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Crop residue harvest for bioenergy production and its implications on soil functioning and plant growth: A review

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ABSTRACT: The use of crop residues as a bioenergy feedstock is considered a potential strategy to mitigate greenhouse gas (GHG) emissions. However, indiscriminate harvesting of crop residues can induce deleterious effects on soil functioning, plant growth and other ecosystem services. Here, we have summarized the information available in the literature to identify and discuss the main trade-offs and synergisms involved in crop residue management for bioenergy production. The data consistently showed that crop residue harvest and the consequent lower input of organic matter into the soil led to C storage depletions over time, reducing cycling, supply and availability of soil nutrients, directly affecting the soil biota. Although the biota regulates key functions in the soil, crop residue can also cause proliferation of some important agricultural pests. In addition, crop residues act as physical barriers that protect the soil against raindrop impact and temperature variations. Therefore, intensive crop residue harvest can cause soil structure degradation, leading to soil compaction and increased risks of erosion. With regard to GHG emissions, there is no consensus about the potential impact of management of crop residue harvest. In general, residue harvest decreases CO_2 and N_2O emissions from the decomposition process, but it has no significant effect on CH_4 emissions. Plant growth responses to soil and microclimate changes due to crop residue harvest are site and crop specific. Adoption of the best management practices can mitigate the adverse impacts of crop residue harvest. Long-term experiments within strategic production regions are essential to understand and monitor the impact of integrated agricultural systems and propose customized solutions for sustainable crop residue management in each region or landscape. Furthermore, private and public investments/cooperations are necessary for a better understanding of the potential environmental, economic and social implications of crop residue use for bioenergy production.

Keywords: bioenergy feedstock, soil quality, sugarcane, greenhouse gas emissions

Introduction

The gradual change in the energy mix, replacing fossil fuels for renewable energy sources is one of the principal ways to mitigate greenhouse gases (GHG) emissions and effects on global warming (IPCC, 2007). Thus, it is expected a significant increase of production of bioenergy in the near future to attend the growing global demand (IRENA, 2016). In this scenario, the stakeholders of this sector have increased their interest and investments to use crop residues as raw material for bioenergy production, mainly cellulosic ethanol (second-generation ethanol - E2G) and bioelectricity (Franco et al., 2013; Karlen and Johnson, 2014).

Currently, most crop residue is left in the field after harvesting and is underused for energy production. However, crop residues have an essential role to sustain and improve the chemical, physical and biological properties and soil processes, contributing to the suitable functioning of soil, plant growth and other environmental services (Blanco-Canqui and Lal, 2009; Lal, 2009; Franzluebbers, 2015; Carvalho et al., 2016; Stavi et al., 2016). Therefore, since that the use of crop residues for energy production is a promising strategy on a global scale, the trade-offs and environmental synergism associated with crop residue management cannot be overlooked. Given the complexity of this subject, most research has been carried out to understand mechanisms and isolated processes, but few studies summarize and integrate this knowledge (e.g., Blanco-Canqui and Lal, 2009; Carvalho et al., 2016), which may support researchers and decision-makers within the bioenergy sector.

Thus, this review aimed to synthesize, gather and discuss information available in the literature on potential impacts of crop residue harvest on soil functioning, GHG emissions as well as its effects on plant growth and yield in areas for bioenergy purposes.

Origin, quality and quantity of crop residues for bioenergy production

Crop residues can be defined as the part of the plant that is left in the field after harvest, varying greatly in properties and decomposition rates (Lal, 2005). Globally, it is estimated that between 2003 and 2013, the production of agricultural residues increased by 33 %,

reaching 5 billion Mg in 2013 (Table 1). The Asian continent is the largest producer of crop residues, 47 % of the total, followed by America (29 %) Europe (16 %) Africa (6 %) and Oceania (2 %) Between 2003 and 2013, the production of crop residues in Brazil increased by approximately 41 %, reaching 432 million Mg. This account for approximately 30 % of the total production of the American continent and 9 % of the global production.

With respect to energy supply, the crop residues of maize (*Zea mays* L.) and sugarcane (*Saccharum* spp. L.) emerge as the most important raw materials for the production of cellulosic ethanol and/or bioelectricity (Table 1). Other crops of global importance such as rice (*Oryza sativa* L.) and wheat (*Triticum aestivum* L.), are highlighted by the sheer volume of residues produced (44 % of the total). However, both rice and wheat play a minor role in the production of biofuels. On the other hand,

Table 1 – Estimate crop residue production* (10^6 Mg) in 2003 and 2013 for Brazil, continents and the world.

Crop	Brazil		America		Asia		Africa		Europe		Oceania		World	
	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013
Cereals														
Barley	0.5	0.4	32.0	31.6	32.1	33.7	8.7	11.0	124.8	128.7	16.1	11.8	213.8	217.1
Corn	48.3	80.5	361.8	522.6	167.6	304.3	45.5	71.6	69.6	117.4	0.5	0.7	645.1	1016.7
Millet	-	-	0.4	0.6	26.3	20.5	23.3	22.6	2.0	0.9	0.05	0.06	52.2	44.8
Oats	0.4	0.4	6.8	6.6	1.1	1.1	0.1	0.2	16.3	14.5	2.0	1.1	26.5	23.8
Rice	15.5	17.6	47.9	54.7	799.2	1012.2	27.7	43.9	4.8	5.8	0.6	1.7	880.5	1118.5
Rye	0.005	0.006	0.8	0.6	1.5	1.7	0.04	0.1	19.3	22.4	0.03	0.05	21.9	25.0
Sorghum	2.7	3.1	34.5	35.2	16.4	13.1	34.6	38.5	0.7	1.7	2.2	3.3	88.5	92.0
Wheat	9.2	8.5	170.4	176.2	367.3	478.2	32.1	42.1	230.5	338.2	39.6	34.9	840.1	1069.7
Total	76.6	110.5	654.6	828.1	1411.5	1864.8	172.0	230.0	468.0	629.6	61.0	53.6	2768.6	3607.6
Legumes														
Beans	3.3	2.9	7.3	7.0	10.0	10.6	3.0	4.8	0.4	0.5	0.05	0.5	20.9	23.1
Broad beans	0.01	0.05	0.1	0.2	2.2	1.4	1.1	0.7	0.7	0.6	0.2	0.2	4.5	3.3
Chick peas	-	-	0.2	0.5	6.2	11.0	0.3	0.5	0.08	0.09	0.1	0.8	7.1	13.1
Groundnut	0.1	0.3	2.6	3.6	24.7	29.9	8.8	11.5	0.008	0.008	0.04	0.02	36.3	45.2
Lentils	-	-	0.6	2.1	2.1	2.2	0.07	0.1	0.04	0.07	0.1	0.3	2.9	4.9
Peas	0.06	0.04	2.3	4.7	2.1	2.2	0.3	0.7	4.5	3.0	0.5	0.2	9.8	10.9
Pulses	-	-	0.02	0.03	1.4	2.1	1.1	1.2	0.8	1.2	0.006	0.5	3.5	5.2
Soybeans	51.9	81.6	162.1	240.8	25.5	27.2	1.0	2.2	1.8	5.9	0.009	0.9	190.6	276.4
Total	55.3	84.8	175.2	258.9	74.2	86.6	15.6	21.7	8.3	11.3	1.0	3.4	275.6	382.1
Oil crops														
Linseed	0.006	0.01	1.0	0.8	0.6	0.7	0.1	0.1	0.2	0.4	0.008	0.009	2.0	2.2
Rapessed	0.08	0.08	11.3	28.8	23.9	35.0	0.1	0.2	17.1	38.3	2.5	6.2	55.1	108.7
Safflower	-	-	0.3	0.2	0.3	0.3	0.01	0.01	0.001	0.0009	0.02	0.004	0.7	0.6
Seed cotton	3.3	5.1	19.7	18.7	53.4	78.8	7.0	6.4	2.0	1.5	1.3	4.0	83.7	109.5
Sesame	0.03	0.01	0.2	0.2	4.3	4.9	1.8	4.2	0.002	0.002	-	-	6.4	9.5
Sunflower	0.07	0.1	5.5	4.6	4.4	6.0	1.2	2.1	16.3	31.8	0.02	0.04	27.5	44.7
Total	3.4	5.3	38.0	53.3	86.9	125.7	10.2	13.0	35.6	72.0	3.8	10.2	175.4	275.2
Sugar crop														
Sugarbeet	-	-	8.4	8.0	7.9	9.4	1.5	3.0	40.2	41.9	-	-	58.1	62.5
Sugarcane	118.8	221.7	193.8	301.7	180.9	223.4	26.8	29.1	0.02	0.001	12.0	8.7	413.7	563.1
Total	118.8	221.7	202.2	309.7	188.8	232.8	28.3	32.1	40.2	41.9	12.0	8.7	471.8	625.6
Tubers														
Potato	0.7	0.8	10.4	10.6	31.3	45.1	3.8	7.5	32.5	28.2	0.4	0.4	78.6	92.0
Sweet potato	0.1	0.1	0.6	0.9	28.8	21.6	3.0	4.8	0.01	0.01	0.1	0.1	32.7	27.6
Total	0.8	0.9	11.0	11.5	60.1	66.7	6.8	12.3	32.5	28.2	0.5	0.5	111.3	119.6
Grand total	254.9	432.2	1081.0	1461.5	1821.5	2376.6	232.9	309.1	584.6	783.0	78.3	76.4	3330.9	5010.1

