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Soil Carbon Mineralization Kinetics as Influenced by Changes in Land Use and Soil Management in the Central Highlands of Ethiopia

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በመካከለኛው የኢትዮጵያ ደጋማ ቦታዎች የአፈር መብላላት (ሚንራላይዜሽን) ሂደትን ለመረዳት ኩታ ገጠም ከሆኑ 5 የመሬት አጠቃቀምና አያያዝ ዘዴዎች (የግጦሽ መሬት፣ የእርሻ ማሳ፣ ወደ በህር ዛፍ የተቀየረ የእርሻ መሬት፣ በኖራ የታከመ ማሳ እና የእዳሪ ማሳ) በተሰበሰቡ የአፈር ናሙናዎች ላይ 62 ቀናት የፈጀ የቤተ ሙከራ ጥናት ተካሂዷል፡፡ ውጤቱ እንደሚያሳየው የጥቅል ካርቦን መብላላትና እና የመብላላት ፍጥነት ከግጦሽ መሬት በሁለቱም የአፈር ጥልቀት (ከ0-10 ሳ.ሜ እና 10-20 ሳ.ሜ) መጠን ከሌሎች የመሬት አጠቃቀም ስልቶች አንፃር ሲታይ በተከታታይነት ከፍተኛ ሆኖ ታይቷል፡፡ ጥቅል የካርቦንዳይኦክሳይድ ልቀት ሲታይ የግጦሽ ማሳ፣ የእርሻ ማሳ፣ የባህር ዛፍ ማሳ፣ የእዳሪ መሬት እና በኖራ የታከመ መሬት ቅደም ተከተላቸውን ጠብቀው ከፍተኛ መጠን ነበራቸው፡፡ ከተፈተሹት 6 የፍጥነት ሀይል ሞዴሎች ውስጥ የመጀመሪያ ደረጃው ሞዴል (ጥቅል ካርቦን ልቀት፣ እምቅ ካርቦን) ($1-e^{-kt}$) የሚባለው የሙከራ አፈሮችን የካርቦን መብላላት መጠን በመግለፅ ረገድ ተመራጭ ሆኗል፡፡ የግጦሽ ማሳ በሁለቱም ጥልቀት ደረጃዎች እና የእርሻ ማሳ በ10- 20 ሳ.ሜ ጥልቀት ከሌሎች የመሬት አጠቃቀም ስልቶች አንፃር ሲታዩ አመርቂ የሆነ እምቅ ለመብላላት የሚሆን ካርቦን እንዳላቸው ማወቅ ተችሏል፡፡በተመሳሳይ መልኩ ኖራ የተጨመረበት ማሳ እና የእዳሪ መሬት ከ10-20 ሳ.ሜ የአፈር ጥልቀት ላይ ከሌሎች መሬት አጠቃቀሞችና አያያዝ ስልቶች አንፃር ሲተይ አመርቂ የሆነ ካርቦንዳይኦክሳይድ ነበራቸው፡፡ በሌላ በኩል ደግሞ የግጦሽ ማሳን የአፈር ተፈጥሮአዊ እና ከብደተ-ህያው ካርቦን ይዘት ከኩታ ገጠም የእርሻ ማሳ፣ የባህር ዛፍ መሬት፣ በኖራ የታከመ ማሳ እና በእዳሪ ማሳ አንፃር ሲታይ በ9.9 በመቶ በአስተማማኝ ደረጃ የበለጠ መሆኑ ታይቷል፡፡ የአፈር ተፈጥሮአዊ እና ከብደተ-ህያው ካርቦን መጠን ከእምቅ መብላላት ከሚችል ካርቦን፣ የፍጥነት ኃይል እና የሁለቱ ብዜት ጋር አዎንታዊ ግኙኝነት ሲኖራቸው ከ$t½$ እና qCO_2 ጋር ግን አሉታዊ ግኙኝነት እንዳላቸው ለታዉቋል፡፡ ስለሆነም ተአካ፣ ከህካ፣ እሚካ እና qCO_2 የመሬት አጠቃቀምና አያያዝ ዘዴዎችን የልዩነት ደረጃ ሁነኛ መለኪያዎች መሆናቸው ታውቋል፡፡

Abstract

Conversions of natural vegetation to other land use and soil management systems are often accompanied by changes in soil properties and have environmental implications. Such changes in land use and agricultural practices affect soil carbon pools and contribute to increased atmospheric CO2 concentrations. Hence, to understand carbon mineralization processes, a 62-day laboratory incubation experiment was carried out using soil samples collected from five adjacent land uses and management systems (grassland, cropland, Eucalyptus plantations, limed land, and fallow land) in the central highlands of Ethiopia. Total carbon mineralized and the mineralization rates were consistently higher in grasslands in both 0-10 cm and 10-20 cm as compared to the other land uses and management systems. The cumulative CO_2 release followed the order: grassland> cropland> Eucalyptus> fallow land> limed land. Among six kinetic models tested, a first-order model [$C_t = C_o (1-e^{-kt})$] was selected and fitted well to describe C mineralization of the experimental data. Grassland in both depths and cropland in the surface layer (0- 10 cm) had significantly higher mean values of potentially mineralizable carbon (C_o) as compared to each depth in different land uses.

Metabolic quotient (qCO_2) observed in limed land and fallow land in 10 -20 cm depth was significantly higher than the other land uses and management systems. Similarly, soils under grassland had significantly ($p<0.001$) higher soil organic carbon (SOC) and microbial biomass carbon (MBC) than the adjacent cropland, Eucalyptus plantations, limed land and fallow land. SOC and MBC were positively correlated with C_o, k and C_o*k, and negatively correlated with $t_{1/2}$ and qCO_2. Hence, SOC, MBC, C_o and qCO_2 were better discriminators among different land uses and management systems, and therefore, could be used as sensitive indicators of ecosystem change in the study area.

