REVIEW ARTICLE

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Composting with additives to improve organic amendments. A review

Justine Barthod[1] · Cornelia Rumpel[1] · Marie-France Dignac[2]

Abstract
Composting and vermicomposting are sustainable strategies to transform organic wastes into organic amendments, valuable as potting media or soil conditioner. However, the negative aspects of these processes are emissions of greenhouse gases and odorous molecules and final product potentially containing toxic compounds. These negative aspects can be limited through the addition of organic, inorganic or biological additives to the composted or vermicomposted mixture. The aims of this review are (1) to present the main characteristics of composting and vermicomposting processes with and without additives, (2) to show the influence of additives on greenhouse gas emissions during waste degradation and (3) to report the effects of additives on the properties of the final products (heavy metal and nutrient contents), in view of their use as a soil conditioner or potting media. Finally, the feasibility and potential environmental benefits of co-composting and co-vermicomposting are discussed. Our results show that additives affect composting parameters such as temperature, pH and moisture and thus have an impact on the composting process. They may be used to reduce gas emissions and mobility of mineral ions. The various additives have contrasting effects on the quality of the final product and its impact on soil quality. The use of worms and additives seems to increase plant available nutrient contents, while decreasing N leaching, heavy metal mobility and composting time. Co-composting and co-vermicomposting strategies need to be locally optimised, involving the generated amendments in a circular economy to improve sustainability of agricultural systems.

Keywords Composting · Vermicomposting · Additives · Waste management · Organic amendment · Potting media

Contents

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1 Introduction

The growing world population and the resulting higher consumption of goods and services has driven a rapid increase of organic wastes originating from households, industry and agriculture (Hoornweg et al. 2013). This situation generates serious environmental issues, calling for safe and sustainable strategies to treat these wastes, from their production to their recycling or elimination. On the other hand, the organic fraction of wastes represents a valuable organic resource, which could be recycled and transformed into nutrient-rich fertiliser and/or soil conditioner (Marshall and Farahbakhsh 2013; Bernstad et al. 2016; Calabi-Floody et al. 2017). Biological degradation during composting and vermicomposting is one of these strategies to transform organic wastes into organic amendments. These amendments can be applied on soils to increase soil carbon stocks and associated ecosystem services (Francou et al. 2005; Lashermes et al. 2009, Peltre et al. 2012; Bernstad et al. 2016) or used as potting media (Hashemimajd et al. 2004; Zaller 2007; Ceglie et al. 2015; Morales et al. 2016; Wang et al. 2017; Malińska et al. 2017). Compost application leads to improved soil structure, reduced erosion and increased water holding capacity (Diacono and Montemurro 2010). However, compost production is associated with greenhouse gas (GHG) emissions such as NO_2, CH_4 or CO_2, contributing to global warming (Bernstad et al. 2016). In addition, immature composts have negative effects on plant germination and, more generally, on plant development (Bernal et al. 2009) and might cause environmental problems such as water pollution and odour emissions (Wang et al. 2016).

Composting is an aerobic process, performed at large scale in windrows or piles (Hobson et al. 2005) or at local scale with a home composter (Andersen et al. 2011). Composting is used to treat manures, green wastes or municipal solid wastes and their fractions (Goyal et al. 2005). Vermicomposting, involving the presence of worms, is used for similar purposes (Chan et al. 2011; Garg et al. 2006; Ngo et al. 2011).

Various substrates might be added to waste during the composting process. They are either organic (Gabhane et al. 2012; Zhang et al. 2017), mineral (Wong et al. 1995; Himanen and Hänninen 2009; Wang et al. 2016), biological (Wakase et al. 2008; Jurado et al. 2015; Awasthi et al. 2017) or a mixture of substrates (Hayawin et al. 2014; Awasthi et al. 2018). Some added substrates are considered as bulking agents, when they only act on the physical structure of the compost (aeration), but most of the time, these substrates have also direct or indirect effects on other composting parameters and can be considered as additives (Villasenor et al. 2011). Additives are used to enhance the composting process by reducing leaching (Steiner et al. 2010) and gas emissions (McCrory and Hobbs 2001; Awasthi et al. 2016a, b), improving compost aeration or accelerating organic matter degradation (Sánchez-García et al. 2015) and improving nutrient content and availability in the final product (Gabhane et al. 2012; Morales et al. 2016). Only few studies assessed the effect of additives on vermicomposting (Wang et al. 2014; Barthod et al. 2016; Malińska et al. 2017) (Fig. 1).

Many studies were published in the last years proposing to use various additives during composting and vermicomposting of different feedstocks, but no critical review exists to compare their benefits. The main aims of this article are to present the different additives used during composting and vermicomposting and to evaluate their effect on the main composting parameters and on the quality of the end product in terms of their effect on soil functions and plant growth. The objective of this review is to give recommendations with regard to additive use during composting and vermicomposting. Finally, perspectives and feasibility of using additives during both processes will be discussed.

2 Biological degradation processes and additives used for organic waste treatment

2.1 The composting process

Composting is defined as a bio-oxidative process leading to organic matter mineralisation and transformation (Zucconi and de Bertoldi 1987). The end product is considered as stabilised and free of phytotoxic agents and pathogens (Wichuk and McCartney 2007). Composting typically consists of three phases: initial activation, followed by a thermophilic and a maturation phase (Fig. 2). The initial activation generally lasts 1–3 days during which simple organic compounds such as sugars are mineralised by microbial communities, producing CO_2, NH_3, organic acids and heat (Bernal et al. 2009). During this phase, the temperature in the composting pile increases. Thereafter, during the thermophilic phase, the temperature reaches a maximum. The optimum temperature range for composting is 40–65 °C (de Bertoldi et al. 1983), allowing to kill pathogens above 55 °C. During this phase, thermophilic microorganisms degrade fats, cellulose and lignin (Bernal et al. 2009). Finally, during the mesophilic phase or maturation, temperature slowly decreases due to reduced microbial activity resulting from a decrease of biodegradable compounds. Composting involves a succession of microbial communities (Insam and de Bertoli 2007; Mehta et al. 2014), according to the temperature of the composting pile. For instance, fungi are not present at a temperature above 60 °C, when bacteria predominate (Klamer and Bååth 1998).

Fig. 1 Worms and additives might change the duration of the composting process and the quality of the final product

Thus, the relative abundance of specific microorganisms and temperature are good indicators of the compost evolution.

Biogenic CO_2 emissions during composting derive from organic matter aerobic decomposition and CH_4 oxidation by aerobic methanotrophic bacteria (Sánchez-García et al. 2015). CO_2 is the most abundant gas emitted during composting (Haug 1993). CH_4 emissions are mostly recorded during the initial and thermophilic phases (Beck-Friis et al. 2000), due to the formation of anaerobic spots. Nitrous oxide (N_2O) emissions can occur during the whole process due to denitrification. NH_3 emissions may also occur during composting, generally during the first 2 weeks. They decrease when the easily degradable materials are exhausted and the degradation rate decreases (Morand et al. 2005; Szanto et al. 2007).

The composting process and compost quality and maturity are controlled by several parameters influencing the microbial activity, such as the initial pH and C/N ratio of the feedstock, their particle size and their distribution, as well as the aeration and humidity of the compost pile (Bernal et al. 2009, Onwosi et al. 2017). The temperature profile during composting is influenced by the composition of the initial feedstock, aeration through pile turning,

compost humidity and the addition of additives or bulking agents (Pasda et al. 2005). It influences compost duration, compost quality and maturity, but also compost sanitisation (Soobhany et al. 2017). CO_2 and NH_3 emissions, mainly produced during the thermophilic stage of organic matter degradation (Maulini-Duran et al. 2014; Pagans et al. 2006), are affected by the C/N ratio of the composted material, the maximum temperature reached during the process and the aeration of the pile (Pagans et al. 2006). The presence of anaerobic zones is responsible for N_2O and CH_4 emissions (Nasini et al. 2016).

