Chapter V

USE OF DIGESTATE AS FERTILIZER

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Introduction

Anaerobic digestion is still an emerging technology in Brazil (Kunz et al., 2009) despite the great potential for the energy use of agro-industrial residues available in the country (Abiogás, 2015). In Europe, this industry is already developed. A report by the European Biogas Association pointed out that there were already 17,240 biogas plants in operation in 2015 on that continent, notably in Germany, where around 60% of these plants were installed (EBA, 2015). Brazil had approximately 150 biogas plants operating in 2016, less than 1% of the installed capacity in Europe (Cibiogás, 2016).

One of the biggest challenges for the development of this industry is the need for the correct disposal of effluent from biodigesters (digestate). If, on the one hand, there are already technologies for the digestate treatment (Chapters VI and VII), aiming at nutrient removal (nitrogen and phosphorus) and enabling the reuse of wastewater or its disposal into receiving water bodies, on the other hand, the use these technologies add costs that impact the economic viability of these projects (Miele et al., 2015). The recycling of digestate as a fertilizer in agriculture removes part of the added cost with the implementation and operation of digestate treatment systems, but aspects related to the supply of nutrients via digestate, the demand for nutrients in agricultural areas available for recycling, and the logistics of fertilizer distribution projects should be considered in these projects, as they also add costs and have technical limitations (Miele et al., 2015; Nicoloso, 2014).

The technical criteria necessary for the correct destination of digestate as a source of nutrients for agriculture will be discussed in this chapter. The concepts that will be exposed here are valid both for larger-scale projects (biogas plants) and for smaller-scale biodigesters for the treatment, for example, of animal manure and other residues from rural properties or decentralized energy generation condominiums (Olivi et al., 2015). The environmental impacts related to the use of digestate as a fertilizer and strategies for its mitigation will also be addressed.

Characterization of digestate as a fertilizer

The digestate quality and its potential for agronomic use depend on several factors, namely: (a) composition and variability of residues used as substrates for biodigestion (e.g., waste and carcasses of dead animals, agro-industry residues, residues, or plant biomass, among others); (b) type of biodigester and biodigestion technology; (c) segregation and loss of nutrients in the substrate and digestate storage structures; (d) efficiency of substrate pre-treatment systems (e.g., separation of phases before the biodigester) and/or digestate treatment; and (e) dilution of substrates and digestate with water. Table 1 shows the amount of nutrients (nitrogen, phosphorus, and potassium) associated with some residues of animal origin, plant biomass, and agro-industrial residues commonly used as substrates in biodigesters.

In addition to differences in the chemical composition and variability among substrates, the different proportions of substrate mixtures to be used in the feeding of the biodigester will also have a major impact on the nutrient composition of the digestate. Therefore, each project must have a specific analysis to determine the supply and nutrient content of the digestate available for recycling as fertilizer in agriculture. The values shown in Table 1 can be used for dimensioning the supply of nutrients via digestate that must be submitted to treatment or recycling in agricultural areas as fertilizer. However, the processes of loss and segregation of nutrients that can occur in the biodigester and effluent treatment or storage systems need to be considered. Vivan et al. (2010) found no significant variation in the concentration of TKN (total Kjeldahl Nitrogen), NH_3-N (N ammoniacal), and P (phosphorus) between the wastewater (liquid swine manure) and the digestate from a covered lagoon biodigester with a hydraulic retention time (HRT) of 45 days. However, reductions in the contents of these nutrients in the order of 50%, 30% and 77%, respectively, were observed after passing the digestate through an anaerobic lagoon with an HRT of 55 days. This reduction in N contents was attributed to ammonia volatilization losses, which can be increased by the mineralization of organic N during the biodigestion process. On the other hand, the reduction in P contents in the digestate was attributed to the physicochemical P precipitation, mostly in the form of calcium phosphate (Steinmetz, 2007). Therefore, P is not lost but segregated, as observed by the increase in the concentrations of this nutrient in the sludge deposited in the digestate storage lagoons (Zanotelli et al., 2005). In general, N losses of 50%–60% are expected for swine manure treated by biodigestion, also considering the digestate storage before its application to the soil (Fatma, 2014). The other nutrients have no considerable losses although the segregation of nutrients between the different types of effluents from biodigesters (e.g., sludge and liquid digestate) should be considered.

Table 1. Amount of nutrients associated with some animal residues commonly used as a substrate for anaerobic digestion.

Type	Source	Substrate production		Nutrient			
		Amount	Unit	N	P_2O_5	K_2O	Unit
Swine and cattle slurry[1]	Swine, finishing	1.64	m^3 $piglet^{-1}$ $year^{-1}$	8.0	4.3	4.00	kg.$piglet^{-1}$.$year^{-1}$
	Swine, nursery	0.84	m^3 $piglet^{-1}$ $year^{-1}$	0.40	0.25	0.35	kg.$piglet^{-1}$.$year^{-1}$
	Swine, PPU	8.32	m^3 sow^{-1} $year^{-1}$	25.7	18.0	19.4	kg.sow^{-1}.$year^{-1}$
	Swine, CC	17.2	m^3 sow^{-1} $year^{-1}$	85.7	49.6	46.9	kg.sow^{-1}.$year^{-1}$
	Cattle, milk	20.0	m^3 $animal^{-1}$ $year^{-1}$	65.6	36.8	61.8	kg.$animal^{-1}$.$year^{-1}$
Litter[2]	Broiler chickens	2.36	kg $animal^{-1}$ $year^{-1}$	67.0	71.0	62.0	g.$animal^{-1}$.$year^{-1}$
Animal carcasses[3]	Swine	75.0	kg (mean weight)	21.5	63.6	35.5	kg.ton^{-1}
	Broiler chickens	2.5	kg (mean weight)	30.5	57.1	24.5	kg.ton^{-1}
Plant biomass[4]	Corn silage	21.0	ton ha^{-1} (DM)	9.5	9.5	12.7	kg.ton^{-1} (DM)
	Sorghum silage	23.1	ton ha^{-1} (DM)	3.5	2.7	4.6	kg.ton^{-1} (DM)
	Sunflower silage	15.9	ton ha^{-1} (DM)	11.8	15.2	28.7	kg.ton^{-1} (DM)
Agro-industrial residues[5]	Sugarcane vinasse	13.0	L L^{-1} de ethanol	0.37	0.60	2.03	kg.m^{-3}
	Filter cake (sugarcane)	35.0	kg ton^{-1} sugarcane	1.40	1.94	0.39	% (DM)

