



Carbon mineralization from composts and food industry wastes added to soil

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Abstract

We have studied the short term C mineralization of six wastes from important food industries, one sludge from a biogas plant and three composts. All the wastes were characterized chemically and fractionated according to the Van Soest method. The fresh wastes were incubated under controlled environment conditions to determine the C mineralization rate. Based on first order mineralization kinetics, we calculated the hypothetical amount of stable C in the wastes as the amount of C that would not be mineralized within one year under field conditions. The percentage of stable organic C in the organic matter was in general much larger in the composts than in the other wastes, but when expressed on dry matter, the non-composted wastes had comparable or larger amounts of stable organic C than the composts and have a considerable potential for supplying organic matter to soils, and hence for C sequestration. The amount of stable organic C could best be predicted by the total N content of the wastes ($R_a^2 = 0.855$), whereas the results of the fractionation had very little predictive power, probably due to problems related to the high ash content of some of the wastes. An index that combined stable organic C and N and P content in the wastes was calculated to assess possible limitations for applying these wastes in agriculture. Under current nutrient legislation in Western Europe, a number of these wastes will only be usable in small amounts, but these and other food industry wastes could still prove to be valuable soil amendments in nutrient poor situations, for increasing soil organic C content and supplying nutrients.

Introduction

Wastes from a number of food industry plants (e.g. sugar beet processing, meat and fish processing, dairies, vegetable processing, breweries) are potentially valuable resources for use in agriculture, both for supplying nutrients and for replenishing soil organic matter. These wastes contain large concentrations of organic C (Gendebien et al. 2001) and have the advantage over other industrial wastes and sewage sludges that they contain in general very small amounts of toxic compounds (Gendebien et al. 2001). Moreover, they are at present used only to a limited extent in agriculture and are often still being disposed of by landfilling or incineration. It is expected that

landspreading on agricultural land will increase following the implementation of Community and National regulations which restrict disposal of organic-rich materials in landfills and which require treatment of organic-rich industrial effluent from different branches of the food and drink and other sectors. The addition of organic matter to soils has gained new importance recently as a consequence of worldwide environmental concerns about the elevation of atmospheric concentrations of greenhouse gasses, leading to global warming. The Kyoto protocol, aimed at reducing greenhouse gas emissions, allows in its articles 3.3 and 3.4 for accounting of atmospheric CO₂ removal through C sequestration in soils. Smith et al. (2000) suggested that Europe could meet its Kyoto commit-

ment by implementing a number of agricultural land management options, including the use of straw, manures and sewage sludges. However, some of these materials are limited in their availability, e.g. straw and farmyard manure, or pose environmental problems such as the large concentrations of toxic compounds in sewage sludge (Lerch et al. 1992). In view of this, food industry wastes have potential to be used as a substitute for other organic resources.

There exists a considerable amount of experimental data on C mineralization and the amount of stable organic matter in pure chemical compounds such as glucose and (hemi)cellulose (e.g. Mary et al. 1993; Sørensen 1983; Voroney et al. 1989) or materials such as crop residues (e.g. De Neve et al. 1996; Roper 1985; Ladd et al. 1981), composts (e.g. Beloso et al. 1993; Vandervreken 1996; Pascual et al. 1998), manure (e.g. Sørensen and Jensen 1995; Baumann and Schmidt 1987) or sewage sludge (e.g. Bernal et al. 1998; Lerch et al. 1992). However, very little is known about the above mentioned food industry wastes in terms of carbon mineralization and the stability of the organic carbon they contain. Moreover, one should take into account the nutrient content in these wastes, as high concentrations of notably N and P might put restrictions on their use in a number of countries.

In this study we have assessed the carbon mineralization and the amount of stable organic carbon in wastes from the food industry and composts upon application to soil, and the possible constraints in the use of these wastes in agriculture as imposed by their nutrient content.

