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Water and Bromide Dynamics in a Soil Amended with Different Urban Composts

Maha Chalhoub, Yves Coquet,* and Pierre Vachier

ABSTRACT

Urban waste compost additions to soil can increase soil organic matter content and improve soil physical conditions, leading to agronomic and environmental benefits. The need for information still exists to evaluate more precisely the effects of urban waste compost on soil physical properties. Three types of urban waste composts, a biowaste compost (BIO), a municipal solid waste compost (MSW), and a co-compost of green waste and sewage sludge (GWS), were applied once every 2 yr on a loamy soil for 10 yr. The effects of the three composts on soil water and solute transport dynamics were tested. Soil water matric head and water content were monitored using tensiometers and time domain reflectometry probes, respectively. A Br^- tracer experiment was also conducted to evaluate the effect of compost application on nonsorbing solute transport. Water content measurements showed that the application of composts significantly ($P < 0.05$) affected soil water content in the plow layer, with average increases of $0.03 \text{ cm}^3 \text{ cm}^{-3}$ for the GWS and MSW compost, and $0.015 \text{ cm}^3 \text{ cm}^{-3}$ for the BIO compost compared with a control without organic amendment. Bromide tracing during the wetting period showed that the application of urban waste composts did not affect the soil's potential for leaching. The application of composts did reduce soil evaporation during the spring, however, which in turn favored downward Br^- migration in the soil.

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Abbreviations: BIO, biowaste compost; CDE, convection–dispersion equation; CONT, control receiving no organic amendment; FYM, farmyard manure; GWS, co-compost of green wastes and sewage sludge; MSW, municipal solid waste compost; SCM, stochastic–convective model; TDR, time-domain reflectometry.

Urban waste management is a major concern around the world due to the continuous increase in the urban population and economic growth (McCarthy, 1994). Composting is a method to manage and dispose of urban wastes by application to cultivated land. Waste composting creates a relatively low-cost product that can be used in agriculture for agronomic, economic, and environmental advantages (Stratton and Rechcigl, 1998; Wolkowski, 2003; Hansen et al., 2006; Hargreaves et al., 2008). Like other organic amendments, urban waste compost addition to the soil tends to affect its physical properties (Khaleel et al., 1981; Sikora et al., 2002). Compost addition increases soil aggregate stability (Hernando et al., 1989; Annabi et al., 2007), decreases bulk density (Giusquiani et al., 1995; Wong et al., 1999), and increases porosity and water holding capacity (Hernando et al., 1989; Soumare et al., 2003; Pagliai et al., 2004). The reported effects of compost amendments on soil hydraulic conductivity are poorly documented and sometimes contradictory. Some studies performed on a variety of soils, from sandy to clayey, have shown increased saturated hydraulic conductivities after compost application (Khaleel et al., 1981; Aggelides and Londra, 2000; Strauss, 2003; Celik et al., 2004; Paradelo et al., 2009), while no effect was found in other studies (Lindsay and Logan, 1998). Schneider et al. (2009)

observed a decrease in saturated and near-saturated hydraulic conductivity after compost application on a silt loam soil, but the effect had disappeared 2 mo after compost application. Pot et al. (2011) studied the effect of compost application on solute hydrodynamic dispersion in undisturbed silt loam soil columns and found no statistically relevant effect.

Methodological differences can induce substantial variability in the estimated soil physical properties (Paige and Hillel, 1993). Laboratory methods have been used to study the effect of urban waste compost on soil physical properties (Lindsay and Logan, 1998; Foley and Cooperband, 2002; Ebertseder et Gutser, 2003; Pot et al., 2011) rather than field measurements, which are costly and time consuming. Field methods are more representative than laboratory methods, however, because soils are characterized under natural conditions (Scott, 2000; Mermoud and Xu, 2006). Only a few field-scale studies have examined the evolution of soil hydraulic properties after waste amendment. Kluge and Bolduan (2003) found that compost application increased water retention in soil, but this effect was dependent on the soil texture. Paradelo et al. (2009) showed that grape marc vermicompost increased soil hydraulic conductivity at an application rate of 120 Mg ha^{-1} , while no effect was detected at 60 Mg ha^{-1} . None of these studies looked at eventual effects on solute transport properties.

