



Review

Crop residue stabilization and application to agricultural and degraded soils: A review



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ABSTRACT

Agricultural activities produce vast amounts of organic residues including straw, unmarketable or culled fruit and vegetables, post-harvest or post-processing wastes, clippings and residuals from forestry or pruning operations, and animal manure. Improper disposal of these materials may produce undesirable environmental (e.g. odors or insect refuges) and health impacts. On the other hand, agricultural residues are of interest to various industries and sectors of the economy due to their energy content (i.e., for combustion), their potential use as feedstock to produce biofuels and/or fine chemicals, or as a soil amendments for polluted or degraded soils when composted. Our objective is review new biotechnologies that could be used to manage these residues for land application and remediation of contaminated and eroded soils. Bibliographic information is complemented through a comprehensive review of the physico-chemical fundamental mechanisms involved in the transformation and stabilization of organic matter by biotic and abiotic soil components.

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1. Introduction

Organic residues remaining after crop or wood harvesting in agriculture and forest industries can be transformed into energy, biofuels, high-value chemicals, forage or soil amendments (Liu et al., 2008; Schnitzer et al., 2014). Agricultural wastes are classified into two categories, crop residues (i.e., plant residues such as roots, leaves, straw, stubbles, among others), and residues from agro-industrial activities which are by-products of post-harvest and food processing, among others. Crop residues are one of the major organic wastes, with an annual production of about 74 Tg of dry crop residues produced worldwide. The global annual production of rice and wheat crop residues is about 45 Tg (Kim and Dale, 2004) whereby they may become a potential environmental problem or useful feedstock for new byproducts.

Crop residues are an important source of nutrients (Azam, 1990), which can positively influence the biological, chemical and physical properties of soil. The incorporation of crop residues increases nitrogen mineralization (Singh, 1995) as well as organic carbon (C) and total nitrogen (N) in soil (Cassman et al., 1996). In this context, several studies have shown that the incorporation of crop residues into soil improves tilth, reduces erosion processes, prevents nutrient losses by run-off and leaching, and increases microbial biomass (Lal, 1980; Lal et al., 1980; Maurya and Lal, 1981; Bukert et al., 2000; Shah et al., 2003; Shafi et al., 2007). Consequently, when organic residues are incorporated to the soil, detrimental or erosive processes decrease due to the high amount of organic matter (OM) that they supply. This is a very important aspect to take into account in both soil remediation and amelioration practices, since degraded and polluted soils usually have lower organic matter content (Lal, 2003).

Mining is an industrial activity, which may have a deleterious effect on soils, agro ecosystems and wildlife. Extensive mining produces a large amount of metal-enriched solid and liquid effluents, which can pollute air, water, and soils (Meier et al., 2012). The productivity of contaminated soils by mining activities requires subsequent expensive reclamation practices. Current and future reclamation technologies are designed to minimize such deleterious impacts through the use of socially friendly technologies, such as organic amendments from crop residues, which can even cause a prolonged sequestration of potentially toxic elements (PTE) such as metals (Pb, Cu, Zn, Cd, Fe) and metalloids (As, Se) into the organic matrix of soils.

Recent research indicates that incorporation of excessive residues as a soil amendment may have an adverse impacts on soil, environment and crop yield (Jiang et al., 2012) because many soil functions require mature and stable organic matter (Senesi, 1989; Senesi and Brunetti, 1996; Senesi and Plaza, 2007; Brunetti et al., 2008; Plaza and Senesi, 2009; Guenet et al., 2012). For these reasons, further studies concerning transformation and stabilization processes of organic wastes, prior to soil application are imperative. Ensuring a high supply and availability of nutrients for crops, and the long-term storage of atmospheric C in soils, will be favored if organic residues are well stabilized before their application to soil. Composting is one of the most used technologies for agricultural residues treatment. This technology reduces volume and weight by about 50% resulting in a stable product that can be used to enhance chemical and physical properties of soils for

agricultural purposes (Sánchez-Monedero et al., 2002). Alternative composting methods to stabilize C in composts are being developed. These methods enhance their quality and the physical, chemical and biological fertility of soils (Bolan et al., 2012).

Composting as technology has been widely described and summarized in several contributions as review articles and books chapters since decades. These articles describe different aspects of composting such as composts feedstock, production methods and properties of composts, composting parameters, methodologies for the characterization of the final product, biochemical transformations of organic matter, microorganisms involved in the process and the effects of compost in different types of soils (e.g. Agricultural, polluted or degraded soils), soils properties, plant growth and pathogen protection (de Bertoldi et al., 1983; Senesi, 1989; Schaub and Leonard, 1996; Senesi and Brunetti, 1996; Senesi et al., 1996; Scheuerell and Mahaffee, 2002; Chen, 2003; Mason, 2006; Larney and Hao, 2007; Bernal et al., 2009; Plaza and Senesi, 2009; Park et al., 2011; Mohammad et al., 2012; Singh and Kalamdhad, 2012; Li et al., 2013; Mekki et al., 2013; Anand and Apul, 2014; Han et al., 2014; Cavagnaro, 2015) among others, which contribute significantly to the compost science knowledge. However, we think is necessary to summarize new findings and novel approaches described in current scientific articles due to the relevance of this topic for agriculture and environment.

Thus, our objective is to review the most relevant and current information regarding compost stabilization, use and application of composted agricultural residue, with emphasis on organic-metal complexes and metal-OM interactions. We hope that in the near future such knowledge may be useful for developing novel approaches, processes and technologies for the production and application of biotical and chemically stabilized composted organic amendments in polluted and/or degraded soils. Furthermore, we hope that these new composting approaches and technologies may be also used as tools for C sequestration in soils based on scientific evidence.

Table 1
Annual production of the wheat straw and crop residues in the main producers' countries.

References	Country	Residues	Annual production (10 ⁶ Mg y ⁻¹)
Li et al. (2012)	Canada	Wheat straw	37.4
		Total crop residues	82.4
Scarlat et al. (2010)	European Union	Wheat straw	108.6
		Total crop residues	258.0
Liu et al. (2008) Jiang et al. (2012)	China	Wheat straw	126.6
		Total crop residues	630.0
Cala-Rodríguez et al. (2003)	Spain	Wheat straw	10.0
		Total crop residues	38.0
Schomberg et al. (1994), Lal (2005)	United States	Wheat straw	74.0
		Total crop residues	421.6

Table 2
Annual production of rice residues in the world, modified from Lim et al. (2012).