Controlling compost parameters and influencing them with additives have major impacts in terms of compost quality (maturity, heavy metal contents, nutrient content and bioavailability) and environmental impacts of composting, such as production of greenhouse gases and other volatile compounds such as NH_3, sulphur-containing compounds and volatile organic compounds (Maulini-Duran et al. 2014; Cerda et al. 2018).

2.2 The vermicomposting process

During vermicomposting, temperature ranges from 25 to 37 °C, depending on the worm species. Generally, worm activities and development are negatively affected by higher or lower temperatures. Four species are extensively used in vermicomposting facilities. They comprise two tropical species [*Eudrilus eugeniae* (Reinecke et al. 1992; Kurien and Ramasamy 2006; Hayawin et al. 2014) and *Perionyx excavatus* (Suthar and Singh 2008)] and two temperate ones [*Eisenia andrei* and *Eisenia fetida* (Kaushik and Garg 2003; Garg et al. 2006; Malińska et al. 2017; Alavi et al. 2017)]. These species have been classified as epigeic worms and correspond to worms, which mainly feed on fresh organic matter such as litter, manure and compost (Bouché 1977). Most vermicomposting facilities and most studies are using the worms *Eisenia andrei* and *Eisenia fetida* due to their high rate of consumption, digestion and assimilation of organic matter, adaptation to a wide range of environmental factors, short life cycles, high reproductive rates and endurance and resistance during handling (Domínguez and Edwards 2010).

Although pathogen removal has been recently demonstrated to occur during transit in the worm gut (Soobhany et al. 2017) as well as during composting, combining vermicomposting to composting is the preferred process, because it favours the removal of pathogen microorganisms and increases organic matter decomposition rate (Frederickson et al. 1997, 2007; Wang et al. 2014). Wastes are composted during a few weeks, until the end of the thermophilic phase. Then, after some days at high temperature, pre-mature compost is cooled by spreading it on vermicomposting beds (thin layers).
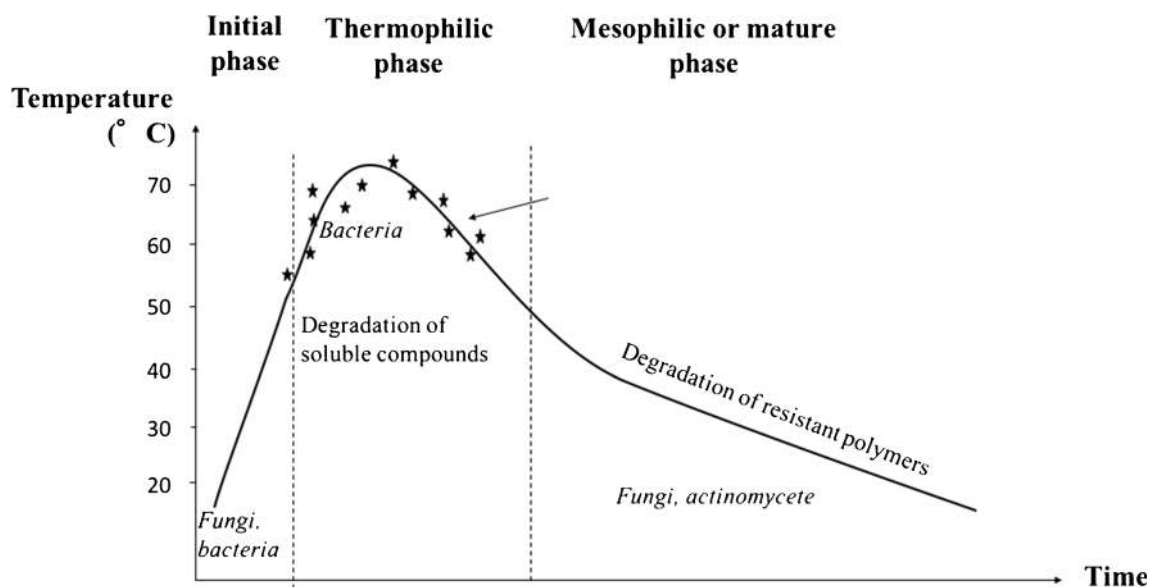
Fig. 2 Temperature changes during composting and corresponding microbial communities and organic compounds. Stars indicate the phases when the removal of pathogen microorganisms takes place (after Insam and de Bertoli 2007). Mineral, organic as well as biological additives have been shown to stimulate the microbial activity, leading to an earlier start and a longer duration of the thermophilic phase as compared to regular composting

Vermicomposting produces GHG, similar to composting. Worm activity accelerates and enhances organic matter decomposition leading to higher CO_2 emissions than those of regular composting (Chan et al. 2011; Nigussie et al. 2017). The emissions of N_2O are either increased (Hobson et al. 2005) or decreased (Wang et al. 2014; Nigussie et al. 2017) compared to composting, probably depending on the feedstock materials.

As already noted for composting, vermicomposting parameters must be controlled because they influence the rate of decomposition, the quality of the end product and the amount of GHG emitted. In addition, for vermicomposting, worm mortality must be avoided, by regulating, in particular, the temperature and the moisture content. As for composting, the parameters controlling vermicomposting may be influenced by the addition of mineral, organic or biological substrates (Suthar 2009; Li et al. 2009; Malińska et al. 2017). For vermicomposting, the choice of additives is also driven by their possible negative impacts on worm development (Malińska et al. 2016).

2.3 Types and sources of additives used during biological waste treatment

A wide variety of additives, classified in three categories (mineral, organic or biological) may be used to enhance composting and vermicomposting processes and/or the quality of the final product (Bernal et al. 2009; Onwosi et al. 2017). A non-exhaustive list of the common additives is given in Table 1.

Biological additives refer to microorganisms, which are inoculated to a compost or vermicompost pile. These microorganisms are generally isolated from composts during the thermophilic phase, cultivated and sold as a commercial solution. Effective microbes and vertical transmitter bacteria are common commercial additives (Manu et al. 2017). Few manufacturers reveal the identity, fate and functions of the microbes present in the commercial solutions. However, in most of the commercial microbiological additives, *Alcaligenes*, *Bacillus*, *Clostridium*, *Enterococcus* and *Lactobacillus* microorganisms are present (Sasaki et al. 2006; Wakase et al. 2008). These microorganisms are involved in the ammonia assimilation during composting (Wakase et al. 2008) or in the decomposition of lignocellulose (Jurado et al. 2015).

Organic additives cover a large variety of products: residual straws, mature composts, refuse from green waste compost screening, grass clippings, crushed hardwood materials, crushed wood pallets, bark and cornstalks (Doublet et al. 2011). When choosing organic additives, attention must be paid to the C/N ratio of the initial mixtures to ensure organic matter degradation and prevent N leaching during composting (Doublet et al. 2011). Furthermore, biochar, a highly aromatic pyrolysis product (Lehmann and Joseph 2015), has recently received great interest as highly stabilised organic additive for composting (Dias et al. 2010; Waqas et al. 2017) and vermicomposting (Malińska

Table 1 The different types of additives used during composting and their possible effects