A field study was conducted to test (i) whether or not urban compost additions modify water dynamics and inert solute transport in bare soil, and (ii) if different types of composts lead to different impacts on these processes. Three urban waste composts from municipal wastes were examined and compared with a reference farmyard manure and a control treatment without organic amendment. Soil water matric head and water content were monitored using tensiometers and time-domain reflectometry (TDR) probes, respectively. A Br^- tracer study was performed to evaluate compost application effects on nonsorbing conservative solute transport in soil. The study started just after compost application in autumn and covered the subsequent soil wetting period, which is critical for pollutant release and leaching.

MATERIALS AND METHODS

Study Site and Compost Characteristics

A long-term field experiment has been performed in Feucherolles (Yvelines, France) since 1998 to characterize the agronomic value and environmental impacts of urban composts (Houot et al., 2002). Three different composts were used in this study: (i) a biowaste compost (BIO) produced by co-composting green wastes and the organic fraction of source-separated municipal wastes; (ii) a municipal solid waste compost (MSW) made from mechanically separated organic fractions after dry and clean packaging were selectively collected; and (iii) a co-compost of green wastes, wood chips, and sewage sludge (GWS). The three compost treatments were compared with a farmyard manure (FYM) applied at the same C application rate, $4 \text{ Mg organic C ha}^{-1}$, and to a control receiving no organic amendment (CONT). The properties of the different organic amendments are reported in Table 1. The soil at the 6-ha study site is a silt loam Albeluvisol (World Reference Base classification) or Glossudalf (U.S. soil taxonomy). The soil was formed from a 2.5-m-thick loess deposit overlying a buhrstone clay bed that crowns the sands and sandstones of the Sables de Fontainebleau formation (Nicole et al., 2003). The particle size fractions of the 0- to 32-cm surface layer were $15.0 \pm 1.0\%$ clay, $78.2 \pm 2.0\%$ silt, and $6.7 \pm 0.9\%$ sand. Wheat (*Triticum aestivum* L.) and maize (*Zea mays* L.) were cultivated in rotation.

Composts and manure were applied to 10- by 45-m plots on wheat stubble once every 2 yr in September 1998, 2000, 2002, 2004, and 2006. A supplementary compost application was made in September 2007 on barley (*Hordeum vulgare* L.) stubble when maize could not be cropped because nearby fields were infested with corn rootworm (*Diabrotica virgifera virgifera* Leconte). After each application, organic amendments are incorporated to a depth of 25 cm by chisel plowing the day after application. Mineral N fertilization was used on each treatment as a complement to the organic amendments. The soil was plowed once every year in October to November to a depth of 28 cm with a four-furrow moldboard plow. After the last application in September 2007, the soil was plowed on 22 Oct. 2007. A 7-cm-deep seedbed was prepared on 28 Apr. 2008 before maize sowing. The field experiment includes four blocks, each one divided into five plots (Fig. 1). Each plot corresponds to a treatment, and the treatments are distributed in a split-plot design. One plot per treatment were selected and used for similar experiments and measurements. Some physical and physicochemical properties of the topsoil of the five selected plots are reported in Table 2.

Field Measurements

Daily weather data were recorded at a meteorological station located 500 m away from the field experiment. Data included rainfall, air temperature, air humidity, wind speed, and net radiation.

A 2-m-long, 1-m-wide, and 2-m-deep trench was dug in each of the five selected plots (Fig. 1) and instrumented with one tensiometer and one TDR probe (rod length of 20 cm) at the 20-, 40-, 60-, 80-, 100-, 130-, and 160-cm depth. The tensiometers had 7.5-cm-long, 9.9-mm-diameter ceramic ends mounted on 40-cm (for the tensiometers at the 20-cm depth) and 55-cm polyvinyl chloride tubes connected to Hg manometers. Measurements were performed three times per week from November 2007 until June 2008, but the 20-cm tensiometer readings were stopped earlier because of tillage for seedbed preparation. Because no replicates were available, TDR and tensiometer data were used only for qualitative analysis of the soil water dynamics in the selected plots. No statistical test could be performed on the TDR and tensiometer data. Average bulk density profiles were measured using three 500-cm³ soil cores (9 cm by 8.4-cm diameter) taken from each soil horizon at the time the TDR probes and tensiometers were installed.

