Decomposition of organic materials and its effect on transport of nitrogen, phosphorous and water in unsaturated soil

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Abstract

A simultaneous transport of water, phosphorous, and nitrogen and organic matter decomposition were studied experimentally and numerically. Greenhouse experiments were achieved for collecting data of organic carbon, nitrogen, phosphorous and water contents. A numerical model (LEACHM) was used for simulating these data. Simulated results were compared with experimental data. The experiments were conducted in one-end open PVC columns (0.056-m ID and 0.30-m high) using unsaturated sandy soil under the atmospheric condition of Buridah, Al-Qassim, KSA. Municipal solid wastes (fermented partially) were mixed with the sandy soil. A 0.05-m mixed soil layer was located at six different depths (6 treatments) within the soil columns. Two replicates were assigned for each treatment. The soil or the mixed layer had initial water content of 0.0802 m^3/m^3. The initial carbon in the soil and mixed layer were 0.117 and 0.451%, respectively. The experiment lasted two months with application distilled water periodically at the open end of soil columns. Both predicted and measured final water showed nonlinear distributions. The model overestimated slightly the water content in comparison to the observed data. The stored water in a soil column decreased as the depth of mixed layer increased. The predicted and observed concentrations of organic carbon, NH_4^+, NO_3^-, and available phosphorous behaved similarly. The beak concentrations of the aforementioned variables occurred at the depth of the mixed layer. The organic matter decomposition was limited because the soil water content was low and the duration of experiment was short. The results of the study could encourage using the LEACH as a tool for organic matter management strategy and monitoring the fate and transport of plant nutrient as N and P in soil.
Keywords: LEACHM, organic carbon, decomposition, water movement, modeling, nitrogen, phosphorous.

1   Introduction

Soil organic matter improves root growth, uptake of minerals and aids the plant in the physiological activities. It increases mobilization of nutrients both major and minor from the soil to plant roots. It also produces growth promoting substances and higher nitrogen fixation by bacteria. It is essential in the formation of soil aggregates and soil structure which has a direct bearing on soil aeration. Acids released, dissolve insoluble phosphates and make it available. Decomposition of organic matter is variously referred to as oxidation, metabolism, degradation and mineralization. Organic matter is first oxidized by molecular oxygen, and the products (or metabolites) of the reaction are carbon dioxide and recycled nutrients Froelich et al. [1] and Drever [2]. For example, the aerobic respiration of organic matter yields 106 moles of carbon dioxide (CO_2), 16 moles of ammonia (NH_3) and one mole of orthophosphate (PO_4) (reaction 1). If sufficient dissolved oxygen is present, bacteria oxidize the NH_4 produced in reaction 1 to nitrite (NO_2) and then to nitrate (NO_3) in a process called nitrification. The decomposition of organic matter may take several months to several years to complete. The study of soil organic matter decomposition is important for assessing the problems of organic matter depletion in soil, long term soil fertility and sustained productivity of soil. A variety of soil parameters based models have been used by several researchers to represent short and long-term changes in single state variable model form to represent the decline of organic matter in cultivated soils. In the original CENTRUY model Parton et al. [3], the soil organic matter (SOM) consists of three fractions. They constitute (i) an active fraction (active SOM) of soil C consisting of live microbes and microbial products, along with soil organic matter (SOM) with a short turnover time (1-5 years); (ii) a pool of C (slow SOM) that is physically protected and/or in chemical forms with more biological resistance to decomposition, with an intermediate turnover time (20-40 years), and (iii) a fraction that is protected with the longest turnover time (200-1500 years). According to CENTRUY model, the rate of decomposition, k, of soil organic matter depends on soil types, soil texture (silt + clay content), rainfall, soil temperature, potential evapotranspiration, initial soil organic carbon content and the maximum decomposition rate parameter. The k value for active pools ranges from 0.29 yr^{-1} van Veen and Paul [4] to 0.66 yr^{-1} Parton et al. [5], Jenkinson [6]. Intermediate pools k values range from 0.02 yr^{-1} Jenkinson [6] to 0.04 yr^{-1} Parton et al. [5] to 0.003 yr^{-1} van Veen and Paul [4].

In the LEACHM model Hutson and Wagenet [7], it is assumed three organic pools: plant residue, manure and soil humus, each of which is defined in terms of C:N:P ratios. Inorganic N pools include urea, nitrate and ammonium. There are two inorganic P pools: a labile pool which is always in local equilibrium with sorbed and solution phases, and a bound pool which is in kinetically-controlled equilibrium with solution P. P sorption is described by Freundlich isotherms. Mineralization rate constants refer to transformation of organic carbon, which is split into three transformation pathways: into soil humus, into biomass, which remains an integral part of organic pool, and into carbon dioxide. The relative

sizes of these pathways are defined by the synthesis efficiency factor and the humification fraction. The efficiency factor is the fraction of C mineralized that is converted to humus and biomass rather than CO_2, while the humification factor defines the relative amounts of humus and biomass produced. LEACHM uses adaptations of the concepts and equations described by Johnsson et al. [8]. C:N ration for biomass and humus determine N released or consumed during mineralization. Thus C:N ratios of residue and manure decrease during mineralization; if insufficient N is available to satisfy the C:N ratio of the products, the rate slows. Inorganic N is subject to nitrification and denitrification. Denitrification rates increase with increasing water content and insufficient organic C needs to be present to support denitrification. Although organic pools are regarded as insoluble in LEACHM, this only means that they are not subject to transport. Conceptually, organic C can provide a C substrate for denitrification. Amonium volatilization is a complex, pH-dependent process. No attempt is made to simulate the various processes which contribute to and influence NH_3 volatilization. For simplicity, NH_3 volatilization is described assuming first-order kinetics, and is a function of a rate constant (which is not adjust for temperature or water content) and the concentration of NH_4 in the upper soil segment. LEACHM simulates flows between these pools in each soil segment as well as on the soil surface. C and N cycling is based on the procedures described by Johnsson et al. [8] but with additional pools and pathways. The inorganic P model was based on concepts described by Shaviv and Shachar [9]. In Shaviv and Shachar's model the strongly bound P pool was considered to be a precipitated form of P having a very low solubility; in the LEACHM this pool can be considered to be a precipitate or to follow a sorption isotherm. The labile pool is always in local equilibrium, but sorption to or desorption from the bound pool is kinetic, where the bound pool as well as the labile pool are described by Freundlich isotherms.

