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DYNAMIC OF ORGANIC MATTER DECOMPOSITION AND ITS EFFECT ON SOME MICRONUTRIENTS AVAILABILITY IN SOME SULAIMANI SOILS GOVERNORATE

A dissertation
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Agricultural Science / Soil
(Soil chemistry)

By
Kamal Hama Karim Hama

B. Sc. Agriculture, Soil Science
Sulaimani University 1981
M. Sc. Agriculture, Soil Science
Mosul University 1988

Supervised by
Assit. Prof.
Dr. Said Ismail. Al - Sulaivany

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بِسْمِ اللَّهِ الرَّحْمَٰنِ الرَّحِيمِ

وَاضْرِبْ لَهُم مَّثَلَ الْحَيَوٰةِ الدُّنْيَا كَمَاءٍ أَنزَلْنَٰهُ مِنَ
السَّمَاءِ فَاخْتَلَطَ بِهِ نَبَاتُ الْأَرْضِ فَأَصْبَحَ هَشِيمًا
تَذْرُوهُ الرِّيَٰحُ وَكَانَ اللَّهُ عَلَىٰ كُلِّ شَيْءٍ مُّقْتَدِرًا ﴿٤٥﴾

سُورَةُ الْكَهْفِ الْآيَةِ (٤٥)

Cite for them the example of this life as water that we send down from the sky to produce plants of the earth, and then they turn into hay that is blown away by the wind. GOD is able to do all things. [18:45]

Declaration

We certify that this thesis was prepared under my supervision at the department of soil and water, College of Agriculture, University of Sulaimani and hereby recommended to be accepted in partial fulfillment of the requirements for the degree of PhD in Agriculture, Soil Science (Soil Chemistry).

Signature

Supervisor: Dr. Said. I. Al - Sulaivany
Title: Assist. Prof.
Address: Dean of the College of Postgraduate Studies
University of Salahaddin / Erbil
Date: 17 / 12 / 2007

In view of the above recommendation, I forward this thesis for debate by the examining committee.

Signature: Salih N.

Name: Salh N. Majid
Head of Soil and Water Science Department
College of Agriculture
University of Sulaimani
Date: 17 / 12 / 2007

As an examining committee, we certify that we have read this thesis, and examined the student in its contents. In our opinion, it is adequate with **(Excellent)** standing as a thesis for the degree in PhD in Agriculture / soil and water science, (soil chemistry)

Signature:
Dr. Salman Khalaf Essa
Professor
(Chairman)

Signature:
Dr. Shafieq Chlab Al- Kaysi
Professor
(Member)

Signature:
Dr. Mohammad Ali Jamal Al- Obaidi
Assist. prof.
(Member)

Signature: J.K.Kass
Dr. Jabbar Kadhium Kassim
Assist. prof.
(Member)

A. O. Esmail
Signature:
Dr. Akram Othman Esmail
Assist. prof .
(Member)

Signature
Dr. Said I.Al –Sulaivany
Assist. prof.
(Member – Supervisor)

Approved for the College Committee of Graduate Studies

Signature: M. Karim
Dr. May I. Jaff
Assist. prof.
(Dean of the College of Agriculture)
Date: 17 / 12 / 2007

Dedication

I DEDICATE THIS TO:

THE SPIRIT OF OUR PROPHET (MOHAMMAD), THE
SYMBOL JUSTICE, LOYALTY, AND LOVE
BIG KURDISTAN
MY LATE PARENTS
MY SISTERS AND BROTHERS
MY LATE BROTHER (JALAL) WHO ENCOURAGED ME TO
LOVE KNOWLEDGE WITH OUT GETTING A PRODUCT
MY LOVELY WIFE (NIHAYAT) WHO SUFFERED MUCH
DURING THE FULFILLMENT OF THE RESEARCH PROJECT
MY SON (RANJ) WHO ACCOMPANIED ME ALL THE
TIME DURING THE RESEARCH PROJECT
MY DAUGHTERS (REZHNA, DIA, AND HAZHA)

Kamal Hama Karim

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Kamal Hama Karim

Summary and Conclusions

The purpose of this research was to study the factors affecting soil organic matter (SOM) decomposition such as different temperatures, which represent the common ones in different seasons, available water, and application manure rates, at various periods (2, 4, 8, and 12) months. In order to fulfill this purpose two locations have been selected in Sulaimani governorate, and then surface soil samples were taken at (0 – 30) cm at two locations {(Kalar (Shirwana castle), and Arbat (Kharajian village)} in Sulaimani area, Iraqi Kurdistan region. The first location was previously cultivated by okra, while the second one was exploited by wheat; the two locations were varied with respect to texture, rainfed, and total calcium carbonate equivalent, whereas they were similar with respect to organic matter content.

A laboratory incubation experiment was conducted in the soil and water science department, college of agriculture , university of Sulaimani , Iraqi Kurdistan region during (2003 – 2004) in order to measure soil organic carbon dynamics, factorial experiment of five factors was used in completely randomized design (CRD) with two replicates. The factors used were:

- Soil locations (Kalar and Arbat)
- Temperatures (5 , 28 , and 45) C^o
- Sheep manure applications rates (0.5, 1.5, and 2.5) g/100g
- Available water (25, 50, and 100) g/100g
- Incubation periods (2 , 4 , 8 , and 12) months

200 g soils were taken from each soil in a small tightly capped bright jars (13 cm high and 6 cm wide) and treated with (0.5, 1.5, and 2.5) g/100g sheep manure. Moisture content was adjusted to (25 , 50 , and 100) g/100g of available water , and then incubated under different temperatures (5, 28 , and 45 C^o) at different interval incubations (2 , 4 ,8 and 12) months , three periods (4 , 8 , and 12) months were used for determining humic acid (HA) , fulvic acid (FA) , and humin at 28 C^o. TOC g/100g , Eh , EC, pH , and available (Fe, Mn , Cu, and Zn) were measured at (2 , 4 , 8 , 12) months. A sample has been taken at 28C^o and 50 g/100g available water without manure application at each period for the purpose of comparison.

Different mathematical models were used to describe carbon mineralization such as Zero-Order, First – Order, Second –Order, Hyperbolic, Parabolic diffusion, Modified Elovich, Power function, Logarithmic function equations. The model that gives the highest value of determination coefficient (R^2) and the minimum value of standard error (SE) of estimate was considered the best model equation. The solubility diagram was used to determine the solubility of (Fe, Mn, Cu, and Zn) in the soil solution. This study has shown the following results:

Average means of (TSOC remaining g/100g) were (1.674 and 1.816) g/100g for Kalar and Arbat at 5 C^o respectively , whereas they were (1.600 and 1.665) g/100g and (1.337 , 1.048) g/100g at 28 and 45 C^o for the two locations respectively . In Kalar and Arbat locations, 5C^o was superior significantly at $p \leq 0.01$ to 28C^o and 45 C^o with regard to (TSOC remaining g/100g) during incubation periods. Average means of TSOC remaining g/100g were (1.574 and 1.576) g/100g for Kalar and Arbat consequently at 25 g/100g available water. While they were (1.525, 1.515) g/100g and (1.513, 1.438) g/100g for (50 and 100) g/100g of available water at the two locations respectively.

In Kalar location, 25 g/100g available water was superior significantly to (50 and 100) g/100g, whereas it was superior significantly at $p \leq 0.01$ to 100 g/100g available water at Arbat with respect to average means of TSOC remaining g/100g. While 2.5 g/100g manure application rate was superior significantly to (0.5 and 1.5) g/100g with regard to TSOC remaining g/100g for both locations. In terms of incubation period, 4 months incubation period was superior significantly to (2, 8, and 12) periods with regard to TSOC remaining g/100g for Kalar and Arbat locations.

A hyperbolic model compared to the various kinetic models is considered as the best model for describing SOC mineralization since it has given the highest R^2 and the lowest SE of estimate which are (0.870 , .082) and (0.760 , 0.100) for R^2 and SE of estimate at Kalar and Arbat locations respectively. It can be noticed that with an increase of manure application rate will lead to the decrease of half –life's. Since the half-life's is depended on the rate of decomposition, while no differences for the values of half –life's were seen at (25, 50, 100) g/100g of available water for both locations .Whereas the highest values of half life's were 16.021 and 17.147 months at 45 and 28C^o, but the lowest values were 13.667 and 14.009 months at 28 and 45 C for Kalar and Arbat locations respectively.

The decomposition rates (K) were increased with increasing manure application rates and available water except at 100 g/100g available water for both locations, while the

(K) values were decreased with increasing temperatures at Kalar and Arbat locations. Whereas, the Q_{10} values increased with increasing the manure application rates, but nearly remain constant with different available water added. On the contrary, Q_{10} values decreased with increasing temperature for both locations. It can be concluded that SOC decomposition in soils was sensitive to changes in temperature (Q_{10}). Manure application rates at 2.5 g/100g was superior significantly at ($p \leq 0.01$) to (0.5 and 1.5) g/100g in Kalar location with regard to average mean of C – humin g/100g except 1.5 g/100g manure application rates and 8 months incubation periods. While 2.5 g/100g manure application rates and (4, 8) months incubation periods were superior to all others except 1.5 g/100g manure application rate and 4 months incubation periods for Arbat location. On the other hand, there were no significant differences at ($p \leq 0.01$) between different available water with regard to C- humin g/100g for Arbat location. Whereas, at Kalar location (25 and 100) g/100g available water and 8 incubation periods were superior significantly to 50 g/100g While, the interaction of $28C^o$ and 8 months period was superior significantly with regard to C- humin g/100g at ($p \leq 0.01$) to (12 and 4) months period in Kalar location. But in Arbat location $28C^o$ and 4 months incubation period was superior significantly at ($p \leq 0.01$) to (8 and 12) months , with no difference between the latest two periods for Arbat location , whereas there is significant difference between (4 and 12) months in Kalar location with regard to average mean of C – humin g/100g.

The results indicated that the 2.5 g/100g manure application rate was superior significantly with regard to average mean of C- HA g/100g at $p \leq 0.01$ to the other two rates for both locations , also the available water had affected the C – HA g/100g at level 100 g/100g in Kalar location as compared to 50 and 25 g/100g While no significant differences were observed for the three available water percent in Arbat location. The results indicated that 8 months incubation period in Kalar location was superior significantly with respect to C – HA, g/100g at $p \leq 0.01$ to the other two periods, in contrast no significant differences were observed between the three periods in Arbat location. In Kalar location 0.5 and 1.5 g/100g manure application rates were more effective on FA g/100g than 2.5 g/100g, whereas 1.5 g/100g was superior significantly at $p \leq 0.01$ to the other two manure application rates in Arbat location. In the term of available water, (25 and 50) g/100g available water were different significantly from 100 g/100g available water with respect to average mean of C- FA g/100g in Kalar location, but no significant different were noticed for the three available water g/100g in Arbat location. On the other hand, 4 months incubation periods was superior to the other two periods in Kalar location, while 12 months period was superior to the other two periods in Arbat location in contrast to Kalar location.

The results indicated that FA was increased in comparison with HA. This was expected since the rates of decomposition were increased. Generally, the highest HR g/100g values were observed at 4 months period for all the studied parameters, whereas, there are fluctuations at 8 and 12 months. Humification rate percentage expressed as indicator for humification rate, where higher value of HR indicates higher humification process.

The results show that $5C^o$ was superior significantly at $p \leq 0.01$ with regard to average mean of available Fe to 28 and 45 C^o for both locations. On the other hand, the results show that 2.5 and 1.5 g/100g manure application rates were superior significantly at $p \leq 0.01$ with regard to available Fe to the 0.5 g/100g at Kalar location. Whereas in Arbat location 2.5 g/100g manure application rate was superior significantly at $p \leq 0.01$ to the 1.5 and 0.5 g/100g manure application rates. With respect to available water, there were no significant differences at $p \leq 0.01$ between different available water with regard to available Fe for both locations. On the other hand , 2 months incubation period was superior significantly with regard to available Fe at $p \leq 0.01$ to the other three periods (4 , 8 , and 12) months for Kalar location , while 8 months incubation period was superior significantly at $p \leq 0.01$ to the three periods (2 , 4 and 12) months for Arbat location. The results show that $5C^o$ was superior significantly with regard to available Mn at $p \leq 0.01$ to 28 and 45 C^o during incubation periods for Kalar and Arbat locations. With regard to available Mn, the 2.5 g/100g manure application rate was superior significantly to (0.5 and 1.5) g/100g at $p \leq 0.01$ during incubation periods for Kalar and Arbat locations, while the 25 g/100g available water was superior significantly with regard to available Mn at $p \leq 0.01$ to all others for Kalar location. Whereas the 50 g/100g available water was superior significantly with regard to Mn at $p \leq 0.01$ to all others except 25 g/100g available water for Arbat location. In terms of incubation period , 2 months incubation period was superior significantly with regard to available Mn at $p \leq 0.01$ to (4, 8 and 12) months at Kalar and Arbat locations. With regard to available Cu, 5 C^o was superior significantly at $p \leq 0.01$ to 28 and 45 C^o during incubation periods at Kalar location. While 45 C^o was superior significantly at $p \leq 0.01$ to the others except 5 C^o at Arbat location. On the other hand, 2.5 g/100g manure application rate was superior significantly with regard to available Cu at $p \leq 0.01$ to 0.5 and 1.5 g/100g during incubation periods at Kalar and Arbat locations. In addition, the results show that the 25 g/100g available water was superior significantly with regard to available Cu at $p \leq 0.01$ to (50 and 100) g/100g at Kalar location. Whereas, there was no significant differences between different available water with regard to available Cu at Arbat location and, the results show that 12 months was superior significantly with regard to available Cu at $p \leq 0.01$ to (2, 4, 8) periods for Kalar and Arbat locations.

The results show that 5C^o was superior significantly with regard to available Zn at $p \leq 0.01$ to 28 and 45 C^o at Kalar and Arbat locations. On the other hand, the results show that 2.5 g/100g manure application rate was superior significantly with regard to available Zn at $p \leq 0.01$ to (0.5 and 1.5) g/100g at Kalar location, while 2.5 g/100g manure application rate was superior significantly to the others except 1.5 g/100g at Arbat location. Also the results show that the 25 g/100g available water was superior significantly with regard to available Zn at $p \leq 0.01$ to (50 and 100) g/100g at Kalar location, while there was no significant differences between different available water with regard to available Zn at Arbat location . The results show that 2 months incubation period was superior significantly with regard to available Zn at $p \leq 0.01$ to (4, 8, 12) periods at Kalar and Arbat locations.

On the other hand, the solubility of Fe is undersaturated with respect to fresh precipitate at different temperatures, manure application rates, available water, and incubation periods for Kalar and Arbat locations. While Mn solubility is considered undersaturated with respect to $MnCO_3$ at all mentioned parameters except at (25, 50 and 100) g /100g available water, which is between rhodochrosite (MnCO3) and γ – manganite (MnOOH) minerals, at Kalar location. While at Arbat location, the points are located on or below the $MnCO_3$ line,. So the solubility of Mn is undersaturated or in equilibrium with respect to $MnCO_3$ for all parameters. On the other hand, the solubility of copper is oversaturated with respect to Soil - Cu at all mentioned parameters for Kalar and Arbat locations. Finally, both minerals (ZnO and Zn_2SiO_4) may control the solubility of Zn.

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List of abbreviations

ANOVA = analysis of variance
As = arsenic
C / N = carbon nitrogen ratio
C = carbon
^{14}C = radio carbon
C- FA = carbon as fulvic acid
C- HA = carbon as humic acid
C- Humin = carbon as humin
C_a = active carbon pool
CEC = cation exchange capacity
CEC_v = variable charge of cation exchange capacity
C_{min} = mineralized carbon
$Cmol_c$ = centimole of charge
C_o = initial carbon or potentially mineralizable carbon pool
C_r = resistant carbon pool
CRD = completely randomized design
C_s = slow carbon pool
CT = conventional tillage
C_t = total carbon pool
DOM = dissolved organic matter
DTPA = diethylene triamine pentaacitic acid
E_4 = absorbance at 465 nm
E_6 = absorbance at 665 nm
ECe = electrical conductivity of extract
ECEC = effective cation exchange capacity
ESR = electron spin resonance
Exp. = exponential
FC = field capacity
FTIR = Fourier transform infrared
FV = fulvic acid
FVs = fulvic acids
GC = gas chromatography
HA = humic acid
HAs = humic acids
HCL = hydrochloric acid
HR = humification rate
HS = humic substances

I = ionic strength
IOM = inert organic matter
K = decomposition rate
LMW = low molecular weight
mmolc = mmole of charge
MRT = mean residence time
MS = mass spectroscopy
MSE = mean square of error
$Na_4P_2O_7$ = sodium pyrophosphate
NMR = nuclear magnetic resonance
NOM = natural organic matter
NT = no tillage
P = probability
POM = particulate organic matter
ppt. = precipitation
Q_{10} = increase in K value for every 10 C^o increase
R^2 = determination coefficient
SE = standard error
SOC = soil organic carbon
Soln. = solution
SOM = soil organic matter
SSAS = Soil Science Society of America.
$t_{1/2}$ = half life
TEA = triethanolamine
TMAH = tetramethylammonium hydroxide
TOC = total organic carbon
TSOC = total soil organic carbon
USDA = unite state department of agriculture

1. Introduction

World Population is currently estimated 6.3 billions increasing annually by 77 millions, the rapid growth in population has major impacts on land , water , pollution , and other factors related to the food supply, and the importance of soil fertility and conservation continues to increase in order fulfill population's demand. Food security is a major objective, which all nations of the world seek to achieve through the promotion of the agricultural sector vertically and horizontally. The use of chemical fertilizers is important to achieve this object. Increase in prices of fertilizers in recent years, and environmental problems caused by those fertilizers make us hand search for chemical fertilization alternatives. Currently some countries in the world especially the low income ones resort to use organic fertilizers since they are cheap and low harmful for human beings. Global food, fiber, and energy needs continue to escalate with rising population necessitating the cultivation of available land. However, much of the available land can be characterized as marginal in another word 78 % of the total earth's surface area is categorized as unsuitable for agriculture (Lal and stewart, 1992).

Degradation refers to a significant loss of soil quality, which requires specific and significant remedial measures to restore soil functions (Bouma, 1997). In a recent assessment of global soil degradation, the United Nations Environmental Program (UNEP) states that approximately 40 % of agricultural soils have been adversely degraded by anthropogenic causes. Level of productivity is economically unfeasible. Soil organic matter (SOM) strongly affects soil properties such as water infiltration rate, erodibility, water holding capacity, nutrient cycling , retention of nutrient by cation exchange , the availability of nutrients (e.g. , NH_4^+ , NO_3^- , PO_4^{-3} , SO_4^{-2}) trace elements by mineralization , soil buffering capacity , chelates metallic ions which cause the availability of some nutrients , and decreases the toxicity of other ions (Stevenson, 1994; Campbell et al., 1996; Francioso et al., 2000; Wander and Yang, 2000). On a biological level, (organic matter) OM improves microbial activity, adsorbs pollutants such as Pb and Cu and certain pesticides (Brady and Weil , 1996).It has been suggested that proper management of (soil organic matter) SOM is the heart of sustainable agriculture (Weil, 1992). Recent research has also recognized SOM as a central indicator of soil quality and health (Soil and Water Conservation Society, 1995). For example, a decline in SOM (biological oxidation or erosion) significantly reduced the N supply and resulted in a deterioration of soil physical conditions, leading to crop yield reduction (Greer et al., 1996). Therefore, it is important to maintain proper levels of SOM to sustain soil productivity. Intensive agricultural practices change SOM characteristics greatly, generally a substantial loss of soil organic carbon (SOC). Consequently, small changes in the SOM content are significant to the agricultural production of the region.

As there are few studies concerning OM decomposition in our region, the research was selected and its main objectives are:

- study the factors affecting SOM decomposition such as: different temperatures which represent the common temperature in different seasons, available water percent, and application manure rates during incubation periods.
- find accurate models to describe the kinetics of carbon mineralization from OM added under standard laboratory conditions.
- determine the humification rate of SOM decomposition and evaluate the fractionation of SOM and to measure the decomposition rate of SOM decomposition and also to estimate the half-life decomposition of the added organic residue.
- study the relationship of OM fractions and micronutrients (Fe, Mn, Cu, Zn) and to study the effect of OM added on solubility equilibria of some micronutrients (e.g. Fe, Mn, Cu, and Zn)
- predict the best time for adding the manure (sheep) to the soil.

2. LITERATURE REVIEW

2.1 Definition of soil organic matter and its components

Researchers have made several definitions for soil organic matter (SOM) is defined as the organic fraction of soil exclusive of undecayed plant and animal residue (SSSA, 1987). SOM can be classified on a continuum from easily decomposable material (simple sugars and proteins), slowly decomposable or physically protected (cellulose and hemi-cellulose) and recalcitrant or passive materials (lignins, waxes, chitins, and polyphenols) (Gregorich et al., 1994). Stevenson, (1994) defines SOM as including litter, microbial biomass associated with the light organic fraction (< 2.0 g cm^{-3}), and humus. The primary plant nutrient found in SOM is carbon comprising 45 to 58% by weight. The decomposition and resynthesis of carbon and these materials are mediated by microbial activity that comminute plant litter, churn soil, and facilitate nutrient turnover rates. Soil humus is one product of this resynthesis and catabolic processes and is defined as the soil fraction that no longer visibly resembles the organic material from which it was derived (McBride, 1994).

While the term “Soil organic matter” (SOM) has been used in different ways to describe the organic constituents of soil. SOM will be used as defined by Baldock and Skjemstad, (1999) as “all organic materials found in soils irrespective of origin or state of decomposition”. Since SOM consists of C, H, O, N, P and S, it is difficult to actually measure the SOM content and most analytical methods determine the (SOC) content and estimate SOM through a conversion factor. The amount of SOC that exists in any given soil is determined by the balance between the rates of organic carbon input (vegetation, roots) and output (CO_2 from microbial decomposition).

SOM is a complex mixture, which influences a number of soil properties and nutrient cycling, and is itself influenced in kind and amount by land-use, soil type, climate and vegetation (Loveland and Webb ,2002).

SOM can be divided into (i) light fraction, which consists of mineral-free organic matter composed of partly decomposed plant and animal residues, which turn over rapidly and have a specific density considerably lower than that of soil minerals; and (ii) heavy fraction, composed of more processed decomposition products, which turn over more slowly and have a high specific density because of their intimate association to soil minerals (Christensen, 1992; Barrios et al., 1996).

OM in agricultural soils consists mainly of plant biopolymer residues (e.g., polysaccharides, lignin, proteins, and cuticular materials), materials derived from them via decomposition processes, microbial tissues, and humic substances (HS) (Stevenson, 1994). With humification, plant residues are transformed via chemical, biological, and physical processes into more stable forms (humus). Therefore, humification and degradation processes result in a loss of the characteristic signals of structurally identifiable materials (such as plant biopolymers). SOM is a physically and chemically heterogeneous mixture of organic compounds of plant, animal, and microbial origin, and has components at different stages of decomposition.

SOM will be used as defined by Franzmeier et al., (1985); Magdoff et al., (1996); and Paul, (1997) as OM in soils is represented by plant debris or litter in various stages of decomposition through to humus and includes the living organisms in the soil. Above ground, plants (phytomass) are generally excluded from discussions of SOM, but living roots are generally included. The following definitions will be followed:

Soil Organic Matter: Natural C-containing organic materials living or dead, but excluding charcoal.

Phytomass: It is the above ground portion of materials of plant origin usually living, but may also include standing dead trees.

Microbial Biomass: It is the living population of soil microrganisms.

Litter: It comprises the dead plant and animal debris on the soil surface.

Macroorganic Matter: Organic fragments from any source which are > 250µm (generally less decomposed than humus).

Organic Carbon: The carbon content is commonly used to characterize the amount of OM in soils.

Organic matter = 1.724 x percent organic carbon. ------------------ 1

Humus: Material remaining in soils after removal of macro organic matter (generally material that has been more extensively physically and/or biochemically transformed because of soil forming processes than macro organic matter). There are two major classes: the nonhumic substances (e.g. amino acids, lipids, carbohydrates) and humic substances (a series of high-molecular-weight amorphous compounds).

Humic Acids (HAs): Dark-colored amorphous materials that can be extracted from the soil by a variety of reagents, such as strong bases or neutral salts and that are insoluble in dilute acid. This implies that HAs contain primarily acidic functional groups, such as phenolic or carboxylic groups. HAs are composed of molecules with molecular weights in the range 20,000 to 1,360,000. They are considered to be polymerization products of fulvic acids and other decay products.

Fulvic Acids (FAs): The organic materials that are extracted with humic acid but remain in solution upon acidification with dilute acid. This implies that FAs contain acidic functional groups since it is soluble in strong bases and extracted with HAs that FAs also contain basic groups since it remains in solution upon acidification. FAs are composed of molecules with molecular weights in the range 275 to 2,100. They are considered to be decay products of higher plants and microbial residue.

Humin: The strong base insoluble fraction.

SOC is an important index of soil quality because of its relationship to crop productivity. Optimum levels of SOM can be managed through crop rotation, fertility maintenance including use of inorganic fertilizers and organic manures, tillage methods, and other cropping system components (Janzen et al., 1998). Table (1) presents fertility status of OC g /100g in soil.

OM contents for 28 various Iraqi Kurdistan region soils were found to be in the range of (0.280 – 3.350) g / 100g (Qadir, 2001).

Table (1) the interpretation of the status of OC g /100g in the soil and its relationship to the soil fertilities, (adapted from Baruah and Barthakur, 1999).

Organic carbon g /100g	Comments
< 0.5	Low
0.50-0.75	Medium
>0.75	High

Biologically active components of the organic soil fraction include polysaccharides, amino sugars, nucleotides, and organic sulfur and phosphorus compounds. Humus, a water – insoluble material that biodegrades very slowly, makes up the bulk of SOM. The organic compounds in soil are summarized in Table (2).

Table (2) Major classes of organic compounds in soil, (adapted from Manahan, 2000)

Compound type	composition	significance
Humus	Degradation-resistant residue from plant decay , largely C , H , and O	Most abundant organic component, improves soil physical properties, exchanges nutrients, reservoir of fixed N
Fats, resins, and waxes	Lipids extractable by organic solvents	Generally, only several percent of soil organic , may adversely affect soil physical properties by repelling water perhaps phytotoxic
Saccharides	Cellulose, starches, hemi cellulose, gums	Major food source for soil microorganisms, help stabilize soil aggregates
N – Containg organics	Nitrogen bound to humus amino acids , amino sugars, other compounds	provide nitrogen for soil fertility
Phosphorous compounds	phosphate esters, inositol phosphates (phytic acid), phospholipids	Source of plant phosphate

The amount of C stored in soils is twice the amount of C in the atmosphere and three times the amount of C stored in living plants (Kimble and Stewart, 1995).

In reality SOM is a complex material (Fig. 1) which is built up out of several different types of organic species with different characteristics properties.

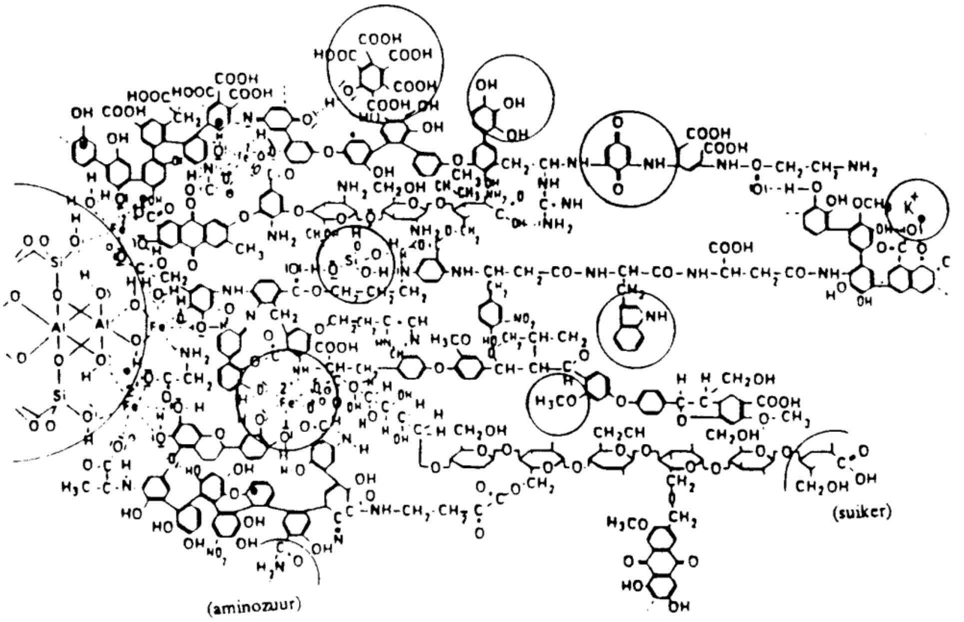


Fig (1) Proposed model for the structure of SOM (adapted from Hoefs and Lagas, 1992).

2.2 Factors affecting soil organic matter decomposition

Several factors influence the processes related to formation and evolution of SOM, including organisms, C/N ratio, climate, clay mineralogy, texture, and agricultural management.

2.2.1 Organisms

Microbial communities are responsible for SOM cycling and thus for maintaining soil fertility. Fungi are heterotrophic plants larger than the bacteria; those that live on the dead tissues of organic substances are saprophytic. Fungi may be regarded as the scavengers who will decompose in soil almost anything of organic nature that bacteria cannot tackle and many of them serve as food for the bacteria. In acid soils, the fungi are the main decomposers of cellulose as under acidic conditions, bacteria and actinomycetes become inactive .Also, *penicillium* and *trichoderma* take part in cellulose decomposition, whereas in other soils the fungi species are *Aspergillus, fusarium, etc*. Certain species of fungi, *Alternaria*, *Aspergillus*, etc.

produce substances similar to humic substances in soil and may be important in the synthesis of soil humus. The black or dark brown color of soil humus may be due to their presence. Actinomycetes have characteristics, which are transitional between bacteria and fungi and sometimes called fungi-like bacteria. Actinomycetes are more abundant in dry soil than in grassland and pasture soils than in the cultivated soils. They are responsible for the decomposition of the more resistant OM of soil and produce a number of dark black to brown pigments, probably contributing to the dark color of soil humus. Microbial biomass is part of the active pool of soil organic matter that plays focal roles in decomposition of organic materials, nutrient cycling, and biophysical manipulation of soil structure (Franzluebbers et al., (1999).

Soil bacteria, the agents of decomposition, use carbon mainly as fuel and nitrogen as building material for their bodies and for the production of the intricate organic compounds that result from their activity. Fresh OM is characterized as a rule by a large amount of carbon in relation to nitrogen. It has a wide carbon-nitrogen ratio, so far as the bacteria are concerned, a wide ratio of fuel to building material. Such fresh material--straw, for example, may have a ratio that is too wide, so that it decomposes very slowly. If the ratio is less wide, decomposition may be more actively carried on. The carbon will then be rapidly used up as fuel while the nitrogen is held or treasured without appreciable loss (USDA, 1938 and Pascual et al., 1999).

Hetrotrophic microorganisms act on organic materials and degrade them in soil. Consequently, nutrients are made available in soil. The extent of availability of nutrient depends not only on the type of organic additives (Debnath et al., 1991), but also on the build up of autochthonous microorganisms (Debnath et al .,1994).Fresh OM is decomposed more quickly first it is attacked by soil fauna, so that it is more easily accessible to fungi and bacteria .Soil fauna such as termites ,earthworms, woodlice, mites and sprigtails ,respire only 5 to 10 % of the decomposable organic C, and their contribution to the total biomass of decomposers is even smaller. Nitrogen added to the soil as fertilizers or crop residues is immobilized by the soil biomass and subsequently transformed into organic forms that may be relatively resistant to mineralization (Broadbent and Nakashima, 1974; Olson and Swallow, 1984).

2.2.2 The C /N ratio

The C / N ratio used to predict decomposition rates. The C/N ratio of SOM, depending on the C/N ratio of the vegetation and degree of decomposition, can vary between 12-16 but may be much higher in plant litter or in environments where SOC decomposition is restricted (e.g. peats) (Baldock and Nelson, 1999; Baldock and Skjemstad , 2000).

Table (3) presents a summary of C/N and lignin/N ratios for various forms of residues (modified from Praveen – Kumare et al., 2003)

Residue quality	C/N	Lignin/N
Highly decomposable	<18	<4.5-5
Moderate	18-27	5.5-7
Slow	28-60	7.5-15
Least	>60	>15

When decay has proceeded to the point where the carbon-nitrogen ratio is significantly decreased, a residue of a more stable nature is produced. Thereafter the carbon-nitrogen ratio is narrower and remains more constant. This corresponds more nearly to the condition that holds in the case of the OM in virgin soils. Its further decay, which is slow because of the relatively low level of carbon, liberates nitrogen in place of storing or preserving it. Because of its high carbon content, the decomposition of fresh OM requires additional soluble nitrogen to be used as building material by the microorganisms, which obtain it from the soil, often exhausting the supply to a degree that is damaging to a growing crop (USDA, 1938).

Some studies have shown, for instance, that elevated CO_2 can cause (i) an increase in soil N availability (Körner and Arnone, 1992; Zak et al., 1993; and Hungate et al., 1997) (ii) decreased N availability, because of N immobilization in high C to N ratio–litter and/or labile organic compounds (Diaz et al., 1993; Cotrufo et al., 1994; Bernston and Bazzaz, 1996) among the added organic wastes, *subabul* leaves released highest amount of nitrogen due to high N content and narrow C / N ratio (Alexander ,1977). As incubation period in creased, mineralization of nutrients increased due to decomposition of added organic wastes . Sarmah and Bordoloi, (1994) showed that of the various sources of OM, the CO_2 evolution was recorded under S.rostrata followed by rice straw and farmyard manure. This variation may be attributed to their varying C / N ratio. S.rotrata had a narrow C / N ratio 15.5 than the rice straw 69.1 and therefore, decomposed more rapidly. Nitrogen rich materials decompose rapidly (Alexander,1961).Contrary to this, farmyard manure decomposed slowly in spite of having a narrow C / N ratio as this had already undergone through process of decomposition. Rapid mineralization of N in soils treated with S. *rostrata was probably* due to narrow C/ N ratio accompanied by rapid decomposition (Parr and Papendick ,1978). On the contrary, conspicuous decrease in mineral N content in soils treated with rice straw may be due to wider C / N ratio of the material and immobilization of N by microorganisms. In this case, the rate of decrease was highest in the initial stages of incubation. Piccolo et al., (2004) showed that the few years under elephant grass has resulted in a significant increase in particulate OC, probably due to the relatively slow rate of decomposition of fresh residue, characterized by high dry matter input and C / N ratios. Singh et al., (1994) found that carbon content of the humin fraction increased from 21.3 to 28.6 % , whereas, it decreased from 34.9 to 23.7 % by lowering the C / N ratio of the rice straw through the application of fertilizer N. Similarly the distribution of carbon was affected in wheat straw amended soil. Also, they suggested that the application of crop residues of wider C / N ratio are likely to stabilize the N in Slowly biodegradable fractions and on along term can improve the health of the soil.

2.2.3 Climate

In the presence of excess water, water logging, anaerobic condition occur the aerobes become suppressed and inactive. In the absence of adequate moisture in soil, some of the microbes die due to tissue dehydration and some of them change their form into resting stages of spores or cysts. Temperature is the most important environmental factor influencing the biological processes and the microbial activity. When the temperature is low, the number and activity of microorganisms fall. Most of the soil organisms are mesophiles and grow well between $15C^o$ and $45C^o$. A temperature of $37C^o$ is considered to be optimum for most mesophiles. Microbes consume oxygen from soil air and give out carbon dioxide. In the absence of such gaseous exchange, carbon dioxide accumulates in soil air and becomes toxic to the microbes. Rate of oxygen intake and simultaneous evolution of carbon dioxide are measures of microbial activity. Direct sunlight is injurious to most of the microorganisms except algae. The accumulation of OM in soil is strongly influenced by temperature and by the availability of oxygen. Since the rate of biodegradation decreases with decreasing temperature, OM does not degrade rapidly in colder climates and tends to build up in soil. In water and in waterlogged soils, decaying vegetation does not have easy access to oxygen, and OM accumulates. The organic content may reach 90% in areas where plants grow and decay in soil saturated with water, (Manahan, 2000). The rate of lignin degradation is less affected by changes in temperature and nitrogen availability than other major plant components during decomposition, (Sarkanen and Ludwig, 1971). Another factor for slow decomposition is that lignin decomposers are relatively slowly growing microorganisms (Witkamp and Drift, 1963). As a result, the rate of mass loss follows a steady - state for the rest of the decomposition process. Soil organic matter decomposition rates are nearly five times greater in warm, wet tropical and subtropical regions than under temperate conditions (Sanchez and Logan, 1992).

The soil moisture content also has a remarkable effect of SOM decomposition and accumulation. Waterlogged soils tend to accumulate OM because the microbial processes, in particular decomposition and mineralization, are slowed down. In aquic moisture regimes, the drainage and soil aeration is poor (anaerobic conditions). Anaerobic oxidation of organic residues is less efficient than aerobic oxidation. If OM is accumulated, the soil development is towards organic soils (Histosols). Histosols generally form in wet, poorly aerated sites, such as shallow lakes and ponds, depression areas, swamps, and bogs and are the end product of natural eutrophication. Alexander, (1977) reported that the optimum water content for maximum activity of soil microorganisms is commonly greatest at about 60 to 80 % of the water holding capacity of the soil.

The water content of the soil as well as the temperature is one of the most important environmental factors affecting the rate of degradation of SOM. Murwira et al., (1990) found that the best decomposition rate of SOM is occurred at %50 of water holding capacity, while more than %50 adversely affected the decomposition rate due to the low activity of organisms.

2.2.4 Texture

Texture, among several other factors, may play an important role in controlling the distribution and activity of soil microbial biomass and, in turn, the size and quality of SOM pools, and the related dynamics of N and S in soil. Separation of the primary SOM pools by particle-size fractionation was expected to contribute to elucidate how different SOM fractions could be affected by texture in Pampean soils (Galantini et al., 2004).

Bayer et al., (2000) reported that SOC associated with sand and silt fractions was less humified than that associated with the finer-size fractions. According to Baldock et al., (1992) the OM in the silt-sized fraction is less decomposed, with higher C/N ratio and higher O–alkyl C concentrations, than OM in the <20-µm fraction. In contrast, the silt- (2- to 20-µm) and clay- (<2-µm) sized fractions had much lower C/N ratios than observed for the larger sized fractions. This is because of microbial alterations of SOM and the stabilization of the finer fractions by microbial products including, fungal hyphae, and bacterial cells or colonies encrusted with clay particles, (Oades and Waters, 1991). As much as 40 to 60% of the microbial biomass may be associated with microaggregates 2 to 20 µm, depending on the amount and type of clay (Monrozier et al. ,1991).

Lanpo et al., (2006) concluded that total SOC was significantly related to the clay plus silt content of soils. The correlation between the proportion of clay and silt sized particles and the amount of SOC associated with this fraction was significantly greater in Black soils compared to Baijiang soils. Yuri et al., (2005) found that the SOC concentration was directly and linearly correlated with the combined clay + silt (but not to clay) content for all depths. The intercept and slope of these linear relations decreased with depth following exponential and logarithmic functions ($P < 0.001$, $R^2 = 0.97$ and 0.95, respectively). Roscoe. et al., (2001) observed that 89–91% of the TOC accumulated in the clay+silt fraction. Hevia et al .,(2003) found that potential SOM losses can be larger in fine-textured soils (up to 54.3 Mg C ha^{-1}) than in coarse-textured (up to 35.7 Mg C ha^{-1}). Alessandra et al., (2002) found that silt and clay fractions contained the largest amount of organic carbon (60–95% of total organic carbon), and free light fraction was the most sensitive pool of organic carbon to detect changes in SOM due to soil tillage and crop rotations. Bayer et al., (2006) concluded that the original C stocks in 0–20 cm layer of soils under native Cerrado were higher in the clayey (54.0 Mg ha^{-1}) than in the sandy clay loam soil (35.4 Mg ha^{-1}), suggesting a higher physical stability of OM associated with variable clay minerals in the clayey Oxisol. Yong, (2007) concluded that sandy soils had greater C and N sequestration than fine-texture soils. The proportion of C and N in POM to total organic C and N increased as sand content increased. It is evident from this study that the soils of northwest China have large potential to sequester C and the conversion of crops to alfalfa could be one of the effective strategies to improve C and N sequestration in this region.

Igwe and Nwokocha, (2006) found that micro aggregate-associated SOC was also low with most of the SOC protected by the <63 µm fractions. Clay content was negatively correlated with <2, 63–2, and <63 µm associated SOC (*r* = –0.45*, –0.42*, –0.40*, respectively). The mineralization rate of clay SOM is twice that of observed for silt. OM associated with clay accounts for most of the mineralization in soils (about 60%), silt contributes about 30% and sand 10%. The greater mineralization rate from clay than from silt-associated SOM may result from differences in the bioavailability of the particle-bound SOM (Christensen, 1992).

2.3 Functions of SOM in soils

The functions of SOM can be broadly classified into three groups: biological, physical, and chemical (Fig.2). These groups are not static entities and dynamic interactions occur between these three major components.

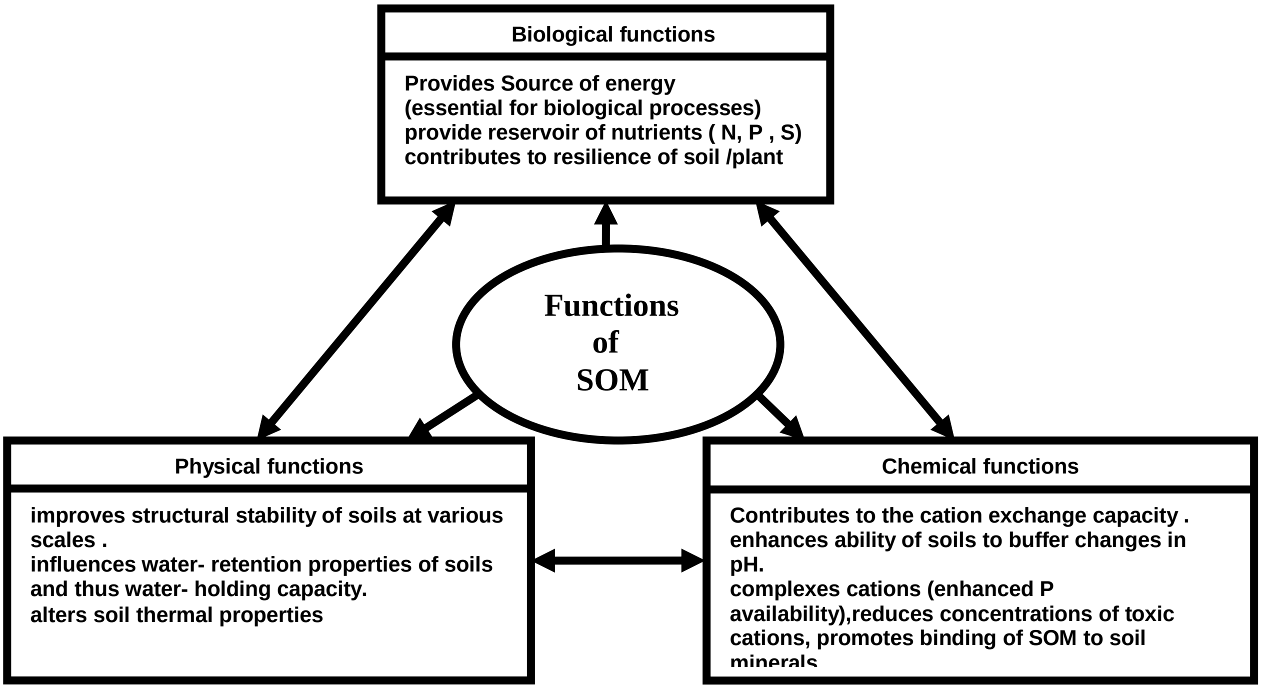


Fig. (2) Functions ascribed to SOM. Note that interactions occur between the different soil function (modified from Baldock and Skjemstad, 1999)

It is well established that addition of SOM can not only reduce bulk density (D_b) increase water holding capacity, and effectively increase soil aggregate stability.

For example, De Jong, (1983) and Haynes and Naidu, (1998) found an increase in water content with increasing SOC content and Wolf and Snyder, (2003) stated that an increase of 1% SOM can add 1.5% additional moisture by volume at FC. The effect of usually dark brown or black SOM on soil color is important not only for soil classification purposes, but also for ensuring good thermal properties, which in turn contribute to soil warming and promote biological processes (Baldock and Nelson, ,1999). Krull .et al., (2004) showed that SOM contributes increase in the variable-charge CEC (CEC_v) and can account for up to 70% of the ECEC (at the pH and soil solution concentration of the soil in the field. This is known as the ECEC) in highly weathered soils. Functional groups (e.g. carboxylic acids) of SOM are believed to be one of the main contributors to CEC_v as they provide negatively charged sites. The availability of different functional groups (e.g. carboxylic, phenolic, acidic alcoholic, amine, amide, and others) allows SOM to act as a buffer over a wide range of soil pH values. pH contributes to CEC as dissociation of functional groups at pH>5 increases the number of negatively charged sites; in addition a decrease in CEC at low pH might be related to blockage of exchange sites by Fe and Al. Apart from functional groups, smaller particle size fractions (especially the organo-mineral clay fraction) had a greater influence on CEC than coarser fractions. In addition, plays a role for retention pesticides and other organic chemicals, chemical fertilizer and manure application can both increase the CEC of the soil. Soil organic matter is a large reservoir of C that can act as a sink or source of atmospheric CO_2 (Lugo and Brown, 1993). It is also an important source of inorganic nutrients for plant production in natural and managed ecosystems.

2.4 Soil organic matter pools and SOC analytical methods.

Due to the difficulties in measuring SOM directly, it is substituted by measurement of SOC (Baldock and Skjemstad, 1999). A convenient way to calculate SOM is by multiplying the OC % by a factor; however, conversion factors vary between 1.4 and 3.3 (Rasmussen and Collins, 1991) and this large range is due to the inherent differences between soils. Most commonly, a conversion factor of 1.72 is used (Baldock and Skjemstad, 1999). Therefore, to ensure consistency and allow reliable comparison of data, it is advantageous to report results as SOC rather than as SOM. Determination of SOC by wet oxidation is typically made by acid dichromate oxidation (Kalembasa and Jenkinson, 1973), also known as the Walkley-Black method (no heating) or Heanes method (externally heated and addition of concentrated sulphuric acid). However, several workers have found that the Walkley-Black method does not determine organic C quantitatively and depending on soil type, recoveries can vary from 56% to 100% (summarized in Skjemstad and Taylor, 1999). Determination of SOC by dry combustion converts all carbon in the presence of oxygen to CO_2 during a heating process.

2.5 Extraction and fractionation of soil organic matter.

The structure techniques used for SOM analysis are as follows:

-Extraction technique

-Purification technique

-fractionation technique

- Characterization technique

The extraction part deals with methods used to get the SOM separated from the soil mineral matter. As far as extraction technique are concerned there are two categories: bulk extractions and sequential extraction methods. A humin extraction method was also found. The Purification part is necessary because of the incomplete separation of the OM from the mineral matter, and necessary to lower the ash content and make functional groups accessible to the reagents used in subsequent characterization steps. Fractionation techniques are all more or less based on the chromatography principle and can be divided into the following categories: fractionation based on differences in molecular size, chemical properties, and mobility in electrical fields. In the field of characterization techniques, two divisions can be made: spectroscopical methods were employed to study the chemical structure of bulk SOM (NMR, ESR, FTIR, TMAH, MS, GC, Fluorescence, and E_4 / E_6 ratio) Table (4) summarized different extractants used for extracting organic soil constituents.

Table (4) Reagents used for extraction of organic constituents from soil (Stevenson 1982)

Type of material	Extractant	Organic matter extracted %
Humic substances	NaOH	to 80 %
	Mild extractants	
	$Na_4P_2O_7$ and other	
	Organic chelates: acetyloacetone, cupferron hydroxyquinoline	to 30 %
	Formic acid (HCOOH)	to 55 %

From these data, it is evident that, extraction of HS from soil with NaOH solution leads to recovery of approximately two – thirds of the SOM. It is known that the amount of OM extracted from soil with caustic alkali increase with time of extraction, but the longer the extraction period and the more alkaline solution; the greater will be the chemical changes (Tatjana et al., 2001).

Generally, SOM fractionation procedures seek to partition the humified and non-humified substances. Two categories that have been predominantly used in laboratory analysis to isolate OM are chemical and physical fractionation. Classical chemical fractionation is based on the different solubility of organic material in alkali and/or acid solutions such as NaOH (sodium hydroxide) or $Na_4P_2O_7$ (sodium pyrophosphate) (Turchenek and Oades, 1979; Stevenson, 1994; Cheng and Mollina, 1995). When dealing with a soil source, separation of HAs and FAs from humin is usually obtained by alkaline extraction. Undesirable feature of this isolation method is that, overcome by isolation in N_2 atmosphere. However, for isolation of even less altered material, the presence of O_2 in the very alkaline extraction solution should not be neglected. NaOH and $Na_4P_2O_7$ at a 0.5 *N* to 0.1 *M* concentration, both extractants have a high pH, which disrupts H- bonds and dissolves SOC very effectively, and more humic micelles will be dispersed into solution.