Materials and methods

We selected six wastes from the food industry, according to their relative importance for Belgian conditions based on annual amounts produced (Gendebien et al. 2001). The sludges were obtained from a dairy industry plant (DAI), a brewery (BRE), an abattoir (ABA), a gelatine production plant (GEL), a fruit processing plant (FRU) and a potato processing plant (POT). The sludges were collected before stabilization, i.e. not dried and without addition of CaCO_3 . Additionally, we selected the sludge (FER) from a biogas plant, resulting from the anaerobic fermentation of a mixture of organic waste from food industries, fruit and vegetable waste compost and pig slurry. We also included three composts, namely green

waste compost (GWC), vegetable, fruit and garden waste compost (VFG), and spent mushroom compost (SMC). The composts were also included in the experiment as a kind of reference materials, because more is known about their stability and C mineralization behaviour than about the other organic wastes. The chemical characteristics of the wastes are summarized in Table 1. Dry matter was determined by oven drying at 60 °C, organic C content was measured by wet oxidation at elevated temperature, total N was measured using a modified Kjeldahl procedure to include mineral N, nitrate and ammonia N were determined separately by steam distillation with MgO (and addition of Devarda alloy for the nitrate determination), and total P was measured colorimetrically in a HNO_3 extract of the ashed materials through complexation with nitrovanadomolybdate. The organic wastes were also fractionated chemically according to the Van Soest method (Van Soest 1963) to determine the amounts of hemicellulose, cellulose and lignin.

The wastes were incubated under controlled environment conditions for measuring C mineralization. The soil used for the incubations was a sandy loam soil (3.5% clay, 32.5% silt, 64% sand) with a small organic C (0.755%) and organic N (0.049%) content, and a $\text{pH}_{\text{H}_2\text{O}}$ of 5.82. We had chosen this particular soil with small organic C content to more clearly see the C mineralization of the added organic matter. The soil was dried at room temperature until air dry and sieved on a 2 mm mesh. Variable amounts of the fresh organic wastes were mixed thoroughly with 350 g of dry soil. The amounts of organic waste incorporated varied between 10 (for VFG) and 30 g (FER), depending on the dry matter content of the wastes. The soil–waste mixture was placed in glass jars with an inner diameter of 0.104 m. The soil–residue mixture was compacted in a vertical plane using a wooden cylinder (De Neve and Hofman 2000) to obtain a bulk density of approximately 1.3 Mg m^{-3} and the water content was adjusted to 80% of field capacity (0.17 g g^{-1} soil at pF 2) by the addition of distilled water, taking into account the water content in the air dry soil and in the organic wastes added. A control treatment (soil without addition of organic wastes) was included to determine C mineralization from native soil organic matter. The initial weights of the jars were recorded. Small vials containing 15 ml 1 M NaOH solution were placed in the jars to trap evolved CO_2 . The jars were closed with airtight seals and incubated in duplicate at 21 ± 1 °C. Samples were taken regularly

Table 1. Chemical composition of the wastes.

Waste	Dry matter (% of fresh)	C _{org} (% of DM)	N _{tot} (% of DM)	N _{min} (% of DM)	C:N ratio	P _{tot} (% of DM)
GWC	51.01	17.7	0.96	0.038	18.3	0.26
VFG	79.36	14.3	1.23	0.059	11.6	0.39
SMC	32.39	35.0	2.11	N.D.	14.0	0.53
FER	1.43	29.6	3.46	0.045	10.1	1.32
BRE	5.57	25.5	2.81	0.110	9.1	1.39
FRU	4.03	38.3	5.97	0.337	6.4	1.68
POT	4.88	25.6	3.34	0.034	7.7	2.22
GEL	38.15	33.9	2.52	0.021	13.4	1.11
DAI	5.06	41.8	7.95	0.892	5.3	3.02
ABA	5.51	41.2	6.09	0.447	6.8	1.44

N.D.: not determined.

during the 39-day incubation period by removing the NaOH vials. Amounts of evolved CO₂ were measured by titration of the NaOH with 1 M HCl to pH 8.3 in the presence of BaCl₂ (Anderson 1982). The net amount of CO₂ produced from the organic wastes was determined from the difference between CO₂ evolved from the waste-amended soil and CO₂ evolved from the control soil. After removal of the vials, the glass jars were left open for 2 h to allow replenishment of oxygen. Soil moisture content was adjusted, fresh vials containing NaOH were added, and the jars were sealed again to continue the C mineralization measurements.