[1]Calculated based on Nicoloso and Oliveira (2016) and Miele et al. (2015). Manure and nutrient production per housed animal: finishing units, considering 3.26 lots of finishing pigs per year; nursery, considering piglets up to 28 days old; the unit for piglet production units (PPU) and complete cycle (CC) is hosted sow, considering 2.35 farrows per year, 12 piglets per farrow, and 11.5 finished piglets per sow per farrow. [2]Calculated based on Nicoloso et al. (2016a) and considering 13 poultry housed per square meter, 0.10 m litter thickness, 600 kg litter density per cubic meter, and litter change every 15 lots of 42 days and 7 days apart. [3]TEC-DAM (2017). [4]Oliveira et al. (2010). [5]Soares et al. (2014); DM: dry matter.

A field survey carried out in a microbasin in the Santa Catarina State, Brazil, showed that the digestate from covered lagoon biodigesters treating the same type of substrate (e.g., swine manure) had high variability in terms of N, P_2O_5, and K_2O contents (Table 2). The biodigesters had similar characteristics despite the distinct origin of substrates (type of farm). In this case, the high variability of results was attributed to differences in the farm manure management (waste of water), biodigester operation (some of them had systems for separating coarse solids from the wastewater), occurrence of rainwater inlet in some facilities (poorly oriented drainage of the terrain), and, mainly, long digestate storage time in some of these units, which allowed P precipitation into the lagoon sludge, considerably reducing the P_2O_5 content of the liquid digestate (Olivi et al., 2015).

Table 2. Characterization of digestate from covered lagoon biodigesters treating liquid swine manure (Olivi et al., 2015).

Type of farm	Housed animals		Biofertilizer				
	Number	Category	TS	N	NH_3-N	P_2O_5	K_2O
				mg.L^{-1}	mg.L^{-1}	mg.L^{-1}	mg.L^{-1}
PPU	280	Sows	2.3	550	508	71	384
PPU	400	Sows	14.8	2.008	1,527	850	576
PPU	300	Sows	9.9	1,718	1,401	370	715
PPU	150	Sows	3.1	862	783	86	515
GFU	250	Swine	38.5	4,089	2,568	1,670	1,257
GFU	750	Swine	4.2	987	954	31	919
GFU	1,000	Swine	27.0	2,232	1,301	940	934
GFU	260	Swine	3.6	771	731	41	909
CC	150	Sows	1.7	125	94	29	447
NU	1,500	Piglets	19.4	2,376	1,843	352	1,438
Mean			13.1	1,644	1,232	435	866
Standard deviation			12.0	1,133	707	520	381

PPU: piglet producing unit; GFU: growing and finishing unit; CC: complete cycle; NU: nursery unit.

Furthermore, the use of different practices or processes for managing and treating the digestate (e.g., phase separation, composting, and drying) will also affect the availability of nutrients in the fertilizer.

A preliminary study for the construction of a biogas plant for treating different agricultural residues (swine manure, swine carcasses, poultry hatchery waste, sludge from a slaughterhouse treatment system, and poultry litter) using complete-mix biodigesters determined that two types of effluents would be produced in that plant: liquid digestate and organic compound obtained after a phase separation process of the effluent from the biodigester (Brasil, 2015; Nicoloso, 2014). The digestate and organic compound characteristics expected to be generated in the biogas plant are shown in Table 3.

Table 3. Characteristics of digestate, biodigester sludge, solid residue, and organic compound obtained under different arrangements of complete mix biodigesters in a biogas plant and upflow biodigester in a swine manure treatment plant.

Source	Fertilizer	DM (%)	N	P_2O_5	K_2O
			$kg.m^{-3}$ or $kg.ton^{-1}$ (wet basis)		
Biogas plant[1]	Liquid digestate	2.5	2.5	1.1	2.1
	Organic compound	25.0	93.0	121.0	47.0
SMTP[2]	Liquid digestate	1.0	2.0	0.6	1.1
	Biodigester sludge	6.5	5.1	7.2	1.1
	Solid residue	28.0	6.9	7.4	2.1
	Organic compound	54.5	8.5	12.1	3.3

[1]Complete mix biodigester treating a mixture of substrates (swine manure, swine carcasses, poultry hatchery waste, sludge from a slaughterhouse effluent treatment system, and poultry litter). Source: Brazil (2015) and Nicoloso (2014). [2]Swine manure treatment plant of Embrapa Swine and Poultry. Source: Nicoloso et al. (unpublished data).

The liquid digestate and organic compound expected to be generated in the biogas plant would present drastically different chemical composition and dry matter content. Similarly, high variability is observed regarding the composition of the different fertilizers obtained in a swine manure treatment plant (SMTP), where different treatment systems were installed, including rotary sieve brush, flotation-settling tank, upflow biodigester, and a composting system for the solid fraction of swine manure separated on the sieve (Table 3). Differences regarding the concentration and form in which the nutrients are available in fertilizers (organic or mineral) obtained from different treatment processes will considerably affect their agronomic efficiency, as discussed later (Nicoloso et al., 2016a). However, the logistics, cost, and feasibility of transporting and distributing fertilizers are also affected (Miele et al., 2015; Nicoloso, 2014).

The results presented here show that the high variability of nutrient content in the digestate and other organic fertilizers makes laboratory analysis essential for fertilizer characterization (Nicoloso et al., 2016a). The analysis of fertilizer will allow its application at adequate doses in agricultural areas, supplying the crop demand for nutrients without excess in the soil and avoiding environmental impacts.

Criteria for the agronomic use of digestate

Fertilizer (mineral or organic) application to the soil aims to supply the nutrient demand of crops so that they express their productive potential. Plants explore the soil through their root system in search of water and nutrients, which can be originated in the soil or come from the applied fertilizer. Thus, more fertile soils require the application of lower doses of fertilizers than soils that have lower contents of available nutrients, as fertile soils can supply higher amounts of macronutrients (N, P, K, Ca, Mg, and S) and micronutrients (B, Cl, Cu, Fe, Mn, Mo, Co, Ni, and Zn) to the plants.