In addition, pyrophosphate forms complexes with Al and Fe, so that these metals are removed from organic complexes, removing the materials that are associated with polyvalent cations. Subsequent acidification of the extract to a pH of 2 divides the substances into three fractions: humic acid (HA), fulvic acid (FA), and humin. Humic acids are soluble in alkali solutions but precipitate at a pH of 2. Humic acids are the most stable fractions and increase in predominance as SOM degrades (Paul and Collins, 1998). FAs are soluble in both alkaline and acid solutions, but not at neutral pH (Sanchez, 1998). Humin is insoluble in water at all pH levels. Sanchez, (1998) states that the humin fraction is closely associated with clay particles and tightly bound by polyvalent cations. The limitations of chemical fractionation based on solubility are that alkaline extracts dissolve silica altering the soils structural framework and exposing organic constituents to oxidative processes thereby changing the inherent nature of the substance. The procedure is laborious, time-consuming, and only suitable for a very small number of samples. These three fractions have varying turnover rates with FA 23 having a rate of a few hundred years and the other two fractions even slower at thousands of years (Collins et al., 1997). As an alternative to chemical fractionation, physical fractionation procedures are used to recover the light fraction and partially decomposed products, to determine the role of different OM in formation of water-stable aggregates, and to establish the biological and environmental significance of OM in organo-mineral complexes (Stevenson, 1994). Elliott and Cambardella, (1991) suggest that chemical extraction methods provide information on the type of OM present in the extracts, but physical separations provide information on where the OM is located within 24 the soil matrix. Generally, physical fractionation is achieved by disruption or separation. Disruption involves three methods: sonication, shaking, and chemical. Methods used under separation techniques include dry and wet sieving, sedimentation, and density flotation. Recent studies have attempted to physically isolate the non-humified SOM components, the light fraction and macro-organic matter, according to particle size or specific density (Spycher et al. ,1983; Strickland and Sollins ,1987; Wander et al. ,1994; Bremer et al,. 1995; Ellert and Gregorich, 1995; Meijboom et al., 1995; Barrios et al., 1996; Barrios et al., 1997). The light fraction has been recognized as an important substrate for biologically mediated soil processes, and as an indicator of soil fertility. Christensen (1992) emphasizes that physical fractionation methods are chemically less destructive, than the classical solubility extraction with acid and base reagents, so the soils resemble their natural state and function.

2.6 Humification of soil organic matter

In fact, soil is a complex, multicomponent system of interacting materials, and the properties of soil result from the net effect of all these interactions. One of the major problems in communicating in the field of humic substances (HS) is the lack of precise definitions for unambiguously specifying the various fractions. Unfortunately, the terminology is not used in a consistent manner. The term humus is used by some soil scientists synonymously with SOM that is to denote all organic material in the soil, including HS. Contemporary, the term humus is frequently used to represent only the HS. The term SOM is generally used to represent the organic constituents in the soil, including undecayed plant and animal tissues, their partial decomposition products, and the soil biomass. Thus, this term includes:

(i)-identifiable, high-molecular-weight organic materials such as polysaccharides and proteins.

(ii)-Simpler substances such as sugars, amino acids, and other small molecules, humic substances.

It is likely that SOM contains most if not all of the organic compounds synthesized by living organisms. Humus is separated from the non HS such as carbohydrates (a major fraction of soil carbon), fats, waxes, alkanes, peptides, amino acids, proteins, lipids and organic acids by the fact that distinct chemical formulae can be written for these non HS. Most small molecules of non-humic substances are rapidly degraded by microorganisms within the soil. In contrast soil humus is slow to decompose (degrade) under natural soil conditions. When in combination with soil minerals soil humus can persist in the soil for several hundred years. Humus is the major SOM component, making up 65% to 75% of the total. Humus assumes an important role as a fertility component of all soils, far in excess of the percentage contribution it makes to the total soil mass. Distribution SOM is shown on the following picture.

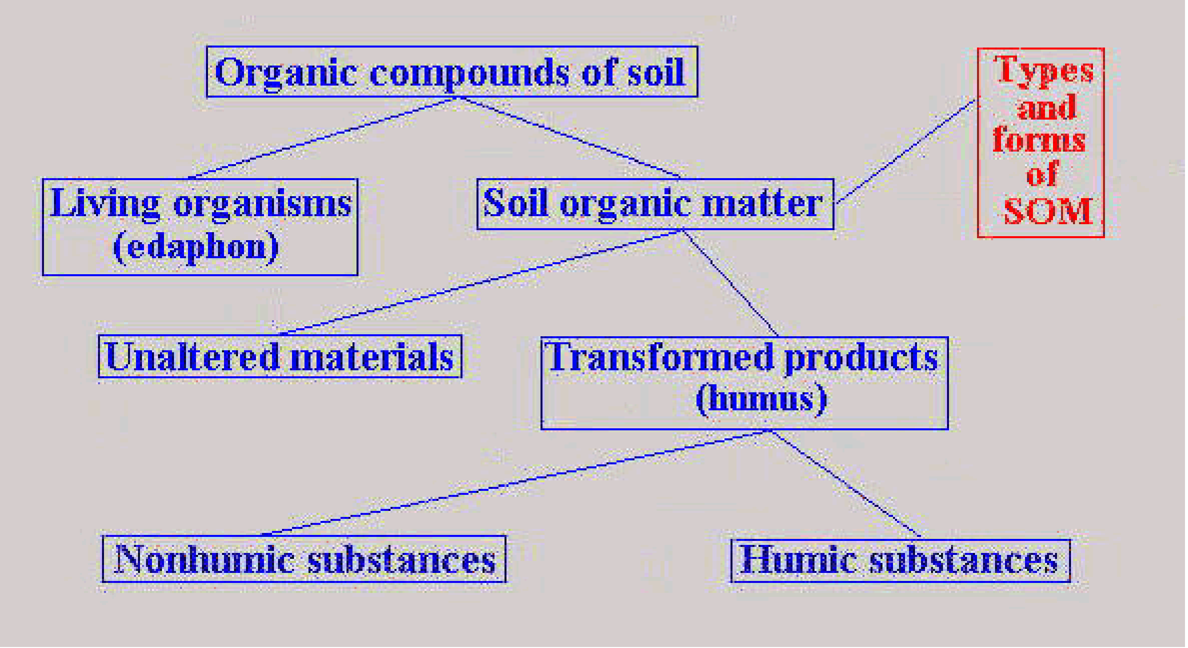


Fig (3) Distribution of SOM compounds.

Organic compounds of soil - live organisms and their undecomposed, partly decomposed, and completely decomposed remains as well as products of their transformation.

Living organisms alive - edaphon.

SOM - non-living components which are a heterogeneous mixture composed largely of products resulting from micro bal and chemical transformations of organic debris. SOM can exist in different morphological patterns, which are the bases of the classification of so-called forms and types of humus.

Unaltered materials - fresh and non-transformed components of older debris.

Transformed products - (humus) - bearing no morphological resemblance to the structures from which they were derived. These transformed components are referred to as the humification process products.

HS are the most widespread natural organic materials occurring in soils, waters and sediments. They are regarded as complex chemical structures, which are more stable than their precursors and have lost their chemical characteristics. They are generally divided into three classes of materials on the basis of their alkaline and acid solubility (FAs which are soluble under all pH conditions, HAs soluble at pH >2 and humin insoluble at all PHs). Humin and humic acids generally represent the major fraction of HS and appear to display similar analytical characteristics and chemical structure (Schulten and Schnitzer, 1997). Soil HAs are mainly derived from the biochemical degradation of plant and animal residues and from microbial synthetic activity and constitute a significant fraction of the SOM (20% of the TOM (Schulten and Schnitzer, 1997; Grasset and Ambles, 1998 a, b). While, humic substances (HS) were defined by Stevenson (1994); Ruttimann-Johnson and Lamar, (1996); Shevchenko and Baily, (1996) as follows: HS are relatively high molecular – weight, dark – colored organic materials that are produced during chemical and biological transformation of plant, animal, and humin wastes. HS are major organic components of soils, natural waters, marine and lake sediments and coals. The end product of composting HS resembles SOM. It has been suggested that the rapid humification process that takes place during composting can serve as a model for the process in soil (Jimenez and Garcia, 1992; Ciavatta et al., 1993; and Adani et al., 1995). On the other hand, Stevenson, (1994) hypothesized that the formation of polyaromatic HS is associated with phenoloxidase, laccase, and peroxidase activities in the soil. Laccase - like enzymes oxidize the phenolic nucleus by one electron, forming a phenoxy free radical, which causes spontaneous polymerization (Suflita and Bollag, 1981;

Thurston, 1994; Yarpolov et al. 1994; and Youn et al., 1995; Call and Mueke, 1997). The synthesis of polyphenols is an essential step in the formation of HS from phenols, quinones, carbohydrates, and N sugars. Lignin is the most probable source for much of the carbon in HA and FA in soils.

Humic substances- a series of relatively high -molecular-weight, brown to black colored substances formed by secondary synthesis reactions. The term is used as a generic name to describe to colored material or its fractions obtained on the basis of solubility characteristics:

Humic acids - the fraction of HS that is not soluble in water under acidic conditions (pH < 2) but is soluble at higher pH values. They can be extracted from soil by various reagents and which is insoluble in dilute acid. HAs are the major extractable component of soil HS. They are dark brown to black in color.

Fulvic acids - the fraction of HS that is soluble in water under all pH conditions. They remain in solution after removal of HA by acidification. FAs are light yellow to yellow-brown in color.

Humin - the fraction of HS that is not soluble in water at any pH value and in alkali. Humins are black in color.

Many investigators now believe that all dark colored HS are part of a system of closely related, but not completely identical, high - molecular - weight polymers. According to this concept, differences between HAs and FAs can be explained by variations in molecular weight, numbers of functional groups (carboxyl, phenolic OH), and extent of polymerization.

Table (5) chemical characteristics of HA and FA fractions of soil humus (adapted from Stevenson, 1994)

Property	Humic Acid	Fulvic Acid
Color	black	yellow
Molecular weight (Daltons)	80% > 10^5	80% < 10^3
Elemental analysis (%)		
Carbon	57	49
Oxygen	34	45
Hydrogen	4	5
Nitrogen	3	2
Sulfur	1	2
Phosphorous	0.3	0.3
Functional groups		
Total acidity (Cmol kg^{-1} dry ash free)	700	1,200
Carboxyl (Cmol kg^{-1} dry ash free)	400	800
Phenolic hydroxyl (Cmol kg^{-1} dry ash free)	300	400
Alcoholic OH (Cmolkg $^{-1}$)	345	600
Quinoid (Cmolkg $^{-1}$)	255	600
Ketonic (Cmolkg $^{-1}$)	285	270
Methoxy	55	75
Polyphenolics	high	low
Polysaccharides	low	high

The low - molecular - weight FAs have higher oxygen but lower carbon contents than the high - molecular - weight HAs. FAS contain more functional groups of an acidic nature, particularly COOH. The total acidity of FAs (1200 Cmol kg^{-1}) are considerably higher than for HAs (700 Cmol kg^{-1}).

Another important difference is that while the oxygen in FAs can be accounted for largely in known functional groups (COOH, OH, C=O), a high portion of the oxygen in HAs seems to occur as a structural component of the nucleus. Is apparent that HS consist of a heterogeneous mixture of compounds for which no single structural formula will suffice. HAs are thought to be complex aromatic macromolecules with amino acids, amino sugars, peptides, aliphatic compounds involved in linkages between the aromatic groups. The hypothetical structure for HA, shown in figure (4) , contains free and bound phenolic OH groups, quinone structures, nitrogen and oxygen as bridge units and COOH groups variously placed on aromatic rings.

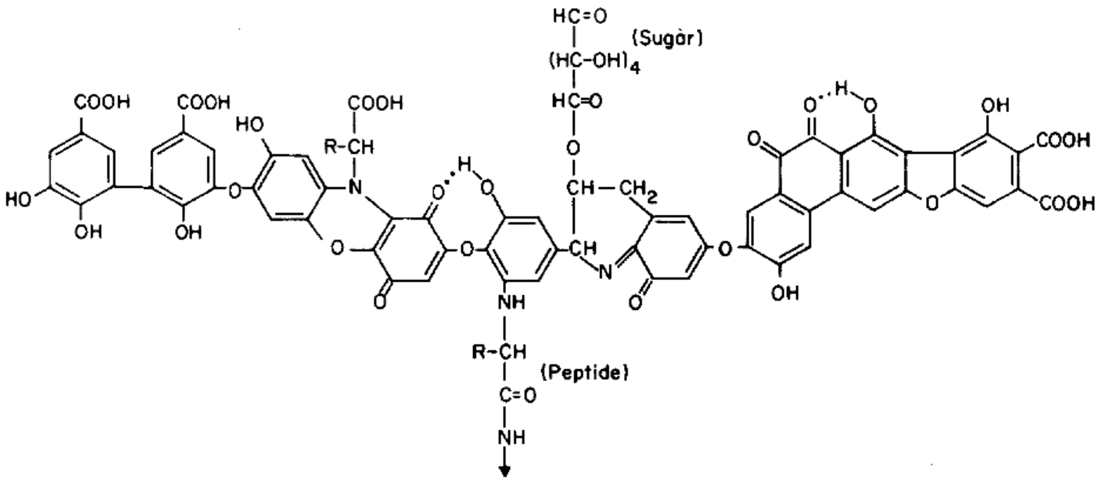


Fig (4). Structure of HA proposed by Fuchs
(Stevenson, 1994).

The hypothetical model structure of FA (Buffle's model) contains both aromatic and aliphatic structures, both extensively substituted with oxygen - containing functional groups.

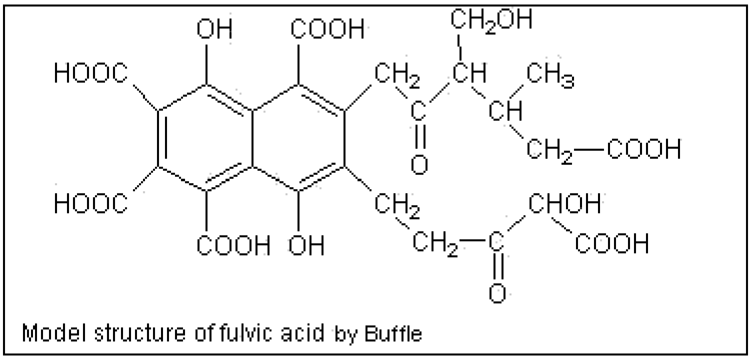


Fig (5) Structure of FA proposed by Buffle (Stenvenson, 1994)

The percentage of the humus which occurs in the various humic fractions varies considerably from one soil type to another. The humus of forest soils is characterized by a high content of FAs while the humus of peat and grassland soils is high in HAs. The HA / FA ratio usually, but not always, decreases with increasing depth.

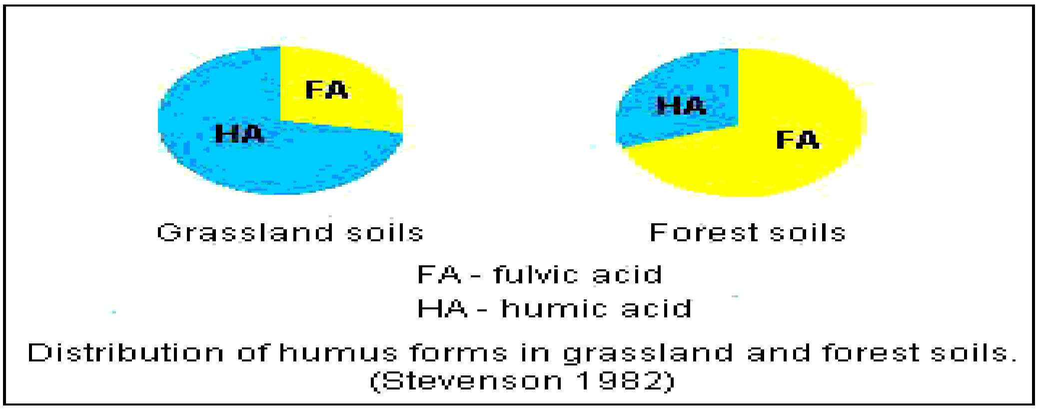


Fig (6) distribution of humus forms

The process by which humus is formed is called humification. Soil humus is similar to its lignin precursors, but has more carboxylic acid groups. Part of each molecule of HS is nonpolar and hydrophobic, and part is polar and hydrophilic. Such molecules are called amphiphiles, and they form micelles in which the nonpolar parts compose the inside of small colloidal particles and the polar functional groups are on the outside. Amphiphilic humic substances probably also form bilayer surface coatings on mineral grains in soil. An increase in nitrogen/carbon ratio is a significant feature of the transformation of plant biomass to humus through the humification process. This ratio starts at approximately 1/100 in fresh plant biomass. During humification, microorganisms convert organic carbon to CO_2 to obtain energy. Simultaneously, the bacterial action incorporates bound nitrogen with the compounds produced by the decay processes. The result is a nitrogen/carbon ratio of about 1/10 upon completion of humification. As a general rule, therefore, humus is relatively rich in organically bound nitrogen. HS influence soil properties to a degree out of proportion to their small percentage in soil.

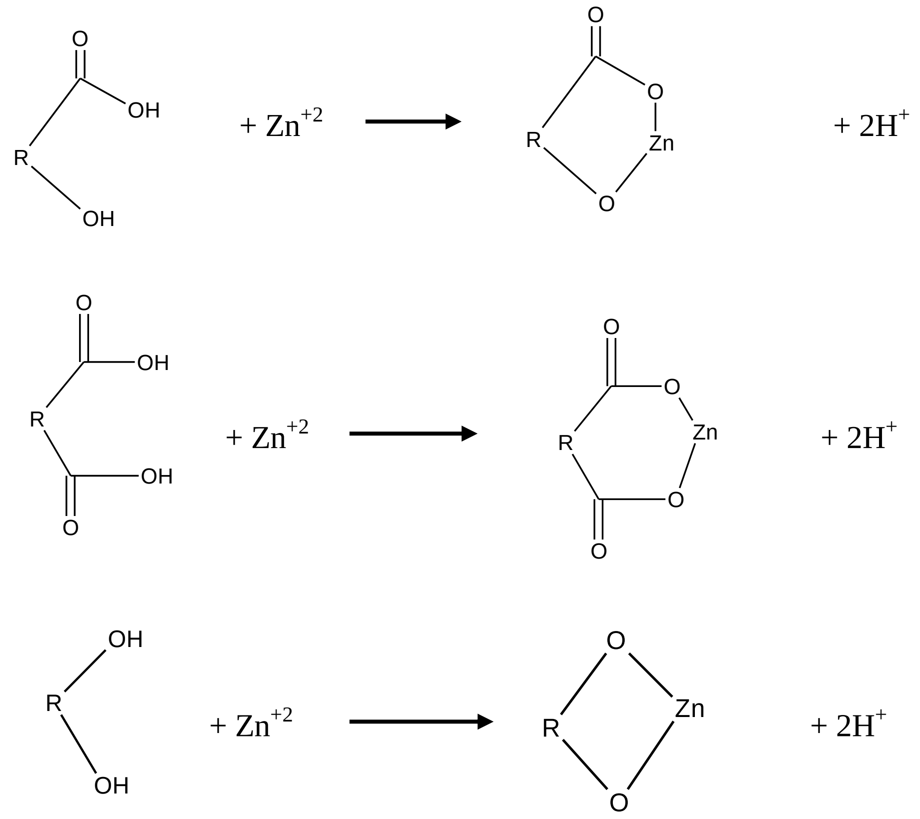
2.7 Relationship of organic matter fraction with micronutrients (Fe, Mn, Cu, and Zn)

A chelate complex is formed when two or more coordinate positions about the metal ion are occupied by donor groups of a single ligand to form an internal ring structure. In soil role of ligands fulfillment simple organic compounds and functional groups of humic substances. The order of decreasing affinity of organic groupings for metal ions is approximately as follows:

R-C= C-OH > -NH_2 > -N=N- > =N > -COO^- > -O- > C=O
enolate amine azo ring N carboxylate ether carbonyl

The relative ability of transition metal ions to form complex ions is quite regular in order:
$Mn^{2+} < Fe^{2+} < Co^{2+} < Ni^{2+} < Cu^{2+} < Zn^{2+}$ for the divalent cations and $Cr^{3+} = Mn^{3+} > Fe^{3+} < Co^{3+}$ for the traivalent cations.The strongest complexing divalent cation is Cu (II) , while Fe(III) is the weakest complexing trivalent transition metal ion , but is stronger than other trivalent cations such as Al^{3+} and the lanthanides, (Bohn, et al.,1985)

The complexing ability of HAs and FAs results largely from their content of oxygen-containing functional groups, such as COOH, phenolic OH and C = O group. Soil organic constituents form both soluble and insoluble complexes with metal ions and thereby play a dual role in soil. Low - molecular - weight compounds (biochemicals, fulvic acids) bring about the solubilization of metal ions and affect their transport to plant roots. In contrast, high - molecular - weight compounds (e.g. humic acids) function as a "sink" for polyvalent cations. Natural complexing agents are of considerable importance in weathering processes and in the movement of sesquioxides into the subsoil (Stenvenson, 1982).Many organic compounds found in the soil form complexes with various metal ions in soil and water. When a metal ion such as Zn^{+2} combines with an electron donor (certain organic compounds), the substance formed is known as a complex or chelate. Examples of Zn^{+2} chelates are shown in Fig. (7).



R=Soil humus

Fig (7) Possible Zn^{+2} chelates formed with soil organic matter functional groups. (adapted from Sylvia , et al. , 2005)

The bond between the electron – accepting metal (Zn^{+2}) and the electron-donating group may be covalent. Metal complexes are important in micronutrient and heavy metal availability to plants. It is generally thought that most metal complexes are more bioavailable to plants and other organisms and more mobile in soil than the corresponding ionized form. Micronutrient availability is facilitated by chelate formation, and it has demonstrated that up to 75% of the copper in soil solution in a chelated form. However, chelate formation by heavy metals such as nickel and cadmium would be undesirable if it resulted in increased plant uptake, certain organic compounds in SOM form chelates with Fe and Al. The chelates facilitate the downward movement of Fe and Al through the soil profile in a process known as cheluviation. The chelates are subsequently deposited in the lower soil horizons, and the resulting redistribution of Fe and Al is important in soil profile development, (Sylvia, et al., 2005).

Trace elements in livestock and poultry manure by-products have recently been reviewed by Bolan et al., (2004). Trace elements including As, Co, Cu, Fe, Mn, Se, and Zn are added as feed supplements to prevent diseases and to improve weight gain and feed conversion· In general, the largest enrichment of manure occurs for As, Cu, and Zn (Bolan et al., 2004). OM is the main adsorptive phase to sequester cationic metals in manure by-products. Sequential extraction results suggest that most (>60%) Cu and Zn is organically complexed in dairy cattle and poultry manure Bolan et al. (2003, 2004). Phenols are considered important agents for Cu complexing by SOM (Lewis and Broadbent, 1961; Senesi, 1992). Thus, their prevalence in submerged soils may well contribute to the greater abundance of Cu^{+2} complexes and perhaps also to the greater number of Fe^{+3} complexes in these HA samples. Further, Cu-specific adsorption can promote aggregation of polyaromatic molecules (Bartoli et al., 1987), which might affect cycling of other SOM-bound nutrients. Carboxyl groups can also complex Cu^{+2}, Fe^{+3}, and VO^{+2}, but their contribution to increased metal complexation in submerged soils is uncertain, given their decreasing relative abundance with increasing soil submergence as found here and previously by ^{13}C NMR (Olk et al., 1996). An increase in soil pH, for example, increases the precipitate form of Cu, while decreasing the organically bound form (Sims, 1986; Sims and Kline, 1991; Zhu and Alva, 1993). The higher pH and carbonate contents may also facilitate Cu precipitation as $Cu(OH)_2$ and/or $CuCO_3$ (Lindsay, 1979). In the presence of DOM, the Cu sorption capacity decreased markedly for both soils, but the effect on the calcareous soil was greater than that on the acidic sandy loam. It was assumed that DOM might complex with Cu in different binding forms at various pH values as follows (Qin and Mao, 1993 ;McBride, 1994) :

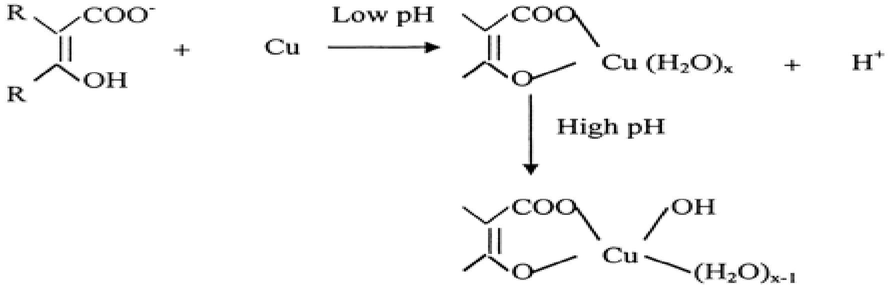


Fig (8) Complexation between DOM and copper in different binding forms at various pH.

Hydroxyl groups bound with Cu could be easily ionized in high pH to yield a negative charge. Consequently, the Cu–DOM complex bearing a negative charge would be repelled by the soils of the same charge through which Cu sorption was reduced. Copper is one of the metals of chief concern in this regard, as it has been shown to be particularly toxic to microbes, with many species of fungi and N-fixing bacteria being especially sensitive (McGrath et al.,1988,; Chaudri et al.,1992; Dahlin et al.,1997). Copper is also one of the potentially toxic metals that is present in relatively high concentrations in many biosolids (commonly 500–3000 ug /g, dry weight). This, combined with the knowledge that Cu associates strongly with organic matter that can degrade over time, suggests that it is vital to determine the likely long-term impact of Cu and other sewage biosolid metals on soil and ecosystem health so that applications of the material can be kept at safe levels. Almas and Singh, (2001) concluded that the concentration of soluble Zn increased correspondingly with the OM addition (from 1.69 to 5.76 and from 22.83 to 23.93 µmol kg^{-1} soil in extracts of clay loam and loamy soils respectively). The addition of OM to loamy soil may thus have increased the uptake in two ways. First, the addition of soluble organic acids may have competed with other reversible sorption phases for the metals, thereby increasing the potential mobility of the investigated metals by formation of soluble organo–metallic complexes. Second, the addition of fresh pig manure, which was the OM source used, improved the growth conditions through the additional nutrients provided and, by this practice, increased the uptake of Cd and Zn. Although the OM source provided contained Cd and Zn, the addition of these metals was relatively small as compared with the amount of naturally present metals in clay loam soil. Karen and Erich ,2003) suggested that copper binding capacity was modeled as involving the noncompetitive interaction of two binding site classes: (i) a strong binding (L1) class defined by inner-sphere complexation and (ii) a weak binding (L2) class defined by outer-sphere complexation.

The model equation is defined as:

$$Y = (B_{\max1}X)/({}^{c}K_{Cu1} + X) + (B_{\max2}X)/({}^{c}K_{Cu2} + X) \text{ ------- } 2$$

and assumes 1:1 binding stoichiometry. The terms B_{max1} and ${}^{c}K_{Cu1}$ and B_{max2} and ${}^{c}K_{Cu2}$ refer to the maximum binding capacities and conditional stability constants of binding site classes L1 and L2, respectively. The independent (*X*) and dependent (*Y*) variables define the initial solution Cu concentration and the corresponding equilibrium concentration of organically complexed Cu, respectively. In some cases, there is a strong interaction between the organic and inorganic portions of soil. This is especially true of the strong complexes formed between clays and HA (FA) compounds. In many soils, 50-100% of SOC is complexed with clay. These complexes play a role in determining the physical properties of soil, soil fertility, and stabilization of SOM.

One of the mechanisms for the chemical binding between clay colloidal particles and humic organic particles is probably of the flocculation type in which anionic organic molecules with carboxylic acid functional groups serve as bridges in combination with cations to bind clay colloidal particles together as a flocculation. Support is given to this hypothesis by the known ability of NH_4^{+1}, Al^{+3}, Ca^{+2}, and Fe^{+3} cations to stimulate clay-organic complex formation. The synthesis, chemical reactions, and biodegradation of humic materials are affected by interaction with clays. The lower-molecular-weight FAs may be bound to clay, occupying spaces in layers in the clay (Manahan, 2000). Plaza et al., (2005) showed that complexation reactions involving (HS) play a key role in the speciation of Cu (II) ions in soil and, therefore, in controlling its biological availability, accumulation and mobility. Khan, et al.,(2005), concluded that the desorption of Cu decreased with increasing soil pH. The irreversible retention of Cu might be the result of complex formation with Cu at high pH in pasture soils. JoaquÃn et al., (2005) analyzed that the effect of the application of the solid phase of pig slurry to agricultural calcareous soils on the input and availability of Fe, Mn, Cu, and Zn. The experimental design was based on a field experiment using, as fertilizers two manures derived from the solid phase of pig slurry (composted, [C] and noncomposted [NC]). Soil samples were taken at 0, 14, 33, 53, 73, and 103 days. Available Fe, Cu, Mn, and Zn were extracted with DTPA (0.005 M DTPA, 0.01 N $CaCl_2$, and 0.1 N TEA at pH 7) and determined by using atomic absorption spectrophotometry. A significant effect of manure application on available Fe, Cu, Mn, and Zn was observed. Zn and, to a lesser degree, Fe DTPA contents in amended soils were increased by composted and noncomposted pig manure, and Mn was increased by the noncomposted material. Extractable Cu was lower in amended soils, than in the controls. In general, the composted pig manure induced less variation of the studied micronutrient contents in amended soils than the noncomposted manure. *Sinha* et al., (1978) concluded that soils which undergo alternate cycles of reversible, oxidation-reduction processes were found to maintain higher solubility of iron and manganese than other soils.

2.8 Decomposition of soil organic matter.

In the first stage of decomposition, litter reaching the soil surface decreases in volume due to the activity of earthworms and other soil animals (Lee, 1974; Striganova and Bienkowski, 2000; Topoliantz et al., 2002). Following this stage, microbial decomposition and intense leaching of solublizied organic substances are mostly responsible for litter breakdown (Lee, 1974). The next stage is decomposition of organic substances by microbial cells producing CO_2, NH_3, and organic acids (Chen et al., 2001; Haynes and Mokolobate, 2001). Awide range of organic compounds are produced by microorganisms. They include low molecular organic acids or ligands such as malonic, fumaric, succinic acids (Cieslinski et al., 1997), and soluble humic molecules (Conte and Piccolo, 2002; Negre et al., 2002; Piccolo et al., 2003). The last stage is gradual decomposition of more resistant substances such as lignin. Lignin decomposition is a key factor influencing the rate of formation of humic substances. This partly due to retarding effect of macronutrients in soil to the lignin decomposition (Berg, 2000), even though it is not the case with N (Hobbie, 2000). Either by the help of soil micro-organism, or by the formation of polyphenol, or by condensation of sugar and amino acids, the end product of lignin decomposition is humic substances. Lignin decomposition is one of four major pathways in the formation of humic substances in soils (Stevenson, 1994). In this pathway, lignin will be broken into products with low molecular weight. It can also be modified to lose methoxyl (OCH_3) groups in order to generate o-hydroxyphenol and to oxidize aliphatic chains to form COOH groups. Finally, the modified material is converted to HAs and FAs. The half-life of lignin can reach more than 10 years (Ganjegunte et al., 2004). It can be assumed that the more lignin content in organic materials the more resistant the materials from decomposition. The dominant factor responsible for decomposition rates of such material is the C / N ratio, (Jhorar et al., 1991; Mishra et al., 2001).The higher the C:N ratio the slower the decomposition of the materials (Rahn et al., 2003) because the material contain higher amounts of lignin, wax, and chitin which are more resistant to decomposition.

Artoshy , (2004) concluded that the rates of the added organic manure at (1.5 and 3) g /100g levels were faster in the soil , which high in active / total $CaCO_3$,and also concluded that sixty days of incubation were enough to complete the decomposition of the most easily decomposable parts of the added sheep manure.

For example, Martin et al., (1980) conclude from the results of their research that major portions of lignin carbons are incorporated into the more resistant components of the soil humus while polysaccharide carbons are metabolized and utilized for energy and synthesis of cell polymers, primarily proteins and poly-saccarides. Some of the latter are stabilized in the humus but still susceptible to acid hydrolysis. Kiem and Kogel- Knabner, (2003) propose that lignin and polysaccharides differ in their long-term stabilities, sugars , mainly those of microbial origin, are stabilized in the soil within fine separates while lignin is associated mainly with the coarse fractions. These workers believe that lignin, usually considered recalcitrant by resisting microbial attack, may not be subject to stabilization processes in the soil, which are effective over the long-term and may therefore not contribute to the refractory C- pool. In a similar vein, Gleixner et al.,(2002) concluded that polysaccharides, and proteins ,amino acids, and compounds such as chitin can be preserved in the soil during decomposition and humification but lignin is severely biodegraded and no longer present in its initial form. Schnitzer et al.,(2006) found that the cultivated soil has lost 59% of its initial OM (51% loss of OM based on SOC) , while the cultivated soil was relatively richer in carbohydrates, phenols and lignin monomers, alkyl aromatics , N – containing compounds and low – molecular weight organics but poorer in lignin dimmers, lipids, sterols and suberins than the virgin soil. Increases in carbohydrates and N- containing compounds in the cultivated soil may be the results of microbial syntheses, while increasing concentrations of phenols, lignin monomers and low – molecular weight organics may have arisen from the degradation of complex lignins ,lipids , sterols , and suberins, also they showed that with increasing time of cultivation , aliphatic are preferentially degraded while aromatics are more resistant to biodegradation and survive in the soil for longer periods of time. Physical fractionation of organic matter based on either particle size or density has proven useful to determine the fate and decomposition of crop residues in soil (Christensen, 1992). It has been found that the light fraction of crop residues rapidly decomposes (Shields and Paul,1973) and that residue particle size decreases with a large proportion of the residual material becoming heavier because of close association with the clay- and silt-size fractions (Christensen, 1992; Aita et al., 1997). Fresh residues decompose rapidly in soil environments and typically, over half of the C in fresh residue is mineralized to CO_2 within a few weeks or months (Ajwa and Tabatabai, 1994). By contrast, the C in recalcitrant clay–humic complexes has mean residence times in soils of 500 to 3000 yr. (Oades, 1988, 1989; Stevenson, 1994).

The supplies of nutrients depend on OM mineralization during its decomposition. Decomposition includes the breakdown of litter and transfers of OM and nutrients to the soil. It is an essentially biological process which may be influenced basically by substrate quality and environmental conditions, (Swift *et al.*, 1979; Broder and Wagner, 1988). Environmental conditions are related to aeration, moisture condition, pH, and temperature. Substrate quality is related to the C organic chemical composition of the litter. The decomposition rates of leaf litter have been predicted by lignin: N ratio from the leaves (Taylor *et al.*, 1989). SOM is composed of a variety of fractions or pools with different decomposition rates. These include a microbial biomass fraction (Sparling (1991); a labile or actively cycling fraction; and a stable or recalcitrant fraction (Tate, 1987; Theng et al., 1989; Ellert and Gregorich, 1995). These different fractions of SOM lead to a variety of biological and physical methods for studying SOM dynamics. The microbial biomass fraction is a more sensitive indicator of OM dynamics than total SOM (Sparling, 1991). Total SOM is not a sensitive or accurate indicator of short-term changes in SOM. The actively cycling fraction is more sensitive to management, necessitating a distinction between the actively cycling and more stable fractions of SOM (Ellert and Gregorich, 1995). The conditions related to the tendency of soil C to become stabilized or sequestered have been classified as recalcitrance, interactions between molecules, and physical accessibility (Sollins et al., 1996).

Recalcitrance refers to molecular-level characteristics such as elemental composition, functional groups, and molecular conformation that influence the inherent biodegradability of the molecule. *Interactions* refers to the intermolecular interactions with other inorganic or organic molecules that influence biodegradability—for example, complexation with Al. Physical accessibility refers to the location of the molecule with respect to enzymes or cells .For example, substrates may be located in sub micrometer pores too small for microbial cells to enter (Sollins et al.,1996). These conditions do not necessarily occur exclusively. The degree of decomposition increases with decreasing particle size. Plant fragments are found in the light and coarse fractions (Hassink and Dalenberg, 1996) and HS accumulate preferentially in the heavy and fine fractions (Tisdall and Oades, 1982).

Ekschmitt, et al., (2005) conclude that SOM is stabilized by a complex of mechanisms that constrain decomposition rates, several of which are not based on substrate quality or soil conditions, but on the biology of the decomposing soil organisms. Mineralization is an important process that can provide mineral nutrients for plants and microorganisms by means of biochemical oxidation of SOM Marinari et al., (2007). Johansson, (1986) indicated that a considerable proportion of the terrestrial C is present in the form of lignin and carbohydrates and that SOM is predominantly derived from their decomposition. Lignin decomposes slowly and represents a recalcitrant fraction in soil and litter (Guggenberger et al., 1994). During decomposition of lignin, intermolecular bonds between phenylpropanoid components of the lignin are severed and oxidized, and phenolic derivatives are released. As biodegradation progresses, the fractions derived from lignin in soil become increasingly acidic (carboxylic) (Haider, 1992; Kögel-Knabner, 1993). Thus, acid to aldehyde ratio of phenolic moieties can be used to estimate the extent of lignin biotransformation (Zech and Kögel-Knabner, 1994; Solomon et al., 2000). At constant temperature and moisture, microbial activity and the release of nutrients decline exponentially with time as a result of decreasing carbon availability (Chodak et al., 2001). Microbial respiration or the CO_2 production of aerated compost is often used as an indicator for activity or stability of compost (Wu and Ma, 2002).

2.8.1 Kinetics of soil organic matter.

During the late 1970s, and particularly in the 1980s, soil chemists increasingly realized that while the study of equilibrium processes and reactions in soils were important, that most field reactions were seldom at equilibrium. It became apparent that one needed to understand the rates of soil chemical processes to accurately predict the fate/transport of ions and organic compounds in soil and water environments, including the development of predictive models. Another major impediment in promotion of kinetics as a major area in soil chemistry was the complexity of the topic, particularly as kinetics is applied to heterogeneous soil systems, and the lack of good methods to measure both rapid and slow soil chemical reactions. Over the past two decades, soil chemists have developed and employed an array of batch and flow methods to measure reactions over time scales of minutes and longer , and have used chemical relaxation methods, such as pressure-jump and concentration-jump, to study reaction rates on millisecond time scales. These methods are discussed in a number of books and reviews (Sparks, 1989; Amacher, 1991; Sparks and Zhang, 1991; and Sparks et al., 1996). Without question, in the last two decades, kinetics has become a major theme in soil chemistry. There is no space to discuss all the significant contributions that soil chemists have made in the area of kinetics; however, the reader can consult a number of references authored by soil chemists (Sparks, 2000; Sparks,et al., 1991; and Sposito, 1994).

Chemical kinetics is the description of the rate of change in the concentration of reaction of reactants in a chemical reaction or the velocity of reaction can be expressed as a simple function of the concentrations of certain of its reactants. This expression is the rate equation for the reaction which has the general form,

Rate of reaction = constant x [reactant (s)]^{n} ------------------------------------3

The value of exponent (n) in its rate equation is the order of reaction. This is the rate at which the reactants are transformed into products. This may take place by a biotic or by biological systems, such as microbial metabolism, since a rate is a change in quantity that occurs with time, the change we are most concerned with is the change in the concentration of our contaminants into new chemical compounds

Reaction rate = change in product concentration / corresponding change in time ----- 4

and,

Reaction rate = change in reactant concentration / corresponding change in time---- 5

In environmental degradation, the change in product concentration will be decreasing proportionately with the reactant concentration, so, for substrate *X*, the kinetics looks like:

Rate = $-\Delta(X) / \Delta t$ ---6

The negative sign denotes that the reactant concentration is decreasing. It stands to reason then that the degradation product *Y* resulting from the concentration will be increasing in proportion to the decreasing concentration of the substrate *X*, and the reaction rate for *Y* is: Rate = $-\Delta(Y) / \Delta t$ --7

Table 6 (summarized the reaction orders of 0, 1, 2 and n. Reactions with order 3 are very rare, and extremely unlikely to occur.

Table (6) summary for reaction orders 0, 1, 2 and n.

	Zero Order	First Order	Second Order	n-th Order
Rate Law	$-\frac{d[A]}{dt} = k$	$-\frac{d[A]}{dt} = k[A]$	$-\frac{d[A]}{dt} = k[A]^2$	$-\frac{d[A]}{dt} = k[A]^n$
Integrated Rate Law	$[A] = [A]_0 - kt$	$[A] = [A]_0 e^{-kt}$	$\frac{1}{[A]} = \frac{1}{[A]_0} + kt$	$\frac{1}{[A]^{n-1}} = \frac{1}{{[A]_0}^{n-1}} + (n-1)kt$ [Except first order]
Units of Rate Constant k	$\frac{M}{s}$	$\frac{1}{s}$	$\frac{1}{M \cdot s}$	$\frac{1}{M^{n-1} \cdot s}$
Linear Plot to determine k	$[A]$ vs. t	$\ln([A])$ vs. t	$\frac{1}{[A]}$ vs. t	$\frac{1}{[A]^{n-1}}$ vs. t [Except first order]
Half-life	$t_{1/2} = \frac{[A]_0}{2k}$	$t_{1/2} = \frac{\ln(2)}{k}$	$t_{1/2} = \frac{1}{[A]_0 k}$	$t_{1/2} = \frac{2^{n-1} - 1}{(n-1)k[A_0]^{n-1}}$ [Except first order]

The mathematical description of in vitro carbon and nitrogen mineralization is another interesting approach to characterizing SOM. The exponential model was widely used to describe the carbon and nitrogen mineralization process (Stanford and Smith, 1972; Riffaldi et al., 1996). From this model, the potentially mineralizable carbon pool of soils (C_0) may be estimated. C_0 is assumed to be a readily mineralizable carbon component which mineralized at a constant rate (k) proportional to the size of the pool. Another single-component model is the hyperbolic model, which considers that the time to mineralize 50% of C_0 pool ($t_{1/2}$) increases as the incubation time gets longer, according to the rise of carbon chemical protection.

Alvertz. and Alvertz, (2000) used the following models for describing carbon mineralization kinetics, as shown in Table (7).

Table (7) Models used to describe carbon mineralization kinetics.

Model	Equation
Exponential	$\text{Cmin} = C_0(1 - e^{-kt})$
(Stanford and Smith, 1972)	
Hyperbolic	$\text{Cmin} = C_0\ t/t_{1/2}\ t$
(Juma et al., 1984)	
Doubled exponential	$\text{Cmin} = C_L(1 - e^{-k\text{L}t}) + C_R(1 - e^{-k\text{R}t})$
(Bonde and Rosswall, 1987)	
Exponential plus linear	$\text{Cmin} = C_L(1 - e^{-k\text{L}t}) + C\ t$
(Bonde et al., 1988)	
Exponential plus a constant	$\text{Cmin} = C_L + C_R(1 - e^{-k\text{R}t})$
(Jones, 1984)	

Cmin = mineralized carbon, C_0 = potentially mineralizable carbon pool, k = mineralization constant, fraction mineralized per time unit, t = time, $t_{1/2}$ = semidecomposition time, time needed for the decomposition of 50% of C_0, C_L = carbon labile pool, k_L = labile pool mineralization constant, C_R = carbon resistant pool, k_R = resistant pool mineralization constant, C = resistant pool mineralization rate, resistant carbon mineralized per time unit.

Also they concluded that both the exponential and hyperbolic models provided satisfactory fits of the cumulative CO_2-C production data in all samples. The double-exponential model gave higher coefficients of correlation than the one-component models, but it could only be fitted to the mineralization data of 20 soil samples. The exponential plus linear model showed a good fit to the carbon mineralization data, but estimated negative resistant pool mineralization constants (C) for eight samples. The exponential plus a constant model also described satisfactory the in vitro mineralization, but calculated negative values for the carbon labile pool (C_0) in 10 samples. The potentially mineralizable carbon pool estimated by the exponential model (exponential C_0) and the hyperbolic model (hyperbolic C_0) were highly correlated between each other. Additionally, their semi decomposition times ($\bar{t}$) were positive and highly associated too. The hyperbolic model gave C_0 values that were about 55% greater and $\bar{t}$values 2 to 2.8 times greater than those estimated by the exponential model. When these functions were adjusted to mineralization data sets corresponding to shorter incubation time periods (i.e., 130 or 100 d), both models estimated lower potentially mineralizable carbon pools (C_0), but still gave high r^2 values. The exponential model was selected because of its simplicity and the goodness of fit for all samples. The exponential C_0 was closely and linearly related to the mineralized C, with a slope of 1.07. Thus, similar to the relationships previously noted for mineralized C, the exponential C_0 was highly correlated with biomass C ($r^2 = 0.750$, $P < 0.001$; not shown) and with light fraction C. Otherwise, the exponential C_0 showed low correlation with the heavy fraction C $(r^2 = 0.410;\ P < 0.001)$ and with the medium fraction C $(r^2 = 0.250,\ P < 0.001)$. The mineralization constant of the exponential model (exponential *k*) presented a positive but weak correlation with the ratio light fraction C/total C $(r^2 = 0.200,\ P < 0.05)$, and a negative association with the ratio heavy fraction C/total C $(r^2 = 0.240,\ P < 0.01)$. Webb et al., (2003) calculated that SOM is equated to SOC, and most models calculate turnover of C rather than SOM. Changes in SOC may be expressed as:

Δ SOC = Annual mineralization of SOC - stabilized C residue from annual additions of organic matter (OM)-- 8

In arable soils, annual ΔSOC will usually be negative since most of the fresh OM is mineralized within the season of application (Matus and Rodríguez, 1994) and the amount of OM stabilized is likely to be smaller than the amount of SOC mineralized. The exponential decrease of SOC may be expressed as:

$$C_e = C - (C_o - C_e)^{a - kt} \quad \text{-- } 9$$

where C_e equals equilibrium SOC, *C* equals current SOC, C_o equals initial SOC, *a* equals calculated constant, *k* equals decomposition constant, and *t* equals time. Wilts et al.,

(2004) found that the original SOM (SOC_i) ranged from 88 to 107 Mg ha^{-1} , while A_t ranged from 62.8 to 72.1 Mg ha^{-1}. Assuming a first-order reaction, decomposition was expressed as $A_t = SOC_i\ e^{-kt}$, where t was time (yr) and k was an apparent decomposition constant. The k ranged from 0.0115 to 0.0146 with a mean of 0.0136. On the basis of k, the mean half-life was 77 yr, which was not influenced by any treatments. The first-order decomposition rate constant of lignin, one of the slowest decomposing substances present in original plant material has been reported as 0.18 yr^{-1} a half decay time ($t_{1/2}$) of 3.8 yr in a temperate soil (Lynch,1991). Lipson et al., (2001) calculated that both process were plotted as rate vs. temperature, and were fit to an exponential curve. For reference, Q_{10} (means the increase in k value for 10 C^o increases) was calculated from exponential coefficient (k) using the formula:

$Q_{10} = e^{(10k)}$ -- 10

Rethemeyer et al., (2005) found that the ^{14}C values of the humin fraction, isolated from samples of different field trials at the agricultural station Rotthalmünster (Germany), yielded ^{14}C decreases of about 30% to 54% from the surface soil to 65 cm depth. These results indicate a progressive enrichment of stable organic compounds with increasing soil depth. In contrast, a minor decline in ^{14}C concentrations of the HA fraction, which mostly showed higher ^{14}C values than the humin, reflect the translocation of modern organic carbon towards greater depth. Bayer et al., (2006) concluded that the one-compartment C model $C_t=C_0e^{-k_2t}+k_1A/k_2(1-e^{-k_2t})$ is being long used to simulate soil organic C (SOC) stocks. C_t is the SOC stock at the time t; C_0, the initial SOC stock; k_2, the annual rate of SOC loss (mainly mineralization and erosion); k_1, the annual rate to which the added C is incorporated into SOC; and A, the annual C addition. The component $C_0e^{-k_2t}$ expresses the decay of C_0 and, for a time t, corresponds to the remains of C_0 ($C_{0\ \mathrm{remains}}$). The component $k_1A/k_2(1-e^{-k_2t})$ refers, at time t, to the stock of SOC derived from C crops (C_{crop}). . A linear equation ($C_t = a + bA$) between the SOC stocks measured at the 13th year (0–17.5 cm) and the mean annual C additions was fitted for CT and NT. This equation is equivalent to the equation of the model $C_t=C_0e^{-k_2t}+k_1A/k_2(1-e^{-k_2t})$, so that $a=C_0e^{-k_2t}$ and $bA=k_1A/k_2(1-e^{-k_2t})$. Such equivalences thus allow the calculation of k_1 and k_2. NT soil had a lower rate of C loss ($k_2 = 0.019$ $year^{-1}$) than CT soil ($k_2 = 0.040$ $year^{-1}$), while k_1 was not affected by tillage (0.148 $year^{-1}$ under CT and 0.146 $year^{-1}$ under NT).

The SOC half-life ($t_{1/2}$ = ln $2/k_2$) was 36 years in NT and 17 years in CT, reflecting the slower C turnover in NT. The effects of NT on the SOC stocks relates to the maintenance of the initial *C* stocks (higher $C_{0\ \mathrm{remais}}$), while increments in C_{crop} are imparted mainly by crop additions. Torri et al., (2003) studied that the mineralization of carbon from sewage sludge was conducted in a pot experiment during an aerobic incubation with three representative soils of the Pampean Region of Argentina. Surface horizons (0-15 cm) from a Typic Hapludoll, a Typic Natraquoll, and a Typic Argiudoll were used. Samples were collected a year after sludge addition. Expressed as percentages, more than 50% of the added sludge carbon mineralized during the first 60 days. However, 29-45% remained in the soils a year after application. The best fit to carbon mineralization data was provided by a first-order exponential plus a constant kinetic model. Percent readily mineralizable carbon (%CLS) values and first-order rate constants (k) of the sludge treated soils ranged from 53 to 58 and 0.035 to 0.030 day^{-1} (Typic Hapludoll, Typic Argiudoll) and 71 with a k value of 0.07 (Typic Natraquoll). Thus, carbon mineralization of sewage sludge in the soils studied was independent of soil texture. The higher pH of the Natraquoll seemed to have favored an intense microbial activity, and explained the larger labile carbon pool of sewage sludge in this soil compared to the other two soils studied. Fayez,(2006) using laboratory incubations for 60 days to determine the nitrogen and C mineralization rates . Results show that in this calcareous wetland soil, cultivation decreased TSOC and total N contents, while TOC / total N ratio, bulk density, pH, and extractable P and available K levels remain unaffected. Cultivation resulted in a significant increase in soil C and N mineralization.