*Crop residues production was estimated by equation: R = G * r, where R is the residue production of the crop; G is grain production based on FAOSTAT data (http://faostat.fao.org/site/291/default.aspx); r is straw/grain ration of the crop, based on Lal (2005).

soybean [*Glycine max* (L.) Merr.] and cotton (*Gossipium hirsutum* L.) residues, although available at large quantities (approximately 663 million Mg), are considered non-collectable because of the small quantity produced per area and fast decomposition rates (Lal, 2005).

In Brazil, since the 2000s, the gradual elimination of sugarcane straw burning has promoted a notable increase in the quantity of residue left on the soil after harvesting (Manechini et al., 2005). This has made available a large quantity of straw that could be used for energy production. The amount of straw produced depends on factors such as sugarcane variety and edaphoclimatic conditions (Leal et al., 2013). On average, the straw represents 9 to 19 % of the sugarcane dry weight (Landell et al., 2013), varying from 7.4 to 24.3 Mg ha^{-1} (average 14 Mg ha^{-1}) in Brazil (Paes and Oliveira, 2005).

Sugarcane straw has great potential for heat production, electricity and cellulosic ethanol (Santos et al., 2012). It is comprised mainly of cellulose (40-44 %), hemicellulose (30-32 %) and lignin (22-25 %) (Gomez et al., 2010), representing one third of the energy potential of the crop (Santos et al., 2012). Pereira et al. (2015) verified that the fermentability of hydrolysates from different sugarcane residues were in order: straw > tops > bagasse, indicating the large potential of straw and tops for the production of cellulosic ethanol (Pereira et al., 2015).

Short-term governmental projections (2016-2025) indicate that Brazil may produce 10 billion liters of cellulosic ethanol, half of current production of existing plants, another 1.5 billion liters arising from the increase in sugarcane milling and finally, 3.5 billion liters from new plants to be commissioned from 2020 onward (UNCTAD, 2016). In terms of bioelectricity, the burning of sugarcane residues supplied 4 % of all the energy consumed in Brazil in 2015 (EPE, 2015), with potential to reach 18 % in 2020/21 (EPE, 2013).

The use of crop residues from maize also has high potential for bioenergy production (Mitchell et al., 2016). Although this crop is grown in various countries worldwide, maize residue management is highlighted in the United States, the world´s largest producer of ethanol from maize grains (ERS, 2016). After grain harvest, about 10 Mg ha^{-1} of dry material is left in the field (Lal, 2005), a substantial source of biomass for cellulosic ethanol production. Wilhelm et al. (2004) cited that maize residue could provide about 1.7 times more C than barley (*Hordeum vulgare* L.), oat (*Avena sativa* L.), sorghum [*Sorghum bicolor* (L.) Moench], wheat, soybean and sunflower (*Helianthus annuus* L.) residues based on production level. The carbohydrate content of the maize biomass fractions (leaves and stem from the superior third, cobs, leaves and stem from the inferior third of the plant) is quite variable. However, these variations had little or no effect on the potential for biofuel production, suggesting that the residual mass volume is the most influential factor (Mourtzinis et al., 2016).

The United States has a limited potential to increase the area planted with maize. Thus, the adoption of the best management practices and the use of plant breeding are essential to meet the national target of 61 billion L yr^{-1} of biofuel derived from crop residues by 2022 (Bonner et al., 2014). Recently, governmental estimates indicated that in 2040, from 588 to 936 million tons of dry material will be available for harvest, of which 153 to 161 million tons will be from maize (U.S. Department of Energy, 2016).

Forest crop residues from wood, paper and cellulose industry are also reported as potential raw materials for bioenergy production (Koizumi, 2015), although the productive processes are incipient in comparison to agricultural residues. For arboreal species become economically feasible, it needs to produce wood with medium to high density and easily dryable, besides biochemical composition adequate for the production of bioenergy (Long et al., 2015), as well as be mechanically harvested throughout the year. A good example of a potential raw material is the eucalyptus (*Eucalyptus* spp.), which attends the largest part of these criteria and has high potential for productivity in short rotation periods (Seebaluck and Leal, 2015).

Impacts of bioenergy crop residues on soil properties

i) Crop residues *vs* soil chemical properties

The harvest of crop residues has changed the dynamics and availability of nutrients in the soil (e.g., Trivelin et al., 2013; Khanal et al., 2014; Karlen et al., 2014, 2015; Adler et al., 2015; Tan and Liu, 2015). Moreover, crop residues are an important source of nutrients for subsequent crops (Table 2). Accordingly, residue harvest for energy purposes might increase the demand for mineral fertilizers (Trivelin et al., 2013; Karlen et al., 2015). However, the impact of this practice on soil chemical properties are dependent on the harvest rates adopted (Franco et al., 2013; Khanal et al., 2014; Karlen et al., 2015).

In the short term, the harvest of residues could significantly alter the levels of P and K, whilst, in the long term, the impacts are more relevant on the levels of N, Ca, Mg, S and micronutrients (Sawyer and Mallarino, 2007; Jeschke and Heggenstaller, 2012). However, Karlen et al. (2015) showed that most research was carried out under temperate climates and did not evaluate the secondary

Table 2 – Nutrient content in dry mass of maize and sugarcane residues.

Crop residue	N	P	K	Ca	Mg	S
			g kg^{-1}			
Maize[§]	5.9-6.5	0.5-0.7	7.9-12.8	1.8-2.2	1.4-1.8	0.4-0.7
Sugarcane[†]	4.4-12.3	0.8-1.3	3.1-17.2	3.3-13.1	1.1-6.0	0.7-4.0

[§]Karlen et al. (2015), Mourtzinis et al. (2016); [†]Fortes et al. (2012), Trivelin et al. (2013), Franco et al. (2013).

macronutrients and micronutrients, due to the increase in the cost of analyses and the crop removes only low quantities of these elements not changing the fertilization programs. In contrast, the evaluation of these nutrients might be important in tropical regions, which have highly weathered, nutrient poor and naturally acidic soils.