Regions	Residues	Annual production (10 ⁶ Mg y ⁻¹)
Africa	Rice straw	24.5
	Rice husk	4.9
Asia	Rice straw	618.0
	Rice husk	123.65
Europe	Rice straw	4.10
	Rice husk	0.82
Oceania	Rice straw	0.29
	Rice husk	0.06
Americas	Rice straw	38.10
	Rice husk	7.62
World	Rice straw	685.24
	Rice husk	137.05

2. Crop residues production

As a consequence of the increase in world population, there has been a rise in the demand for food and food production. This has generated huge amounts of agricultural residues. As stated earlier in this article, agricultural residues can be classified conventionally into two categories: crop residues and agro-industrial residues (Tripathi et al., 1998). Crop residues refer to plant remains staying in the field after crop harvest. Crop residues include wheat straw, shafts and shoots, among others. On the other hand, agro-industrial residues are byproducts of the postharvest processes generated from cleaning, sieving, milling (Lim et al., 2012) or residues left by the food processing industry. Worldwide, straw, roots, shaft and other tissues of corn, wheat and rice are the main crop residues annually produced, which account for 40.6%, 24.2% and 15.7% respectively (Jiang et al., 2012; Schnitzer et al., 2014). For example, it has been estimated that the total production of crop residues only in China during 2009 was about 807 million tons, and the average yield was 716 million tons in the past 10 years (Jiang et al., 2012). Total yields and annual production of crop residues and the major producer countries and regions are summarized in Tables 1 and 2.

Kim and Dale (2004) estimated annual production rates of main crops produced around the world. For instance, they estimated that the maize annual production is about 520 Tg of dry matter. They also reported that major production regions are North America (42%), Asia (26%), Europe (12%) and South America (9%). Regarding maize, the highest grain yields occurs in North America, in which U.S. average grain yield is 7.2 Mg/ha. About 5% of corn in the world is wasted, generating about 203.6 Tg of corn stover annually (Kim and Dale, 2004). On the other hand, the annual global production of dry rice grain is about 526 Tg. In this sense, Asia is the leading production region of rice with over 90% of global production and the largest yield occur in Australia. About 4.8% of rice annually production is lost as waste (Kim and Dale, 2004) and about 22 Tg of these loses residues occur in Asia. Finally, the annual global production of wheat dry grain is about 529 Tg, with Asia (43%) and Europe (32%) representing the main regions. North America is the third largest production region with 15% of global wheat production (Kim and Dale, 2004). The latter authors noticed that grain yield of wheat ranges from 1.7 to 4.1 dry Mg ha⁻¹. About 20 Tg of dry wheat residues (4% of global production) is lost as waste. About 10 Tg of dry wheat in Asia ends up in the waste stream. Although wheat straw is being used as an animal feed, the scope of paddy straw as cattle feed is limited due to its high silica content (>12%) (Gai and Nain, 2010).

2.1. Potential environmental issues and economic benefits

Production of large amounts of lignocellulosic materials such as straw and shafts play a useful role in dealing with potential environmental problems. For example, the full harvest of crop residues and soil tillage may give rise to soil erosion and decrease soil OM content (Mann et al., 2002). Under anaerobic conditions N immobilization by soil OM can occur (Olk et al., 2002; Schmidt-Rohr et al., 2004), due to the binding of proteinaceous compounds to the lignin-derived phenolic component (Olk et al., 2006). Moreover, some studies have shown that when organic waste, such as crop residues, are used as a soil amendment the application of these fresh materials may cause several adverse effects on soils due to the lack of mature and stable OM (Senesi and Brunetti, 1996; Brunetti et al., 2008). These effects include, among others: (i) an increase of the mineralization rate of native soil organic C through extended microbial oxidation, (ii) the induction of anaerobic conditions by mineralization of large amounts of non-stabilized organic C, (iii) the associated extended O₂-consumption, and (iv) the alteration of soil pH (Senesi and Brunetti, 1996). In addition, these fresh residues may be phytotoxic due to the production of ammonia, ethylene oxide and organic acids (Tam and Tiquia, 1994). Otherwise, in some places their combustion have a negative impact on air quality (Madariaga, 2003). For this reason, environmental friendly treatments of agricultural residues are required.

3. Transformation and stabilization of agricultural residues

Crop residues treatment, their safe use and disposal become important issues for human society. With the increase in agricultural wastes, landfill space has become the limiting factor for disposal (Amir et al., 2003). Thus, residue processing and recycling become interesting alternatives for their treatment. The application of agricultural residues (e.g. crop residues, animal manures) to soil have become since decades interesting environmental and agricultural practice for maintaining and enhancing soil OM, being known that this remediates degraded soils and supplies crop nutrients. Thereby, it is well known that stabilization of these residues before their application to soil may help increase their residence time by reducing their oxidation to CO₂ in soil agro-ecosystems. Some recent methods of stabilizing agricultural wastes are reviewed in the article.

3.1. Composting

Composting is one of the most important and common technologies for the treatment of agricultural residues. Composting reduces volume and weight about 50% and results in a stable product that can be used to enhance chemical and physical properties of soils for agricultural purposes (Senesi, 1989). In general, aerobic composting is defined as a microbiological process that converts waste into organic manure rich in plant nutrients and humus (Bewick, 1980; de Bertoldi et al., 1983; Senesi, 1989; Schaub and Leonard, 1996; Senesi and Brunetti, 1996; Sharma et al., 1999; Senesi and Plaza, 2007). Composting is a technology for waste treatment and an important way for stabilization of organic residues. Different phases of composting process have been recognized and defined. The composting process, especially when it is carried out in batch mode, can be divided into phases characterized by temperature. The phases are conventionally defined in terms of the kinds of bacterial and fungal populations that thrive in different temperature ranges, i.e., psychrophilic (optimum ≤ 13 °C), mesophilic (21–48 °C), and thermophilic (45–68 °C) (de Bertoldi et al., 1983; Chen and Inbar, 1993; Haug, 1993; Mason, 2006; Hubbe et al., 2010; Li et al., 2013).

Table 3
Type of Microorganisms involved in the different stage of composting. Modified from Hubbe et al. (2010).