2.9 Solubility equilibria of iron, manganese, copper, and zinc in soils.

The iron in most primary minerals (ferromagnesian and oxide phases) occurs in the ferrous ($Fe^{+2)}$ oxidation state, which is more soluble than ferric iron (Fe^{+3}). There are several different kinds of iron oxides, each with a different solubility (Schwertmann, 1988). Free iron minerals that occur in soil are used as a key characterization for soils and for soil horizon. The solubility of Fe^{+3} is usually controlled by the most soluble oxide present ; thus, freshly precipitated amorphous magnetite or siderite (Soil – Fe (OH $)_3$) is the most soluble Fe^{+3} oxide and generally controls the activity of Fe^{+3} and solubility of Fe^{+2} in soils, depending on redox and CO_2 (Bodek et al., 1988) . Under oxidizing conditions (pe +pH > 11.5) , soil – Fe – (OH $)_3$ (which is intermediate in solubility to amorphous hydroxide and crystalline oxide) controls the solubility ; below 11.5, magnetite (Fe_3O_4) is the stable phase until siderite ($FeCO_3$) form , as determined by the partial pressure of CO_2 (g) (Lindsay, 1979).

Iron contents for five various soils were found to be in the range of 3.45 – 4.89 g /100g (Al – Aazami ,1981 as stated by Rahi, 1991), and available iron four north Iraqi soils was 17.03 ug /g when extracted by ammonium method (al – Malik , 1986 as stated by Rahi, 1991). Values of available iron extracted by EDTA $+NH_4OAc$, DTPA $+NH_4HCO_3$, DTPA+ $CaCl_2$+TEA , NH_4OAc , EDTA + $(NH_4)_2CO_3$, and HCL + H_2SO_4 metods were (17 .624 , 16.458 , 14.285 ,4.472 , 2.705 , and 1.681) ug / g respectively (Abdullah , 2006).

Other Fe^{+3} oxides are listed in order of decreasing solubilities:

- $Fe(OH)_3$ ((soil – Fe)
- γ- Fe_2O_3 (maghemite)
- γ- FeOOH (lepidocrocite)
- α - Fe_2O_3 (hematite)
- α-Fe OOH (goethite)

The solubility of $Fe(OH)_3$ (amorphous) is 3,630 times more soluble than that goethite (Lindsay, 1979). The reactions of iron in the processes of weathering are depending largely on the Eh – pH system of the environment and on the stage of oxidation of the iron compounds involved. The electron potential for the Fe (III) / Fe (II) is 0.771 V (Bodek et al., 1988). The general rule governing the mobilization and fixation of iron are that oxidizing and alkaline conditions promote the precipitation of insoluble iron Fe^{+3} oxides and / or oxyhydroxide, whereas acidic and reducing conditions promote the solution of ferrous (Fe^{+2}) compounds. The released iron readily precipitates as oxides and hydroxides, but it substitutes for Mg and Al in other minerals and often complexes with organic ligand (Kabata – Pendias and Penias, 1992). The iron oxide that is present in soil is dependent upon the moisture content, pH, and oxygen content of the soil. In wet, but oxic soil conditions, the iron oxide typically exists in the hydrated ferric oxide state. In wet and hypoxic soil condition, the typical iron oxide is in the ferrous state.

The solubility of inorganic Fe in well – aerated soils is controlled by the dissolution and precipitation of Fe^{+3} oxides. The concentration of Fe^{+3} is related to pH , decreasing from 10^{-8} to10^{-20} *M* as pH increases from 4 to 8 (Römheld, and Marschner ,1986). High HCO_3 concentrations decrease the availability of Fe in calcareous soils (Bloom and Inskeep, 1988).

The manganese content of the lithosphere content is approximately 900ug /g , and soils generally contain from 20 to 3000 ug/g with an average of 600 ug / g , this average would supply 10^{-1} *M* manganese in the soil solution at 10 % moisture if it all dissolved. Manganese is less abundant than either aluminum or iron (Lindsay, 1979). Total manganese content in Iraqi soils was between 495 and 1650 ug /g (Rahi et al., 1987 and Barzan, 1984 as stated by Rahi, 1991). Manganese is a fairly common element. It is the fifth most abundant metal in the Earth's crust and the second most common trace metal after iron, found at about 1/50 the abundance of iron. Manganese is distinctive for being able to exist in a great number of oxidation states, from 0 to +7. In nature, however, it is primarily found in the Mn (II), Mn (III), and Mn (IV) states.

The Mn^{2+} (manganous) cation is the most important soluble form of manganese in nature, though certain important Mn (II) salts such as manganous carbonate (rhodochrosite) have only low or negligible solubility. The Mn^{3+} (manganic) ion is unstable in neutral solution unless strongly complexed; it rapidly disproportionates to Mn^{2+} and MnO_2. Mn (III) and Mn (IV) are generally found as insoluble oxides or hydrous oxides, Mn (IV) most notably as MnO_2. These oxides are brown- or black-colored. Mn(II) oxidation can lead to a variety of oxides, depending on the exact conditions of oxidation; some possibilities are ß-, and γ-MnO_2, ß-, and γ-MnOOH, Mn_2O_3, Mn_3O_4, and $Mn(OH)_3$ (Stumm and Morgan,1970). Any oxidation not rigidly controlled is likely to result in a nonstoichiometric mixture of these oxides. Such mixed oxides are often referred to as "MnO_x" for convenience, with *x* ranging from 1.0 to 2.0. Furthermore, some of the oxides are metastable and convert to higher oxides upon aging (Hem and Lind, 1983). An important complication of studying Mn (II) oxidation is that manganese oxides provide binding sites for trace metals, including Mn^{2+}, and that MnO_x catalyzes oxidation of bound Mn (II). The oxidation of aqueous Mn^{2+} to MnO_2 is energetically favorable under neutral aerobic conditions. However, the kinetics of the reaction is such that spontaneous oxidation does not occur measurably until the pH increases above about 8 or 9.

The average copper content of the lithosphere is reported to be 70 ug /g, whereas soils generally range from 2 to 100 ug /g. The average content for soils is estimated at

30 ug/ g, which represents a maximum concentration of $10^{-2.33\,M}$ Cu in the soil solution at 10 % moisture. Total Cu content in Iraq was 23.5 – 72.5 ug /g (cited from Rahi et al., 1991). Copper is slightly less abundant in soils than zinc. The minerals governing the solubility of Cu^{+2} in soils are not known (Lindsay, 1979). Recently, heavy metal ion activities controlled by solubility–precipitation and sorption–desorption reactions in soil have been proposed to be one of the essential parameters for evaluating the toxicity of these metals to organisms (McBride, 1989; Sauve et al., 1998). Lighthart et al., (1983) reported that dissolved Cu was predicted to inhibit soil microbial respiration when the free ion activity reached a value of 0.01 to 0.1 m*M* in the soil solution. However, no effect of Cu ion activity in the soil solution was observed on then and C- mineralization rates in field-aged Cu-enriched soils (Minnich and McBride, 1986). The Cu concentration in soil solution is usually very low, ranging between 10^{-8} and 10^{-6} *M* (Lindsay, 1979).

The zinc content of the lithosphere is approximately 80 ug /g, and the common range for soils is 10 – 300 ug /g with average content of 50 ug /g. Zinc is considered a trace element in soils. If all of zinc present in an average soil were in solution at 10 % moisture, it would comprise only $10^{-2.12}$ *M* zinc. Some studies have shown that total Zn content in some Iraqi soils was ranged between 19 and 68.6 ug /g (Nuhood, 1981 and Rahi, 1987 as stated by Rahi, 1991). The specific minerals controlling the activity of Zn^{+2} in soils are not known. The solubility of Zinc in soil solution must be quantified to evaluate bioavailability and transport of zinc in soils. Zinc solubility can be represented by the total concentration of Zn in solution or by Zn^{+2} activity. The activity of Zn^{+2} represents Zn availability to plants and can be used to predict possible solid phase that control Zn solubility in the soil. Depending on the soil and soil properties, different Zn precipitates may form in soils and control Zn solubility at different levels (Lindsay, 1979). Zinc has been the subject of much research in agriculture because it is an essential micronutrient for plants (e.g., Tiller et al., 1972 ; Sadiq, 1991). Zinc deficiencies commonly occur in Colorado and other Western states because of high soil pH. In contrast, Zn in high concentrations can be toxic to plants and animals; a subject of current environmental research (e.g., Barbarick et al., 1997; Lerch et al., 1990).It appears that Zn availability in some alkaline soils may be controlled by franklinite. Ma and Lindsay, (1990) estimated free Zn activity by chelation in Colorado soils. They found that Zn^{2+} activities for uncontaminated soils were near the soil-Zn line (Lindsay, 1979) and that the solid phase that controls Zn availability could be franklinite in alkaline soils. The solubility of Zn and the mechanisms that control Zn solubility may vary with soil properties, such as pH, organic matter content, and clay content. McBride and Blasiak, (1979) state that different adsorption mechanisms are likely to control Zn solubility at different pH values. Singh and Abrol, (1985) found that precipitation of willemite (Zn_2SiO_4) was likely at pH >7.9 in the sodic soils they studied. They also found that precipitation or adsorption may occur between pH 6 and 7.9 and that adsorption may occur below pH 6. Jeffrey and Uren, (1983) conclude that at neutral to alkaline pH, specific adsorption of a hydrolyzed form of Zn (e.g., $Zn\ [OH]^{+}$) may account for low soluble Zn concentrations. Various studies have shown that there may be changes in Zn solubility with other soil properties. Zinc may bind to Fe, Mn, and Al oxides; clays; or organic matter in soils. Iron, Mn, and Al oxides contain surface-hydroxyl functional groups that may strongly bind metals, with increased adsorption at high pH (Sposito, 1984).

3. MATERIALS AND METHODS

3.1 Soil sampling

Surface soil samples were taken at (0 – 30) cm at two locations {(Kalar (Shirwana castle)}, and Arbat (Kharajian village) in Sulaimani area Iraqi Kurdistan region. The first location was previously cultivated by okra, while the second one was exploited by wheat, the two locations were varied in respect to texture, rainfed, and total calcium carbonate, whereas they were similar with respect to organic matter content. Both soils were classified as (Typic Torrifluvents and Vertic Calcixerepts) successively according to Soil Survey Staff, (1994). Representative soil samples were taken from each location, composite soil samples were air dried and passed through a 2 mm sieve, soil samples were stored in plastic bags prior to use. Some of the chemical and physical characteristics were listed in the Table (8).

Table (8) some physical and chemical characteristics of the studied soils

	Soil location	
Characteristics	Kalar (Shirwana Castle)	Arbat (Kharajian village)
Texture	Loam	Silty Clay
Sand (g /100g)	41.18	8.15
Silt (g /100g)	40.99	43.39
Clay (g /100g)	17.83	48.46
Bulk density (Mg m $^{-3}$)	1.7	1.49
Field capacity (g /100g)	20.28	32.44
Permanent wilting point (g /100g)	10.81	21.53
Maximum water holding Capacity (g /100g)	40.56	64.88
PH (H_2O 1: 5)	8.01	7.97
EC_e (dS m $^{-1)}$ at 25 C°	0.48	0.25

Continued

Cation exchange capacity (Cmol$_C$ / kg)	31.83	51.56
Calcium carbonate equivalent (g /100g)	30.00	21.25
*Organic matter (g /100g)	2.21	2.17
*TSOC remaining (g /100g)	1.28	1.26
*C- Humin (g /100g)	0. 42	0.63
*C- HA (g/100g)	0.09	0.08
*C-FA (g /100g)	0.77	0.55
Total nitrogen (g /100g)	0.86	0.66
Available potassium (ug / g)	66.30	284.70
Available phosphorus (ug / g)	18.60	17.90
*Available Fe (ug / g)	14.25	6.95
*Available Mn ((ug / g)	2.86	11.25
*Available Cu (ug / g)	0.61	0.64
*Available Zn (ug / g)	0.35	0.39
Calcium (mmol$_C$ L^{-1})	3.89	2.73
Magnesium (mmol$_C$ L^{-1})	1.05	0.74
Sodium (mmol$_C$ L^{-1})	1.04	0.34
Potassium (mmol$_C$ L^{-1})	0.15	0.06
Sulphate (mmol$_C$ L^{-1})	1.59	1.61
Chloride (mmol$_C$ L^{-1})	1.00	1.08
Carbonate (mmol$_C$ L^{-1})	0.00	0.00
Bicarbonate (mmol$_C$ L^{-1})	3.57	4.10

*A sample has been taken at 28C^o and 50 g/ 100g available water without manure application at each period for the purpose of comparison.

3.2 Preparing of organic wastes

Sheep manure has been taken from Maladaoud village / Sulaimani area, air-dried, and ground to pass through a 0.5 mm sieve. Table (9) shows some chemical characteristics of the manure.

Table (9) Nutrient content of sheep manure

Type of manure	N (g / 100g)	P_2O_5 (g /100g)	K_2O (g /100g)	TOC (g /100g)	OM (g /100g)
Sheep	1.80	1.40	2.70	31.76	54.76

3. 3 Physical analysis

3.3.1 Particle size distribution

Particle size distribution was performed by the pipette method as described in Van Reeuwijk, (1995).

3.3. 2 Field capacity and permanent wilting point

Field capacity and permanent wilting point were calculated according to equations provided by Karim, (2000).

3.3.3 Bulk density

Bulk density was determined by core method according to Black et al., (1965).

3.3.4 Saturation percentage

Saturation percentage was determined as described in Ryan John et al., (2001).

3.4 Chemical analysis

3.4.1 Electrical conductivity

Electrical conductivity was measured with EC – meter model D 8120 and adjusted to 25 C° as described in Van Reeuwijk, (1995).

3.4.2 Soil pH

pH was measured with pH – meter model Hans Herbert Mennerich (geotechnik) Hanover as described in Van Reeuwijk ,(1995).

3.4.3 Organic matter

Oxidizable organic matter was determined by the Walkley and Black, wet dichromate oxidation procedure as described by Nelson and Sommers, (1982).

3.4.4 Total nitrogen

Total nitrogen was determined by the Kjeldahl digestion-distillation method according to Bremner and Mulvaney, (1982).

3.4.5 Available phosphorus

Available phosphorous was determined by using Olsen procedure as described in Ryan John et al., (2001).

3.4.6 Available potassium

Available potassium was extracted by using $1N$ NH_4 OA_C solution (pH = 7) and determined by flame photometer as described in Ryan John et al., (2001).

3.4.7 Cation- exchange capacity

Cation exchange capacity was determined by means of 1 N $NaOA_C$ solution (pH = 8.2) as described in Ryan John et al., (2001).

3.4.8 Calcium carbonate equivalent

Calcium carbonate was determined by acid neutralization method according to Richards as described in Rowell, (1996).

3.4.9 Soluble cations and anions

Soluble cations and anions were determined according to the procedures outlined by Page et al., (1982).

3. 5 Experimental Design

The experiment was conducted in the laboratory of soil and water science department, college of agriculture , university of Sulaimani ,Iraqi Kurdistan region during (2003 – 2004) in order to measure SOC dynamics by soil incubation in the laboratory to allow biological mineralization of carbon, factorial experiment of five factors was used in completely randomized design (CRD) with two replicates. The factors used were:

- Soil locations (Kalar and Arbat)
- Temperatures (5 , 28 , and 45) C°
- Sheep manure applications rates (0.5, 1.5, and 2.5) g /100g
- Available water (25, 50, and 100) g /100g .
- Incubation periods (2 , 4 , 8 , and 12) months

200 gm soils were taken from each soil in a small tightly capped bright jars (13 cm high and 6 cm wide) and treated with (0.5 , 1.5 , and 2.5) g /100g sheep manure. Moisture content was adjusted to (25 , 50 , and 100) g /100g of available water by weighing the jars daily and adding water to replenish the moisture deficit , three periods (4 , 8 , and 12) months were used for determining humic acid (HA) , fulvic acid (FA) , and humin at 28 C° , and then incubated at (5 , 28, 45 C°) for time intervals (2 , 4 , 8 , 12) months , TOC g /100g , Eh , EC, pH , and available (Fe, Mn, Cu ,and Zn) were measured at (2 , 4 , 8 , 12)months. A sample has been taken at 28C°and 50 g /100g available water without manure application at each period for the purpose of comparison.

3.6 Extraction and fractionation of soil organic matter

OM fractionation was determined according to the method described by (Schnitzer, (1982). Briefly, If the soil sample under investigation contains free carbonates as indicated by reaction with dilute HCl, allow the soil to stand in contact at room temperature with an excess acid 0.05*N* HCl until bubbling cease. Then wash out excess acid with distilled water, thinly spread the soil on a glass plate and allow it to dry at room temperature. Weigh 10 g of air – dry soil into a 200 ml propylene flask, add 100 ml of extractant (0.1 *N* NaOH), and displace the air flask by N_2. Stopper separate the soluble material FA from the coagulate HA by centrifugation. Take them to dryness on a rotary evaporator at about 40 C° as shown in (Fig. 9). OC for each portion was estimated and FA calculated by the difference as shown below:

Fulvic acid – C = Total - C – [(humin – C) + (humic acid - C)] --------------------------- 11

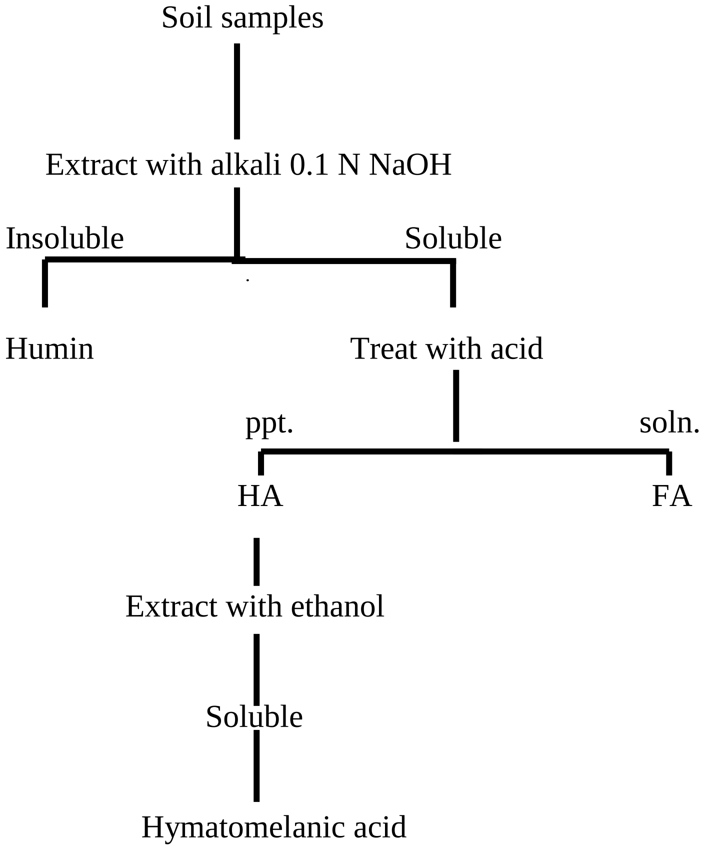


Fig.(9) Extraction and fractionation of humic substances (adapted from Schnitzer, 1982)

3.7 Available (Fe, Mn, Cu, Zn)

DTPA –extractable micronutrients were determined by the methods described by Lindsay and Norvell, (1978). Briefly, weigh 10 g of soil add 20 ml of the {(DTPA extracting solution which consisted of (0.005 *M* DTPA, 0.01 *M* $CaCl_{2,}$ and 0.1 *M* TEA)}. Shake continuously for 2 hours, preferably on a horizontal shaker, and filter through Whatman No. 42 filter paper. If the filtrate is cloudy, refilter as necessary. Prepare, at least, 10 standards using DTPA as the matrix for each element with a range 0 to 3 ug/ ml for Zn and Cu and from 0 to 20 ug/ ml for Fe and Mn. First take readings for the standards, then, measure the element from the filtrate by atomic absorption spectrophotometer. A blank solution, containing all reagents excepting the soil, should be run to correct for contamination.

3.8 Kinetics of soil organic mater decomposition

Different mathematical models were used to describe carbon mineralization as described by (Sparks, 1989; Alvarez, and Alvarez, 2000), as shown in Table (10).The model that gives the highest value of determination coefficient (R^2) and the minimum value of standard error (SE) of estimate was considered the best model equation. It is also useful to be able to calculate the half- life,$t_{1/2}$, time required for 50 g /100g of the substrate to be transformed, or time needed for the decomposition of 50 g /100g of $C_{0.}$

Table (10) Models used to described carbon mineralization kinetics

Kinetic model	(Kinetic equation) in		
	Plotting of variables	linerarized form	half- life
Zero-Order	$C_t = C_o - Kt$	C_t vs. t	$t_{1/2} = C_o / 2K$
First – Order	$\ln C_t = \ln C_o - Kt$	$\ln C_t$ vs t	$t_{1/2} = 0.693 / K$
Second –Order	$1/ C_t = 1/ C_o + Kt$	$1/ C_t$ vs. t	$t_{1/2} = 1/ K C_o$
Hyperbolic	$C_t = C_o t / t_{1/2} t$	Ct / t vs. t	$t_{1/2} = C_o$ / slope
Parabolic diffusion	$C_t = C_o - K t^{1/2}$	C_t vs. $t^{1/2}$	
Modified Elovich	$C_t = 1/K \ln C_oK + 1/K \ln t$	C_t vs. lnt	
Power function	$\ln C_t = \ln C_o + K\ln t$	lnCt vs. lnt	
Logarithmic function	$C_t = C_o + K\ln t$	Ct vs. lnt	

Where:

C_t = the mineralized carbon or the concentration of substrate remaining at time (t.)

Co = potentially mineralizable carbon pool or the initial concentration of substrate at time = 0

K = mineralization constant or rate constant

t= time

$t_{1/2}$ = semi decomposition time (half life) .

The value of standard error of estimate was calculated according to the following equation

Standard error of estimate = $\{\sum (C_o - C_t)^2 /n -2\}^{1/2}$ -------------------------------------- 12

Where C_o = calculated mineralized carbon

C_t = measured mineralized carbon

n = number of measurement

3. 9 Solubility Equilibria of (Fe, Mn, Cu, and Zn) in the soil

The soil samples were treated with DTPA buffer solution as described in Section (3.7) earlier. In the DTPA filtrate (EC, pH, Eh) measured and extractable Fe, Mn, Cu, and Zn were determined by using atomic absorption spectrophotometer. The solubility diagram of these elements was conducted as described by Lindsay, (1979). Log of each activity Fe, Mn, Cu and Zn against either pH or pe + pH incase of redox potential was used in order to determine the solubility of those cations in the soil solution.

Eh (millivolts) = 59.2 pe --- 13

The redox relationships can be expressed in terms of either pe or Eh.

The redox parameters pe and Eh are both defined as zero for the standard hydrogen half –cell. (Lindsay, 1979)

The activity coefficients of these elements calculated by using Davies equation that uses the ionic strength (I) of the solution (Davies, 1962 and Lindsay, 1979):

$-\text{Log } fm = B_{dh} Z^2_m [A^{1/2} / (1 + A^{1/2}) - 0.3 A]$ -------------------------------------- 14

Where fm is an activity coefficient of an ion M, B_{dh} is the Debye – Huckel limiting law parameter as indicated by Sposito, (1989), Zm is the valence of species M, and A is the ionic strength (I) in mol / L was estimated by the following equation

I = 0.013 x EC ---15

Where EC is the electrical conductivity of the DTPA solution in dS m^{-1} (Griffin and Jurinak, 1973). The activities of these elements were calculated by their concentrations multiplying by activity coefficient.

3.10 Statistical analysis

Statgraphics software package (Manugistics, Inc., Rockvill, MD) were used to conduct the analysis of variance (ANOVA) for a completely randomized design. Significance of treatment means were determined at $P \leq 0.01$ with the least square means test, according to Duncan's Multiple Range test and the relationships between the different variable were evaluated by regression analysis and tested by their F values.

4. Results and Discussion

4.1 Soil organic matter decomposition

4.1.2 Effects of temperature

A laboratory incubation experiment was conducted under different temperatures (5 , 28 , and 45 C^o) at different interval incubations (2 , 4 ,8 and 12) months in order to study changes in total soil organic carbon remaining (TSOC remaining g/100g) in soil. Fig. (10) shows that 5C^o was superior significantly at $p \leq 0.01$ to 28C^o and 45 C^o with regard to (TSOC remaining g/100g) during incubation periods for Kalar and Arbat locations. Average mean of (TSOC remaining g/100g) were(1.674 and 1.816) g /100g for Kalar and Arbat at 5 C^o respectively , whereas they were (1.600 and 1.665) g /100g and (1.337 , 1.048) g /100g at 28 and 45 C^o for the two locations respectively .

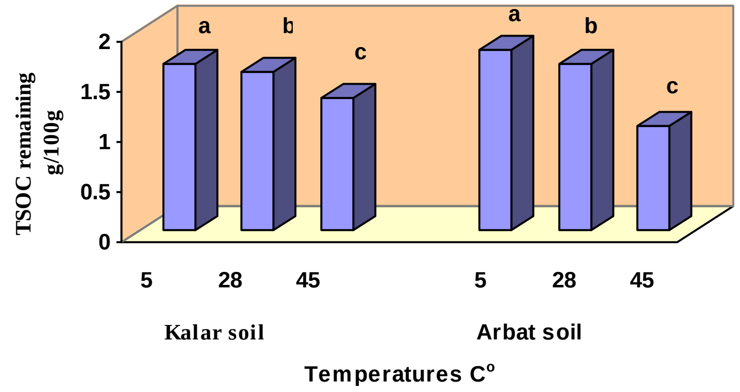


Fig. (10) Effect of temperatures on average mean of TSOC remaining g /100g during incubation periods in studied soils.

These results may be due to the following: changes in microbial diversity adapted to high temperatures, which may favor microorganisms to be capable of mineralization and disadvantage immobilizer populations (Zogg et al., 1997). Changes in temperature are, also considered likely to have a greater influence on microbial activity and microbial processes such as N mineralization and immobilization in more extreme environments at lower and higher temperature (Hoyle et al., 2006).Low temperature limits both microbial metabolism and enzyme activity, so the SOC mineralization will decrease (Fox and Van Cleve, 1983). Increase in temperatures caused by climatic change may give rise to the increase of turnover of (SOM) as a result SOC will reduce (Webb, et al., 2003).

4.1.3 Effect of available water

Fig. (11) shows that 25 g /100g available water was superior significantly to (50 and 100) g /100g for Kalar location, whereas it was superior significantly at $p \leq 0.01$ to 100 g /100g available water at Arbat with respect to average mean of TSOC remaining g /100g . Average means of TSOC remaining g /100g were (1.574 and 1.576) g /100g for Kalar and Arbat consequently at 25 g /100g available water. While they were (1.525, 1.515) g /100g and (1.513, 1.438) g /100g for 50 and 100 g /100g of available water at the two locations respectively.

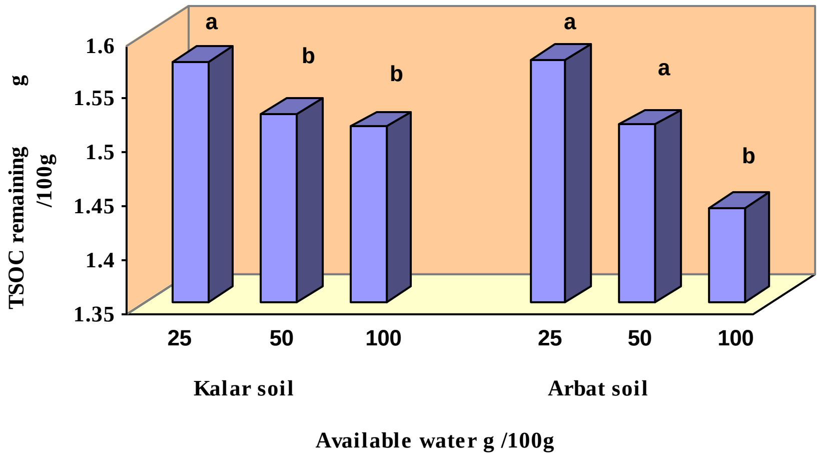


Fig. (11) Effect of available water on average mean of TSOC remaining g /100g during incubation periods in studied soils.Values with the same letter among treatments are not significantly different (P $\leq$ 0.01)

These results may be due to the OM degradation in wet lands, which is often limited by the availability of electron acceptors rather than electron donors (Reddy and Angelo, 1994; Amador and Jones, 1995; McLatchey and Redy, 1998). Heterotrophic microbial activity was higher under drained conditions than under flooded conditions, and was highest under aerobic conditions also activities decreased when other electron acceptors were utilized, (Wright and Reddy, 2001).The soil moisture condition seemed to limit the rate of decomposition in the studied area. These results indicate that changes in soil moisture can lead to an increase in carbon mineralization during incubation periods, (Rey, et al, 2005).

4.1.4 Effect of manure application rate

Fig. (12) shows that (2.5 g /100g) manure application rate was superior significantly to (0.5 and 1.5) g /100g with regard to TSOC remaining g /100g for both locations. Average means of TSOC remaining g /100g were (1.715 , 1.670) g /100g for

Kalar and Arbat locations consequently at 2.5 g /100g manure application rate , while they were (1.337 , 1.000) g /100g and (1.559 , 1.580) g /100g for 0.5 and 1.5 g /100g manure application rates consequently.

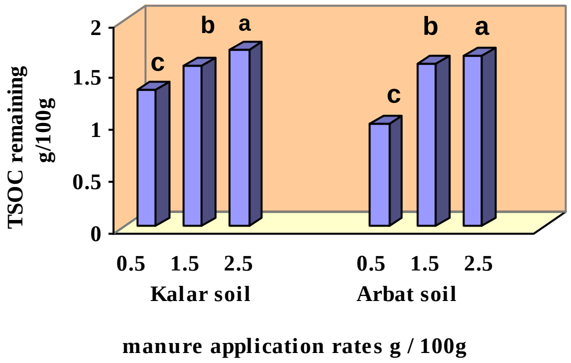


Fig. (12) Effect of manure application rates on average mean of TSOC remaining g/100g during incubation periods in studied soils.

Generally, an application of OM to the soil causes an increase in the ratio of microbial biomass / TOC (Powlson et al., 1987). Min et al., (2003) concluded that long -term application of dairy manure slurries significantly increased TOC, microbial biomass, potential mineralization , extractable and labile carbon pool, respectively. Many studies have indicated a strong positive relationship between the amount of carbon incorporated into soil , either from crop residues or external sources such as manure, and total SOC content (Havlin et al ., 1990 ; Paustian et al ., 1992) . The results showed that with the increasing rate of manure application rate, the TSOC remaining g /100g increased. The results were expected to be due to high OM. These results are in general agreement with those reported by (Plaza, et al., 2003; Bokhtiar and Sakuria, 2005; and Nelly, et al., 2006).

4.1.5 Effect of incubation periods

Fig. (13) shows that 4 months incubation period was superior significantly to (2 , 8 , 12) periods with regard to TSOC remaining g /100g for Kalar and Arbat locations. Average mean of TSOC remaining g /100g were (0.989 , 0.829) g /100g , (1.996 , 1.988) g /100g , (1.455, 1.405) g /100g and (1.709 , 1.817) g /100g for (2 , 4 , 8 ,12) months period at the two locations respectively. From the Fig. (20) , it can be noticed that TSOC remaining g /100g decreased sharply at 2 months period then increased at 4 months but decreased again at 8 months and finally increased for the second time at 12 months .

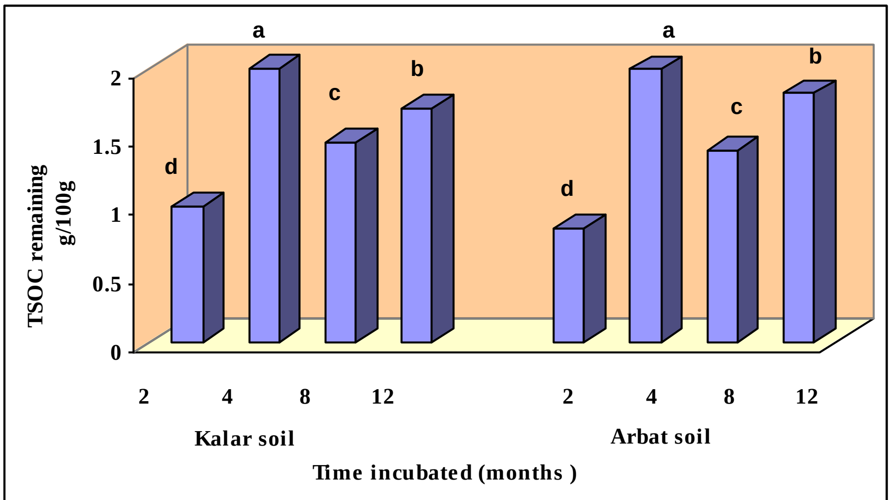


Fig.(13) Effect of incubation periods on average mean of TSOC remaining g/100g in studied soils.

These fluctuations may be due to the increase number of different microorganisms after addition of sheep manure or the easily biodegradable pool of organic carbon was gradually exhausted at 2 months of incubation (Goyal et al., 1993), but the increase of TSOC remaining g /100g at 4 months may be due to microbial biomass C immobilized and subsequently released as enzymes or as dead microbial cell fragments (Vance and Chapin, 2001). On the other hand, most of the decrease in the TSOM appears to be due to the degradation of the SOM, these results are in agreement with the finding of (Chander et al., 1995) who found that the soil microbial C increased after sewage sludge addition and this initial increase of biomass C declined with prolonged incubation. In order to well understand the actual vibration of TSOC, further studies are needed on the role of the type of soil microorganism dominant in addition to their activity at each stage of incubation periods.

4.1.6 Effect of temperature and incubation period interactions.

The results in Table (11) show that the highest mean value of TSOC remaining g /100g , which differ significantly at $p \leq 0.01$ from the other interactions, were (2.545 and 2.559) g /100g at (28C^{o} and 5C^{o}) and 4 months incubation period for Kalar and Arbat locations respectively , while the lowest mean values were (0.813 and 0.715) g /100g at 45C^{o} and 2 months incubation period for Kalar and Arbat consequently . These results might be due to both or each effect of temperature and incubation periods individually on TSOC remaining g /100g as mentioned before in section (4.1.2)

4.1.7. Effect of available water and incubation period interactions

Table (11) indicates that the interactions of 4 months incubation period with 25 g /100g of available water gave the highest mean values of TSOC remaining g /100g that were (2.051 and 2.086) g /100g for Kalar and Arbat locations subsequently , which were superior significantly at $p \leq 0.01$ to the other interactions except for interactions of 4 months period at 50 g /100g of available water which gave (1.996 and 2.006) g /100g for Kalar and Arbat locations consequently . On the other hand, the lowest values of TSOC remaining g /100g (0.970 and 0.798) g /100g are not significantly different from (25 and 50) g /100g of available water at 2 months period, which gave (1.000 , 0.997) g /100g and (0.843 , 0.846) g /100g for Kalar and Arbat locations subsequently. The interaction between these two parameters was uneffected on SOC decomposition.

4.1.8 Effect of manure application rate and incubation period interactions

The highest mean values of TSOC remaining g /100g were (2.225 and 2.150) g /100g at 2.5 g /100g of manure application rate and 4 months incubation periods for Kalar and Arbat locations respectively , which were superior to all other interactions except 1.5 g /100g of manure application rate and 4 months for Arbat locations as shown in Table (11) . On the other hand, the least values of TSOC remaining g /100g were (0.788 and 0.614) g /100g at 0.5 g /100g of manure application rate and 2 months incubation period for Kalar and Arbat locations subsequently. From the Table (11) it is noticed that the increase in temperature as well as available water will give rise to the decrease of TSOC remaining g /100g . In contrast, the increase of manure application rate percent results in an increase of TSOC g /100g remaining in Kalar soil. Furthermore, the increase of available water will give rise to decrease of TSOC remaining g /100g at (2, 4, 8, 12) months and an increase of TSOC remaining g /100g at 8 month.

While TSOC remaining g /100g of Kalar location was 1.280 g /100g but it was 1.260 g /100g for Arbat. In general results indicated that TSOC remaining g /100g used for comparison was lower than the studied parameters except (0.5 , 1.5 , 2.5) g /100g manure application rates and 2 months incubation periods for both location (Table 8). Generally, the higher value of TSOC remaining g /100g indicates that the lower decomposition of SOM was due to the less activity of microorganisms, which has been expected. These results might be due to the effect of both or each of manure application rates and incubation periods individually on TSOC remaining g /100g as mentioned before in section (4.1.4).

Table (11) Changes in mean values of TSOC remaining g /100g as functions of temperatures, available water and manure application rates during incubation periods in studied soils.

Kalar soil									
Time incubated (months)	Temperatures C°			Available Water g /100g			Manure application rates g /100g		
	5	28	45	25	50	100	0.5	1.5	2.5
2	1.137 e	1.017 f	0.813 g	1.000 f	0.997 f	0.970 f	0.788 h	0.986 g	1.193 f
4	2.058 b	2.545 a	1.385 d	2.051 a	1.996 ab	1.941 b	1.766 d	1.997 b	2.225 a
8	1.383d	1.447 d	1.534 c	1.496 e	1.452 e	1.416 e	1.268 f	1.536 e	1.560 e
12	2.119 b	1.393d	1.615 c	1.749 c	1.653 d	1.724 cd	1.527 e	1.716 d	1.883 c
Arbat soil									
2	0.929 fg	0.843 gh	0.715 h	0.846 e	0.843 e	0.798 e	0.614 g	0.825 f	1.048 e
4	2.559 a	1.992 c	1.412 e	2.086 a	2.006 a	1.871 b	1.791 c	2.022 ab	2.150 a
8	1.660 d	1.544 de	1.012 f	1.378 d	1.416d	1.421 d	0.928 ef	1.533 d	1.755c
12	2.114 c	2.281 b	1.056 f	1.995 a	1.794bc	1.662 c	1.748 c	1.939 b	1.764 c

Values followed by the same letters for each treatment in combination with time incubation are not different Significantly, between themselves according to Duncan Multiple Range test at ($p \leq 0.01$)

4.2 Humification of soil organic matter

The results presented in Fig. (14) show that manure application rate at 2.5 g /100g was superior significantly at ($p \leq 0.01$) to 0.5 and 1.5 g /100g in Kalar soil with regard to average mean of C – humin g /100g , but there was no significant difference between 0.5 and 1.5 for Arbat location

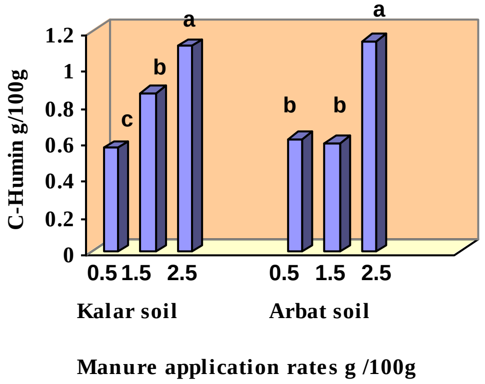


Fig. (14) Effect of manure application rates on average mean of C- humin g/100g during incubation periods in studied soils.

On the other hand, there were no significant differences at ($p \leq 0.01$) between different available water with regard to average mean of C- humin g /100g for both locations as shown in Fig. (15).

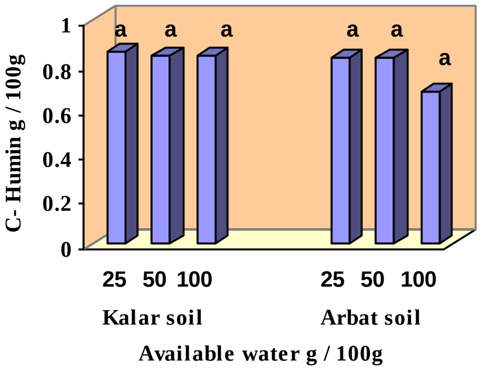


Fig. (15) Effect of available water on average mean of C-humin g / 100g during incubation periods in studied soils.

Also the results presented in Fig. (16) show that 8 months period was superior significantly at ($p \leq 0.01$) to (12 and 4) months period in Kalar location, while in Arbat location 4 months incubation period was superior significantly at ($p \leq 0.01$) to (8 and 12) months , with no difference between the latest two periods for Arbat location , whereas there is significant difference between (4 and 12) months in Kalar location with regard to average mean of C- humin g /100g .

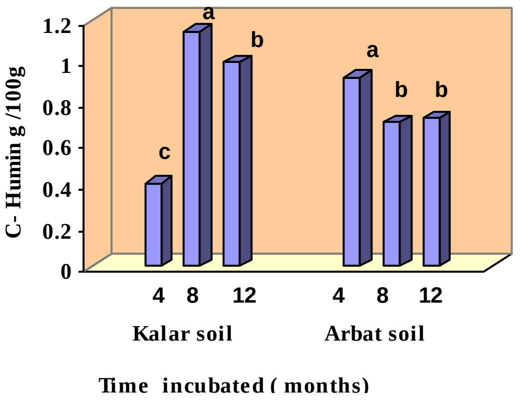


Fig. (16) Effect of incubation periods on average mean of C- humin g / 100g in studied soils.

The effect of manure application rate on the average mean of C–HA g /100g were shown in Fig (17). The results indicated that the 2.5 g /100g manure application rate was superior significantly with regard to C-HA g /100g at $p \leq 0.01$ to the other two rates for both locations.

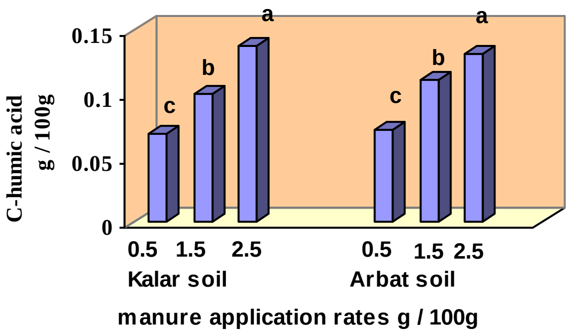


Fig. (17) Effect of manure application rates on average mean of C- humic acid g /100g during incubation periods in studied soils.

In addition, the available water had affected the average mean of C – HA g /100g at level 100 g /100g in Kalar location as compared to (50 and 25) g /100g. While no significant differences were observed for the three available water percent in Arbat location Fig. (18), this may be due to the variation in the texture of these soils at the two locations or microbial activities, and mechanisms of OM decomposition.

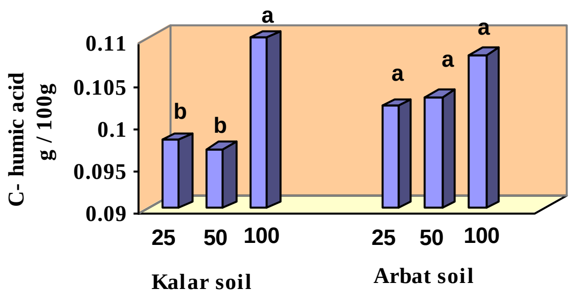


Fig. (18) Effect of available water on average mean of C- humic acid g / 100g during incubation periods in studied soils .

The studies of different incubation periods on the average mean of C – HA g /100g were shown in Fig.(19), the results indicated that 8 months incubation period in Kalar location was superior significantly with regard to average mean of C- HA g /100g at $p \leq 0.01$ to the other two periods , in contrast no significant differences were observed between the three incubation periods in Arbat location.

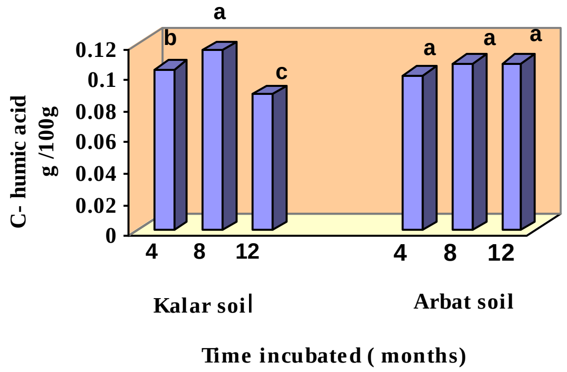


Fig. (19) Effect of incubation periods on average mean of C- humic acid g /100g in studied soils.

Other experiments were conducted to study the effect of manure application rates, available water and incubation periods on FA fraction. These results are shown in Fig.(20 , 21 , 22) were indicated that 0.5 and 1.5 g /100g manure application rates in Kalar location were more effective on FA g /100g content than 2.5 g /100g, whereas 1.5 g /100g was superior significantly at $p \leq 0.01$ to the other two manure application rates in Arbat location. The 25 g /100g available water was different significantly from 100 g /100g available water with respect to average mean of C- FA g /100g in Kalar location, but no significant different were noticed for the three available water g /100g in

Arbat location, this may be due to the variation in soil texture classes . On the other hand, 4 months incubation periods was superior to the other two periods in Kalar location , while 12 months period was superior to the other two periods in Arbat location in contrast to Karla location with respect to average mean of C- FA g /100g , which indicate the obvious effect of soil factor on FA formation.

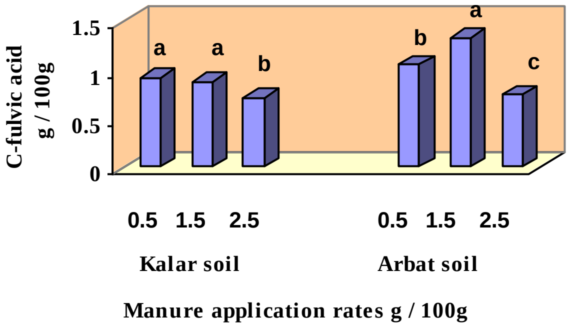


Fig.(20) Effect of manure application rates on average mean of C - fulvic acid g / 100g during incubation periods in studied soils.

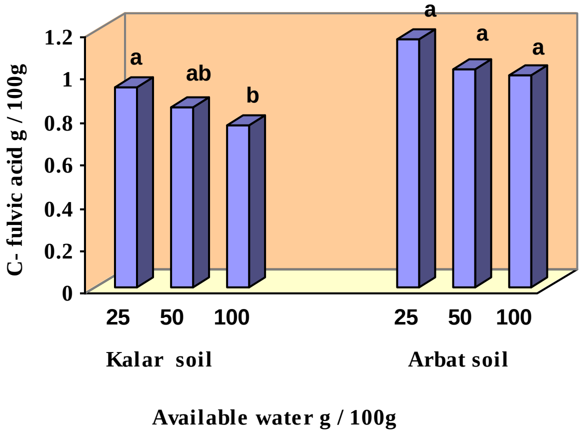


Fig.(21) Effect of available water on average mean of C- fulvic acid g / 100g during incubation periods in studied soils.

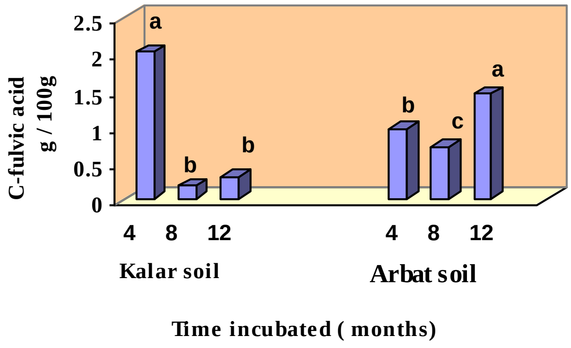


Fig. (22) Effect of incubation periods on average mean of C- fulvic acid g / 100g in studied soils.

The effect of temperature , available water , and manure application rate on the mean values of TSOC remaining g /100g , C – humin g /100g , C- HA g /100g ,C – FA g /100g and HR g /100g were shown in Tables (12 and 13)which are indicated that the highest mean values of TSOC remaining g /100g was 2.544 at 28C^{o} and 4 months incubation periods for Kalar location , which was superior significantly to the other two periods (8 and 12 months). While the highest mean values of TSOC remaining g /100g was 2.279 at 28C^{o} and 12 months incubation periods for Arbat location, which was superior significantly to the other two periods (4 and 8) months.

On the other hand, the highest mean values of TSOC remaining g /100g was 2.680 at 25 g /100g available water and 4 months incubation periods for Kalar location, which was superior significantly to all other interactions except 50 g /100g available water 4 months. Whereas the highest mean values of TSOC remaining g /100g was 2.721at 25 g /100g available water and 12 months incubation periods for Arbat location, which was superior significantly to the (50 and 100) g /100g available water. Also, the highest mean values of TSOC remaining g /100g was 2.675 at 2.5 g /100g manure application rate and 4 months incubation periods for Kalar location , which was superior significantly to all other interactions except 1.5 g /100g manure application rate and 4 months incubation periods. While the highest mean values of TSOC remaining g /100g was 2.555 at 1.5 g /100g manure application rate and 12 months incubation periods for Arbat location , which was superior significantly at $p \leq 0.01$ to all other interactions except 0.5 g /100g manure application rate and 12 months incubation periods. These may be due to the ease of degradable OC which was rapidly consumed by the organisms, these results are in agreement with Leifeld et al. (2001) who indicated that fresh OM or light fraction , which consists of mineral – free OM composed of partly decomposed plant and animal residues , which turn over rapidly and have a specific density considerably lower than that of soil minerals.