Introduction

Conversions of natural vegetation to other land use and soil management systems are often accompanied by changes in soil properties and have environmental implications. Such changes in land use and agricultural practices have affected soil carbon pools and contributed to increased atmospheric CO_2 concentrations (Smith and Conen, 2004). Soils play an important role as a sink or source of large quantities of carbon to the atmosphere through soil respiration. Globally, C losses from land use changes have been steadily increasing over the last century, approaching rates of about 1.4 Pg C yr^{-1} (Le Quéré, 2010). Therefore, understanding the manifestation of the biological activities of soils from different land uses and variations occurring in the organic carbon pool has paramount importance in sustainable management of ecosystem. Soil respiration is a fundamental process in the carbon cycle and represents the main pathway whereby carbon fixed by the soil is returned to the atmosphere (Fernandez *et al*., 2006). Thus, even small changes in soil respiration may greatly influence atmospheric carbon and heat balance (Veenendaal *et al*., 2004; Kane *et al*., 2005). Consequently, soil respiration has received considerable attention in recent years because of the release of large quantities of CO_2 from the soils to the atmosphere. Therefore, assessing the impact of land use changes on soil respiration is of vital significance to understand the interactions between belowground metabolism and regional carbon budgets (Sheng *et al*, 2010). Particularly, the description of the dynamics of C mineralization in incubation studies by fitting the experimental data to kinetic models could be of great interest for the prediction of the ability of soils in supplying potentially mineralizable organic carbon and, more generally, for the organic matter balance (Alessandro *et al*., 2014).

Previous reports have found that a zero-order equation more adequately describes C mineralization (Seyfried and Rao, 1988). However, a first-order equation has been frequently used to describe the carbon mineralization process of SOC (Dossa *et al*, 2009; Aulen *et al*, 2012). Alternatives to the simple first-order model, Delphin (1988) successfully used a two-part parabolic equation that assumes soil organic carbon can be divided into two components, a labile fraction and a more recalcitrant one, each decaying exponentially at rates characterized by its own constant (k and h, respectively). Putting the above points in view, numerous field and laboratory incubation studies have been conducted worldwide (Campos *et al*., 2006; Davidson *et al*., 2006; Sheng *et al*., 2010; Fazle et al., 2014). However, most of these studies on the effects of land use changes on soil respiration have only focused on temperate environments (Carlisle *et al*., 2006; Kellman *et al*., 2006; Alessandro *et al*., 2014) and tropical Latin America (Salimon *et al*., 2004; Campos, 2006). On the other hand, soil microbial activities, populations and communities are governed by site characteristics, such as soil type and texture, temperature, moisture or pH. It also varies with type of vegetation and its management practices, environmental conditions and land use types (Frank *et al*., 2006).

In the highlands of Ethiopia, where low soil pH and associated problems are among the major impediment to agricultural productivity, conversions of natural vegetation/agricultural land to other land use and management systems have been in practice since several decades. However, the native soil organic carbon dynamics in different land uses and its concomitant contribution to CO_2 fluxes have so far rarely been investigated, and virtually nothing is known about CO_2 emissions from different land uses and management systems. Given that soil microbial parameters are affected by land use, quantifying them can help evaluate the changes in soil microbial functions driven by changes in land use (Bastida, 2006).

The objectives of this to present results of study on assessing the C-mineralization potentials of soils from five adjacent land uses and management systems (grassland, cropland, *Eucalyptus* plantation, limed land and fallow land) by laboratory incubation; the effectiveness of some commonly used decay models for describing rates and amounts of C mineralization; and evaluations on selected soil microbial parameters and their relationships with the C mineralization parameters derived from the best model.

Materials and Methods

Site characteristics

Soil samples were collected from Wetabecha Minjaro, (9° 05′ 55″ N; 38° 36′ 21″ E) in the central highlands of Ethiopia. The area receives 1100 mm of rainfall annually. The mean decadal **(2005 - 2014)** monthly maximum and mean decadal monthly minimum temperatures are 23.3 °C and 8.7 °C, respectively. According to the local agro-climatic classifications, it belongs to moist highland agro-climatic zone with two distinctive rainy periods; the main rainy season, which occurs from June to September, and the short rainy season extending from February to April. The soils are classified as Nitisols with deep, red, well- drained tropical soils (IUSS, 2006). In these soils, soil acidity ($pH< 5$) and associated low nutrient availability are constraints to crop production.

Soil sampling and respiration measurements

In July, August and September, 2012, composite soil samples were randomly collected from eight sites in each land use and management systems that encompass five adjacent land uses and management systems (grassland, cropland, *Eucalyptus* plantation, limed land and fallow land) in two depths (0- 10 cm and 10- 20 cm) from plots of 10 x 10 m area. Descriptions of the land uses are given below (Table 1). Besides using the eight sites as replicates, samples were also replicated twice in the laboratory during incubation. The samples for *Eucalyptus* plantations (*Eucalyptus globulus* L.) were collected from plantations that were 6 -7 years old. In this context, fallow land was referred to mean a resting period of 18 months without crop cultivation. The samples from limed lands were collected after three years of liming acid soils. Prior to analysis, all the samples were air-dried, thoroughly mixed, and sieved at 2 mm, plant roots and other residues were removed from the samples. Soil particle size distribution was determined by hydrometric method (Bouyoucos, 1962). Soil pH was determined by using a pH meter in a 1:2.5 soil/water suspension, soil organic carbon by Walkley and Black method (Walkley and Black, 1934). Total nitrogen (TN) was analyzed by wet oxidation procedure of the kjeldahl digestion, distillation and titration method (Bremner and Mulvaney, 1982).

Table 1. Descriptions of land use/management systems of the study area

Land use/management	Brief descriptions
Cropland	Areas that were under barley cultivation at the time of soil sampling; had no history of liming
Grassland	Huge areas of land with no cropping practice, trees or settlements; totally dominated by natural grasses and used for grazing
Fallow land	Abandoned previously croplands from cultivation for a period of 18 months to restore soil fertility
Limed land	Lime applied fields to counteract soil acidity and its associated problems
Eucalyptus plantations	Areas occupied by *Eucalyptus globulus* plantations (6 -7 years of age)

Microbiological analysis

For measuring microbial activity, thirty grams of dry soil samples were wetted to 60 % of water holding capacity and incubated in 0.5 L air tight jars at 28 °C. The CO_2 evolved was trapped in plastic vials containing 10 ml of 0.5 M NaOH. An empty vial without was used a control. The moisture content was kept constant by adding distilled water and weighing at each sampling date. The amount of CO_2 evolved was measured after 8, 18, 26, 36, 45, 54 and 62 days of incubation by titrating with 0.5 M HCl against a phenolphthalein indicator after precipitation with $BaCl_2$ (0.5 M). Soil microbial biomass carbon (MBC) and nitrogen (MBN) were determined by the chloroform fumigation extraction method, using 0.5 M K_2SO_4 as extractant (Vance et al., 1987). Carbon and nitrogen contents in the fumigated and non-fumigated extracts were determined using SKALAR FormacsHT total organic carbon analyzer for liquid samples. The metabolic quotient qCO_2 [(mg CO_2-C. h^{-1}. g^{-1} MBC] was calculated from respiration values with formula: qCO_2 = [(CO_2-C mineralized/MBC] (Anderson and Domsch, 1985). It is considered as an index of microbial efficiency in utilizing the available resources (high efficiency for low values of qCO_2).