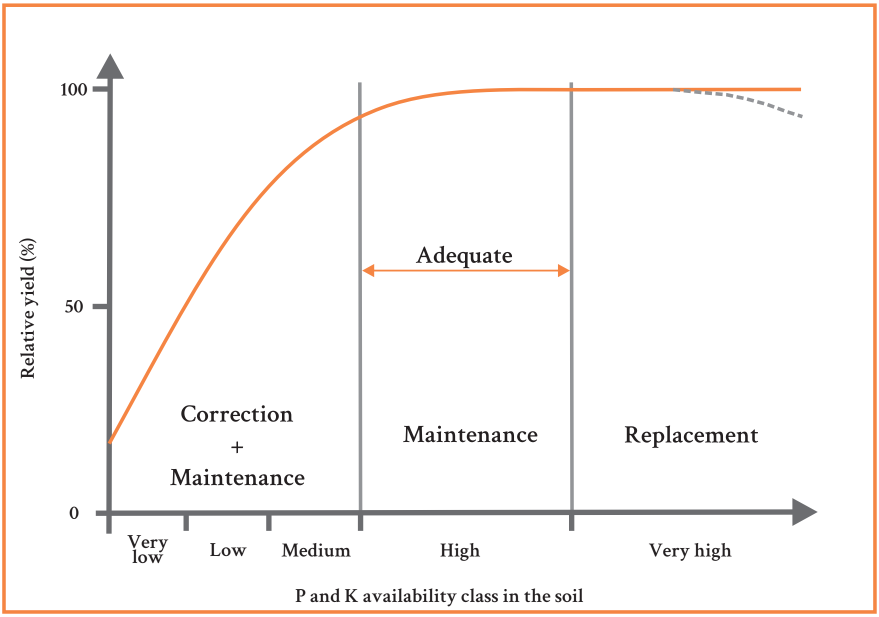
In general, fertilization recommendations aim to establish the most technically and economically efficient N, P, and K doses for different crops (Gatiboni et al., 2016). The focus on these three nutrients for fertilizer recommendation occurs because Ca and Mg are supplied through liming, S is recommended preventively for more demanding crops, and micronutrients are supplied in adequate amounts by the soil, without the need for their application via fertilizers, except under specific soil, climate, and crop conditions (Gatiboni et al., 2016). N recommendations are based on soil organic matter content and its decomposition rate, N cycling in the soil-plant system, losses of N applied via fertilizers (e.g., leaching, volatilization, and immobilization), and N demand by crops. Therefore, the construction of soil fertility in terms of N supply to plants is related to an increase in soil organic matter stocks in the long term and not directly to the application of nitrogen fertilizers. P and K fertilization recommendations are based on their availability in the soil, their losses when applied via fertilizers (e.g., adsorption and leaching), and their demand by crops. In this sense, three fertilization concepts are established for P and K recommendation, namely: corrective, maintenance, and replacement fertilization (Gatiboni et al., 2016).

Correction fertilization aims to raise P and K contents in the soil to the "critical content" of the crops (Figure 1). The critical content represents the concentration of P and K available in the soil necessary for a yield of approximately 90% of the maximum production of the crop to be fertilized. Crop yield below this critical content shows a high response to fertilization and an increase in soil P and K contents. Soils from Rio Grande do Sul and Santa Catarina present correction rates varying from 40 kg.P_2O_5.ha^{-1} to 160 kg.P_2O_5.ha^{-1} and 30 kg.K_2O.ha^{-1} to 120 kg.K_2O.ha^{-1}, according to their availability classes (very low, low, or medium) in the soil (De Bona, 2016). These doses recommended as correction fertilization have been determined only to increase soil nutrients contents, not considering that part of these nutrients is absorbed and exported by the plants. Thus, a maintenance dose must be added to this correction dose to meet the demand for P and K by crops. A significant increase in crop yield is not expected due to an increase in P and K contents in the soil above the critical level. Therefore, maintenance fertilization aims only to add the amounts of P and K removed by crops and exported through grains, forage, or biomass and also replace the losses of these nutrients in the soil, keeping the P and K contents stable in a range considered suitable for crop development ("high" nutrient availability class). On the other hand, replacement fertilization aims to add the amounts of P and K exported by crops and is recommended for soils with contents classified as "very high". Applying only the prescribed replacement doses can result in a reduction of P and K contents in the soil over time due to the nutrient losses that are likely to occur. Table 4 shows the amounts of N, P_2O_5, and K_2O suggested for maintenance and replacement fertilizations of the main grain crops grown in Rio Grande do Sul and Santa Catarina (De Bona, 2016).

Table 4. Maintenance fertilization and phosphorus and potassium replacement for the main grain crops grown in the Rio Grande do Sul and Santa Catarina States (adapted from De Bona, 2016).

Crop	Reference yield[1]	Maintenance fertilization[2]			Replacement fertilization[3]		
	ton.ha^{-1}	N (SOM 2,6%-5%)[4]	P_2O_5	K_2O	N	P_2O_5	K_2O
		---------------- kg.ha^{-1} ----------------			----------- kg.ton^{-1} --------		
Canola	1,5	=40+20*(EY-RY)	=30+20*(EY-RY)	=25+15*(EY-RY)	20	15	12
Barley	3,0	=40+30*(EY-RY)	=45+15*(EY-RY)	=30+10*(EY-RY)	20	10	6
Sunflower	2,0	=40+20*(EY-RY)	=30+15*(EY-RY)	=30+15*(EY-RY)	25	14	6
Corn	6,0	=70+15*(EY-RY)	=90+15*(EY-RY)	=60+10*(EY-RY)	16	8	6
Soybean	3,0	=0	=45+15*(EY-RY)	=75+25*(EY-RY)	60	14	20
Sorghum	4,0	=55+15*(EY-RY)	=50+15*(EY-RY)	=35+10*(EY-RY)	15	8	4
Wheat	3,0	=60+30*(EY-RY)	=45+15*(EY-RY)	=30+10*(EY-RY)	22	10	8

[1]Reference yield is the minimum yield considered in this fertilization recommendation system. [2]The maintenance fertilization for phosphorus and potassium is calculated as a function of an additional maintenance dose to be applied according to the desired expected yield (EY) relative to the reference yield (RY). [3]Replacement fertilization as a function of nutrient export for each ton of grain produced. [4]Nitrogen fertilization for soils with organic matter (SOM) contents between 2.6% and 5%, medium-yielding predecessor crop.