Waste	Additives	Improvement of process parameters						Decrease of gas emissions				Compost quality			Soil health	References
		Temperature (°C)	Duration	Microbe activity	Moisture	Aeration	pH	CO$_2$	CH$_4$	GHG	Odour (N, S)	C seq.	N, P	Heavy metals		
Manure	Minerals							G				C				Bolan et al. (2012)
Manure	Charcoal						P	G			G					Chen et al. (2010)
Manure	Biochar							G				C				Chowdhury et al. (2014)
Manure	Biochar					P										Czekała et al. (2016)
Manure	Biochar		P													Dias et al. (2010)
Manure	Chestnuts, leaf litter													C		Guerra-Rodriguez et al. (2006)
Manure	Phosphogypsum									G						Hao et al. (2005); Luo et al. (2013)
Manure	Compost, inoculum		P													Kato and Miura (2008)
Manure	Biochar, sawdust										G					Khan et al. (2014)
Manure	Ash										G					Koivula et al. (2004)
Manure	Straw					P										Kulcu and Yaldiz (2007)
Manure	Zeolite										G		C			Lefcourt and Meisinger (2001)
Manure	Bentonite				P								C			Li et al. (2012)
Manure	Rock P													C		Lu et al. (2014)
Manure	Rice straw												C			Lü et al. (2013)
Manure	Elemental S						P	G			G					Mahimairaja et al. (1994)
Manure	Rock P												C		S	Nishanth and Biswas (2008); Billah and Bano (2015)
Manure	Rock P, phosphogypsum														S	Oliveira and Ferrera (2014)
Manure	Mg hydroxide, phosphoric acid										G					Ren et al. (2010); Jeong and Hwang (2005)
Manure	Biochar										G					Steiner et al. (2010)
Manure	Zeolite						P									Venglovsky et al. (2005)
FW	Sawdust				P											Chang and Chen (2010)
FW	Ash														S	Kuba et al. (2008)
FW	Inoculum	P		P			P					C				Manu et al. (2017)
FW	Biochar	P			P		P									Waqas et al. (2017)
FW	Fly ash; lime						P									Wong et al. (2009)
FW	Cornstalk, sawdust									G			C			Yang et al. (2013)
FW	Na acetate			P												Yu and Huang (2009)
GW	Biochar, clay							G				C				Barthod et al. (2016)
GW	Ash												C			Belyaeva and Haynes (2009)
GW	Rock P, sediment			P	P								C			Zhang et al. (2017)
GW	Jaggery, fly ash			P									C			Gabhane et al. (2012)
GW	Biochar, spent mushroom			P												Zhang and Sun (2014)

Table 1 (continued)

Waste	Additives	Improvement of process parameters						Decrease of gas emissions				Compost quality			Soil health	References
		Temperature (°C)	Duration	Microbe activity	Moisture	Aeration	pH	CO₂	CH₄	GHG	Odour (N, S)	C seq.	N, P	Heavy metals		
MSW	Bagasse, paper, peanut shell, sawdust						P									Iqbal et al. (2010)
MSW	Rice straw					P					G					Shao et al. (2014)
MSW	Zeolite										G					Turan and Ergun (2007)
MSW	Zeolite										G					Wang et al. (2014)
MSW	Plastic tubes, woodchips					P				G						Maulini-Duran et al. (2014)
MSW	Inoculum											C				Wei et al. (2007)
SWS	Zeolite, lime									G						Awasthi et al. (2016a, b)
SWS	Charcoal										G			C		Hua et al. (2009)
SWS	Woodchips, rice husk				P											Morisaki et al. (1989); Eftoda and McCartney (2004)
SWS	Rice husk			P												Nakasaki et al. (1986)
SWS	Fly ash, red mud, lime						P									Wong et al. (1997); Fang et al. (1999); Wong et al. (2009)
SWS	Lime													C		Wong and Selvam (2009)
SWS	Lime						P							C		Fang and Wong (1999)
SWS	Zeolite													C		Nissen et al. (2000); Zorpas et al. (2000); Villasenor et al. (2011)
SWS	Red mud, fly ash													C		Qiao and Ho (1997); He et al. (2016); Wang et al. (2013a)
Rice straw	Red mud													C	S	Zhou et al. (2017)

P = improvement of compost process parameters; G = decrease of gas emissions; C = improvement of compost quality; S = improvement of soil health

C seq. C sequestration, FW food waste, GW green waste, MSW municipal solid waste, SWS sewage sludge

et al. 2017; Barthod et al. 2016). As the production of biochar was shown to yield highly aromatic materials with high stability when added to soil, biochar may enhance the carbon sequestration potential of composts and vermicomposts (Lehmann et al. 2006; Lehmann 2007), thus mitigating climate change. A great variety of biochars is produced, with different properties depending on the initial feedstock and pyrolysis temperature (Zhao et al. 2013). Generally higher temperatures lead to greater aromatisation of the material, decreasing its surface area, cation exchange capacity and content of volatile compounds (Lehmann et al. 2006).

The main inorganic or mineral additives are lime, clays or industrial wastes, for example red mud or fly ash. Red mud is a by-product of the industrial alumina production (Wang et al. 2008), and fly ash is a waste product of clean coal combustion used to mitigate gas emissions (e.g. used in power plants). Therefore, the main advantages of these alkaline materials are their high availability (Gomes et al. 2016) and their low cost as industrial wastes.

Minerals such as zeolite or Ca-bentonite became popular in the last decades due to their physical and chemical properties to adsorb heavy metals, in particular in soils and water purification systems (Lopez et al. 1998; Wang et al. 2016). Pure zeolite is easily synthesised by the slow crystallisation of a silica-alumina matrix. Finally, pure minerals such as clay, extracted from quarries or soils, are increasingly used with the aim of reducing greenhouse gas emissions during composting (Bolan et al. 2012; Barthod et al. 2016). These minerals are sometimes added during vermicomposting (Li et al. 2009; Hayawin et al. 2014; Barthod et al. 2016).

3 Effects of additives on the composting process

During composting, the naturally developing microbial community can be affected by additives, directly, when adding an inoculum, or indirectly, when adding organic or mineral materials inducing changes in the aeration, temperature, moisture content, pH, nutrient availability, etc.

3.1 Effects of additives on composting temperature profiles

The temperature profile is a good indicator of the microbial activity during composting (Haug 1993) and important to assess the removal of pathogens through high temperature (Gea et al. 2007). It can change due to the presence of additives. Mineral, organic as well as biological additives

have been shown to stimulate the microbial activity, leading to an earlier start and a longer duration of the thermophilic phase as compared to regular composting (Fang and Wong 1999; Chen et al. 2010; Doublet et al. 2011; Himanen and Hänninen 2011; Gabhane et al. 2012; Jiang et al. 2015a; Morales et al. 2016). The duration of the thermophilic phase increased from 2 to 3 weeks following the addition of commercial products containing zeolite, kaolinite, chalk, ashes and sulfates (Himanen and Hänninen 2009) or biochar addition during biowaste and food waste composting, thereby shortening the composting process. For example, compost stability was already achieved after 50–60 days when amended with biochar (Waqas et al. 2017). Rapid temperature increases were observed following the addition of biochar as well as mineral and polymer additives (zeolite, jaggery and polyethylene glycol) during composting of animal manure, food waste and green waste (Chen et al. 2010; Czekała et al. 2016; Waqas et al. 2017; Venglovsky et al. 2005; Gabhane et al. 2012). Similar temperature profiles were recorded for compost with biological or organic additives (Manu et al. 2017; Nakasaki and Hirai 2017). The more rapid temperature increase with these additives may be due to the increase of microbial biomass and activity. However, some additives such as bentonite (Li et al. 2012), phosphogypsum and lime (Gabhane et al. 2012) do not change the temperature profile, suggesting that they do not affect microbial biomass.

3.2 Additives to stimulate microbial activity

The additives influence the microbial communities of composts through their effect on temperature, moisture and aeration of the pile (Fuchs 2009), which is due to their contents of nutrients and readily available forms of carbon. For example, adding jaggery increased the number of microorganisms and thus enhanced the enzymatic degradation of cellulose during composting of green wastes (Gabhane et al. 2012). Similar results were obtained with the addition of fish pond sediments, spent mushroom substrate and biochar (Zhang and Sun 2014; Zhang et al. 2017). The effect of biochars on microbial activity was probably due to their effect on microbial habitat and protection from grazers (Meng et al. 2013; Wei et al. 2014). Biochar, due to its porous structure, might also enhance microbial activity through moisture and aeration control (Waqas et al. 2017), which has an effect on compost temperature as discussed in the previous section. However, a biochar application rate higher than 20% is not recommended since it may hinder organic matter biodegradation (Liu et al. 2017; Xiao et al. 2017). Alkaline substrates such as fly ash and lime are also additives with high nutrient contents; however, they do not favour microbial activity

due to their high pH (Fang et al. 1999). At high rates, fly ash addition inhibited phosphatase, β-glucosidase or dehydrogenase but did not affect urease activity, indicating that some stages of N-containing compound degradation were unchanged (Wong et al. 1997).