Carbon mineralization kinetic models

Descriptive and graphical analyses of the CO_2 respiration values obtained during the 62 days incubation from different land use/management and depths were carried out to detect anomalies. In addition, six different models were used to describe the CO_2 respiration (Table 2). They were obtained from the scientific literature. The models were tested to know which one was the best to our data. The convergence, the values of adjusted coefficient of determination ($R^2_{adj.}$), the squared sum error (SSE) and the mean squared error (MSE) were important criterions for choosing the best model. Model fitting were carried out with MODEL procedures of the SAS/STAT® statistical program (SAS Institute Inc., 2001).

Table 2. Kinetic models used in to describe soil carbon mineralization

Model	Equation	References
First order	$C_t = C_o(1-e^{-kt})$	Murwira et al. (1990)
First order special	$C_t = C_o(1-e^{-kt})+C_1$	Jones (1984)
Linearized power function	$C_t = kt^m$	Standford and Smith, (1972)
Zero order	$C_t = a+ kt$	Seyfried and Rao (1988)
Two simultaneous reactions	$C_t = C_1(1-e^{-kt})+C_2(1-e^{-ht})$	Delphin (1988)
Special model	$C_t=C_1(1-e^{-k^*t})+ht$	Bonde and Rosswall (1987)

Note: *C_t, C_o, C_1 and C_2= Cumulative carbon mineralized after time t (mg C-CO_2/kg soil), potentially, easily, slowly mineralizable carbon (mg C-CO_2/kg soil), respectively; a= intercept; k, m and h= rate constants (day^{-1}); t= time from the start of incubation.*

Biochemical parameters related to organic carbon

Once the model was fitted, different parameters were determined to analyze: a) the potentially mineralizable C (C_o), b) the rate constant of carbon mineralization (k) that represents the slope of the curve, c) the initial potential rate of C mineralization (C_o*k), as the product between C_o and k, and d) the half-life time of carbon ($t_{1/2}$), that represents the time needed for half of the initial C to decay away necessary to reach a half of the maximum mineralization. A mixed model was applied in order to detect significant differences in the measured variables as a function of land uses and management systems in two depths. The depth was considered as a repeated measures factor. The statistical model was expressed as follows: [Eq. 1]

$$Y_{ij;k} = \mu + \alpha_i + \beta_j + \gamma_k + \beta\gamma_{jk} + \varepsilon_{ij;k} \qquad \text{[Eq. 1]}$$

with i=1,…,8 for the sites, j=1,…,5 for the land uses and k=1,2 for the two depths, and being:
$Y_{ij;k} =$ Observed values of the dependent variable for the land use j at depth k in site i.

$\mu =$ general mean effect; $\alpha_i =$ main effect of the site I; $\beta_j =$ main effect of the land use j; $\gamma_k =$ main effect of the depth k; $\beta\gamma_{jk} =$ interaction effect of the land use j with the depth k; $\varepsilon_{ij;k} =$ random error in the dependent variable for the land use j at depth k in the site i. The assumptions for the model were:

- $\varepsilon_{ij;k} \sim N(0, \sigma_k^2)$, with σ_k^2= random variance for errors at depth k.
- $Cov(\varepsilon_{ij;k}, \varepsilon_{i'j';k'}) = \begin{cases} \omega & \text{if } i = i', j = j' \text{ and } k \neq k' \\ 0 & \text{if } i \neq i' \text{ or } j \neq j' \end{cases}$, with ω=covariance between errors at different depths. Therefore, the model included three variance parameters, which were estimated using the restricted maximum likelihood method (REML). Finally, Tukey´s HSD procedure was used for multiple comparisons of mean physical, chemical, and microbiological properties of the soil under different land use systems.

Results and Discussions

Soil respiration rates

Carbon dioxide-C mineralization rates during the 62-days incubation period followed similar general pattern across in all land uses and management systems; an initial increase at the beginning of the incubation followed gradual decreases as the incubation time progresses (Figures 1 and 2).

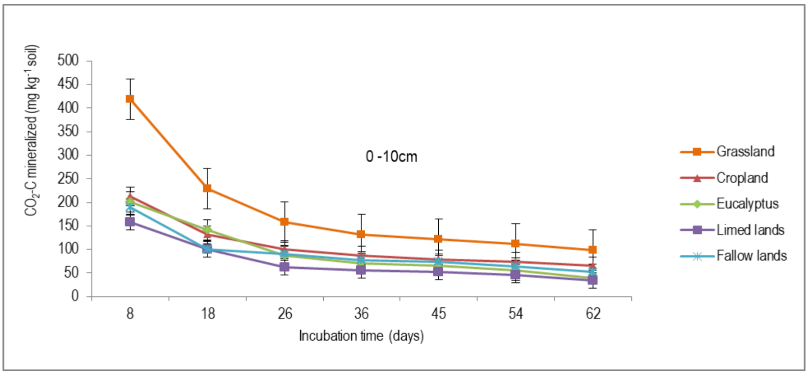

Figure 1. Carbon mineralization rates for different land uses during 62 days of laboratory incubation with standard error (n=8).

The higher amount CO_2 evolved at initial stage indicate a rapid depletion of an easily mineralizable fraction (labile SOC) while the slow-steady phases in which mineralization declined to a fairly constant rate indicate that the most active fraction has exhausted and the resistant and stable fraction of SOC was being mineralized (Wander et al. 1994).