The highest mean values of C – humin g /100g was 1.146 at 28 C^o and 8 months incubation periods for Kalar location which was superior significantly at $p \leq 0.01$ to the other periods (4 and 12) months, while the highest mean values of C – humin g /100g was 0.922 at 28 C^o and 4 months incubation periods for Arbat location , which was superior significantly at $p \leq 0.01$ to the other two periods (8 and 12) months. On the other hand, the highest mean values of C – humin g /100g was 1.314 at 50 g /100g available water and 8 months incubation periods, which was superior significantly to all other interactions except 25 g /100g available water and 8 and 12 months for Kalar soil . Whereas the interactions of 4 months incubation periods with 25 g /100g available water gave the highest mean values of C – humin g /100g that was 0.986, which was superior significantly at $p \leq 0.01$ to all other interactions except (50 g /100g available water g /100g and 4, 8, and 12 months incubation periods), (25 g /100g and 12 months) and (100 g /100g and 4 months incubation periods) for Arbat location.

Table (12) Changes in mean values of TSOC remaining g /100g, C- humin , C- humic acid and C- fulvic – acid g /100g as functions of temperatures , available water and manure application rates during incubation periods in Kalar soil.

Humification Products	**Temperatures C^{o}			Available Water g / 100g									Manure application rates g /100g								
	28			25			50			100			0.5			1.5			2.5		
	Time incubated (months)																				
	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12
TSOCremaining g /100g	2.544a	1.466b	1.392b	2.68a	1.533c	1.46c	2.541ab	1.448c	1.365c	2.412b	1.355c	1.35c	2.34b	1.147f	1.183f	2.619a	1.514de	1.415e	2.675a	1.677c	1.578cd
C- Humin g /100g	0.404c	1.146a	0.955b	0.265d	1.182ab	1.132ab	0.416d	1.314a	0.806c	0.532d	0.941bc	1.046bc	0.551c	0.586c	0.559c	0.183d	1.391a	1.007b	0.479c	1.461a	1.418a
$C_{HA\ g/100g}$	0.103b	0.115a	0.087c	0.101bc	0.107b	0.086d	0.102bc	0.105bc	0.085d	0.107b	0.132a	0.091cd	0.091ef	0.063g	0.05g	0.102de	0.111cd	0.086f	0.117bc	0.171a	0.126b
$C_{FA\ g\ 100g}$	2.037a	0.205b	0.35b	2.314a	0.244e	0.243e	2.023b	0.029f	0.474d	1.773c	0.282e	0.213e	1.698c	0.498d	0.574 d	2.334a	0.012f	0.322e	2.079b	0.045f	0.034 f
*HR g /100g	84.119	21.828	31.394	90.112	22.896	22.519	83.628	9.254	40.952	77.944	30.554	22.519	76.453	48.910	52.747	93.013	8.124	28.834	82.093	12.880	10.139

Vales in horizontal row followed by the same letters are not different significantly between themselves according to Duncan Multiple Range test at ($p \leq 0.01$).
*** HR % (Humification rate) = C (HA + FA) * 100 / TOC ---16 (calculated according to Ciavatta et al ., 1990).**
**** Only one level of temperature ($28C^{o}$) has been used representing optimum temperature.**

Table (13) Changes in mean values of TSOC remainin g /100g g , C- humin , C- humic acid and C- fulvic – acid g /100g as functions of temperatures , available water and manure application rates during incubation periods in Arbat soil.

Humification Products	**Temperatures C^{o}			Available Water g /100g									Manure application rates g /100g								
	28			25			50			100			0.5			1.5			2.5		
	Time incubated (months)																				
	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12	4	8	12
TSOC remaining g /100g	1.991b	1.542 c	2.279 a	2.006 bc	1.546de	2.721a	2.02 bc	1.664 d	2.183 b	1.948c	1.42 b	1.946c	1.727d	1.002f	2.503a	2.017c	1.564e	2.555a	2.23b	2.061bc	1.792d
C- Humin g /100g	0.922a	0.701b	0.722b	0.986a	0.624bc	0.884abc	0.951ab	0.892abc	0.663ab	0.829abc	0.589c	0.62bc	0.276d	0.8bc	0.761c	1.103ab	0.22d	0.446d	1.386a	1.083abc	0.961bc
$C_{HA\ g/100g}$	0.099a	0.107a	0.104a	0.099a	0.105a	0.103a	0.1a	0.106a	0.104a	0.098a	0.11a	0.115a	0.088de	0.072ef	0.055f	0.1cd	0.111cd	0.121bc	0.11cd	0.137ab	0.146a
$C_{FA\ g/100g}$	0.97b	0.734c	1.453a	0.921cde	0.817de	1.734a	0.969cde	0.666e	1.416b	1.021cd	0.721de	1.211bc	1.363b	0.13d	1.687a	0.814c	1.233b	1.988a	0.734c	0.841c	0.685c
*HR g /100g	53.692	54.540	68.319	50.847	59.638	67.512	52.921	46.394	69.629	57.444	58.521	68.140	84.019	20.160	69.596	45.315	85.934	82.544	37.848	47.453	46.373

Vales in horizontal row followed by the same letters are not different significantly between themselves according to Duncan Multiple Range test at ($p \leq 0.01$).
*** HR g /100g (Humification rate) = C (HA + FA) * 100 / TOC (calculated according to Ciavatta et al., .1990).**
**** Only one level of temperature ($28C^{o}$) has been used representing optimum temperature.**

Also, the highest mean values of C – humin g /100g was 1.461 at 2.5 g /100g manure application rate and 8 months incubation periods , which was superior significantly to all other interactions except (2.5 g /100g manure application rate and 12 months) and (1.5 g /100g manure application rate and 8 months incubation periods) for Kalar location. The highest mean values of C- humin g /100g was 1.386 g /100g at 2.5 g /100g manure application rate and 4 months incubation periods for Arbat location, which was superior significantly at (2.5 g /100g manure application rate and 8 months incubation periods) and (1.5 g /100g manure application rate and 4 months incubation periods). While the mean value of C- humin g /100g of Kalar location was 0.420 g /100g but it was 0.630 g /100g for Arbat, in general results indicated that C- humin g /100g used for comparison was lower than the studied parameters except 1.5 g /100g manure application rate and 4 months incubation periods for Kalar location and 0.5 g /100g manure application rate and 4 months incubation periods as well as 1.5 g /100g manure application rate and 8, 12 months incubation periods for Arbat location as shown in (Table 8). This may be due to the microbial origin of HS which consists of the metabolic activity including : polysaccharides , polyrenoids and amino sugars as a result the turnover occurs rapidly leading to re – mineralization and converting to the FA (Al – Saadi , 1997) . The results indicated that C – humin were very low due to the higher HR at the lower incubation period. These organic compounds, which composed organic materials are fresh and easily decomposed leaving very low C – humin.

The mean values of C – HA g /100g was 0.115 at 28 C^o and 8 months incubation periods for Kalar location which was superior significantly at $p \leq 0.01$ to the other two periods (4 and 12) months. On the other hand, the highest mean values of C – HA g /100g was 0.132 at 100 g /100g available water and 8 months incubation periods for Kalar location, which was superior significantly to all other interactions. On the contrary, in Arbat location the mean value of C – HA g /100g were not significantly affected by temperature at 28C^o and available water added. But the interactions of 8 months period with 2.5 g /100g manure application rate gave the highest mean value of C – HA g /100g which were 0.171 , that was superior significantly at $p \leq 0.01$ to all other interactions for Kalar location . Whereas, the interactions of 12 and 8 months with 2.5 g /100g manure application rate gave (0.146 and 0.137) mean values of C- HA g /100g for Arbat location, which were the highest mean values compare to all other interactions. While the mean value of C- HA g /100g of Kalar location was 0.090 g /100g but it was 0.080 g /100g for Arbat. In general results indicated that C- HA g /100g used for comparison was lower than the studied parameters except 0.5 g /100g manure application rate and 8,12 months incubation periods and 1.5 g /100g manure application rate and 12 months incubation periods for Kalar location and 0.5 manure application rate and 8 ,12 months incubation periods for Arbat location as shown in (Table 8). This may be due to the attachment of some humic substances, to the minerals in soil forming metal binding and affecting their solubility and other properties (Schulten And Schnitzer, 1995; and Rumpel et al., 2000).

The highest mean values of C – FA g /100g was 2.037 g /100g at 28C^o and 4 months incubation period for Kalar location, which was superior significantly at $p \leq 0.01$ to the other two periods (8 and 12) months. Whereas the highest mean values of C-FA g /100g was 1.453 at 28 C^o and 12 months incubation period for Arbat location, which was superior significantly to the other two periods (4 and 8) months. On the other hand, the highest mean values of C – FA g /100g was 2.314 at 25 g /100g available water and 4 months incubation period for Kalar location, which was superior significantly to all interactions. While the highest mean values of C – FA g /100g was 1.734 at 25 g /100g available water and 12 months incubation period for Arbat location, which was superior significantly to all other interactions. Also, the values of C – FA g /100g was 2.334 at 1.5 g /100g manure application rate and 4 months incubation period for Kalar location, which was superior significantly to all other interactions. While in Arbat location the highest mean values of C – FA g /100g was 1.988 at 1.5 g /100g manure application rate and 12 months incubation period for Arbat location , which was superior significantly to all interactions except 0.5 g /100g manure application rate and 12 months incubation period. While the mean value of C- FA g /100g of Kalar location was 0.770 g /100g but it was 0.550 g /100g for Arbat, in general results indicated that mean values of C- FA g /100g used for comparison was lower than the studied parameters except (0.5 , 1.5 , 2.5) g /100g manure application rates and 8,12 months incubation periods for Kalar location and 0.5 manure application rate and 8 months incubation periods for Arbat location as shown in (Table 8). This may be due to the FA fraction more rapidly mineralized than HA. (Qualls, 2004).

The results in Tables (12 and 13) indicated that FA were increased in comparison with HA, this was expected since the rate of decomposition were increased. On the other hand, the FA mineralized was not leached through laboratory experiments, which results increase in FA percent. However, in natural soil body the FA percent would be less due to high solubility of FA in water and leached through soil profile. So, the FA will be less than the HA. Generally, the highest HR g /100g values were observed at 4 months period for all the studied parameters at Kalar location, whereas, there are fluctuations at 8 and 12 months for Arbat location. The fluctuation in the result of HR may be due to different activity and type of microorganisms or death and decomposition of microorganisms itself leading to decrease in C / N ratio, which increase in decomposition. In addition, it could be due to the change in the environmental conditions such as the changes in type and concentration of enzymes, which may be affecting on the activity of other types of microorganisms.

Humification rate g /100g expressed as indicator for humification process, where higher value of HR indicates higher humification process (Valarini et al., 2002 and Yang et al.,2004). This may be due to humic substances containing higher aromatic chains compared to aliphatic chains. Dick, et al., (1999); and Aranda and Oyonarte, (2006) who concluded that HS are inherently difficult for microbes to mineralize, and this property can contribute to the sequestration of C in soil and their specific hypothesis were (i) the HA fraction would be mineralized with $t_{1/2}$ on the order of years, (ii) the HA fraction would be mineralized most slowly of all fractions, (iii) the FA fraction would be more rapidly than the HA fraction.

4.3 The kinetics of organic matter decomposition

Chemical changes in soils are relatively simple to study in comparison to biological changes, where the activity of microorganisms controls the rates, often through the production of an enzyme. The rates depend on the factors controlling microbial activity: such as temperature, water supply, nutrient supply and aeration along with the availability of the materials being broken down called the substrate. The rate of decomposition of OM depends on much biological process. Although the loss of C can be followed by measuring the soil's respiration rate, or the decrease in OC can be measured directly, the overall process is complex. Mathematical analysis has, however, proved useful in extending our understanding of the factors governing decomposition rates. It based on the concept that SOM is composed of various fractions, which break down at different rates with standard kinetics analysis applied to each fraction, (Rowell, 1996).

Several equations have been used to compare the goodness of fit of different mathematical models to describe the kinetic of SOC mineralization ,these models include : Zero-Order, First – Order , Second –Order , Hyperbolic , Parabolic diffusion , Modified

Elovich , Power function , Logarithmic function equations . Table (14) shows the values of determination coefficient (R^2) and standard error (SE) of estimate of these various kinetic equations for Kalar and Arbat locations. Hyperbolic model compared to the various kinetic models is considered as the best model for describing SOC mineralization since it has given the highest R^2 and the lowest SE of estimate which were (0.870 , .082) and (0.760 , 0.100) for (R^2) and SE of estimate at Kalar and Arbat locations respectively. These results are in agreement with Alvarez and Alvarez, (2000) who indicated that the exponential and hyperbolic models showed a good description for SOC mineralization data ($R^2 > 0.982$).

Tables (15, 16 and 17) show the values of R^2 and SE of estimate of hyperbolic model at various temperature, manure application rates, and available water for the studied soils. The highest values of R^2 (0.963 , 0.899) were noticed at 45 C° as compared with others and also gives the lowest SE of estimate (0.030 and 0.055) for Kalar And Arbat location respectively at 45 C°. These results were expected to be due to the higher SOC mineralization at 45C° as compared to 5 And 28 C° as mentioned before in section (4.1)

Table (14) the determination coefficient (R^2) and standard error (SE) of estimate of various kinetic equations used to describe the degradation of organic matter in studied soils.

Equations	Kalar		Arbat	
	R^2	SE	R^2	SE
Zero- order	0.140	0.480	0.24	0.54
First-order	0.240	0.328	0.312	0.399
Second –order	0.270	0.230	0.366	0.313
Hyperbolic	0.870	0.082	0.760	0.100
Modified Elovich	0.256	0.451	0.364	0.503
Logarithmic function	0.250	0.450	0.364	0.503
Parabolic diffusion	0.190	0.470	0.297	0.529
Power function	0.340	0.290	0.454	0.355

On the other hand no variations in the values of R^2 and SE of estimate of hyperbolic model were observed at (1.5 and 2.5) g /100g manure application rates for Kalar location , while their values did change at (1.5 and 2.5) g /100g which gave (0.757, 0.101) ,and (0.902 , 0.077) respectively at Arbat location. On the contrary the highest and the lowest values of R^2 and SE of estimate were noticed at (0.5 g /100g) manure application rates which were (0.825 ,0.081) and (0.554 , 0.125) at Kalar and Arbat locations respectively . This may be due to the increase of microbial activity and the death of cells which cause an in increase substrate availability, (Giller et al., 1998). No significant differences for the values of R^2 and SE of estimate of hyperbolic model were seen at (25, 50, and 100) g /100g available water at Kalar location. But the highest and the lowest values of (R^2) and SE of estimate of hyperbolic model were recorded at (100 g /100g) available water which were (0.805 , 0.087) , compared to (0.725 ,0.115) and (0779 , 0.101) for 25 and 50 g /100g at Arbat location respectively. This may be due to the soil type, which differs in soil texture as shown in Table (8). This affects the water holding capacity of the soil that may have influence on the decomposition rate, which results from the microbial activity.

Table (15) The determination coefficient (R^2) and standard error (SE) of estimate of Hyperbolic equation at different temperatures (C^o) used to describe the degradation of organic matter in studied soils.

Equation	Kalar soil						Arbat soil					
	Temperatures C^o						Temperatures C^o					
	5		28		45		5		28		45	
	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE
Hyperbolic	0.851	0.100	0.786	0.142	0.963	0.030	0.675	0.153	0.739	0.098	0.899	0.055

Table (16)The determination coefficient (R^2) and standard error (SE) of estimate of Hyperbolic equation at different manure application rates used to describe the degradation of organic matter in studied soils.

Equation	Kalar soil						Arbat soil					
	Manure application rates g /100g						Manure application rates g /100g					
	0.5		1.5		2.5		0.5		1.5		2.5	
	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE
Hyperbolic	0.825	0.081	0.888	0.078	0.892	0.093	0.554	0.125	0.757	0.101	0.902	0.077

Table (17)T he determination coefficient (R^2) and standard error (SE) of estimate of Hyperbolic equation at different available water used to describe the degradation of organic matter in studied soils.

Equation	Kalar soil						Arbat soil					
	Available water g /100g						Available water g /100g					
	25		50		100		25		50		100	
	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE	R^2	SE
Hyperbolic	0.872	0.086	0.885	0.081	0.875	0.081	0.725	0.115	0.779	0.101	0.805	0.087

Tables (18, 19, and 20) shows changes in mean values of half –life's, decomposition rates and Q_{10} as a function of different manure application rates during incubation periods. Values of half- life's ranged from 14.090 to 15.010 and 14.700 to 16.300 months at 2.5 and 0.5 g /100g manure application rates for Kalar and Arbat locations respectively. It can be noticed that with an increase of manure application rate will lead to decrease of half –life's. Since the half-life's is depended on the rate of decomposition, it will be expected at high rate decomposition, the high life's will be low. While no differences for the values of half –life's were seen at (25, 50, 100) g /100g of available water for both location. While the highest values of half-life's were 16.021 and 17.147 at 45 and 28C^o, whereas the lowest values were 13.667 and 14.009 months at 28 and 45C^o for Kalar and Arbat locations respectively. This may be due to different biological activities at each stage of decomposition. These results are in agreement with the findings of (Lynch, 1991,Guiraud and Marol , 1992) who showed that the plant materials that form SOM contain compounds that decompose with half – decay times ($t_{1/2}$) which vary from days to a few years and the decomposition rate of lignin , one of the slowest decomposing substances present in the original plant material, has been reported to be 0.18 yr^{-1} (corresponding to a half time of decay , or $t_{1/2}$ Of 3.8 yr) in a temperate soil.

The results in Tables (18, 19, and 20) showed that the decomposition rates (K) were increased with increasing manure application rates and the highest K values have been obtained at 50 g /100g available water for both locations. While the (K) values were decreased with increasing temperatures at Kalar and Arbat locations. This may be due to soil texture and different microbial activity. Theses results are in agreement with Wilts, et al., (2004).

The results from Tables (18, 19 , and 20) indicated that the Q_{10} values increased with increasing the manure application rates , but nearly remain constant with different available water added , simply because moisture percent was not limited to the growth of microorganisms. On the contrary, Q_{10} values decreased with increasing temperature for both locations particularly at 45C^o. It can be concluded that SOC decomposition in soils was sensitive to changes in temperature (Q_{10}).These results are in agreement with results reported by (Lipson , et al. 2001 , and Rey, et al . 2005) who found that a temperature sensitivity factor (Q_{10}) of 3.3 , 2.7 and 2.2 for the 0-5cm , 5-10 cm and 10-20 cm layers depth , respectively . The sensitivity of C_{min} to fluctuating moisture depended on temperature. However, the (Q_{10}) was not significantly affected by soil moisture.

Table (18) Changes in the mean values of half life's, Decomposition rates, and Q_{10} as functions of different manure application rates during incubation periods in studied soils.

Parameters	Kalar soil			Arbat soil		
	Manure application rates g /100g			Manure application rates g /100g		
	0.5	1.5	2.5	0.5	1.5	2.5
Half life£ (month)	15.010	14.690	14.090	16.300	16.170	14.700
Decomposition rate €(month $^{-1}$)	0.032	0.040	0.049	0.025	0.032	0.043
*Q_{10}	1.389	1.491	1.632	1.295	1.389	1.544

Table (19) Changes in the mean values of half-life's, Decomposition rates, and Q_{10} as functions of different available water during incubation periods in studied soils

Parameters	Kalar soil			Arbat soil		
	Available water g /100g			Available water g /100g		
	25	50	100	25	50	100
Half life (month)	14.600	14.380	14.600	15.794	15.417	15.521
Decomposition rate (month $^{-1}$)	0..041	0.041	0.039	0.034	0.035	0.032
Q_{10}	1.514	1.517	1.487	1.411	1.419	1.388

Table (20) Changes in the mean values of half life's, Decomposition rates, and Q_{10} as functions of different temperatures during incubation periods in studied soils.

Parameters	Kalar soil			Arbat soil		
	Temperatures C^o			Temperatures C^o		
	5	28	45	5	28	45
Half life (month)	14.584	13.667	16.021	15.622	17.147	14.009
Decomposition rate (month $^{-1}$)	0.044	0.050	0.028	0.040	0.030	0.030
Q_{10}	1.557	1.653	1.327	1.503	1.357	1.360

£ The half life was estimated from the regression line of hyperbolic model
($t_{1/2} = C_o$ / slope) ---17

€ The decomposition rates were calculated by determining the slope of the regression line of hyperbolic model.

* The Q_{10} was calculated from the exponential coefficient according to the following Formula: $Q_{10} = \exp(10\,K)$ ------------------------------------ 18
calculated according to Lipson , et al. 2001)

4.4 Relationship of organic matter fraction with micronutrients (Fe, Mn, Cu, and Zn)

The results presented in Fig. (23) show that $5C^{o}$ was superior significantly at $p \leq 0.01$ with regard to average mean values of available Fe to 28 and 45 C^{o} for both locations. This may be due to the formation of chelated iron complex with SOM, which results in higher availability of Fe in soil solution because of the increase in the rate of decomposition as a function of temperature increase, which results in oxidizing and precipitating Fe as Fe $(OH)_3$.

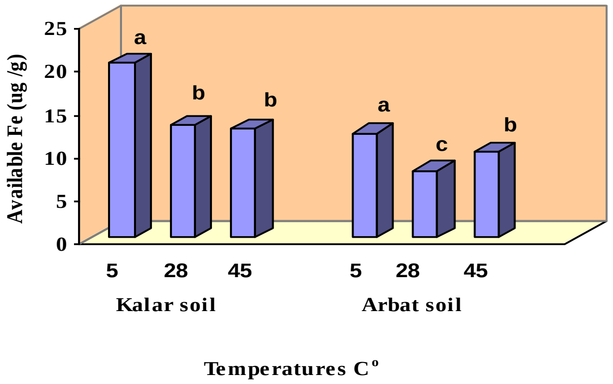


Fig. (23) Effect of Temperatures C° on average mean of available Fe in (ug /g) during incubation periods in studied soils.

On the other hand, Fig. (24) shows that 2.5 g /100g manure application rates was superior significantly at $p \leq 0.01$ with regard to average mean values of available Fe to the 0.5 g /100g at Kalar location. Whereas in Arbat location 2.5 g /100g manure application rate was superior significantly at $p \leq 0.01$ to the 1.5 and 0.5 g /100g manure application rates. This may be attributed to the high percentage of HA g /100g at 2.5 g /100g manure application rate due to carboxyl (COOH) , hydroxyl (OH) and carbonyl (C = O) functional groups in humic substances which are capable of forming complexes with Fe and this has an important role on iron availability. These results are in agreement with those reported by (Tripathi et al.,1993 ;Caselles et al., 2005, and Wei et al. , 2006) who showed that the available Fe in soils was largely influenced by OC content , and there was significant correlation between DTPA – Fe and OC (r = 0.55 **). On the other hand , Hallorans et al., (2004) indicated that materials such as sewage sludge ,animal manure , humates and compost may be rich in iron and in metal binding biochemicals that help keep iron and other metals in solution through chelation and also stimulate chemical and biological reactions which make iron more available.

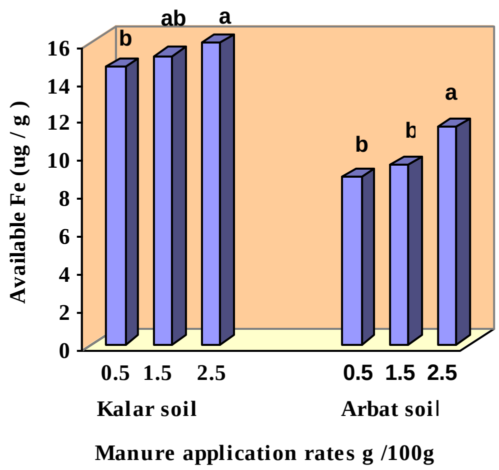


Fig. (24) Effect of manure application rates on the average mean of available Fe in (ug / g) during incubation periods in studied soils.

With respect to available water, there were no significant differences at $p \leq 0.01$ between different available water with regard to average mean of available Fe for both locations as shown in Fig. (25). This may be due to variation in texture class between the soils. Whereas, Wheeler et al., (1985) found that at 80 g /100g F.C iron was less available than when soil was waterlogged and as a result, conditions and solubilization of reducing forms of iron would increase.

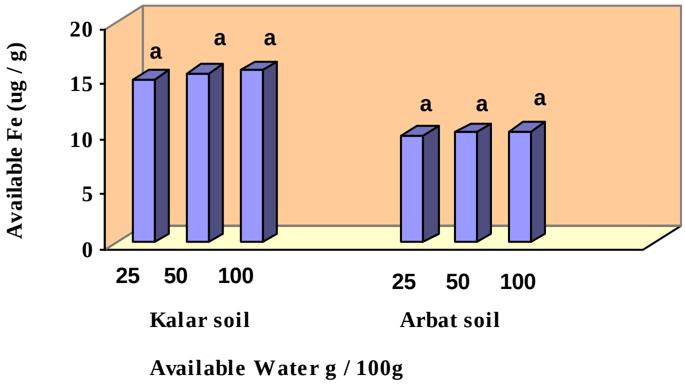


Fig. (25) Effect of available water on average mean of available Fe in (ug / g) during incubation periods in studied soils .

Finally, 2 months incubation period was superior significantly with regard to average mean of available Fe at $p \leq 0.01$ to the other three periods (4 , 8 , and 12) months for Kalar location , while 8 months incubation period was superior significantly at $p \leq 0.01$ to the three periods (2 , 4 and 12) months for Arbat location Fig.(26). This may be due to the activity of soil microbial processes such as mineralization and immobilization at each stage of SOM decomposition during incubation periods.

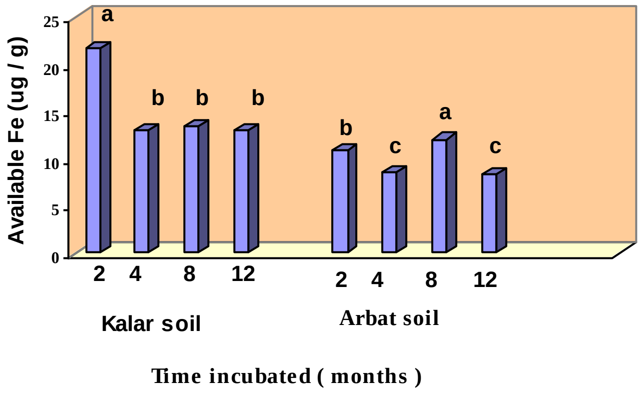


Fig . (26) Effect of incubation periods on average mean of available Fe in (ug / g) in

The results in Fig. (27) show that $5C^o$ was superior significantly with regard to average mean of available Mn at $p \leq 0.01$ to 28 and 45 C^o during incubation periods for Kalar and Arbat locations. Mn also behaved the same behavior as Fe when associated with OM, since it tends to precipitate as MnO_2 and thus is less available in soil solution. Wet weather favors the presence of Mn^{2+}, whereas warm dry conditions encourage the formation of less available oxidized forms of Mn. Dry weather either induces or aggravates Mn deficiency, particularly in fruit trees. Increasing soil temperature during the growing season improves Mn uptake, presumably because of greater plant growth and root activity (Tisdale et al., 1993).

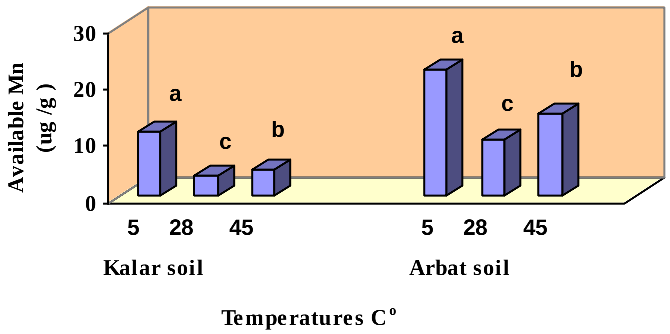


Fig. (27) Effect of temperatures C° on average mean of available Mn in (ug /g) during incubation periods.

With regard to available Mn, at 2.5 g /100g manure application rate was superior significantly to 0.5 and 1.5 g /100g at $p \leq 0.01$ during incubation periods for Kalar and Arbat locations as shown in Fig. (28). This may be due to the bioavailability of Mn in soil which is strongly influenced by the amount and the quality of OM that can react with it , forming complexes and chelates of varying stability (Leita et al. ,1999). Availability of Mn^{2+} can be strongly influenced by reactions with OM. Additions of natural organic materials such as peat moss, compost, and wheat and clover straw have increased the solution and exchangeable Mn (Tisdale et al., 1993).

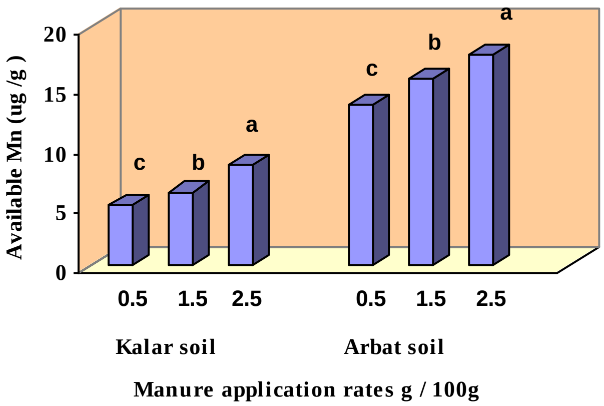


Fig. (28) Effect of manure application rates on available Mn in (ug / g) during incubation periods in studied soils.

Also, the results presented in Fig. (29) show that the 25 g /100g available water was superior significantly with regard to available Mn, at $p \leq$ 0.01 to all others at Kalar location. While the (25 and 50) g /100g available water were superior significantly at $p \leq$ 0.01 to 100 g /100g available water at Arbat location with regard to available Mn .

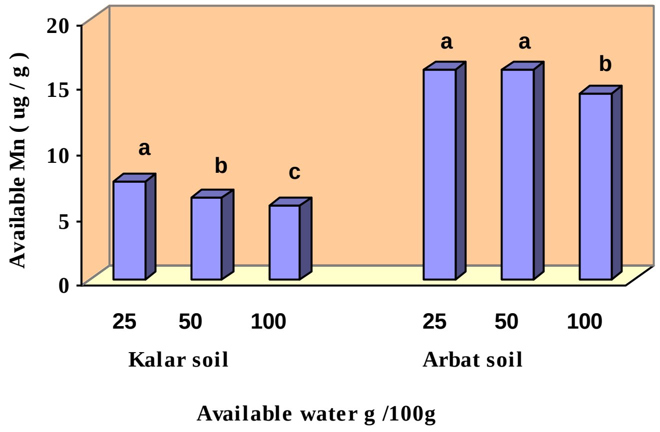


Fig. (29) Effect of available water on average mean of available Mn in (ug /g) during incubation periods.

In terms of incubation period Fig. (30) shows that 2 months incubation period was superior significantly with regard to available Mn at $p \leq 0.01$ to (4, 8 and 12) months at both Kalar and Arbat locations. The availability of Mn decreased at 8 months incubation period as compared to the other periods, and it may be due to the precipitation of soluble Mn as MnO_2 during this period. The high value of available Mn at 2 months incubation period may be due to the fresh organic residue and high percentage of humification rate as indicated in Table 21, which decomposed rapidly at the beginning of the incubation period, leading to an increase in soluble complexes with OM, and as a result, the availability of Mn in soil solution will increase as well.

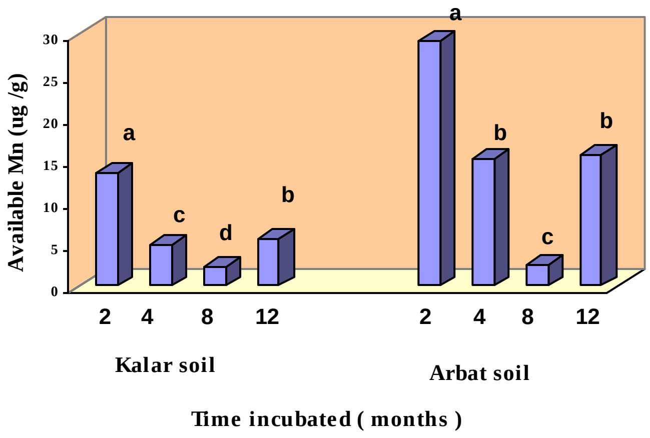


Fig. (30) Effect of incubation periods on average mean of available Mn in (ug / g) in studied soils.

With regard to available Cu, in Fig. (31) shows that 5 C^o was superior significantly at p 0.01 to 28 and 45 C^o during Incubation periods at Kalar location.
While 45 C^o was superior significantly at $p \leq 0.01$ to the others except 5 C^o at Arbat location. In Kalar location available Cu decreased with increasing temperature and carbonate minerals contents , may be due to its precipitation as $Cu(OH)_2$ and or $CuCO_3$ (Lindsay ,1979). While in Arbat location, higher temperature led to an increase of available Cu, this may be due to the formation of soluble complex of Cu with FA. (Hodgson et al., 1966)

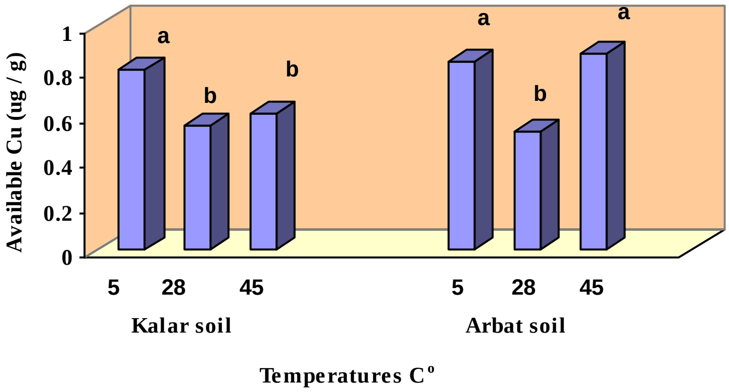


Fig . (31) Effect of temperatures on average mean of available Cu in (ug /g) during incubation periods in studied soils.

On the other hand, Fig. (32) shows that 2.5 g /100g manure application rate was superior significantly with regard to average mean of available Cu at $p \leq 0.01$ to 0.5 and 1.5 g /100g during incubation periods at Kalar and Arbat locations. These results are in agreement with Randhawa and Singh, (1996) and Sauve et al. (1996) who showed that most Cu exists as organic complexes in soil solution, and all the chemical fractions of Cu were significantly related with OC status in soils. Hodgson et al, (1966) reported that the association of Cu with FA formed soluble complexes but with HA formed insoluble complexes. Therefore, SOM can lead to an increase or decrease of the availability of Cu by either binding or increasing its mobility.

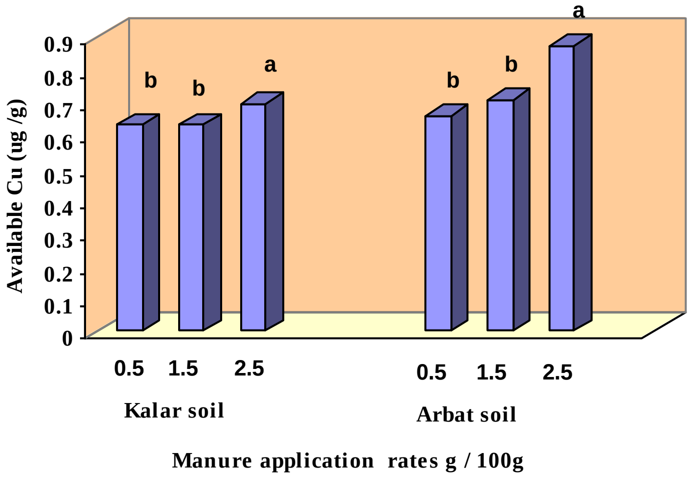


Fig. (32) Effect of manure application rates on average mean of available Cu in (ug /g) during incubation periods in studied soils.

Also, the results presented in Fig.(33) show that the 25 g /100g available water was superior significantly with regard to average mean of available Cu at $p \leq 0.01$ to (50 and 100) g /100g at Kalar location . Whereas there was no significant differences between different available water with regard to available Cu at Arbat location. These results are in agreement with results reported by Ponizovsky, et al. (2006) who showed that copper concentration decreased with the increase of moisture content.

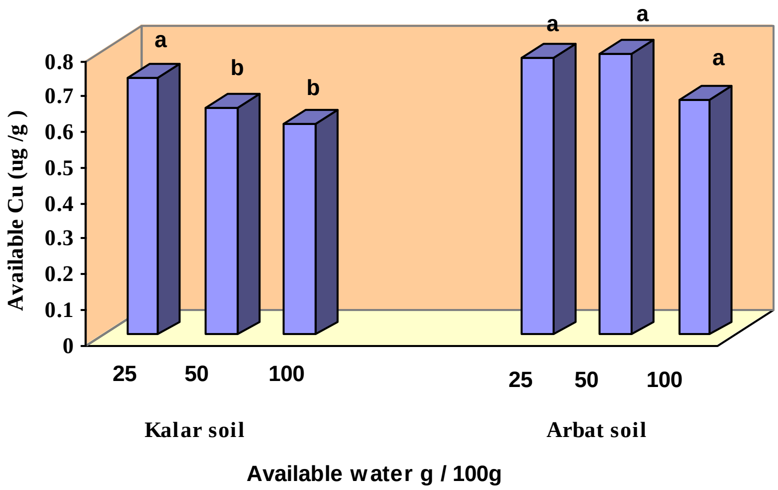


Fig. (33) Effect of available water on average mean of available Cu in (ug /g) during incubation periods in studied soils.

Finally, the results in Fig. (34) show that 12 months was superior significantly with regard to average mean of available Cu at $p \leq 0.01$ to (2, 4, 8) periods for Kalar and Arbat locations. The mobility of Cu increased as incubation time increased, may be due to the addition of OM with incubation period leading to an increase of Cu solubility because of organo – metallic complexes formation. These results are in agreement with Usman, et al., (2004).

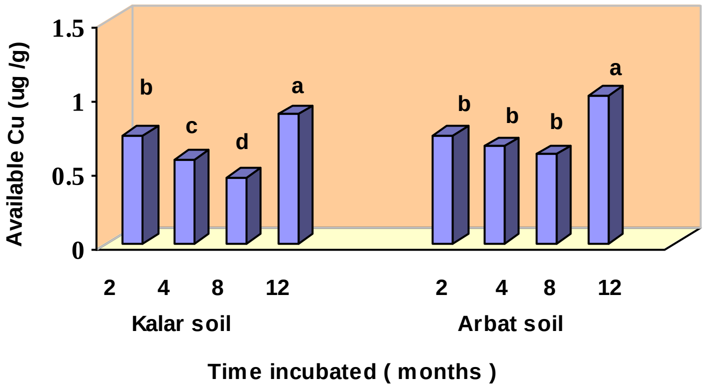


Fig. (34) Effect of incubation periods on average mean of available Cu in (ug /g) in studied soils.

The results presented in Fig. (35) show that $5C^o$ was superior significantly with regard to available Zn at $p \leq 0.01$ to 28 and 45 C^o at Kalar and Arbat locations. Higher temperature and carbonate contents led to the decrease of available Zn may be due to the precipitation of Zn as $Zn (OH)_2$ and or $ZnCO_3$. These results are in agreement with Almas and Singh, (2001) and Usman et al., (2004).

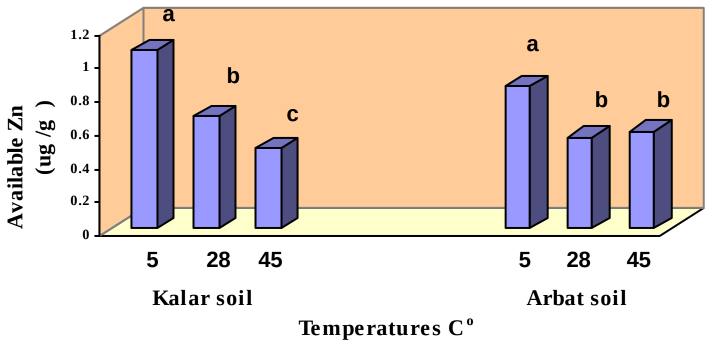


Fig. (35) Effect of temperatures on average mean of available Zn in (ug /g) during incubation periods in studied soils .

On the other hand, Fig. (36) shows that 2.5 g /100g manure application rate was superior significantly with regard to average mean of available Zn at $p \leq 0.01$ to 0.5 and

1.5 g /100g at Kalar location . While 2.5 g /100g manure application rate was superior significantly to the others except 1.5 g /100g at Arbat location. These results are in agreement with Almas and Singh, (2001) who reported that the addition of OM to soil might thus have increased the uptake of Zn either by increasing the potential mobility of the investigated Zn by formation of soluble organo - metallic complexes or improving the growth conditions of microorganisms through the additional nutrients provided.

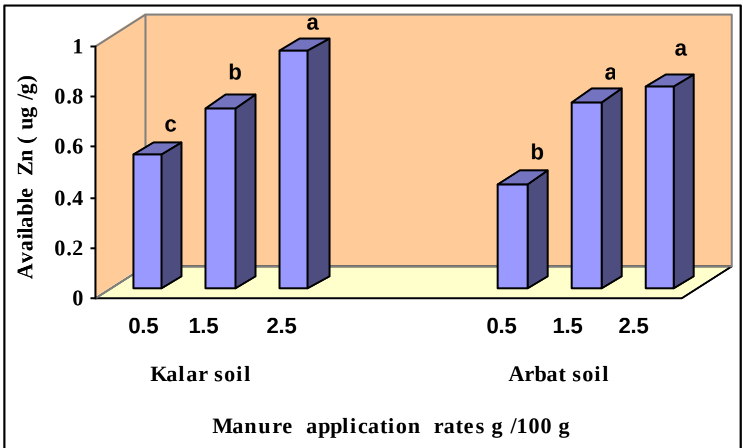


Fig. (36) Effect of manure application rates on average mean of available Zn in (ug /g) during incubation periods in studied soils.

Also the results presented in Fig. (37) show that the 25 g /100g available water was superior significantly with regard to available Zn at $p \leq 0.01$ to 50 and 100 g /100g at Kalar location, while there was no significant differences between different available water regard to available Zn at Arbat location .

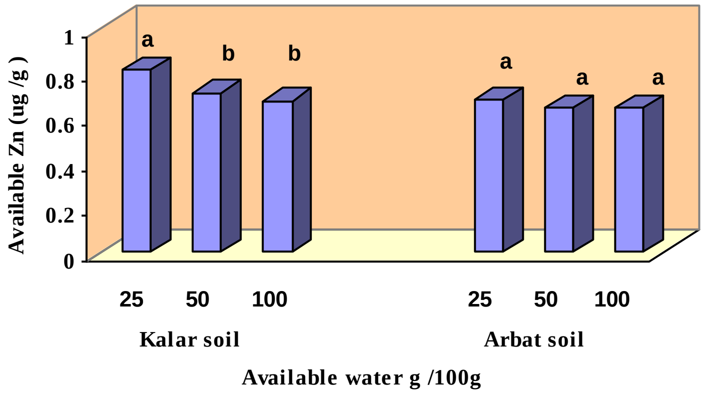


Fig. (37) Effect of available water on average mean of available Zn in (ug /g) during incubation periods in studied soils.

Finally, with regard available Zn the results in Fig. (38) show that 2 months incubation period was superior significantly at $p \leq 0.01$ to (4, 8, 12) periods at Kalar and Arbat locations. These results are in agreement with Usman et al., (2004) who showed that the mobility of Zn decreased as incubation time increased. This may be attributed to the tendency of soluble Zn to be precipitated by $CaCO_3$ in calcareous soil during incubation period.

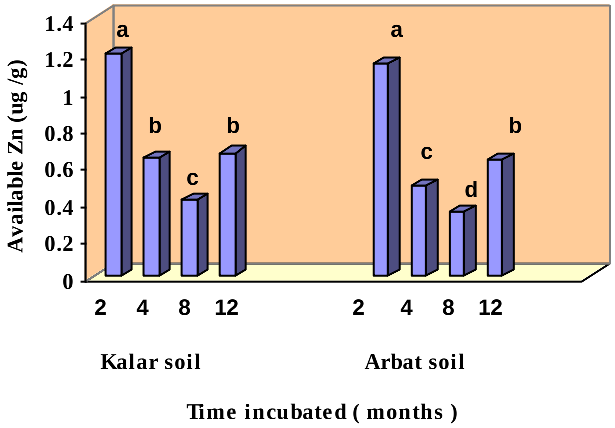


Fig. (38) Effect of incubation periods on average mean of available Zn in (ug /g) in studied soils.

The effects of temperatures, available water and manure application rates on the mean values of available (Fe, Mn, Cu, and Zn) were shown in Tables (21, 22, 23, and 24). The highest mean values of available Fe were 35.682 and 16.815 ug /g at 5 C^o and 2 months incubation periods at Kalar and Arbat locations respectively , which were superior significantly at $p \leq 0.01$ to all other interactions , whereas the lowest mean values were 10.648 and 5.469 ug /g at 45 and 28 C^o and 12 months incubation period. With respect to available water, the highest mean value was 23.618 ug /g at 25 g /100g available water and 2 months incubation periods, which was superior significantly at $p \leq 0.01$ to all other interactions in Kalar location. While the highest mean value of available Fe (12.209 ug /g) was superior to all other interactions except (25 , 50 and 100) g /100g available water and 2 months as well as 25 and 100 g /100g available water and 8 months incubation periods for Arbat location. Finally , the highest mean values of available Fe superior to all others was 22.258 ug /g at 0.5 and 2.5 g /100g manure application rates and 2 months incubation period for Kalar location . Whereas the highest mean value for Arbat location 12.817 ug /g was noticed at 2.5 g /100g manure application rate and 4 months incubation period, which was superior significantly to all other interactions except 1.5 g /100g and 2 months incubation period and (0.5, 1.5 and 2.5) g /100g manure application rate and 8 months. While the available Fe of Kalar location was 14.250 ug /g but it was 6.950 for Arbat. Results indicated that available Fe used for comparison was lower than the studied parameters except (0.5 ,1.5 , 2.5) g /100g manure application rates and 4 , 8 , 12 months incubation periods for Kalar location and (0.5 , 1.5) g /100g manure application rates and 4 months incubation period for Arbat location as shown in (Table 8). In general low temperature, short incubation period, and high manure application rate increase the availability of iron in both locations, with fluctuation in the effect of moisture content. It may be due to relatively high HR g /100g. The content of DTPA–Fe is considered adequate in these soils, which contained more than 4.5 ug / g where, critical level of iron is 4.5 ug / g in the soil as suggested by (Lindsay and Norvell, 1978). These results might be due to both or each effect of temperature, available water and manure application rates, and incubation periods individually as mentioned before.

Table (21) Changes in mean values of available Fe in (ug /g) as functions of temperatures, available water, and manure application rates during incubation periods in studied soils.

Kalar soil									
Time incubated (months)	Temperatures C°			Available Water g /100g			Manure application rates g /100g		
	5	28	45	25	50	100	0.5	1.5	2.5
2	35.682 a	16.997 b	12.734 ef	23.618 a	22.167 b	19.627 c	22.258 a	20.897 b	22.258 a
4	14.187 d	10.648 g	13.987 de	12.552 efg	12.273 fg	13.996 d	11.458 g	14.006 cd	13.361 cdef
8	15.101 cd	12.405 f	12.746 ef	13.074 defg	13.422 defg	13.756 de	12.765 defg	13.663 cde	13.824 cde
12	15.999 bc	12.137 f	10.829 g	13.317 defg	13.550 def	12.099 g	12.189 fg	12.500 efg	14.276 c
Arbat soil									
2	16.815 a	8.471 c	7.201 cd	11.101 a	11.010 a	10.375 ab	9.922 c	12.099 ab	10.466 bc
4	8.289 c	6.022 d	11.184 b	8.430 c	8.466 c	8.599 bc	6.566 b	6.112 b	12.817 a
8	12.180 b	11.102 b	12.965 b	12.135 a	12.209 a	11.904 a	11.776 abc	12.074 ab	12.398 a
12	11.157 b	5.469 d	8.453 c	8.864 bc	8.380 c	7.836 c	7.283 b	7.637 b	10.159 c

Values followed by the same letters for each treatment in combination with time incubation are not different significantly between themselves according to Duncan's Multiple Range test at ($p \leq 0.01$).

Table (22) shows that the highest mean values of available Mn were 31.355 and 55.485 ug /g at 5 C^o and 2 months incubation periods in Kalar and Arbat location respectively , which were superior to all other interactions . While the lowest value of available Mn for Kalar location was 1.560 ug /g at 28 C^o and 8 months incubation period. On the contrary, the lowest value in Arbat location was 1.827 ug /g, which did not differ significantly at $p \leq 0.01$ from (5 and 45 C^0) and 8 months incubation period. With regard to available water, the highest mean value of available Mn was 13.379 ug /g at 25 g /100g available water and 2 months incubation period in Kalar location, which was superior significantly to all other interactions except 50 g /100g available water and 2 months incubation period. While the highest mean value was 32.203 ug /g at 25 g /100g available water and 2 months incubation period for Arbat location, which was superior significantly to all other interactions. Finally the highest mean value of available Mn was 16.316 ug /g at 2.5 g /100g manure application rate and 2 months incubation period which was superior significantly at $p \leq 0.01$ to all other interactions for Kalar location. While the highest mean value of available Mn was 29.696 at 1.5 g /100g manure application rate and 2 months incubation period for Arbat location, which was superior significantly to all other interactions except 2.5 g /100g manure application rate and 2 months incubation period. While the available Mn of Kalar location was 2.860 ug /g but it was 11.250 ug/ g for Arbat. Generally results indicated that available Mn used for comparison was lower than the studied parameters except (0.5, 1.5, 2.5) g /100g manure application rates and 8 months incubation periods for both locations as shown in (Table 8). Therefore, 2 months incubation, low temperature, and moisture content up to 50 g /100g with relatively high manure application rate increase the Mn availability in studied soils. The mean value of DTPA- Mn is much higher than the critical limit of 1.0 ug /g (FAO, 1980; Soltanpour, 1985 and Martens and Lindsay, 1990). These results might be due to both or each effect of temperature, available water and manure application rates, and incubation periods individually as mentioned before.