While mineral additives can change the microbial activity indirectly, microbial consortium addition directly changes the microbial community and activity of the compost. A commercial microbial additive changed the temperature profile of the composting process and the ammonia emissions due to the increase in the mesophilic and thermophilic bacteria compared to the regular composting without additives (Sasaki et al. 2006). The non-dominant microbes of commercial consortium influenced more the compost microbial composition than the dominant ones (Wakase et al. 2008). Isolation of microbial communities prepared by incubation mixtures of cow manure, soil and straw allowed for the selection of microbial colonies with specific degradation functions (Liu et al. 2011). These preparations also increased the duration of the thermophilic phase and were thus able to enhance the microbial diversity of compost (Liu et al. 2011).

Moreover, as discussed above, zeolite, biochar and polyethylene glycol increase microbial activity at doses mostly lower than 5%. Their addition may thus be a good way to reduce the length of the composting process.

3.3 Additives to improve aeration

An optimal aeration during composting is required as explained by Gao et al. (2010). A low aeration rate might lead to anaerobic conditions, while a high aeration rate might result in excessive cooling, thus preventing thermophilic conditions. The common way to enhance aeration and thus favour the microbial activity during composting is through mechanically turning the composted material (Chowdhury et al. 2014; Chen et al. 2015; Manu et al. 2017) and forced aeration through pipes (Ogunwande and Osunade 2011). These actions are not necessary in the case of vermicomposting due to worm activity. The costs of the pile turning or forced aeration may be reduced by the use of bulking agents, such as biochar, residual straws, woodchips or sawdust and crushed branches, since they enhance the natural aeration and porosity of the composting pile (Kulcu and Yaldiz 2007; Czekała et al. 2016). Due to their numerous pores and low moisture content, these bulking agents support the formation of inter- and intraparticle voids (Iqbal et al. 2010). Biochar addition can also improve compost aeration due to its porous structure (Waqas et al. 2017). Moreover, the

biological oxidation rate is directly related to the surface area exposed to the microbial attack (de Bertoldi et al. 1983).

3.4 Additives to regulate moisture content

The moisture content during composting influences the oxygen uptake rate and thus the microbial activity and the degradation rate. An optimal moisture content hence decreases the compost maturation time (Iqbal et al. 2015 b). The optimal water content for organic matter biodegradation has been estimated within 50–70% (moisture content in wet basis: ratio of the weight of water to the weight of the material) (Richard et al. 2002). However, some organic wastes have higher or lower moisture content; for example, sewage sludge has a moisture content ranging from 80 to 90%. Such high humidity may favour anaerobic conditions and thus odour production during composting (Jolanun et al. 2008). The bulking agents commonly chosen to offset the high moisture content of organic wastes are fibrous materials (Miner et al. 2001, Eftoda and McCartney 2004, Doublet et al. 2011), which can absorb part of the leachate. Leachate absorption may be achieved by cornstalk, sawdust or spent mushroom substrate (Yang et al. 2013). Furthermore, Chang and Chen (2010) showed that increasing water absorption capacity by sawdust addition also resulted in higher degradation rate due to more air flow through the particles.

On the contrary, water loss during the first days of composting might delay the composting process and necessitates water sprinkling. The addition of materials with high water retention properties, such as clays, might limit water losses. Li et al. (2012) showed that the initial moisture decrease was buffered by the presence of bentonite, due to its swelling performance. During composting of green wastes, the water holding capacity was also increased by the addition of ash (Belyaeva and Haynes 2009) or phosphate rock (Zhang and Sun 2017). However, some additives such as eggshells had no influence on water holding capacity (Soares et al. 2017) and may even have a negative effect on the biological activity.

3.5 Additives to buffer pH

The pH varies during composting with a decrease during the early stages and an increase during the later stages (Onwosi et al. 2017), which affects microbial activities. Some additives are used to increase the pH and thus enhance the composting of acid feedstocks, such as food waste (Wong et al. 2009). The addition of an inoculum consortium was reported to result in a pH increase from

4.3 to 6.3 during food waste composting (Manu et al. 2017). This may be explained by the degradation of acids along with organic matter through enhanced biological activity. The use of bulking agents such as bagasse, paper, peanut shell, sawdust and Ca-bentonite might also increase the pH during composting (Iqbal et al. 2010; Wang et al. 2016), similar to fly ash, lime or red mud addition (Fang and Wong 1999; Gabhane et al. 2012; An et al. 2012; Chowdhury et al. 2016). However, such alkaline additives might inhibit the metabolic activity. For example, Wong et al. (1997) observed less thermophilic bacteria in the initial phase, when sludge was co-composted with 25% fly ash.

The increase of pH during the thermophilic phase was offset by using bamboo charcoal or zeolite, which was explained by the capacity of these additives to adsorb the ammonia produced by the mineralisation of organic nitrogen (Chen et al. 2010; Venglovsky et al. 2005). A lower pH may also decrease nitrogen loss by avoiding ammonia volatilisation, which occurs at high pH (Chen et al. 2010). Finally, the addition of elemental sulphur considerably decreased the pH during poultry manure composting (Mahimairaja et al. 1994), mainly due to the oxidation of elemental sulphur producing H_2SO_4, thus increasing the H^+ ion concentration.

4 Influence of additives on gas emissions during composting

4.1 Odour emissions

Sulphur-containing compounds and NH_3 are the main odorous gases emitted during composting. Emission of NH_3 not only causes environmental pollution and odorous nuisances but also decreases the agronomic value of the compost (Ermolaev et al. 2015). We found no reference in the literature on vermicomposting and odour emissions, likely due to the fact that worm activities allow a continuous aeration of the composting pile, which limits the formation of anaerobic zones responsible for odour emissions. By improving oxygen transfer inside the composting pile of municipal wastes through the addition of rice straw (1:5), Shao et al. (2014) decreased the cumulative malodorous sulphur-containing gases. Similar results have been obtained after ash (Koivula et al. 2004) and biochar (Steiner et al. 2010; Hua et al. 2009; Khan et al. 2014) addition.

Another way to reduce the odour emissions is to trap the nitrogen excess and thus reduce the ammonia contents in the composting pile. Natural zeolite may be added for this purpose (Lefcourt and Meisinger 2001; Turan 2008; Wang et al. 2014; Awasthi et al. 2016b), the odour reduction being proportional to the amount of zeolite added. Zeolite addition on top of the pile may reduce ammonia emissions (Witter and Lopez-Real 1988; Bernal et al. 1993). Biochar was found to decrease ammonia losses when N-rich wastes were composted (Steiner et al. 2010). However, additives may also increase ammonia losses due to the enhancement of microbial activities (Waqas et al. 2017; Jiang et al. 2015b). By contrast, organic additives such as sawdust, cornstalk and spent mushroom substrates did not influence ammonia emissions (Yang et al. 2013; Yuan et al. 2015).

Chemical additives (a nitrogen electron acceptor, sodium nitrate and sodium nitrite or $FeCl_3$) may be used to control odours related to sulphur-containing compounds by reducing the emissions of dimethyl sulphide and dimethyl disulphide, without altering the composting process (Yuan et al. 2015; Zang et al. 2017). A number of chemical additives may be used to reduce ammonia losses during composting, for example absorbent mixtures of magnesium hydroxide and phosphoric acid (Jeong and Hwang 2005; Ren et al. 2010), calcium superphosphate (Zhang et al. 2017) or other phosphate and magnesium salts (Jiang et al. 2016). Some specific additives such as dicyandiamide were used to inhibit the nitrification process in order to decrease the ammonia emissions (Jiang et al. 2016).