On Nov. 28 2007, a tracer study was started on a 10-m-long, 3-m-wide area (Fig. 1) within each of the five selected plots, 35 m away from the TDR trench. A 73 g L⁻¹ KBr solution was applied with a sprayer at a rate of 67 g Br⁻ m⁻². Soil samples for Br⁻ analysis were taken 13 d after Br⁻ application (DAA) (11 Dec. 2007, T1), 56 DAA (23 Jan. 2008, T2), 99 DAA (6 Mar. 2008, T3), 125 DAA (1 Apr. 2008, T4), and 191 DAA (6 June 2008, T5), which corresponded to 57, 98, 150, 226, and 366 mm of cumulative rainfall, respectively, since Br⁻ application. At each sampling date, six soil cores were taken 1.8 m from each other along 10-m transects oriented perpendicularly to the main dimension of the plot (Fig. 1b). Five of these six soil cores were collected with a hydraulic auger (1.6-cm i.d.) and one (noted R4 in Fig. 1b) with a manual auger (3.5-cm i.d.). The different replicates were labeled in the same order along each transect at the different sampling dates (Fig. 1b). The cores were 60 cm long for T1 and 120 cm long for T2 to T5. They were separated into 5-cm increments in the depth zone where the Br⁻ peak was likely to occur, i.e., from the 0- to 20-cm depth for T1, from 0 to 40 cm for T2, from 0 to 60 cm for T3,

and between 30 and 60 cm for T4. The rest of the cores were divided into 10-cm increments to limit the number of samples to analyze. For the same reason, the five small-core replicates collected with the hydraulic auger were mixed together according to depth to give one average profile per plot, except for the GWS plot. For the GWS plot, each core replicate was analyzed separately from the others to provide a better idea of the within-plot variability. Finally, six replicate profiles were available for the GWS plot at each date, while only two were available for the other plots, one of them resulting from pooling five sampled profiles. Before Br^- extraction, each sample had its gravimetric water content measured and converted to volumetric water content using the average bulk density profiles. These volumetric water contents were used to statistically detect significant differences between treatments as well as for TDR probe calibration.

Soil extraction was made using a 2:1 soil/deionized water ratio. Samples were shaken for 24 h, and then decanted to separate the supernatant, which was then pushed through a 0.45 μm nylon filter (AIT, France). The Br^- concentrations were analyzed using a Waters HPLC Alliance chain equipped with an anion exchange column (A-2 anion 7u 100 mm X 4.6 mm, Alltech) and a photodiode array detector (Waters 996, Waters Corp., Milford, MA). A volume of 50 μm was injected into the column with a mobile phase at a flow rate of 2 mL min^{-1} . The mobile phase was 2.2 mmol L^{-1} Na_2CO_3 and 2.8 mmol L^{-1} NaHCO_3 . The detection wavelength was 195 nm and the Br^- retention time was 6.8 min.

Data Analysis

The climatic water balance was calculated from measured rainfall and from potential soil evaporation calculated with the Penman (1948) equation.

Spatial moment analysis (Freyberg, 1986; Sun et al., 2001; Das et al., 2005; Govindaraju and Das, 2007) was used to characterize each Br^- concentration profile. Spatial moments were used to characterize (i) the total mass moving through the soil profile (zeroth moment, μ_0), (ii) the distance traveled by the center of mass of solute (first normalized moment, Z_1), and (iii) the variance around the mean travel depth, $\text{Var}_z = Z_2 - Z_1^2$ (where Z_2 is the second normalized moment) (Valocchi, 1985; Freyberg, 1986). The general equation for the normalized spatial moment of order N is

$$Z_N = \frac{m_N}{m_0} = \frac{\int_0^{\infty} z^N \rho_b C_r dZ}{\int_0^{\infty} \rho_b C_r dZ} \quad [1]$$

where μ_0 [M L^{-2}] is the total mass of Br^- recovered per unit area, z [L] is depth, ρ_b [M L^{-3}] is bulk density, and C_r [M M^{-1}] is the total resident concentration expressed as mass of Br^- per mass of dry soil. Mass recovery m_r for resident concentrations (dimensionless) is defined as the ratio of μ_0 to the total applied mass per unit area. The standard deviation was calculated for each moment from the six (for GWS) or two (for the other treatments) replicate profiles at each sampling time.