Results and discussion

The results of the Van Soest analysis are given in Table 2. For a number of wastes the ADF (acid detergent fiber) fraction was slightly higher than the NDF (neutral detergent fiber) fraction, and for these no data for hemicellulose are given. The reason for the negative difference (NDF – ADF) and for the

Table 2. Crude fiber analysis of the wastes (expressed in % of dry matter) according to the Van Soest analysis.

Waste	Hemicellulose	Cellulose	Lignin
GWC	–	12.42	14.11
VFG	3.30	9.41	18.61
SMC	–	16.62	19.61
FER	11.13	6.77	4.44
BRE	3.65	1.48	14.10
FRU	24.36	6.70	4.26
POT	29.41	1.59	3.40
GEL	–	0.72	1.15
DAI	35.32	4.93	17.42
ABA	20.90	18.67	4.94

–: NDF smaller than ADF.

rather large variability in hemicellulose and lignin between replicates is the high ash content in a number of wastes, which complicated the fractionation.

The organic wastes had a very diverse pattern of C mineralization (Figure 1). The composted materials, namely GWC, VFG and SMC showed a (very) small C mineralization with only between 1.8 and 8.8% of the total C being mineralized by the end of the incubation (Table 3). The other (uncomposted) organic wastes were in general less stable. DAI mineralized very fast, with over 50% of the added organic C mineralized by the end of the incubation. GEL, BRE, POT and FRU had similar percentages of C being mineralized by the end of the incubation, but in the case of FRU the leveling off of C mineralization was much less pronounced than for the first three wastes, indicating that important additional C mineralization could be expected if the incubation was to be continued for a longer period. The same is true for ABA (almost no leveling off) as compared to FER (very clear leveling off after 10 days). The variability in a number of treatments was large, which was mainly due to the fact that the wastes were incorporated as fresh matter and that a number of these wastes were not easily homogenized in fresh condition. The amounts of organic C incorporated were not the same for the different wastes, which is due to practical limitations of the incubation method. The heterogeneity of the wastes imposes a minimum on the amount of organic material that can be added to the soil, whereas the very low dry matter content of some of the wastes imposes a maximum on the amount which can be added (addition of too much material in that case would result in a too high soil water content). However, within the range of the amounts of organic C incorporated here (from 0.43 to 5.55 g kg⁻¹ soil) it was not expected that the amounts of organic C added