4.2 GHG emissions

The amounts of GHG released during composting of different initial feedstock with different additives are shown in Table 2. GHG emissions from co-composting processes have been largely studied, but few data are available for vermicomposting processes and even less for co-vermicomposting. Moreover, among GHG gases, only CH_4 and N_2O emissions during composting are taken into consideration in national GHG inventories, while CO_2 emissions of biogenic origin are not accounted for (IPCC 2014). The amount of N_2O and CH_4 emitted during the composting process may depend on the feedstock and on the aeration conditions in the compost pile. Aeration in the compost pile may be enhanced by the addition of bulking agents or by mechanical turning. The proportion of added bulking agent and the turning frequency of the composting pile are thus important process parameters, which need to be controlled for regulating GHG emissions, in particular gaseous N losses (Szanto et al. 2007; Maulini-Duran et al. 2014; Morales et al. 2016).

N losses in the form of N_2O can occur under aerobic conditions due to incomplete nitrification/denitrification or anaerobic conditions, when a lack of O_2 leads to nitrate accumulation. Additives affect differently the N_2O emissions according to the initial feedstock and the type of additive. For instance, mineral additives such as phosphogypsum (Hao et al. 2005; Luo et al. 2013) significantly reduced N_2O emissions during manure composting, probably by increasing SO_4^{2-} content of

Table 2 Gaseous emissions during composting with different feedstocks associated with additives

Initial feedstock	Process	Bulking agent/additives	Duration (days)	Gaseous emissions (reported on 100 days)			References
				CO_2 (g kg^{-1})	N_2O (g kg^{-1})	CH_4 (g kg^{-1})	
Straw-bedded cattle feedlot manure	Composting	Cereal straw	99	166.67	9.01	0.078	Hao et al. (2001)
Straw-bedded cattle feedlot manure	Composting	Woodchips	99	147.07	9.02	0.085	
Sewage sludge	Composting	Cotton waste	105	239.77	–	–	Bernal et al. (1998)
Poultry manure, olive mill wastewater	Composting	Cotton waste	105	200.89	–	–	
Sewage sludge, olive mill wastewater	Composting	Cotton waste	140	196.62	–	–	
Sewage sludge	Composting	Maize straw	112	269.36	–	–	
Sewage sludge, olive mill wastewater	Composting	Maize straw	119	260.19	–	–	
Pig slurry, poultry manure	Composting	Sweet sorghum bagasse	112	210.86	–	–	
City refuse	Composting	Sweet sorghum bagasse	133	179.31	–	–	
Duck manure	Pre-composting + vermicomposting	Reed straw	77	400.91	0.12	<0.01	Chowdhury et al. (2014)
Duck manure	Pre-composting + vermicomposting	Reed straw and zeolite	77	405.84	0.14	<0.01	Chowdhury et al. (2014)
Cattle manure	Composting	Barley straw	31	300	0.08	1.19	Chowdhury et al. (2014)
Cattle manure	Composting	Barley straw	31	274.19	0.08	2.41	
Hen manure	Composting	Barley straw	31	254.84	0.09	0.42	Chowdhury et al. (2014)
Hen manure	Composting	Barley straw	31	225.81	0.10	0.41	
Hen manure	Composting	Biochar + barley straw	31	200.00	0.08	0.09	
Hen manure	Composting	Biochar + barley straw	31	174.19	0.08	0.07	
Animal manure + food waste (75:25)	Composting	Plastic tubes (3:1)	28	225.71	0.02	0.08	Chowdhury et al. (2014)
Animal manure + food waste (75:25)	Composting	Plastic tubes (6:1)	28	233.21	0.02	0.13	
Animal manure + food waste (75:25)	Composting	Woodchips (3:1)	28	171.43	0.03	0.12	
Animal manure + food waste (75:25)	Composting	Woodchips (6:1)	28	178.57	0.05	0.14	
Animal manure + food waste (75:25)	Composting	Biochar (3:1)	28	147.14	0.01	0.02	
Animal manure + food waste (75:25)	Composting	Biochar (6:1)	28	162.14	0.01	0.04	
Animal manure + food waste (75:25)	Composting	Barley straw (3:1)	28	179.64	0.01	0.08	
Animal manure + food waste (75:25)	Composting	Lupin residues (3:1)	28	218.57	0.01	0.68	
Kitchen waste	Composting	Sawdust	28	–	0.78	0.15	Yang et al. (2013)
Kitchen waste	Composting	Cornstalk	28	–	0.40	–	
Kitchen waste	Composting	Spent mushroom substrate	28	–	1.04	0.05	
Municipal wastes	Composting	Woodchips	24	–	<0.01	0.01	Maulini-Duran et al. (2014)
Municipal wastes	Composting	Polyethylene tubes	24	–	<0.01	<0.01	
Pig manure	Composting	Mature compost + cornstalk	30	–	0.69	2.22	Luo et al. (2014)

the compost or by altering the nitrification process. No effect was observed after the addition of woodchips and polyethylene tubes as bulking agents during municipal waste composting (Maulini-Duran et al. 2014), while sawdust addition significantly reduced N_2O emissions during kitchen waste composting (Yang et al. 2013).

In a composting pile, CH_4 emissions generally occur due to a lack of oxygen as a result of excessive moisture or presence of anaerobic zones (Amlinger et al. 2008). Contrary to N_2O emissions, CH_4 emissions appear to be more dependent on the properties of the additives than those of the initial feedstock. Thus, to reduce CH_4 emissions, two types of additives might be applied, affecting directly or indirectly the carbon cycle: (1) biological additives in the form of methanotrophic bacteria (Luo et al. 2014) to enhance CH_4 oxidation and thus reduce CH_4 emissions and (2) organic additives as bulking agents to limit anaerobic zones (Yang et al. 2013; Maulini-Duran et al. 2014) or to trap emitted gases (Awasthi et al. 2016a). Physical properties of additives such as particle size may impact their efficiency for reducing CH_4 emissions (Yang et al. 2013). For example, compact zones leading to even increased CH_4 emissions may be formed after the addition of some specific organic bulking agents, such as sawdust and spent mushroom (Maulini-Duran et al. 2014), while small biochar particles may trap more gas than big ones due to increased surface area.

CO_2 emissions may be impacted by the type of additive, in particular the decomposability of organic additives. These emissions are generally increased by the addition of readily decomposable organic materials (paper, straw, peat materials, etc.) (Mahimairaja et al. 1994) and decreased by the addition of organic materials rich in lignin or other slowly degrading organic compounds (Mahimairaja et al. 1994; Chowdhury et al. 2014). Biochar addition during composting has contradictory effects on CO_2 emissions compared to regular compost, either increasing (Czekała et al. 2016; Wu et al. 2017) or decreasing them (Chowdhury et al. 2014). Such contradictory effects were also observed for vermicomposting (Barthod et al. 2016). Organic additives in the form of bulking materials, such as plastic or pumice, increase CO_2 emissions through aeration improvement and microbial activity enhancement (Wu et al. 2015; Czekała et al. 2016). In conclusion, reducing CO_2 emissions from composting without altering the biodegradation process appears difficult. Chemical and mineral additives have the potential to limit CO_2 emissions, for example by trapping emitted CO_2 with an adsorbent, such as red mud, or by protecting carbon from decomposition through associations with clays or amorphous hydroxyl-Al (Bolan et al. 2012; Haynes and Zhou 2015; Barthod et al. 2016).

In conclusion, the effect of additives on the composting process and on gas emissions might be contradictory. The additives must be chosen according to their possible positive and negative effects (Fig. 3), in view of developing amendments with specific properties.

5 Quality of co-compost for use as potting media or soil amendments: influence of additives on nutrient availability, metal availability and soil fertility

Soil health and functions, such as fertility, can be improved by compost and vermicompost amendment, through their effects on soil structure and porosity, soil microbial biomass and soil nutrient contents (Thangarajan et al. 2013). In addition to soil application, composts and vermicomposts are largely used as horticultural potting media, because they may replace peat, which is a limited resource (Hashemimajd et al. 2004; Zaller 2007). The presence of heavy metals or high content of nutrients in composts might cause important adverse effects on animal and human health (Senesil et al. 1999) and on plant growth. Therefore, the nutrient and heavy metal contents as well as their respective availability and mobility in the compost are important parameters to control. Additives can be used to improve the agronomic value of composts related to their nutrient availability or metal mobility.