Because the number of replicates differed among treatments, unbalanced ANOVAs were used to test for effects of organic amendment. Bonferroni tests (Sheskin, 2004) were used to statistically detect differences in volumetric water content or Br^- concentration spatial moments

between treatments. The replicates that resulted from the pooling of five different profiles were given a weight five times that of the other replicates. Whether there were significant differences in Br^- mean travel depth between the two types of core sampling (hydraulic vs. manual cores) were determined by *t*-tests.

To get insight into the type of solute transport process in the studied soil, analytical solutions of the convection–dispersion equation (CDE) and stochastic–convective or stream–tubes model (SCM) were fitted to the measured resident Br^- concentration profiles using CXTFIT (Toride et al., 1999), using a steady-state flow approximation. For the CDE, flow rate and soil dispersivity were fitted together with the duration of the solute pulse at the soil surface. For the SCM, the average flow rate and its standard deviation were fitted while soil dispersivity was set at 1 cm in each stream tube. Pulse duration was also fitted to get a proper Br^- mass balance. To test for the robustness of the CDE and SCM in describing solute transport, the soil dispersivity and the flow rate standard deviation, respectively, obtained from the concentration profiles measured at T1 were used to predict the concentration profiles at T2 to T5.

RESULTS AND DISCUSSION

Soil Water Dynamics

The cumulative climatic water balance increased between the Br^- tracer application date and the end of March (Fig. 2). The first four soil coring dates (T1–T4) occurred during this wetting period. During that time, the soil water content in the upper 20 cm was near or larger than field capacity, taken here as -100 cm matric head (see the distance from the gravity line to the hydraulic head profiles in Fig. 3).

Between T0 and T4, frequent rainfalls established a soil water regime close to steady state, as shown by the limited variation in water content and hydraulic head with time; however, the soil profile was not completely recharged by the date of Br^- application (T0). The hydraulic head profile (Fig. 3b) showed that the infiltration front was located between the 80- and 130-cm depths at that time. It reached the bottom of the profile on 5 Dec. 2007. After that date and until the end of March 2008, the hydraulic head gradient remained close to unity below the 40-cm depth (Fig. 3b, 3d, and 3f), which means that during that time, water below that depth moved downward due to gravity only. Between T2 and T3, a rainstorm (17 mm) occurred on 1 February and caused a transient increase in water content (Fig. 3c) and hydraulic head (Fig. 3d) in the upper horizons. Afterward, a lack of rainfall (Fig. 2) caused the water content and hydraulic head to decrease at the top of the profile. During the same time, the rest of the hydraulic head profile moved parallel to the gravitational gradient line due to free drainage at the bottom of the profile. The weekly water balance was largely positive between T3 and T4 (Fig. 2), so that the hydraulic head profile stayed parallel and close to the gravity gradient line during that time (Fig. 3f).

After T4 (no data available at T5), the high evaporative demand led to a negative water balance (Fig. 2) and created an upward water flux in the topsoil and a zero-flux plane at 60 cm until 27 May (Fig. 3f). The decrease in water content was limited to the upper horizons (Fig. 3e) and did not exceed $0.06 \text{ cm}^3 \text{ cm}^{-3}$ at the 20-cm depth during this evaporative period. The evaporative profile disappeared after an intense rain (36 mm) on 28 May 2008 (Fig. 2), which caused a noticeable increase in water content along the profile (Fig. 3e). The hydraulic head

profile observed immediately after the event (28 May 2008 in Fig. 3f) is interesting because of its heterogeneous behavior. Although the infiltration front did not pass 1 m, a quasi-vertical gradient at the bottom of the soil profile was observed and was probably caused by a transient perched water table occurring above the buhrstone clay subsurface layer located at the 2.5-m depth. Such a transient water table in the deep soil had already been observed at the same site during large rainfall events (Nicole et al., 2003) and was assumed to be due to preferential water infiltration along macropores down below the soil profile.

All treatment plots had approximately the same soil water profile dynamics (Fig. 4). At any time during the experiment, water content differences between treatment plots did not exceed $0.05 \text{ cm}^3 \text{ cm}^{-3}$. While the measured hydraulic heads were very similar at all depths for all plots (data not shown), the measured water contents differed along the profile (Fig. 4). The eluvial (E) horizon at 40 cm retained less water than the illuvial (Bt) horizons at 60 to 130 cm at all times. This is consistent with the soil textural differences created by pedogenic clay translocation processes (Diab et al., 1988).