Source: Adapted from Gatiboni et al. (2016).

Figure 1. Relative yield of crops as a function of P and K content in the soil and indications for correction, maintenance, and replacement fertilizations.

The data shown in Table 4 allow determining the available amounts of nutrients to be applied to the aforementioned crops considering the expected yield projected with fertilization. However, organic fertilizers may have reduced efficiency compared to mineral fertilizers because part of the nutrients is in forms unavailable to plants (Nicoloso et al., 2016a). In general, organic fertilizers with a higher proportion of nutrients in the organic form and high lignin and fiber contents have a lower decomposition rate in the soil and, therefore, a lower release and availability of nutrients for plants. For instance, poultry litter has an agronomic efficiency index for nitrogen of 0.5% or 50% (Table 5). It means that only 50% of the total N content present in the fertilizer will be available for the 1st cultivation after application to the soil (immediate effect). However, poultry litter still has a residual effect of 20% for N, which will be available for the subsequent crop (2nd cultivation). Table 5 lists the agronomic efficiency indices of some organic fertilizers often available in regions of intensive animal production.

Table 5. Mean values of nutrient efficiency of different organic fertilizers applied to the soil in two successive cultivations (Nicoloso et al., 2016a).

Fertilizer	Cultivation	Nutrient[1]		
			P	K
Poultry litter	1st cultivation (immediate effect)	0.5	0.8	1.0
	2nd cultivation (residual effect)	0.2	0.2	0.0
Swine slurry	1st cultivation	0.8	0.9	1.0
	2nd cultivation	0.0	0.1	0.0
Cattle slurry	1st cultivation	0.5	0.8	1.0
	2nd cultivation	0.2	0.2	0.0
Organic compost from swine manure[2]	1st cultivation	0.2	0.7	1.0
	2nd cultivation	0.0	0.3	0.0

[1]Total nutrients (mineral + organic). [2]Considering shavings and/or sawdust as substrate.

The organic fertilizer dose to be applied to the soil must consider the specific recommendations for the different classes of soil fertility, crop demand and expected yield, and content and agronomic efficiency index of the fertilizer to be used, being calculated according to the equations described below (Nicoloso et al., 2016a):

Solid fertilizers

$$\boldsymbol{A} = \frac{\boldsymbol{QD}}{\left(\left(\frac{\boldsymbol{B}}{\boldsymbol{100}}\right) \times \left(\frac{\boldsymbol{C}}{\boldsymbol{100}}\right) \times \boldsymbol{D}\right)}$$

Equation 1

Liquid fertilizers

$$\boldsymbol{A} = \frac{\boldsymbol{QD}}{(\boldsymbol{C} \times \boldsymbol{D})\boldsymbol{A}}$$

Equation 2

Where:

A = Organic fertilizer dose to be applied to the soil ($kg.ha^{-1}$ for solids and $m^3.ha^{-1}$ for liquids)

B = Dry matter content of the solid organic fertilizer (%)

C = Concentration of N, P_2O_5, or K_2O in the organic fertilizer (% for solids and $kg.m^{-3}$ for liquids)
D = Fertilizer agronomic efficiency index. The term "B/100" can be eliminated from the equation for solid fertilizers in which the nutrient content is expressed on a wet basis.

Considering, for example, the mean data of nutrient concentration shown in Table 2 to calculate the amount of digestate (considering an efficiency index similar to swine manure, as shown in Table 5) to be applied for maintenance fertilization in the corn crop with a productivity expectation of 12 tons per hectare (Table 4), we can use Equation 2 as described below:

a) To meet the demand for N: A = 160/1.6 x 0.8 = 125 $m^3.ha^{-1}$.
b) To meet the demand for P_2O_5: A = 180/0.4 x 0.9 = 500 $m^3.ha^{-1}$.
c) To meet the demand for K_2O: A = 120/0.8 x 1.0 = 150 $m^3.ha^{-1}$.

The option for the highest dose (500 $m^3.ha^{-1}$) to meet the demand for P_2O_5 would result in an excessive application of 480 $kg.N.ha^{-1}$ and 280 $kg.K_2O.ha^{-1}$, which should be avoided to mitigate possible environmental impacts, especially related to nitrate and potassium leaching, ammonia volatilization, and nitrous oxide emission (Aita et al, 2014). In this case, the technically correct option would be to opt for the lowest dose (125 $m^3 \cdot ha^{-1}$) to meet the demand for N by the corn crop and complement the fertilization with P and K using another source of mineral fertilizer (Nicoloso et al., 2016a). Table 6 shows the results of an experiment of four growing seasons of corn fertilized with different sources of fertilizers (mineral, liquid swine manure, swine manure digestate, organic compound from swine manure, and control without fertilization) in a Nitisol (26% clay) under no-tillage and conventional tillage systems (Nicoloso et al., unpublished data). In this experiment, the total N dose applied to all treatments was 140 $kg.N.ha^{-1}$ only in the corn crop (spring/summer). P and K applications were carried out to meet the corn demand, according to De Bona (2016).

Tabela 6. Nitrogen absorption, biomass production, and grain yield of corn (accumulated from four growing seasons) fertilized with mineral fertilizer, liquid swine manure, swine manure digestate, and organic compound from swine manure under conventional and no-tillage systems (adapted from Nicoloso et al., unpublished data).