The objective of this study was to develop an integrated methodology for execution of LEACHM model developed by Hutson and Wagenet [7] for assessment soil organic matter decomposition and release of nitrogen and phosphorous nutrients. In addition, modeling water, N, and P movement in unsaturated soil is considered. In this model, it is considered only "active SOM" fraction of soil organic matter with a short time. In soil system, a continuing loss of soil humus carbon (decomposition loss) taken place by microbial oxidization.

2 Materials and methods

2.1 Greenhouse experiments

An experiment was achieved in Al-Qassim region, Saudi Arabia. Al-Qassim lies between 40 to 45 E longitude and 25 N to 30 N latitude approximately. The climate of the region is dry. The average rainfall of the area ranges from 100 to 200 millimeters. The ambient air temperature (T_{air}) of Saudi Arabia varies greatly from season to season and from region to region. T_{air} in the summer is about 43-48°C during the daytime and 32-36°C during the night time. The

microclimate is such that soil water evaporation readily occurs. Soil materials were sampled from a surface layer (0.0-0.3 m depth) from the Agriculture and Veterinary Collage farm, Al-Qassim University. The soil materials were obtained by excavation, then air-dried and ground to pass a 2-mm sieve. The soil was composed of sand materials. A soil material batch was wetted with distilled water to obtain an initial volumetric water content of 0.0802 m^3/m^3. Another batch of soil was mixed with municipal solid waste (partially fermented) at 1% then wetted with an initial water content of 0.0802 m^3/m^3. Each soil batch was covered and stored at $20^{o}C$ for 4 days. PVC cylinders (0.056-m ID and 0.3-m high) were closed at the bottom ends using epoxy-sealed PVC lids. Six treatments were obtained based upon the layer depth of mixed soil layer within the soil column. The treatments are shown in Table 1. The thickness of mixed layer was 0.05 m. Two soil columns replicates were packed at bulk densities of approximately 1514 kg m^{-3} for each treatment. The soil columns were buried vertically within a bare soil field with exposing the open upper end to the natural atmosphere of Al-Qassim region. Distilled water was poured at the open of soil column at different time as shown in Table 2. The soil temperatures at both ends of soil column were recorded. The soil columns were sectioned into a 0.05-m increment. The soil of each increment was divided into two parts: a part for water content and organic carbon and a part for ammonium, nitrate, and phosphorous determinations. The soil water contents were determined gravimetrically in the increments. Available P was extracted using the sodium bicarbonate (Olsen et al. [10]) then was determined by the chlorostannous phosphomolybdic acid method (Jackson [11]). The organic matter was estimated using the Walkely and Black method (Nelson and Sommers [12]). The nitrogen in the soil samples was determined by distillation (Page et al. [13]).

2.2 Theoretical analysis

LEACHM model devolved by Hutson and Wagenet [7] was used to describe water and chemicals transfer, and organic matter decomposition in a soil under nonisothermal condition. A brief outline of LEACHM is presented below for completeness, with additional details found elsewhere (Hutson and Wagenet [7]).

2.2.1 Water flow
The soil water flow equation for transient vertical flow derived from Darcy's law and the continuity equation, is:

Table 1:   The depth of municipal waste with the soil column.

Treatment number	Depth of the mixed layer, m
1	0.0 0.05
2	0.05-0.10
3	0.10-0.15
4	0.15-0.20
5	0.20-0.25
6	0.25-0.30

Table 2:   Amount and time of water application (m/column).

Date of application	Amount of applied water, m	Date of application	Amount of applied water, m
26/7/2003	0.01016	2/9	0.00203
29/7	0.02032	7/9	0.00203
6/8	0.01016	10/9	0.00203
10/8	0.01016	13/9	0.00203
26/8	0.00203	15/9	0.00203
30/8	0.00203	20/9	0.00203

$$\frac{\partial \theta}{\partial t} = \frac{\partial \psi}{\partial t} C_w = -\frac{\partial q}{\partial z} - U(z,t) \tag{1}$$

where θ is water content (L^3 L^{-3}), q is water flux density (L/T), defined as q=-K(Ψ)(δH/δz), H is hydraulic head (L), defined as H=Ψ-z, Ψ is soil water matric potential and z is depth), K is hydraulic conductivity (L/T), t is time (T), C_w is differential water capacity (L^{-1}), and U is a sink term representing water lost per unit time by transpiration (T^{-1}). The later term is neglected under the present study because there is no plant. The water transfer properties (Ψ -θ, and hydraulic conductivity, K) were described in detail by Campbell [14].

2.2.2 Decomposition of organic C, N and P

The rate of loss of soil organic matter (SOM) is based on the instantaneous decay constant, k, which expresses the proportion of the pool that turns over per unit time. We simulated soil organic decompostion for two months using the LEACHM model. The simulated results for total soil C were compared to the measured values from the greenhous experiment. First-order mineralization rate constants determine the overall decomposition rate of the humus pools. Decomposition follows first-order kinetics:

$$\frac{dC_i}{dt} = -\mu_{mi} C_i \tag{2}$$

where C_i represents the concentration of humus, μ_{mi} is first-order rate constants, and t is the time.

Organic N and P associated with the decomposing C pool is released. C:N and C:P ratios defined for humus how much of the N and P released by mineralization are assigned to the humus pools. Dentrification process is neglected in the present work because the soil was unsaturated.