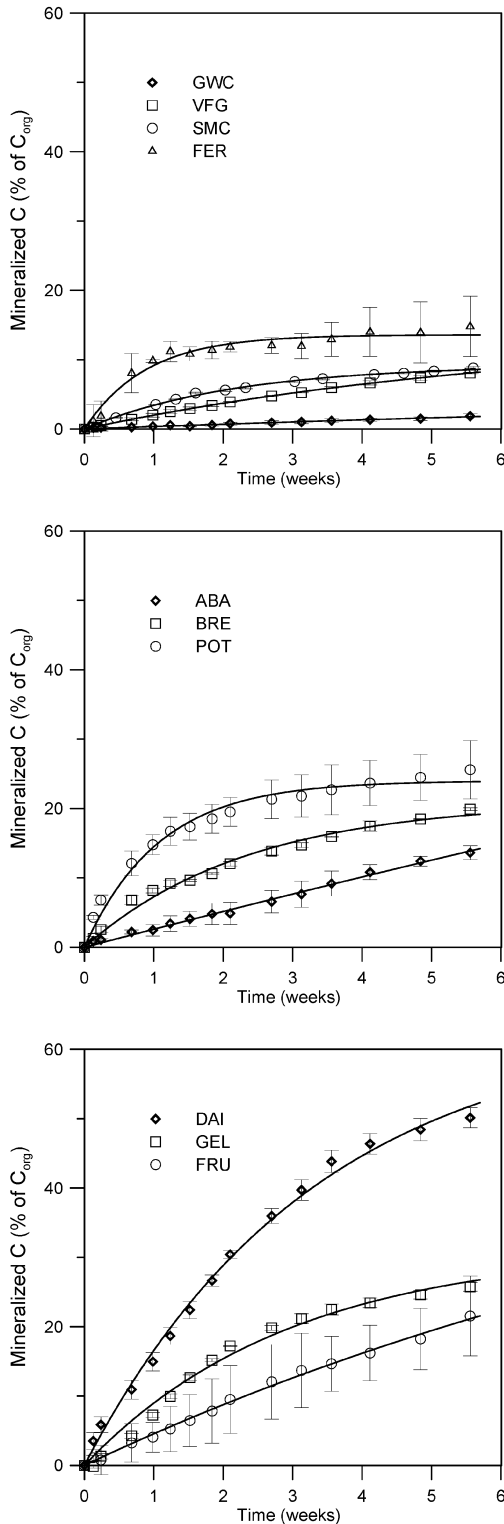


Figure 1. C mineralization during aerobic incubation of the waste materials, with the first order model (solid line) fitted to the data. Bars represent ± 1 standard deviation.

would have influenced the mineralization rates (Jenkinson 1977; Kuo et al. 1997).

A mere comparison of the amounts of C mineralized at the end of the 39-day period (which we will further refer to as OC_{39}) does not take into account the expected C mineralization in the period following the end of this incubation. In order to more objectively compare the C mineralization data between the wastes, we fitted the first order mineralization model to the net C mineralization data of the organic waste amended treatments:

$$C_{\min}(t) = OC_A(1 - e^{-kt}) \quad (1)$$

where $C_{\min}(t)$ is the amount of organic C mineralized at time t , OC_A is the amount of mineralizable C expressed as percentage of total added organic C, and k is the first order mineralization rate (Table 3). The first order model could be fitted to all treatments, but in some cases the value of OC_A will be unreliable, as indicated by the standard error on the estimated parameters. This is mainly the case in the treatments where C mineralization did not level off clearly within the time of the incubation (as for FRU and ABA). Ranking of the organic wastes according to the value of OC_A yielded a different picture than ranking according to the amounts of C mineralized after 39 days (OC_{39}), e.g. FER has a smaller value of OC_A than VFG, and ABA has the largest value of OC_A of all wastes considered, being higher than 100%. An OC_A value of more than 100% is possible, and would normally be considered as a priming effect (Dalenberg and Jager 1989; Kuzyakov et al. 2000). However, in this case the fact that the C mineralization did not level off clearly is the most probable explanation for this high value of OC_A .

To accommodate the cases where C mineralization did not level off and hence yielded very uncertain OC_A values, we considered a period equivalent to one year under field conditions. According to Hénin and Dupuis (1945) organic materials added to soil become part of the soil organic matter one year after addition to the soil, the fraction of the organic material left in the soil after one year being the humification coefficient. The fraction remaining in the soil after one year can therefore be considered to represent the stable organic matter in an organic material. This approach has been applied in recent simulation models of C sequestration in soils (Vleeshouwers and Verhagen 2001). The approximate length of an incubation carried out at 21 °C that would be equivalent to one year

Table 3. C mineralized at the end of the incubation (OC_{39}), C mineralization parameters of the first order model $C_{\min}(t) = OC_A (1 - e^{-kt})$ ($C_{\min}(t)$: amount of organic C mineralized at time t , OC_A : amount of mineralizable C expressed as percentage of total added organic C, k : first order mineralization rate) and amounts of stable organic C ($100 - OC_{21}$) in the wastes. Bracketed values are standard errors of the parameter estimates.

Waste	OC_{39} (% of added C_{org})	OC_A (% of added C_{org})	k (week ⁻¹)	$100 - OC_{21}$ (% of added C_{org})	$100 - OC_{21}$ (% of dry matter)
GWC	1.84	7.70 (5.52)	0.047 (0.037)	95.3	16.9
VFG	8.07	14.1 (0.37)	0.153 (0.005)	86.6	12.4
SMC	8.80	9.25 (0.17)	0.463 (0.019)	90.8	26.9
FER	14.8	13.6 (0.4)	1.120 (0.121)	86.4	30.2
BRE	19.9	21.1 (0.9)	0.423 (0.038)	78.9	20.1
FRU	21.5	57.3 (10.0)	0.083 (0.017)	53.6	20.5
POT	25.6	24.0 (0.6)	0.927 (0.075)	76.0	19.5
GEL	25.7	31.3 (2.1)	0.338 (0.041)	68.7	23.3
DAI	50.1	63.8 (2.6)	0.301 (0.021)	36.4	15.2
ABA	13.6	152 (173)	0.017 (0.019)	55.7	22.9