Table (22) Changes in mean values of available Mn in (ug /g) as functions of temperatures, available water, and manure application rates during incubation periods in studied soils.

Time incubated (months)	Temperatures C°			Available Water g /100g			Manure application rates g /100g		
	5	28	45	25	50	100	0.5	1.5	2.5
Kalar soil									
2	31.355 a	4.931 d	4.012 ef	13.379 a	13.131 a	11.787 b	10.750 c	13.232 b	16.316 a
4	6.847 c	4.258 def	3.715 f	6.196 c	4.684 de	3.941 e	3.668g	4.772 f	6.381 e
8	2.677 g	1.560 h	2.356 g	2.295 f	2.144 f	2.155 f	2.178 h	2.344 h	2.072 h
12	4.811 de	3.936 f	8.100 b	6.418 c	5.389 d	5.041 d	3.730 g	4.310 fg	8.807 d
Arbat soil									
2	55.485 a	14.275 e	17.612 d	32.203 a	29.469 b	25.699 c	28.124 b	29.696 a	29.551 a
4	19.201 c	14.388 e	11.640 f	15.712 e	15.566 e	13.951 f	13.753 e	14.327 e	17.148 d
8	2.824 h	1.827 h	2.820 h	2.827 h	2.378 h	2.267 h	2.529 g	2.434 g	2.509 g
12	12.025 f	8.456 g	26.216 b	13.969 f	17.348 d	15.381 e	9.417 f	16.024 d	21.256 c

Values followed by the same letters for each treatment in combination with time incubation are not different significantly between themselves according to Duncan Multiple range test at ($p \leq 0.01$).

The results in Table (23) show that the highest mean value of available Cu was 1.108 ug /g for Kalar location at 5 C^o and 2 months incubation period, which was superior significantly to all other interactions, while the lowest value was 0.209 ug /g at 28 C^o and 8 months incubation period. Whereas the highest mean value was 1.174 ug /g at 45 C^o and 12 months incubation period in Arbat location , which was superior significantly to all other interactions except 5 C^o and (2 and 12) months incubation period. With respect to available water, the highest mean value of Cu was 0.960 ug /g in Kalar location has been obtained at 25 g /100g available water and 12 months incubation period, which was superior significantly to all other interactions except 50 g /100g available water and 12 months incubation period. While the highest mean values were (1.054,1.050 ug /g) in Arbat location at 25 g /100g and 50 g /100g available water and 12 months incubation period, while the highest mean values in Arbat location after 12 months incubation period, and there are no significant differences between the effects of moisture levels on Cu availability. Finally, the highest mean values of available Cu were 1.069 and 1.326 ug /g at 2.5 g /100g manure application rate and 12 months incubation period for Kalar and Arbat locations respectively, which were superior significantly to all other interactions. The available Cu of Kalar location was 0.610 ug /g but it was 0.640 ug /g for Arbat., Generally results indicated that available Cu used for comparison was lower than the studied parameters except (0.5 ,1.5 , 2.5) g /100g manure application rates and 4 , 8 months incubation periods for Kalar location and 0.5 g /100g manure application rate and 4 months incubation period in addition to (0.5 , 1.5) g /100g manure application rates and 8 months incubation periods for Arbat location as mentioned in (Table 8).The mean value of Cu is much higher than the critical limit of 0.2 ug /g soil. (FAO , 1980 ; Soltanpour , 1985 and Martens and Lindsay , 1990). Its deficiency does not seem to be a problem in these soils. These results might be due to both or each effect of temperature, available water and manure application rates and incubation periods individually as mentioned before.

Table (23) Changes in mean values of available Cu in (ug /g) as functions of temperatures, available water, and manure application rates during incubation periods in studied soils.

Kalar soil									
Time incubated (months)	Temperatures C°			Available Water g /100g			Manure application rates g /100g		
	5	28	45	25	50	100	0.5	1.5	2.5
2	1.108 a	0.560 de	0.513 de	0.801 bc	0.712 cd	0.669 d	0.789 b	0.700 bc	0.692 bcd
4	0.622 cd	0.537 de	0.537 de	0.665 d	0.490 ef	0.540 e	0.509 efg	0.607 cde	0.579 def
8	0.496 e	0.209 f	0.634 cd	0.460 ef	0.465 ef	0.414 f	0.465 fg	0.435 g	0.440 g
12	0.970 b	0.906 b	0.731 c	0.960 a	0.884 ab	0.762 cd	0.752 b	0.785 b	1.069 a
Arbat soil									
2	1.015 ab	0.568 ef	0.603 ef	0.774 bc	0.723 bcd	0.688 cde	0.750 b	0.774 b	0.661 bc
4	0.614 ef	0.537 f	0.813 cd	0.642 cde	0.754 bcd	0.568 de	0.537 c	0.661 bc	0.766 b
8	0.650 def	0.234 g	0.911 bc	0.660 cde	0.644 cde	0.491 e	0.521 c	0.537 c	0.737 b
12	1.069 ab	0.758 cde	1.174 a	1.054 a	1.050 a	0.898 ab	0.820 b	0.855 b	1.326 a

Values followed by the same letters for each treatment in combination with time incubation are not different significantly between themselves according to Duncan's Multiple Range test at ($p \leq 0.01$).

The results in Table (24) show that the highest mean of available Zn values were 1.773 and 1.467 ug /g at 5 C^o and 2 months incubation period for Kalar and Arbat locations respectively , which were superior significantly at $p \leq 0.01$ to all other interactions. On the other hand, the highest mean value of available Zn was 1.312 ug /g at 25 g /100g available water and 2 months incubation period for Kalar location, which were superior significantly to all other interactions except 50 g /100g available water and 2 months incubation period. Whereas, the highest mean value in Arbat location was 1.261 ug /g at 100 g /100g available water and 2 months incubation period, which was superior significantly to all other interactions. Finally, the highest mean value of available Zn was 1.507 ug /g at 2.5 g /100g manure application rate and 2 months incubation period at Kalar location , which was superior significantly at $p \leq 0.01$ to all other interactions . While the highest mean value was 1.407 ug /g in Arbat location at 1.5 g /100g manure application rate and 2 months incubation period, which was superior significantly to all other interactions. The available Zn of Kalar location was 0.350 ug /g but it was 0.390 ug /g for Arbat, generally results indicated that available Zn used for comparison was lower than the studied parameters except 0.5 g /100g manure application rate and 8 months incubation periods for Kalar location and 0.5 g /100g manure application rate and 4 months incubation period in addition to (0.5 , 1.5) g /100g manure application rates and 8 months incubation periods for Arbat location as shown in (Table 8). In general the mean values of available Zn is higher than the critical limit of 0.5 ug /g soil at short period of time 2 - 4 months and there is a decline in mean values of available at last incubation periods particularly at 4 and 8 months.(FAO , 1980 ; Soltanpour , 1985 and Martens and Lindsay , 1990). Its deficiency does not seem to be a problem in these soils. These results might be due to both or each effect of temperature, available water and manure application rates, and incubation periods individually as mentioned before.

Table (24) Changes in mean values of available Zn in (ug /g) as functions of temperatures, available water, and manure application rates during incubation periods in studied soils.

Kalar soil									
Time incubated (months)	Temperatures C°			Available Water g /100g			Manure application rates g /100g		
	5	28	45	25	50	100	0.5	1.5	2.5
2	1.773 a	0.961 b	0.884 b	1.312 a	1.201 ab	1.104 b	0.946 c	1.164 b	1.507 a
4	0.912 b	0.678 c	0.329 d	0.721 c	0.621c	0.578 c	0.426 ef	0.641 d	0.852 c
8	0.609 c	0.332 d	0.304 d	0.546 c	0.338 d	0.360 d	0.310 f	0.430 ef	0.504 de
12	0.934 b	0.686 c	0.381d	0.665 c	0.681 c	0.655 c	0.455 ef	0.635 d	0.911 c
Arbat soil									
2	1.467 a	0.678 c	1.298 b	1.110 b	1.072 b	1.261 a	0.761 c	1.407 a	1.275 b
4	0.682 c	0.532 d	0.252 e	0.549 d	0.508 de	0.409 ef	0.280 f	0.538 de	0.648 cd
8	0.481 d	0.280 e	0.275 e	0.363 f	0.383 ef	0.289 f	0.143 g	0.352 f	0.541 de
12	0.746 c	0.664 c	0.484 d	0.688 c	0.606 cd	0.601 cd	0.486 e	0.669 cd	0.738 c

Values followed by the same letters for each treatment in combination with time incubation are not different significantly between themselves according to Duncan's Multiple Range test at ($p \leq 0.01$).

4.5 Solubility equilibria of iron, manganese, copper, and zinc in soil.

Concentrations of soluble ions (Fe, Mn, Cu and Zn) are too low in soil solution so the chelation method has been used in order to determine their activities (Lindsay, 1979). Effects of various parameters such as different temperatures, available water, and manure application rate and incubation period on the solubility equilibria of the above mentioned ions have been studied. Log of each Fe, Mn, Cu and Zn against pH or pe + pH incase of redox potential was used in order to determine the solubility of those cations in the soil solution. Effects of those parameters on the solubility of iron are shown in Fig. (39 ,40,41,42,43,44,45,and 46) and appendices (1 , 2 , 3 and 4) .The points are located below the fresh precipitate { $Fe_3(OH)_8$ }line so the solubility of Fe is undersaturated with respect to fresh precipitate at all mentioned parameters at Kalar and Arbat location. Lindsay, (1979) indicated that freshly precipitated $Fe_3(OH)_8$ (amorp) has been hypothesized to form in reduced soils. Iron was associated with SOM, which formed chelated iron complexes, and Fe would release into soil solution because of SOM degradation, iron is mobile and at oxidation condition, it will precipitate as Fe oxyhydroxide. It has well known that OC acts as the inhibitor for the crystallization of Fe oxyhydroxide (Cornell and Schwertmann, 1979). At the equilibrium redox corresponding to 0.2 atm oxygen (g) , the activities of Fe^{+2} and Fe^{+3} become equal at pH (7.57) so the pe + pH of soils is generally lower than(20.61), the activity of Fe^{+2} in soils is most often greater than that of Fe^{+3} . The electron activity in soils controls the ratio of Fe^{+3} to Fe^{+2} in solution , when pe =13.04 , the ratio of Fe^{+2} / Fe^{+3} is unity (Lidnsay , 1979). On the other hand , Alexander, (1977) reported that microorganisms play an important roles in the stability of iron ion in soil , it had been found that some types of bacteria will affect on Fe^{+2} oxidation to Fe^{+3} and form precipitated of $Fe(OH)_3$,while other types of bacteria act on oxidation of OM and release Fe and precipitate as pyrite.

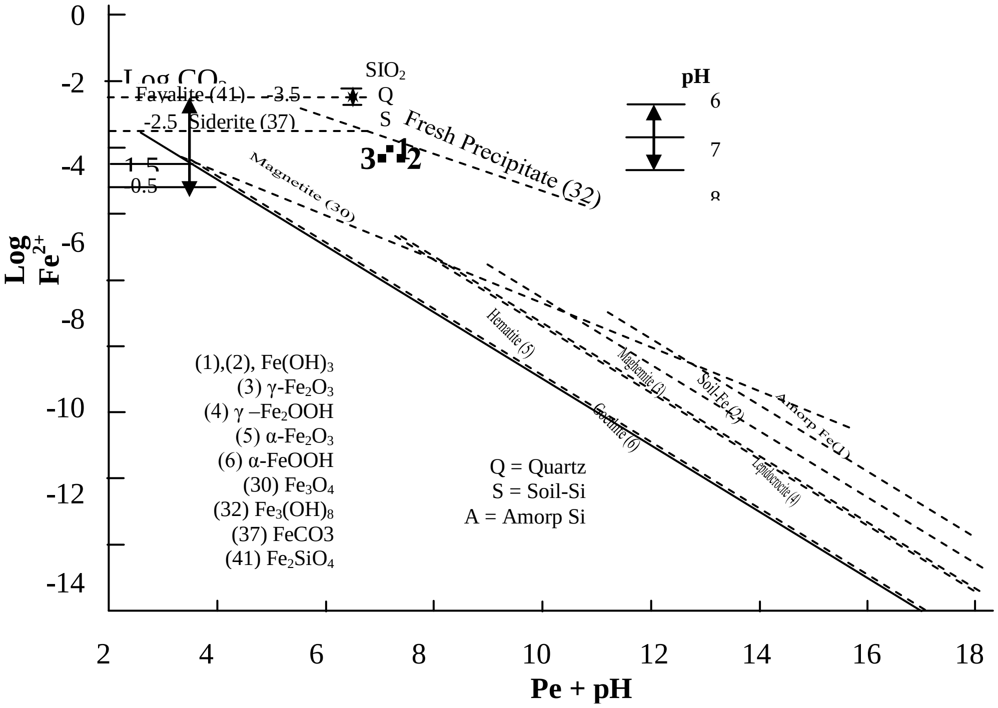


Fig. (39) The effect of different temperatures on the solubility of iron at Kalar location. Points (1, 2 and 3 represent 5, 28 and 45 C^{o} respectively).

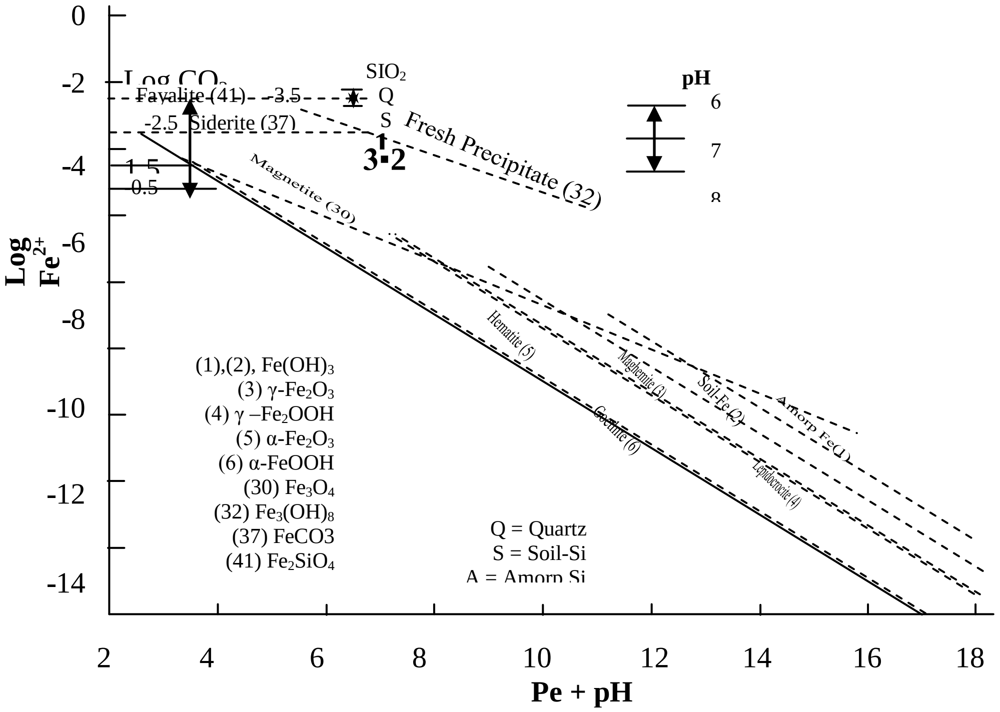


Fig.(40) The effect of different manure application rates on the solubility of iron at Kalar location. Points (1, 2 and 3 represent (0.5, 1.5 and 2.5) g /100g manure application rates respectively).

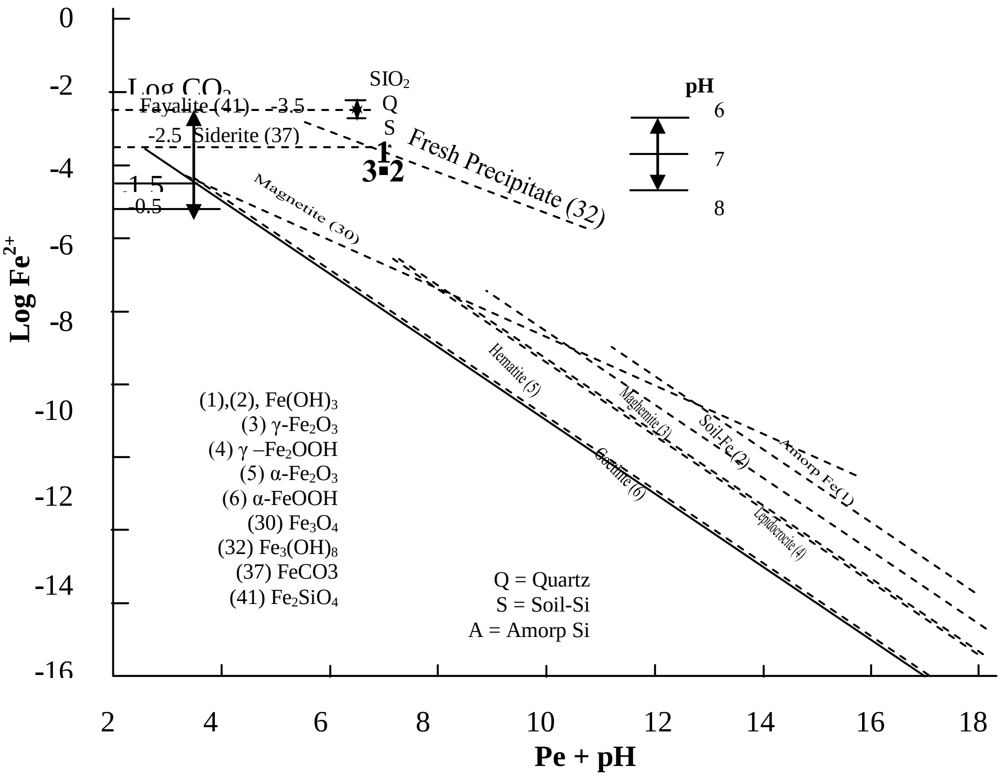


Fig.(41) The effect of different available water on the solubility of iron at Kalar location. Points (1, 2 and 3 represent (25, 50 and 100) g /100g available water respectively).

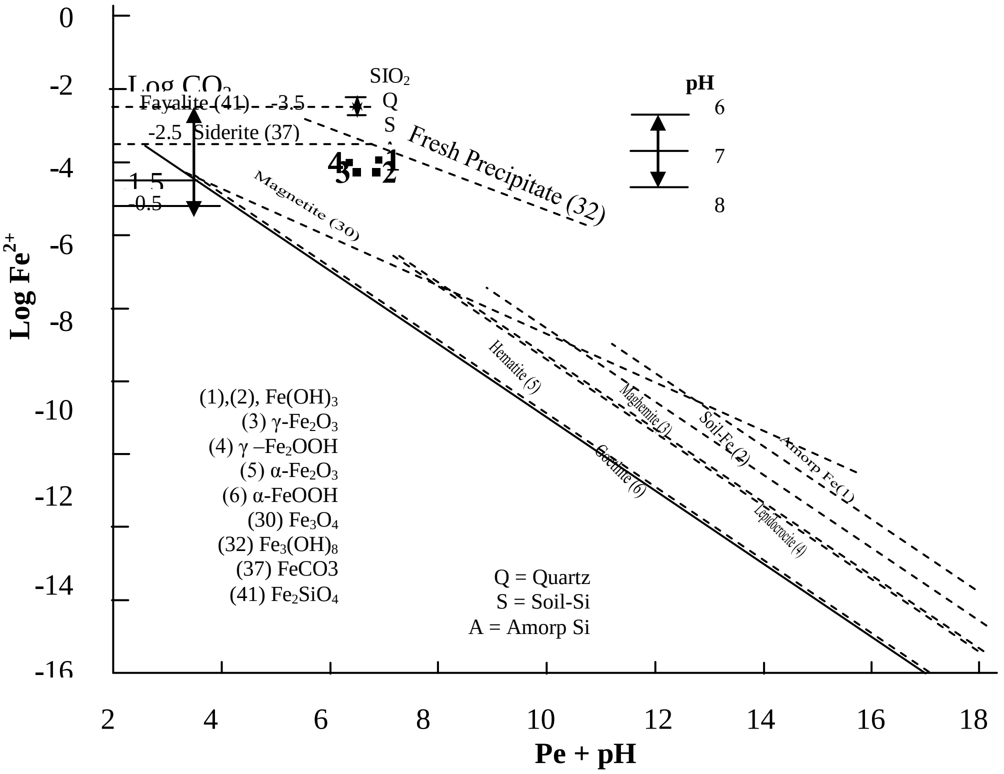


Fig.(42) The effect of different incubation periods on the solubility of iron at Kalar location. Points (1, 2, 3 and 4 represent 2, 4, 8 and 12 months incubation periods respectively).

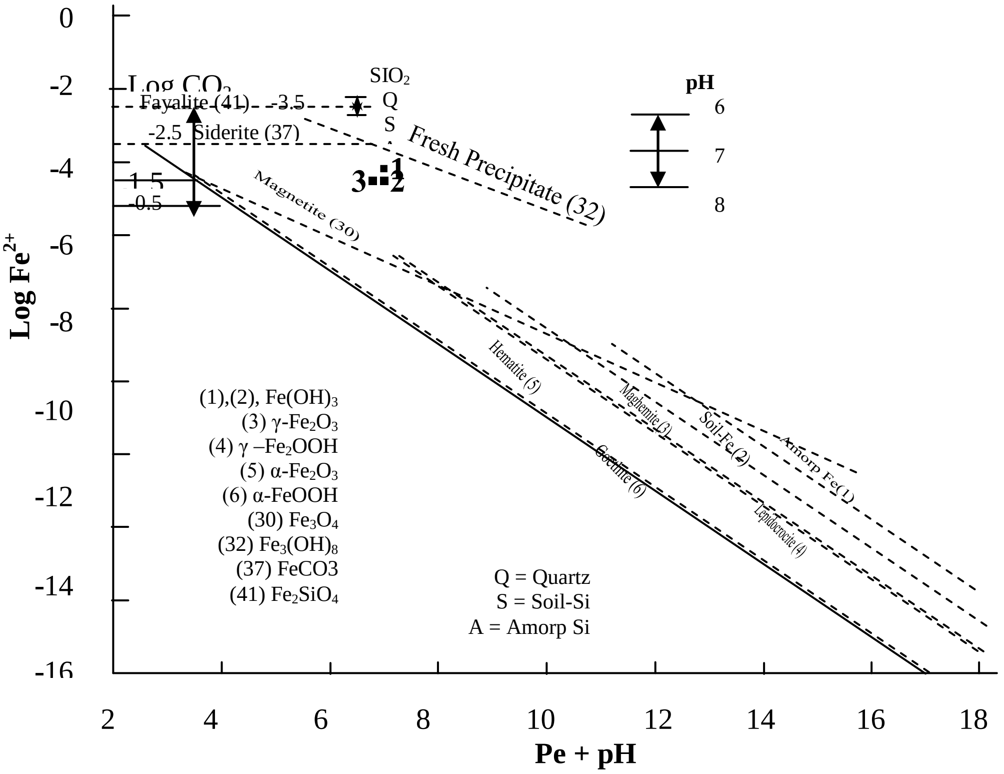


Fig. (43) The effect of different temperatures on the solubility of iron at Arbat location. Points (1, 2 and 3 represent 5, 28 and 45 C^{o} respectively).

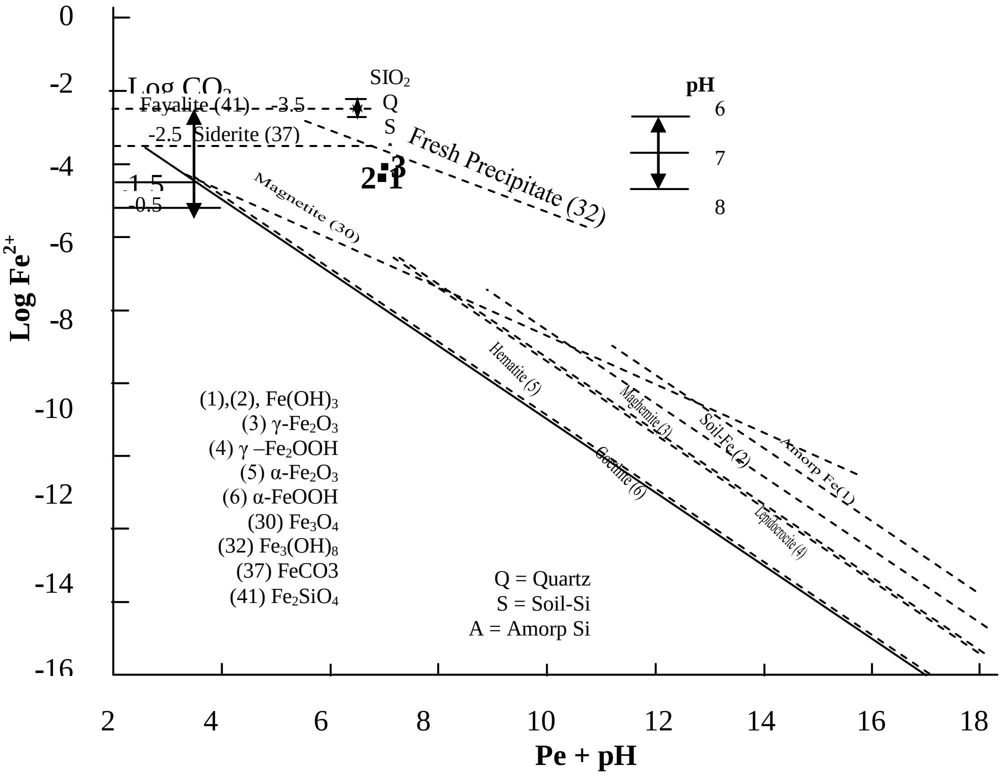


Fig. (44) The effect of different manure application rates on the solubility of iron at Arbat location. Points (1, 2 and 3 represent (0.5, 1.5 and 2.5) g /100g manure application rates respectively).

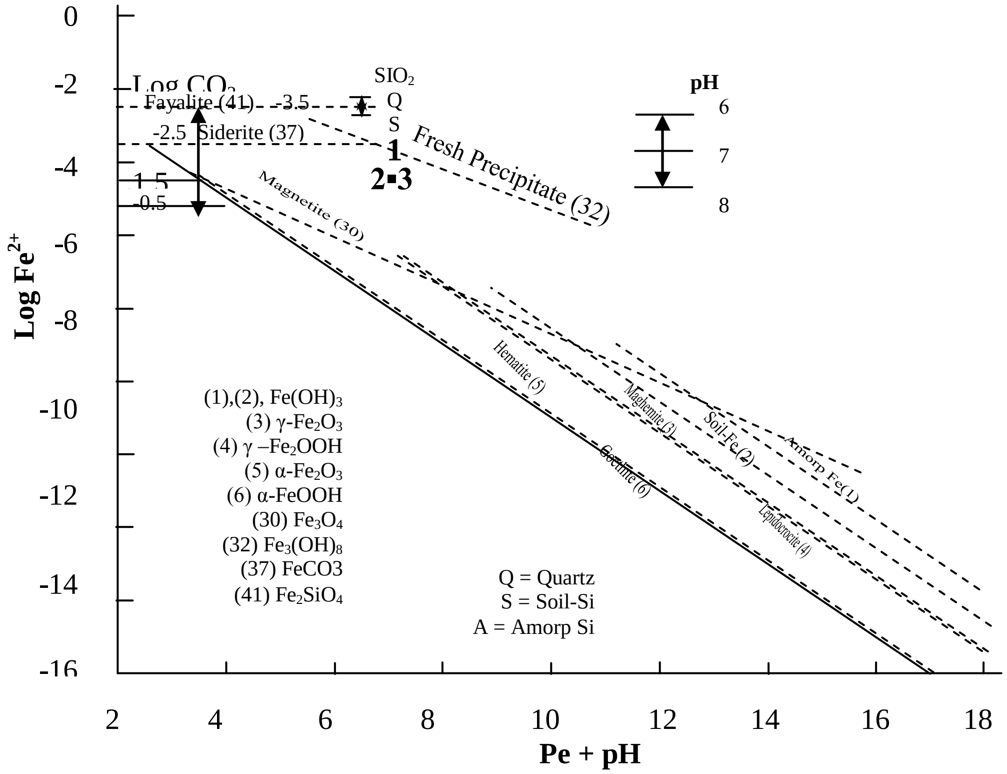


Fig. (45) The effect of different available water on the solubility of iron at Arbat location. Points (1, 2 and 3 represent (25, 50 and 100) g /100g available water respectively).

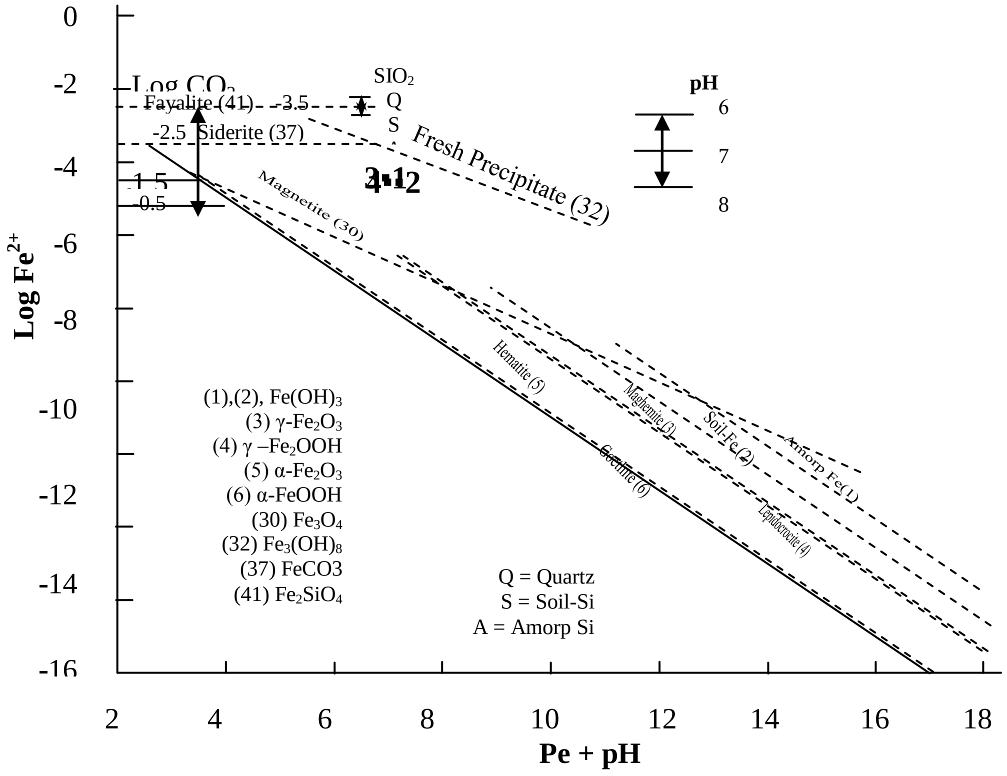


Fig. (46) The effect of different incubation periods on the availability of iron at Arbat location. Points (1, 2, 3 and 4 represent 2, 4, 8 and 12 months incubation periods respectively).

The results presented in Fig (47,48,49 ,50 ,51,52,53,54) and appendices (1, 2 , 3 and 4) show the effects of different temperatures, available water, manure application rates and incubation periods on the solubility of manganese. The points are located below rhodochrosite (MnCO3$_{)}$ mineral. So Mn solubility is considered undersaturated with respect to $MnCO_3$ at all mentioned parameters, at Kalar location. While at Arbat location, the points are located on or below the $MnCO_3$ line,. So the solubility of Mn is undersaturated or in equilibrium with respect to $MnCO_3$ for all parameters except at (25, 50 and 100) g /100g available water, which is between rhodochrosite (MnCO3$_{)}$ and γ – manganite (MnOOH) minerals. It is very difficult to determine the mineral, which controls the solubility of Mn at pH more than 7.0 (Moore and Patrich, 1989).Differences in solubility of Mn between Kalar and Arbat locations may be due to variations in soil texture (Shuman, 1985 as stated by Al – Tamimi, 1997) or it can be attributed to $CaCO_3$. The low solubility of Mn in kalar as compared to Arbat can be described by what has been mentioned by McBride, (1979) who found that strong chemical adsorption of Mn^{+2} on the surface of $CaCO_3$ with increasing carbonate content which may lead to reduce availability of Mn. The addition of 1.5 and 2.5 g /100g of manure application rates decrease the solubility of Mn (undersaturated with respect to $MnCO_3$). Manganese behaves the same manner as Fe^{+2} in depending on redox potential since it is oxidized to Mn^{+3} and Mn^{+4} leading to the decrease of both ions at high soil pH as far as plant growth is concer (Mascagin and Cox , 1985) .

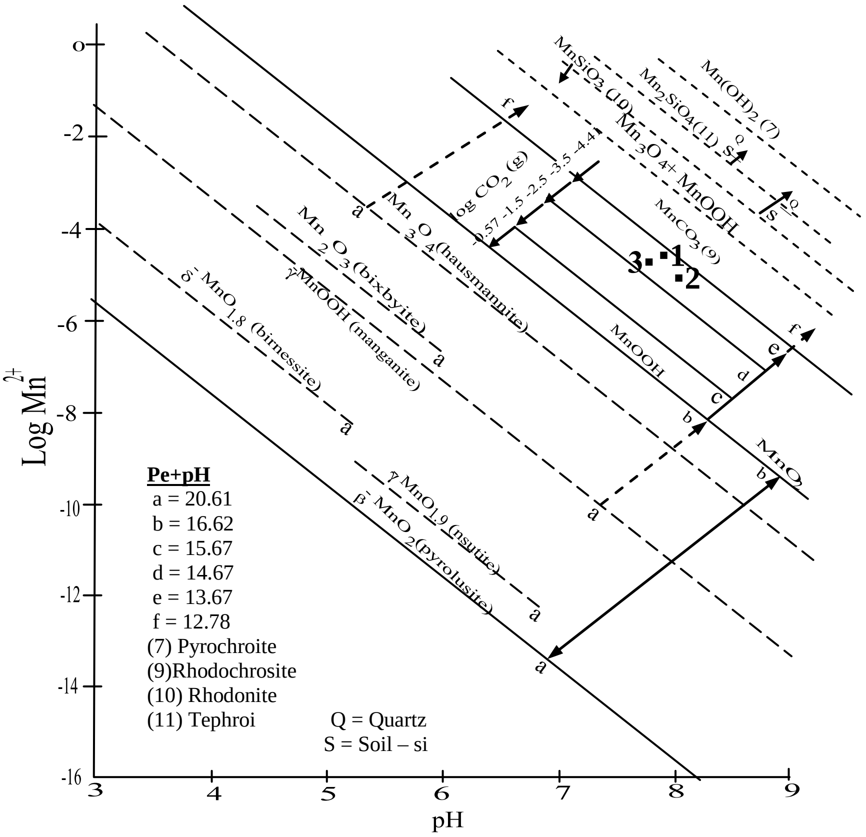


Fig. (47)The effect of different temperatures on the solubility of manganese at Kalar location .Points (1 , 2 and 3 represent 5 , 28 and 45 C^o respectively .)

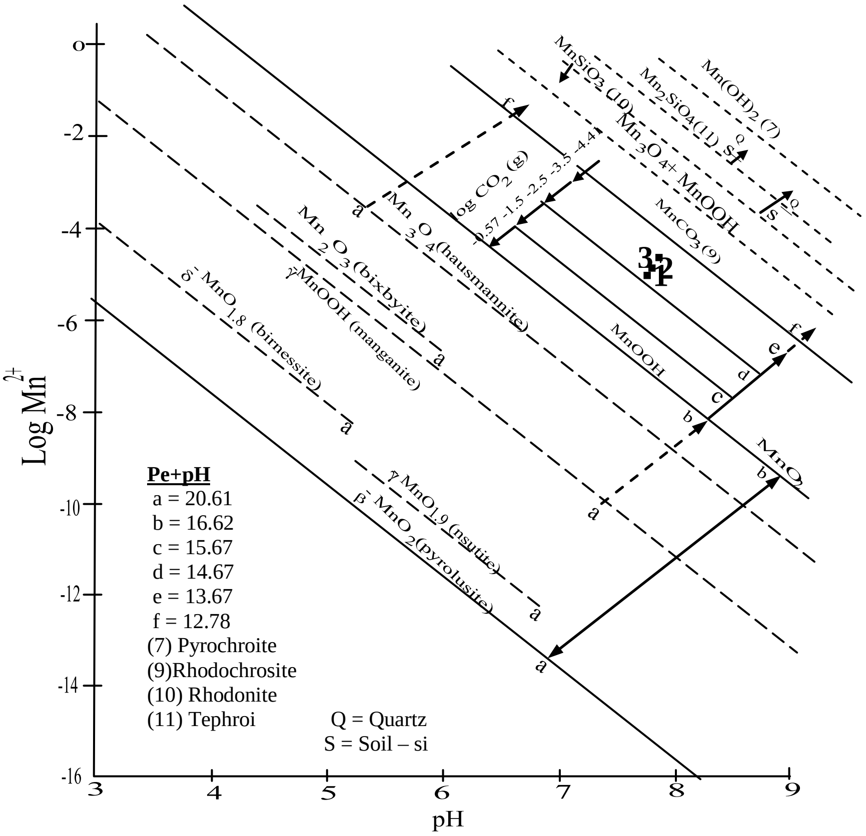


Fig. (48)The effect of different manure application rates on the solubility of manganese at Kalar location. Points (1 , 2 and 3 represent (0.5 , 1.5 and 2.5) g /100g manure application rates respectively).

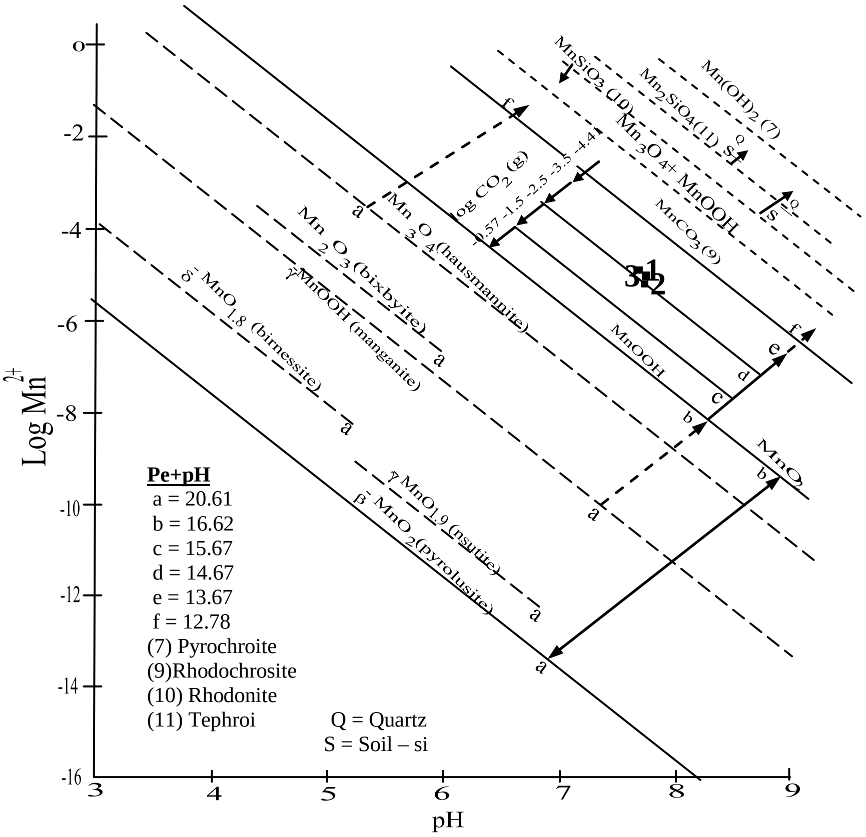


Fig. (49)The effect of different available water on the solubility of manganese at Kalar location . Points (1 , 2 and 3 represent 25 , 50 and 100 g / 100g available water respectively .)

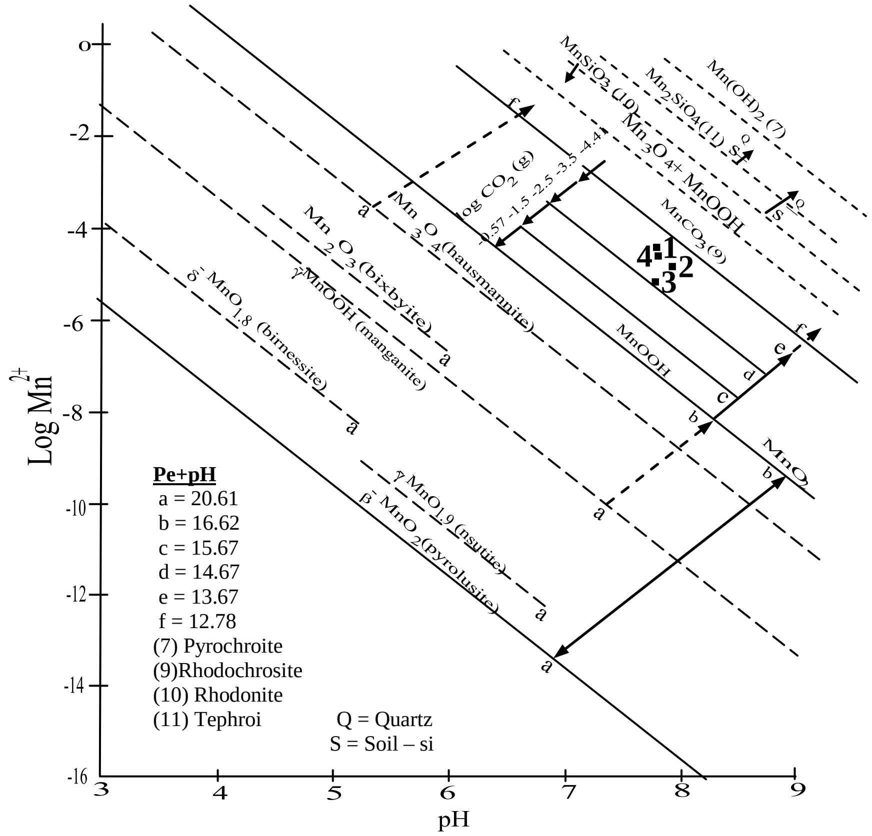


Fig. (50)The effect of different incubation periods on the solubility of manganese at Kalar location .Points (1 , 2 ,3 and 4 represent 2 ,4 , 8 and 12 months respectively .)

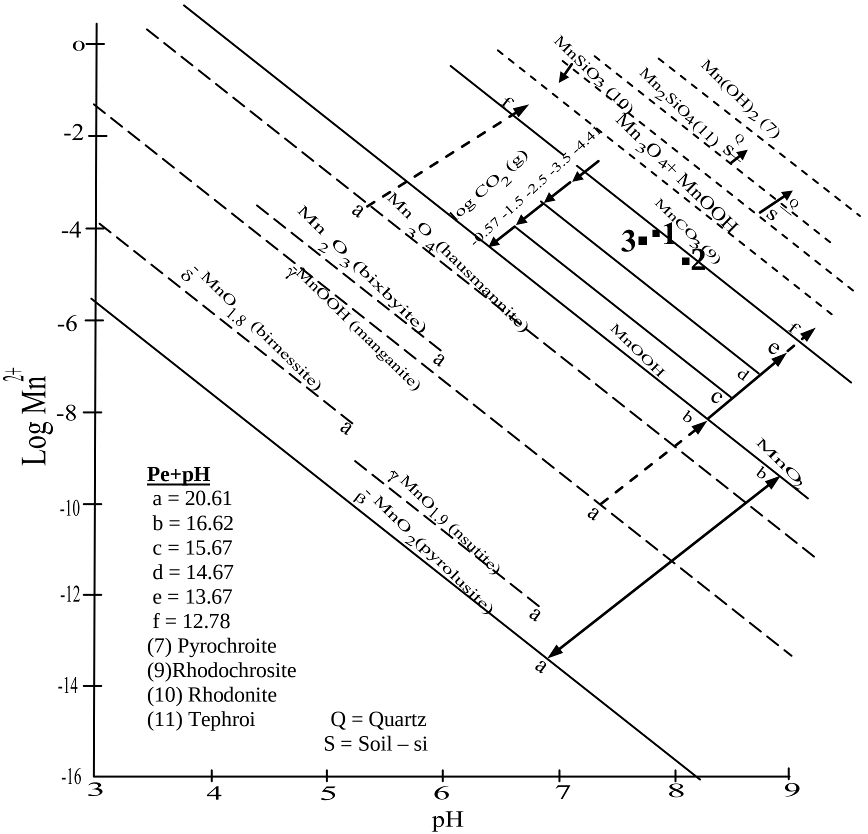


Fig. (51)The effect of different temperatures on the solubility of manganese at Arbat location . Points (1 , 2 and 3 represent 5 , 28 and 45 C^{o} respectively .)

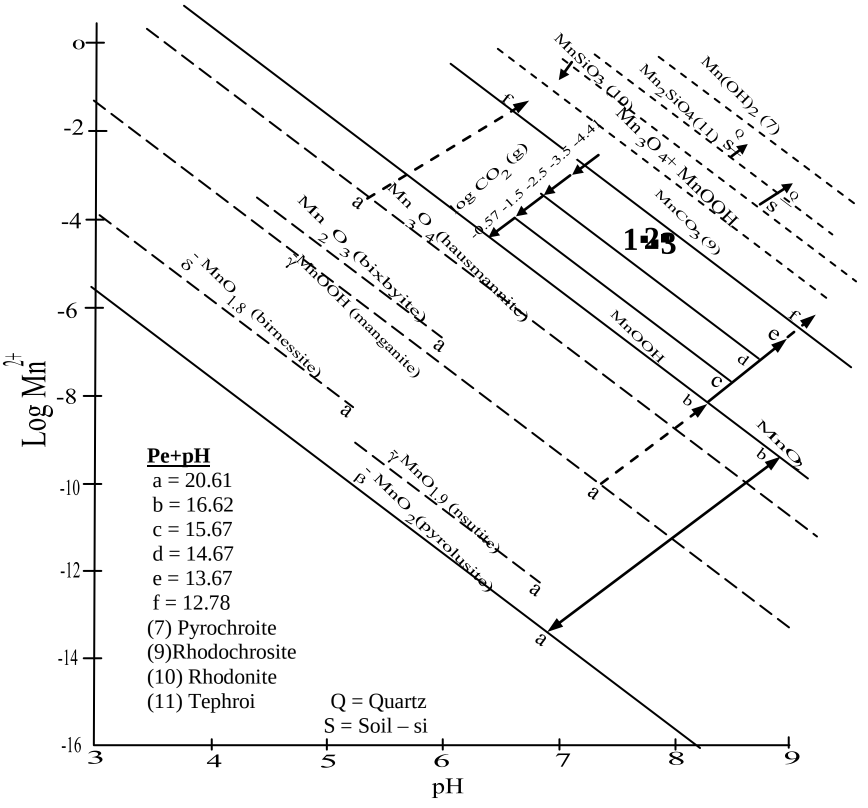


Fig. (52)The effect of different manure application rates on the solubility of manganese at Arbat location. Points (1 , 2 and 3 represent (0.5 , 1.5 and 2.5) g /100g manure application rates respectively).

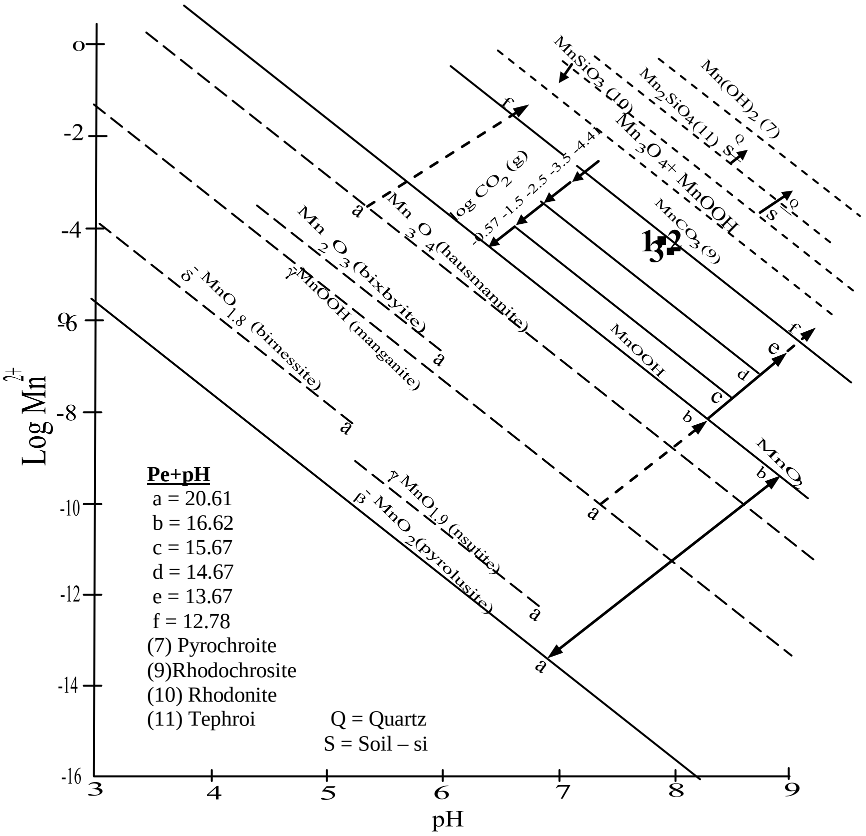


Fig. (53)The effect of different available water on the solubility of manganese at Arbat location .Points (1 , 2 and 3 represent 25 , 50 and 100 g / 100g available water respectively .)