Parameter	Tillage system	Fertilization					Mean
		CTR	MIN	SS	SMD	COMP	
		---------- $kg.ha^{-1}$ ----------					
Nitrogen	CT	599	759	751	741	647	700 ns
	NT	536	680	782	711	583	659
	Mean	567 b[1]	719 a	766 a	726 a	615 b	679
Biomass	NT	43.716	47.548	50.620	49.890	46.652	47.685 ns
	PD	39.808	46.794	51.004	49.901	43.393	46.180
	Mean	41.762 c	47.171 ab	50.812 a	49.895 ab	45.023 bc	46.932
Grains	NT	32.108	35.158	37.198	36.292	33.756	34.902 A
	PD	28.477	33.754	36.952	36.538	30.092	33.163 B
	Mean	30.293 d	34.456 bc	37.075 a	36.415 ab	31.924 cd	34.032

CTR: control without fertilization; MIN: mineral fertilization; SS: liquid swine slurry swine manure; SMD: swine manure digestate; COMP: organic compound from swine manure; CT: conventional tillage; PD: no-tillage; ns: not significant. [1]Means followed by the same lowercase letter in the row and uppercase letter in the column do not differ from each other by t-test ($p<0.05$).

No differences were observed between tillage systems for N accumulation and biomass production in the corn crop. However, corn grain yield was higher in conventional tillage areas due to a higher mineralization rate of soil organic matter induced by soil tillage. N accumulation and biomass production in corn were similar between treatments that received mineral fertilizer (urea), liquid swine manure (SS), and swine manure digestate (SMD). Grain yield was higher in the treatment that received SS than in the treatment with mineral fertilizer. The SMD treatment had intermediate productivity, not differing from each other. The treatment that received organic compost (COMP) had lower N accumulation, biomass production, and corn grain yield than the other treatments, indicating the lower N availability of this fertilizer (Nicoloso et al., 2016a) (Table 5). These results show that digestate and other organic fertilizers can efficiently and safely replace mineral fertilizers when the technical criteria set out here are observed, reducing the production costs in agriculture (Miele et al., 2015).

Requeriments of agricultural areas for digestate recycling

The dimensioning of the agricultural area necessary for the disposal of effluents from a biodigester combines the concepts discussed earlier in this chapter, namely: nutrient supply by the digestate and nutrient demand in the agricultural area. These same principles allow performing the reverse calculation to dimension the substrate offer and the biodigester size as a function of the agricultural area available for digestate recycling. This analysis is valid for small biodigesters operating on rural properties and large-scale biogas plants. However, this dimensioning must be carried out considering both factors (nutrient demand and supply) in the long term.

As previously discussed (Figure 1), correction fertilization aims to increase soil nutrient contents (P and K) to adequately supply the crop demand, reducing fertilizer consumption. However, only maintenance fertilization is used when the critical nutrient content in the soil is reached, keeping the crop productivity close to the productive potential, and replacing the loss of nutrients in the soil. In this sense, the recommendation for maintenance fertilization is the dose to be used for dimensioning the demand for nutrients to keep the soil nutrient contents stable and the enterprise sustainable in the long term (Nicoloso and Oliveira, 2016). The option for dimensioning considering the recommendations for soil fertility correction would cause the gradual and excessive accumulation of nutrients in the soil, with negative effects on the environment over time. Similarly, the digestate supply dimensioning according to the replacement recommendations would promote a reduction in soil fertility and the need for the additional input of mineral fertilizers, as these recommendations do not predict soil nutrient loss. Thus, the dimensioning of nutrient supply and demand can be determined from the following equation (adapted from Nicoloso and Oliveira, 2016):

$$\boldsymbol{\Sigma}\left[\boldsymbol{NS}\times\frac{(\mathbf{100}-\boldsymbol{L})}{(\mathbf{100}\times\boldsymbol{AE})}\right]=\mathbf{\Sigma ND}-\mathbf{\Sigma NSM}$$

Equation 3

Where:

NS = Mean annual nutrient supply (N, P_2O_5, or K_2O) in the substrates that will feed the biodigester, plant, or enterprise under analysis (kg.year^{-1}).

L = Nutrient losses (N, P_2O_5, or K_2O) that occur during the biodigestion process, treatment, and storage of substrates and effluents (%).

AE = Agronomic efficiency index of nutrients (N, P_2O_5, or K_2O) of each effluent.

ND = Mean annual nutrient demand (maintenance recommendation for N, P_2O_5, or K_2O) in agricultural areas available for recycling effluents from the biodigester, plant, or enterprise under analysis (kg.year^{-1}).

NSM = Mean annual nutrient supply from mineral sources or other organic sources used in the fertilization of agricultural areas available for recycling effluents from the biodigester, plant, or enterprise under analysis (kg.year^{-1}).

The determination of the average annual demand for nutrients in agricultural areas receiving digestate and other liquid effluents and solid waste generated by the biodigester, plant, or enterprise under analysis considers the used crop system, which normally varies over the years. Thus, the ideal is to carry out long-term planning (>4 years) for fertilizer use (Fatma, 2014). Another important factor is to determine which nutrient (N, P_2O_5, or K_2O) will be used as a limiting factor for the dimensioning. Usually, P or N is used as a limiting nutrient, as K has little relevance from an environmental point of view for most residues. The sugarcane vinasse is an exception due to the high K concentration compared to other nutrients in this residue (Soares et al., 2014). P is used as a limiting nutrient for residues of animal origin (e.g., swine manure), as its supply in this type of residue meets the demand for this nutrient in most crops, without promoting an excessive supply of N or K in the soil (Fatma, 2014; Nicoloso and Oliveira, 2016).

Environmental limits for digestate application

Excessive fertilizer application may cause significant environmental impacts regardless of their mineral or organic origins, especially due to increased nutrient loss in the soil and the transfer to the environment (Aita et al., 2014; Escosteguy et al., 2016; Soares et al., 2014). Thus, numerous research initiatives have been seeking to establish indicators and critical environmental limits (CELs) of nutrient availability in the soil to reduce environmental pollution risks. CELs can be considered as indicator values of soil quality that impose limits on fertilizer application to the soil. In this sense, CELs can be used by regulatory and supervisory agencies to establish maximum acceptable doses or even prohibit the application of any source of nutrients to the soil, including digestate, agro-industrial residues, or mineral fertilizers. However, CELs cannot be confused with soil nutrient availability classes determined for fertilization purposes (Gatiboni et al., 2016), as not always soil nutrient contents classified as "very high" from an agronomic point of view (Figure 1) indicate a potentially deleterious effect on the environment (Escosteguy et al., 2016).

