrogen Mobility

A strategically managed nutrient plan is required to reduce N loss from biosolids amendments while promoting plant production. The main forms of N in biosolids are organic N (~95%), NO_3^- (~4%), and NH_4^+ (~1%) (Henry et al., 2000; Daniels et al., 2001). After land application of biosolids, some fraction of the organic N is mineralized by microorganisms in the soil and transformed to mobile inorganic forms (Prasad and Power, 1997). Nitrate is highly mobile in soil and can be readily leached through the soil profile. Otherwise, inorganic species (NH_3, NO_3^-, and NH_4^+) are potentially available for plant uptake, apart from any losses to volatilization, immobilization, or leaching (Prasad and Power, 1997). The main areas for considerations are (i) mineralization rates of the biosolids, (ii) volatilization rate of NH_4^+–N, (iii) NO_3^-–N concentrations in the biosolids at application time, (iv) rate of N uptake by vegetation, and (v) existing soil conditions and fertility (Haynes et al., 2009).

Biosolids application rates are a critical step in land application. For example, 18 mo after the application of biosolids to a vineyard soil at rates of 10, 30, and 90 Mg ha^{-1} (fresh weight), soil organic matter increased by up to 3000 mg kg^{-1} and inorganic N concentrations by 5 to 26 kg N ha^{-1} (Korboulewsky et al., 2002). Despite the benefits of higher application rates, only the lowest rate of 10 Mg ha^{-1} was regarded as a safe application rate with no N leaching from the soil. Similar recommended application rates on cotton (*Gossypium hirsutum* L.) crops were reported by Samaras et al. (2008) and on sunflower (*Helianthus annuus* L.) by Lavado (2006). Many other studies report that application rates ranging from 10 to 30 Mg ha^{-1} avoid excessive N leaching from soils and do not pose an undue risk to the environment or human health (Binder et al., 2002; Brenton et al., 2007; Rajendram et al., 2011). Even biosolids that have been through advanced treatment systems can result in high levels of NO_3^-–N leaching if they introduce a high N load combined with a low C/N ratio. Therefore, higher rates of application will require some additional amendment to restrict nutrient leaching.

Organic Amendments to Blend with Biosolids

Few practicable amendments are low cost, easily transportable, and readily available. One example is sawdust and wood waste from commercial timber logging and sawmilling (Schipper and Vojvodic-Vukovic, 2000). Gerwing et al. (1996) reported that logging in eastern Amazonia produced some 24.7 m^3 ha^{-1} of wood waste. Decomposition of untreated wood waste added to a soil increases greenhouse gas emissions; although conversion to biochar partly addresses this concern (Gholz et al., 2000), pyrolyzation is a costly and time-consuming process. Greenhouse gas emissions resulting from the addition of wood waste to soil should be balanced against the alternatives, namely incineration or landfilling, which will ultimately result in the conversion of the wood to CO_2 or CH_4, respectively. An alternative to wood waste is lignite, which is abundant worldwide; recoverable world lignite resources are approximately 195×10^9 t, with 333×10^6 t being located in New Zealand (WEC, 2010).

Physicochemical characteristics of potential organic amendments (Table 4) determine the sorption of contaminants when this material is blended with biosolids. A porous structure is a prominent feature that is key to nutrient absorption or water and nutrient retention by biochar (McLaughlin et al., 2012). With lignite, there is a positive correlation between the porous texture and the molecular size of the sorbed species (Pope, 1984). Both materials have high C/N ratios (43–1300), which also has a critical influence on N mobility within soil (McLaren and Cameron, 1996; Kwiatkowska et al., 2008).

We found recently that lignite and most biochars are not effective in reducing NO_3^- leaching from soils amended with biosolids in a low-fertility soil (Paramashivam et al., 2016a). Lignite lessened the beneficial growth effects of adding biosolids to soil and exacerbated N_2O production. However, sawdust and partially pyrolyzed biochars (resulting from heterogeneous temperatures in the pyrolysis kiln) have provided convincing results from laboratory and glasshouse trails when applied with biosolids to the same soils (Paramashivam et al., 2016b). In batch sorption and column leaching experiments, biochars and fresh sawdust failed to sorb NO_3^-, but NO_3^- leaching was reduced by *Pinus radiata* D. Don sawdust with a low moisture content. One type of low-temperature (350°C) biochar also effectively sorbed NH_4^+, reducing leaching from columns by 40 to 80%. Blending biosolids with some organic materials can reduce the environmental impact of biosolids application. These findings require further testing in the field.