2.2.3 Flow of nitrogen and phosphorus

The movement of miscible solute (i.e., N and P elements) through a soil is assumed in LEACHM to be accomplished by chemical diffusion in the liquid phase in

response to an aqueous concentration gradient and convection of the solute as the result of movement of water flow in which the solute is dissolved. That is:

$$J_t = J_d + J_c \qquad (3)$$

where J_t is total solute flux ($M/L^2 T$), J_d and J_c are diffusion and convection fluxes in the liquid phase, respectively.

The diffusion flux in a soil solution can be obtained from:

$$J_d = -D_p(\theta)\frac{\partial C_l}{\partial z} \qquad (4)$$

where $D_p(\theta) = D_o \, \varepsilon \, exp(b*\theta)$, is the diffusion coefficient of solute in porous media and ε and b are empirical constants Olsen and Kemper [10]. Values of ε ranged from 0.05 to 0.01 and b=10. C_l is the concentration in the liquid phase (M/L^2). D_o is the diffusion coefficient of solute in a free-water system (L^2/T). The diffusion coefficient was described in details Nassar and Horton [15] and Lyman et al. [16].

The convective flux of a solute is usually represented as:

$$J_c = -\theta D_m(q)\frac{\partial C_l}{\partial z} + qC_l \qquad (5)$$

where $D_m(q) = \lambda|v|$, is the mechanical dispersion coefficient that describes mixing between large and small pores, $v=q/\theta$, is the average pore velocity and λ is the dispersivity (L) and its value ranged between 0.5 Δz to $2\Delta z$, and Δz is node spacing (L).

The total steady state solute flux in the liquid phase in a porous media, J_t can be described as:

$$J_t = -\theta D_m(q)\left(\frac{\partial C_l}{\partial z}\right) - D_p(\theta)\frac{\partial C_l}{\partial z} + qC_l \qquad (6)$$

The nonsteady-state equation for the solute transport can be written as:

$$\frac{\partial C_t}{\partial t} = -\frac{\partial J_t}{\partial z} \pm \Phi \qquad (7)$$

where $C_t = \theta C_l + \rho_b C_s$, is the total solute concentration in liquid and sorbed phases (M/L^3), ρ_b is the bulk density of soil (M/L^3), C_s is the concentration of solute in the sorbed phase (M/M) (i,e. phosphorous or ammonium) and Φ is sources or sinks of solute ($M/L^3 T$). Φ is considered negligible for N compounds for the present study because we did not add mineral fertilizer.

LEACHM model can reflect sorption using curvilinear Freundlich isotherms. This is required in order to describe P sorption realistically. The Freundlich isotherm is defined as

$$C_s = k_f C_L^{n_f} \qquad (8)$$

C_s is sorbed concentration (mg kg-1), C_L (mg L^{-1}) is solution concentration, and k_f ($mg^{1-nf} kg^{-1} L^{nf}$) and n_f are constants.

In the case of ammonium, the sorbed ammonium concentration can be described as

$$C_s = K_d C_{L,} \qquad (9)$$

K_d is a partition coefficient (L/M).

2.3 Initial and boundary condition

The initial conditions associated with Eqs. (1) nd (7) are given by

$$\theta(z,0) = \theta_i, \; C_t(z,0) = C_i \qquad (10)$$

The upper boundary conditions for water and solute are given in terms of net mass fluxes by

$$q(0,t) = \zeta E \quad \text{(under the evaporation conduction)} \qquad (11)$$

$$J_t(0,t) = 0.0 \; \text{(for a non-volatile compound)} \qquad (12)$$

or

$$q(0,t) = \inf \; \text{(under ponding condition)} \qquad (13)$$

$$J_t(0,t) = 0.0 \; \text{(no added N or P compounds in the applied water)}$$

where ζ is the evaporation pan coefficient, E is the potential evaporation rate (LT^{-1}), inf (LT^{-1}) is the infiltration rate .

The lower boundary condition for water and solute (N and P solutes) are:

$$q \; (l,t) = 0.0 \qquad (14)$$

$$j_t(l,t) = 0.0 \qquad (15)$$

2.4 Model and results evaluation

Model ability to predict a variable should be evaluated. Visual comparison of simulated and observed data provide a quick and often comprehensive mean of assessing the accuracy of model prediction. However, quantitative evaluation of the model is recommended. In the present work, the mean error (ME) and root mean square error (RMSE) are used as criteria for evaluating the model Milly [17]. The ME values can be estimated as:

$$ME = \tfrac{1}{N} \sum_{i=1}^{N} (V_p - V_m) \qquad (16)$$

and RMSE can be estimated as:

$$RMSE = \left(\frac{1}{N} \sum_{i=1}^{N} (V_p - V_m)^2 \right)^{1/2} \qquad (17)$$

V_m is the observed value of a variable (θ, or C_l), V_p is the predicted value of a variable, and N is number of observations for the variable.

2.5 Soil parameters and characterization

The retention curve data (Ψ vs. θ) and the saturated hydraulic conductivity were measured for the sandy soil used in the present study. The relation of Ψ vs. θ was fitted to the Campbell's function [14]. The coefficients of the function, Ψ_e and b, are shown in Table 3. The saturated hydraulic conductivity, Ks, of the soil was measured in a laboratory soil column using a constant head method Klute and Dirksen [18]. The unsaturated hydraulic conductivity, $K(\Psi)$, was estimated from knowledge of K_s and the retention curve data Campbell [14]. For more details, the reader is referred to Al-Salamah [19].

Table 3:   Input characterization data for LEACHM model.