An ANOVA revealed a significant effect ($P < 0.05$) of organic amendments throughout the experiment, although two replicates only were available per plot (except the GWS plot). Bonferroni tests ($P < 0.05$) showed that the water content in the 0- to 30-cm tilled layer was on average $0.03 \text{ cm}^3 \text{ cm}^{-3}$ larger in the GWS and MSW plots and $0.015 \text{ cm}^3 \text{ cm}^{-3}$ larger in the BIO and FYM plots than in the CONT plot. This result is consistent with experiments reviewed by Khaleel et al. (1981), which showed increased water contents at -33 kPa for soils treated with organic wastes. Other researchers have also found that soils treated with organic amendments had larger water retention (Hollis et al., 1977; Emerson, 1995; Rawls et al., 2003). Hudson (1994) showed that the water content at field capacity increased by 3.2 times the increase in the organic matter content. In our study, after 10 yr of compost applications at a rate of $4 \text{ Mg organic C ha}^{-1}$, the organic matter content increased by 0.3% for the MSW, 0.6% for the GWS and FYM, and 0.7% for the BIO treatment (Table 2; Houot et al., 2007), which gives values of the water content increase/organic matter increase ratio between 2.1 and 10 for the BIO and MSW treatments, respectively. This variability is probably due to different organic matter qualities of the amendments. The BIO compost had the highest lignin content and lowest cellulose and hemicellulose contents, while the MSW compost had the opposite biochemical characteristics (Table 1).

Bromide Transport in the Green Waste–Sewage Sludge Compost Treatment Plot

Each profile sampled in the GWS treatment plot was analyzed separately for its Br^- content to gain information on the local variability of soil transport properties. The six Br^- concentration profiles taken at each sampling date looked similar to each other (Fig. 5). Some concentration profiles had two maximum values (Replicate R1 for T1, R3 for T2, R1 for T3, and R4 for T4). Such bimodal profiles are characteristic of soils with macropores or heterogeneous soils that exhibit funneled or fingered flows (Ghodrati and Jury, 1990; Coquet et al., 2005b). Bimodal profiles have been observed in field experiments as a result of heterogeneities in the plowed layer (Petersen et al., 2001; Coquet et al., 2005a). The glossic Btgd horizon might also account for bimodal profiles, with Br^- tracer partitioned between fast-moving interped zones and slower moving zones within peds (Diab et al., 1988; Vanderborght et al., 2001). Bromide transport heterogeneity remained large throughout the experiment. Some replicates (e.g., R4 at T1, R5 at

T2, R3 at T3, and R1 at T5) were different from the others and had their Br^- center of mass significantly deeper than in the other profiles. At T2, the Br^- center of mass was approximately 38 cm for R5 and between 20 and 25 cm for the other replicates. The limited rainfall between T2 and T3 (Fig. 2) produced only limited changes in the T3 concentration profiles compared with the profiles observed at T2 (Fig. 5b and 5c). Later on, the Br^- peak widened and the maximum concentration value decreased as Br^- moved deeper into the soil (Fig. 5d and 5e).

The CDE model was fitted to the different single concentration profiles measured at the different sampling dates using CXTFIT (Toride et al., 1999). The R^2 values ranged from 0.863 to 0.994, except for profiles exhibiting increased Br^- concentrations in the soil surface horizon due to soil evaporation, whose R^2 ranged between 0.729 and 0.791. Local soil dispersivity calculated from each replicate Br^- concentration profile was highly variable (Fig. 6). Coefficients of variation of this local dispersivity calculated from the six simultaneous profiles of the GWS treatment were 77% for the first sampling date and about 50% for the other sampling dates (Table 3). Variability in local soil dispersivity can be explained by spatial variations in local water velocities related to soil structure heterogeneities such as those found in the plowed layer (Coquet et al., 2005a) or in Btgd horizons. The arithmetic average value of the dispersivities estimated from the individual profiles was similar to the dispersivity estimated from the averaged concentration profile (Fig. 6).

Two types of samplers were used to collect soil profile samples. The Br^- profile sampled at T1 with the cylinder (Replicate R4) was more spread out than the other profiles sampled with the hydraulic auger (Fig. 5a), but this was not true at the other sampling dates. No significant difference was observed between the two sampling devices based on t -tests. Consequently, the average Br^- concentration profile for the GWS treatment was calculated by averaging the six profiles regardless of the sampling device used.