Sorption of Xenobiotics and Heavy Metals

Biodegradation is the major pathway to breakdown xenobiotic compounds (Piveteau et al., 2001; Ye et al., 2004), ranging from days and weeks to years [some compounds, e.g., 1,1,1-trichloro-2,2-bis(4-chlorophenyl)ethane (DDT), take up to decades] (Fetzner, 2000). Organic amendments can promote biodegradation via several mechanisms. Organic amendments that contain an easily oxidized C source, such as low-molecular-weight organic acids, will promote the growth of microorganisms, which may in turn degrade some organic contaminants (Martin et al., 2014). Microorganisms such as white rot fungi [an assemblage of species, including *Pleurotus ostreatus* (Jacq. ex Fr.) P. Kumm and *Trametes versicolor* (L.) Lloyd.] that degrade lignin in wood waste can enhance the degradation of persistent organic pollutants such as pentachlorophenol (Mileski et al., 1988). Microbial inhibition in biosolids caused by poor aeration, low pH, or high concentrations of metals may be reduced by mixing the biosolids

Table 4. Key parameters potential organic amendments that could be mixed with biosolids. Values in brackets represent the standard deviation of the mean.

Material	HTTs†	pH (H_2O)	C	N	H	O	CEC‡	BS§	Ash	Volatiles	Fixed C	References
	°C		%				$cmol_c\ kg^{-1}$		%			
Pine sawdust	–	5.7–7.6	51 (0.04)	0.06 (0.001)	n/d¶	n/d	10.6	448 (6.8)	1.4 (0.5)	72.4 (4)	n/g	(Paramashivam et al., 2016b), (Wilén et al., 2004)
Pine (commercially pyrolyzed char)	350	6.9	78 (0.08)	0.06 (0.02)	n/d	n/d	9.1	88 (2.2)	3.3	53	36.2	(Paramashivam et al., 2016b)
Feedstock for pyrolyzation												
Pine	350	5.1	71	0.1	5	25	29	22	1.2 (0.6)	45 (12)	55 (12)	(Enders et al., 2012), (Pereira Calvelo et al., 2011),
	400	6.9	77	0.6	4.6	14.5	30	0.1>	2.3 (0.8)	38.2 (4.5)	59.6 (4.1)	(Hina et al., 2010)
	550	6.1–10	83 (2.8)	0.4 (0.2)	3.3 (0.1)	12 (2.8)	25	12	3.4 (0.9)	27 (6.5)	69 (5.2)	
Corn stover	500	6.5–9.9	54 (10)	1.1 (0.2)	3 (0.3)	10 (2.5)	40 (12)	142	33 (8)	29 (2.4)	38 (13)	(Enders et al., 2012), (Brewer et al., 2011)
	550	6.6–10	55 (18)	0.9 (0.3)	2.8 (0.5)	13 (1)	28	n/a	29 (15)	34 (3.7)	37 (12)	
	600	6.7–10	51 (20)	0.9 (0.3)	2.2 (0.2)	9.3 (0.05)	33 (6.5)	172	36 (20)	21 (3.5)	43 (17)	
Switchgrass	450	6.2	38	0.5	2.2	9	28	n/a	50	17	32	(Enders et al., 2012), (Brewer et al., 2011)
	500	6.6	52 (12)	1.2 (0.7)	2	2.4	26	n/a	55	32.3 (21)	36 (4.3)	
	550	6.7	42	0.5	2	5	26	n/a	50	14	35	
Poplar	400	7.2–9	55 (17)	0.6 (0.2)	3.4 (1.1)	38 (19)	n/a	n/a	2.9 (0.9)	35	61	(Pereira Calvelo et al., 2011),
	550	8.8	76	1	3.6	13	n/a	n/a	6.5	28	66	
Oak	400	4.6–6.9	64 (11)	0.4 (0.2)	3.2	17.1	51.3 (9.9)	15	0.8	37 (4.5)	58	(Enders et al., 2012), (Zhang et al., 2015)
	600	6.4–9.5	72 (13)	0.4 (0.09)	2.5	8.5	54.8 (42)	7.5	1.3	20.2 (7.9)	71	
Oak (with steam activation)	400	9.6	30.3	0.5	n/a	n/a	36.2	n/a	n/a	n/a	n/a	(Zhang et al., 2015)
	600	9.6	37.1	0.4	n/a	n/a	42.6	n/a	n/a	n/a	n/a	
Lignite												
New Vale, New Zealand	–	4.5	86 (0.4)	1.3 (0.01)	n/a	n/a	43.6 (0.8)	115.3 (0.5)	n/a	n/a	n/a	(Paramashivam et al., 2016a).
5 coal mine sites from Central Europe	–	5.1–6.2	63.4 (0.2)	0.9 (0.04)	5.5 (0.2)	27.4 (0.4)	20–70	n/g	19 (1.2)	n/g	n/g	(Janos et al., 2011)
International humic substance society(IHSS)	–	4.2	65	1.2	5	28	72	n/g	13	n/g	n/g	(Janos et al., 2011)