Although N is one of the most studied nutrients due to its high potential for environmental impact derived from its rapid transformations and losses in the soil, there are currently no CEL indicators in Brazil relating the concentrations of this nutrient in the soil to the environmental pollution risk. Moreover, we need to consider that more than 90% of the N in the soil is associated with SOM and, therefore, the total N contents are not good indicators of environmental risk. Initiatives to establish CEL for N are, therefore, based on the most abundant reactive forms of this nutrient, such as N in the form of nitrate. The Water Protection Act (2008) in Canada (province of Manitoba) establishes that nitrogen fertilization should be planned so that the residual amount of NO_3 (nitrate) in the 0 cm–60 cm soil layer at the end of the crop cycle is not higher than 33 kg ha^{-1} to 157 $kg.ha^{-1}$, according to land use classes. In Europe, the Nitrates Directive 91/676/EEC does not set limits on nitrate in the soil, but it prohibits the waste or manure application during the winter and limits the doses of these residues to up to 170 kg to 250 kg of $N.ha^{-1}$, according to the country, in areas identified as vulnerable to groundwater contamination by this nutrient. The purpose of this legislation is to ensure that the nitrate content in groundwater and surface water in these regions does not reach the critical limit of 50 $mg.L^{-1}$ (van Grinsven et al., 2012). In Brazil, Conama Resolution 420/2009, based on Ordinance 518/2004 of the Ministry of Health, establishes the limit level of nitrate in groundwater at 10 $mg.L^{-1}$ (Brasil, 2009). This value should not be confused as a limit for nitrate concentration in the soil solution. Moreover, member countries of the European Union have also established national programs to control air pollution to reduce ammonia and nitrous oxide emissions from agricultural sources (Loyon et al., 2016). These programs are based on the adoption of good management practices and nitrogen fertilizer application, such as acidification and injection of liquid waste into the soil, incorporation of manure and solid mineral fertilizers, use of urease and nitrification inhibitors, split application, irrigation control, and verification of climate and soil conditions at the time of application (Unece, 2014).

Gatiboni et al. (2015) performed a first approximation to establish critical environmental limits for P (CEL-P) in soils that receive frequent organic residue applications. The developed method allows calculating the maximum available P content that can exist in the soil without high risks of its transference to the environment, considering the soil clay content. The calculation equation is described below:

$$\boldsymbol{CEL - P = 40 + \%Clay}$$ *Equation 4*

Where:

CEL-P = Maximum available P content determined by the Mehlich-1 method (mg. dm^{-3}) that soil can present without a high risk of pollution

%Clay = Soil clay content expressed as a percentage. This indicator has been adjusted and is only valid for the 0 cm–10 cm soil layer

According to the proposed method, sandy soils are more sensitive, whereas clayey soils can support higher amounts of P without making them available in large amounts to the environment. Briefly, the soil is considered a safe reservoir of P when its contents are below the CEL-P, even if these contents are classified as “very high” relative to P availability for crops (Gatiboni et al., 2016). However, soil can become a P source for the environment when its contents exceed this limit value, promoting the eutrophication of surface water reservoirs when lost from agricultural areas, mainly by runoff. This methodology is currently used by the Environmental Foundation of the State of Santa Catarina (Fatma, 2014) to classify the environmental risk of soils with the application of swine manure. However, the authors emphasize that the method is an incipient proposal and lacks a more intense field calibration and the inclusion in the model of factors other than soil texture, such as terrain slope and soil conservation practices, which can also affect soil P losses.

Although K is not considered a nutrient with high potential for environmental impact in most situations, the application of high doses of sugarcane vinasse or other effluents containing high K concentrations may promote excessive K accumulation, affecting soil and water quality. The excessive K accumulation in the soil in areas where vinasse is recycled as fertilizer can impair Ca absorption, promoting its deficiency in the plant (Vitti; Mazza, 2002) and soil salinization in extreme situations through the concomitant supply of Na and Cl by this effluent (Soares et al., 2014). The increase in soil K contents also causes its higher mobility in the soil profile and higher contamination risks of the water table. The consumption of water with high K contents can promote metabolic diseases in individuals with renal dysfunction (Rocha, 2009). The Environmental Company of São Paulo State established limits for vinasse application based on the K saturation in the soil cation exchange capacity (CEC) and the capacity to extract and export this nutrient by crops

(Cetesb, 2006). According to "Technical Standard P4.231 – Vinasse: criteria and procedures for application to agricultural soil", a maximum of 5% of the CEC can be occupied by K, considering the 0 cm–80 cm layer of soil depth.

Other elements, especially micronutrients and heavy metals, do not present a large number of regionalized studies in Brazil establishing CELs. However, Conama resolution 420/2009 establishes soil quality guiding values regarding the presence of some trace elements (Cd, Pb, Co, Cu, Cr, Hg, Ni, Zn, and V) for the entire Brazilian territory (Brasil, 2009). Despite this, these values need to be validated regionally both for the definition of quality reference values (QRV), indicating the natural abundance of a certain element in the soil without anthropogenic influence, and for CEL establishment. A survey carried out to define QRVs in soils of the plateau region of the state of Rio Grande do Sul found higher values for Co, Cu, Cr, and Ni than the prevention (PRV) and investigation reference values (IRV) indicated by Conama resolution (Fepam, 2014). These data reinforce the need for the development of regionalized CELs, especially for micronutrients or trace elements, which present high variability according to the type of material that originated the soil.

Environmental indicators of soil quality, such as CEL-P and others, aims to establish limits and guide the rational use of fertilizers in a technically correct and environmentally safe manner. The indiscriminate disposal of digestate or other agro-industrial residues directly on the soil, although accepted in the past (Decree-Law 303/1967; Brasil, 1967) is currently an inadmissible practice due to immediate and cumulative environmental impacts. The modernization of the environmental legislation in Brazil and other countries has advanced in this direction, requiring environmental licensing of areas where agro-industrial residues are applied according to the size of the enterprise (Cetesb, 2006; Fatma, 2014). The environmental licensing process includes the preparation of an environmental impact study and report, planning for residue recycling in available agricultural areas, and soil quality monitoring based on CELs and specific quality standards for each type of agro-industrial activity.