References	Composting stage	Involved organism in the composting stage	Type of (micro)organism
Steel et al. (2010)	Thermophilic the beginning of composting process, (immediately after of heat peak)	<i>Rhabditidae</i> sp. <i>Panagrolaimidae</i> sp. <i>Diplogasteridae</i> sp. <i>Cephalobidae</i> sp.	Nematode Nematode Nematode Nematode
	At the cooling and maturation of the composting procedure	<i>Mononchoides</i> sp. <i>Dytilenchus filimus</i>	Nematode Nematode
Silva et al. (2009)	Mesophilic (composting before mixing the pile)	<i>Bacillus</i> sp. <i>Paenibacillus</i> sp. <i>Actinomycetes</i> <i>Aspergillus fumigatus</i>	Bacteria Bacteria Filamentous bacteria Fungi
Días et al. (2009)	At the end of composting procedure (samples in different points)	<i>Aspergillus fumigatus</i> <i>Emericella</i> sp. <i>Aspergillus ochraceus</i> <i>Aspergillus terreus</i> <i>Penicillium oxalicum</i>	Fungi Fungi Fungi Fungi Fungi
Chang et al. (2009), Tuomela et al. (2000)	At the cooling and maturation	<i>Thermoactinomyces</i> sp.	Yeast
Rebollido et al. (2008)	Mesophilic	<i>Pseudomonas</i> sp. <i>Azotobacter</i> sp. <i>Azospirillum</i> sp. <i>Bacillus</i> sp.	Bacteria Bacteria Bacteria Bacteria
		<i>Chaetomium</i> sp. <i>Thermophile</i> sp. <i>Malbranchea sulfurea</i> <i>Thermomyces lanuginosus</i> <i>Torula thermophila</i>	Fungi Fungi Fungi
Taiwo and Oso (2004)	Mesophilic	<i>Feecal coliformes</i> <i>Streptococcus</i> sp. <i>Proteus</i> sp. <i>Serratia</i> sp. <i>Bacillus</i> sp.	Bacteria Bacteria Bacteria Bacteria Bacteria
	Thermophilic	<i>Aspergillus</i> sp. <i>Fusarium</i> sp. <i>Penicillium</i> sp. <i>Humicola</i> sp. <i>Mycothypha</i> sp. <i>Scopulariopsis</i> sp. <i>Cephalosporium</i> sp. <i>Trichothesium</i> sp.	Fungi Fungi Fungi Fungi Fungi Fungi Fungi Fungi
	Cooling down stage	<i>Aspergillus</i> sp. <i>Fusarium</i> sp. <i>Penicillium</i> sp. <i>Cladosporium</i> sp. <i>Mycothypha</i> sp. <i>Scopulariopsis</i> sp. <i>Coprinus</i> sp. <i>Cephalosporium</i> sp. <i>Trichothecium</i> sp.	Fungi Fungi Fungi Fungi Fungi Fungi Fungi Fungi
Hassen et al. (2001)	At the cooling of composting procedure Mesophilic and thermophilic At the remainder of composting cycle	<i>Escherichia coli</i> <i>Streptococcus</i> sp. <i>Bacillus</i> sp.	Bacteria Bacteria Bacteria
De Bertoldi et al. (1983)	Thermophilic	<i>Absidia ramosa</i> <i>Allescheria terrestris</i> <i>Mucor pusillus</i> <i>Chaetomium thermophilum</i> <i>Talaromyces thermophilis</i> <i>Aspergillus fumigatus</i> <i>Humicola insolens</i> <i>Humicola lanuginosa</i> <i>Lenzites</i> sp. <i>Penicillium duponti</i> <i>Scytalidium thermophilum</i> <i>Sporotrichum thermophile</i> <i>Thermoascus</i> sp.	Fungi

Table 3 (continued)

References	Composting stage	Involved organism in the composting stage	Type of (micro)organism
	Meshophilic	<i>Allescheria terrestris</i> <i>Chaetomium</i> sp. <i>Dasyscypha</i> sp. <i>Emericella nidulans</i> <i>Mollisia</i> sp. <i>Thermoascus aurantiacus</i>	Ascomycota
		<i>Armillaria mellea</i> <i>Clitopilus insitus</i> <i>Coprinus cinereus</i> <i>Fomes</i> sp. <i>Lentinus lepiakus</i> <i>Lenzites trabea</i> <i>Pleurotus ostreatus</i> <i>Polyporus versicolor</i>	Basidiomycota
		<i>Alternaria tenuis</i> <i>Artrobotrys oligospora</i> <i>Aspergillus amstelodami</i> <i>Aspergillus</i> sp. <i>Botryotrichum piluliferum</i> <i>Cephalophora tropica</i> <i>Cephalosporiopsis alpina</i> <i>Cephalosporium</i> sp. <i>Cephalosporium</i> sp. <i>Cephalosporium</i> sp. <i>Cephalosporium</i> sp. <i>Cladosporium herbarum</i> <i>Doratomyces</i> sp. <i>Geotrichum candidum</i> <i>Gliocladium</i> sp. <i>Gliomastox murorum</i> <i>Graphium</i> sp. <i>Harpographium</i> sp. <i>Leptographium lundbergii</i> <i>Paecilomyces</i> sp. <i>Penicillium</i> sp. <i>Penicillium</i> sp. <i>Phlyctaena</i> sp. <i>Rhinocladiella artrovirens</i> <i>Scopulariopsis brevicaulis</i> <i>Scopulariopsis</i> sp. <i>Sporotrichum thermophile</i> <i>Stachybotrys</i> sp. <i>Trichoderma viride</i>	Deuteromycota

During composting, bio-macromolecules such as lignin, cellulose, hemicellulose, starch, among others, are partially transformed or decomposed by microorganisms into more simple molecules. The main microorganisms involved in the different phases of composting process include some groups of mesophilic such as *Escherichia*, *Streptococcus*, *Bacillus* and *Serratia*. At thermophilic and cooling down phases fungi such as *Aspergillus* sp., *Fusarium* sp. and *Penicillium* sp. appear to predominate (Table 3) (de Bertoldi et al., 1983; Hubbe et al., 2010; Mohammad et al., 2012).

During composting the biodegradable organic compounds are mineralized, whereas part of the remaining organic materials become precursors of humic substances (HSs) (de Bertoldi et al., 1983; Senesi, 1989; Schaub and Leonard, 1996; Senesi and Brunetti, 1996; Senesi and Plaza, 2007). During the decay and transformation processes of biomolecules (from plants tissues and other dead organism) HSs are formed by secondary biotic synthesis reactions or re-synthesis (Stevenson, 1994; Schnitzer and Monreal, 2011). The HSs are considered as the final product of OM stabilization by composting. The HSs have a crucial role in the main environmental reactions such as reductive and oxidative reactions, sorption and complexation of pollutants and soil C sequestration (Senesi, 1992a; Bradley et al., 1998; Coates et al., 1998; Plaza and Senesi, 2009; Park et al., 2011; Singh and Kalamdhad, 2012).