The CO_2 release in 0 -10 cm followed the order: grassland> cropland> fallow land> *Eucalyptus*> limed land. However, in 10 -20 cm the order was; grassland> fallow land > *Eucalyptus*> limed land> cropland. In grassland, the emission was higher in both 0 -10 cm and 10-20 cm than the other land uses and management systems. The higher CO_2 release in grassland could be attributed to the higher organic matter content as compared to cropland, *Eucalyptus*, fallow land and limed land. Mukhopadhyay and Maiti (2014) reported higher CO_2 flux under grassland as compared to afforested land because of higher root density in grasslands, which conserves moisture and enhances root and microbial respiration. Similarly, Chen et al. (2010) also reported significantly higher microbial respiration in grasslands as compared to other land uses in North Eastern Tibetan plateau. High rates of soil respiration can occur because of large pool of labile C substrates or rapid oxidation of smaller pool (Islam and Weil, 2000). Thus, high basal respiration may indicate ecological stress and degradation or a high level of ecosystem productivity.

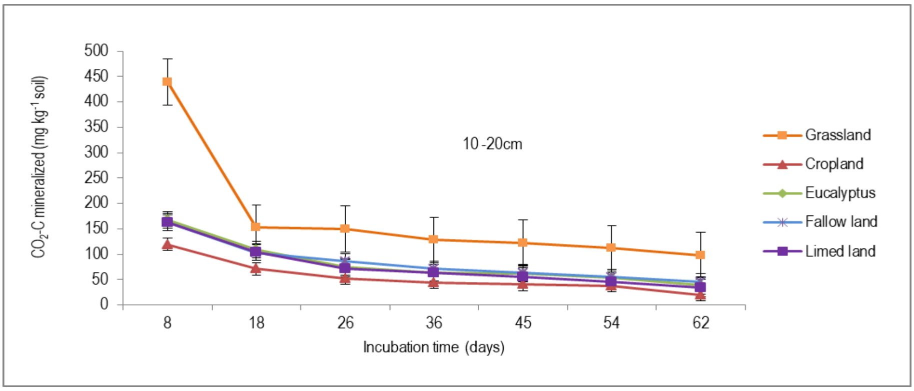

Figure 2. Carbon mineralization rates for different land uses during 62 days of laboratory incubation with standard error (n=8).

Cumulative CO_2-C mineralization

The cumulative CO_2 evolved during 62 days incubation period as affected by different land uses and management under the two depths is presented in Figure 3. In general, at any given time, cumulative CO_2 evolved was significantly greater in grassland as compared to other land uses and management systems. The cumulative CO_2 production was in the order: grassland> *Eucalyptus*> fallow land> cropland> limed land over the entire incubation period. The cumulative amount of CO_2 released from surface (0-10 cm) and sub-surface (10-20 cm) soil samples were also greater in grassland (Figure 3). However, the four land uses, i.e. cropland, fallow land, Eucalyptus and limed land were not significantly ($p \geq 0.05$) different from each in both depths, except in cropland, where significantly lower cumulative CO_2 was recorded in 10- 20 cm soil depth.

According to Frank et al. (2006), higher SOC and MBC lead to higher soil respiration, and lowest values of soil respiration corresponded to sites with lowest MBC. The relatively greater C to be mineralized in grasslands indicates that grasslands contained easily decomposable organic matter than the other land uses and management systems. This finding is in line with the results of Haiqing *et al*. (2009), where they reported higher mineralized carbon under grassland and reduced tillage due to fewer disturbances, promotion, and stabilization of aggregates compared to plowed soils in Southern Germany. Nonetheless, the decrease in the C mineralization in 10- 20 cm might be due to lower organic carbon content and relatively smaller number of microbes as soil depth increases. These results agree with the reports of Taylor *et al.* (2002), where reduced activities of microbial and fungal were reported in deeper versus surface soil layers.

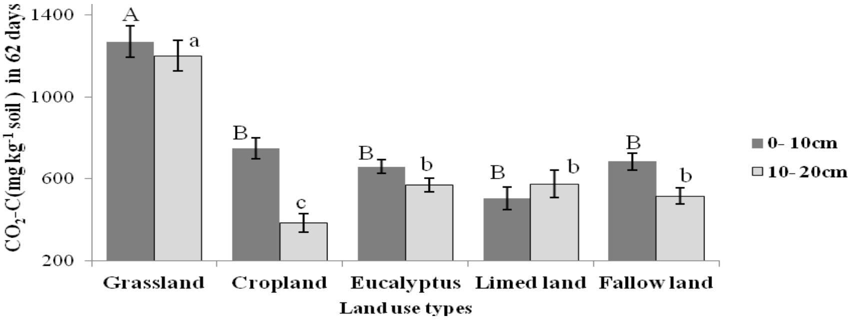

Figure 3. Cumulative CO_2 evolved at the end of 62 days of laboratory incubation for the different land uses and soil depth. Bars are standard error of the mean (n=8). Similar letters above the vertical bars within similar depth across land uses denote no significant differences at $p<0.05$. Upper case letters refer to 0- 10 cm and lower case letters refer to 10- 20 cm depths.

Carbon mineralization kinetics

Kinetic models used to describe soil carbon mineralization were presented in Table 1. In the process of selecting the best model, some of the models showed lack of convergence in most of the sites, land uses, and depths (models 5 and 6). Models 3 and 4 showed higher values of the statistical parameters, but the values were smaller than those obtained in model 1. Therefore, based on convergence, statistical parameters of the fitting, and the significance of the estimated parameters, the first-order model [$C_t = C_o (1-e^{-kt})$] was selected for the experimental data. This kinetic model provided a good fit to C mineralization with R^2 ranging from 0.90 to 0.99 for all land uses and management systems.

The first- order kinetic model used to describe the C mineralization process of soil organic matter assumes that the microbial biomass is constant and the rate of decomposition only depends on the available substrate. Many researchers have fitted C mineralization data with first-order model (Dossa *et al*., 2009; Aulen *et al*., 2012). Analysis with PROC MIXED showed significant interaction between land use/management and soil depth for the C_o, k, $t_{1/2}$ and qCO_2 parameters (Table 3). This implies that the values of each parameter in each land use/management depended on the depth of soil sampling (Table 3). In the surface layer (0-10 cm), significantly higher mean values of C_o were observed in grassland. However, this value was not significantly different from mean value of C_o observed in cropland.

Potentially mineralizable carbon recorded in cropland and fallow land did not differ statistically. Similarly, C_o values that were recorded in *Eucalyptus* and limed land were not statistically different from each other in both 0-10 and 10-20 cm soil depth. But it was significantly lower than cropland and fallow land in 0- 10 cm depth. In 10-20 cm depth, the lowest C_o was observed in cropland, which was statistically different from grassland and fallow land. Lower values of C_o either suggest a lower activity of microbial community or residues more difficult to decompose, due to a different chemical composition. In general, microbial biomass carbon is commonly described as a living or active pool in models that simulate organic C turnover in soils, and the size of this pool directly affects the model outputs (Probert et al., 1998). Therefore, differences in SOC and MBC could substantially contribute to the differences observed in the outputs of carbon kinetic models in this study.