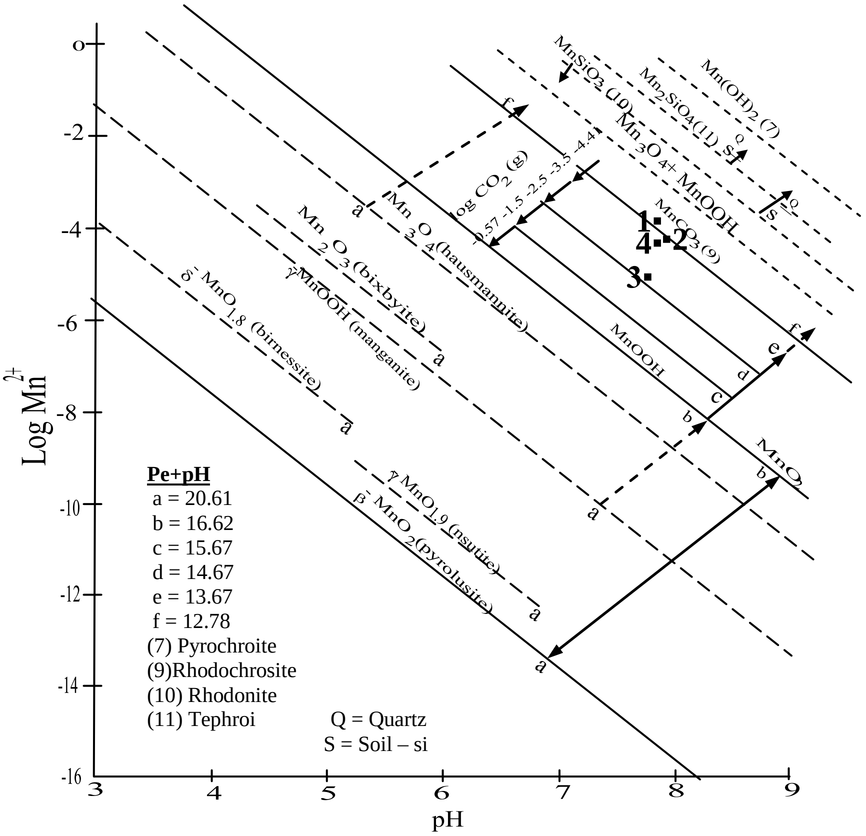


Fig. (54)The effect of different incubation periods on the solubility of manganese at Arbat location .Points (1 , 2 ,3 and 4 represent 2 ,4 , 8 and 12 months respectively .)

The results presented in Fig. (55,56,57,58,59,60,61,62) and appendices (1 , 2 , 3 , and 4) show the effects of different temperatures, available water, manure application rates and incubation periods on the solubility of copper. The points are located above the soil –Cu, so the solubility of Cu is supersaturated with respect to soil Cu at all mentioned parameters for Kalar and Arbat locations. Below pe + pH of 11.53 where magnetite controls iron solubility cupric ferrite becomes even more soluble. The solubility of copper is dependent on redox conditions, so Cu^{+1} and Cu^{+2} activities are equal at pe = 2.62 and their ratio changes 10 – fold for each unit change in pe (Lindsay, 1979). Generally, the availability of Cu^{+2} in soil is dependent on several factors including soil pH, texture and interactions with other nutrients. Increased soil pH reduces the availability of Cu^{+2} through increasing adsorption of Cu at cation exchange sites. While, Al-tamemi, (1997) indicated that Cu solubility was controlled by malachite mineral at calcareous soil. On the other hand, copper hydroxide and carbonate has no role on the solubility of Cu in soil in spite of malachite mineral formed from mixing both chemicals component (carbonate and hydroxides) under alkaline conditions and high $CaCO_3$ content (McBride and Boldin, 1984).

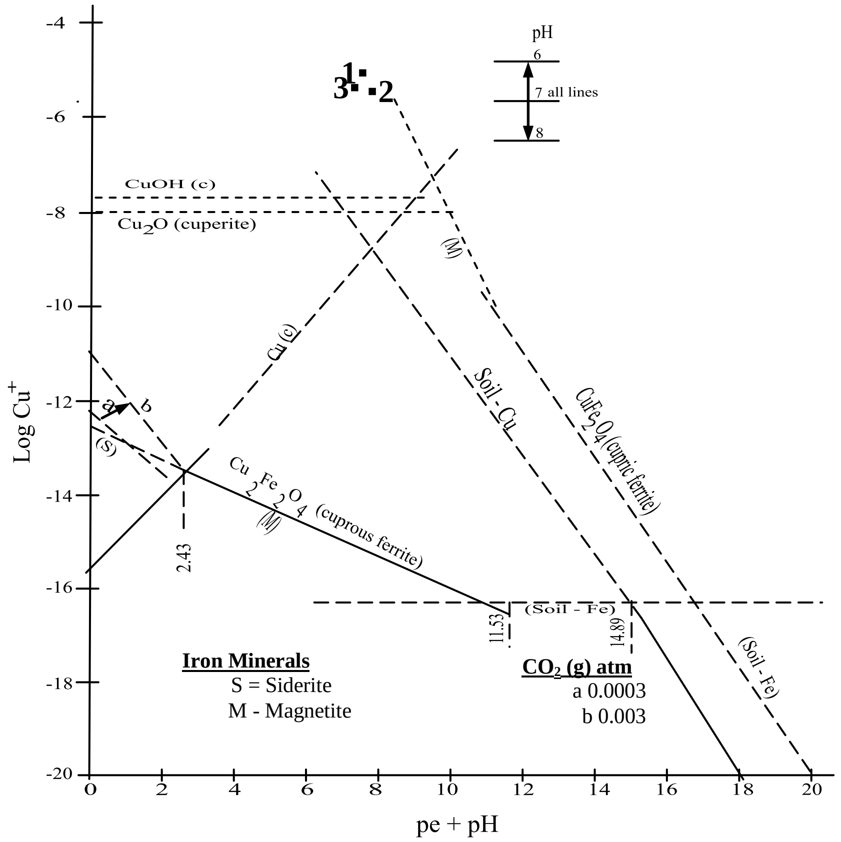


Fig. (55)The effect of different temperatures on the solubility of copper at Kalar location. Points (1 , 2 and 3 represent 5 , 28 and 45 C^o respectively .)

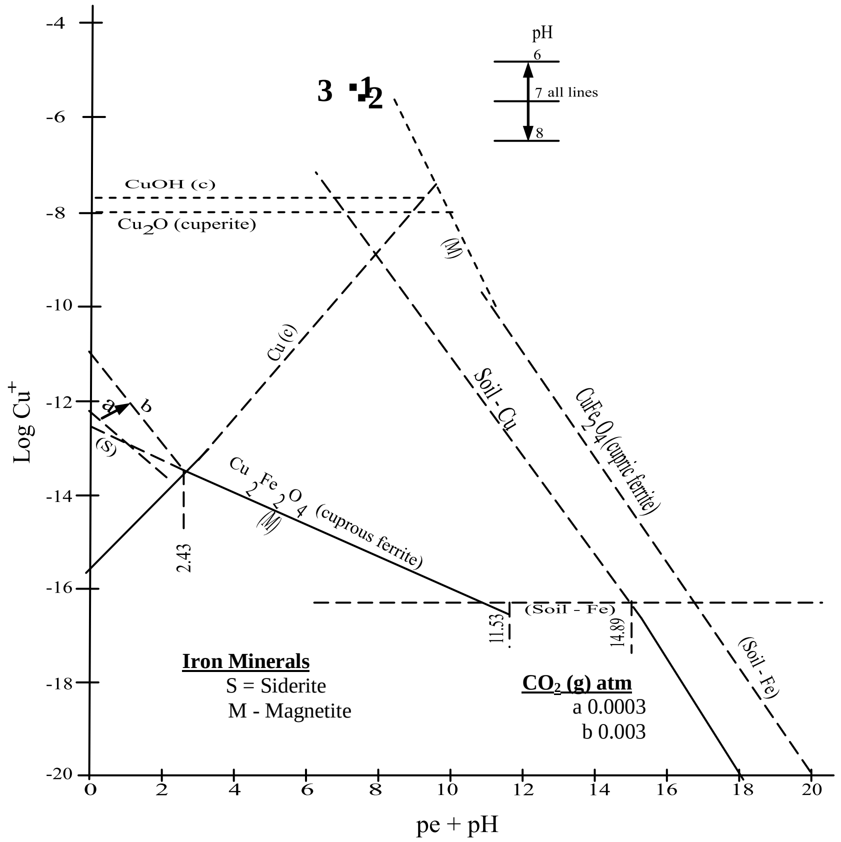


Fig. (56)The effect of different manure application rates on the solubility of copper at Kalar location .Points (1 , 2 and 3 represent (0.5 , 1.5, 2.5) g /100g manure application rates respectively)

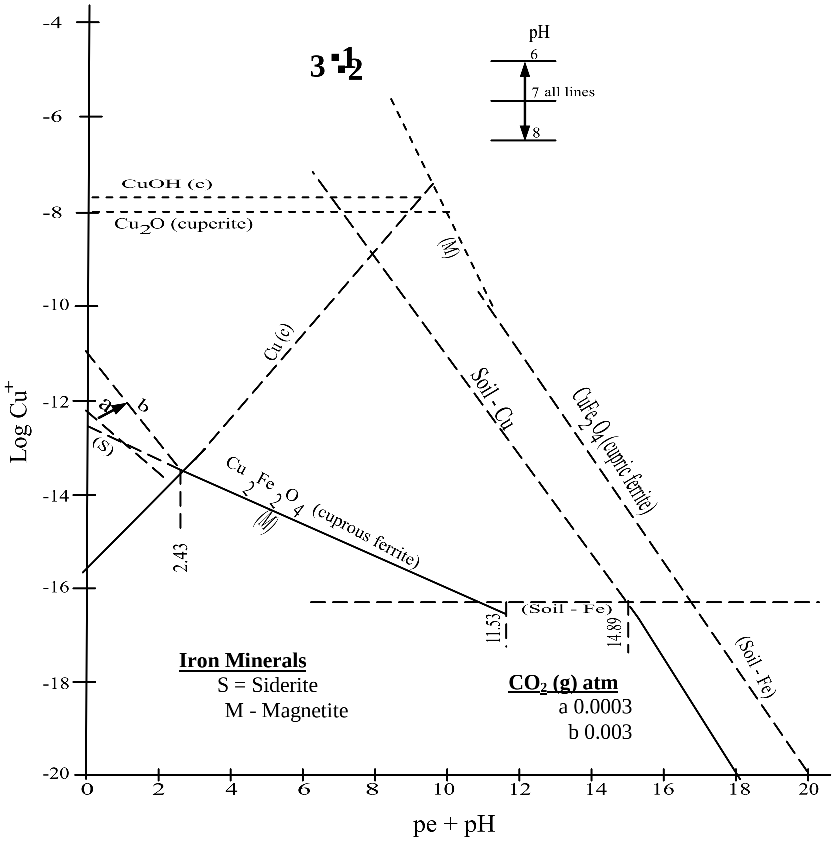


Fig. (57)The effect of different available water on the solubility of copper at Kalar location. Points (1 , 2 and 3 represent (25 , 50 and 100) g /100g available water respectively .)

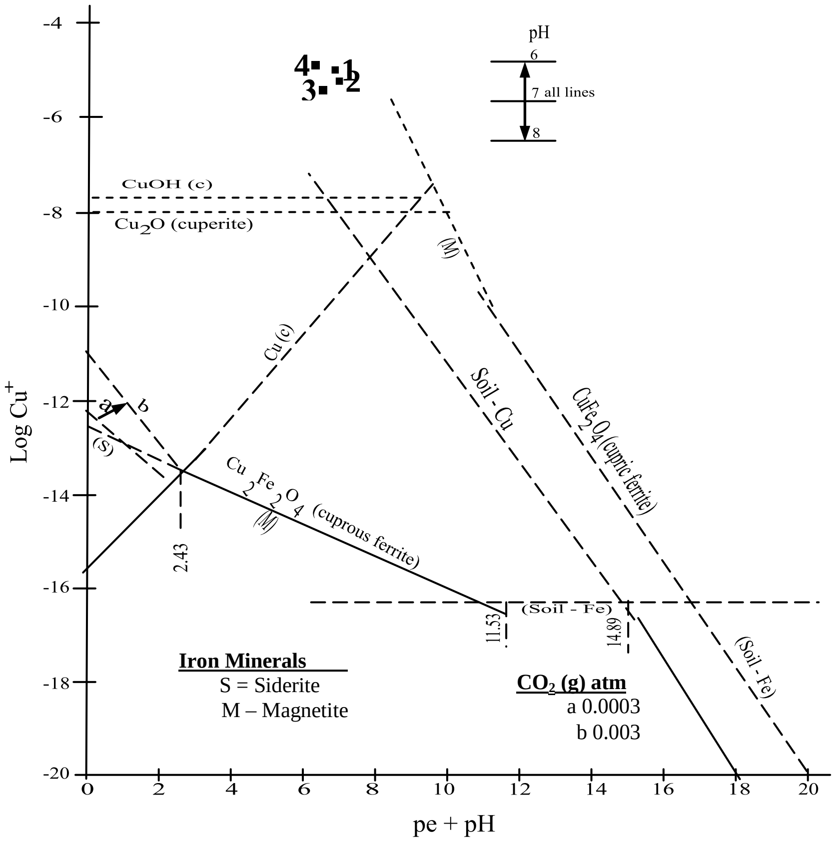


Fig. (58): The effect of different incubation periods on the availability of copper at Kalar location. Points (1 , 2 , 3 and 4 represent 2 , 4 , 8 and 12 months incubation periods respectively)

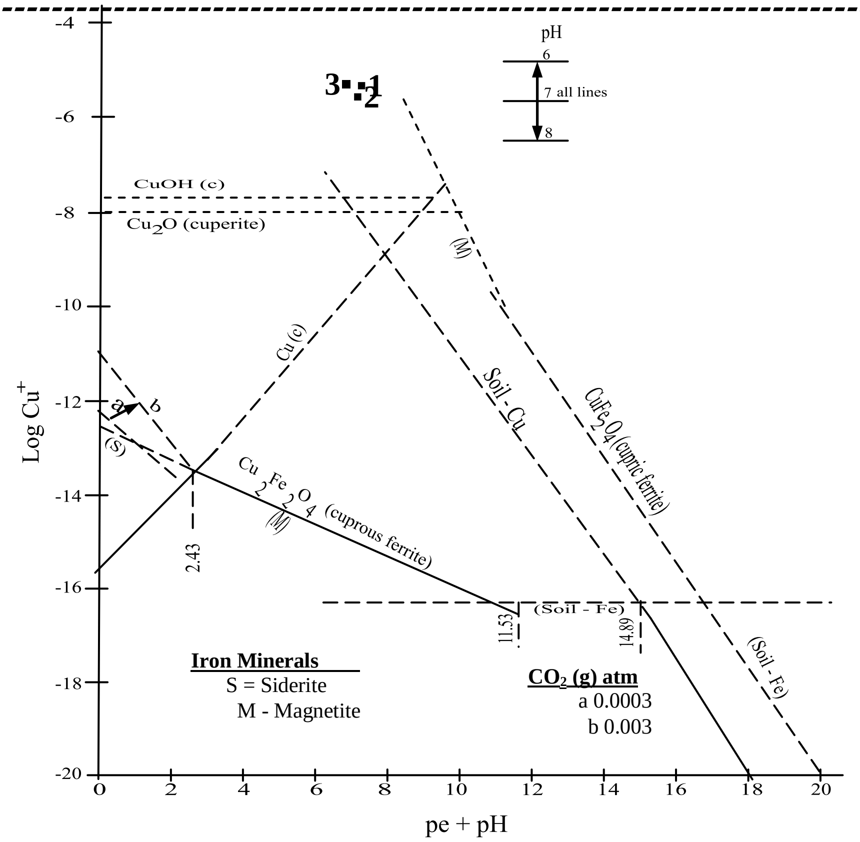


Fig. (59)The effect of different temperatures on the solubility of copper at Arbat location. Points (1 , 2 and 3 represent 5 , 28 and 45 C^{o} respectively .)

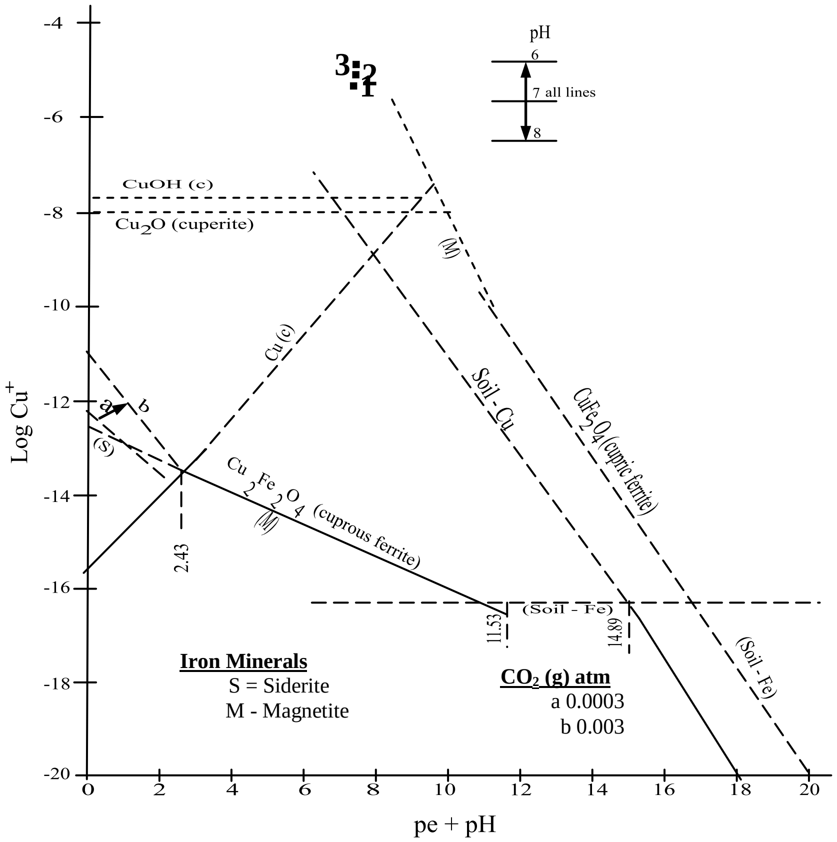


Fig. (60)The effect of different manure application rates on the solubility of copper at Arbat location .Points (1 , 2 and 3 represent (0.5, 1.5, 2.5) g /100g manure application rates respectively)

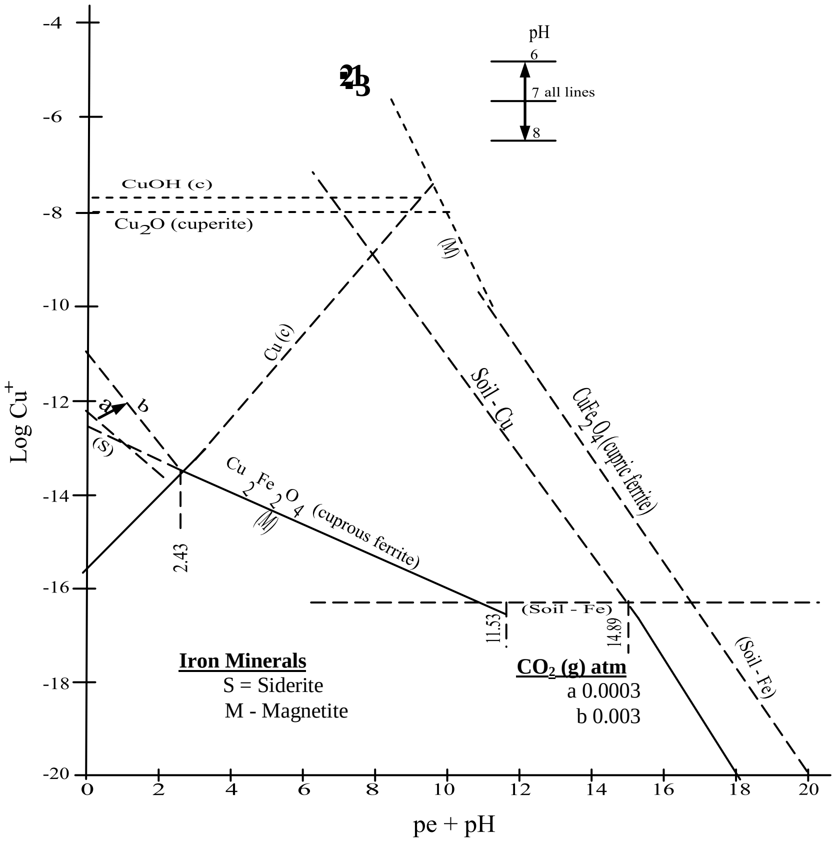


Fig. (61)The effect of different available water on the solubility of copper at Arbat location. Points (1 , 2 and 3 represent (25 , 50 and 100) g /100g available water respectively.)

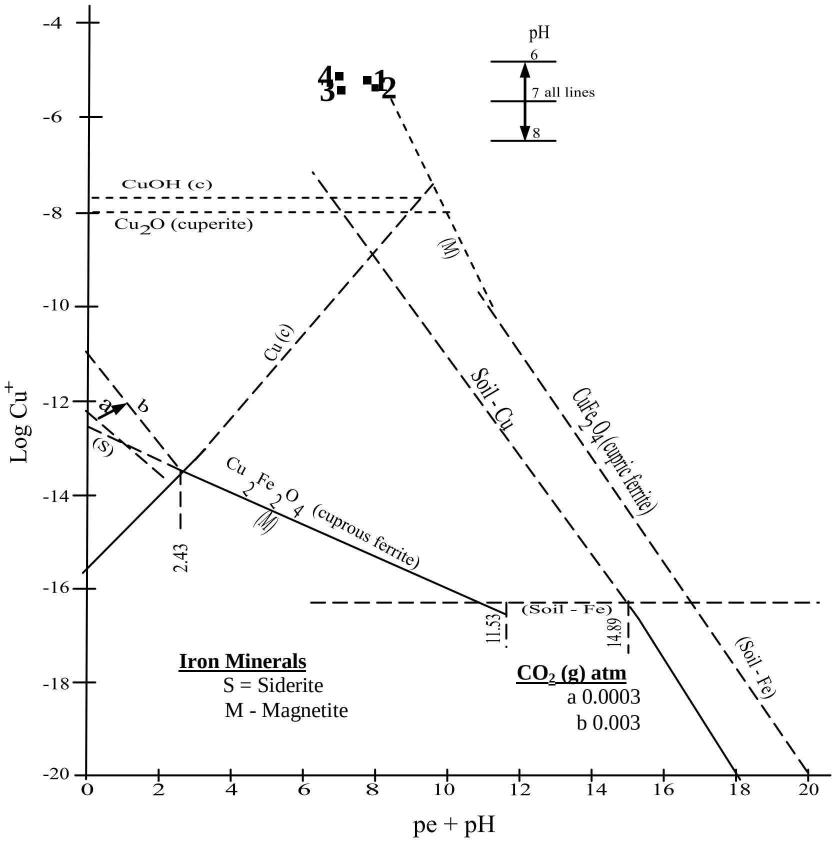


Fig. (62) The effect of different incubation periods on the availability of copper at Arbat location. Points (1 , 2 , 3 and 4 represent 2 , 4 , 8 and 12 months incubation periods respectively.)

Effects of different temperatures, available water, manure application rates, and incubation periods on the solubility of zinc are shown in Fig. (63,64 ,65 ,66, 67 ,68,69,and 70) and appendices (1 , 2 , 3 and 4).The points are located between ZnO (zincite) and Zn_2SiO_4 (willemite) minerals ,therefore the solubility of Zn may be controlled by both minerals. Generally, solubility of Zn decrease slightly with increased temperature between 5 to 45 C^o in both locations. Increasing of manure application increases the solubility of Zn in both locations, it can be attributed to formation of soluble chelated Zn compounds in soil solution . On the other hand, there is no obvious effects of moisture content on solubility of Zn in studied soils , and the fluctuation in solubility can be attributed to other factors, such as $CaCO_3$ content and redox potential , extended period of time for 12 months decrease Zn solubility in studied soils , and the highest values of solubility were obtained at 2 months of incubation periods. The results are in agreement with those reported by Singh et al., (1983) and Al - tamimi, (1997) who found that precipitation or adsorption of willemite may occur between pH 6 and 7.9 and that adsorption may occur below pH 6. Zn solubility changes, as a function of pH, at low pH adsorption appears to control the solubility, while at high pH precipitation will be the limiting factor for the solubility. Gupta et al., (1987) suggest that at high pH, precipitation reactions control Zn solubility, whereas at neutral to acidic pH, specifically adsorbed Zn may control Zn solubility. On the other hand, Kathryn et al., (2002) concluded that OC might play a role in the control of free Zn solubility according to the regression equation:

$$\text{Log} (Zn^{+2}) = 1.63\ OC - 11.9 \quad \text{---} \ 19$$

In neutral to alkaline soils, $Zn (OH)^+$ is a dominant solution species of Zn that may adsorb to soil and replace one H^+:

$$Zn (OH)^+ + LH \longleftrightarrow Zn (OH) L + H^+ \quad \text{--} \ 20$$

Where L is an organic ligand in the soil capable of complexing zinc.

The minerals controlling the solubility of Zn are $Zn (OH)_{2 (C)}$ and $Zn_5(OH)_6(CO_3)_2$ when the soil pH is high and rich in $CaCO_3$ (Misra and Tiwari ,1966) . On the other hand, Leggett, (1978) and Awad, (1984) found that Zn in corporate with $Si (OH)_4$ in calcareous soil forming insoluble precipitated compounds which control the solubility of Zn.

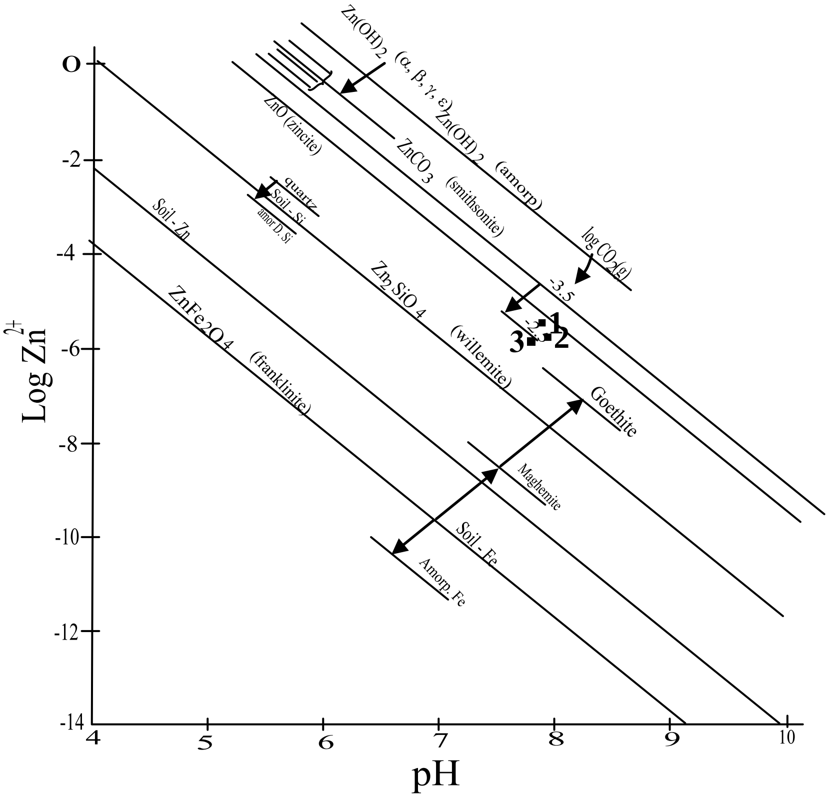


Fig. (63)The effect of different temperatures on the solubility of zinc at Kalar location. Points(1 , 2 and 3 represent 5 , 28 and 45 C^o respectively .)

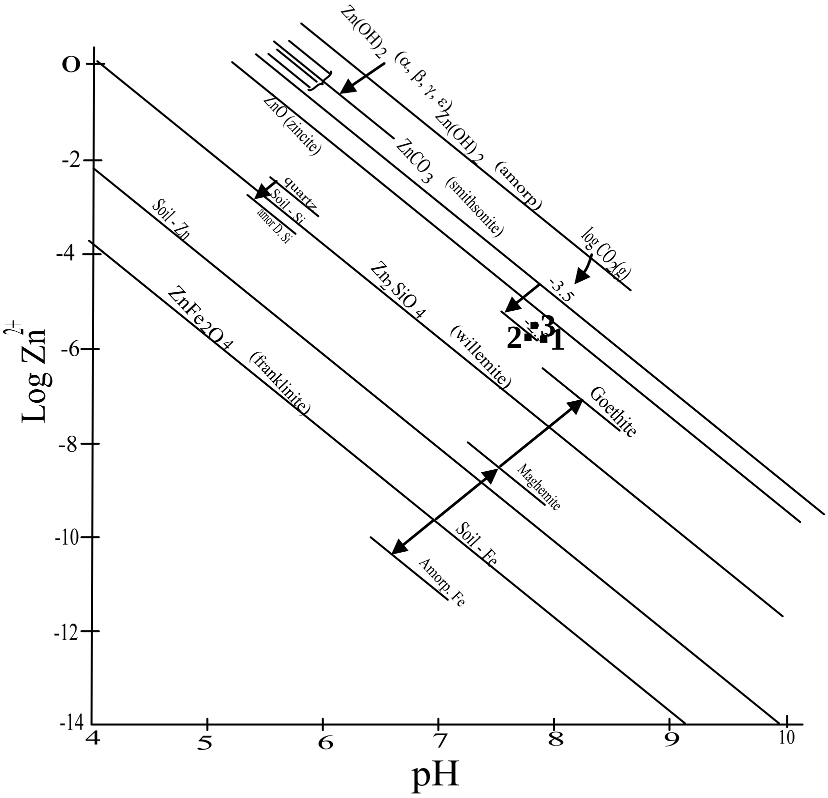


Fig. (64)The effect of different manure application rates on the solubility of zinc at Kalar location .Points (1 , 2 and 3 represent (0.5 , 1.5 and 2.5) g /100g manure application rates respectively).

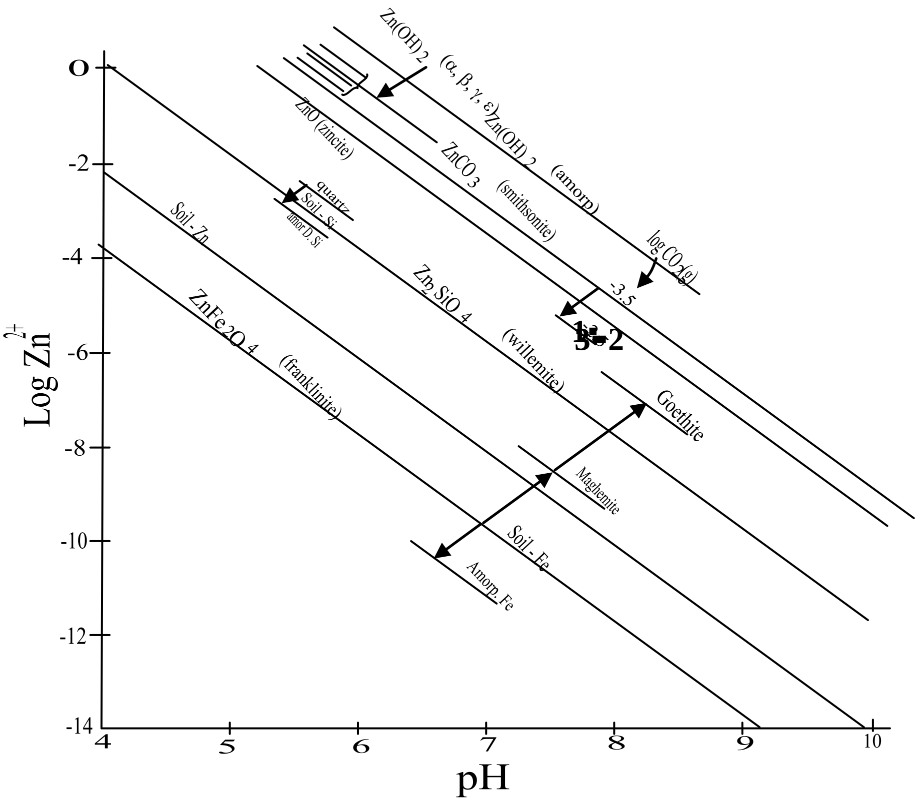


Fig. (65)The effect of different available water on the solubility of zinc at Kalar location. Points(1 , 2 and 3 represent 25,50 and 100) g /100g available water respectively .)

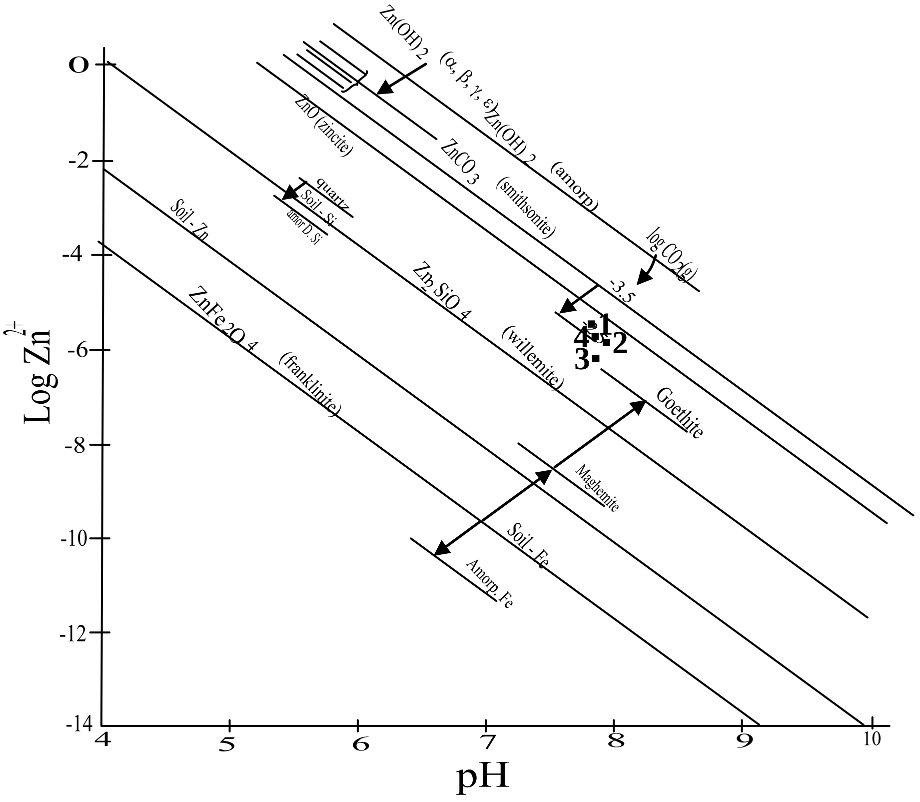


Fig. (66) The effect of different incubation periods on the availability of Zinc at Kalar location . Points (1 , 2 and 3 and 4 represent 2 , 4 , 8 and 12 months incubation periods .respectively)

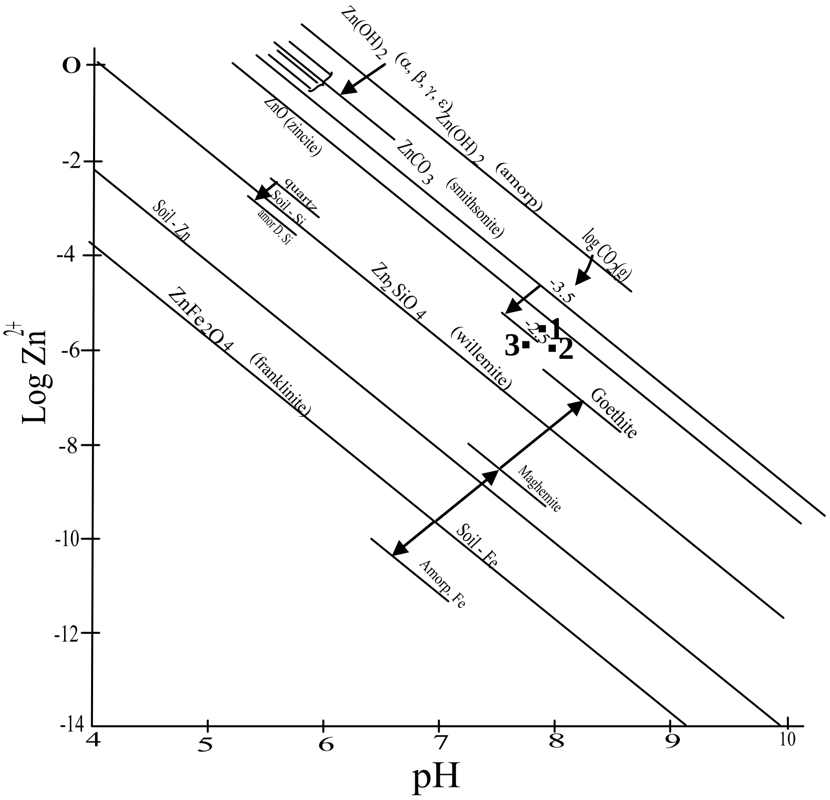


Fig. (67)The effect of different temperatures on the solubility of zinc at Arbat location. Points(1 , 2 and 3 represent 5 , 28 and 45 C^o respectively .)

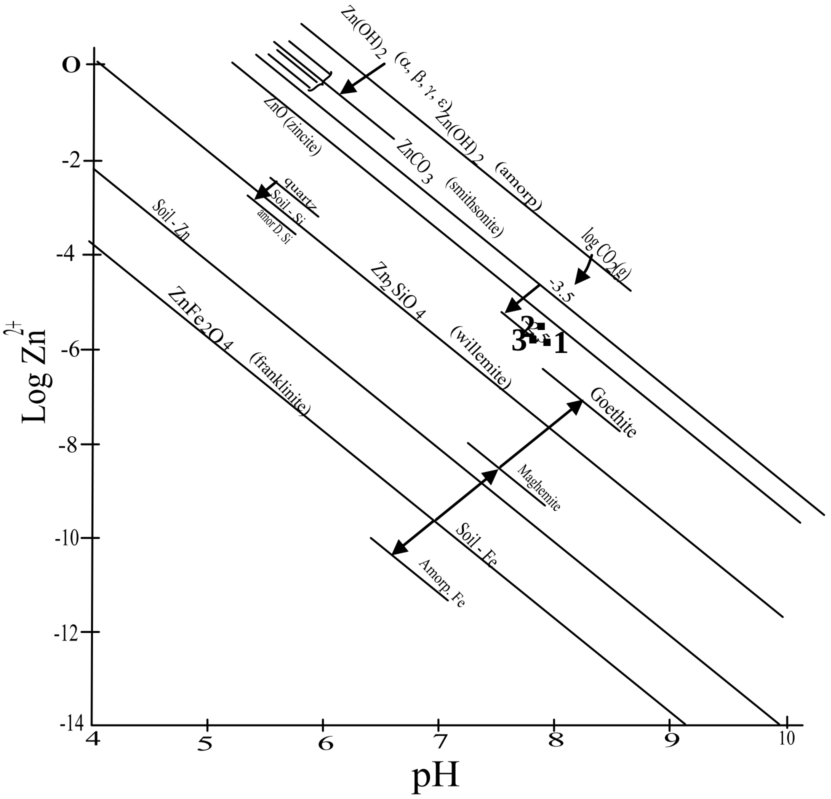


Fig. (68)The effect of different manure application rates on the solubility of zinc at Arbat location .Points (1 , 2 and 3 represent 0.5 , 1.5 and 2.5) g /100g manure application rate. respectively).

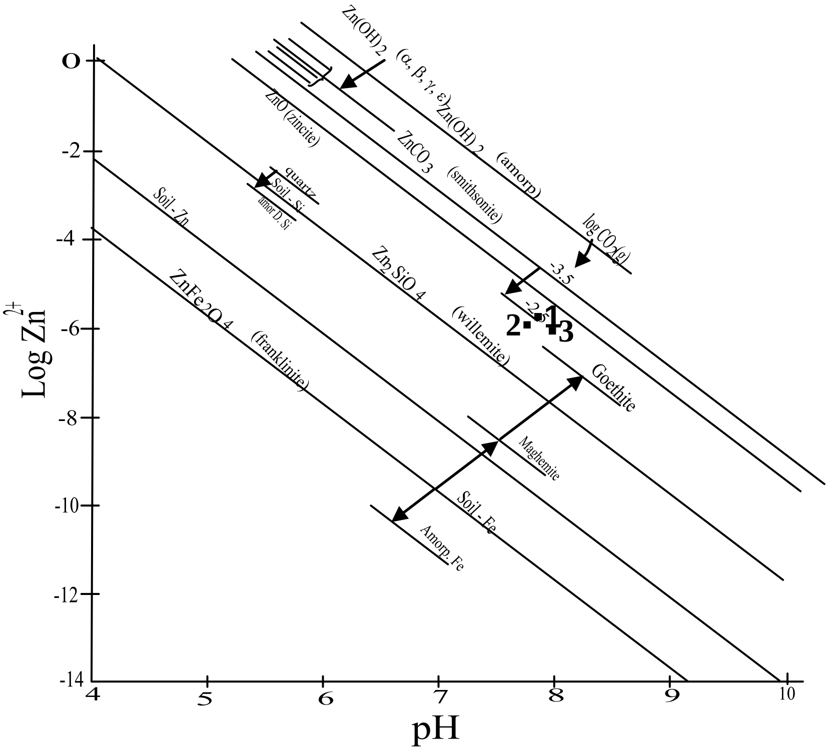


Fig. (69)The effect of different available water on the solubility of Zinc at Arbat location. Points (1 , 2 and 3 represent 25 , 50 and 100 g /100g available water .respectively .)

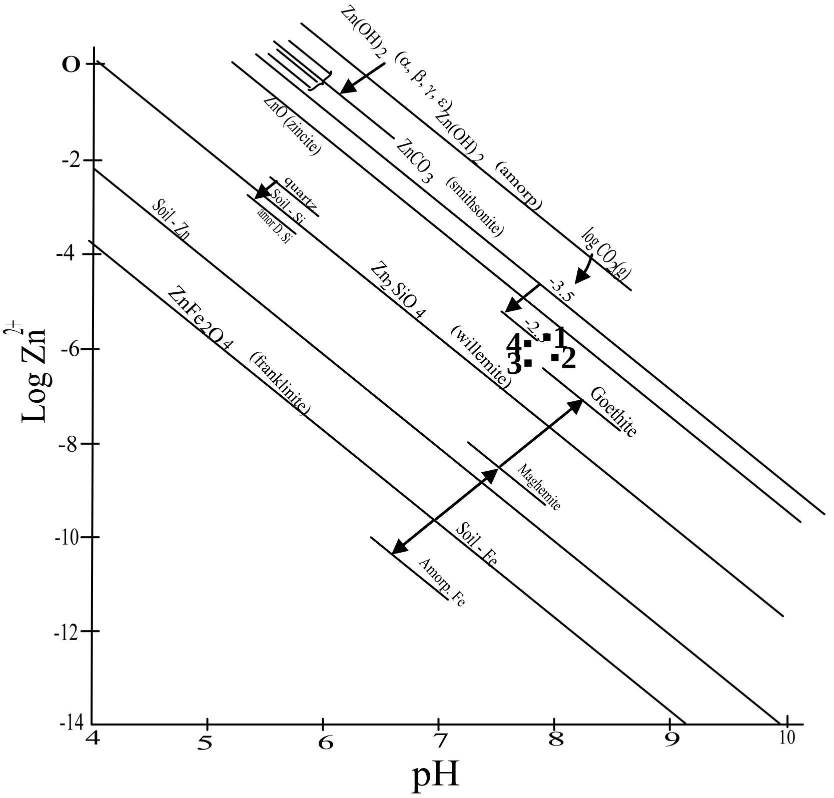


Fig. (70) The effect of different incubation periods on the solubility of Zinc at Arbat location . Points (1 , 2 and 3 and 4 represent 2 , 4 , 8 and 12 months incubation periods respectively)

5. Recommendations

Based on the previous results the following recommendations can be presented:

● In order to understand the actual fluctuation of TSOC further studies are needed on the roles of the types of soil microorganisms dominant in addition to their activities at each stage of incubation periods.

● 2.5 g /100g manure application rates due to each of Carboxyl (COOH) , Hydroxyl (OH) and carbonyl (C= O) functional groups in humic substances which are capable of forming organo - metallic Complexes with (Fe , Mn , Cu , and Zn) as a result of the availability of those ions will increase in soil solution.

● The sheep manure of 0.5 mm size should be applied to agricultural soils in winter, when it is subjected to decomposition by the soil biota which leading to the decrease of turnover of SOM as a result SOC will increase.

● Mostly organic fertilizers should be used as alternatives of the chemical fertilizers, since the latest are expensive and harmful for the environment.

● Field experiments should be carried out in order to be familiar with the factors studies.

● Two months incubation period of sheep manure application is sufficient for Fe, Mn and Zn , since maximum availability of those ions occurred during that period. Whereas twelve months incubation period was optimum for Cu.

● Further studies will be required to study the role of the type of carbonate content on the rate of OM decomposition.

6. References

Abdullah ,S.M.S. 2006. Availabilty indices and physico – chemical behavior of iron in calcareous soil. M. Sc. Thesis. Soil and Water Department. College of Agriculture. University of Sulaimani.

Adani, F.; P. L. Genevini ; and F. Tambone.1995. A new index of organic matter stability .Compost Sci. Utiliz.**3**: 25 – 37.

Aita C.; S. Recous; and D.A. Angers. 1997. Short-term kinetics of residual wheat straw C and N under field conditions: characterization by $^{13}C^{15}N$ tracing and soil particle size fractionation. Eur. J. Soil Sci.; **48:283**-294.

Ajwa, H.A.; and M.A. Tabatabai . 1994. Decomposition of different organic materials in soils. Biol. Fertil. Soils **18**:175–182.

Al – Saadi, E.S. S.1997. Effects of some organic residue application on organic matter and nitrogen dynamic in soil at Jadria location. M .Sc. Thesis. Soil Sci. department. College of Agriculture . University of Baghdad. Iraq. (In Arabic)

Alessandra A. F; L O. D Pedro ; P. D.S Henrique; A.S. Carlos and D S. F. Francisco. 2002. Soil organic carbon and fractions of a Rhodic Ferralsol under the influence of tillage and crop rotation systems in southern Brazil. Soil and tillage Res. Vol. **64** . Issues. 3 – 4. pp. 221 – 230.

Alexander, M. 1961. Introduction to soil microbiology , Indian reprint, John Wiley and Sons, New York.

Alexander, M.1977. Introduction to soil Microbiology, John Wiley and Sons, New York.

Almas .A.R. and B.R. Singh . 2001. Plant uptake of cadmium – 109 and Zinc -65 at Different Temperature and Organic matter levels. J. Environ. Qual. **30**: 869 - 877.

Al-Tamimi, H. J .H. 1997.The chemical behavior of trace elements fertilizers common and artificial chelates producing from humic acids and their efficiency in calcareous soils .PhD .thesis .College of agriculture. University of Basrah. Iraq. (In Arabic)

Alvarez, R. and C.R. Alvarez. 2000. Soil organic matter pools and their association with carbon mineralization kinetics . Soil Sci. Soc. Am. J. **64** : 184- 189.

Amacher, M.C. 1991. Methods of obtaining and analyzing kinetic data. In: Sparks, D.L., Suarez, D.L. Rates of Soil Chemical Processes. Soil Sci. Soc. Am, Madison, WI, pp. 19–59.

Amador, J.A., and R.D. Jones. 1995. Carbon mineralization in pristine and phosphorus enriched peat soils of the Florida Everglades. Soil Sci. **159:** 129 – 141.

Aranda. V. and. C. Oyonarte. 2006. Charactertics of organic matter in soil, surface horizons derived from calcareous and metamorphic rocks and different vegetation types from the Mediterranean high – mountains in SE Spain. European J. of Soil Biology . Vol. No.**42** , Issue 4 , pp. . 247 - 258.

Artoshy , M. 2004. Kinetics of sheep manure decomposition and enzymatic activity in calcareous soils. M. Sc. Thesis . Soil and Water Department. College of Agriculture. University of Dohuk.

Awad, K. M. 1984. Solubility and equilibria of applied zinc in some Iraqi soils. Zanco . **2.** 85 – 100.

Baldock, J.A.; J.M. Oades ; A.G. Waters ; X. Peng ; A.M. Vassalo; and. M.A Wilson. 1992. Aspects of the chemical structure of soil organic materials as revealed by solid-state ^{13}C NMR spectroscopy. Biogeochemistry **16**:1–42.

Baldock, J. A. and, P. N. Nelson. 1999. Soil Organic Matter. In 'Handbook of Soil Science. (Ed M. E. Sumner.) p. B25-B84. (CRC Press: Boca Raton, USA.)

Baldock, J. A. and, J. O. Skjemstad .1999. Soil organic carbon/soil organic matter. In 'Soil Analysis: an Interpretation Manual. (Eds K. I. Peverill, L. A. Sparrow, and D. J. Reuter.) pp. 159-170. (CSIRO Publishing: Collingwood.)

Baldock, J. A. and J. O. Skjemstad . 2000. Role of the soil matrix and minerals in protecting natural organic materials against biological attack. Organic Geochemistry **31**, 697-710.

Barbarick, K.A.; J.A. Ippolito and D.G. Westfall . 1997. Sewage biosolids cumulative effects on extractable-soil and grain elemental concentrations. J. Environ. Qual. **26**:1696–1702.

Barrios E., R.J. Buresh, and J.I. Sprent .1996. Nitrogen mineralization in density fractions of soil organic matter from maize legume cropping systems. Soil Biol. Biochem**.28** (10/11): 1459-1465.