Treatment Effect on Bromide Transport

Mean Br^- mass recoveries, m_r , were between 68 and 104% for all treatments (Table 3). No systematic explanation was found for the occasional mass balance deficits. The m_r coefficient of variation measured in the GWS treatment increased from 20% at T1 to 45% at T4 and then decreased to 9% at T5. In December (T1), the mean travel depths were similar for all treatments (Table 3). A positive cumulative water balance of 130 mm (Fig. 2) from T0 to T4 was required for the Br^- center of mass to reach a depth of approximately 0.43 m in all treatments (Table 3). The ANOVAs performed on the Z_1 data at each sampling date showed no significant difference ($P > 0.05$) between treatments from T1 to T4. The T1 to T4 sampling campaigns were performed during a wetting period, with low evaporative demand. All treatments could be considered to have the same leaching potential during this wetting period, when soil water content was near field capacity (Fig. 3). This result is consistent with the very limited and transient effect of compost addition on soil near-saturated hydraulic conductivity observed by Schneider et al. (2009) in the same plots.

After T4, the soil surface dried out from evaporation (Fig. 2). Hydraulic head profiles observed during this drying period (Fig. 3f) indicated an upward flow in the upper horizons, except just after 28 May 2008. Percolation continued but was limited to deep horizons below the zero-flux plane. As a consequence, the Br^- transport to deep layers was limited. The mean Br^- travel depth moved about 2 to 7 cm deeper between T4 and T5 in the treatments receiving

organic amendments (Table 3). For the control plot, the mean travel depth moved upward, about 6 cm above that at T4. Bonferroni tests on the Z_1 moment of the Br^- concentration profiles at T5 showed a significant difference ($P < 0.05$) between the CONT and the GWS and MSW treatments. To further study treatment effects on the Br^- distribution in the soil during the evaporative stage, we calculated the amount of water held in the soil layer between the surface and the Br^- mean depth for each treatment. The amount of water stored above the mean travel depth was calculated from the soil volumetric water content profiles (Fig. 4). The differences in the water amount stored above the mean Br^- travel depth at T5 were related to the differences in mean travel depth between treatments. The lowest amount, 110 mm, was observed in the CONT treatment associated with the lowest mean travel depth (39 cm). For the other treatments, water storage above the mean Br^- travel depth was 160 mm for GWS and MSW, 140 mm for FYM, and 130 mm for BIO. Organic amendments, and especially the GWS and MSW treatments, appeared to be efficient in reducing water evaporation from bare soil. Abdel-Nasser et al. (2007) also found that organic fertilizer application to a sandy soil increased the water holding capacity and reduced cumulative evaporation.

The increase in solute longitudinal spreading with mean travel depth was similar for all treatments (Fig. 7) except for the CONT and FYM treatments. Solute dispersion around the mean travel depth was significantly larger ($P < 0.05$) from T3 to T5 for the CONT treatment and at T5 for the FYM treatment. The CONT treatment had the highest solute spreading for all dates (Table 3).

The CDE was fitted to the average concentration profile at each date and for each treatment (Table 3). The dispersivity values obtained increased with travel depth and time (Fig. 6), showing that the CDE model was unable to predict concentration profiles at all times from a single profile taken at a particular time. The linear increase in dispersivity with travel depth suggests that the transport process would be better described by the SCM (Simmons, 1982). To compare the CDE and SCM performances in describing Br^- transport, each transport model was fitted to the average concentration profile measured at T1 and used to predict the other profiles at T2 to T5, assuming either a constant dispersivity value for the CDE or a constant standard deviation of the water flow rate for the SCM (Fig. 8). Globally, the SCM performed better than the CDE, with average R^2 values of 0.782 and 0.660, respectively. The SCM clearly outperformed the CDE for the CONT and GWS treatments (Fig. 8), with average R^2 values of 0.707 vs. 0.530 and 0.866 vs. 0.383, respectively, but less so for the BIO treatment (0.886 vs. 0.848). On the other hand, the CDE was more adapted than the SCM to describe Br^- transport in the MSW and FYM treatments (data not shown), with average R^2 values of 0.850 vs. 0.766 and 0.739 vs. 0.684, respectively. Furthermore, the best transport model could be different according to the sampling date. For instance, the average concentration profiles measured at T1, T2, and T5 in the BIO treatment were better described by the SCM, while profiles measured at T3 and T4 were better described by the CDE (Fig. 8). Looking more closely to these two profiles, it can be seen that the actual transport process is somehow in between the two extreme lateral mixing regimes represented by the CDE and the SCM (Flühler et al., 1996). Vanderborght et al. (2001) linked the incomplete mixing regime of solutes in an Albeluvisol to the pedogenetic morphological features of the soil profile. Elementary components of the heterogeneous Btgd horizons might have different hydraulic properties (Frison et al., 2009).