Table 3. Parameters of microbial mineralization activity of soils sampled from five land uses and two depths, estimated according to first-order equation

Depth (cm)	LUM	C_o (mg CO_2-C /kg soil)	k (day^{-1})	$t_{1/2}$ (day)	qCO_2[(mg CO_2-C h^{-1}) /g MBC]
0-10	Grassland	1387.5A a	0.03411A a	20.7B a	1.06BC a
	Cropland	1339.2AB a	0.01468C b	67.0A a	1.23BC a
	Eucalyptus	744.7BC a	0.03302A a	22.4B a	1.13BC a
	Limed land	523.0BC a	0.03132AB a	23.1B a	0.75C b
	Fallow land	928.2B a	0.02013BC a	37.9AB a	1.74AB b
10-20	Grassland	1325.3A a	0.03243A a	22.8C a	1.26BC a
	Cropland	453.6C b	0.02914A a	25.9BC b	1.01C a
	Eucalyptus	706.6BC a	0.02670AB a	28.7BC a	1.61BC a
	Limed land	681.0BC a	0.02879A a	25.0BC a	1.99AB a
	Fallow land	950.4AB a	0.01605B a	56.2AB a	2.60A a

Note: *Different upper case letters show significant differences in each depth between different land uses; different lower case letters show significant differences in each land use between the two depths. LUM: land use and management,* C_o*: potentially mineralizable carbon, k: rate constant of carbon mineralization,* $t_{1/2}$*: half- lifetime,* qCO_2*: metabolic quotient.*

Rate constant (k) for C mineralization was significantly higher in soils of grasslands in 0-10 cm soil profile. However, it was not statistically different from soils of *Eucalyptus* and limed land with similar depth. The k values observed in cropland in 0- 10 cm soil depth were significantly lower than the other land uses/management systems studied. In 10- 20 cm soil profile, mean k values were similar in all land uses and management systems except fallow land where the lowest mean value of k was recorded. However, k value recorded in fallow land and *Eucalyptus* were not different. All values of k fell within relatively narrow range except cropland and fallow. This implies that microbial respiration and metabolized organic compounds are similar or have the same degree of availability (Riffaldi et al., 1996; Alessandro et al., 2014), and no significant correlations between k and both potentially and easily mineralizable, indicating that differences in k values among soils cannot be attributed to differences in the relative sizes of the C pools (Riffaldi et al., 1996). Cropland exhibited significantly higher values of $t_{1/2}$ as compared to the other land uses and management systems in 0 -10 cm soil depth. However, significant changes in $t_{1/2}$ were not detected among grassland, *Eucalyptus*, limed land and fallow land in 0-10 cm soil depth. In 10- 20 cm, fallow land recorded higher values of $t_{1/2}$, which was not statistically different from the other land uses except grassland. The half-life time observed in grassland was lower than the other land uses and management systems. Half-life time is a more clearly interpretable parameter, where high levels are associated with ecosystem stresses (Anderson and Domsch, 2010). Therefore, the higher half-life time in fallow land and cropland indicates lower carbon mineralization rates.

Even though, higher contents of MBC and soil respiration generally indicate better soil quality, these parameters do not always show the same change tendency. Thus, metabolic quotient (qCO_2) is used to evaluate the efficiency of soil microbial biomass in utilizing the organic carbon compounds (Anderson and Domsch, 1990). In 0- 10 cm, the qCO_2 observed in fallow land was higher than the other land uses and management system. Higher values of qCO_2 imply a higher requirement of maintenance energy or lower metabolic efficiency in the utilization of both the native organic matter and the added plant material. The mean value of qCO_2 recorded in limed was comparable with the other land uses except fallow land. Similar to 0- 10 cm, the qCO_2 observed in fallow land in 0-20 cm was significantly higher than the other land uses and management system except limed land. However, there was no evidence of significant differences among grassland, cropland, and *Eucalyptus* in 10- 20 cm. In ecological terms, a high qCO_2 reflects a high maintenance carbon demand, and if the soil system cannot replenish the carbon which is lost through respiration, microbial biomass must decline (Anderson and Domsch, 2010). The lower qCO_2 value observed in 0-10 cm soil depth indicate more stable ecosystems, with greater efficiency of microorganisms to convert organic waste into microbial biomass and with greater sustainability. Our results are consistent with results published elsewhere (Gil-Sotres et al., 2005). Agricultural liming which is generally used to overcome the problem of acidification, would result in higher abundance and diversity of detritivorous soil fauna such as some species of earthworms (Bishop, 2003); contribute to an improved organic matter decomposition and nutrient mineralization (Bradford et al., 2002). Hence, the beneficial effect of liming was conspicuously observed by significantly lowering the values of qCO_2 only in 0- 10cm soil depth. However, the applied lime could not change the qCO_2 value in 10- 20 cm soil profile due to the fact that lime moves very slowly in soil and its ameliorative effect is limited to the top few centimeters of the soil profile. The qCO_2 also indicates the changes in microbial activity between natural and disturbed ecosystems more clearly (Islam and Weil, 2000). Hence, the relatively higher qCO_2 values observed in soils of fallow land in both depths and limed land in 10-20 cm indicate greater stress of the microbial community compared to soils with more stable ecosystems (Islam and Weil, 2000). Soil chemistry affecting substrate availability also changes dramatically with depth. In this study, results of the carbon kinetics parameters were significantly affected by depth of sampling in some of the land uses. Cropland showed significantly higher mean values of C_o and $t_{1/2}$ in 0- 10 cm as compared to 10- 20 cm soil depth. In 0- 10 cm, mean values of qCO_2 recorded in limed land and fallow land were significantly lower than the values in 10-20 cm. Generally, soils of grassland and Eucalyptus were not affected by depth of sampling relatively implying stable ecosystem in these land use systems.