5.1 Reduction of environmental hazards due to heavy metals

Depending on the initial feedstock, heavy metals might be present in the composted material (Swati and Hait 2017). Heavy metal contents are high in composts originating from animal manure or sewage sludge. During composting and vermicomposting, heavy metals can react with organic matter and their speciation might change. For instance, manure composts may contain high levels of Zn, Cd, Pb and Cu and their application on soils may lead to an excessive input of heavy metals (Chen et al. 2010). Using additives during composting is a way to reduce heavy metal availability, resulting in a marketable, safe material. Heavy metal mobility, in particular of Cu, Zn, Cd, Pb and Mn in water-soluble form, can be reduced by organic and mineral additives. Organic additives like bamboo charcoal, chestnut/leaf litter and biochar were used for this purpose (Chen et al. 2010 Guerra-Rodríguez et al. 2006; Lau et al. 2001). Mineral additives favorable to reduce heavy metal mobility are fly ash, Ca-bentonite, phosphate rock, lime or zeolite (Wang et al. 2016; Lu et al. 2014; Fang and Wong 1999; Wong et al. 2009; Nissen et al. 2000; Zorpas et al. 2000; Villasenor et al. 2011). Mixtures of organic and mineral additives in the form of lime and biochar were also effective for reducing heavy metal mobility in composts (Awasthi et al. 2016a). When using mineral additives during composting, the decrease of the labile fractions of heavy metals in manure or sewage sludge is mainly due to the

Fig. 3 The potential negative and positive effects of mineral, organic and biological additives. A trade-off must be found between these effects when choosing an additive
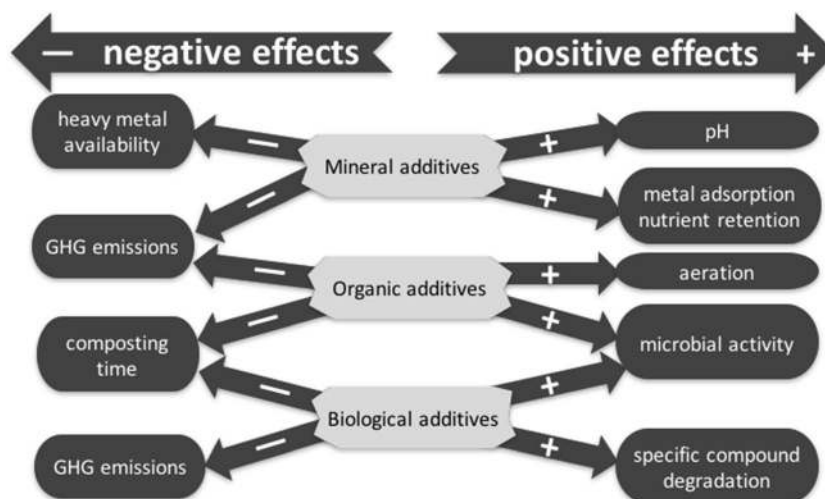
formation of organo-metal complexes between additives and metals (Wang et al. 2013b) or a complexation of the metal ions (Lu et al. 2014). Lime and biochar limit heavy metal mobility also due to a pH effect.

Co-vermicomposting may be used to decrease heavy metal hazards, since worms act as bio-accumulators of heavy metals (He et al. 2016) or may change the interaction of heavy metals with organic matter (Li et al. 2009; Swati and Hait 2017). Organic additives, such as biochar, straw and sawdust, decreased the bioavailability of heavy metals to worms during vermicomposting (Malińska et al. 2017) and reduced the concentration of heavy metals in vermicomposts (He et al. 2016). During sewage sludge composting, worms improved the efficiency of additives such as fly ash or sawdust by increasing the exchangeable fraction of essential nutrients such as Zn and Cu and decreasing the soluble forms of non-essential elements such as Cr and Pb (Wang et al. 2013a). Mineral additives were found to decrease heavy metal content and availability in co-vermicomposts (Hayawin et al. 2014; He et al. 2016).

The application of co-compost to soil may also reduce soil environmental hazards related to heavy metal-polluted soils. Zhou et al. (2017) recommended the application of co-composts (obtained with red mud) on polluted soils to decrease the ecological risk of heavy metals by efficiently reducing soil heavy metal availability and increasing soil microbial biomass.

5.2 Influence of additives on nutrient contents and availability

The major possible negative effect of additives on composts or vermicomposts is a decrease of their nutrient content due to dilution (Banegas et al. 2007) or a decrease of nutrient availability due to adsorption on mineral additives. In some cases, the adsorption of excess nutrients from compost is beneficial

to avoid environmental risks, such as eutrophication, after compost application to soil.

Mineral additives, such as coal fly ash, zeolite and alum, reduce compost-soluble P contents compared to regular composting of sewage sludge, due to the formation of insoluble phosphate compounds (Menon et al. 1993; Fang et al. 1999) or adsorption and formation of insoluble Al-phosphate intermediates (Lefcourt and Meisinger 2001; Belyaeva and Haynes 2012). Likewise, zeolite addition may retain soluble N (NH_4^+) (Li et al. 2008) and lead to a significant decrease in the C/N ratio and an increase in total nitrogen, total available phosphorous and total potassium (Hayawin et al. 2014).

Inversely, some mineral additives, such as phosphate rock (Nishanth and Biswas 2008; Billah and Bano 2015), lime and fly ash (Gabhane et al. 2012; Zhang et al. 2017) or medical stone (Wang et al. 2017), are used to increase nutrient contents of final composts made from nutrient-poor feedstocks. Nitrogen content increased in the presence of 2.5% bentonite during composting (Li et al. 2012) or with both bacteria (N_2-fixing and P-solubilising) and phosphate rock during vermicomposting (Busato et al. 2012; Unuofin and Mnkeni 2014).

The use of organic additives in the form of branches, pallets and cornstalks during sludge composting had a positive effect on N availability in the final composts by enhancing the N organisation and limiting N losses by volatilisation during composting (Doublet et al. 2011). The effect of biochar on nutrient leaching from compost is contradictory (Chen et al. 2010; Iqbal et al. 2015a; Wu et al. 2017). Co-composting of food waste with biochar increased concentrations of NH_4^+ and NO_3^- (Waqas et al. 2017).

Worms may increase the bioavailability of N and P during vermicomposting with mineral additives. Unuofin and Mnkeni (2014) recommended to use a high worm density, up to 22.5 g kg^{-1}, in order to maximise phosphorus availability in co-vermicompost with phosphate rock.

Co-composts used as potting media or soil amendment can improve fertility through two mechanisms: (1) because they are rich in nutrients when they are made with additives rich in available nutrients and/or (2) because they increase microbial biomass, which plays a pivotal role in nutrient cycling. In this way, soil application of co-composts obtained with phosphate rock and bacteria inoculum resulted in an increase of not only the soil microbial biomass but also the microbial biomass phosphorus, inducing high amounts of available phosphorus (Meena and Biswas 2014).

However, these effects are highly dependent on pedoclimatic conditions, which influence the availability of nutrients, such as P or N, and their leaching. For instance, in acidic soils, P ions may precipitate, thereby decreasing P availability (Hinsinger 2001), while nitrogen contents decrease when the pH falls below 5 (Lucas and Davis 1961). Thus, to improve the fertility of acid tropical soils, it was recommended to apply a co-compost of wood ash with high pH to retain Al and increase the bio-availability of exchangeable cations (Bougnom et al. 2009). The application of co-composts (rate and additives) should be optimised and adapted to the soil properties to improve soil health and avoid environmental risks (Roca-Pérez et al. 2009).