Humification of simple organic molecules produces changes in the chemical characteristics of the OM constituents, which are usually considered as quality indices for the end product of composting. These indices are usually based on the ratios between different extractable alkali fractions of OM such as humic acids (HA), fulvic acids (FA), total extractable carbon and absorbance of acetone extract (Iglesias and Pérez, 1989; Senesi et al., 1989; Senesi, 1992b; Romero et al., 2007; Khan et al., 2009). Different types of raw materials are used for composting and generate various types of HSs with different chemical compositions (de Bertoldi et al., 1983; Senesi, 1989; Senesi et al., 2002; Larney and Hao, 2007; Senesi and Plaza, 2007; Bernal et al., 2009; Mohammad et al., 2012; Li et al., 2013). For example, Brunetti et al. (2007) found differences among HS compositions in different organic amendments and native soils, highlighting that HA like-substances from the organic amendments exhibit higher H, N, S and phenolic OH group contents, and lower C/N and C/H ratios, O, carboxylic, total acidity and organic free radical contents than native soil. Thus, Brunetti et al. (2008) reported that aliphatic and polysaccharide structures and S- and N-containing groups of HA-like fractions of organic amendments are partially incorporated into native soil HAs, hence, determining modifications to various extents of their composition, structure and reactivity. In the soil environment, chemical components of HA such as polyketides and humin, are extremely resistant

to biodegradation (Schnitzer and Monreal, 2011). Their half-decay time ($t_{1/2}$) reach thousands years. For example, ^{14}C dating has shown that soil OM in whole and aggregate size fraction in soils from the Great Plains and Eastern regions of North America, or from volcanic environments is in the order of hundreds-to-thousands of years old (Monreal et al., 1997; Grinhut et al., 2007).

The resulting product of the composting process is used frequently for agricultural purposes such as amelioration of soil characteristics (as structure and porosity) and enhancing the nutrient supply for crop growth (Bewick, 1980; de Bertoldi et al., 1983; Senesi, 1989; Plaza and Senesi, 2009; Mekki et al., 2013; Cavagnaro, 2015). For example, Tejada and Gonzales (2003) found that the application of crushed cotton gin residues compost was of great agricultural interest due mainly to its high OM content, which produced an increase in soil microbial activity, structural stability and porosity. Carbon sequestration is another effect of composting process which is currently receiving an increased interest (Pipatti and Wihersaari, 1998; Lou and Nair, 2009). Lynch et al. (2006) found that some compost sources, such as corn residues, had a high recalcitrance in soils after being applied. The above authors noticed that one year after application, 75–95% of the applied compost C remained in soil, and hence composts increase OM recalcitrance with prolonged storage in non-mineral soil fractions. Moreover, Bustamante et al. (2010) found that applying compost could be a great contribution to the reduction of atmospheric CO_2 , because it increases C storages in soils after compost application. However, authors as Godbout et al. (2010) has shown that a part of the C source resulting from the composting process eventually becomes mineralized in the soil and returned to the atmosphere as CO_2 (Senesi, 1989).

New approaches have been developed for enhancing the residence time of HS produced by composting in soil before being decomposed (Godbout et al., 2010). Decreasing the large amount of energy needed for producing the optimal conditions of biological composting at optimum moisture and temperature, and decreasing the long composting time needed for OM humification (90–365 days) has been studied by Alfano et al. (2008). Some novel composting technologies which aim to promote the stabilization of the composted organic amendment by physico-chemical interactions has been analyzed by Bolan et al. (2012) and Qi et al. (2012a,b), and are reviewed in the following sections.

3.2. Biotransformation of crop residues by saprophytic fungi

Lignin, hemicelluloses and celluloses are major components of crop tissues. Lignin is a complex aromatic macromolecule that provides strength and rigidity to cell walls and tissues of all vascular plants by acting as glue between the polysaccharide filaments and fibers (Monties and Fukushima, 2001). Lignin is a chemically heterogeneous molecule which is linked by several covalent bonds (Brunow, 2001). These bonds cannot be cleaved by hydrolytic enzymes as in most other natural polymers (cellulose, starch, proteins, etc.) (Hofrichter, 2002). Lignin tends to be less abundant than the cellulose or hemicellulose components of biomass during the composting processes (Tuomela et al., 2000; Eiland et al., 2001a,b; Jouraiphy et al., 2005; Huang et al. 2006; Albuquerque et al., 2009; Spaccini et al., 2009). In the future, the addition of organisms such as saprophytic fungi that can act over the chemical bonds in lignin could be a way to accelerate the transformation and humification of residues with high lignin content.

It is well known that saprophytic fungi act as primary, secondary and tertiary decomposers in natural ecosystems (Grinhut et al., 2007). Moreover, fungi are the most efficient degraders of macromolecules such as lignin, hemicelluloses and celluloses and humic substances (HS). Fungi are considered more efficient than bacteria because most of bacteria cannot degrade various of these

substances (Esham et al., 2000; Machnikowska et al., 2002; Filip and Tesarova, 2004). However, recent studies showed that certain bacterial isolates have an enhanced capability for lignin breakdown, where apparently extracellular enzymes as peroxidases and laccase appear to be involved (Vargas-Garcia et al., 2007; Bugg et al., 2011).

Saprophytic fungi secrete non-specific oxidoreductases extracellular enzymes, such as manganese peroxidase (MnP), laccase (LAC) and lignin peroxidase (LiP). Apparently, MnP and LiP production is only limited to certain basidiomycetous fungi (Grinhut et al., 2007). In this context, two ecophysiological groups have been described as able to secrete these enzymes: white-rot and litter decomposers (WRF and LDF) (Dix and Webster, 1995), which secrete MnP in multiple forms into their surrounding microenvironment. When these enzymes are secreted by saprophytic fungi, decomposition of lignin (making up the so-called ligninolytic system) lead to the formation of unstable compounds (e.g. phenoxy radicals), which can undergo subsequent condensation and polymerization (humification) (Stevenson, 1994; Chefetz et al., 1998; Grinhut et al., 2007). The pathway followed by each enzymatic product (degradation or polymerization) is probably dependent not only on the enzymes and substrates involved, but also on reaction conditions, such as pH, humidity, percent oxygen and electrical conductivity (Grinhut et al., 2007). In addition, other enzymes such as cellulases, hemicellulases and esterases which depolymerize cellulose, hemicellulose and glucosides are also considered to be extremely important in the degradation process of lignocellulosic biomass (Tabka et al., 2006).

About 8500 species of basidiomycetes have been described, which are lignocellulose-degrading saprotrophs and half of them occur in soils, wood and fallen plant litter (Lynch and Thorn, 2006). Moreover, these groups of fungi are able to degrade recalcitrant molecules as well as HS. Based on their large size and stability, HS are not likely-metabolized intracellularly by microbial cells, instead they appear to be initially degraded by extracellular non-specific enzymes (Kastner and Hofrichter, 2001). It has been observed that white-rot fungi and litter decomposer fungi can degrade and transform macromolecules only in presence of other easily degradable C sources. For a review about this topic see Grinhut et al. (2007).