Soil organic carbon, MBC, Ct and C_o*k

Soils under grassland had significantly ($p<0.001$) higher SOC and MBC than the adjacent cropland, *Eucalyptus* plantations, limed land and fallow land (Table 4). However, mean values of SOC among cropland, *Eucalyptus,* limed land and fallow land did not show any evidence of statistical difference. The lower levels of SOC in soils of these land uses/management systems could be due to a combination of lower carbon inputs through biomass return and greater C losses by soil erosion after tillage. Losses in soil C caused by the conversion of natural vegetation to cultivated land is well documented. Cultivation of land that was previously covered in perennial vegetation, SOC can be rapidly lost due to enhanced C decomposition and erosion brought about by soil disturbance (Lal, 2005; Van der Werf *et al*., 2009). Guo and Gifford (2002) conducted a meta-analysis of land-use change experiments and showed that converting grassland to croplands caused significant loss of SOC, whereas conversion of forestry to grassland did not result in SOC loss in all cases.

The increased carbon content in grasslands suggests that grassland soil could serve as a C sink in a given ecosystem. The fact that MBC declines with depth partly explains the lower microbial activity in 10- 20 cm soil depth. Clay content controls availability of exchangeable ions, and organic matter controls the abundance and availability of dissolved and available sorbed organic nutrients, both decreases with depth (Konopka and Turco, 1991).

Table 4. Soil organic carbon, MBC, Ct, C_o*k as affected by land use and soil depth, with standard error and p-values of ANOVA

LUM	SOC [g/kg soil]	MBC [mg/kg soil]	Ct [mg CO_2-C /kg soil]	C_o*k [mg C-CO_2 /kg soil day]
Grassland	43.6a	763.7a	614.3a	45.2a
Cropland	23.2b	342.6b	243.7bc	15.6b
Eucalyptus	24.5b	345.0b	311.7b	21.2b
Limed land	24.0b	335.7b	255.7bc	18.6b
Fallow land	23.2b	190.8c	224.1c	14.9b
SE	1.27	35.25	19.56	1.63
Depth (D) (cm)				
0- 10	28.5	485.4a	362.7a	24.9a
10 -20	26.9	305.8b	297.1b	21.3b
SE	0.81	18.46	12.54	1.04
ANOVA				
LUM	***	***	***	***
D	Ns	***	***	*
LUM x D	Ns	ns	ns	ns

Note: LUM= land use/management, SOC: organic carbon, MBC= microbial biomass carbon, Ct: total carbon mineralized in 62 days, C_o*k: initial potential rate of C mineralization, SE: standard error, †Mean values with similar letters in each column indicate non-significant differences at *$p<0.05$, ** $p< 0.01$, *** 0.001, ns = non-significant

The total amount of carbon mineralized in 62 days of incubation period in grassland was significantly superior to the other land uses. However, Ct observed in fallow land was the lowest, and was not different from cropland and limed land. Similarly, mean values of

C_o*k recorded in grassland was higher than the other land uses and management systems. Statistically, cropland, *Eucalyptus*, limed land and fallow land did not differ from each other in mean values of C_o*k. As expected, Ct and C_o*k were significantly higher in 0- 10 cm depth compared to 10- 20 cm. Because, higher values of SOC and MBC were recorded in 0- 10 cm and these parameters have high association with Ct and C_o*k. The microbial activity decreased clearly with increasing depth in response to the decreasing labile C pools (Agnelli *et al*., 2004).

Relationships between different soil variables

The ability of SOC to predict C_o in 62-days was evaluated by sketching a linear regression using eighty observations (Figure 4). Good fits of correlation between MBC or OC and C mineralization parameters suggests that C mineralization depends on the MBC and OC availability to microbial activity.

Figure 5 shows the relationship between qCO_2 and MBC: SOC, where both soil biological parameters are inversely related. The figure shows that higher values of qCO_2 reflect difficulties in the use of organic substrates by microbial biomass due to low levels of MBC: SOC. Klose *et al*. (2004) also reported that a low MBC: SOC ratio indicates a reduced pool of available carbon in soil.

Results clearly showed that soils that have values of MBC: SOC higher than 1.5% had efficient microorganisms in utilizing the available substrates than soil with MBC: SOC lower than 1.5%. The observed increase in the ratio of MBC: SOC in the grassland as compared to other land uses might be due to lesser disturbances and the presence of readily available C as compared to the other land uses. Powlson (1994) also reported MBC to SOC ratio as an effective early warning of the improvement or deterioration of soil quality.

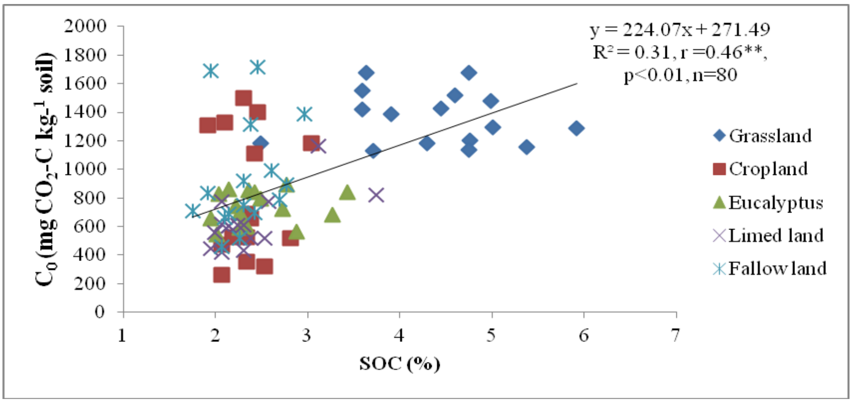

Fig. 4. Linear regression between C_o and SOC

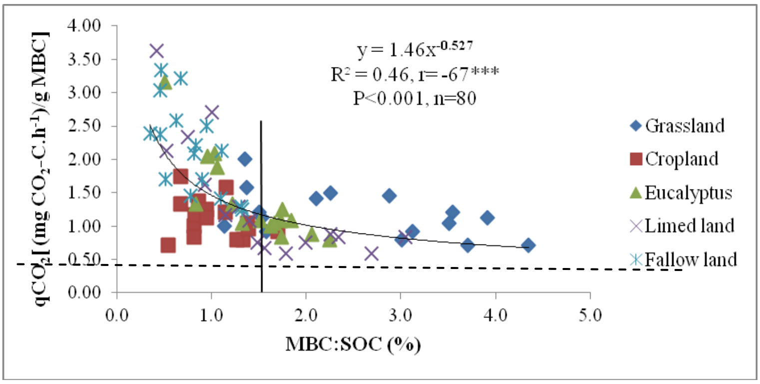

Fig. 5. Linear regression between qCO_2 and MBC: SOC

Simple correlation coefficients (r) between some selected physical, chemical, and microbiological parameters are presented in Table 5. Results demonstrated a clear positive correlation between SOC and C mineralization parameters (C_t, C_o, k and C_o*k) estimated according to first-order mineralization model. Similarly, these C mineralization parameters had significant positive associations with MBC. However, both SOC and MBC were negative correlated with $t_{1/2}$ and qCO_2. These findings are in agreement with the findings of Beck *et al*. (1997) who reported significant correlations between the above C-mineralization parameters with both SOC and MBC.