Barrios, E. F. K; R.J. Buresh; and. J.I. Sprent. 1997. Light fraction soil organic matter and available nitrogen following trees and maize. Soil Sci. Soc. Am. J. **61:** 826-831.

Bartoli F.; A. Hatira ; J.C. Andre; J.M. Portal . 1987. Proprietes fluorescentes et colloidales d'une solution organique de Podzol au cours du processus de complexation par le cuivre. Soil Biol. Biochem. ; **19**:355-362.

Baruah.T.C and H. P. Barthakur. 1999. A textbook of soil analysis .UBS. Publishers distributors Ltd.

Bayer, C. L.; N. Martin; J. Mielniczuk; and C.A. Ceretta . 2000. Effect of no-till cropping systems on soil organic matter in a sandy clay loam Acrisol from southern Brazil monitored by electron spin resonance and nuclear magnetic resonance. Soil Tillage Res. **53:** 95–104.

Bayer.C, L..N Martin, J. Mielniczuk, A. Pavinato and J. Dieckow. 2006. Carbon sequestration in two Brazilian Cerrado soils under no-till. Soil and Tillage Res.. Vol. **86**. Issues. 2. pp. 237 – 245.

Berg, B. 2000. Litter decomposition and organic matter turnover in northern forest soils. Forest Ecology and Management. **133** : 13 -22.

Bernston, G.M.; and F.A. Bazzaz. 1996. Belowground positive and negative feedbacks on CO_2 and growth enhancement. Plant Soil :**187**:119–134.

Black C. A. (Ed.). 1965. Methods of soil analysis, pp. 374-399. Part 1. Physical properties. AM. Soc. of Agro. Inc, publisher, Madison, Wisconsin, U.S. A.

Bloom, P. R.; and W. P. Inskeep. 1988. Factors affecting bicarbonate chemistry and iron chlorosis in soils. J . Plant Nutr. **9** : 215 – 228

Bodek, I.; W. J. Lyman ; W. F. Reehl ; and D.H. Rosenblatt. 1988. Environmental Inorganic Chemistry: Properties, Processes, and Estimation Methods. SETAC Special Publication Series, B.T. Walton and R.A. Conway, editors. Pergamon Press. New York

Bohn H.; McNeal B.; and G. O'Connor .1985. Soil chemistry. 2nd edition. John Wiley and Sons, Inc.

Bokhtiar. S. M. and K. Sakurai . 2005. Integrated use of organic manure and chemical fertilizer on growth, yield, and quality of sugarcane in high ganes river floodplain soils of Bangladesh. Communications in soil science and plant analysis .Vol. **36,** No. 13 -14 .pp. 1823 -1837.

Bolan, N.S.; M.A Khan.; D.C Donaldson; D.C Adriano; and C. Matthew . 2003. Distribution and bioavailability of copper in a farm effluent. Sci. Total Environ. **309**:225–236.

Bolan, N.S.; D.C. Adriano; and S. Mahimairaja . 2004. Distribution and bioavailability of trace elements in livestock and poultry manure by-products. Crit. Rev. Environ. Sci. Technol. **34**:291–338.

Bouma, J. 1997. Soil environmental quality: A European perspective. J. Environ. Qual. **26**:26-31.

Brady, N.C.; and R.R. Weil.1996. The nature and properties of soils. 11th ed. Prentice-Hall Inc., Upper Saddle River, NJ.

Bremner J. M. and C. S. Mulvaney . 1982. Nitrogen - Total. P. 595-622. In A.L. page et al., (ed.) Methods of soil analysis. Part 2. chemical and microbiological properties. Agron. Monogr. 9.ASA,Madison,WI.

Bremer E.; B.B. Ellert; and H.H. Janzen.1995. Total and light-fraction carbon dynamic during four decades after cropping changes. Soil Sci. Soc. Am. J. **59**:1398-1403.

Broadbent, F.E.; and T. Nakashima . 1974. Mineralization of carbon and nitrogen in soil amended with carbon-13 and nitrogen-15 labeled plant material. Soil Sci. Soc. Am. J. **38**:313–315.

Broder , M.W. ; and G.H. Wanger .1988. Microbial colonization and decomposition of corn , wheat and soybean residue . Soil Sci Soc. Am. J. **52**: 112 – 117.

Call, H. P.; and I. Mucke. 1997. History, overview and application of mediated lignolytic systems, especially laccase – mediator- system (lignozym-process.). J. Biotechnol. **53:** 163 – 202.

Campbell, C.A.; B.G. McConkey; R.P. Zentner; F. Selles; and D. Curtin. 1996. Long-term effects of tillage and crop rotations on soil organic C and total N in a clay soil in southwestern Saskatchewan. Canadian J. Soil Sci. **76:**395–401.

Caselles M J; M Raul ; D. P. M Maria ; P. E. Aurelia ; P. Copncepcian ; and A. Enrique. 2005. Fe, Cu, Mn, Zn Input and Availability in Calcareous Soils Amended with the Solid Phase of pig Slurry. Communication in Soil Science and Plant Analysis .Vol. **36**. No. 4-6. pp. 525 – 534 .

Chander, K.; P.C. Brookes ; and S.A. Harding . 1995. Microbial biomass dynamic following addition of metal – enriched sewage sludge to a sandy loam, Soil. Biol. biochem. **27**. 1409 -1421.

Chaudri, A.M.; S.P McGrath.; and K.E. Giller.1992. Metal tolerance of isolates of *Rhizobium leguminosarum* biovar *trifolii* from soil contaminated by past applications of sewage sludge. Soil Biol. Biochem. **24**:83–88.

Cheng H.H. and J.A.E. Molina . 1995. In search of bioreactive soil organic carbon: The chlorosis in soils. J. Plant Nutr. **9**: 215-228.

Chen, M.; M. Wang ; C. P. H Chiu ; and H. King. 2001. Determination of low molecular weight dicarboxylic acids and organic functional groups in rhizosphere and bulk soils of Tsuga and Yushania in a temperate rain forest. Plant and Soil. **23**: 37 – 44.

Chodak, M. W.; B Ludwig ; and F. Beese. 2001. Effect of temperature on the mineralisation of C and N of fresh and mature compost in sandy material. J. Plant Nutr. Soil Sci. **164**:289–294.

Christensen B.T. 1992. Physical fractionation of soil and organic matter in primary particle size density separates. Adv. Soil Sci.; 20:1-90.

Ciavatta . C. M; Govi ; A. L. Vittori ; and P. Sequi. 1990. J. Chromatogr. **509:** 141-146. cited from Dellabate et al ., 1997 . Humic substances as indicators of soil organic matter evolution in Vertisols. Scientific registration number : 178 . Symposium n^{o} :12 . Presentation :poster.

Ciavatta, C. ; M. Govi ; L. Pasotti ; and P. Sequi. 1993. Changes in organic matter during stabilization of compost from municipal solid waste . Bioresource Technol. **43**: 141 – 145.

Cieslinski, G.; K.C.J. Rees; A.M. Szmigielska ; and P.M. Huang. 1997. Low molecular weight organic acids released from roots of durum wheat and flax into sterile nutrient solutions. J. of Plant Nutrition. **20**: 753 – 764.

Collins, H.P.; E.A. Paul ; K. Faustian ; and. E.T. Elliott. 1997. Characterization of soil organic carbon relative to its stability and turnover. pp. 51-72. In E.A. Paul, K.Paustian, E.T. Elliot, and C.V.Cole (ed.) Soil organic matter in temperate agroecosystems: long-term experiments in North America. CRC Press, Boca Raton, FL.

Conte P.; and A. Piccolo. 2002. Effect of concentration on the self – assembling of dissolved humic substances. In A .Violante, et al. (ed.) 2002. 3rd Symposium on soil mineral- organic matter- microorganism interactions and ecosystem health . Naples – Capri, Italy. Elsevier Science B.V., Amsterdam, Netherlands.

Cornell, R.M. and U Schwertmann. 1979. Influence of organic anions on the crystallization of ferrihydite. Clays and clay Minerals .**27**: 402 – 410.

Cotrufo, M.F. ; P. Ineson ; and A.P. Rowland . 1994. Decomposition of tree leaf litters grown under elevated CO_2: Effect of litter quality. Plant Soil :**163**:121–130.

Dahlin, S.; E. Witter ; A. Martensson ; A. Turner ; and E. Baath .1997. Where's the limit? Changes in the microbiological properties of agricultural soils at low levels of metal contamination. Soil Biol. Biochem. **29**:1405–1415

Davies, C. W. 1962 . Ion association. Butterworths .London

De Jong, R.1983. Soil water desorption curves estimated from limited data. Canadian J. of Soil Sci. **63**, 697-703.

Debnath, A.; A. C Das; and D. Mukherjee. 1991. Environ. Ecol. 9.315.

Debnath, A.; A. C Das; and D. Mukherjee.1994. Microbial. Res. **149**, 195.

Diaz, S.; J.P. Grime; J. Harris; and E. McPherson . 1993. Evidence for feedback mechanism limiting plant response to elevated carbon dioxide. Nature **365:** 616–617.

Dick D. P.; B. Peter ; and H. Helmut.1999. Influence of extractant and soil type on molecular characteristics of humic substances from two Brazilian soils. J Braz. Chem. Soc.. Vol. **10**. No.2 . 140 -145.

Ekschmitt K; L Manqiang; V Silke; F Oliver and W. Volkmar. 2005. Strategies used by soil biota to overcome soil organic matter stability — why is dead organic matter left over in the soil? Geoderma.Vol.**128.**Issues 1- 2. pp. 167- 176.

Ellert, B.H.; and E.G. Gregorich .1995. Management-induced changes in the actively cycling fractions of soil organic matter.pp.119-138. In W.W. McFee and J.M. Kelly (ed.) Carbon forms and functions in forest soils. Eighth North American Forest Soils Conference. SSSA, Madison, WI.

Elliott, E.T. ; and C. Cambardella.1991. Physical separation of soil organic matter. Agric.Ecosystems Environ., **34**:407-419.

FAO.1980. Soil testing and plant analysis. Bull. No.**38** / I, Food and Agriculture Organization, Rome, Italy.

Fayez R. 2006**.** Carbon and N mineralization as affected by soil cultivation and crop residue in a calcareous wetland ecosystem in Central Iran. Agriculture Ecosystem and Environment. Vol. **112 .**Issue 1.pp. 13 -20.

Fox, J. F.; K. Van Cleve, 1983. Relationships between cellulose decomposition, Jenny's k, forest – floor nitrogen, and soil temperature in Alaskan taiga forests. Canadian J. of Forest Res. **13,** 789 – 794.

Francioso, O.; C. Ciavatta ; S. Sanchez-Cortes; V. Tugnoli ; L. Sitti ; and C. Gessa. 2000. Spectroscopic characterization of soil organic matter in long-term amendment trials. Soil Sci. **165**:495–504

Franzluebbers, A. J.; R.L. Haney; and F. M. Hons.1999 Relationships of chloroform Fumigation – incubation to soil organic matter pools. Soil Biology and Biochemistry **31**: 395 – 405

Franzmeier D.P.; G.D. Lemme ; and R.J. Miles. 1985. Organic Carbon in Soils of North Central United States. Soil Sci. Soc. Am. J., **49:** 702 - 708.

Galantini J.A ; N. Senesi ; G. Brunetti ; and R. Rosell. 2004. Influence of texture on organic matter distribution and quality and nitrogen and sulphur status in semiarid Pampean grassland soils of Argentina. Geoderma.Vol. **123.** Issuses. 1 -2. pp. 143 -152.

Ganjegunte, G.K.; L.M. Condron; P.W Clinton; M.R Davis; and N. Mahieu. 2004. Decomposition and nutrient release from radiate pine (*Pinus radiate)* coarse woody debris. Forest Ecology and Management **187:** 197 – 211.

Giller K.E.; E. Witter; S.P. and McGrath.1998. Toxicity of heavy metals to microorganisms and microbial processes in agricultural soils; a review. Soil Biology and Biochemistry **30**. 1389 – 1414.

Gleixner, G.; N. Poirier; B .T Bol . 2002. Molecular dynamics of organic matter in a cultivated soil.Org.Geochem. **33,** 357 -366.

Goyal, S.; M.M. Mishra ; S.S. Dhankar ; K.K. Kapoor ; and R. Batra. 1993. Microbial biomass turnover and enzyme activities following the application of farmyard manure to field soils with and with out previous long –term applications . Biol. Fertil. Soils. **15**. 60 -64.

Grasset, L.; and A. Ambles. 1998a. Structure of humin and humic acid from an acidic soil as revealed by phase transfer catalyzed hydrolysis .org. geochem. **29**, 881 – 891

Grasset L.; A. Ambles. 1998b. Structure study of soil humic acid and humin using anew preparative thermochemolysis technique. J. Anal. App. Pyrolysis. **47**, 1 – 12.

Greer, K.J.; D.W. Anderson ; and J.J. Schoenau. 1996. Soil erosion, organic matter decline and soil quality indicators. pp. 21–33. *In* G.M. Coen and H.S.

Vanderpluym (ed.) Proc. of Soil Quality Assessment for the Prairies Workshop, Edmonton, Alberta.

Gregorich, E.G.; M.R. Carter; D.A. Angers; C.M. Monreal; and B.H. Ellert. 1994.Towards a minimum data set to assess soil organic matter quality in agricultural soils. Can. J. Soil Sci. **74:**367-385.

Griffin, R. A.; and J. J, Jurinak.1973. Estimation of activity coefficients from the electrical conductivity of natural aquatic systems and soils extracts. Soil Sci. **116**: 26 – 30.

Guggenberger, G. ; B.T. Christensen ; and W. Zech .1994. Land – use effects on the composition of organic matter in particle – size separates of soil I. Lignin and carbohydrate signature , Eur. J. Soil Sci. **45** : 449 - 458

Guiraud, G.; and C. Marol. 1992. Influence of temperature on mineralization kinetics with a nitrification inhibitor (mixture of dicyandiamide and ammonium thiosulphate). Biology and Fertility of Soils. Vol. **13**. No. 1. pp. 1 – 5.

Gupta, R.K.,. V. D. Elshout and I.P. Abrol. 1987. Effect of pH on zinc adsorption-precipitation reactions in an alkali soil. Soil Sci. **143**:198–204.

Haider, K. 1992. Problems related to the humification processes in soils of temperate climates. pp. 55–94. In J.M. Bollag and G. Stotzky (ed.) Soil biochemistry. Vol. **7.** Marcel Dekker, New York.

Hallorans J.M.; W.C. Lindemann; and R. Steiner.2004. Iron Characterization in Manure Amended soils. Communication in Soil Science and Plant Analysis .Vol. **35**. No.15 - 16. pp .2345 –2356 .

Hassink, J.; J.W. Dalenberg. 1996. Decomposition and transfer of plant residue ^{14}C between size and density fractions in soil. Plant soil.**179:** 159- 169.

Havlin, J. L.; D.E. Kissel; L.D Maddux.; M.M Classen; and J. H Long. 1990. Crop rotation and tillage effects on soil organic carbon and nitrogen. Soil Sci. Soc.Am. J**. 54,** 448 -452.

Haynes, R. J. ; and M. S. Mokolobate. 2001. Amelioration of Al toxicity and P deficiency in acid soils by additions of organic residues: a critical review of the phenomenon and the mechanisms involved. Nutrient Cycling in Agroecosystems **59**, 47-63.

Haynes, R. J.; and R. Naidu. 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. Nutrient Cycling in Agroecosystems : **51**: 123-137.

Hem, J. D. ; and C. J. Lind.1983. Nonequilibrium models for predicting forms of precipitated manganese oxides. Geochim. Cosmochim. Acta .**47**:2037-2046.

Hevia, G. G.; D. E. Buschiazzo; E. N. Hepper; A. M. Urioste ; and E. L. Antón. 2003.Organic matter in size fractions of soils of the semiarid Argentina. Effects of climate, soil texture and management. Geoderma. Vol. **116.** Issues 3 - 4. pp. 265 - 277.

Hobbie, S.E.2000. Interactions between litter lignin and soil nitrogen availability during leaf litter decomposition in a Hawaiian Montane Forest. Ecosystems **3:** 484 – 494.

Hodgson, J.F.; W. L Lindsay; and J. F. Trierweiler. 1966. Micronutrient cation complexes in soil solution: Complexing of zinc, copper in displaced solution from calcareous soils. Soil Sci .Soc. Am. Proc. **30**: 723 -726.

Hoefs M. J.L. ; and D.P. Lagas . 1992 . Literature review on : Soil organic matter characterization . Rapport 725701002

Hoyle F.C.; D. V. Murphy; and I.R.P. Fillery. 2006. Temperature and stubble management influence microbial CO_2 – C evolution and gross N transformation rates. Soil Biology & Biochemistry **38:** 71 – 80

Hungate B.A.; E.A. Holland; R.B. Jackson; F.S. Chapin III; C.B. Field; H.A. Mooney; and C.B. Field.1997. The fate of carbon in grasslands under carbon dioxide enrichment. Nature **388**:576–579.

Igwe.C.A; and D.Nwokocha. 2006. Soil organic matter fractions and microaggrigates in a Ultisol under cultivation and secondary forest in south – eastern Nigeria Aust. J. of soil Res.: **44** (6) 627 – 635.

Janzen, H.H.; C.A. Capbell; E.G. Gregorich; and B.H. Ellert . 1998. Soil carbon dynamics in Canadian agroecosystems. In: Lal , R., Kimble J.M., Follett; R.F. Stewart; B.A.(EDS), Soil processes and carbon cycles. CRS Press, Boca Raton , FL , pp. . 57 – 80.

Jeffery, J.J.; and N.C. Uren. 1983. Copper and zinc species in the soil solution and the effects of soil pH. Aust. J. Soil Res. **21**:479–488.

Jhorar, B.S., V. Phogat, and R.S.Malik. 1991. Kinetics of composting rice straw with glue waste at different carbon : nitrogen ratios in a semiarid environment. Arid Soil Research and Rehabilitation **5**: 297 -306.

Jimenez, E., and P. Garcia. 1992. Determination of maturity indices for city refuse composts. Agric. Ecosyt. Environ. **38**: 331 – 343.

JoaquÃn M. C. ; R. Moral ; M. D.P. Murcia ; A. P. Espinosa ; C. Paredes ; and E. AgullÃ. 2005. Fe, Cu, Mn, and Zn Input and Availability in Calcareous Soils Amended with the Solid Phase of Pig Slurry. Communications in Soil Science and Plant Analysis. Vol. **36**, No. 4-6 pp. 525 – 534.

Johansson, M.B. 1986. The chemical composition of needle and leaf litter from Scots pine, Norway spruce and White birch in Scandinavian forests. Forestry **68**:49–61.

Kabata-Pendias, A., and H. Pendias. 1992. Trace Elements in Soils and Plants, second ed. CRC Press, Boca Raton, FL.

Kalembasa, S. J. ; and D. S. Jenkinson. 1973. A comparative study of titrimetric and gravimetric methods for the determination of organic carbon in soil. Journal of the Science of Food and Agriculture **24**, 1085-1090.

Karen A. M. ; and M. S. Erich. 2003.Influence of Organic Matter Decomposition on Soluble Carbon and Its Copper-Binding Capacity. J. Environ. Qual. **32**: 2122 - 2131.

Karim, T. H.2000. Models to predict water retention of Iraqi soils. Indian J. of soil Sci. **47**: 19 -23.

Kathryn M. Catlett; D. M. Heil ; W. L. Lindsay; and M. H. Ebinger. 2002 . Soil chemical properties controlling zinc activity in 18 Colorado soils. Soil Sci. Soc. Am. J. **66** : 1182-1189.

Khan, M .A. R.; N. S. Bolan ; and A. D. MacKay.2005. Adsorption and Desorption of Copper in Pasture Soils. Communication in soil science and plant analysis.Vol.**36.**No.17 – 18.pp. 2461 – 2487.

Kiem, R.; Kogel - Knabner I. 2003. Contribution of lignin and polysaccharides to the refractory carbon pool in C- depleted arable soils. Soil Biol. Biochem. **15**, 101-118.

Kimble, L.; and B.A. Stewart. 1995. World soils as a source or sink for radiatively active gasses. pp. 1–7. In R. Lal et al. (ed.) Soils and global change. CRC Press, Boca Raton, FL.

Kögel-Knabner, I.1993. Biodegradation and humification processes in forest soils. Soil Biochem. **8:**101–135.

Körner, C. ; and J.A. Arnone.1992. Responses to elevated carbon dioxide in artificial tropical ecosystems. Science **257**:1672–1675.

Krull E. S.; O. J. Skjemstad; and J. A. Baldock. 2004. Functios of soil organic matter and the effect on soil properties. GRDC Project No CSO 00029.

Lal, R., and B.A. Stewart.1992. Need for soil restoration. Adv. Soil Sci**.** **17:**1-11.

Lanpo Z.; Y. Sun ; X. Zhang ; X. Yang ; and C.F. Drury.2006. Soil organic carbon in clay and silt sized particles in Chinese mollisols: Relationship to the predicted capacity. Geoderma. Vol. **132.** Issues. 3 – 4 .pp. 315 – 323.

Lee, K.E. 1974.The significance of soil animals in organic matter decomposition and mineral cycling in tropical forest and savanna ecosystems. Soil and Fertilizers. **38:** 898

Leggett, G. E. 1978. Interaction of monomeric silicic copper and zinc and chemical charges of the precipitates with Aging. Soil Sci .Soc. Am.J. **42**: 262 – 266.

Leifeld J., S. Siebert, and I. Kogel – Knabner. 2001. Stabilization of composted organic after application to a humus – free sandy mining soil. J. Environ.Qual. **30**. 602 – 607.

Leita, L., M.D. Nobili and C. Mondini. 1999. influence of inorganic and organic fertilization on soil microbial biomass, metabolic quotient and heavy metal bioavailability , Biol. Fertil. Soils. **28**. 371 – 376.

Lerch, R.N., K.A. Barbarick, D.G. Westfall, R.H. Follett, T.M. McBride, and W.F. Owen. 1990. Sustainable rates of sewage sludge for dryland winter wheat production. I. Soil nitrogen and heavy metals. J. Prod. Agric. **3**:60–65.

Lewis T.E., F.E. Broadbent. 1961. Soil organic matter–metal complexes: 4. Nature and properties of exchange sites. Soil Sci.; **91**:393-399.

Lighthart, B., J. Baham, and V.V. Volk. 1983. Microbial respiration and chemical speciation in metal-amended soils. J. Environ. Qual and chemical speciation in metal-amended soils. J. Environ. Qual.

Lindsay, W.L. 1979. Chemical equilibria in soils. John Wiley & Sons, New York.

Lindsay, W.L. and W.A. Norvell. 1978. Development of DTPA micronutrient soil test for zinc, iron, manganese and copper. Soil Sci. Soc. Am. Proc. **42:** 421 – 428.

Lipson, D. A., T. K. Raab, S. K. Schmidt and R. K. Monson. 2001**.** An empirical model of amino acid transformations in an alpine soil. Soil Biology and Biochemistry **.33**: 189 – 198.

Loveland. P. and J. Webb. 2002. Is there a critical level of organic matter in the agricultural soils of temperate regions: Soil and Tillage Res. Vol. **70.**, Issue 1. , pp. 1 - 18

Lugo, A.E., and S. Brown.1993. Management of tropical soils as sink or sources of atmospheric carbon. Plant Soil **149**:27–41

Lynch, J. M. 1991. Sources and fate of soil organic matter .p. 231 -238. In E.S. Wilson (ed.) Advances in soil organic matter research, the impact on agriculture and environment. Royal Society of Chemistry, Cambridge.

Ma, Q., and W.L. Lindsay. 1990. Divalent zinc activity in arid-zone soils obtained by chelation. Soil Sci. Soc. Am. J. **54**:719–722.

Magdoff F.R., M.A. Tabatabai, and E.A. Hanlon, Jr. (Eds.). 1996. Soil Organic Matter: Analysis and Interpretation. Soil Science of America, Madison, WI.

Manahan, S.E. 2000. Soil environmental chemistry. Boca Raton: CRC Press LLC,

Marinari .S., K. Liburdi, G. Masciandaro, B. Ceccanti B. and S. Grego**.** 2007**.** Humification-mineralization pyrolytic indices and carbon fractions of soil under organic and conventional management in central Italy. Soil and Tillage Res. Vol. **92**, Issues 1-2, , P.10-17

Martens, D.C, and W .L Lindsay.1990. Testing soils for copper, iron manganese and zinc .p.229 – 264. In R. L. Westerman (Ed), Soil testing and plant analysis, 3rd ed., Soil Sci. Soc. Am., Madison, WI, USA.

Martin, J.P., K Haider, G. Kassim .1980. Biodegradation and stabilization after 2 years of specific crop, lignin, and polysaccharide carbons in soils. Soil. Sci. Soc. Am.J**. 44,**1250 -1255.

Mascagni, J. H.T., and F.R.Cox. 1985. Evaluation of inorganic and organic manganese fertilizer sources. Soil Sci. Soc. Am. J.**49**: 458 - 461.

Matus, F.J., and J. Rodríguez. 1994. A simple model for estimating the contribution of nitrogen mineralization to the nitrogen supply of crops from a stabilised pool of soil organic matter and recent organic input. Plant Soil **162:**259–271.

McBride, M.B.1979. Chemisorption and precipitation of Mn $^{+2}$ at $CaCO_3$ surfaces. Soil Sci. Soc. Am.J.**43:** 693 - 698.

McBride, M. B.; and Blasiak, J .J. 1979. Zinc and copper solubility as a function of pH in an acid soils . Soil Sci. Soc. Am. J. **43**: 866 – 870.

McBride, M.B., and D.R. Bouldin. 1984. Long term reactions of copper (II) in a contaminated calcareous soil. Soil Sci Soc. Am. J .**48**: 56 -59.

McBride M., Sauve S., and Hendershot. W. 1997. Solubility control of Cu, Zn, Cd and Pb in contaminated soils. Eur. J. Soil Sci. **48**:337–346.

McBride, M.B. 1989. Reactions controlling heavy metal solubility in soils. Adv. Soil Sci. **10**:1–56.

McBride, M.B. 1994. Environmental chemistry of soils. Oxford University Press, New York.

McGrath, S.P., P.C. Brookes, and K.E. Giller. 1988. Effects of potentially toxic metals in soil derived from past applications of sewage sludge on nitrogen fixation by *Trifolium repens* L. Soil Biol. Biochem. **20**:415– 424.

McLatchey, G.P., and K.R. Reddy. 1998. Regulation of organic matter decomposition and nutrient release in a wetland soil. J.Environ. Qual. **27**; 1268 – 1274.

Meijboom, F.W., J. Hassink, and M Van Noordwijk. 1995. Density frcationation of soil macroorganic matter using silica suspensions. Soil Biol. Biochem. **27**:1109-1111.

Min. D.H. ; K.R. Islam, L.R.Vough ,and R.R. Weil. 2003 . Dairy manure effect on soil quality properties and carbon sequestration in alfalfa orchard grass systems. Communications in soil science and plant analysis . Vol. **34**. No. 5-6. pp. 781 – 790.

Minnich, M.M., and M.B. McBride. 1986. Effect of copper activity on carbon and nitrogen mineralization in field-aged copper-enriched soils. Plant Soil. **91**:231–240.

Mishra, B., P.K.Sharma, and K.F. Bronson. 2001. Kinetics of wheat straw decomposition and nitrogen mineralization in rice field soil. J. of the Indian Soc. of Soil Sci. **49 :** 249 – 254.

Misra .S.G and R.G. Tiwari. 1966. Retention and release of copper and zinc by some Indian soils. Soil Sci. **101**: 465 – 471.

Monrozier, L.J., J.N. Ladd, R.W. Fitzpatrick, R.C. Foster, and M. Raupach. 1991. Components and microbial biomass content of size fractions in soils of contrasting aggregation. Geoderma **49**:37–62.

Moore, P.A.J., and W.H. Patrich. 1989. Manganese availability and uptake by rice sulfate soils. Soil Sci Soc. Am.J. **53:** 104 – 109.

Murwira, H. K; H kirchmann, and M. J Swift. 1990.The effect of moisture on the decomposition rate of cattle manure .plant and soil. Vol.**122**. No. 2. pp. 197 – 199.

Negre M., D. V. S. Spera, L. Ferraris, and M. Gennari. 2002. Effect of the chemical composition of soil humic acids on their viscosity, surface pressure, and morphology. Soil Sci.. **167:** 636 – 651.

Nelly B., R. D. Faulkner, A. R. Till and P.R. Poulton. 2006. Long –term management impacts on soil C, N. and physical fertility. Soil and Tillage Res.. Vol. **91**. Issues 1 -2. pp. 30 -38 .

Nelson D. W. and L.E. Sommers. 1982. Total carbon, organic carbon and organic matter. P.539 -580. In . page, A. L. ,Miller, R., H., Keeney, DR. (eds.) Methods of soil analysis. Part 2. chemical and microbiological properties. Agron. Monogr. 9.ASA,Madison,WI.

Oades, J.M. 1988. The retention of organic matter in soils. Biogeochemistry **5**:35–80.

Oades, J.M. 1989. An introduction to organic matter in mineral soils. pp. 89–159. *In* J.B. Dixon and S.B. Weed (ed.) Minerals in soil environments. 2nd ed. SSSA, Madison, WI.

Oades, J.M., and A.G. Waters. 1991. Aggregate hierarchy in soils. Aust. J. Soil Sci. **29**:815–828.

Olk D.C., K.G. Cassman, E.W. Randall, P. Kinchesh, L.J. Sanger, J.M. Anderson. 1996. Changes in chemical properties of organic matter with intensified rice cropping in tropical lowland soil. Eur. J. Soil Sci.; **47**:293-303.

Olson, R.V., and C.W. Swallow. 1984. Fate of labeled nitrogen fertilizer applied to winter wheat for five years. Soil Sci. Soc. Am. J. **48**:583–586.

Page, A. L.; R. H. Miller; D. R. Keeney. 1982. Methods of soil analysis. part 2 . Chemical and microbiological properties. 2nd edition Agronomy No.9. ASA, Madison, Wisconsin USA.

Parr,J. F.; and R.I. Papendick. 1978. Crop residue management system (Ed. W .R. Oschwald) ASA, Madison, W. I.

Pascual, J.A., C. Garcia, C., and T. Hernandez. 1999.Lasting microbiological and biochemical effects of the addition of municipal solid waste to an arid soil . Biol. Fertil. Soils. **30**: 1-6

Paul E.A. (Eds.). 1997. Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America. CRC Press.

Paul, E.A., and H.P. Collins. 1998. The characteristics of soil organic matter relative to nutrient cycling. p.181-197. *In* R. Lal, W.H. Blum, C. Valentine, and B.A. Stewart.(ed.) Methods for assessment of soil degradation. Advances in Soil Science. CRCPress, Boca Raton.

Paustian, K., W. J Parton. , J. Person. 1992. Modeling soil organic matter in organic amended and nitrogen fertilized long –term plots. Soil Sci. Soc. Am. J**. 56,** 476 – 488.

Piccolo, A., .Conte P, R. Spaccini, and M. Chiarella. 2003. Effects of some dicarboxylic acids on the association of dissolved humic substances. Biology and Fertility of Soils. **37:** 255 -259.

Piccolo. G.A, J. A .Galantini, and R. A. Rosell. 2004. Organic carbon fractions in a yerba mate plantation on a subtropical kandihumult of Argentina. Geoderma. Vol. **123.** Issues 3 - 4. pp. 333 – 341.

Plaza. C., N. Senesi, A. Polo, G. Brunetti, J. C. Garcia – Gil and V. D Orazio. 2003. Soil fulvic acid properties as a means to assess the use of pig slurry amendment. Soil and Tillage Re. Vol. **74** , Issue 2 ,pp. 179 – 190.

Plaza.C., V. D'Orazio; and N. Senesi. 2005. Copper(II) complexation of humic acids from the first generation of EUROSOILS by total luminescence spectroscopy . Geoderma .Vol. **125**, Issues 1-2 , pp.177-186

Ponizovsky A. A., D. M. Metzler, H.E. Allen, and A. J. Ackerman. 2006. The effect moisture content on the release of organic matter and copper to soil solutions. Geoderma . Vol. **135**. pp. 204 -215.

Powlson , D. S., P.C. Brookes, and , B. T Christensen.1987. Measurements of soil microbial biomass provides an early indication of changes in total soil organic matter due to straw incorporation, Soil Biol. Biochem . **19**.159 -164.

Praveen-Kumar, T.; C Jagadish ; P. Jitendra ; and K. Shyam. 2003. A rapid method for assessment of plant residue quality. J. of Plant Nutrition and Soil Sci. **166**, 662-666.

Qadir. K. S. 2001. Study of erodobility of soils in Iraqi Kurdistan region. M. Sc. Thesis . Soil and Water Department. College of Agriculture. University of Sulaimani.

Qin, C.N., and S.S. Mao. 1993. Environmental soil science. China Agric. Press, Beijing.

Qullas. G. R. 2004. Biodegradability of humic substances and other fractions of decomposing leaf litter. Soil Sci. Soc. Am. J. **68**: 1705 -1712.

Rahi, H. S. ; I. I. Khzair ; and M.A. Jamal Al – Obaidi. 1991. Soil chemical analysis. Mosul . university press.

Rahn, C.R, G.D. Bending, M.K. Turner, and R.D. Lillywhite. 2003. Management of N mineralization from crop residues of high N content using amendment materials of varying quality .Soil Use and Management . **19**: 193 – 200.

Randhawa. H. S. and S. P. Singh. 1996. Copper fractions in Soils of Punjab. J. of the Indian Soc.of Soil Sci .Vol. **44.** No. 2. pp. 243 – 246.

Rasmussen, P. E. and H. P. Collins. 1991. Long-term impacts of tillage, fertilizer, and crop residue on soil organic matter in temperate semiarid regions. *Advances in Agronomy* **45**, 93-134.

Reddy, K.R., and E. M.D Angelo. 1994. Soil processes regulating water quality in wetlands. P. 309 -324. In W. J. Mitsch (ed.) Global wetlands; Old word and new. Elsevier, Amsterdam.

Rethemeyer J. , C. Kramer, G. Gleixner , B. John , T. Yamashita, H. Flessa , N. Andersen, M. J. Nadeau and P. .M. Grootes**.** 2005**.** Transformation of organic matter in agricultural soils: radiocarbon concentration versus soil depth. Geoderma. Vol. **128**, Issues 1 -2, pp. 94 – 105.

Rey. A., Petsikos C., P.G. Jarvis and J. Grace. 2005. Effect of temperature and moisture on rates of carbon mineralization in a Mediterranean oak forest soil under controlled and field conditions. European Journal of Soil Sci. Vol. **56,** Issue 5 . pp .589.

Riffaldi R., A Saviozzi, R. Lev-Minzi. 1996. Carbon mineralization kinetics as influenced by soil properties. Biol. Fertil. Soil; **22** .293 -298

Römheld, V., and H. Marschner. 1986. Mobilization of iron in the rhizosphere of different plant species. In: Advances in Plant Nutrition, Vol. **2**, B. Tinker and A. Läuchli, editors. Praeger Scientific, New York, pp. 153 – 204.

Roscoe. R, P. Buurman, E. J. Velthorst and C. A. Vasconcellos. 2001. Soil organic matter dynamics in density and particle size fractions as revealed by the $^{13}C/^{12}C$ isotopic ratio in a Cerrado's oxisol. Geoderma. Vol. **104.** Issues 3 - 4. pp. 185 -202.

Rowell .D. L.1996. Soil science: Methods and applications. Longman Group UK Limited. pp. 50 – 51.

Rumple. C.; I. Kogel – Knabner, H. Knicker, R.F. Huttl. 2000. Composition and distribution of organic matter in physical fractions of a rehabilitated mine soil rich in lignite – derived carbon. Geoderma. **98**: 177 – 192.

Ruttimann –Johnson C., and R . T. Lamar. 1996. Polymerization of pentachlorophenol and ferulic acid by fungal extracellular lignin-degrading enzymes. Appl. Environ. Microbial. **62.**3890 -3893.

Ryan J., G. Estefan, and Abdul Rashid. 2001. Soil and plant analysis laboratory manual.2nd edition .International center for agriculture research in the dry areas (ICARDA).

Sadiq, M. 1991. Solubility and speciation of zinc in calcareous soils. Water Air Soil Pollut. 57/58:411– 421.

Sanchez, F.G. 1998. Soil organic matter and soil productivity: Searching for the missing link. P.543- 556. *In* R.A. Mickler and S. Fox (ed.) The productivity and sustainability of southern forest ecosystems in a changing environment. Springer Publ., New York.

Sanchez, P.A., and T.J. Logan. 1992. Myths and science about the chemistry and fertility of soils in the tropics. p. 35–46. *In* R. Lal and P.A. Sanchez (ed.) Myths and science of soil of the tropics. SSSA Spec. Publ. **29**. SSSA, Madison, WI.

Sarkanen, K. V., and C.H. Ludwig. 1971. Lignins: occurrence, formation, structure, and reactions. New York: Wiley – Interscience.

Sarmah. A. C. and P. K. Bordoloi .1994. Decomposition of organic matter in soils in relation to mineralization of carbon and nutrient availability. Journal of the Indian Society of soil science, Vol. **42:** No. 2. pp. 199 – 203.

Sauve , S., N. Cook, W.H. Hendershot , and M.B. McBride. 1996. Linking plant tissue concentrations and soil copper pools in urban contaminated soils. Environ. Pollut. **94**: 153 – 157.

Sauve, S., A. Dumestre, M. McBride, and W. Hendershot. 1998. Derivation of soil quality criteria using predicted chemical speciation,. of Pb^{+2} and Cu^{+2} Environ. Toxicol. Chem. **17:**1481–1489.

Schnitzer M., D.F.E. McArthur, H.R. Schulten, L. M Kozak, P.M. Hauang. 2006. Long-term cultivation effects on the quantity and quality of organic matter in selected Canadian prairie soils. Geoderma. **130**: 141 – 156.

Schnitzer, M. 1982. Organic matter characterization. P. 581-594. In A.L.Page et al. (ed) Methods of soil analysis . Part 2. 2nd ed. Agron. Monogr. 9.ASA, Soils and global change. CRC Press, Boca Raton, FL.SSSA, Madison, WI.

Schulten, H. R. and M. Schnitzer. 1995. Three dimensional models for humic acids and soil organic matter. Naturwiss.**82**: 487 - 498.

Schulten, H. R., Schnitzer, M. 1997. Chemical model structures for soil organic matter and soils .Soil Sci. **162,** 115 – 130.

Schwertmann, U. 1988. Occurrence and formation of iron oxides in various pedoenvironments. In: *Iron in Soils and Clay Minerals*, J. W. Stucke et al., editors. NATO ASI Series C127. D. Reidel Publ. Co., Boston, pp. 267-308.

Senesi, N. 1992. Binding mechanisms of some pesticides to soil humic substances. Sci. Total Environ. **123**:63–76

Shevchenko, S.M. and G.W. Bailey. 1996. Life after death: lignin – humic relationships reexamined. Crit. Rev. Environ. Sci. Technol. **26:** 95 -154

Shields J.A., and Paul E.A.1973. Decomposition of C^{14}-labelled plant material under field conditions. Can. J. Soil Sci.; **53**:297-306.

Sims J.T.1986. Soil pH effects on the distribution and plant availability of manganese, copper, and zinc. Soil Sci. Soc. Am. J.**50:**367-373.

Sims J.T., J.S. Kline. 1991. Chemical fractionation and plant uptake of heavy metals in soils amended with co-composted sewage sludge. J. Environ. Qual.; **20**:387-395

Singh, M. V., R. Chhabra, and I. P. Abrol. 1983. Factors affecting DTPA – extractable zinc in sodic soils. Soil Sci. **136.** 359 – 366.

Singh, M.V., and I.P. Abrol. 1985. Solubility and adsorption of zinc in a sodic soil. Soil Sci. **140:**406–411.

Singh.G.P; V.Beri; and B.S.Sidhu. 1994. Humification of rice and wheat residues in soil. J. of the Indian Soc. of Soil Sci., Vol.**43.**No.1.p.17 -20.

Sinha, M.K, S.K Dhillon, K.S Dhillon and S. Dyanand. 1978. Solubility relationships of iron, manganese, copper and zinc in alkaline and calcareous soils. Australian Journal of Soil Research .Vol. **16** .No. 1 pp.19 -26.

Skjemstad, J. O. and J. A. Taylor. 1999. Does the Walkley-Black method determine soil charcoal? Communications in Soil Science and Plant Analysis **30**, 2299-2310.

Soil and Water Conservation Society. 1995. Farming for a better environment-A white paper. Soil Water Conserv. Soc. Ankeny, IA

Soil Science Society of America. 1987. Glossary of soil science terms. SSSA, Madison, WI.

Soil Survey Staff.1994 . Key to soil taxonomy. Pocahontas press, Blackburg. VA.

Sollins, P., P. Homann, and B.A. Caldwell. 1996. Stabilisation and destabilisation of soil organic matter: Mechanisms and controls. Geoderma **74:**65–105.

Solomon, D., J. Lehmann, and W. Zech. 2000. Land use effects on soil organic matter properties of chromic Luvisols in the semiarid tropics: Carbon, nitrogen, lignin and carbohydrates. Agric. Eco. Environ. **78**:203–213.

Soltanpour, P.N. 1985. Use of ammonium bicarbonate-DTPA soil test to evaluate elemental availability and toxicity. Commun. Soil Sci. Plant Anal**.** **16:** 323 – 338.

Sparks. D. L. 1989. Kinetics of soil chemical processes. pp. 4 -31. ACADEMIC PRESS,INC. Harcourt Brace Jovanovich, Publishers.

Sparks, D.L., Suarez D., and L. ŽEds. 1991. Rates of Soil Chemical Processes. SSSA Spec. Publ., vol. **27,** Soil Sci. Soc. Am, Madison, WI

Sparks, D.L., P.C. Zhang. 1991. Relaxation methods for studying kinetics of soil chemical phenomena. In:Sparks, D.L., Suarez, D.L.ŽEds.., Rates of Soil Chemical Processes. SSSA Spec. Publ. No. **27**, pp. 61–94, Madison, WI.

Sparks, D.L., S.E. Fendorf, I.V. Toner, C.V. Carski.1996. Kinetic methods and measurements. In: Sparks, D.L. ŽEd.., Methods of Soil Analysis: Chemical

Methods. SSSA Book Ser. No. 5, Soil Sci. Soc. Am., Madison, WI, pp. 1275–1307.

Sparks, D.L. 2000. Kinetics and mechanisms of soil chemical reactions. In: Sumner, M.E.ŽEd.., Handbook of Soil Science. CRC Press, Boca Raton, FL, p. B-123, Last-Page B-168.

Sparling, G.P. 1991. Organic matter and soil microbial biomass C as indicators of sustainable land use, pp. 563 – 580 , J. Dunmanski, et al ., eds. Evaluation for sustainable land management in the developing world. International Board for Soil Research and Management Inc. (IBSRAM) , Bangkok.

Sposito, G. 1984. The surface chemistry of soils. Oxford University Press, New York.

Sposito, G. 1989. The soil solution. P.80. In the chemistry of soils. Oxford Univ. Press, Oxford.

Sposito, G.1994. Chemical Equilibria and Kinetics in Soils. Wiley, New York.

Spycher, G., P. Sollins, and S. Rose. 1983. Carbon and nitrogen in the light fraction of a forest soil: Vertical distribution and seasonal patterns. Soil Sci. 135(2):79-87

Stanford G., S.J. Smith. 1972. Nitrogen mineralization potentials of soils. Soil Sci. Soc. Am. Proc.; **36:465** - 472

Stevenson F.J. 1982. Humus chemistry. Genesis, composition, reactions. John Wiley and Sons.Cited from Tatana A., D. Andelkovic, J. Perovic, M. Purenvic, and P . Polic. 2001. Decrease of oxygen interference on humic acid structure alteration during isolation . Physics , Chemistry and Technology . Vol. 2. No. 3. pp. 163 – 171.

Stevenson, F.J. 1994. Humus Chemistry: Genesis, Composition, Reactions (2nd ed). John Wiley and Sons Inc, New York, USA. Cited from Paul , E.A. , S. J. Morris , J Six , K . Paustian, and E.G. Gregorich .2003. Interpretation of soil carbon and nitrogen dynamics in agriculture and afforested soils. Soil Sci. Am. J . **67**: 1620 – 1628.

Strickland, T.C., and P. Sollins.1987. Improved method for separating light- and heavy fraction organic material from soil. Soil Sci. Soc. Am. J. **51**:1390-1393.

Striganova, B. R., and P. Bienkowski. 2000. Rates of decomposition processes in mountain soil of the Sudeten as a function of edaphic – climatic and biotic factors. Biology Bulletin.**27**: 619 – 628.

Stumm, W. and J. J. Morgan. 1970. Aquatic chemistry: an introduction emphasizing. chemical equilibria in natural waters. Wiley-Interscience, New York

Suflita, J.M., and J.M. Bollag. 1981. Polymerization of phenolic compounds by soil – enzyme complex. Soil Sci. Soc.Am. J. **45**: 297-302.

Swift, M.J., O.W. Heal, and J.M. Anderson. 1979. Decomposition in terrestrial ecosystems. 371 p. Blackwell Scientific Publications, London, England.

Sylvia, M . D. Jeffry; G. Peter.; and D. A. Zuberer. 2005. Principles and applications of soil microbiology .2nd edition. Pearson Prentice Hall. Upper Saddle River, New Jersey.

Tate R.L. 1987 Ecosystem management and soil organic matter level, in: Wiley, J., Sons Inc., (Eds.), Soil Organic Matter. Biol. Ecol. Effects, pp. 260–280.

Tatjana A. , D. Andelkovic , J. Perovic , M. Purenovic , and P. Polic. 2001. Decrease of Oxygen interference on humic acid structure alteration during isolation. Physics, chemistry and Technology .Vol. **2.** , No. 3. pp. 163 – 171.

Taylor, B.R., D. Parkinson, and W.F.J. Parsons. 1989. Nitrogen and lignin content as predictors of litter decay rates: A microcosm test**.** Ecology **70:**97-104.

Theng, B.K.G., Tate, K.R., Sollins, P., Nadkarni, N. and Tate, R.L. 1989. Constituents of organic matter in temperate and tropical soils. Pages 5-32 in D.C. Coleman, J.M. Oades, and G. Uehara, Eds. Dynamics of soil organic matter in tropical ecosystems, NifTAL Project, University of Hawaii. USA.

Thurston, C. F. 1994. The structure and function of fungal laccase. Microbiology . **140:** 19 -26.

Tiller, K.G., J.L. Honeysett, and M.P.C. de Vries. 1972. Soil zinc and its uptake by plants. Aust. J. Soil Res. **10:**165–182.

Tisdall, J.M, J.M Oades.1982.Organic matter and water stable aggregates in soils. J Soil Sci. **33:** 141 -163

Tisdale S. L.; W. L. Nelson; J. D. Beaton and J. L. Havlin. 1993. Soil fertility and fertilizers. 5th edition. Prentice- Hall, Inc.

Topolianz, S., J.F. Ponge, D. Arrouays, S. Ballof, and P.Lavelle.2002. Effect of organic manure and the endogeic earthworm *Pontoscolex corethrurus* (Oligochaeta: Glossoscolecidae) on soil fertility and bean production. Biology and fertility of Soils**.36**: 313 -319.

Torri S., R. Alvarez, and R. Lavado. 2003**.** Mineralization of Carbon from Sewage Sludge in Three Soils of the Argentine Pampas. Communications in Soil Science and plant Analysis. Vol. **34**. No. 13 -14. pp. 2035 -2043.

Tripathi D; S Karan. ; and G.P. Upadhyay. 1993 . Distribution of micronutrients in some representative soil profiles of Himachal Pradesh. Journal of the Indian Society of Soil Science. Vol. **42.** No. 1. pp. 143 - 145

Turchenek, L.W., and J.M. Oades.1979. Fractionation of organo-mineral complexes by sedimentation and density techniques. Geoderma, **21**:311-343.

U.S. Dept. of Agric. 1938. Soils and men, yearbook of Agriculture, pp. 347 -360.

Usman, A. R. A.; Y. Kuzyakov, and K. Stahr. 2004. Dynamics of organic mineralization and the mobile fraction of heavy metals in a calcareous soil incubated with organic wastes. Water, Air, and Soil Pollution. **158:** 401 – 418.

Valarini J.P.; M. C. D. Alvarez; J. M Gasco; F. Guerrero; H. Tokeshi. 2002 .Integrated evaluation of soil quality after the incorporation of organic matter and microorganisms. Brazilian Journal of microbiology: **33**: 35 – 40.

Van Reeuwijk. L.R. 1995. Procedures for soil analysis .p. 3-8. 5th edition .International soil reference and information center, technical paper 9.

Vance, E.D.; and F.S. Chapin. 2001. Substrate limitations to microbial activity in taiga forest floors. Soil Biology & Biochemistry. **33:** 173 – 188.

Wander, M.M., and X. Yang. 2000. Influence of tillage on the dynamics of loose and occluded particulate and humified organic matter fractions. Soil Biol. Biochem. **32**:1151–1160.

Wander, M.M., S.J. Traina, B.R. Stinner, and S.E. Peters. 1994. Organic and conventional management effects on biologically active soil organic matter pools. Soil Sci. Soc. Am. J. **58**:1130-1139.

Webb. J., P. Bellamy, P. J. Loveland and G. Goodlass. 2003. Crop Residue Returns and Equilibrium Soil Organic Carbon in England and Wales. Soil Sci. Soc. of Am. J. **67**:928-936

Wei X., M. Hao, M. Shao, and W. J. Gale. 2006. Changes in soil properties and the availability of soil micronutrients after 18 years of cropping and fertilization. Soil and Tillage Res.. Vol. **91**. Issues 1-2 .pp. 120 – 130.