Crop residues with high C:N ratios may induce soil N immobilization by microorganisms and can even increase the short-term demand for N fertilizers (Trivelin et al., 2013; Karlen et al., 2014; Ferreira et al., 2016; Meier and Thorburn, 2016). Ferreira et al. (2016) verified an average N recovery from sugarcane straw after three crop cycles of 7.6 kg ha^{-1} or 16 % of the initial N content in straw, representing a limited contribution to crop nutrition (2 % of total N needs) in the short term. However, long-time maintenance of sugarcane straw promotes a gradual increase in the soil N, reducing N fertilization rates in sugarcane crops (Robertson and Thorburn, 2007; Trivelin et al., 2013; Costa et al., 2014; Ferreira et al., 2016; Meier and Thorburn, 2016). On the other hand, Khanal et al. (2014) reported a reduction of N in the soil due to maize stover harvesting in Iowa, the United States. Therefore, from an environmental protection perspective, partial harvest of maize residues in regions with high N content in the soil could reduce the potentially leachable N and consequently the pollution of water bodies (Khanal et al., 2014).

Short-term reduction in soil P availability (Sawyer and Mallarino, 2007; Jeschke and Heggenstaller, 2012) after crop residue harvest is related to: (i) less P released during the mineralization process, and (ii) the lower adsorption of P to the soil minerals induced by increased soil organic matter (SOM) in systems with high C input via crop residues (Reddy et al., 2014). The quantity of P removed in the maize grains is five-fold higher than the P content in the residue (Avila-Segura et al., 2011). However, the quantity of this element in the residue should not be neglected, since 0.76 kg ha^{-1} of P is removed from the field per ton of maize stover (Karlen et al., 2014). Sugarcane straw contains about 0.41 kg ha^{-1} P per ton of straw (Trivelin et al., 2013), which represents 40 % of the P uptaken by the crop (Oliveira et al., 2010).

In plants, ionic K is not part of any organic structure and is released rapidly from the plant residue (Calonego et al., 2012). Crop residues present high concentrations of K (Table 1) thus the maintenance of this residue contributes to soil K accumulation (Sawyer and Mallarino, 2007; Calonego et al., 2012). Maize residues contain 73 % of the total K extracted by the plant (Karlen et al., 2015) and the exportation of this nutrient could reach 62 kg ha^{-1} (Karlen et al., 2014). In sugarcane, 42 % of the total K extracted by the plant is in the straw (Oliveira et al., 2010) and the removal of K is approximately 80 kg ha^{-1} (Franco et al., 2013).

Crop residue harvesting negatively affects the cycling, storage and availability of nutrients in the soil, whether in the short or the long-term (Tan and Liu, 2015). Thus, the correct replacement of the nutrient removed

in crop residues should be perfomed in order to ensure sustainability to the system (Karlen et al., 2011a; Adler et al., 2015). Furthermore, some management strategies of residue harvesting could minimize nutrient removal. Several studies have shown that the concentration of nutrients vary depending on the part of the plant analyzed (e.g., Avila-Segura et al., 2011; Franco et al., 2013; Trivelin et al., 2013; Karlen et al., 2015). Therefore, whenever operationally possible, it should be collected only the parts of the plant where nutrient concentration is the lowest (e.g., dry lower leaves of sugarcane plants, dry cobs and/ or leaves from the lowest parts of the maize plants) could minimize nutrient removal due to crop residue harvesting for bioenergy production.

ii) Crop residues *vs* soil physical properties

The maintenance of crop residues in the field is a key factor for the proper functioning of many soil physical and hydraulic processes. Most studies have shown that crop residue harvest is associated with soil structural degradation (Wilhelm et al., 2004; Blanco-Canqui and Lal, 2009; Blanco-Canqui, 2013; Tormena et al., 2016), mainly because of lower C inputs into the soil (Karlen et al., 2011b; Tormena et al., 2016), the absence of mechanical protection that disperses the pressure caused by machine traffic (Braida et al., 2006) and additional wheel traffic over the field to collect and remove crop residue (Wilhelm et al., 2004). Structural degradation of soils induced by crop residue harvest is related to decreased macro-aggregation and aggregate stability (Wegner et al., 2015; Johnson et al., 2016), which consequently lead to soil compaction (Wilhelm et al., 2004; Blanco-Canqui and Lal, 2007; Roque et al., 2010). Soil compaction decreases macroporosity (Osborne et al., 2014) and increases mechanical resistance to root penetration (Rosim et al., 2012; Tormena et al., 2017), which decreases water availability to plants (Tormena et al., 2017).

Crop residues that cover the soil act as physical barriers, making it less susceptible to the erosive action of raindrops and wind (Hammerbeck et al., 2012; Franzluebbers, 2015; Johnson et al., 2016). Moreover, the maintenance of crop residues favors infiltration (Blanco-Canqui et al., 2007; Johnson et al., 2016; Valim et al., 2016) and storage of water in the soil (Peres et al., 2010; Xiukang et al., 2015; Tormena et al., 2017). Peres et al. (2010) observed that the presence of 15 Mg ha^{-1} of sugarcane straw covering the soil reduced water loss (0-20 cm layer) by approximately half when compared to uncovered soil. The higher infiltration rate and water storage reduce runoff speed, decreasing losses of soil (Blanco-Canqui and Lal, 2009) and nutrients (Martins Filho et al., 2009; Silva et al., 2012; Sousa et al., 2012). In a climate change scenario, the maintenance of crop residues on the soil might decrease the effects of droughts, as well as prevent soil losses by erosion due to the occurrence of more frequent heavy rains.

In addition, crop residues could act as a thermal isolator reducing the heat exchange between the soil and the atmosphere, consequently reducing the amplitude

of diurnal soil temperurate. This impact is greater when the residue is fresh and decreases as the residue decays (Wilhelm et al., 2004). The temperature of soil-surface layer (seeding layer) might be reduced by 5 to 10 °C when the soil is covered with maize stover (Lal, 2009) and 2 °C when covered with sugarcane straw (Moitinho et al., 2013). In addition, the effects of residues on soil temperature are a complex set of processes that result in less evaporation of water from the soil when it is covered with crop residues (Wilhelm et al., 2004).

iii) Crop residues *vs* soil biological properties

Biological attributes are more sensitive to soil management changes when compared to chemical and physical attributes (Kaschuk et al., 2011). Recently, there has been an increase in the number of studies to identify and quantify the effects of crop residue harvest on the soil biota. Under no-tillage, maize stover harvest has a significant influence on the ratio fungus: bacteria, especially in the topsoil, but the effects on the soil microbial community structure depend on the duration of this system, as well as the type of soil and climatic conditions (Lehman et al., 2014). The maintenance of maize stover contributes to stability of the microbial community through the seasons (Lou et al., 2011). Moreover, the retention of maize residues could significantly affect the structure of the bacterial and fungal communities, with an increase in the abundance of the genes responsible for N fixation (njfH) and denitrification (napA) (Wakelin et al., 2007).

Studies have shown the beneficial effects of crop residues maintenance on the control of some soil pests and diseases. Qi et al. (2015) reported that high concentrations (> 0.12 g mL^{-1}) of compounds released during the decomposition of maize stover negatively affects germination, number and weight of the sclerotia of fungi *Rhizoctonia* and *Bipolaris.* Govaerts et al. (2007) observed that the increase in energy supply of microorganisms by keeping maize crop residues support the development of antagonists and predators in the soil, decreasing the population of phytoparasitic nematodes.

The harvest of residues from different bioenergy crops (*Andropogon gerardii*, *Miscanthus* × *giganteus*, *Sorghum bicolor*; *Panicum virgatum*; cv. *Shawnee*; and maize) showed no impact on total microbial biomass or on the microbial groups in annual or perennial systems. However, the restructuring of the soil microbial community is a gradual process, directly associated with changes in the quantity of organic material inputs, and therefore, should be considered a long-term effect (Orr et al., 2015).