Parameters	Values
ρ (bulk density) (kg m^{-3})	1514
θ_i (initial water content) (m^{-3} m^{-3})	0.0802
θ_s (saturated water content) (m^{-3} m^{-3})	0.4286
K_s (m s^{-1})	2.25x10^{-5}
Clay, (%)	1.9
Sand (%)	96.3
Silt (%)	1.8
CaCO$_3$ (%)	3.72
Ci (P), ppm	13.7 (17.0)[*]
Ci (NO$_3$), ppm	20.04 (37.22)[*]
Ci (NH$_4$), ppm	10.18 (16.54)[*]
Ci (O.C) (%)	0.117 (0.451)[*]
D_o (m^2 s^{-1})	6.478x10^{-10}
k_d (L/kg) (NH$_4$)	3 (NH$_4$)
μ_{mi} (day^{-1})	0.0025
k_f (L/kg)	100 (P)
n_f	0.6 (P)
C:N (humus)	10:1
C:P (humus)	50:1
Ψ_e (kPa)	-0.38
b	1.81

[*] These numbers represent the concentration in the municipal waste mixed layer through the soil column.
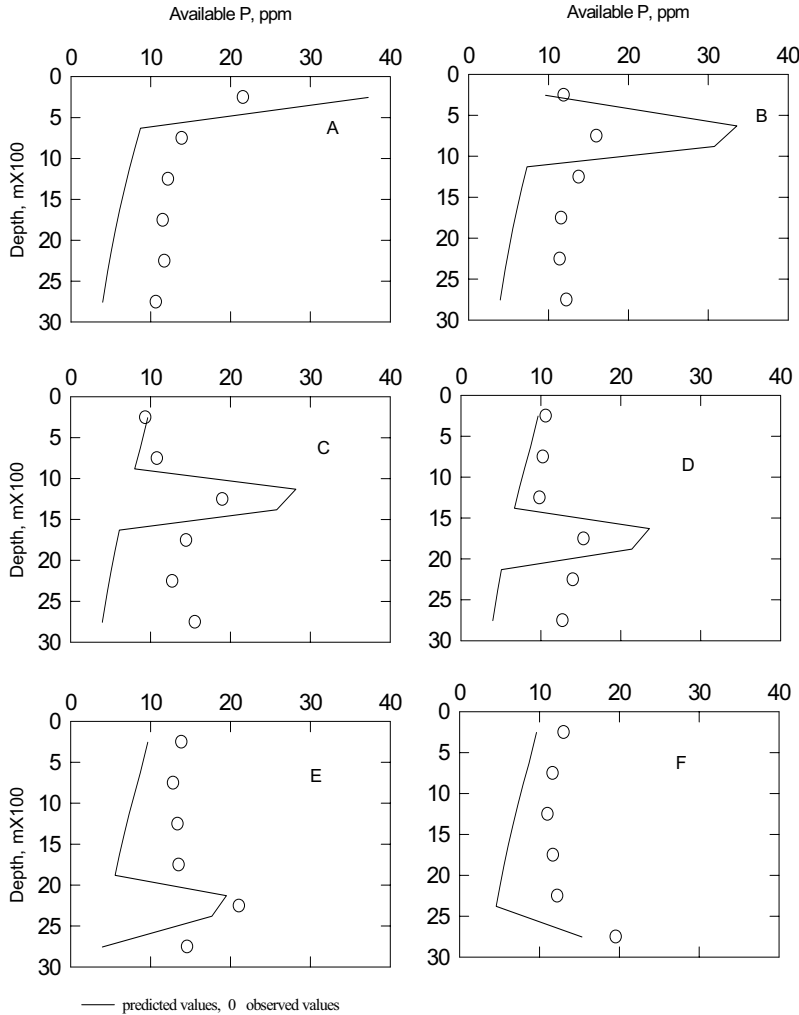
Figure 1:   The predicted and observed available phosphorous. A, B, C, D, E, and F referred to the depths of municipal waste layer 0.0-0.05, 0.05-0.1, 0.1-0.15, 0.15-0.2, 0.2-0.25, and 0.25-0.3 m, respectively.

3   Results and discussion

3.1 Predicated and measured nutrients concentration

Measured and predicted available phosphorous, P, concentrations are compared in Figure 1 for time of 60 d. Initial soil carbon, nitrogen and phosphorus concentration are provided to the model as input variables (Table 3). The predicted values of P followed the trend of the observed values. Both observed and

predicted values showed the greatest P concentration in the municipal waste layer. The great value of P is due to two reasons. The first reason is the high initial concentration of P in the municipal layer. The second reason is the low movement of P that is controlled by the presence of calcium carbonate (3.72%) and low soil water content. The average means of P within most of the soil column were lower than the initial P concentration (14.25 ppm) which indicated that some of labile P is bounded to the soil particle. The model overestimated the predicted P in the layer of municipal waste. This might be due to the high mineralization coefficient (0.0025d[1]) used for the simulation followed by high released P. The soil posses' low percentage of water contents that results low unsaturated hydraulic conductivity. So, movement of phosphorous in soil is limited well. It is expected that some of the released phosphorus will be bound to the calcium carbonate present in soil. When the municipal waste layer was either at the 0-0.05 or 0.05-0.10 m-depth, the predicted P was greater than the observed. This discrepancy is due to the lower movement of water by model or the high mineralization rate of SOM. In general, the measured value of phosphorous showed little difference within the soil column. Phosphate forms are more soluble Ca^{2+} and Mg^{2+} at pH values near neutrality, and difficulty soluble Ca^{2+} compounds at higher pH Bohn et al. [20]. Phosphate availability also tends to decrease at high soil pH, because of precipitation as insoluble calcium phosphate compounds Bohn et al. [20]. The long-term capacity of most soils to adsorb phosphate is an order of magnitude greater than the amounts of phosphate added as fertilizer Bohn et al. [20].