Good fits of correlation (r = 0.63, 0.43 and 0.62) were also found between TN, and C_t, C_o and C_o*k respectively. Correlations between qCO_2 and MBN was negative and statistically significant ($p<0.001$). This might be due to high values of qCO_2 that are associated with soil degradation brought about by poor agricultural management practices (Islam and Weil, 2000).

Table 5. Spearman correlation coefficients between C-mineralization parameters of the first order model and some selected soil characteristics

Parameter	Sand	Clay	pH	SOC	TN	MBC	MBN
C_t	ns	0.294*	0.370**	0.659**	0.632**	0.653**	0.446**
C_o	ns	0.350*	ns	0.456**	0.431**	0.384**	0.307*
K	0.274*	ns	0.404**	0.330*	0.350*	0.346*	ns
C_o**k*	ns	0.278*	0.389**	0.639**	0.618**	0.586**	0.353*
$t_{1/2}$	-0.274*	ns	-0.404**	-0.330*	-0.350*	-0.346*	ns
qCO_2	*ns*	*0.241**	*-0.390***	*ns*	*ns*	*-0.668****	*-0.651****

Note: *Ct=total carbon mineralized in 62days, C_o=potentially mineralizable C, k=C mineralization rate, C_o*K= initial potential rate of C mineralization, $t_{1/2}$= half- life time SOC= soil organic carbon, TN=total nitrogen, MBC= microbial biomass carbon, MBN= microbial biomass nitrogen, ns= non-significant, *p<0.05, **p<0.01 and ***p<0.001*

Conclusions

The amounts of C mineralized during 62-days of soil incubation period were different in different land uses and management systems. Grassland had higher C-CO_2 mineralized than adjacent cropland, *Eucalyptus* plantations, limed land and fallow lands, indicating high rates of biological activity and C cycling relative to the land uses. Among six kinetic models tested, a first-order model [$C_t = C_o (1-e^{-kt})$] provided a good fit to C mineralization data with R^2 ranging from 0.90 to 0.99. Land use change from grassland to other land uses/management systems drastically decreased soil SOC and MBC. Grassland promoted soil SOC accumulation and decomposition both in surface and subsurface soil compared to cropland, *Eucalyptus* plantations, limed land and fallow land. Parameters estimated according to first-order model like C_o, Co*k and qCO_2 were better discriminators among different land uses and management systems. Hence, these parameters along with SOC and MBC were sensitive to land use change, and could be considered as a good indicator of soil quality change in the study area.

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References

Agnelli A, J Ascher, G Corti, MT Ceccherini, P Nannipieri, and G Pietramellara, 2004. Distribution of microbial communities in a forest soil profile investigated by microbial biomass, soil respiration, and DGGE of total extracellular DNA. Soil biology and Biochemistry 36: 859-868.

Alessandro S, V Giacomo, and C Roberto. 2014. Carbon mineralization kinetics in soils under urban environment. Applied Soil Ecology 73: 64-69

Anderson TH and KH Domsch. 2010. Soil microbial biomass: the eco-physiological approach. Soil Biology and Biochemistry 42: 2039-2043.

Anderson TH and KH Domsch. 1985. Determination of eco-physiological maintenance requirements of soil organisms in a dormant state. Biol. Fert. Soils 19: 269-279.

Anderson TH, KH Domsch. 1990. Application of eco-physiological quotients (qCO_2 and Qd) on microbial biomasses from soils of different cropping histories. Soil Biology and Biochemistry 22: 251-255.

Aulen M, B Shipley, and R Bradley. 2012. Prediction of in situ root decomposition rates in an interspecific contest from chemical and morphological traits. Ann. Bot. 109: 287-297.

Bastida F, JL Moreno, T Hernández, and C García. 2006. Microbiological activity in a soil 15 years after its devegetation, Soil Biol. Biochem. 38: 2503-2507.

Beck T, RG Joergensen,E Kandeler Makeschin, F Nuss, HB Oberholzer, and S Scheu. 1997. An inter-laboratory comparison of ten different ways of measuring soil microbial carbon. Soil Biol. Biochem. 29: 1023 -1032.

Bishop HO. 2003. The Ecology of Earthworms and Their Impact on Carbon Distribution and Chemical Characteristics of Soil. PhD Thesis submitted to University of Stirling, Stirling, UK.

Bonde TA and T Rosswall. 1987. Seasonal variation of potentially mineralizable nitrogen in four cropping systems. Soil Science Society of America Journal 51: 1508-1514.

Bouyoucos GJ. 1962. Hydrometer method improved for making particle size analysis of soils. Agron. J. 54: 464 -465.

Bradford MA, GM Tordoff, T Eggers, TH Jones, and JE Newington. 2002. Microbiota, fauna, and mesh size interactions in litter decomposition. Oikos. 99: 317-323.

Bremner JM amd CS Mulvaney. 1982. Nitrogen-total. *In:* Page AL, *et al*., (eds). Methods of Soil Analysis Part 2 (American Society of Agronomy, Madison, Wisconsin. p. 595-624.

Campos CA. 2006. Response of soil surface CO_2-C flux to land use changes in a tropical cloud forest (Mexico). Forest Ecology and Management 234: 305–312.

Carlisle EA, KL Steenwerth, and DR Smart. 2006. Effects of land use on soil respiration: conversion of oak woodlands to vineyards. Journal of Environment Quality 35: 1396-1404.

Chen DD, SH Zhang, SK Dong, XT Wang, and GZ Du. 2010. Effects of land use on soil nutrients and microbial biomass of an alpine region on the North Eastern Tibetan plateau. China. land degradation and development. 21: 446-452.

Davidson EA, IA Janssens, and YQ Luo. 2006. On the variability of respiration in terrestrial ecosystems: moving beyond Q10. Global Change Biology 12: 154- 164.