Several studies conducted around the world have shown that the maintenance of sugarcane straw promotes increases in microbial biomass (Graham and Haynes, 2006; Souza et al., 2012; Paredes Jr. et al., 2015) and microbial community diversity (Rachid et al., 2013, 2016; Liao et al., 2014), especially in surface soil layers. In terms of soil macrofauna, despite the land-use change sequence, native vegetation-pasture-sugarcane induces a

significant reduction in the diversity of macroinvertebrates (Franco et al., 2016), the remaining straw support the soil biota as a food source, improving microclimate conditions and as a shelter in sugarcane areas (Portilho et al., 2011; Benazzi et al., 2013). Portilho et al. (2011) verified a significant reduction in the richness and diversity of soil epigeic invertebrates in plots under total sugarcane straw removal when compared to those without straw removal. The presence of sugarcane straw supported the soil macrofauna, although increased quantities did not cause significant effects (Abreu et al., 2014). Moreover, the maintenance of sugarcane straw on the soil might promote the multiplication of natural enemies and inhibit proliferation of phytoparasitic nematodes that cause significant damage to sugarcane plantations (Dinardo-Miranda and Fracasso, 2013). On the other hand, the maintenance of sugarcane straw also supports the proliferation of important sugarcane pests such as the root spittlebug [*Mahanarva fimbriolata* (Hemiptera; Cercopidae)] and the sugarcane weevil (*Sphenophorus levis* Vaurie (Coleoptera: Curculionidae) (Dinardo-Miranda and Fracasso, 2013), and if these pests are not properly controlled, they may cause significant losses in sugarcane yields.

iv) Crop residues *vs* soil C stocks

Soil carbon stocks in agricultural areas depend predominantly on two factors: i) the dynamics of SOM decomposition and ii) C input into the soil. Therefore, crop residues are the principal C input in agricultural soils (Poeplau and Don, 2015). In Brazil, the higher biomass production of the main crops, associated with C depletion at a large rate of the soils, determine a positive soil C balance in most sites where crop residues are kept in the field (Figure 1).
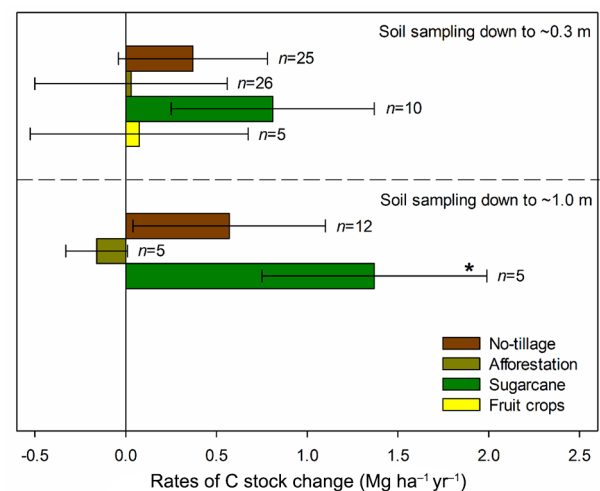

Figure 1 – Rates of C stock change (Mg ha^{-1} yr^{-1}) in the mainly Brazilian land uses related to high crop residues inputs. Estimates obtained from the studies in Appendix 1; Bars means standard deviation; *It includes two studies with soil sampling down to 0-0.6 m.

Conservationist practices that prioritize the maintenance of crop residues, such as no-tillage, are considered one of the principal strategies for soil C accretion. Based on data obtained from studies published in the last 10 years, we calculated a rate of C change of 0.37 ± 0.41 Mg ha^{-1} yr^{-1} at the 0-0.3 m soil layer in no-tillage areas in Brazil (Figure 1). If the layer below one meter is considered, the rate increases to 0.57 ± 0.53 Mg ha^{-1} yr^{-1}.

For a long time, C increases in no-tillage were attributed to absence of soil disturbance (e.g., Lal, 2004), because the natural aggregation of the soil is kept, protecting SOM against microbial respiration. However, this mechanism might not be the main reason for the increase in soil C in no-tillage areas, especially in highly weathered soils (Campos et al., 2011). In this context, the maintenance of crop residues emerges as the principal strategy to increase C stocks in areas using no-tillage system. Through an extensive revision of 276 studies conductes worldwide, West and Post (2002) showed a direct relationship between the input of crop residues and the increase in soil C in no-tillage areas. In Brazil, soil C stocks showed a linear relationship with the annual input of crop residues, regardless of the soil preparation system (no-tillage or conventional tillage) (Campos et al., 2011). Recently, Carvalho et al. (2017) predicted soil C changes using the DayCent model in Nebraska (USA) induced by maize stover removal and tillage practices. The results showed that at 75 % to 100 % stover removal rates in areas under conventional tillage, reductions close to 1 Mg ha^{-1} yr^{-1} of soil C stocks were observed; whereas under no-tillage, even at 100 % maize stover removal, soil C stocks increased in relation to the baseline under conventional tillage.

In contrast to what occurs in areas with annual crops under no-tillage, the literature is unanimous in affirming that the principal factor associated with the increase in soil C stocks in sugarcane crops with the maintenance of the sugarcane straw (Cerri et al., 2011; Meir and Thorburn, 2016; Carvalho et al., 2017; Oliveira et al., 2017). Historically, the practice of burning before harvest was common in sugarcane crops in Brazil and is related to soil C stock depletion (Galdos et al., 2009; Cerri et al., 2011). With the gradual elimination of burning, a large quantity of sugarcane straw has been left on the soil. As these residues decay, part of the C is emitted as CO_2 and part is incorporated into the soil, promoting increases in C stocks (Galdos et al., 2009; Cerri et al., 2011; Thorburn et al., 2012; Meir and Thorburn, 2016; Carvalho et al., 2017; Oliveira et al., 2017). In Brazil, field available data show an annual rate of C stock change in the soil in sugarcane crops without burning of 0.81 ± 0.56 and 1.37 ± 0.62 Mg ha^{-1}yr^{-1} for the 0-0.3 m and 0-1.0 m layers, respectively (Appendix 1). In two recent estudies conducted by Carvalho et al. (2017) and Oliveira et al. (2017), 30-year soil C changes induced by sugarcane straw management were predicted using the DayCent model. Carvalho et al. (2017) verified that the maintenance of sugarcane straw in the field increased

soil C by 0.19 and 0.09 Mg ha^{-1} yr^{-1} at two study sites, whereas straw removal rates promoted slight decreases in soil C stocks compared to areas without straw removal management. Oliveira et al. (2017) found a similar rate of soil C stocks accretion (0.11 Mg ha^{-1} yr^{-1}) when sugarcane straw is left on the field; however, they verified a significant decrease in soil C stocks (0.19 Mg ha^{-1} yr^{-1}) under 75 % straw removal.

During the whole cycle, forest species deposit a large quantity of residues onto the soil due to the natural process of senescence. Furthermore, at harvesting, part of the biomass is left on the soil. The high C input in these areas is associated with increases in soil C stocks in afforestation areas worldwide (e.g., Paul et al., 2002; Shi et al., 2015). In Brazil, studies assessing soil C changes in areas under forests are divergent, while some report C accretion (Lima et al., 2006; Zinn et al., 2011), others point to equilibrium (Cook et al., 2014; Fialho and Zinn, 2014) and even reductions in C stocks in these areas (Soares et al., 2013). Based on these studies, changes in soil C stock averaged 0.03 ± 0.53 and -0.16 ± 0.17 Mg ha^{-1} yr^{-1}, in the 0-0.3 m and 0-1.0 m layers, respectively, in cultivated forests in Brazil (Figure 1). Although there are divergent results about the potential of forest plantations to increase soil C stocks (Appendix 1), the harvest of crop residue in these areas is clearly associated with a negative impact on SOM in different regions of the world (Kumaraswamy et al., 2014; Epron et al., 2015) and in Brazil (Versini et al., 2014). Furthermore, the maintenance of forest residues has a positive correlation with forest yield in Brazilian plantations (Gonçalves et al., 2013).