Figures 2 and 3 show the predicted, and observed ammonium and nitrate concentrations, respectively. Both the predicted and observed values followed a similar trend. The municipal waste layer possessed the greatest concentration of either form of nitrogen. The predicted values of all forms of nitrogen showed good agreement with the observed ones within all soil columns of most of the treatments. When the municipal waste layer was located in the 0-0.05 layer, the LEACHM model overestimated all of the nitrogen forms in this layer. This discrepancy in the first layer in nitrogen concentration can due to the high convective transport calculated by the model for the nitrate and the low nitrogen loss in gaseous phases. The amount of N_2 and N_2O lost during oxidation is normally small and often negligible Bohn et al. [20]. The average mean concentration of ammonium within all soil column treatment was less than the initial concentration of ammonium (11.24 ppm), while nitrate concentration means were greater than the initial concentration (24.57 ppm). Some of ammonium might be converted to nitrate as a result of nitrification process. Relative to other exchangeable cations, however, the amount of exchangeable NH_4 is almost invariably small. Ammonium ions are much less mobile than NO_3 and are much less likely to be lost through denitrification, although ammonia readily volatilizes from the surface of alkaline soils. In the desert soil, for example, the absence of water greatly hinders the oxidation rate of organic materials at soil surface Bohn et al. [20]. Nitrogen may also be lost to the atmosphere as NH_3; which may occur whenever NH_4 is present at the soil surface, especially at high pH (above 7.0) and high temperature by volatilization when surface applied Barber [21]. The nitrate concentration was high when the municipal waste was located at either depth of 0.0-0.05 or 0.05 -0.10 m. So, high concentration
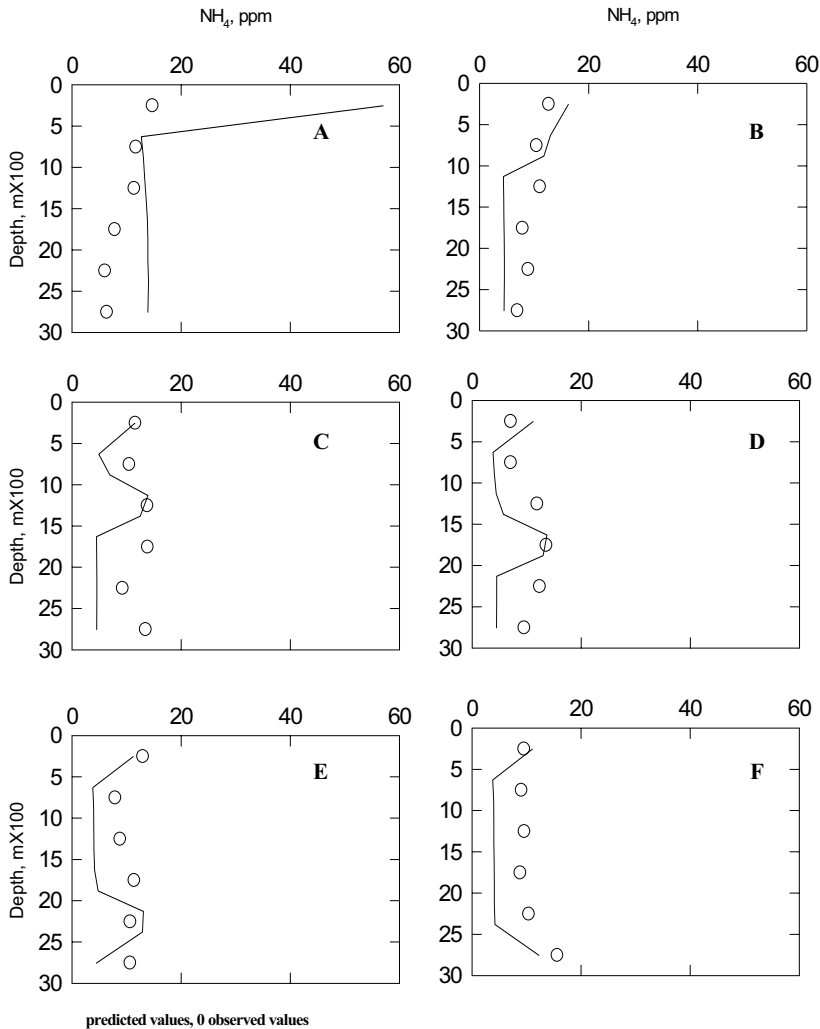
Figure 2:   The predicted and observed amonium. A, B, C, D, E, and F referred to the depths of municipal waste layer 0.0-0.05, 0.05-0.1, 0.1-0.15, 0.15-0.2, 0.2-0.25, and 0.25-0.3 m, respectively.

of NO_3 in the top layers might be due to the high convective flux of nitrate and the high nitrification processes in these layers. The NO_3 solutions flowed through the soil column almost as quickly as the water because they are not adsorbed by soil constitutes Bohn et al. [20]. They also reported that in well–aerated soils, with adequate moisture, moderate temperature, NH_4 are converted to NO_3 in a matter of weeks. Cumulative net nitrogen that was mineralized was linearly related to the

square root of time Barber [21]. Values for the mineralization potential varied from 20 to 300 mg/kg of air-dry soil. This potential was 5 to 40% of total nitrogen in the soil, with an average 18% for 39 soils studied. The mineralization rate constant averaged 0.054 ± 0.009/week. Barber [22] measured the reduction in soil organic matter content in plots that were followed for six years; the rate was 2.4% per year while Larson et al. [23] obtained a value of 1.9% per year in Iowa.
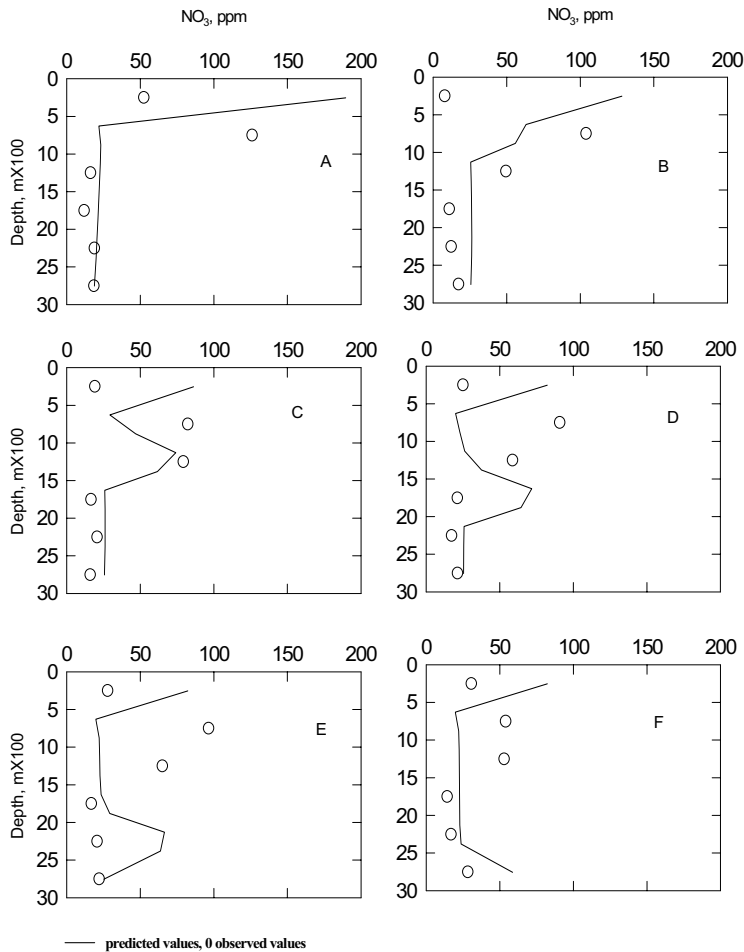