As a consequence of their extracellular enzymatic activity, WRF and LDF have been widely used for degrading lignocellulosic compounds of agricultural residues (crop residues), and thus, reducing the time-course of the stabilization into humic substances. For example WRF are well known for biodelignification of wood chips (Eriksson and Kirk, 1985), wheat straw (Tsang et al., 1987; Arora, 1995) and bamboo sticks (Reid, 1989).

4. Stabilization of OM by composting of crop residues

4.1. Biotic stabilization

Two main mechanisms are responsible for OM stabilization during composting: biochemical alteration and physicochemical protection. Biochemical alteration occurs when simple organic compounds are transformed by biotic (microorganism) processes into more resistant and larger molecular size forms of OM. These transformations are usually both referred to as “humification processes” and altered or transformed products as “humus” material (Senesi and Plaza, 2007). Humification is believed to be the primary process by which simple OM components are stabilized under the form of HSs (Stevenson, 1994). Under this context, the aim of any treatment for organic residues, such as composting, is to promote OM humification by increasing its biostability and maturity, thus improving its potential as an organic fertilizer and

Table 4
Abiotic additives used for organic compounds and compost stabilization.

References	Amendment or organic substrate	Abiotic additives	Mechanism
Qi et al. (2012a,b), Nishimoto et al. (2013)	Glycine, glucose Catechol	Steel slag(Fe(III) and Mn(IV) oxides)	Polycondensation reactions of humic precursors as quinones amino acids and saccharides
Brunetti et al. (2008)	Oil mill wastewater	Manganese oxide	Condensation, oxidative polymerization, ring cleavage, decarboxilization, and dealkylation reactions
Bolan et al. (2012)	Poultry manure and cow manure compost, green waste biochar	Goethite, Gibbsite, Allophane	Physicochemical protection of OM by complexation, reduced bacterial activity, low pH and poor availability of nutrients
Xu et al. (2012) Himanen and Hänninen (2009)	Sewage sludge Peat mixture	Fly ashes Sulfates and oxides of iron, magnesium, manganese, and zinc mixed with clay; calcium hydroxide, peroxide, and oxide	Promoting humification process pH alteration, decrease water-soluble ammonia among others

avoiding adverse environmental effects that may occur in soil (Senesi et al., 1989; Senesi and Brunetti, 1996; Plaza et al., 2002). Recently, Schnitzer and Monreal (2011) indicated that a microbial and biochemical humification path proceeds via synthesis of polyketides. In the latter process, intracellular and complex multi-enzyme systems cooperate to catalyze processes of condensation, aliphatic chain elongation, polyaromatic ring formation (aromatases) and a few other reactions. As a result of the latter biotic process, a large stable and chemically diverse alkyloaromatic structures are formed that serve as the backbone of HAs.

4.2. Abiotic stabilization

4.2.1. Stabilization by metal oxides

Recent studies have shown that abiotic oxidation of OM by oxides such as iron oxides (FeOx), aluminum oxides (AlOx) and manganese oxides (MnOx) have a positive effect on organic residues, increasing the residence time and accelerating the humification process of organic matter in soils (Brunetti et al., 2008; Qi et al., 2012a,b; Saidy et al., 2012). Humification by abiotic agents has been described for several agricultural wastes. In particular, MnOx have shown to be more reactive catalyzers than other soil mineral constituents such as AlOx and FeOx in the polymerization of phenolic compounds based on their high oxidation potential and high specific surface reactivity (Shindo and Huang, 1984; Wang and Huang, 2000). In this context, Brunetti et al. (2008) found that manganese(II) oxide (MnO₂) was able to catalyze efficiently the humification of oil mill wastewater promoting the chemical OM alteration, possibly acting in some reactions such as condensation, oxidative polymerization, ring cleavage, decarboxilization, and dealkylation reactions (Jokic et al., 2004).

The use of abiotic humification process by inorganic oxidants has shown various advantages. This method has a fast processing time, and no selectivity for different OM sources. Qi et al. (2012a,b) used “steel slag”, which is characterized by a high content of calcium, iron, manganese and silicon, all of which are present mainly as oxides that are tightly bound to the slag matrix. Qi et al. (2012b) reported that Fe(III) and Mn(IV)-oxides in steel slag can act as effective oxidants and substantially accelerate poly-condensation of model humic precursors (glycine, glucose and catechol) to form FA, which are subsequently transformed into HA. Moreover, Qi et al. (2012a,b) found that humification reactions can inhibit the release of metals from steel slag under acidic conditions, which presents environmental advantages. Xu et al. (2012) found similar results in stabilized sewage sludge using fly ash as catalyst. For these reasons, this method of abiotic composting has

attracted particular attention, and currently a series of novelty materials are being used for humification of organic amendment (Table 4).

4.2.2. Stabilization by clay colloids

The application of clay materials as stabilizing agents (goethite, gibbsite, allophane, among others) aims at decreasing the decomposition rate of HS. For example, it has been described that allophane spherules, a non-crystalline or ‘short-range order’ aluminosilicate (Parfit, 1990; Brigatti et al., 2006; Abidin et al., 2007; Hashizume and Theng, 2007; Creton et al., 2008), have an important role in soil OM stabilization (Chenu and Plante, 2006). Calabi-Floody et al. (2011) found that in volcanic soils a significant amount (11.8%) of carbon is so strongly held by nanoclay surfaces as to resist repeated treatments with hydrogen peroxide. About 50% of this carbon is identifiable as black carbon-derived compounds, suggesting that allophanic nanoclays in volcanic soils play an important role in carbon stabilization. The role of some clay minerals in the stabilization of compost (poultry and cow manures) has been evaluated in a recent study (Bolan et al., 2012). Results showed that the addition of clay materials to compost decreased the rate of decomposition, thereby increasing the stabilization and half-life of C. Half-life value of poultry compost manure increased from 139 days to 620, 806 and 474 days with the addition of goethite, gibbsite and allophane, respectively (Bolan et al., 2012). In this case, the decrease in decomposition rate may be attributed to the interaction between C of OM and Al and Fe oxides mainly through the formation of stable organo-mineral complexes. In addition, Schnitzer and Monreal (2011) indicated that polymerization and cross-linking of alkyloaromatic moieties of HAs by surfaces of clay colloids result in stabilizing HA macromolecules and soil organic matter.

Bolan et al. (2012) and others suggests that OM preservation from stabilized compost occurs in a similar way as that in soils, basically influenced by a number of processes including: (i) physicochemical protection of OM by complexation with iron, aluminum and allophane in these cases (Eusterhues et al., 2003; Mikkuta et al., 2006; Matus et al., 2008; Calabi-Floody et al., 2011; Garrido and Matus, 2012); (ii) reduced bacterial activity that results from the presence of free iron and aluminum; and (iii) low pH and poor availability of nutrients, especially phosphorus to soil microorganisms involved in OM degradation (Parfitt, 2009).