Mitigation of greenhouse gases due to the agronomic use of digestate

In the agricultural sector, greenhouse gas (GHG) mitigation strategies can be summarized as: (a) reduction of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions; (b) replacement of GHG emissions from fossil fuels by renewable energy sources; and (c) atmospheric CO_2 sequestration by photosynthesis and its storage in stable or slow cycling compartments in the global C cycle (Smith et al., 2007). It is noteworthy that CH_4 and N_2O have a global warming potential (GWP) 28 and 265 times higher than CO_2, respectively (Myhre et al., 2014). Ammonia is not considered a GHG, but it can also indirectly affect N_2O emission during and after its nitrification when it returns to the soil (Singh et al., 2008).

Worldwide, the agricultural sector has the potential to offset approximately 10% of anthropogenic GHG emissions at their current levels, while in Brazil it can reach from 20% to 30% of the country's GHG emissions (Bayer, 2007). It is estimated that 89% of the technical potential for GHG mitigation in this sector is related to soil C sequestration, 9% is associated with the reduction of CH_4 emissions (flooded rice, ruminant management, treatment of waste and agro-industrial residues), and 2% is dependent on the reduction of soil N_2O emissions through the management of nitrogen fertilization (Smith et al., 2007).

Biodigesters and composting are currently the most widespread technologies to treat swine manure in Brazil (Kunz et al., 2009). Biodigesters have good potential for GHG mitigation, as CH_4 produced by the anaerobic decomposition of manure and other organic residues can be converted into CO_2 by controlled biogas burning (Kunz et al., 2009). In this sense, the ABC Plan (Low Carbon Emission Agriculture) of the Brazilian Government provides for the treatment of 4.4 million tons of manure through biodigestion or composting by 2020 (Barros et al., 2015) as one of the strategies for Brazil to meet the GHG emissions mitigation commitments (Intended Nationally Determined Contributions – iNDC) submitted to the Paris Agreement (Brasil, 2016).

However, GHG mitigation strategies employed in the agricultural sector can affect more than one GHG by more than one mechanism in processes that can even be opposed. Thus, the net benefit of adopting these strategies must be assessed by the combined effect on all GHGs (Robertson and Grace, 2004; Schils et al., 2005; Koga et al., 2006). Furthermore, the effect of a mitigation strategy can vary in time differently between GHGs: some can be mitigated indefinitely, while others are temporarily affected (Six et al., 2004; Marland et al., 2003). Thus, the GHG emissions that occur after the biodigestion or composting process, when the organic compound, digestate, sludge, and other effluents from biodigesters are applied to the soil as fertilizers, need to be considered regarding the treatment of manure and other agro-industrial residues.

The application of animal manure and other organic residues to the soil, especially those rich in ammoniacal nitrogen, is expected to accelerate the decomposition (and CO_2 emission into the atmosphere) of crop residues (N-poor grass straw). However, Aita et al. (2006) did not observe this effect when adding liquid swine manure to black oat crop residues (C/N = 44/1). In this case, the oat straw did not show a sufficiently high C/N ratio and, therefore, the microbial population did not need external mineral N for the decomposition of crop residues. Moreover, the authors reported that the occurrence of rain after manure distribution on crop residues may have transported the ammoniacal N applied to the soil with the manure beyond the residue decomposition zone. However, Grave et al. (2015a) observed an increase in the CO_2 emissions from soil fertilized with liquid swine manure only in the first 30 days after its application. On the other hand, the soil fertilized with swine manure treated by biodigestion did not show the same increase. Therefore, this effect was attributed to the decomposition of C applied to the soil by manure and not to the decomposition of crop residues (wheat straw) present in the soil. Field experiments have shown, in some situations, only an initial peak in CH_4 emission in the first hours after manure application, which has been attributed to CH_4 that is dissolved in the effluent (Sherlock et al., 2002). Thus, the application of organic fertilizers, especially those treated by biodigestion, has a limited effect on the increase in soil CO_2 and CH_4 emissions. However, these fertilizers can significantly contribute to the sequestration of atmospheric CO_2 and its stabilization as soil organic matter.

The impact of organic fertilizers on soil C sequestration rates depends on the quantity and quality of the residue to be applied. Mafra et al. (2014) observed a linear increase in soil C sequestration rates (–0.21 Mg C $ha^{-1}.yr^{-1}$ to 1.69 Mg C $ha^{-1}.yr^{-1}$) due to an increase in liquid swine manure application rates (0 $m^3.ha^{-1}.yr^{-1}$ to 200 $m^3.ha^{-1}.yr^{-1}$) on an Oxisol cultivated with corn and black oat. Although a large proportion of this increase in C sequestration rates is related to nutrient input to the soil and higher biomass production by corn and oat, another fraction can be directly attributed to C input by swine manure. However, residues characterized by a higher proportion of recalcitrant C and slowly decomposing in the soil, such as residues that undergo composting (Grave et al., 2015a), may have a higher impact on soil C accumulation. Nicoloso et al. (2016b) observed that C sequestration rates in a Chernozem cultivated with corn and fertilized with liquid cattle manure increased significantly when the fertilizer source was replaced by organic compound generated from cafeteria waste, considering the same N input to the soil from both sources. Conversely, the treatment of agro-industrial waste and residues by biodigestion can reduce C content in the digestate and limit soil C sequestration rates. Grave et al. (2015a) observed that the treatment of liquid swine manure by biodigestion reduced C input to the soil by approximately 50% compared to untreated manure. After three years of application of different sources of organic fertilizers for corn (140 $kg.N.ha^{-1}$), these authors did not observe significant differences between C stocks in soil fertilized with digestate and mineral fertilizers (unpublished data). Thus, the possible increase in C stocks in soil fertilized with digestate and other effluents containing low C contents can be attributed mainly to the input of nutrients and improvement in soil fertility rather than to a direct C input by the organic fertilizer.