Few studies have assessed the impacts of fruit crops on soil C changes in Brazil (Appendix 1). Based on these studies, a rate of C accretion of 0.074 ± 0.6 Mg ha^{-1} yr^{-1} for the 0-0.3 m layer was observed (Figure 1).

As discussed above, the maintenance of crop residues on the soil is one of the main factors associated with the positive soil C balance for some crops (Figure 1). Therefore, increasing crop residue harvest for bioenergy production can be associated with decreases in soil C stocks in areas with annual crops (e.g., Anderson-Teixeira et al., 2009; Blanco-Canqui, 2013; Liska et al., 2014; Wu et al., 2015; Carvalho et al., 2017), sugarcane (e.g., Carvalho et al., 2017; Oliveira et al., 2017) and afforestation (e.g., Hudiburg et al., 2011). Certainly, energy production from alternative biomass is essential for the transition to a low-C economy (Sanchez et al., 2015). However, since the use of bioenergy is supposed to mitigate GHG emissions, soil C losses and GHG emissions associated with crop residue harvest should be carefully evaluated.

In Brazil, the main scenario of crop residue harvest for bioenergy production occurs in sugarcane areas. However, only a few studies evaluate soil C stock responses to sugarcane straw harvest in these areas (Carvalho et al., 2017; Oliveira et al., 2017). Therefore, field research is essential to produce knowledge to support making decisions about investments and policies for the sugarcane industry.

Impact of crop residues on GHG emissions

To date, there is no consensus on the real impact of crop residue harvest on GHG emissions of the soil. Some studies claim that the harvest of crop residues stimulate GHG emissions (Carmo et al., 2013; Liska et al., 2014). On the other hand, other studies indicate that residue harvest management does not alter GHG emissions (e.g., Baker et al., 2014; Guzman et al., 2015) or even reduce GHG emissions of the soil (Al-Kaisi and Guzman, 2012; Davis et al., 2012; Bentsen et al., 2014; Jin et al., 2014). Diverse factors affect directly or indirectly the production and emission of GHG, contributing to the variability of the results found in the literature. These factors could be environmental, edaphic and related to the quality and quantity of the material used to cover the soil (Pimentel et al., 2015).

In general, the largest GHG emissions are found when a large quantity of material is left on the soil surface (Carmo et al., 2013; Jin et al., 2014; Signor et al., 2014). Carmo et al. (2013) observed a decrease in emissions of 2.3 times under total straw harvest compared to 14 Mg ha^{-1} or 21 Mg ha^{-1} of sugarcane straw left on soil surface. A similar trend was observed under total maize stover harvest compared to no harvest (Jin et al., 2014).

Generally, an increase in soil CO_2 emissions is observed with the maintenance of the crop residues in the field (Carmo et al., 2013; Baker et al., 2014; Jin et al., 2014; Signor et al., 2014). Jin et al. (2014) observed that maize stover harvest decreased soil total CO_2 emissions by 4 % compared to no stover harvest at nine sites across the US Corn Belt. However, CO_2 from decomposition of residues is not counted in studies that produce inventories of GHG emission in agriculture, because this CO_2 (biogenic) is cycled in the soil-plant-atmosphere continuum and it is distinguished from CO_2 emitted from fossil fuels used for energy production, which accumulates in the atmosphere (Pourhashem et al., 2013). Nevertheless, crop residue harvest could favor CO_2 emissions, due to the increase in temperature and reduction of moisture in the soil (Corradi et al., 2013; Guzman et al., 2015). The harvest of crop residues over the years should reduce organic C in the soil that could cause significant implications for CO_2 emissions (Six et al., 2004).

The maintenance of crop residues on the soil could also promote the immediate increase in N_2O emissions (Carmo et al., 2013; Jin et al., 2014). However, this scenario could be reversed in the long term, when mineralized N from straw becomes available to the plants (Meier and Thorburn, 2016). Thus, N inputs via crop residues can reduce the need for replacement via synthetic N fertilizers (Robertson and Thorburn, 2007; Davis et al., 2012; Trivelin et al., 2013; Costa et al., 2014; Ferreira et al., 2016; Meier and Thorburn, 2016), the main source of N_2O emissions in agriculture (Smith et al., 2007; Guzman et al., 2015). In this sense, GHG emissions of N fertilizers in the agricultural phase have raised doubts about the environmental viability of replacing fossil fuels for maize ethanol (Crutzen et al.,

2008). Residue harvest can also cause an increase in soil temperature, stimulating microbial activity and N mineralization, resulting in higher N_2O emissions from the soil (Mosier et al., 2002).

Signor et al. (2014) observed increases in CH_4 emissions with the maintenance of sugarcane straw. On the other hand, various studies did not find a change in CH_4 emissions from soils covered with straw (Carmo et al., 2013; Baker et al., 2014; Jin et al., 2014). Other studies did not consider CH_4 emission in agricultural soils, due to its low emission in relation to CO_2 and N_2O, lower than 1 and 10 %, respectively (Guzman et al., 2015). Lastly, it is possible that the crop residue harvest does not cause notable changes in factors that affect methanogenesis in the soil, such as redox potential (Agostinetto et al., 2002).

Field measurements of the effects of crop residue harvest on GHG emissions are still limited in Brazil and over the world (Carmo et al., 2013; Baker et al., 2014; Jin et al., 2014; Signor et al., 2014; Pitombo et al., 2015). Most studies on this issue have used model simulations (Searchinger et al., 2008; Davis et al., 2012; Campbell et al., 2014; Liska et al., 2014; Zhao et al., 2015; Cardozo et al., 2016), as well as life cycle analysis assessments (Macedo et al., 2008; Bordonal et al., 2012, 2015). For example, Liska et al. (2014) verified that the production of gasoline emits 7 % less GHG when compared to the production of cellulosic ethanol from maize stover.

Besides the direct effects of using crop residues for bioenergy production, there are indirect effects, such as the decrease on land demand for feedstock production. Most uncertainties regarding biofuel sustainability are based on the GHG emissions associated with land-use change (Searchinger et al., 2008; Lapola et al., 2010). Therefore, cellulosic ethanol production might solve some of these issues, since there is no need to convert more land to increase production. According to Adler et al. (2007), the harvesting of 50 % of the straw could increase ethanol production by 35 % in maize areas, without the need to expand the area, reducing potential GHG emissions.

In general, crop residue harvest might reduce the direct and indirect GHG emissions in agricultural areas, increasing the sustainability of its industrial use. Moreover, the maintenance of a large quantity of straw in the field is improbable due to the economic perspective of cellulosic ethanol and bioelectricity. Therefore, future research should concentrate on responding questions regarding the ideal quantity of straw to be removed for the production of biofuels without increasing the GHG emissions.

Impact of crop residues on growth and productivity of plants

The management of crop residues can have direct and indirect effects on crop yield (Pittelkow et al., 2015). In tropical conditions, studies have shown different responses of plants to the maintenance of sug-

arcane straw on the soil. Tavares et al. (2010) reported that straw maintenance on the soil surface improved tillering and final population of plants, which increased the stalk yield in relation to areas with straw burning. Conversely, Campos et al. (2010) found reduced tilling, growth and productivity of the plants at the end of the cycle when all straw was left on the soil surface.

In temperate zones, the full retention of the straw compromised sugarcane productivity, while the straw burning before harvest was associated with increased stalk yield (Wiedenfeld, 2009; Arceneaux and Selim, 2012). However, the beneficial effects of straw burning on productivity are generally restricted to the short term (Wiedenfeld, 2009; Viator and Wang, 2011).

In sub-tropical conditions, reduced sugarcane tillering (Olivier and Singels, 2006, 2012) and growth (Olivier and Singels, 2012) were observed with moderate amount of straw kept on the soil surface. Although there are possible negative effects of the straw in the crop establishment phase, no reduction in stalk yield was observed (Olivier and Singels, 2006, 2012). The negative effects of straw during the crop establishment phase might be associated to extracts released during the decomposition of sugarcane straw, which contain a large variety of functional groups such as phenols, alcohols, organic acids, ketones, ethers and aldehydes (Villegas et al., 2007). High concentrations of these biochemical substances could be toxic, affecting crop germination (Viator et al., 2006; Villegas et al., 2007).