The physicochemical protection is referred to as organo-mineral interactions between HSs and minerals, which occur at the molecular level from nanometer to micrometer scales. As a result of these interactions, the microbial attack of OM (decomposition) is

prevented. Some physicochemical ways for OM stabilization are the complexation with soil minerals and occlusion within soil aggregates, among others, which produce highly inaccessible OM to decomposers and extracellular enzymes (Monreal et al., 1997; Jastrow et al., 2007). Therefore, the physicochemical protection can be defined as the spatial inaccessibility for biotic decomposers due to several mechanisms such as intercalation, occlusion and interaction with mineral surfaces and metal ions. Mineral surface interaction (Fe-, Al-, Mn-oxides, phyllosilicates) with OM is one of the most important mechanisms for OM stabilization that protects it against the decomposers. Sorption occurs by diverse organo-mineral associations, such as polyvalent cation bridges, hydrogen bonding, van der Waals forces, and interactions with hydrous oxides and aluminosilicates (Jastrow et al., 2007).

4.3. Properties of stable OM in compost

It has been well established that poorly-ordered Fe, Mn and Al-Fe Oxides are the most reactive components that facilitate humification from precursors such as amino acids, polyphenols, and reducing sugars (Wang and Huang, 2000; Chen et al. 2010), this being concomitant with the organo-metal complex formation, among other physicochemical interactions. Currently, these inorganic colloidal agents are an alternative to agricultural residues treatment and stabilization processes which are applied to several agricultural residues such as olive oil mill wastes, poultry manure, cattle manure, among others. Noteworthy, there is no sufficient information about using these inorganic compounds in the humification of organic materials, or in composting processes of lignocellulosic materials. The reader is directed to an article published by Mtui (2009) dealing with recent technologies that have been used for the treatment and pretreatment of lignocellulosic biomass and production of value-added products which include mechanical, physical and biological systems.

5. Biotechnological applications of stabilized agricultural wastes

5.1. Restoration of eroded and degraded soils

Stabilized agricultural residues are usually used as soil amendments for enhancing soil properties (i.e., structure, nutrients, OM content). These amendments have been used for decades in amelioration and remediation of polluted, degraded and eroded soils. Soil erosion affects physically and chemically to soils and degrades its structure, altering soil fertility among other properties (Lal, 2003). This destructive process causes physical loss of topsoil with its constituent nutrients and soil OM, exposing at the surface less fertile subsoil, which is usually characterized by a low structural stability and productivity. This type of erosion adversely affects soil quality and ecosystem functions (Srinivasan et al., 2012). Annually, the global rate of erosion is about 75 billion Mg (Pimentel et al., 1995) and 100 Mg ha⁻¹ for severely eroded soils (Lal, 2003). As a consequence of soil OM losses by erosion, the content of soil organic C is often <1–2%, resulting thus necessary to increase soil OM content, by intensification of crops or by incorporation of organic amendments.

Currently, the use of compost for greenhouse gases (GHG) mitigation has drawn increased attention. Compost application is considered an important tool, because it recycles crop nutrients and decreases the requirements of synthetic fertilizers and pesticides; thereby reducing GHG emissions due to use of fossil fuel associated with their industrial production and application (Cogger, 2005; Favoino and Hogg, 2008). Use of composted OM also increases and accelerates the growth in plants, thereby increasing CO₂

uptake and C storage within the plant and C sequestration in soil (Mondini et al., 2007; Favoino and Hogg, 2008; Arriagada et al., 2014). Composts may also improve tillage and workability of soil, thereby reducing emissions (Favoino and Hogg, 2008; Lou and Nair, 2009).

5.2. Ecological restoration

Soil amendments produced from agricultural residues are widely used for soil and ecological restoration to enhance the health of degraded systems. After environmental degradation, restoration and reclamation efforts can be hampered by poor physico-chemical soil characteristics and reduced diversity in the community of soil microorganisms (Ohsowsky et al., 2012). In general, soil amendment with composts enhances the nutritional characteristics (Allievi et al., 1993), soil properties (Table 5) and microbial activity (Liu et al., 2007). For this reason, compost application in ecological restoration has a positive effect on the vegetation cover. Tandy et al. (2011) reported that in a short-time experiment, compost applied to a nutrient-poor alkaline soil increased OM, nutrient content, and water holding capacity, whilst lowering soil pH. These changes were linked to an increase in vegetation cover. In an eroded soil from the Sonoran desert (USA), however, compost and inoculant (bacteria and arbuscular mycorrhizal fungi) application together with native plants (*Prosopis articulata*, *Parkinsonia mycophylla* and *Parkinsonia florida*), had no long-term beneficial response on vegetation cover in this degraded ecosystem (Bashan et al., 2012). In the latter study and in the short-term (<3 months), plants had a positive response on growth (height, number of branches and diameter of the main stem) until three months after the application of compost. The short-term benefits can be explained by the quick mineralization of some carbon sources from compost (Godbout et al., 2010).

5.3. Remediation of polluted soils

As a consequence of some industrial activities, there are significant levels of pollutants in soils, and other natural systems, which alter the functioning and equilibrium of ecosystems. Industrial activities, such as mining, are associated with important pollution problems in soils (Lottermoser, 2003). The discharge of metal-enriched solid and liquid effluents into the air, water and soils generate a deleterious effect on the ecosystems particularly in soils (Meier et al., 2012). In developing countries, mining is an

Table 5
Effects of stabilized agricultural wastes applied as soil amendment.

References	Effects in soils
Crecchio et al. (2004), Adani et al. (2006, 2007) Bustamante et al. (2010), Bolan et al. (2012)	Enhancing the C stock and increasing the colloidal humified organic matter
Spark et al. (2008), Olivier et al. (2012)	Increasing the activity of microbial communities and VOC emissions by plants
Alguacil et al. (2011)	Promoting the activity of arbuscular mycorrhiza
Stucky and Hudak (2001)	Plant growth promoting
Gaind and Nain, 2010; Srinivasan et al. (2012)	Improving chemical and physical properties of soil
Eriksen et al. (1999), He et al. (2000), Wortman and Walters (2007)	Mineralization of nitrogen and phosphorus content in soil
Tandy et al. (2011), Bashan et al. (2012)	Ecological restoration of eroded soils: Enhances WHC, increases soil OM, promote bacteria and AMF activities
Bolan et al. (2003), Park et al. (2011), Nielsen et al. (2011), González et al. (2012)	Remediation of metal-polluted soils: increasing metal sorption, changes in pH, increase of microbial activities, reduction of metals

important economic activity, being responsible for generating a great amount of pollutants. Solid mine wastes contain a large amount of metals in particulate matter, and also a high volume of wastewater is generated during the mining and metal extraction processes (Gray et al., 2003).