6 Environmental perspectives and feasibility of co-composting

6.1 Co-compost: a potential stimulator of plant growth

Few studies assessed the impact of co-composts on plant growth, and even less took place under field conditions (Chowdhury et al. 2016; Kuba et al. 2008). Moreover, to the best of our knowledge, none assessed the effect of co-vermicomposts on plants. The effect of co-composts on plants is generally tested by assessing the germination index, a biological indicator used to evaluate the toxicity and maturity of compost (Zucconi et al. 1981). A germination index higher than 50% indicates a phytotoxin-free compost. The presence of additives may influence this germination index, since additives have an effect on nutrient availability (see above). Chen et al. (2010) showed that the addition of bamboo charcoal and bamboo vinegar during pig manure composting increased this index up to 95%, suggesting that charcoal can be considered as a plant growth stimulator, while several studies concluded that the presence of mineral additives such as bentonite and alkaline materials inhibits plant growth (Fang and Wong 1999; Samaras et al. 2008; Li et al. 2012). Moreover, the nutrient uptake of plants can be changed by co-compost application. Zayed and Abdel-Motaal (2005) showed that co-compost with biological additives enhanced the plant phosphorous uptake and, in addition, decreased the growth of

bacteria in the rhizosphere. Similarly, mineral additives can increase nutrient availability to plants. For example, phosphate rock-enriched compost supplied more phosphorous to seedling than a usual growing media (Mihreteab et al. 2015) and led to higher biomass yield, uptake and recoveries of P and K when prepared with additional waste mica (Nishanth and Biswas 2008). Moreover, the use of specific amendments able to absorb heavy metals limited metal transfers to plant (Nissen et al. 2000).

Finally, the few field experiments carried out with co-composts showed that they might improve soil fertility and thus potential revegetation of degraded soil. Kuba et al. (2008) showed that a co-compost produced with 16% wood ash was more efficient than mineral and organic fertilisers for increasing plant cover during ski slope revegetation. Similarly, Chowdhury et al. (2016) observed that co-composts (biowastes with alkaline amendment) improved soil fertility when used to revegetate an urban landfill soil (Fig. 4).

6.2 Reduction of GHG emissions and carbon sequestration potential of soil amended with co-compost

The degree of degradation of compost organic matter is an important parameter to consider for their use as soil amendment (Francou et al. 2005). A stable organic matter applied to soil emits less carbon than an instable one (Bernal et al. 1998). Stability is related to compost chemical composition (Dignac et al. 2005). Since lignocellulose, a major component of green waste, is known to be slowly degradable, slowing the composting process (Jurado et al. 2015), stable phenolic compounds and complex aromatic compounds may be used as indicators of stability (Fukushima et al. 2009, Wang et al. 2017). Wei et al. (2007) and Manu et al. (2017) suggested that higher contents of aliphatic carbon chains and carboxyl and hydroxyl groups in the compost produced with biological additives compared to regular compost point to a more stable product. They also observed decreased contributions of cellulose, hemicellulose and lignin in the co-compost compared to the regular compost, indicating that organic matter transformation was facilitated by the presence of a microbial inoculum. Mineral additives may enhance organic matter degradation during composting. For example, Zorpas and Loizidou (2008) showed that the addition of sawdust and zeolite during sewage sludge composting increased the percentage of NaOH-extractable compounds in the final products and thus the number of oxygen-containing functional groups. Finally, organic additives in the form of biochar during poultry manure composting also increased the degree of transformation of the compost (Dias et al. 2010).

Soil can retain carbon for a long time, thus avoiding its mineralisation and release in the form of CO_2 or CH_4. Soil contains about three times more carbon than the atmosphere
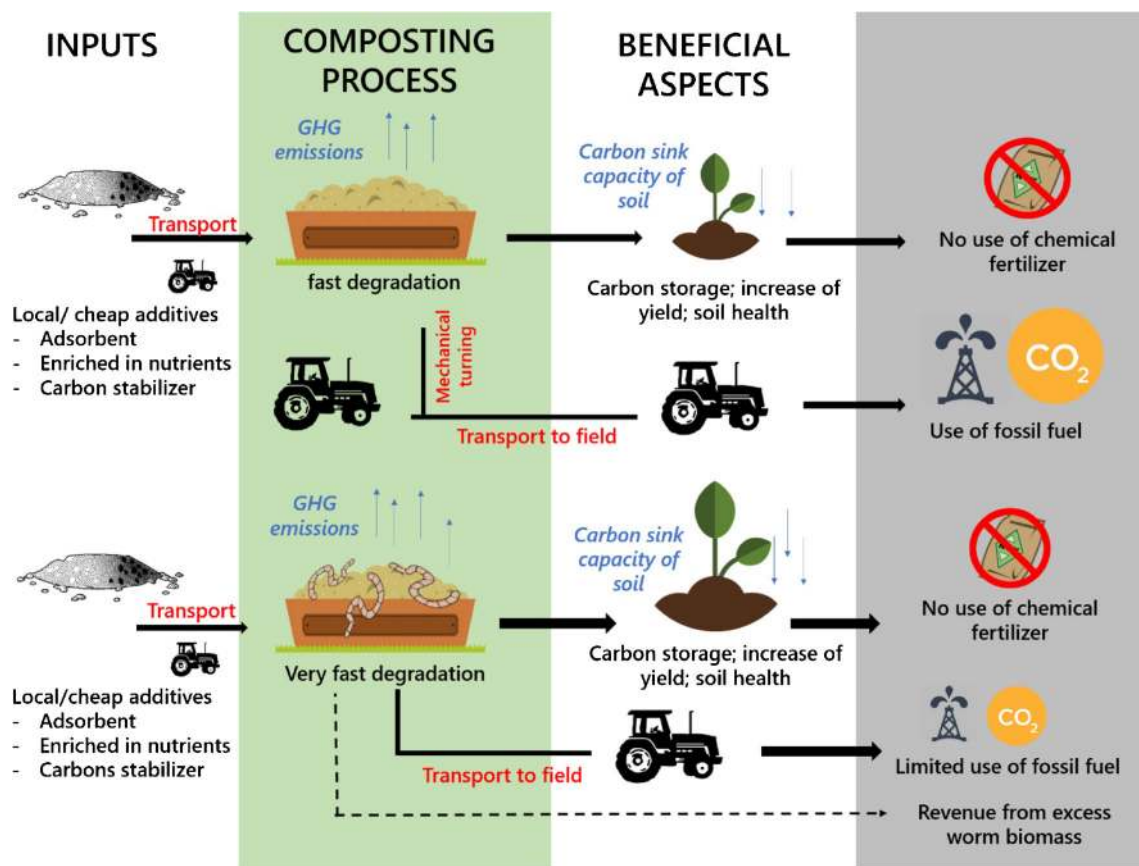
Fig. 4 Main potential environmental differences between co-composting and co-vermicomposting processes

(Eglin et al. 2010), and carbon storage in soils might be one of the solutions to mitigate climate change (IPCC 2014; Lal et al. 2004). This storage may be favoured by agricultural practices adapted to soil and climate conditions (Dignac et al. 2017). Managing organic waste materials and returning their organic C to soil may increase the soils' C sequestration potential in the sense of a circular economy (Chabbi et al. 2017). One strategy to increase soil carbon stocks is the transformation of organic waste through composting and the application of composts or vermicomposts to soils (Ngo et al. 2012). However, these organic amendments show a very fast degradation in soils due to the presence of large amounts of labile organic compounds, consumed by the soil microbial biomass, leading to increased carbon emissions. Favoino and Hogg (2008), using first-order decay kinetics, estimated that between 2 and 10% of the compost carbon applied to soil today would remain in the soil organic matter in a hundred years. However, this estimation depends largely on the type of soil (Busby et al. 2007), on the pedoclimatic conditions and on the type of compost (Peltre et al. 2012). For instance, the mineralisation process in temperate countries takes place rather slowly, while in tropical countries, the mineralisation generally leads to a rapid decline of the added compost. In Table 3, we estimated the amount of carbon potentially

remaining in soil after application of various co-composts, based on the estimation of Favoino and Hogg (2008).