† HTT, highest treatment temperature for that particular pyrolysis.

‡ CEC, cation exchange capacity.

§ BS, base saturation.

¶ n/d, not determined; n/g, not given by the authors; n/a, not analyzed.

with organic amendments (Song and Schobert, 1996; Pehlivan and Arslan, 2007; Chassapis et al., 2009; Knowles et al., 2011; Simmler et al., 2013; Doskocil et al., 2015).

Xenobiotics can be immobilized or entrapped within the micropores of organic soil amendment (Fetzner, 2000). Biochar has been widely used for the sorption of xenobiotic contaminants from the soil (Wang et al., 2010). Zhang et al. (2010) have reported that the phenanthrene sorption linear distribution coefficient of soil amended with biochar was 3.4×10^4 L kg^{-1}, and soil without amendment was only 47 L kg^{-1}.

Heavy metal concentrations in biosolids will be reduced by dilution when the biosolids are mixed with an organic amendment that has low metal concentrations. Amendments such as biochar can increase the pH of biosolids-amended soil, resulting in increased sorption and precipitation of heavy metal cations (Beesley et al., 2011). Heavy metals may be immobilized by sorption onto exchange sites on an organic amendment. Lignite increases the cation exchange capacity of the material due its high humic acid content (Kucerik et al., 2003; Janos et al., 2011). Lignite effectively immobilizes metallic ions and heavy metals such as Cu, Pb, Cd, Ni, and Zn in contaminated soils (Karczewska et al., 1996; Pehlivan et al., 2004; Budaeva et al., 2006; Domańska and Smolinska, 2012; Doskocil and Pekar, 2012) and removes radionuclides and potential toxic metals from wastewater (Mohan and Chander, 2006; Mizera et al., 2007). Unlike the degradation of xenobiotics, the immobilization of heavy metals may not be permanent; the degradation of organic amendments, as well as the biosolids themselves, may result in desorption of heavy metals and their subsequent leaching or plant uptake (Tella et al., 2016). In some cases, organic materials may increase the solubility of heavy metals such as Cu, which can form mobile complexes with dissolved organic C associated with the amendment (Bolan et al., 2003). However, the solubilized Cu complex may not be bioavailable (Kunhikrishnan et al., 2013).

Conclusions

Biosolids can provide organic matter and a rich source of nutrients to improve soil quality, but food chain risks and wider environmental concerns reduce or even prevent its wide-scale application to agricultural land in many countries. Wastewater treatment options are unable to address all of these concerns, but landfilling and incineration do not provide a sustainable alternative. Coapplication with other organic waste streams to land certainly does not always provide a viable solution; however, there is evidence that mixtures containing sawdust and partially pyrolyzed biochars are effective in mitigating excessive nitrate leaching from biosolids-amended soils. Lignite coal waste, while ineffective at mitigating nitrate leaching, may reduce plant uptake of biosolids-borne contaminants such as Cd. Economically and environmentally acceptable solutions that will allow biosolids application to agricultural land remain highly challenging, but the requirement to improve soil quality for forest and mine rehabilitation probably justifies more field investigations. Combining biosolids with other organic wastes to rehabilitate degraded land remains a potentially practicable and sustainable management of these resources.

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