Recent studies have shown that total harvest or maintenance of only 25 % (5 Mg ha^{-1}) of the sugarcane straw reduces crop yield (Aquino and Medina, 2014; Aquino et al., 2015). In contrast, the maintenance of 50 % (10 Mg ha^{-1}) of the straw was sufficient to increase sugarcane root mass, which resulted in increased stalk yield (Aquino and Medina, 2014; Aquino et al., 2015).

In the United States, sustainability of the maize stover harvest depends on the management system. The maintenance of maize residues in no-tillage areas is associated with reduced seedling emergence (Dam et al., 2005) and the standing of plants in the establishment phase (Blanco-Canqui et al., 2006), plant growth (Sindelar et al., 2013), which can decrease grain productivity (Linden et al., 2000; Boomsma et al., 2010). Besides the effect of temperature regulation, the decay of maize stover also releases biochemical compounds that may affect plant growth (Qi et al., 2015). In contrast, stover harvest may also increase grain productivity (Karlen et al., 2014; Pantoja et al., 2015). However, the indiscriminate harvest of maize crop residues, even in areas under no tillage, might affect the system sustainability (Blanco-Canqui and Lal, 2009), mainly in years with low rainfall (Linden et al., 2000).

Some studies indicate that the harvest of approximately 50 % of maize residues had little effect on crop productivity (Kenney et al., 2015; Jin et al., 2015). On the other hand, the harvesting of values above 25 % may reduce grain productivity in the subsequent harvest (Blanco-Canqui and Lal, 2007, 2009). In some cases, whether the residues were kept or fully removed, little effect on maize productivity was observed (Dam et al., 2005). After evaluating different harvest strategies over a 4-year period, Birrell et al. (2014) concluded that maize stover harvest had little effect on the soil quality or crop yield. However, these authors suggested a continuous monitoring of soil fertility, soil C content and crop productivity to establish a sustainable long-term residue harvest management. The lack of a consensus about the quantity of maize crop residues that can be harvested without affecting crop productivity is justifiable, since the response of the crop to this practice depends on the location, soil type, topography and climate (Blanco-Canqui et al., 2006, 2007; Birrell et al., 2014).

Forestry residue harvest for bioenergy production has caused concern about the potential negative long-term effects on soil quality and wood productivity (Achat et al., 2015; Jang et al., 2015). Studies have shown that retention of forest residues increases the nutritional status and the productivity of eucalyptus trees (Mendham et al., 2003; Gonçalves et al., 2007; Rocha et al., 2016). Rocha et al. (2016) observed that the total removal of forest residues decreased wood production by 40 % in the first and 6 % in the second rotation of eucalyptus. On the other hand, in boreal forests, residue harvesting may not affect growth or productivity of the trees (Kaarakka et al., 2014). Global meta-analysis performed by Achat et al. (2015) showed that the harvesting of residues from forestry plantations reduced soil fertility, causing a reduction from 3 to 7 % in the growth of the trees in subsequent cultivations.

Incorporating best management practices for sustainable crop residue management

Crop residues play essential roles to maintain or improve soil quality/health, plant growth and other ecosystem services, as discussed above. However, the benefits provided by crop residues left on field can be lost through indiscriminate harvest to produce bioenergy (e.g., biofuels and bioeletricity). Thus, crop residue management needs to be considered within a holistic view, integrating best management practices (BMPs) to attenuate adverse impacts caused by crop residue harvest. Best management practices that enhance the sustainability of residue harvest include replacement of soil nutrients removed with the residue, use of no-tillage and crop rotation, inclusion of annual or perennial cover crops, and the use of manure and other organic amendments (Blanco-Canqui and Lal, 2009; Karlen et al., 2011a; Blanco-Canqui, 2013; Paustian et al., 2016).

The adoption of conservationist tillage could considerably increase the amount of collectable crop residues for industrial use without compromising soil C stocks (Carvalho et al., 2017). The absence of soil disturbance is one of the main drivers to prevent soil C losses in no-tillage systems (Segnini et al., 2013; Silva-Olaya et al., 2013; Oliveira et al., 2017). In sugarcane, soil till-

age for unburned sugarcane renewal (every 5-6 years) could generate a loss equivalent to 80 % of soil C that is potentially accumulated in this soil layer during one year of cultivation (Silva-Olaya et al., 2013). In annual crops, soil C losses are even higher, since soil tillage is performed every year.

Predictions using the DayCent model indicated that the effects of maize residue harvest on soil C stocks are much more severe under conventional tillage than in no-tillage in the US Corn Belt (Campbell et al., 2014; Carvalho et al., 2017), reaching a soil C loss rate of close to 1 Mg ha^{-1} yr^{-1} (Carvalho et al., 2017). No-tillage rotations that include semi-perennial grasses or legumes also offer a promise to promote soil-profile C sequestration and improve soil resilience after residue removal (Blanco-Canqui, 2013). The sustainability of crop residue management under no-tillage can be enhanced by the introduction of cover crops into the agricultural system (Blanco-Canqui, 2013; Adner et al., 2015; Wegner et al., 2015). According to Blanco-Canqui (2013), no-tillage cover crops can sequester between 0.10 and 1 Mg ha^{-1} yr^{-1} of C in relation to no-tillage without cover crops, depending on cover crop species, soil type, and precipitation input.

Considering the management of sugarcane straw for bioenergy production (cellulosic ethanol and bioelectricity), Oliveira et al. (2017) simulated some scenarios that well illustrated the potenciality of conservacionist management and other BMPs to mitigate the negative effects of straw removal on soil C accretion in Brazil (Figure 2). The authors reported that the rate of soil C loss due to sugarcane straw harvest decreases from 0.19 Mg ha^{-1} yr^{-1} (conventional cultivation) to 0.11 Mg ha^{-1} yr^{-1} when no-tillage is adopted. Although the adoption of no-tillage in sugarcane fields is not commom in Brazil, it is important to consider this practice during crop renovation, mainly in a scenario of straw removal with possible

soil C losses. Moreover, a positive soil C balance could be reached if an alternative C source (e.g., vinasse, filtercake) were added (Figure 2), and an average increase in soil C stocks of 0.25 Mg ha^{-1} yr^{-1} was observed when both practices (no-tillage and organic amendments) were adopted in sugarcane areas with straw removal (Oliveira et al., 2017).

To date, most research has focused on the effects of crop residue harvest on SOM, because of its multiple influences on soil properties and processes. Thus, conservacionist management practices can also increase nutrient cycling, water storage, biological activity as well as the soil resistance to structural degradation (Blanco-Canqui and Lal, 2009; Blanco-Canqui, 2013; Franzluebbers, 2015; Stavi et al., 2016). For instance, Tormena et al. (2016) observed through a visual evaluation method that deleterious impacts of high maize stover harvest on soil structural quality were magnified when a chisel plow was used as compared to no-tillage in Iowa (USA). The introduction of alfafa (cover crop) under no-tillage increased the available water and reduced the potential restriction of penetration resistance to crop growth (Tormena et al., 2017). In addition, no-tillage also can mitigate GHG emissions compared to conventional tillage, as reported by Silva-Olaya et al. (2013) and Jin et al. (2014).

Furthermore, integrated models and variable-removal technologies that allow the recognition of spatial heterogeneities in crop residue production and collection of a sustainable rate of crop residue harvest (i.e., based on soil characteristics, topography and management practices adopted) on the field or sub-field scale should be pursued (for instance, see Muth et al., 2012). It could prioritize a higher rate of residue harvest in areas with higher yields and lower potential for degradation, by erosion for example, minimizing the adverse environmental impacts caused by the spatially uniform management of crop residues.