Reducing the amount of labile carbon through carbon stabilisation in composts and vermicomposts recently received attention (Bolan et al. 2012; Chowdhury et al. 2014; Barthod et al. 2016). To improve carbon stabilisation in the organic amendments, the addition of minerals, such as goethite, montmorillonite or industrial wastes, such as red mud, was used. Some authors observed that the presence of these minerals during composting allowed to increase the half-life of organic amendment carbon and thus increase soil carbon stocks without altering soil properties (Bolan et al. 2012; Chowdhury et al. 2016). The addition of biochar to vermicompost changed the biodegradation of vermicompost organic matter after addition to soil and might thus increase its potential for C sequestration (Ngo et al. 2013). However, little is known about the C stabilisation mechanisms occurring during composting with these minerals and even less is known about the mechanisms occurring during vermicomposting (Barthod et al. 2016). The main hypothesis proposed in these studies is that carbon from fresh organic matter used for composting and vermicomposting is stabilised through similar mechanisms as those occurring in soils. Further studies including the assessment of GHG emissions during composting

Table 3 Estimates of the carbon potentially remaining 100 years after the application of different composts and vermicomposts to soil

Initial waste	Composting process	Additives	Compost C content (mg g^{-1})	C potentially still remaining in soil after 100 years		References
				Minimum (mg g^{-1})	Maximum (mg g^{-1})	
Sewage sludge	Composting	Shredded rice straw	227	4.5	22.7	Roca-Pérez et al. (2009)
		Non-shredded rice straw	184	3.7	18.4	
Biowaste	Composting	Sulphate, oxide iron, magnesium, manganese + zinc mixed with clay	458	9.2	45.8	Himanen and Hänninen (2009)
		Mixture of calcium hydroxide, peroxide and oxide	446	8.9	44.6	
Alperujo	Composting	Grape stalk	470	9.4	47.0	Alburquerque et al. (2006)
		Olive leaf	484	9.7	48.4	
Green waste		Degradable plastic	453	9.1	45.3	Unmar and Mohee (2008)
		Biodegradable plastic	338	6.8	33.8	
Poultry manure	Composting	Wheat straw + 5% biochar	377	7.5	37.7	Czekala et al. (2016)
		Wheat straw + 10% biochar	385	7.7	38.5	
Poultry manure	Composting	Barley waste	401	8.0	40.1	Guerra-Rodríguez et al. (2006)
		Chestnut burr/leaf litter	410	8.2	41.0	
Kitchen waste	Composting	Phosphogypsum	292	5.8	29.2	Yang et al. (2015)
		Superphosphate	280	5.6	28.0	
Biowaste	Composting	15% wood ash	229	4.6	22.9	Juarez et al. (2015)
Green waste	Composting	Fly ash	391	7.8	39.1	Gabhane et al. (2012)
		Phosphogypsum	390	7.8	39.0	
		Jaggery	424	8.5	42.4	
		Lime	411	8.2	41.1	
		Polyethylene glycol	434	8.7	43.4	
Duck manure	Vermicomposting	Reed straw	236	4.72	23.6	Wang et al. (2014)
		Reed straw + zeolite	233	4.66	23.3	

Estimates based on Favoino and Hogg (2008) are calculated as follows: $C_{min} = C_{content} \times 2\%$ and $C_{max} = C_{content} \times 10\%$

and after application to soil are necessary to finally conclude on the global C balance.

6.3 Feasibility of co-composting at large scale: economical and practical aspects of co-composting and co-vermicomposting

Composting and vermicomposting processes are both suitable to produce valuable soil amendments, but their economic potential is dependent on the initial cost, the production volume, the initial quality of the feedstock and the end product prices at a particular location (Lim et al. 2016). Lim et al. (2016) estimated that a composting facility costs about 4.37 million USD annually (mainly related to initial investment and annual operation and maintenance costs) and generates 1.10 million USD of benefits per year. Since composting is a time-consuming process (Onwosi et al. 2017), using additives to shorten composting duration can reduce the costs.

Screening and re-using additives is not cost-effective (Villasenor et al. 2011). The additives must thus be cheap and efficient for producing a high-quality end product. For instance, the composting process can be improved by jaggery and polyethylene glycol, which are expensive (Gabhane et al. 2012), but also by bentonite (Li et al. 2012) or allophane (Bolan et al. 2012), which are cheap and abundant. In addition, the optimal additive to organic waste ratio must be optimised in order to reduce the cost of composting. For instance, compared to 3:1 proportions (sawdust/sludge), a 1:1 proportion was efficient and economic (Banegas et al. 2007). However, at low sawdust proportions, the final product had a lower quality, inducing, for example, a low germination index (Huang et al. 2004). Moreover, with low sawdust proportions, a longer composting process is required compared to high sawdust proportions.

Finally, while selecting additives, several trade-offs have to be considered. Taking into account the additive cost, spatial and temporal accessibility (e.g. agricultural residues collected seasonally) and abundance, the composting process with additives must be adapted to the region, the composting facility localisation and the season. Moreover, vermicomposting also generates revenue through excess worm biomass and improved end products as compared to a regular compost (Gajalakshmi and Abbasi 2002; Ngo et al. 2011; Doan et al. 2015). Thus, we suggest that the presence of worm, combined with an additive, will significantly improve the composting process by increasing nutrient availability, stabilising carbon, shortening compost duration and decreasing GHG emissions (Komakech et al. 2015). Therefore, this process may lead to higher return on investment and lower annual cost than a regular composting with or without additives (Fig. 4). However, to account for the economic advantages and disadvantages of the different composting procedures, a full economic analysis is needed. For example, while

vermicomposting may reduce the cost of pile turning, the space required to treat a similar amount of organic wastes may be much higher than in traditional composting units, thus increasing costs. Moreover, vermicompost may not sanitise the wastes and additional composting may be required.

7 Conclusion

This review examined the effect of mineral, organic and biological additives during composting and vermicomposting procedures on the process parameters, the quality of the final products and their impact on soil ecosystem services (production of food and fibre through improved soil fertility, climate change mitigation through C storage in soil organic matter, etc.). Adding mineral, organic or biological materials during composting and vermicomposting affects composting parameters and may be used to (1) influence the key parameters of the composting process, such as aeration and porosity of the pile, and composting duration by manipulating the thermophilic period; (2) enhance the agronomic value of the final products by increasing nutrient contents and reducing metal mobility or increasing carbon stability; and (3) limit odour and GHG emissions. However, different additives have contrasting effects and costs and should therefore be used appropriately. After optimisation of co-composting processes, co-compost application to soil may improve soil health. On polluted soils, it appears relevant to use compost produced with an additive able to immobilise or reduce the availability of heavy metals, while on degraded soils, using a co-compost produced with organic or mineral additives rich in available nutrients might improve soil fertility. While regular composts are rapidly mineralised after soil application, co-compost application may increase the soil C stabilisation potential. However, negative effects of some co-composts on plant growth were also reported.

Further studies are needed to complete the information on the effects of additives:

- The use of additives during composting leads to end products with contrasting properties, impacting differently the soil properties. Thus, in a view of an ecological application, more studies are needed to establish a direct link between co-compost properties, soil parameters and effects on plant growth, according to the species in order to improve soil health while avoiding environmental risks.
- The quality and quantity of the additives should be considered in order to improve the cost-effectiveness of the system. The development of a cost-effective system based on co-composting or co-vermicomposting should be investigated. According to the available literature, vermicomposting is less expensive than composting and the final products generally show a higher quality than a

regular compost. Therefore, improvement of the environmental impact of the vermicomposting systems should be assessed after the addition of organic, mineral or biological materials.

- Ultimately, region-specific transformation processes should be designed depending on the availability of organic wastes and additives, taking into consideration the availability of organic wastes and additives as well as prevailing pedoclimatic conditions.

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