Figure 3:   The predicted and observed nitrate. A, B, C, D, E, and F referred to the depths of municipal waste layer 0.0-0.05, 0.05-0.10, 0.1-0.015, 0.15-0.20, 0.20-0.25, and 0.25-0.3 m, respectively.
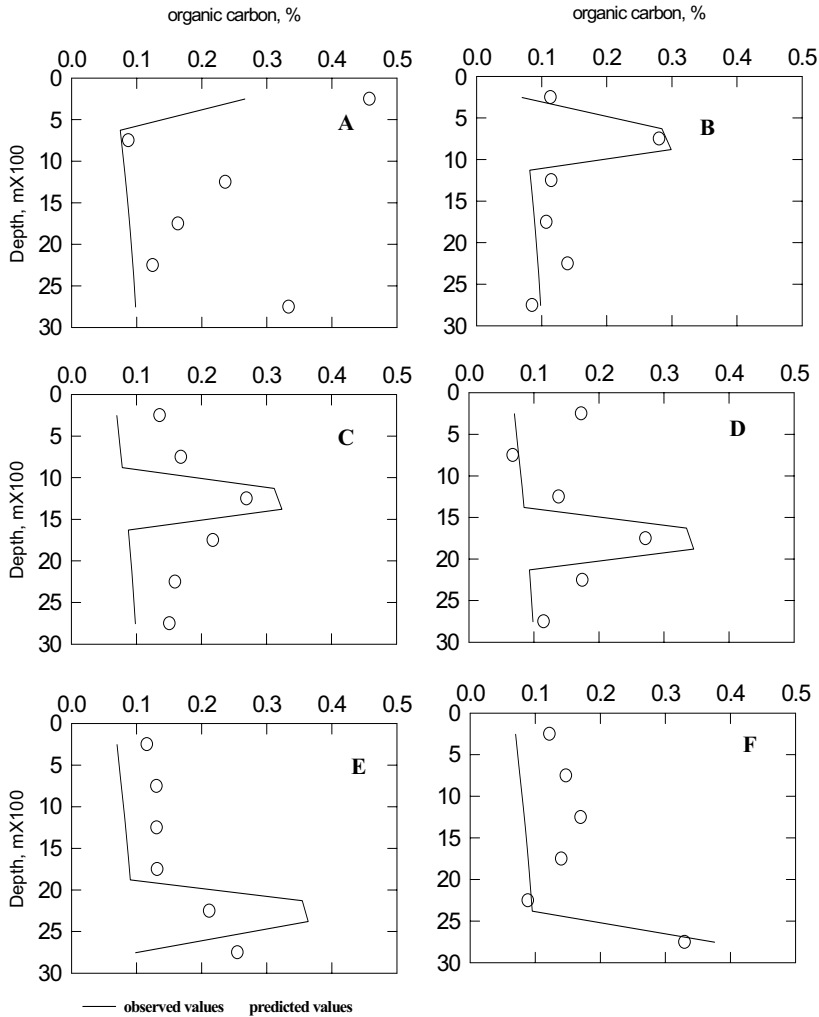
Figure 4:   The predicted and observed organic carbon. A, B, C, D, E, and F
             referred to the depths of municipal waste layer at 0.0-0.05,
             0.05-0.1, 0.1-0.15, 0.15-0.2, 0.2-0.25, and 0.25-0.3 m, respectively.

Figure 4 shows the calculated and observed organic carbon. The LEACHM
model predicted organic carbon values in close agreement with the observed
values within most of the soil columns. The mean of organic carbon was little
less than the initial organic carbon (0.172%) in four treatments. The difference
between the means of theses treatments and the initial organic carbon was small
because of the soil water content was small during running period of the
experiment and the duration of experiment was short (2 months). Parton et al [3]

reported that the rate of decomposition of soil organic matter depends on soil types; soil texture (silt + clay content); rainfall; soil temperature; Potential evapotranspiration; initial soil organic carbon content and the maximum decomposition rate parameter. Parton et al. [24] found that CENTURY model simulated soil C and N levels within 25% of the observed values. Field losses of total C averaged 28% over the life of an old ant nest (between 30 and 60 yr according to Coffin and Lauenroth [25]. Gilmanov et al. [26] found that CENTURY model reproduced the seasonal, mid-term, and in some cases, long-term dynamics in aboveground biomass in a wide range of grassland ecosystems across the former USSR. They attributed model discrepancies to changes in species composition and short-term responses to intermittent rainfall that are missed by the monthly time step of the model. However, decay constants were not always in agreement with conceptual models. The authors' estimated values of active soil organic matter (SOM) turnover (k = 0.007-0.035) were lower than predicted by the three theoretical models van Veen and Paul [4], Parton et al. [5], and Jenkinson [6] to which they made comparisons (k = 0.29-0.66). Gilmanov et al. [26] used CENTURY model output to calculate simulated SOM losses over time due to removal of plants. Over 60 yr, simulated total soil C decreased 63% from 2074 g/[m.sup.2] to 828 g/[m.sup.2]. Their measured loss of total C on ant mounds over an estimated 30-60 yr was approximately 28%. The CENTURY model predicted an annual (k-based) loss of total C of 1.6-2.1%, while measured loss rates (k-based) range from 0.5 to 1.1%.