Final Remarks

The use of crop residues for energy purposes is certainly one of the main alternatives to increase bioenergy production in the next few years. Without the need to expand the cultivated area, the threats against natural ecosystems and food production are greatly reduced. However, various trade-offs associated with crop residue harvest from the field were presented and discussed throughout this literature review and are summarized in Figure 3.

Consistently, literature reports that crop residue harvest has a negative impact on nutrient cycling and availability in the soil. Long-term nutrient removal by crop residues harvest impoverishes the soil, demanding their replacement using increasing quantities of mineral fertilizers. This produces an increase in production costs and greater impact on the environment. Furthermore, the reduction in the input of organic residues into the soil implies a direct reduction in C stocks, which nega-
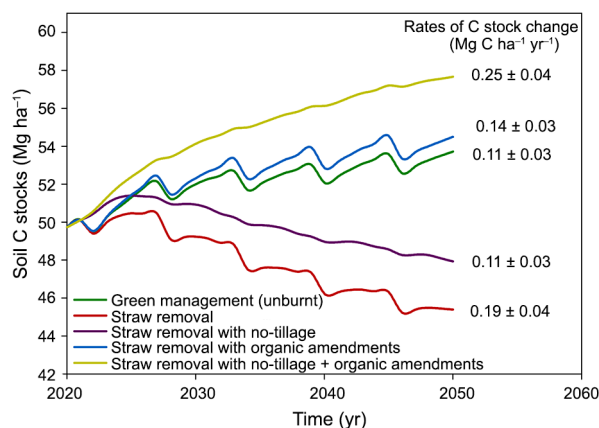

Figure 2 – Simulated soil C stocks at 0-0.3 m layer in sugarcane areas under green management, straw removal, and best management practices in Brazil; The reference for the rates of C stock change is the soil C stock in areas under green management by 2020; Source: Oliveira et al. (2017).
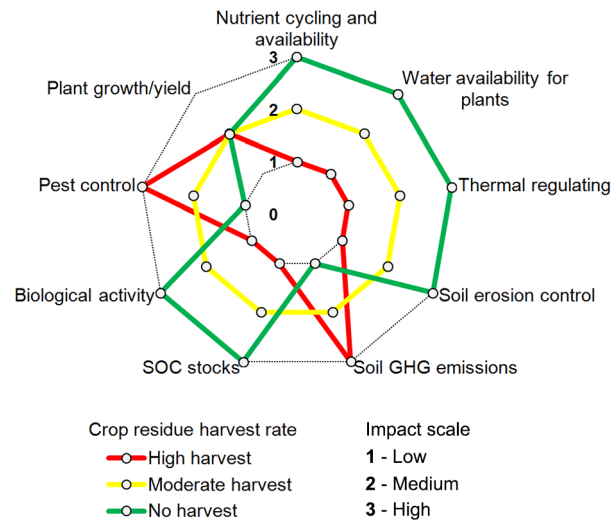
Figure 3 – Impacts of crop residue management on soil functions and plant growth. The soil functions and plant growth were graded for each crop residue harvest rate (high, moderate and low) according to the impact scale (i.e., 1 - low, 2 - medium and 3 - high); Source: adapted from Stavi et al. (2016).

tively affects the soil biota. Deleterious effects from crop residue harvest extend from soil microbial to macrofauna communities. On the other hand, the increase in biological activity in the soil promoted by the maintenance of the residues has not only positive effects on the plants, but it also promotes the proliferation of pests, as it has been consistently reported in sugarcane fields. Crop residue harvest also makes the soil more susceptible to structural degradation, leading to compaction and consequently, resulting in lower infiltration and storage of water, and decreased plant root growth. Furthermore, crop residues act as physical/mechanical barriers that help to protect the soil against raindrop impact, which reduces the risk of erosion. In addition, crop residue mulch acts as a temperature isolator, reducing the amplitude of soil temperature and water evaporation.

The impacts of crop residue management on soil GHG emission are still not fully understood and demand more field studies. In general, the harvesting of residues reduce emissions of CO_2 and N_2O produced by decay and has no effects on CH_4 emission. However, the gradual depletion of C and N stocks in the soil, associated with replacement of N via mineral fertilizers, could induce a negative C balance and higher N_2O emissions in areas where the residues are harvested.

The plant response to crop residue management is site specific. Therefore, in addition to edaphoclimatic conditions that directly affect growth and crop productivity, factors intrinsic to the culture and management practices are determinants in the plant response to residue harvest. However, considering that residue management for bioenergy production is something recent,

most experiments have been running for a few years, hindering assessment of the sustainability of this practice in the long term.

As crop residues have multiple funtions in the soil, affecting directly and indirectly diverse ecosystem services, investments in research to better understand the impact associated with residue management are essential to define strategies for the industrial use of this raw material. Additionally, the integrated planning of crop residue management, including best management practices (e.g. conservacionist tillage, crop rotation and cover crops, nutrient management and/or organic residues) can partially offset the adverse impacts caused by crop residue management, moving towards more sustainable bioenergy production.

Finally, collaboration networks between private and public institutions should be encouraged towards synchronizing the generation of knowledge and its application on a field scale, especially in Brazil, one of the world's major players in the bioenergy sector.

Acknowledgments

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Appendix 1 – Rates of soil C stock change (Mg ha^{-1} yr^{-1}) in the mainly Brazilian land uses related to high crop residues inputs; Data obtained from studies published in the last 10 years (in English).

Local (Lat)[a]	Soil Type	Clay content (g kg^{-1}[b])	Land Use[c]	Sampling depth (m)	C stock change (Mg ha^{-1} yr^{-1}[d])	Reference
			No-Tillage			
30°51' S	Acrisol	220	C-M	0-0.175	0.82	Diekow et al., 2005
30°51' S	Acrisol	220	C-M	0-0.175	0.86	Diekow et al., 2005
25°00' S	Ferralsol	500	C-M-W-S	0-0.2	-0.01	Alburquerque et al., 2015
25°00' S	Ferralsol	500	C-M-W-S	0-0.2	-0.05	Alburquerque et al., 2015
25°00' S	Ferralsol	500	C-M-W-S	0-0.2	0.02	Alburquerque et al., 2015
25°00' S	Ferralsol	500	C-M-S	0-0.2	0.12	Alburquerque et al., 2015
25°00' S	Ferralsol	500	A-M	0-0.2	0.23	Alburquerque et al., 2015
16°15' S	Typic Hapludox	350	M-S	0-0.2	0.30	Bayer et al., 2006
18°31' S	Typic Hapludox	650	M-S-CO	0-0.2	0.60	Bayer et al., 2006
30°51' S	Acrisol	220	C-M-C	0-0.2	0.25	Conceição et al., 2013
30°51' S	Acrisol	220	C-M	0-0.2	0.18	Conceição et al., 2013
28°15' S	Rhodic Ferralsol	630	C-S-C-S-C-M	0-0.3	0.25	Boddey et al., 2010
28°29' S	Rhodic Ferralsol	640	W-S	0-0.3	-0.24	Boddey et al., 2010
28°29' S	Rhodic Ferralsol	640	C-M-W	0-0.3	0.21	Boddey et al., 2010
28°30' S	Rhodic Ferralsol	630	W-S-W-M-C	0-0.3	0.35	Boddey et al., 2010
28°30' S	Rhodic Ferralsol	630	W-S-W-M-C	0-0.3	0.04	Boddey et al., 2010
28°36' S	Oxisol	520	W-S-C	0-0.3	0.29	Campos et al., 2011
28°36' S	Oxisol	520	C-S-C-M-C-W-S	0-0.3	0.51	Campos et al., 2011
12°29' S	Typic Hapludox	730	R-M-S	0-0.3	0.38	Carvalho et al., 2009
SI	Latosols	660	S-M/SO/MI	0-0.3	1.90	Siqueira Neto et al., 2010
30°51' S	Acrisol	220	C-M	0-0.3	0.32	Zanatta et al., 2007
30°51' S	Acrisol	220	C-M	0-0.3	0.40	Zanatta et al., 2007
30°51' S	Acrisol	220	C-M-C	0-0.3	0.65	Zanatta et al., 2007
28°15' S	Rhodic Ferralsol	630	C-S-C-S-C-M	0-1	1.11	Boddey et al., 2010
28°29' S	Rhodic Ferralsol	640	W-S	0-1	-0.22	Boddey et al., 2010
28°29' S	Rhodic Ferralsol	640	C-M-W-S-C-S	0-1	0.52	Boddey et al., 2010
28°30' S	Rhodic Ferralsol	630	W-S-W-M-C-M	0-1	0.85	Boddey et al., 2010
28°30' S	Rhodic Ferralsol	630	W-S-W-M-C	0-1	0.48	Boddey et al., 2010
25°00' S	Ferralsol	500	C-M-W-S	0-1.0	0.06	Alburquerque et al., 2015
25°00' S	Ferralsol	500	C-M-W-S	0-1.0	0.29	Alburquerque et al., 2015

Appendix 1 – Continuation.