When heavy metals and metalloids are incorporated into the soil, they undergo different reactions. Their adsorption and complexation at the interface between the soil solution and solid phases result in their accumulation. Adsorption reactions reduce the formation of surface complexes which can be either physical attachment or bonding of metal ions and molecules onto the surface of another component. Metal adsorption is highly dependent of soil pH; e.g. the sorption of cationic metals increases when increasing pH while that of anionic species decreases when increasing pH (Adriano et al., 2004), and on the presence of other soil organic and inorganic components, redox potential and cation exchange capacity (Naidu et al., 2008). For instance, soil OM has a high affinity for metal cations due to the presence of ligands or functional groups (Senesi, 1992c; Harter and Naidu, 1995). When soil pH increases, H^+ dissociates from functional groups such as carboxyl, phenolic and hydroxyl functional groups, thereby increasing the affinity for metal cations (Park et al., 2011).

Heavy metal and metalloids can form different complexes with soil inorganic surfaces due to various hydroxyl groups with different levels of reactivity. These complexes can be inner-sphere complexes, when metal(loid) ion is directly bound to functional groups of soil particles or outer-sphere complexes when water molecules are interposed in between functional groups of soil particles and metal(loid) ions (Park et al., 2011). In the same sense, organic complexes between metal(loid) and organic acids such as HA and FA, can be affected by different factors such as temperature, soil pH, ionic strength, dominant cations and soil type (texture), being soil pH the main factor on organic-metal complexes formation (Senesi, 1992c; Bolan et al., 2003; Luo et al., 2010). Another important reaction experienced by metals in soils is the precipitation, which is defined as the process of metal(loid) immobilization in presence of inorganic anions such as sulfate, carbonate, hydroxide and phosphate when the soil pH and the metal(loid) concentration are high (Adriano, 2001). Because phosphate are able to precipitate metals, they are the most common compounds used to effectively precipitate heavy metals and metal(loid)s in contaminated soils or water. According to the degree of stability of precipitates between phosphate and metals, Pb forms the most stable precipitate and Zn the lowest one (Bolan et al., 2003). Other important metal stabilizing reactions in the environment are: (a) oxidation/reduction reactions by microorganisms; e.g. reduction of Se(VI) to Se(0) by bacteria; (b) methylation/demethylation, which is considered to be the major process of volatilization of As, Hg and Se in soils, resulting in the release of toxic methyl gases (Cernansky et al., 2009); and (c) the biological modification of local soil environment as a consequence of some microbial processes which can enhance metal(loid) solubility, thereby increasing their bioavailability and potential toxicity (Park et al., 2011).

The use of stable composts may help remediate soils contaminated by mining activities especially if the concentration of one or more metals exceeds the specified threshold level in soil profile. A number of environmental remediation systems using physical, chemical, or biological treatments have been developed in recent decades (Meier et al., 2012), highlighting the immobilization techniques, which are focused on reducing the availability and activity of trace elements, but not on removing the pollutant from the soil. The purpose of immobilization processes is to accelerate natural ones such as sorption, complexation and precipitation reactions using amendment with the aim of decreasing the mobility and bioavailability of pollutants (heavy metals among others) in soils (Bolan et al., 2003).

Inorganic materials (i.e. iron oxides, phosphate and calcium) and/or organic materials) have been used for increasing the stability and complexation of contaminants in polluted soils (Castaldi et al., 2005; Derome, 2009; Nielsen et al., 2011). González et al. (2012) found in a metal-As polluted soil that the combination between an agricultural residue and marble sludge (8% marble sludge with 2% compost) was most effective increasing the immobilization of Cd, Cu, Pb and Zn. None of these treatments, however, proved to be effective in reducing the soluble concentrations of all trace elements involved. Actually, amendment was effective at immobilizing one pollutant, but may increase the mobility of others (Hartley et al., 2004).

Organic amendments usually enhance bioremediation of heavy metal(oid)s through various processes that include immobilization, reduction, volatilization and rhizosphere modification (Park et al., 2011). The immobilization is referred to adsorption reactions, where the organic amendments induced retention of metal(loid)s which is attributed to an increase in surface charge (Clark et al., 2007) and the presence of metal(loid) binding compounds (Gondar and Bernal, 2009). Bolan et al. (2003) found that the incorporation of compost increased the surface charge of the amended soils, which is attributed to the higher pH and surface charge of the composted biosolid. Other organic amendments have been effective in decreasing metal bioavailability by immobilization. For example, Cu is immobilized in soil because of increased the Cu-organic complexes formation, and Cr by reduction from Cr(VI) to Cr(III), and subsequent precipitation as chromic hydroxide (Park et al., 2011). Generally, organic amendment application has shown to have a positive effect in soil improving soil physical characteristics such as particle size distribution, cracking pattern and porosity. Improved soil structural characteristics contribute to prevent the dispersion of metal contaminated particles by formation of water stable soil aggregates (Park et al., 2011).

Further, metal(loid)s are subject to redox reactions which can have a biotic or abiotic origin, being the reduction reaction the most important in some metalloids such as Arsenic. It has been noticed that organic amendments such as compost from crops residues among other, have a positive effect enhancing the Cr and Se reduction (Bolan et al., 2003; Chiu et al., 2009; Hsu et al., 2009), where Cr(IV) is reduced to Cr(III) which is less available to plants. Under anaerobic conditions and increased OM content, As(VI) can be biotically reduced to As(III) resulting in higher toxicity and mobility in the soil environment (Wakao et al., 1988). The amendment application in polluted soil enhances the dissolved organic carbon (DOC) content, and the easily oxidizable fraction of DOC providing the energy source for soil microorganisms involved in the reduction processes of metal(loid) (Chiu et al., 2009). Moreover, DOC and natural OM may be associated with Hg altering its speciation and bioavailability in ecosystems; this form strongly bonds with humic substances and appears to stabilize Hg(II) in soil environment (Park et al., 2011). Another important effect of organic amendment such as crop residues compost application is the modification of rhizospheric environment. As a consequence of these changes (pH, organic acids, soil solution composition, microorganism activity) the metal chemistry can influence transformation, mobility and availability of metal(loid)s in soils (Akhtar and Malik, 2000; Caravaca et al., 2005; Perez-de-Mora et al., 2006). The complexation of heavy metals with soluble organic compounds increases their mobility (Neal and Sposito, 1986). In comparison, organic ligands of stable soil macromolecules (i.e., with aliphatic components) in humic substances may reduce metal mobility and associated environmental toxicities (Monreal et al., 1998). Thus, increasing the content of long-chain components as lipids and alkanes in humified organic materials will help stabilize heavy metals in soils preventing their transfer into environment and food chain.