Several biotic and abiotic processes are involved in the N_2O production and emission in agricultural soils. Heterotrophic and autotrophic nitrification, nitrification coupled with denitrification (different microorganisms), denitrifying nitrification (same microorganism), and denitrification are the main biological processes that control N_2O emissions in aerated soils (although under partial O_2 availability) (Butterbach-Bahl et al., 2013). These processes are mainly controlled by pH, temperature, moisture, oxygen diffusion, and soil C and N availability (Giles et al., 2012). Therefore, soil management and fertilizer application play a major role in regulating the substrate availability for these processes and, consequently, soil N_2O emissions. The increased soil moisture promotes a reduction in oxygen diffusion (e.g., 65%–70% of the porosity filled by water) and an increase in soil nitrate (NO_3-) concentrations prevents its complete denitrification into N_2, contributing to N_2O accumulation as an intermediate metabolite (Panek et al., 2000; Giles et al., 2012).

In this sense, animal manure, especially liquid and with high availability of ammoniacal N and labile C, may favor soil N_2O emissions compared to mineral fertilizers, as observed in different soil and climate situations (Rochette et al., 2004; Perälä et al., 2006; Chantigny et al., 2010; Damasceno, 2010; Schirmann, 2012). This effect of manure on the increase in N2O emissions is attributed to several causes, especially the following:

a) Manure adds labile C to the soil, which is used for biomass and energy production by denitrifying bacteria and other heterotrophic soil microorganisms, reducing O_2 availability through its respiratory activity.

b) The liquid fraction applied to the soil with manure, composed of a mixture of water and urine, also contributes to reducing O_2 availability, an essential condition for N_2O emission through nitrification and denitrification.

c) Ammoniacal N from manure is rapidly nitrified in the soil, which, associated with the reduced O_2 availability, can result in N_2O emission during nitrification and denitrification when the produced NO_3 can be used as an alternative to O_2 in the respiratory chain of denitrifying bacteria.

In addition to these effects attributed to manure on favoring N_2O emissions, other additional factors inherent to no-tillage can contribute to increasing these emissions. The reduction in macroporosity, the soil compaction due to the movement of machines, and moisture preservation are characteristics of no-tillage, which, alone or together, can reduce soil O_2 availability, favoring denitrification. Moreover, soil organic matter (SOM) accumulation and the presence of crop residues in the no-till system increase C availability to heterotrophic bacteria, responsible for denitrification. Thus, animal manure treatment using biodigestion has been an efficient technology to reduce N_2O emissions from soil managed under the no-tillage system (Table 7).

Table 7. Accumulated N_2O emissions (64 days) from a Nitisol fertilized with organic fertilizers under no-tillage and conventional tillage system (Grave et al., 2015b).

Fertilization	Tillage system		t-test (p-value)
	Conventional	No-tillage	
	------------------ $kg.N_2O.ha^{-1}$ ------------------		
CTR	1,42 ± 0,18	1,85 ± 0,73 c[(1)]	0,948
MIN	1,87 ± 0,72	3,52 ± 0,65 ab	0,120
SS	2.55 ± 0.51 B	5.60 ± 1.38 A a	0.050
SMD	2.10 ± 0.40	2.94 ± 1.18 bc	0.606
COMP	1.56 ± 0.13 B	4.67 ± 1.70 A ab	0.017
Teste t (valor p)	0.443	0.004	-

CTR: control without fertilization; MIN: mineral fertilization (urea); SS: swine slurry; SMD: swine manure digestate; COMP: compound from swine manure. 1Means ± standard error (n=4) followed by the same lowercase letter in the column or uppercase letter in the row do not differ from each other by the t-test ($p<0.05$).

The accumulated N_2O emission at 64 days after the application of different sources of fertilizers was higher in the soil managed under the no-tillage system than in the soil submitted to conventional tillage, especially in areas fertilized with liquid swine manure (LSM) without treatment or submitted to composting (COMP) (Grave et al., 2015b). The authors attributed these results to the higher soil moisture content under no-tillage, as N availability and soil labile C contents did not vary between tillage systems. SS application to the soil under the no-tillage system increased N_2O emissions by 59% compared to the soil fertilized with urea (MIN) due to the input of labile C to the soil, which favored the proliferation of denitrifying microorganisms under high moisture and NO_3 availability conditions. These factors prevented the complete denitrification of NO_3 into N_2, resulting in N_2O accumulation as an intermediate metabolite and its emission into the atmosphere. As expected, the treatment of manure by biodigestion (SMD) or composting (COMP) limited the input of labile C and mineral N to the soil, reducing N_2O emissions into the atmosphere by 47% and 17% compared to the soil under no-tillage system and fertilized with SS.

These results are especially relevant for Brazilian agriculture, as Brazil has one of the largest cultivated areas under the no-tillage system in the world (Febrapdp, 2016). In this sense, the treatment of swine manure by biodigestion or composting and its recycling as sources of nutrients for agriculture contribute to the potential of GHG mitigation in the Brazilian agricultural sector by increasing C sequestration rates and mitigating soil N_2O emissions. However, for the potential of these technologies to be fully evaluated, it is essential that the GHG mitigation verified during the treatment of manure and other organic residues, which is currently accounted for in the ABC Plan, is also added to those observed in agricultural areas used for recycling organic fertilizers from different treatment systems

Final remarks

Technologies for the management of agricultural and agro-industrial residues have evolved significantly in recent decades. This evolution was followed by an increase in size and scale of production in rural properties and agribusinesses, providing alternatives for an environmentally adequate destination of residues generated by these activities in response to increasingly restrictive environmental legislation. In this sense, recycling organic residues as a source of nutrients for agriculture has proven to be a technically and economically viable alternative. However, this practice must follow the fundamental principles of fertilizer management and soil fertility already established and constantly refined by research. Failure to comply with these principles and the inadvertent disposal of these residues directly on the soil is a waste of nutrients from both an agronomic and economic point of view and can promote severe environmental impacts. Therefore, the establishment of environmental limits and their adoption by regulatory agencies as references for licensing processes and environmental monitoring is complementary to agronomic recommendations for fertilizer applications to ensure soil and environment quality conservation. Organic residues when properly managed constitute a safe source of nutrients for agriculture that can efficiently replace mineral fertilizers, with positive impacts on the environment and contributing to the economic viability of agricultural and agro-industrial enterprises.

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