25°00′ S	Ferralsol	500	C-M-W-S	0-1.0	0.14	Alburquerque et al., 2015
25°00′ S	Ferralsol	500	C-M-S	0-1.0	0.28	Alburquerque et al., 2015
25°00′ S	Ferralsol	500	A-M	0-1.0	0.50	Alburquerque et al., 2015
30°51′ S	Acrisol	220	C-M	0-1.075	1.40	Diekow et al., 2005
30°51′ S	Acrisol	220	C-M	0-1.075	1.48	Diekow et al., 2005
			Afforestation			
19°14′ S	ud	590	Eucalyptus	0-0.1	0.35	Lima et al., 2006
18°42′ S	ud	700	Eucalyptus	0-0.1	0.57	Lima et al., 2006
19°11′ S	Ultisol	120	Eucalyptus	0-0.1	-0.51	Soares et al., 2013
18°70′ S	Ultisol	100	Eucalyptus	0-0.1	0.38	Soares et al., 2013
18°44′ S	Oxisol	740	Eucalyptus	0-0.1	0.18	Soares et al., 2013
17°50′ S	Oxisol	770	Eucalyptus	0-0.1	-0.80	Soares et al., 2013
29°56′ S	Alfisol	420	Eucalyptus	0-0.1	-0.33	Soares et al., 2013
Rw	rw	rw	Eucalyptus	0-0.2	-1.50	Fialho and Zinn, 2014
19°14′ S	Oxisol/Inceptisol	670	Eucalyptus	0-0.2	0.07	Lima et al., 2011
19°45′ S	Oxisol/Inceptisol	490	Eucalyptus	0-0.2	0.28	Lima et al., 2011
19°57′ S	Oxisol/Inceptisol	485	Eucalyptus	0-0.2	0.15	Lima et al., 2011
18°42′ S	Oxisol/Inceptisol	605	Eucalyptus	0-0.2	1.01	Lima et al., 2011
18°39′ S	Oxisol/Inceptisol	645	Eucalyptus	0-0.2	0.43	Lima et al., 2011
18°46′ S	Oxisol/Inceptisol	760	Eucalyptus	0-0.2	0.43	Lima et al., 2011
19°24′ S	Oxisol/Inceptisol	630	Eucalyptus	0-0.2	0.02	Lima et al., 2011
19°56′ S	Oxisol/Inceptisol	530	Eucalyptus	0-0.2	0.88	Lima et al., 2011
19°28′ S	Red Latosol	730	Pinus	0-0.2	-0.18	Wendling et al., 2010
17°25′ S	Haplustoxes	340	Eucalyptus	0-0.2	-0.03	Zinn et al., 2011
17°24′ S	Haplustoxes	623	Eucalyptus	0-0.2	-0.14	Zinn et al., 2011
17°00′ S	Haplustoxes	867	Eucalyptus	0-0.2	-0.09	Zinn et al., 2011
23°02′ S	Ferralsols	180	Eucalyptus	0-0.3	0.07	Maquere et al., 2008
23°02′ S	Ferralsols	175	Eucalyptus	0-0.3	0.04	Maquere et al., 2008
11°58′ S	Typic Haplustox	100	Eucalyptus	0-0.3	-0.90	Stape et al., 2008
Rw	rw	rw	Eucalyptus	0-0.4	0.30	Fialho and Zinn, 2014
22°40′ S	Typic Hapludox	240	Eucalyptus	0-0.45	0.03	Cook et al., 2014
22°40′ S	Typic Hapludox	210	Pinus	0-0.45	0.00	Cook et al., 2014
23°02′ S	Ferralsols	180	Eucalyptus	0-1	-0.03	Maquere et al., 2008
23°02′ S	Ferralsols	175	Eucalyptus	0-1	-0.01	Maquere et al., 2008
17°25′ S	Haplustoxes	340	Eucalyptus	0-1.0	-0.21	Zinn et al., 2011
17°24′ S	Haplustoxes	623	Eucalyptus	0-1.0	-0.43	Zinn et al., 2011
17°00′ S	Haplustoxes	867	Eucalyptus	0-1.0	-0.14	Zinn et al., 2011
			Sugarcane			
21°22′ S	Hapludox	Clayey	SG GM	0-0.1	0.65	Razafimbelo et al., 2006
21°30′ S	Entisol	370	SG GM	0-0.2	0.4	Canellas et al., 2010
21°25′ S	Haplustult	150	SG GM CT	0-0.2	1.16	Figueiredo et al., 2015
21°25′ S	Haplustult	150	SG GM NT	0-0.2	0.39	Figueiredo et al., 2015
21°22′ S	Typic Hapludox	760	SG GM	0-0.2	1.2	Galdos et al., 2009
Es	es	es	SG GM CT	0-0.3	0.24	Bordonal et al., 2012
Es	es	es	SG GM NT	0-0.3	0.3	Bordonal et al., 2012
15°19′ S	Typic Hapludox	621	SG	0-0.3	1.05	Brandani et al., 2015
Rw	rw	Clay	SG GM	0-0.3	2.04	Cerri et al., 2011
Rw	rw	Sand	SG GM	0-0.3	0.73	Cerri et al., 2011
Sl	sl	sl	SG GM	0-0.6	1.1	Carvalho et al., 2013
21°12′ S	Oxisol	570	SG GM CT	0-0.6	0.67	Segnini et al., 2013
21°12′ S	Oxisol	570	SG GM NT	0-0.6	1.63	Segnini et al., 2013
21°22′ S	Typic Hapludox	760	SG GM	0-1.0	2.5	Galdos et al., 2009
19°18′ S	Haplic Acrisol	250	SG GM	0-1.0	0.93	Pinheiro et al., 2010
			Fruit crops			
10°19′ S	Ultisols	ud	Banana	0-0.3	0.8	Guimarães et al., 2014
10°19′ S	Ultisols	ud	Citrus	0-0.3	-1.69	Guimarães et al., 2014
5°09′ S	Eutric Cambisol	261	Banana	0-0.4	0.09	Assis et al., 2010
5°08′ S	Haplocambid	261	Banana	0-0.5	1.48	Oliveira et al., 2015
5°08′ S	Haplocambid	400	Guava	0-0.5	-0.31	Oliveira et al., 2015

[a]sl = studied carried out in several areas; rw = review; es = studied based on estimates; [b]ud = unavailable data; [c]C = cover or winter crops; M = maize; W = wheat; S = soybean; A = alfalfa; CO = cotton; R = rice; SO = sorghum; MI = millet; SG = sugarcane; GM = green management; NT = no tillage; CT = conventional tillage; [d]calculated considering the native vegetation or previous land use as reference; If soil density were not provided, 1.2 g cm^{-1} would be adopted to calculate soil C stocks.