Weil, R.R. 1992. Inside the heart of sustainable farming. The New Farm. 14 Jan. 43-48.

Wheeler .B.D., M.M Al – Farraj and R. E. D. Cook. 1985. Iron toxicity to plants in base –rich wetlands: comparative effects on the distribution and growth of

Epilobium hirsutuml and Juncus subnodulosus schrank. New phytol. **100:** 653 – 669.

Wilts, A.R., D. C. Reicosky, R. R. Allmaras and C. E. Clapp. 2004. Long-Term Corn Residue Effects: Harvest Alternatives, Soil Carbon Turnover, and Root-Derived Carbon. Soil Sci. Soc. Am. J**. 68:** 1342 – 1351.

Witkamp, M., and J. V. D. Drift. 1963. Breakdown of forest litter in relation to environmental factors. Plant and soil **.15** : 295 – 311.

Wolf, B. and G. H. Snyder. 2003. 'Sustainable Soils: The place of organic matter in sustaining soils and their productivity.' (Food Products Press of the Haworth Press: New York.

Wright A.L. and K.R. Reddy. 2001. Heterotrophic microbial activity in northern Everglades wetland soils. Soil Sci.Soc. Am. J. **65**: 1856 – 1864.

Wu, L., and L.Q. Ma. 2002. Relationship between compost stability and extractable organic carbon. J. Environ. Qual. **31**:1323–1328**.**

Yang. Z. H. ; B. R. Singh ; B. K . Sitaula. 2004. Soil organic carbon fractions under different Land uses in Madrid watershed of Nepal. Communication in soil science and plant analysis. . Vol. **35**. No. 5 -6 .pp. 615- 629.

Yarpolov, A.I., O.V. Skorobogatko, S. S. Vartanov, and S. D. Varfolomeyev. 1994. Laccase properties, catalytic mechanism, and applicability. Appl. Biochem. Biotechnol. **49:** 257 – 280.

Yong Z. S. 2007. Soil carbon and nitrogen sequestration following the conversion of cropland to alfalfa forage land in northwest China. Soil and tillage Res. Vol. **92.** pp. 237 – 245.

Youn, H.D., Y.C. Hah, and S. Kang. 1995. Role of laccase in lignin degradation by white rot fungi . FEMS. Microbial. Lett. **132** : 183 – 188.

Yuri L. Z. , R. Lal and D. V.S. Resck .2005.Texture and organic carbon relations described by a profile pedotransfer function for Brazilian Cerrado soils .Geoderma. Vol. **127.** Issues 1 -2 . pp.168 – 173.

Zak, D.R., K.S. Pregitzer, P.S. Curtis, J.A. Teeri, R. Fogel, and D.L. Randlett. 1993. Elevated atmospheric CO_2 and feedback between carbon and nitrogen cycles. Plant Soil **151**:105–117.

Zech, W., and I. Kögel-Knabner. 1994. Patterns and regulation of organic matter transformation in soils: Litter decomposition and humification. p. 303–334. *In* E.-D. Schulze (ed.) Flux control in biological systems: From enzymes to populations and ecosystems. Academic Press, San Diego, CA.

Zhu B., A.K. Alva. 1993.Distribution of trace metals in some sandy soils under citrus production. Soil Sci. Soc. Am. J.; 57:350-355

Zogg, G.P.; D.R. Zak; D. B. Ringelberg; N.W. MacDonald; K.S Pregitzer. ; D.C White.1997. Compositional and functional shifts in microbial communities due to soil warming. Soil Sci. Soc. Am. J. **61**; 475 – 481.

Appendix (1) Changes in mean values of EC, Eh, pe, pH, Fe, Mn , Cu and Zn as functions of different temperatures during incubation periods in studied soils.

Parameters	Kalar soil			Arbat soil		
	Temperatures C^o			Temperatures C^o		
	5	28	45	5	28	45
EC (dS /m) at 25C^0	7.033	13.157	8.373	5.905	8.805	6.169
Eh (mv)	- 44.747	- 46.997	-33.423	- 45.556	- 49.144	-38.231
pe	- 0.756	- 0.794	- 0.565	- 0.770	- 0.830	- 0.646
pH	7.849	7.96	7.625	7.864	8.002	7.691
pe + pH	7.093	7.166	7.060	7.094	7.172	7.045
Log Fe^{+2}	-4.161	-4.426	-4.390	-4.361	-4.605	-4.452
Log Mn $^{+2}$	-4.402	-4.970	-4.825	-4.087	-4.500	-4.280
Log Cu $^{+1}$	-5.307	-5.485	-5.433	-5.280	-5.497	-5.263
Log Zn $^{+2}$	-5.511	-5.788	-5.882	-5.586	-5.833	-5.758

Appendix (2) Changes in mean values of EC, Eh, pe, pH, Fe, Mn , Cu and Zn as functions of different manure application rates during incubation periods in studied soils.

Parameters	Kalar soil			Arbat soil		
	Manure application rate g / 100g			Manure application rate g / 100g		
	0.5	1.5	2.5	0.5	1.5	2.5
EC (dS /m) at 25C^o	8.420	9.138	11.004	6.116	7.118	7.698
Eh (mv)	- 41.429	- 41.475	- 42.263	- 45.454	- 44.722	- 42.756
pe	- 0.700	- 0.701	- 0.714	- 0.768	- 0.755	- 0.722
pH	7. 804	7. 810	7. 820	7. 873	7. 860	7. 824
pe + pH	7.104	7.109	7.106	7.105	7.105	7.102
Log Fe^{+2}	-4.324	-4.314	-4.318	-4.200	-4.190	-4.119
Log Mn $^{+2}$	-4.776	-4.702	-4.591	-4.313	-4.267	-4.226
Log Cu $^{+1}$	-5.416	-5.416	-5.381	-5.387	-5.360	-5.271
Log Zn $^{+2}$	-5.830	-5.712	-5.616	-5.896	-5.667	-5.644

Appendix (3) Changes in mean values of EC, Eh, pe, pH, Fe, Mn , Cu and Zn as functions of different available water during incubation periods in studied soils.

Parameters	Kalar soil			Arbat soil		
	Available water g / 100g			Available water g / 100g		
	25	50	100	25	50	100
EC (dS /m) at $25C^o$	9.425	9.425	9.712	6.907	6.854	7.170
Eh (mv)	- 40.275	- 42.972	- 41.92	- 43.363	- 44.137	- 45.431
pe	- 0.68	- 0.726	- 0.708	- 0.732	- 0.746	- 0.767
pH	7.787	7.833	7.814	7.835	7.850	7.873
pe + pH	7.107	7.107	7.106	7.103	7.104	7.106
Log Fe^{+2}	-4.310	-4.318	-4.334	-4.459	-4.463	-4.484
Log Mn^{+2}	-4.618	-4.695	-4.741	-4.249	-4.247	-4.306
Log Cu^{+1}	-5.360	-5.319	-5.443	-5.315	-5.309	-5.390
Log Zn^{+2}	-5.663	-5.721	-5.746	-5.703	-5.724	-5.732

Appendix (4) Changes in mean values of EC, Eh, pe, pH, Fe, Mn , Cu and Zn as functions of different incubation periods in studied soils.

Parameters	Kalar soil				Arbat soil			
	Time incubated (months)							
	2	4	8	12	2	4	8	12
EC (dS/m) at $25C^o$	6.937	12.487	10.526	8.038	6.274	7.381	7.909	6.327
Eh (mv)	-36.494	-36.803	- 42.209	-51.383	- 43.979	-37.524	- 44.283	-51.457
pe	- 0.616	- 0.622	- 0.713	- 0.868	- 0.743	- 0.634	- 0.748	- 0.869
pH	7.791	7.914	7.744	7.798	7.891	7.934	7.781	7.804
pe + pH	7.175	7.292	7.031	6.930	7.148	7.300	7.033	6.935
Log Fe^{+2}	-4.126	-4.424	-4.389	-4.370	-4.418	-4.544	-4.400	-4.531
Log Mn^{+2}	-4.330	-4.876	-5.167	-4.727	-3.981	-4.287	-5.079	-4.254
Log Cu^{+1}	-5.347	-5.474	-5.572	-5.274	-5.343	-5.395	-5.436	-5.205
Log Zn^{+2}	-5.452	-5.798	-5.967	-5.728	-5.461	-5.852	-6.013	-5.722

Appendix (5) MSE values for the studied parameters at Kalar and Arbat locations.

Parameters	Values of MSE	
	Kalar	Arbat
- TSOC g /100g remaining	0.009	0.024
- C - humin g /100g	0.026	0.038
- C – HA g /100g	0.00008	0.0002
- C – FA g /100g	0.026	0.036
- Available Fe (ug / g)	6.093	3.926
- Available Mn (ug / g)	0.772	2.027
- Available Cu (ug / g)	0.016	0.043
- Available Zn (ug / g)	0.035	0.021
- pH	0.007	0.014
- EC dS m^{-1}	0.002	0.0003
- Eh mV	22.684	47.586

حکومەتی هەرێمی کوردستانی عێراق
وەزارەتی خوێندنی باڵاو توێژینەوەی زانستی
زانکـۆی سلێمانی
کۆلێژی کشتوکاڵ

دینامیکیەتی شیبوونەوەی ئۆرگانەمژادوکارتێکردنی لەشیانی هێندێتوخمەدەگمەنەکان، لە هێندێ خاکی پارێزگەی (سولەیمانیی) دا

ئەم نامەیە پێشکەشە بە ئەنجومەنی کۆلێجی کشتوکاڵ / زانکۆی سلێمانی کەبەشێکە لە پێداویستیەکانی بەدەستهێنانی پلەی دکتۆرا لە فەلسەفەی زانستی کشتوکاڵ/ خاك
(کیمیای خاك)

لە لایەن:
کەمال حەمە کەریم حەمە
(بکالۆریۆس / زانکۆی سلێمانی 1981)
(ماجستێر / زانکۆی موسڵ 1988)

بەسەرپەرشتی:
پ.ی.د. سەعید ئیسماعیل سلیڤانی

رەزبەر 2707 رەمەزان 1428 تشرینی یەکەم 2007

پوختەو دەرئە نجامەکان

مەبەست لە ئە نجامدانی ئەم توێژینەوەیە ئەو هۆکارانەن کەکار دەکەنـە سـەر شـیبوونەوەی مـژادی ئەنـدامی خـاك وەك (پلەی گەرمی جیاواز کەدەر بڕینە لە پلەی گەرمی وەرزەکانی ساڵ، ڕێژەی جیاواز لە ئاوی سود لێبینراو، وەڕێژەی جیاواز لە پەینی ئاژەڵی بەکار هاتوو) کە هەڵگیرا بوون لەناو ئامێری بە خێوکردندا (Incubator) بۆ ماوەی(2، 4، 8، 12) مانگ.

وە بۆ جێبەجێکردنی ئەم مەبەستەش دوو ناوچە هەڵبژێردران لە پارێزگای سلێمانی لە هەرێمی کوردستانی عێراقدا کـە بریتیبوون لەکەلار لە (قەڵای شێروانە) وە عەربەت لە (گوندی خەراجیان)، ئنجا نمونەی خاکەکان وەرگیران بـە قـوڵایی (صفر – 30سم). یەکەمیان بریتی بوو لەو زەوی یەی کە با میەی تیـا کرابـووە بـەجێهێڵرابوو بـۆ سـاڵی ئاینـدە، وە دووەم شوێنیشـیان پەرێزی گەنم بوو. وە ناوچەکان جیاواز بوون لە ڕووی (شانەی خاکەکانیانەوە، لەگەڵ ڕێژەی مژادی ئەندامیەوە).

تـاقیکردنەوەیەکی تـاقیگەیی بەخێو کردن ئە نجا مدرا لە بەشی خاك و ئاو لە کۆلیجی کشتوکاڵ لـە زانکـۆی سـلێمانی لـە هـەرێمی کوردستانی عێراقدا لە ساڵانی (2003-2004) دا بەمەبەستی پێوانی جەم و جۆڵی کاربۆنی ئەندامی خاك. وە بۆ ئەم مەبەسـتەش تـاقیکردنەوەیەکی هەمەکارییەکان (Factorial) کە پێك هاتبوو لە پێنج کار کەر (factor) لەسەر شێوازی نەخشـەی (CRD) ئە نجام درا کە دووجار دووبارە کرابۆوە بۆ هە ر ئا ستێکی بە کا ر ها توو .

وەکار کەرەکانیش بریتی بوون لە :-

- ناوچەکان (کەلارو عەربەت).
- پلەی گەرمیەکان (5،28،45) س°.
- تێکڕایی پەینی ئاژەڵی بەکار هاتوو (0.5، 1.5، 2.5) گم/100گم.
- ئاوی سود لێبینراو (25، 50،100) گم/100گم.
- ماوەی بەخێوکردن (2، 4، 8، 12) مانگ.

(200 گم) لە هەریەکە لە خاکەکان وەرگیران وەخرانە شووشەیەکی روونی بچـووکەوە کـە سـەرەکەی بەتـەواوەتی تونـد کرابوو کە بەرزی یەکەی (13سـم) وەتیرەکـەی (6 سـم)بوو، ئینجـا پـەینی ئـاژەڵی لـە جـۆری مـەڕ بـە ڕێـژەی (0.5، 1.5، 2.5) گـم/100گـم. تێکـەڵ کـرا لـە لەگـەڵ گڵەکـە (200 غـم) ، وەئاویـان تێکـڕا هـەتا ڕێـژەکانیان گەیەنرایـە (25، 50، 100) گم/100گم لەئاوی سود لێبینراو وە دوای ئەمانە خرانە ئامێری بە خێوکردنەوە لەسەر پلەی (5،28،45) س° دانـران بـۆ مـاوەی (2، 4، 8، 12) مانگ. سێ ماوە بەکار هێنرا کە بریتی بوون لە (4، 8، 12) مانگ بۆ پێوانی ترشـی هیومیـك و فولڤیـك لەگـەڵ هیومیندا. لەیەك پلەی گەرمی دا کەبریتی بوو لە (28س°) بـەڵام (Eh, PH, EC,TOC لەگـەڵ available Fe, Mn, Cu, Zn) پێوران لە ماوەی (2، 4، 8، 12) مانگدا لەسێ پلەی گەرمی جیاوازدا کە بریتـی بـوون لـە (5،28،45) س° وە یـەك

نموونەش وەرگیرا بۆ بەراوردکردن لە پلەی گەرمی 28س°، 50 گم/100گم لەئاوی سود لێبینرابە بێ بە کارهێنانی پەینی ئاژەڵی بەکارهاتوو بۆ هەریەك لە چوار ماوەکە. چەند مۆدێلێکی ماتماتیکی جیاواز بەکار هێنران بۆ باسکردنی پرۆسەی کاربۆنە بەکانزابووەکان.(Carbon mineralization) وە مۆدێلەکانیش بریتی بوون : Zero – Order, First – Order , Hyperbolic , Parabolic diffusion , Modified Elovich , , Power function , Logarithmic function.

ئەو مۆدێلەی کەوا نرخی هاوکۆڵکەی دەست نیشانکردنەکەی (Determination coefficient) بەرزترین نرخ بوو وەنرخی هەڵەی پێوانەییەکەی (Standard error) نزمترین نرخ بوو دانرا بە باشترین مۆدێل بۆ باسکردنی پرۆسەی کاربۆنە بەکانزابووەکان. وەهێڵکاری توانای توانەوەی (Solubility diagram) بەکار هێنرا بۆ خەملاندنی توانەوەی هەریەك لە (Fe, Mn, Cu, Zn) وەتوێژینەوەکە ئەم ئە نجامانەی خوارەوەی بەدەست هێنا :-

تێکڕای بەهای (TSOC remaining گم/100گم) بۆ هەریەك لە ناوچەی کەلارو عەربەت بریتی بوو لە (1.67،1.81) گم/100گم یەك بەدوای یەك لە (5س°) یدا, بەڵام بەهاکانیان لەسەر (28,45س°)بریتیبوون لە (1.65 ،1.60) لەگەڵ (1.04 ، 1.33) گم/100گم بۆ هەردووناوچەکە یەك بەدوای یەکدا. وەمامەڵەی (5س°) کاریگەرییەکی بەهاداری (معنوی) هەبوو لەسەر (TSOC remaining گم/100گم) بەڵام تێکڕای بەهای (TSOC remaining گم/100گم) لەمامەڵەی 25 گم/100گم ئاوی سود لێبینراودا بریتی بووە لە (1.57 گم/100گم) بۆ هەردوو ناوچەی کەلارو عەربەت و وەهەروەها بەهاکانیان بریتی بوون لە (1.51،1.52) گم/100گم لەگەڵ (1.43، 1.51) گم/100گم بۆ هەریەکە لە (50، 100) گم/100گم لە ئاوی سود لێبینراو لە هەردوو ناوچەکەدا یەك بەدوای یەکدا. لەناوچەی کەلار دا مامەڵەی 25 گم/100گم لە ئاوی سود لێبینراو کاریگەرییەکی بەهاداری هەبووە لەسەر (TSOC remaining گم/100گم) بەبەراورد کردنی بەهەردوو مامەڵەی (50، 100) گم/100گم، بەڵام لە ناوچەی عەربەتدا مامەڵەی 100 گم/100گم ئاوی سود لێبینراو کاریگەرییەکی بەهاداری هەبوو لەسەر (TSOC remaining گم/100گم). وەمامەڵەی 2.5 گم/100گم لە تێکڕای پەینی ئاژەڵی بەکار هاتوو کاریگەرییەکی بەهاداری هەبووە لەسەر (TSOC remaining گم/100گم) بەبەراورد کردنی بە مامەڵەکانی تر (0.5، 1.5) گم/100گم لە هەردوو ناوچەکەدا. وە مامەڵەی (4 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر (TSOC remaining گم/100گم) بە هەریەك لە مامەڵەکانی (2،8،12) مانگ بۆ هەردوو ناوچەکە.

وەمۆدێلی (Hyperbolic) باشترین مۆدێل بوو بەبەراورد کردنی بەمۆدێلەکانی تر بۆ باسکردنی پرۆسەی (SOC mineralization) چونکە بەرزترین بەهاکانی R^2 داوە کە بریتی بوون لە (0.87، 0.76) وەنزمترین بەهاکانی (SE) داوە کە بریتی بوون لە (0.08، 0.10) بۆ هەریەك لە ناوچەی کەلارو عەربەت یەك بەدوای یەکدا.

ئەوەی شایانی باسە بووترێت بەبەرزبوونەوەی بەهای تێکڕای پەینی ئاژەڵی بەکار هاتوو بەهای نیوە تەمەن (half life) کەم بووەتەوە چونکە بەهای نیوە تەمەن بەندە لەسەر تێکڕای شی بوونەوەی ماددەی ئەندامی خاك. بەڵام هیچ گۆڕانکاریەك نەبینراوە لە بەهای نیوە تەمەندا لەهەریەك لە (25، 50،100) گم/100گم. لە ئاوی سود لێبینراودا لەهەردوو ناوچەکەدا.

بەڵام بەرزترین بەهای نیوە تەمەن کە بریتی بووە لە (16.02، 17.14) مانگ تۆمار کراوە لە (28، 45) س° دا، وەنزمترین بەهاش بریتی بووە (13.66، 14) مانگ کە تۆمار کراوە لە (28، 45) س° دا، وەنزمترین بەهاش بریتی بووە (13.66، 14) مانگ کە تۆمار کراوە لە (28، 45س°) دا لەهەردوو ناوچەی کەلارو عەربەتا یەك بەدوای یەکدا. بەهای تێکڕای شی بوونەوە (K) زیادی کردووە بەزیادبوونی تێکڕای پەینی ئاژەڵی بەکار هاتوو، ئاوی سودلێبینراو بێجگە لە 100 گم/100گم ئاوی سود لێبینراو نەبێت لە هەردوو ناوچەکەدا. بەڵام بەهای (K) کەمی کردووە لەگەڵ زیاد بوونی پلەی گەرمیدا، وەبەهای (Q_{10}) زیادی کردووە بە زیادبوونی تێکڕای پەینی ئاژەڵی بەکار هاتوو بەڵام بەجێگیری ماوەتەوە لەگەڵ ئاوی سود لێبینراوە جیاوازەکاندا.

لەئە نجامدا دەرکەوت شیبوونەوەی SOC لە خاکدا زۆر هەستیارە بەرامبەر بە گۆڕانکاری لە پلەی گەرمی یە کا ندا.

بەڵام بەگوێرەی C-humim گم/100گم ، مامەڵەی 2.5 گم/100گم لە تێکڕای پەینی ئاژەڵی بەکارهاتوو کاریگەرییەکی بەهاداری هەبووە لەسەر C-humin گم/100گم ئەگەر بەراوردبکرێت بەهەریەك لە مامەڵەی (0.5، 1.5) گم/100گم بێجگە لە بەیێکداچوونی (تداخل) مامەڵەی 1.5 گم/100گم لەگەڵ (8 مانگ) لەناوچەی کەلاردا. بەڵام لە ناوچەی عەربەتدا ، بەیێکداچوونی مامەڵەی 2.5 گم/100گم لە تێکڕای پەینی ئاژەڵی بەکارهاتوو لەگەڵ (4،8 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر C-humin گم/100گم ئەگەر بەراوردبکرێت بەهەریەك لە مامەڵەی (0.5، 1.5) گم/100گم بێجگە لە بەیێکداچوونی مامەڵەی 1.5 گم/100گم لەگەڵ (4 مانگ)دا نە بێت. لەلایەکی ترەوە هیچ کاریگەرییەکی بەهادار نەبینراوە لە نێوان ئاوە سود لێبینراوە جیاوەزەکاندا لەسەر C-hmin گم/100گم لە هەردوو ناوچەکەدا. بەڵام لەناوچەی کەلاردا بەیێکداچوونی مامەڵەی ماوەی (8 مانگ) لەگەڵ(28 س°) کاریگەرییەکی بەهاداری هەبووە لەسەر (C-humin گم/100گم) بە بەراوردکردنی بەمامەڵەکانی (4، 12) مانگەوە. . بەڵام بەیێکداچوونی مامەڵەی ماوەی (4 مانگ) لەگەڵ(28 س°) کاریگەرییەکی بەهاداری هەبووە لەسەر (C-humin گم/100گم) بە بەراوردکردنی بەمامەڵەکانی (8، 12) مانگەوە لە ناوچەی عەربەتدا.

وەئە نجامەکان دەریان خست کە مامەڵەی تێکڕای پەینی ئاژەڵی بەکار هاتوو بەڕێژەی 2.5 گم/100گم کاریگەرییەکی بەهاداری هەبووە لەسەر C- HA گم/100گم بەبەراوردکردنی بەمامەڵەکانی (25، 50) گم/100گم وە لەناوچەی کەلاردا. بەڵام ئاوە سود لێبینراوە جیاوازەکان لە ناوچەی عەربەتدا هیچ کاریگەرییەکی بەهاداری نەبووە لەسەر C-HA گم/100گم .

وەئە نجامەکان وایان پیشان داوە، مامەڵەی (8 مانگ) لەسەر C- HA گم/100گم کاریگەرییەکی بەهاداری هەبووە بەبەراوردکردنی بەدوو مامەڵەکەی ترەوە لە ناوچەی کەلاردا. بەڵام لە ناوچەی عەربەتدا هیچ جیاوازیەك نەبینراوە لە نێوان ماوەکاندا.

وە لەناوچەی کەلاردا مامەڵەی پەینی ئاژەڵی بەکارهاتوو (0.5، 1.5) گم/100گم کاریگەریەکی بەهاداریان هەبووە لەسەر C-FA گم/100گم بەبەراودکردنیان بەمامەڵەی 2.5غم /100 غم وە . بەڵام لە ناوچەی عەربەتدا مامەڵەی 1.5 گم/100گم کاریگەرییەکی بەهاداری هەبووە لەسەر C-FA گم/100گم بەبەراوردکردنی بەمامەڵەکانی ترەوە . وە مامەڵەکانی(25، 50) غم / 100غم لەئاوی سود لێبینراو کاریگەرییەکی بەهاداری هەبووە لەسەر C-FA گم/100گم لەناوچەی کەلاردا بەڵام هیچ جیاوازییەك نەبینراوە لە نێوان مامەڵەی ئاوە سود لێبینراوە جیاوازەکاندا لەناوچەی عەربەتدا، وە مامەڵەی (4 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر C-FA گم/100گم لەناوچەی کەلاردا, بەڵام لەناوچەی عەربەتدا مامەڵەی (12 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر (C- FA گم/100گم). وەئەنجامەکان وایان دەرخست کە بەهای FA زیاتر بووە لەبەهای HA ئەمە ش پێش بینی دەکرا بەهۆی زیادبوونی تێکڕای شی بوونەوەی مژادی ئەندامی خاکەوە.

بەشێوەیەکی گشتی بەرزترین بەهای HR گم/100گم بینرا لە مامەڵەی (4 مانگ) دا بۆ هەموو خەسڵەتە بەکارهاتووەکانی توێژینەوەکە.

وەبەرزونزمی ڕویداوە لە مانگی (8،12) دا سەبارەت بە (HR گم/100گم) لە ناوچەی عەربەتدا . تێکڕای (Humification) بەڵگەیە بۆ ڕودانی پرۆسەی Humification هەتا نرخی (HR گم/100گم) بەرزبێت مانای پرۆسەی Humification زۆربووە.

وەئەنجامەکان پیشانیاندا کەوا مامەڵەی (5س°) لەسەر ئاسنی(Fe) سودلێبینراو کاریگەرییەکی بەهاداری هەبووە بەراوردکردنی بەمامەڵەکان (28،45) س° لەهەردووناوچەی کەلارو عەربەتدا. لەلایەکی ترەوە وا دەرکەوت مامەڵەی تێکڕای پەینی ئاژەڵی بەکار هاتوو بەڕێژەی 2.5 گم/100گم کاریکەرییەکی بەهاداری هەبووە لەسەر Fe سودلێبینراو لەناوچەی کەلارو عەربەتدا. بەڵام هیچ جیاوازییەك نەبینراوە لە نێوان مامەڵەی ئاوە سودلێبینراوە جیاوازەکاندا لەسەر Fe سود لێبینراو لەهەردوو ناوچەکەدا. وەمامەڵەی (2 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر Fe سود لێبینراو بەبەراوردکردنی لەگەڵ مامەڵەکانی تردا لە ناوچەی کەلاردا بەڵام لە ناوچەی عەربەتدا مامەڵەی (8 مانگ) کاریگەرییەکی بەهاداری هەبووە لەسەر Fe سودلێبینراوە.

بەڵام لەڕووی توخمی مەنگەنیزەوە, مامەڵەی (5س°) کاریگەریەکی بەهاداری هەبووە لەسەر Mn سودلێبینراو لەناوچەی کەلارو عەربەتدا بەبەراوردکردنی بەهەریەك لە مامەڵەکانی (28،45) س°. وەمامەڵەی تێکڕای پەینی ئاژەڵی بەکارهاتوو بەڕێژەی 2.5 گم/100گم کاریگەرییەکی بەهاداری هەبووە لەسەر Mn سود لێبینراو بەبەراوردکردنی بەمامەڵەکانی ترەوە لەهەردوو ناوچەی کەلارو عەربەتدا. وەمامەڵەی 25 گم/100گم لەئاوی سودلێبینراو کاریگەریەکی بەهاداری هەبووە لەسەر Mn سودلێبینراو لە ناوچەی کەلاردا بەڵام لەناوچەی عەربەت مامەڵەی (25، 50) گم/100گم کاریگەری بەهاداری هەبووە بێجگەلە

مامەڵەی 100 گم/100گم لەسەر Mn سودلێبینراو نە بێت .وە مامەڵەی (2 مانگ) کاریگەری یەکی بەهاداری هەبووە لەسەر Mn سودلێبینراو ئەگەر بەراوردبکرێت بە مامەڵەکانی (4، 8، 12) مانگەوە.

بەڵام بەگوێرەی توخمی (Cu) وە مامەڵەی (5 س°) کاریگەریەکی بەهاداری هەبووە لەسەر (Cu) سودلێبینراو ئەگەر بەراوردبکرێت بەمامەڵەکانی (28،45) س° لە ناوچەی کەلاردا بەڵام لەناوچەی عەربەتدا مامەڵەی (45 س°) کاریگەرییەکی بەهاداری هەبووە لەسەر Cu سودلێبینراو بێجگە لە مامەڵەی (5 س°) ئەگەر بەراورد بکرێت بەمامەڵەکانی تر. لەلایەکی ترەوە مامەڵەی پەینی ئاژەڵی بەکار هاتوو بەڕێژەی 2.5 گم/100گم کاریگەرییەکی بەهادا ری هەبووە لەسەر (Cu) ی سودلێبینراو ئەگەر بەراورد بکرێت بەمامەڵەکانی تر لەهەردوو ناوچەی کەلارو عەربەتا. وە هەروەها مامەڵەی 25 گم/100گم لە ئاوی سودلێبینراو کاریگەرییەکی بەهاداری هەبووە لەسەر Cu سودلێبینراو ئەگەر بەراور دبکرێت بەمامەڵەکانی تر لەناوچەی کەلاردا. بەڵام لەناوچەی عەربەتدا هیچ جیاوازییەك نەبینراوە لە نێوان مامەڵەی ئاوە سود لێبینراوە جیاوازەکاندا. وەمامەڵەی (12 مانگ) کاریگەریەکی بەهاداری هەبووە لەسەر Cu سودلێبینراو ئەگەر بەراورد بکرێت بەمامەڵەکانی (2، 4، 8) مانگە وە .

بەڵام بەگوێرەی توخمی (Zn) وە مامەڵەی (5 س°) کاریگەرییەکی بەهاداری هەبووە لەسەر (Zn) یی سودلێبینراو ئەگەر بەراوردبکرێت بەمامەڵەکانی تر لە هەردوو ناوچەی کەلارو عەربەتدا. لەلایەکی تروە مامەڵەی تێکڕایی پەینی ئاژەڵی بەکار هاتوو بەڕێژەی 2.5 گم/100گم کاریگەرییەکی بەهاداری هەبووە لەسەر (Zn) ی سودلێبینراو ئەگەر بەراوردبکرێت بەمامەڵەکانی تر بەڵام لە ناوچەی عەربەتدا. مامەڵەی 2.5 گم/100گم کاریگەرییەکی بەهاداری هەبووە لەسەر (Zn) ی سودلێبینراو بێجگە لە مامەڵەی 1.5 گم/100گم ئەگەر بەراوردبکرێت بەمامەڵەکانی ترە وە . وەمامەڵەی 25 گم/100گم ئاوی سودلێبینراو کاریگەرییەکی بەهاداری هەبووە لەسەر (Zn) ی سود لێبینراو ئەگەر بەراوردبکرێت بەمامەڵەکانی تر لە ناوچەی کەلار بەڵام لەناوچەی عەربەت هیچ جیاوازییەك نەبینراوە لە نێوان مامەڵەکانی ئاوە سودلێبینراوە جیاوازەکاندا لەسەر (Zn) ی سود لێبینراو. وەهەروەها مامەڵەی (2 مانگ) کاریگەری یەکی بەهاداری هەبووە لەسەر (Zn) ی سودلێبینراو ئەگەر بەراورد بکرێت بەمامەڵەکانی تر لە هەردوو ناوچەی کەلارو عەربەتدا.

لەلایەکی ترەوە توانای توانەوەی Fe لە حالەتی ژێرتێربووندایە (Undersaturated) لەگەڵ کانزای نیشتووی ئاسندا (Fresh precipitate) یا ن لە حالە تی سە روو تێربووندایە (Supersaturated) لەگەڵ کانزای (Magnetite) لە گە ڵ هە موو ما مە ڵە جیاوازە کاندا لە هەردوو ناوچەی کەلارو عەربەتدا.

بەڵام توانای توانەوەی Mn لە حالە تی ژێر تێربووندایە (Undersaturated) لەگەڵ هەموو ما مە ڵە بەکارهاتووەکاندا لەگەڵ کانزای $MnCO_3$ بێجگە لە ما مەڵەی ئا وی سوود لێبینراودا نەبێت بە ڕ ێژە ی (25، 50 ، 100)گم /100گم چونکەتوانای توانەوەی Mn دە کە وێتە نێوان (MnOOH) – γ وە $MnCO_3$ لەناوچەی کەلاردا.

بەلاّم لە ناوچەی عەربەتدا توانای توانەوەی Mn لەحالە تی ژێر تێربووندایە یا ن هاوسە نگیدایە لەگەڵ هەموو ما مە ڵە بەکار هاتووەکاندا لەگەڵ کانزای $MnCO_3$ دا.

لەلایەکی ترەوە توانای توانەوەی Cu سەرو تێربوونە لەگەڵ هەموو ما مەڵە بەکارهاتووەکاندا لەگەڵ کانزای (Soil- Cu) دا لە هەردوو ناوچەی کەلارو عەربەتدا.

لەکۆتایدا هەردووکانزای ($ZnSiO_4$, ZnO) کۆنترۆڵ دەکەن بەسەر توانای توانەوەی (Zn) دا لە ناوچەی کەلارو عەربەتدا.

حكومة أقليم كوردستان العراق
وزارة التعليم العالي والبحث العلمي
جامعة السليمانية
كلية الزراعة

حركية تحلل المادة العضوية و تأثيرها في جاهزية بعض العناصر الصغرى في بعض ترب محافظة السليمانية

أطروحة
مقدمة الى مجلس كلية الزراعة في جامعة السليمانية كجزء
من متطلبات درجة دكتوراه فلسفة في العلوم الزراعية / التربة
(كيمياء التربة)

من قبل:
كمال حمه كريم حمه
(بكالوريوس / جامعة السليمانية 1981)
(ماجستير / جامعة الموصل 1988)

بإشراف:
أ.م.د. سعيد إسماعيل السليڤانى

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المستخلص والاستنتاجات

استهدف البحث دراسة العوامل المؤثرة في تحلل المادة العضوية (SOM) عند الدرجات الحرارية المختلفة والتي تمثل درجات الحرارة في مواسم السنة الرئيسية ، نسبة الماء الجاهز ومعدلات مختلفة من السماد الحيواني(مخلفات الاغنام) عند فترات زمنية مختلفة تراوحت بين (2،4،8، 12)شهراً وتم الاختيار موقعين مختلفين في محافظة السليمانية اقليم كردستان العراق لاجراء البحث وتم اخذ النماذج من الترب السطحية بعمق (صفر – 30 سم) لكل من موقعي كلار (قلعة شيروانة) وعربت (قرية خراجيان) والموقع الاول مزروع بمحصول الباميا(Okra) اما الموقع الثاني مستغل لزراعة محاصيل الحبوب وقد تباينت تربة كلا الموقعين في النسجة وكاربونات الكالسيوم بينما كانتا متماثلتان في محتواهما من المادة العضوية .

لدراسة حركية تحلل المادة العضوية تم الاستعانة بتجارب الحضن داخل مختبرات قسم علوم التربة والمياه في كلية الزراعة / جامعة السليمانية وباستخدام التجربة العاملية (Factorial experiment) متضمنة خمسة عوامل وباستخدام التصميم العشوائي الكامل (CRD) بمعدل مكررين لكل معاملة .

من اجل دراسة تأثير العوامل التالية :-

1- تأثير التربة (كلار وعربت)
2- نسب الماء الجاهز (25 ،50 ، 100) غم /100غم
3- درجات الحرارة (5 ،28 ، 45) $م^o$
4- معدلات الاضافة للسماد الحيواني (0.5 ،1.5 ، 2.5) غم/100 غم
5- فترات التحضين(2،4،8، 12) شهراً.

تم اخذ نماذج ممثلة لكل معاملة بواقع 200 غم في قناني زجاجية بارتفاع 12 سم وبقطر 6 سم وتم معاملتها بعوامل الدراسة المذكورة انفاً وقد تم تقدير كل من

- نسبة الكاربون العضوي TOCغم /100غم .
- درجة تفاعل التربة . pH
- الجهد الكهربائي Eh .
- تقدير العناصر الغذائية الصغرى الجاهزة (Fe, Mn , Cu , Zn)

تم اجراء التقديرات المذكورة عند الفترات الزمنية المختلفة(2،4،8، 12) شهراً من التحضين واستخدمت عينات التربة الغير معاملة بالمادة العضوية والتي تم حضنها على درجة 28 $م^o$ ونسبة 50غم /100غم من الماء الجاهز لتمثل عينة المقارنة .

كما تم تقدير كل من حامض الدباليك (HA) وحامض الفولفيك (FA) ومن ثم الدبالين في العينات التي تم حضنها على درجة 28 $م^o$.

ومن اجل دراسة حركيات تحلل المادة العضوية والمعدنة الكاربون العضوي استخدم نماذج رياضية مختلفة Models تضمنت:

- معادلة الرتبة صفر **Zero – order equation**
- معادلة الرتبة الاولى **first – order equation**
- معادلة الرتبة الثانية **Second – order equation**
- معادلة القطع الزائد **Hyperbolic equation**
- معادلة القطع المكافي الانتشاري **Parabolic diffusion equation**
- معادلة اولفيج المحورة **Modified Elovich equation**
- معادلة الدالة الاسية **Power function equation**
- معادلة الدالة اللوغارتمية **Logarithmic function equation**

وتم اختبار الموديل الاكفاء استنادا على التحليل الاحصائي من خلال اكبر قيمة لمعامل التقدير R^2 واقل قيمة لخطأ القياسي SE .

وللتنبؤ بجاهزية العناصر الصغرى المدروسة من خلال معيار الذوبانية تم الاستعانة بمخططات الذوبانية Solubility diagrams

وقد امكن التوصل الى النتائج التالية .

- **تراوحت نسب المعدل العام للكاربون العضوي الكلي المتبقي TSOCغم/100غم بين(1.81،1.67) غم /100غم في كل من موقعي كلار وعربت على التوالي ، بينما كانت القيم (1.60 ، 1.66) و (1.33، 1.04) تمثل المعدل العام عند درجة الحرارة 45 م° في كلا الموقعين وعلى التوالي .**
- **وقد كانت لدرجة الحرارة 5 م° تأثيراً معنوياً في قيم المتبقي TSOC غم/100غم عند درجة معنوية 0.01 مقارنة بكل من درجات الحرارة (28 ، 45) م°.وبالنسبة لتأثير المحتوى الرطوبي فقد بلغت اعلى قيمة لل المتبقي TSOC غم/100غم(7 1.5 غم /100غم) في كلا الموقعين كلار وعربت عند نسبة رطوبة جاهزة 25غم/100غم . ممايشير الى انخفاض الرطوبة الى 25غم/100غم من الماء الجاهز في موقع كلار يرافقها زيادة معنوية عالية في قيم المتبقيTSOC غم/100غم مقارنة ب (50 ، 100) غم/100غم من الماء الجاهز، كما اظهرت اضافة السماد بنسبة 2.5غم/100غم تفوقا معنوياً عالياً على النسب (0.5 ، 1.5) غم/100غم من حيث قيم المتبقي TSOC غم/100غم مشيرة الى ان اعلى القيم تم الحصول عليها عند فترة الحضن لمدة 4 اشهر ولكلا الموقعين.**
- **اشارت نتائج التحليل الاحصائي باستخدام التطابق الامثل ان معادلة القطع الزائد Hyperbolic equation هي الافضل في وصف معدنة الكاربون العضوي في تربة من خلال اعلى القيم لمعامل التقدير R^2 واقل القيم لخطأ القياسي SE.**

  ولوحظ ان زيادة معدل اضافة مخلفات الاغنام تؤدي الى تقليل قيمة نصف العمر (Half life's) ووجد بانها تعتمد على معدل التحلل بينما لم تلاحظ فروقات واضحة في قيم نصف العمر عند النسب المختلفة من الماء الجاهز وفي كلا الموقعين . واستنتج بان تحلل الكاربون العضوي في التربة SOC يظهر حساسية واضحة تجاه التغيرات في درجات الحرارة من خلال استخراج قيم Q_{10}.

- وفيما يخص التحلل الجزئي لمادة العضوية فوجدبأن نسبة كاربون الدبالين C-Humin غم/100غم فيها تمثل قيم معنوية عالية عند اضافة السماد بنسة 2.5 غم/100غم مقارنة بالنسبتين(0.5 ، 1.5) غم/100غم في موقع كلار بينما تم الحصول على قيم عالية في موقع عربت عند نفس الاضافة 2.5 غم/100غم وعند فترات الحضن 4 ، 8 اشهر ، بينما لم يلاحظ اي تأثير معنوي لنسب الماء الجاهز في الكاربون الدبالين C-Humin غم/100غم ولكلا الموقعين .

وفيما يتعلق بتأثير فترات الحضن مع الدرجات الحرارة لوحظ الحصول على القيم العالية لكاربون الدبالين عند فترة الحضن 8 اشهر وعند درجة حرارة 28 م^o مقارنة بالفترات 4 ، 12 شهراً في موقع كلار .

مع ملاحظة عدم وجود فروق معنوية بين فترتي الحضن الاخيرتين في موقع عربت .

تبين النتائج بأن تأثير معدل اضافة السماد الحيواني بنسبة 2.5 غم/100غم كان لها تأثير معنوي عالي في زيادة نسبة كاربون حامض الدباليك C-Humic غم/100غم مقارنة بالمعدلين الاخيرين من الاضافة وفي كلا الموقعين كما أثر الماء الجاهز بنسبة 100 غم /100غم في نسبة كاربون الحامض الدباليك عند موقع كلار مقارنة بالنسبتين(25 ، 50) غم /100 غم من الماء الجاهز اما في موقع عربت فلم يلاحظ اي اختلافات معنوية في جميع النتائج المتحصل عليها عند النسب المختلفة من الماء الجاهز . كما اشارت النتائج الى ان فترة الحضن لمدة 8 اشهر في موقع كلار كانت لها تأثير معنوي عالي في زيادة نسبة الكاربون في حامض الدباليك وبالعكسه لم يلاحظ ايه اختلافات معنوية عند الفترات الثلاثة في موقع عربت .

وجد بان النسبتين (0.5 ، 1.5) غم/100 غم من معدلات اضافة السماد الحيواني كانتا اكثر تأثيراً في نسبة كاربون حامض الفولفيك C-Fulvic غم/100 غم مقارنة بنسبة 2.5 غم/100 غم من الاضافة ، مع ملاحظة ان النسبة 1.5 غم/100 غم كانت ذو تأثير معنوي عالي عند مستوي احتمالية 0.01 وذلك في موقع عربت .

وفيما يخص تأثير الماء الجاهز وجد بان 25 غم/100 غم من الماء الجاهز تفوقت معنوياً على النسبة 100 غم/100 غم من الماء الجاهز في التأثير على نسبة كاربون حامض الفولفيك عند موقع كلار بينما لم يلاحظ اية اختلافات معنوية بين النسب الثلاثه للماء الجاهز في تأثيرها على نسبة كاربون حامض الفولفيك عند موقع عربت .

وفيما يخص تأثير الفترات الزمنية وجد بان فترة الحضن لمدة 4 اشهر كانت ذو تاثير معنوي عالي في كاربون حامض فولفيك في موقع كلار وبعكس ذلك وجد ان فترات الحضن الطويله نسبياً (12) شهراً كان لها تأثير معنوي عالي في زيادة كاربون حامض الفولفيك كما تبين من نتائج ان الزيادة الحاصلة في حامض الفولفيك يرافقها زيادة مماثلة في حامض الهيوميك وكان ذلك متوقعاً بسبب ازدياد معدل التحلل . ان اعلى القيم لمعدل التدبل (HRغم/100 غم) كانت عند الفترة 4 اشهر من الحضن ولجميع المعاملات التي تم دراستها مع ملاحظة ان هناك تذبذب في معدلات التحلل عند فترتي الحضن (8 ، 12) شهراً وتستخدم معدلات التدبل (HRغم/100 غم) كمؤشر او دليل على سرعة حدوث عملية التدبل او تكون الدبال Humification .

* اظهرت النتائج بان لدرجات الحرارة تأثيراً واضحاً في تحلل المادة العضوية وجاهزية العناصر الصغرى من خلال انطلاق العناصر الغذائية الصغرى اثناء التحلل او من خلال تفاعلاتها مع نواتج تحلل المادة العضوية . حيث ان درجة الحرارة الواطئة نسبياً 5 م^o لهاتأثير معنوي عالي في زيادة قيم الحديد الجاهز

مقارنة بدرجتي الحرارة(28 ، 45) م^o وفي كلا الموقعين . كما ان الاضافة 2.5 غم/100 غم من معدل السماد العضوي كانت لها تأثير معنوي عالي ($P \geq 0.01$) في زيادة الحديد الجاهز مقارنة بالاضافة 0.5 غم/100 غم في موقع كلار والنسبتين (0.5 ، 1.5) غم/100 غم من الاضافة في موقع عربت.

وفيما يتعلق بتأثير الماء الجاهز على الحديد الجاهز فلم تسجل اية اختلافات معنوية في كلا الموقعين ، ومن ناحية اخرى وجد بان فترات الحضن القصيرة لمدة شهرين كانت لها تأثير معنوي عالي في الحديد الجاهز مقارنة بالفترات الثلاثة اخرى (4 ، 8 ، 12) شهراً في موقع كلار ، بينما كانت للفترة 8 اشهر التأثير المعنوي العالي ($P \geq 0.01$) مقارنة ببقية الفترات في موقع عربت .

وفيما يتعلق بتأثير الحرارة في المنغنيز الجاهز وجد بأن الدرجة الحرارية المنخفضة 5 م^{o}كانت ذات تأثير معنوي عالي مقارنة بالدرجتين (28 ، 45) م^o خلال فترات الحضن المختلفة وفي كلا الموقعين كما ان نسبة 2.5 غم/100 غم من معدل الاضافة لها تأثير معنوي عالي في قيم المنغنيز الجاهز خلال فترات الحضن المختلفة ولكلا الموقعين على التوالي .

ان نسبة 25غم/100 غم من الماء الجاهز له تأثير معنوي عالي في قيم المنغنيز الجاهز مقارنة ببقية النسب في موقع كلار ، وفي موقع عربت وجد ان النسبة(25 ،50) غم/100 غم من الماء الجاهز لهما تأثيراً معنوياً عالياً في زيادة المنغنيز الجاهز مقارنة بالنسبة 25 غم/100 غم من الماء الجاهز .

وفيمايتعلق بالنحاس الجاهز تبين ان درجة الحرارة المنخفضة أيضا 5 م^o ذي تأثير معنوي عالي في موقع كلار، بينما في موقع عربت وجد بأن الحرارة العالية45 م^o كانت لها تأثير معنوي عالي مقارنة ببقية الدرجات الحرارية عدا الدرجة 5 م^o.

كما ان اضافة السماد بنسبة 2.5غم / 100غم يرافقها زيادة في قيم النحاس الجاهز مقارنة بالنسبتين (0.5 ، 1.5) غم / 100غم خلال فترات الحضن المختلفة ولكلا الموقعين ، اضافة الى ان 25 غم/100 غم من الماء الجاهز ذي تأثير معنوي عالي على النحاس الجاهزفي موقع كلار في حين لم يلاحظ اية اختلافات معنوية في قيم النحاس الجاهز عند نسب الماء المختلفة عند موقع عربت .

وكذلك تبين بان فترات الحضن الطويلة نسبياً (12) شهراً لها تأثير معنوي عالي ($P \geq 0.01$) في قيم النحاس الجاهز مقارنة ببقية فترات الحضن وفي كلا الموقعين .

كما استدلت النتائج بان درجة الحرارة المنخفضة نسبيا 5 م^o ذات تأثير معنوي عالي في زيادة جاهزية الخارصين مقارنة ببقية الدرجات الحرارية في كلا الموقعين ، كما ان معدلات الاضافة 2.5 غم/100 غم من السماد الحيواني لها تأثير معنوي عالي ($P \geq 0.01$) في قيم عنصر الخارصين مقارنة ببقية النسب (0.5 ، 1.5) غم/100 غم من الاضافة في موقع كلار .

كما تبين بان 25 غم/ 100 غم من الماء الجاهز لها تأثير معنوي عالي في قيم الخارصين الجاهز مقارنة بالنسب (50 ، 100) غم/100 غم من الماء الجاهز في موقع كلارمع ملاحظة عدم وجود ايه اختلافات معنوية بين قيم الخارصين الجاهز عند نسب الماء الجاهز المختلفة في موقع عربت وكذلك اظهرت النتائج بان فترة الحضن القصيرة نسبياً شهران لها تأثير معنوي عالي ($P \geq 0.01$) في قيم الخارصين الجاهز مقارنة ببقية فترات الحضن ولكلا الموقعين .

ان ذوبانية الحديد استقرت عند حالة تحت التشبع بمعدن الحديد المترسب حديثاً Fresh precipitate او انها تمثل حالة فوق التشبع بمعدن Magnetite وعند المعاملات المختلفة .

بينما لوحظ بان ذوبانية المنغنيز تقع عند حالة تحت التشبع بمعدن كاربونات المنغنيز $MnCO_3$ عند جميع المعاملات المدروسة باستثناء معاملة الماء الجاهز وبنسبة (25 ، 50 ، 100) غم/100 غم من الماء الجاهز حيث انها استقرت بين كل من معدني كاربونات المنغنيز $MnCO_3$ ومعدن المنكانايت γ-manganite (MnOOH) عند موقع كلار في حين اشارت نتائج الذوبانية للمنغنيز في موقع عربت بان معدن $MnCO_3$ هو المعدن المتحكم بالذوبانية وهي تمثل اما حالة الاستقرار او تحت التشبع بالمعدن المذكور .وفيما يتعلق بذوبانية عنصر النحاس فانها تستقر عند حالة فوق التشبع بنحاس التربة Soil-Cu في جميع المعاملات المذكورة سابقاً ولكلا الموقعين فيما وجد بأن المعدنين ZnO, Zn_2SiO_4 يتحكمان باذابة عنصر الخارصين في الترب المعاملة بالمخلفات العضوية ولكلا الموقعين .