3.2 Predicted and measured soil water status

The predicted and measured soil water content distributions in nonisothermal columns are shown in Fig.5. The predicted water content distributions matched the observed values only in trend. There are two distinguished zones for the observed distribution of the soil water contents. The first zone was located in the upper 0.2 m while the second zone was located in the lower 0.1 m layer. The water content changed abruptly from the upper to the lower zones. The mean soil water contents calculated from the observed data were 4.79, 4.17, 4.90, 4.56, 3.58 and 3.69% for the treatments 1, 2, 3, 4, 5, and 6, respectively. The model overestimated the soil water content in the upper zone while it is underestimated the soil water content in the lower zone. The means of soil water (approximately 4.9%) obtained from the predicted values were greater than the observed means. Overestimation of the predicted soil water content is due partially to underestimation of water vapor movement. Similar results were reported by Nassar et al. [27] and Al-Salamah [19]. Several researchers found that temperature gradient has great effect on water flow Globus [28] and Nassar et al. [29].

Figure 6 shows the stored soil water after 60 d as a function of the depth of mixed layer. The stored water content decreased as the depth of mixed layer increased. For example, the stored water contents were 0.0062 and 0.0042 m/column when the mixed layer at 0-0.05 and 0.25-0.30 m, respectively. The stored soil water as a function of the mixed layer depth described using a liner relation. The function described 72% of the variations. Similar results were

reported by Al-Salamah and Nassar [30] when municipal waste was located at the top, mixed in the top layer or located at subsurface layer. According to cumulative evaporation reported by Al-Salamah and Nassar [30], the four application aspects followed the order: the bare soil > mixed layer> subsurface layer> top layer.
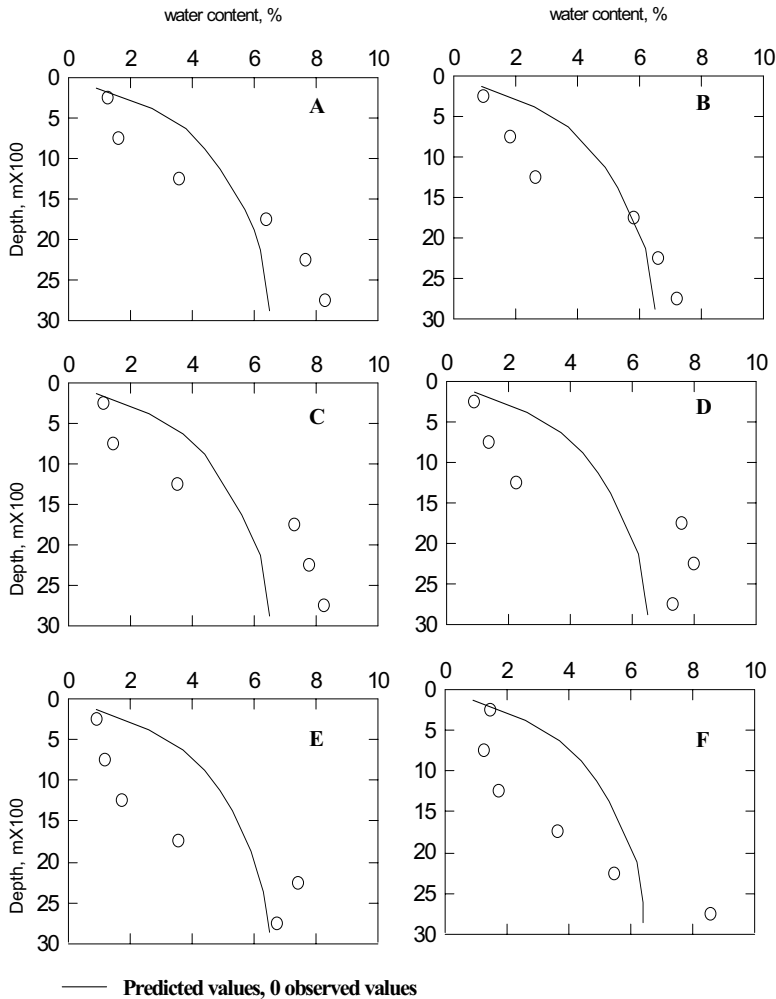