Bioremediation, such as phytoremediation is an important technique used for remediation of heavy metal polluted soils (Brunetti et al., 2012; Ghanem et al., 2013). Plant-mediated decontamination/detoxification processes are commonly referred to as “phytoremediation” (Rajkumar et al., 2012). Phytoremediation is generally considered an inexpensive and environment friendly technique; and it may involve various processes: phytoextraction, phytoimmobilization, phytotransformation, phytodegradation, phytostimulation, phytovolatilization, and rhizofiltration (Ashraf et al., 2010). The main disadvantage of phytoremediation is, however, a slow process that requires several years or decades to reduce metal concentrations in soil at levels that are harmless to humans and other organisms (McGrath and Zhao, 2003). This slowness is due to the limited growth and biomass production of hyperaccumulator plants (Peuke and Rennenberg, 2005). For this reason, enhancing the soil conditions for metal hyperaccumulator plants may be a priority in phytoremediation research. Hajishah et al. (2010) evaluated the phytoextraction of Ni, Cd and Pb using different amendments (compost and EDTA among others) and found that the effectiveness of sugarcane compost to stimulate the accumulation of Cd, Pb in plant shoots were 2.8 and 2.9 times higher than the control respectively. Further, they found that all treatments (compost and EDTA) were higher in terms of solubilizing at different levels soil Pb, Cd and Ni for root uptake and translocation into canola shoots. These results suggested that the amendment used could be an alternative in the remediation and phytoremediation of heavy metal polluted soils. Finally, several microorganisms such as bacteria and mycorrhizal fungi have been described as enhancers of phytoremediation process in heavy metal polluted soils (Arriagada et al., 2009a; Brunetti et al., 2012; Meier et al., 2012).

Actually, soil microorganisms as arbuscular mycorrhizal fungi (AMF) are involved in diverse biochemical processes and nutrient cycling. These processes enhance or accelerate the re-establishment of a plant cover, thereby increasing the stability of polluted ecosystems (Reynolds et al., 1999; Moynahan et al., 2002; Arriagada et al., 2007). In this context, soil microorganisms have developed several mechanisms to resist or tolerate the toxic effects of metals in polluted soils. Particularly, in the case of soil fungi, the tolerance/resistance mechanisms developed include; (a) adsorption of metals to the cell wall surface (biosorption), (b) transportation and cellular incorporation (bioaccumulation), and (c) transformation of metals through redox or methylation reactions (Gadd, 1986, 1993).

The ability of AMF to confer resistance to plants against metal and metalloids has been reported in several studies (Janousková et al., 2005; Hildebrandt et al., 2007; Arriagada et al., 2009b; Aguilera et al., 2011; Meier et al., 2011, 2015; Barea et al., 2013; Cornejo et al., 2013; Seguel et al., 2013). These observations and the positive effect of mycorrhizal symbiosis on the phytoremediation of metal-polluted soils are of great biotechnological interest, as mycorrhizal plants are as effective in extracting metals (e.g., Cu, Cd, Pb, Zn) as non mycorrhizal-hyperaccumulator plants (Huang and Cunningham, 1996; Ebbs and Kochian, 1998; Arriagada et al., 2010). Mycorrhizal plants also improve phytostabilization because metals (Zn, Cd, and Cu) are bound to hyphae and roots without translocating these elements to shoots (Joner and Leyval, 1997, 2001). As such, metals remain in the soil but because they are less bioavailable, toxicity to other organisms is reduced (Leyval et al., 2002). The production of extracellular polymeric substances (EPS), mucopolysaccharides and proteins by plant-associated microbes can also play an important role in complexing toxic metals and in decreasing their mobility in the soils (Rajkumar et al., 2012). However, production and accumulation in soils of glomalin, a glycoprotein produced by AMF, is an important mechanism to enhance phytoremediation process, since glomalin can

immobilize in a stable form several toxic elements, such as Pb, Cu, Zn, Al (González-Chávez et al., 2004; Cornejo et al., 2008; Vodnik et al., 2008; Aguilera et al., 2011; Seguel et al., 2015). Therefore, the manipulation and use of AMF as a tool for polluted soils begins to be considered in phytoremediation programs using mycorrhizal plants (Meier et al., 2012). Further research is required to understand the long-term effects on metal sequestration plant-AMF association and the combined effects of organic amendments application.

Several studies showed that the application of organic amendments as compost derived from agricultural residues have a positive effect on microbial activity and plant growth (Table 3) suggesting their possible use and also the use of phytoremediation plans to enhance the immobilization of metals and promote soil microbial activity. Alguacil et al. (2011) found that the addition of sugar beet waste amendment was a good strategy for the remediation and/or phytostabilization of mine tailing sites. Alguacil et al. (2011) and Meier et al. (2011) have also found that the application of sugar beet waste (treated with temperature) was very efficient with regard to increasing AMF communities in the majority of the host plant species assayed and decreasing the concentration of metals in shoot tissues.

Within this context, Medina et al. (2010) found a positive interaction between a dry olive cake (compost) with a AMF strain (isolated from a Cd-contaminated soil). Such interaction resulted in a highest growth of *Trifolium repens*, which may be associated to enhanced nutrient acquisition and the immobilization of Cd in the compost amended soil. Despite the latter results, the long-term potential of organic amendment from agricultural residues, for metal(loid) immobilizing material, however, is still unknown. Little is known about the nutrient mineralization rates of different amendment materials in soils, especially the long-term effects. Current studies have reported that different organic amendments have high decomposition rates (Godbout et al., 2010). Thus, the positive effects of plant-AMF associations on metal(loid) immobilization in polluted soil could be functioning in a short-term (<1 year), being necessary frequent compost applications. Therefore, to development of improved and stabilized amendments with an increased residence time in soils is still necessary in order to enhance the waste management and promote the remediation of degraded and polluted soils.

6. Conclusions and future prospects

The application of composted organic amendments derived from different crop residues has a positive impact on the physical, chemical and biological properties of soils. The application of organic amendments also helps to immobilize metals and some metalloids through adsorption and complexation reactions, reducing their bioavailability and limiting their further transfer through plant uptake and leaching. Moreover, their joint use (microorganisms, plants, organic residues and inorganic materials) could be a useful tool for enhancing phytostabilization processes in environmental conditions. In this sense, the novel technologies of OM stabilization here reviewed appears as an important tool for the production of improved amendments finalized to agriculture and remediation of polluted and physically degraded soils. Finally, the new strategies for improved compost production based on agricultural residues may stimulate farmers to keep those materials revaluing and reusing them in scenarios as above described.

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