under field conditions was approximated based on the long term yearly average air temperature in Ukkel (Belgium), which is 9.7 °C, and using a temperature dependence model of C mineralization (De Neve et al. 1996):

$$k(T) = k_{\text{opt}} e^{-k(1 - \frac{T}{T_{\text{opt}}})^2} \quad (2)$$

where $k(T)$ is the C mineralization rate at temperature T , k_{opt} is the mineralization rate at optimum temperature, k is a factor expressing the temperature sensitivity of k and T_{opt} is the optimum temperature for C mineralization. An equivalent temperature model, for N mineralization, was shown to give good results for simulating N mineralization under field conditions (De Neve and Hofman 1998). The length of an incubation at 21 °C equivalent to one year under field conditions (at 9.7 °C) was calculated to be 21 weeks. We calculated the amount of organic C, OC_{21} , that would mineralize over a 21-week incubation period (equivalent to one year under field conditions) according to the first order model. The parameter OC_{21} takes into account all data of the incubation, but at the same time reduces errors resulting from the uncertainty in the estimation of OC_A for a number of organic wastes. The difference $100 - OC_{21}$ can be considered as a measure for the amount of stable organic matter in the organic wastes (Table 3), which is a very important parameter in view of the potential use of organic wastes to sequester carbon in soils. This parameter can be used directly as an input in some C sequestration simulation models (Vleeshouwers and Verhagen 2001). When expressed on dry matter basis, the amount of stable organic C in the food industry wastes is comparable to the amounts found in the composts, and is in a number of cases even considerably larger.

The actual measurement of the fraction of stable organic matter by incubations is very labour intensive and expensive. Therefore the (bio)chemical composition of the wastes as determined by standard chemical analysis (Table 1) and by the Van Soest fractionation was used in a multiple regression analysis in order to predict the amount of stable organic matter, i.e. the parameter ($100 - OC_{21}$). We used a stepwise linear regression procedure with the following criteria: the probability of the statistic F to enter a variable was set at 0.05, the probability of F to remove a variable was set at 0.1. The parameter N_{tot} alone explained a very large portion of the variance. The other parameters included in the regression analysis (hemicellulose, cellulose and lignin) explained only a very small additional portion of the residual variance, and did not improve the regression significantly (Table 4). Hence only the parameter N_{tot} was retained by the regression procedure, yielding the following relation:

$$(100 - OC_{21}) = 101 - 7.78\% N_{\text{tot}}, \quad (3)$$

$$R_a^2 = 0.864,$$

significant at $P < 0.001$. The results of the Van Soest fractionation have been used before in finding predictive relationships between chemical composition and C mineralization from organic materials (Henriksen and Breland 1999; Saviozzi et al. 1997). The fact that the results of the Van Soest analysis could not be used here to explain differences in C mineralization of the organic wastes indicates that this type of fractionation is less well suited for the particular type of wastes considered here. According to Trinsoutrot et al. (2000) relations between C mineralization and (bio)chemical composition can be obscured by lack of

available N in soil. We do not think that lack of mineral N has restricted C mineralization of the wastes studied here, as C:N ratios of the wastes were low (between 18.3 and 5.3) and considerable amounts of mineral N were added through the mineral N already present in the wastes at the time of application.