Figure 5:   The predicted and observed volumetric water content. A, B, C, D, E, and F referred to the depths of municipal waste layers 0.0-0.05, 0.05-0.10,   0.1-0.15,   0.15-0.20,   0.20-0.25,   and   0.25-0.30 m, respectively.
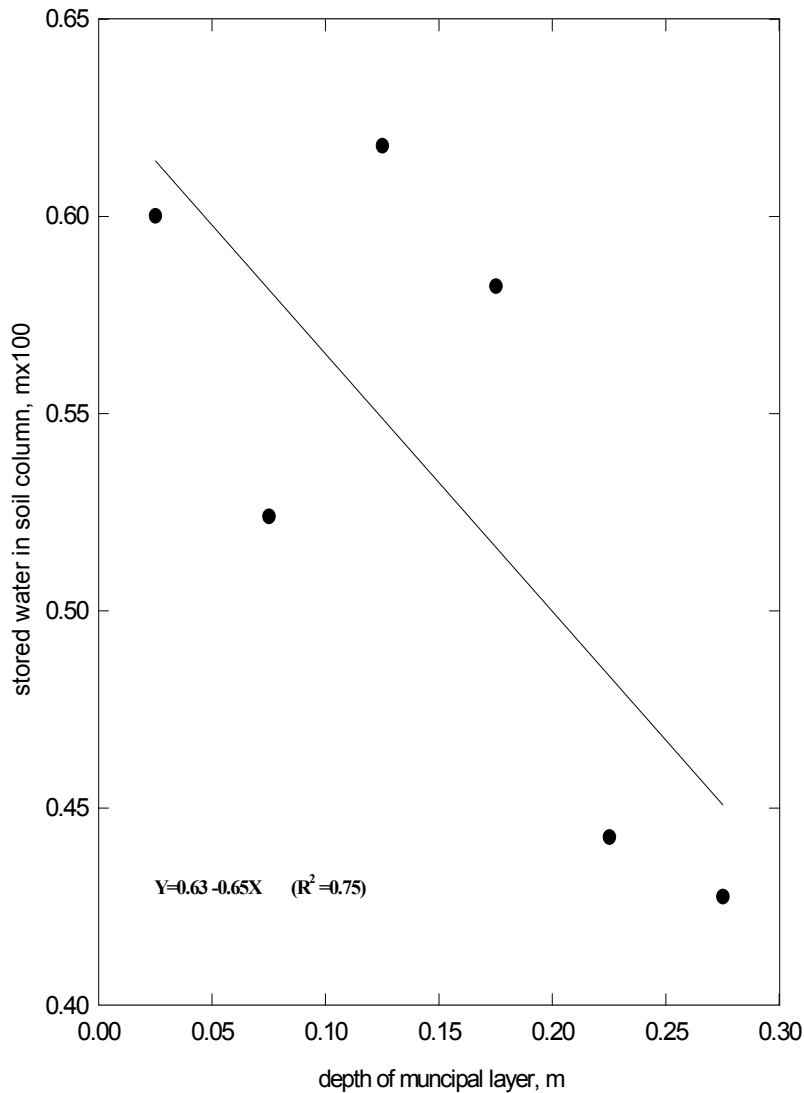
Figure 6:   Effects of municipal wastes depth on the stored water in a soil column.

3.3 Model evaluation

Table 4 shows the observed mean, mean error (ME) and root mean square error (RMSE) for the P, NH_4, NO_3, Organic carbon and water content (Figures 1–5). The observed means of P ranged from 12.11 to 14.86 ppm and ME ranged from 2.71 to 5.94 ppm. The ratio of ME to the observed mean concentration of P varied from treatments to other. This ratio ranged from 0.22 to 0.39. The lower ratio indicates

that high performance of the model for P prediction. The observed means for NH_4 ranged from 9.61 to 12.01 ppm and the ME ranged from 1.82 to 4.63 ppm. The ratio of the ME to the observed mean was 0.18 to 0.38. It can be concluded that the performance of the LEACHM model in the prediction of NH_4 is similar to the prediction of P. The ratio of the ME to the observed means ranged from 0.03 to 0.21, 0.14 to 0.47 and 0.01 to 0.32 for the NO_3, organic carbon content and water content, respectively. It can be concluded that the performance of the LEACHM model in prediction of nitrate is better than the prediction of either water content or organic carbon content. The RMSE values behaved similarly to the ME for P, NH_4, NO_3, organic carbon, and the water content. It can be concluded that the model described the mechanisms of organic matter decomposition and P, NH_4, NO_3 and water flows appropriately under the conditions of the present study.

Table 4:   The observed mean, mean error (ME) and root mean square error (RMSE) for the data presented in Figures 1–5. The symbols are shown in the figures.

Figure No	Param-eters	Figures symbols					
		A	B	C	D	E	F
Fig. (Av..P) (ppm)	Mean	13.56	12.81	13.63	12.11	14.86	13.19
	ME	2.345	2.218333	3.6875	2.7195	5.9425	4.671667
	RMSE	8.43041	8.725044	7.531261	6.101289	6.507956	4.899321
Fig. 2 (NH_4) (ppm)	Mean	9.63	9.61	12.01	10.21	10.34	10.45
	ME	-11.2157	1.828167	4.639667	3.119667	3.504167	3.8685
	RMSE	18.04968	4.004083	5.881585	5.159043	4.683364	4.667167
Fig. 3 (NO_3) (ppm)	Mean	40.56	33.61	38.96	39.02	41.43	32.79
	ME	-8.645	-15.0242	-6.06	-3.46667	1.04	-5.595
	RMSE	70.32737	54.05466	33.64538	43.02575	45.71885	30.90611
Fig. 4 (O. C) %	Mean	0.233	0.140	0.183	0.156	0.162	0.166
	ME	0.116	0.019	0.059	0.029	0.033	0.034
	RMSE	0.142	0.030	0.080	0.065	0.096	0.057
Fig. 5 (Water) %	Mean	4.79	4.17	4.9	4.56	3.58	3.68
	ME	-0.1285	-0.71105	0.025667	-0.32728	-1.29512	-1.2055
	RMSE	1.554069	1.421457	1.701784	1.968665	2.110211	2.193406

4 Summary and conclusion

A greenhouse experiment was achieved for monitoring organic matter decomposition, P, NO_3, NH_4 and water transfer in unsaturated soil column. Municipal wastes material as a source of organic matter was mixed with soil and located at different depths within the soil columns. The soil columns were one-end open and buried vertically in a bare soil field in Al-Qassium region, KSA. Distilled water was added several times during the course of the experiment. Presences of municipal waste mixed layer close to the soil surfaces reduced water losses in comparison to lower depth. The LEACHM model was used to predict soil organic matter decomposition, and P, No_3, NH_4 and water content within the soil columns. The predicted and observed variables were similar in trends. However, there were some discrepancies between the observed and predicted values especially in the

upper portion of soil columns. The application of this model may be very useful for the monitoring soil organic matter decomposition and the fate of nutrients (P, C, and N) in soil. This model can be a good tool in the fertilization strategies for plants. It is also useful for monitoring nitrate transfer and its deterioration effects on water resources.

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