Delphin JE. 1988. Utilisation de cinétique du premier ordre en vue de caracétriser la minéralisation de quelques sols agricoles. Agronomie 8: 289-294.

Dossa EL, M Khouma, I Diedhiou, M Sene, F Kizito, and AN Badiane. 2009. Carbon, nitrogen and phosphorus mineralization potential of semiarid Shelian soils amended with native shrub residues. Geoderma 148: 251-260.

Fazle Rabbi SM, R Brian, V Peter, Heiko Daniel, and M IainYoung,. 2014. Soil organic carbon mineralization rates in aggregates under contrasting land uses. Geoderma 216: 10-18.

Fernandez DP, JC Neff, J Belnap, and RL Reynolds. 2006. Soil respiration in the cold desert environment of the Colorado Plateau (USA): abiotic regulators and thresholds. Biogeochemistry 78: 247-265.

Frank AB, MA Liebig, and DL Tanaka. 2006: Management effects on soil CO_2 efflux in Northern semiarid grassland and cropland. Soil till. Res. 89: 78- 85.

Gil-Sotres F, C Trasar-Cepeda, MC Leirós, and S Seoane. 2005. Different approaches to evaluating soil quality using biochemical properties. Soil Biol. and Biochemistry 37, 877-887.

Guo LB and RM Gifford. 2002. Soil carbon stocks and land use change: a meta-analysis. Global Change Biology 8: 345-360.

Haiqing C, F Mingsheng, B Norbert, S Karl, and K Yakov. 2009. Effect of land use types on decomposition of 14_C-labelled maize residue (*Zea mays* L.). European Journal of Soil Biology 45: 123 -130.

Islam KR and RR Weil. 2000. Soil quality indicator properties in mid-Atlantic soils as influenced by conservation management. J. soil sci. plant anal. 29: 2269 -2284.

IUSS Working Group WRB. 2006. World reference base for soil resources 2006. World Soil Resources Reports No. 103. FAO, Rome.

Jones CA. 1984. Estimation of an active fraction of soil nitrogen. Commun Soil Sci. Plant Anal. 15: 23-32.

Kane ES, DW Valentine, EAG Schuur, and K Dutta. 2005. Soil carbon stabilization along climate and stand productivity gradients in black spruce forests of interior Alaska. Canadian Journal of Forest Research 35: 2118- 2129.

Kellman L, H Beltrami, and D Risk. 2006. Changes in seasonal soil respiration with pasture conversion to forest in Atlantic Canada. Biogeochemistry 82, 101- 109.

Klose S, KD Wernecke, and F Makeschin. 2004. Soil microbial biomass and enzyme activities in forest soils exposed to atmospheric depositions from a lignite power plant. Soil Biology and Biochemistry 36: 1913 -1923.

Konopka A and R Turco. 1991. Biodegradation of organic compounds in vadose zone and aquifer sediments. Appl. Environ. Microbiol. 57: 2260-2268.

Lal R. 2005. Forest soils and carbon sequestration. Forest Ecology and Management 220: 242-258.

Le Quéré C. 2010. Trends in the land and ocean carbon uptake. Current opinion in Environmental Sustainability 4: 219-224.

Mukhopadhyay S and SK Maiti. 2014. Soil CO_2 flux in grassland, afforested land and reclaimed coalmine overburden dumps: A case study. Land Degra. Dev. 25: 216-227.

Murwira HK, H Kirchmann, and MJ Swift. 1990. The effect of moisture on the decomposition rate of cattle manure. Plant and Soil 122: 197-199.

Powlson DS. 1994 The soil microbial biomass: before, beyond and back. *In*: Ritz K, J Dighton, KE Giller (eds) Beyond the biomass. Wiley, Chichester p. 3-20.

Probert ME, JP Dimes, BA Keating, RC Dalal, and WM Strong. 1998. APSIM's water and nitrogen modules and simulation of the dynamics of water and nitrogen in fallow systems. Agricultural Systems 56: 1–28.

Riffaldi R, A Saviozzi, and R Levi-Minzi. 1996. Carbon mineralization kinetics as influenced by soil properties. Biol. Fertil. Soils 22: 293-298.

Salimon CI, EA Davidson, RL Victoria, and AWF Melo. 2004. CO_2 flux from soil in pastures and forests in southwestern Amazonia. Global Change Biology 10: 833- 843.

SAS Institute inc. 2001. SAS/STAT version 9.1, user's guide. Cary, NC, USA.

Seyfried, M.S, Rao, P.S.C. 1988. Kinetics of nitrogen mineralization in Costa Rican soils: Model evaluation and pretreatment effects. Plant and Soil 106: 159-169.

Sheng HAO, Y Yusheng, ZJ Yang, G Chen, XIE Nsheng, JIA Gou, and Z Shuangquan. 2010. The dynamic response of soil respiration to land-use changes in subtropical China. Global Change Biology 16: 1107-1121.

Smith KA and F Conen. 2004. Impacts of land management on fluxes of trace greenhouse gases. Soil Use and Management 20: 255-263.

Standford G and SJ Smith. 1972. Nitrogen mineralization potentials of soils. Soil Sci. Soc. Am. Proc. 36: 465-472.

Taylor JP, B Wilson, MS Mills, and RG Burns. 2002. Comparison of microbial numbers and enzymatic activities in surface soils and subsoils using various techniques. Soil Biol. Biochem. 34:387-401.

Van der Werf GR, DC Morton, and RS DeFries. 2009. CO_2 emissions from forest loss. Nature Geoscience 2: 737-738. D.

Vance ED, PC Brookes, and S Jenkinson. 1987. An extraction method for measuring soil microbial, biomass C. Soil Biol. Biochem. 19: 703 -707.

Veenendaal EM, O Kolle, and J Lloyd. 2004. Seasonal variation in energy fluxes and carbon dioxide exchange for a broad-leaved semi-arid savanna (Mopane woodland) in Southern Africa. Global Change Biology 10: 318-328.

Walkley A and CA Black. 1934. An examination of the Degtjareff method for determining soil organic matter, and proposed modification of the chromic acid titration method. Soil Sci. 37: 29 -38.

Wander MM, SJ Traina, BR Stinner, and SE Peters. 1994. Organic and conventional management effects on biologically active soil organic matter pools. Soil Sci. Soc. Am. J. 58: 1130-1139.