Although some of the organic wastes considered here may contribute significantly to stable organic matter build-up in soil, a number of these wastes contain large amounts of macronutrients, notably N and P, which might put a constraint on the amounts that can be applied on agricultural soils in view of the vigouring legislation in different countries. To assess the input of nutrients that would go along with the use of these wastes as sources of organic matter, we have calculated the amount of organic waste dry matter and nutrients that would have to be applied to supply 1 ton of stable organic matter (the stable organic matter referring to the parameter $100-OC_{21}$), and the amount of nutrients this represents (Table 5). From these calculations it is clear that mainly the P content of a number of these wastes may put severe restrictions on their use. Notably DAI and POT waste application could result in excessive P supply to soil. As an example, in Flanders (Northern part of Belgium) the fixed limits for N and P application to arable land are between 200–250 kg N ha⁻¹ yr⁻¹ and 45–55 kg P ha⁻¹ yr⁻¹, respectively. In order to weigh up the C sequestration potential and the nutrient supply upon application of these wastes, we calculated the following index, which takes into account that (under Flemish regulations) P is about 5 times more restrictive than N:

Table 5. Amounts of waste dry matter and nutrients (N and P) that would have to be applied to soil to supply 1 ton of stable organic C per ha, and value of the index (Equation 4).

Waste	Dry matter (tons ha ⁻¹)	N (kg ha ⁻¹)	P (kg ha ⁻¹)	Index value
GWC	5.93	57	15	7.52
VFG	8.08	100	32	3.88
SMC	3.72	79	20	5.65
FER	3.31	115	44	3.00
BRE	4.97	139	69	2.06
FRU	4.87	291	82	1.43
POT	5.14	172	114	1.35
GEL	4.29	108	48	2.88
DAI	6.58	523	199	0.66
ABA	4.36	266	63	1.73

$$\text{index} = \%OC * (100 - OC_{21}) / [(\%N_{\text{tot}} + 5 * \%P_{\text{tot}}) * 100] \quad (4)$$

with %OC the percentage of organic C in the waste dry matter, and $(100-OC_{21})$ the percentage of stable organic C in the total waste organic C. The composts have the highest index, but there is also a clear separation between the uncomposted wastes, with FER and GEL having an index that is 4–5 times larger than DAI.

Calculation of this index gives a fast indication of the possibilities and limitations for use of these wastes as soil amendments under the conditions and restrictions imposed. Obviously, by changing the relative importance of N and P in the denominator of Equation 4, other situations than the ones specified here can be covered as well. This type of index could thus be used to screen the organic wastes for their suitability for

Table 4. Results of the stepwise linear regression analysis between waste (bio)chemical composition and amount of stable organic carbon ($100 - OC_{21}$).

Model ^a	Variable	Coefficient	Std. error	Significance	R ²	R _a ²
1	Constant	98.8	11.2	0.013	0.905	0.714
	N _{tot}	-0.717	0.301	0.140		
	Cellulose	0.466	0.695	0.571		
	Hemicellulose	-0.146	0.560	0.819		
2	Constant	98.0	7.6	0.001	0.904	0.808
	N _{tot}	-0.718	0.247	0.062		
	Cellulose	0.436	0.540	0.479		
	Hemicellulose	-0.131	0.451	0.790		
3	Constant	97.6	6.5	0.000	0.901	0.852
	N _{tot}	-0.775	0.130	0.004		
	Cellulose	0.443	0.474	0.402		
4	Constant	101	5.2	0.000	0.880	0.855
	N _{tot}	-0.778	0.129	0.002		

^a Dependent variable: $(100 - OC_{21})$.

application in agriculture. In soils and regions where nutrient application restrictions apply, as is the case in large parts of Western Europe, wastes with higher values of the index will be preferred as sources of stable organic C. However, in situations where the N and P status is low, the food industry wastes with low value of the index could prove to be good soil amendments with still considerable organic C supplying capacity when nutrient availability is problematic. It should be stressed once more that the suitability of these wastes for use in agriculture is largely determined by the absence or very small concentrations of toxic compounds such as heavy metals and organic contaminants. This obviously would exclude wastes from most other industries from similar applications.

Conclusions

The organic carbon contained in food industry wastes was shown to be less stable than the organic carbon in a number of composted materials. However, these wastes can still contribute considerably to organic matter supply to soil due to their high organic carbon content, which, in combination with their low content of toxic compounds, makes them potentially suitable for use in agriculture. The macronutrient content (N and P) of these wastes is in general large, which will probably put strong restrictions on their use in large parts of Western Europe. An index reflecting the balance between stable organic carbon and macronutrient content was developed and can be useful in classifying these waste products according to their suitability for landspreading, depending on the local soil fertility status and restrictions on nutrient application.

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