Between T4 and T5, evaporation was larger than between the previous sampling dates. Alternate upward and downward flows may also have increased solute dispersion because of variations in pore-water velocity magnitude and direction (Bechtold et al., 2011). This could explain the better performance of the SCM than the CDE for the concentration profiles measured at T5. The large solute spreading observed in the CONT treatment at T5 (Table 3) can be explained by the larger contribution of upward flow caused by larger evaporative water losses than in the other treatments (Fig. 7).

CONCLUSIONS

Ten years of biennial compost or farmyard manure applications to soil only slightly affected the water regime during the recharge period. From December to early June, the water content in the 0- to 30-cm plow layer was on average $0.03 \text{ cm}^3 \text{ cm}^{-3}$ larger for the plots that received MSW or GWS and $0.015 \text{ cm}^3 \text{ cm}^{-3}$ larger for those that received BIO or FYM than the control plot. During the high evaporative demand period (April–June), organic amendment applications, and especially those of the composts, reduced the soil cumulative evaporation.

Regarding inert solute transport, the control and FYM treatments had larger solute spreading at the end of the experiment in June than the treatments with organic amendments. The Br^- mean travel depth decreased between April and June in the control plot, while it increased in the other plots. This could be explained by the larger cumulative evaporation observed in the control plot during this period. The dispersivity was highly variable locally along 10-m transects. Coefficients of variation for dispersivity ranged between 11 and 175%. This variability may be explained by local variations in water velocities related to soil structure heterogeneities such as those found in the plowed layer or in the Btg horizon. The bimodal profiles occasionally found for Br^- transport were consistent with this hypothesis.

Dispersivity increased with travel depth. The SCM was generally better adapted than the CDE to predict Br^- transport in the studied soil, especially at the end of the experiment when soil evaporation influenced Br^- transport; however, the CDE sometimes outperformed the SCM for particular combinations of treatments and sampling times.

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Fig. 1. (a) Locations of time-domain reflectometry probes, tensiometers, and the Br⁻ experiment within each plot, and (b) locations (crosses) of the six replicates (R1–R6) taken from the soil for Br⁻ concentration measurements at the different sampling dates (T1–T5).

Fig. 2. Temporal evolution of cumulative and weekly climatic water balances. Points show the application date (T0) and the five sampling dates (T1–T5).

Fig. 3. Volumetric water content (θ) and hydraulic head (H) profiles during the tracer experiment for the co-compost of green wastes, wood chips and sewage sludge (GWS) treatment: (a) and (b) between sampling dates 28 Nov. 2007 (T0) and 23 Jan. 2008 (T2); (c) and (d) between T2 and 6 Mar. 2008 (T3); and (e) and (f) after T3.

Fig. 4. Water content (θ) profiles measured during the tracer experiment at sampling date T1 (11 Dec. 2007), T2 (23 Jan. 2008), T3 (6 Mar. 2008), T4 (1 Apr. 2008), and T5 (6 June 2008) in the various treatment plots: co-compost of green wastes, wood chips and sewage sludges (GWS), municipal solid waste compost (MSW), biowaste compost (BIO), farmyard manure (FYM), and the control (CONT). Error bars indicate ± 1 standard error of the mean.

Fig. 5. Bromide concentration profiles in the soil solution for the six replicates (R1–R6) of the co-compost of green wastes, wood chips, and sewage sludge (GWS) treatment on (a) 11 Dec. 2007 (T1, 13 d after application [DAA]), (b) 23 Jan. 2008 (T2, 56 DAA), (c) 6 Mar. 2008 (T3, 99 DAA), (d) 1 Apr. 2008 (T4, 125 DAA), and (e) 6 June 2008 (T5, 191 DAA).

Fig. 6. Soil dispersivity calculated with CXTFIT for the co-compost of green wastes, wood chips, and sewage sludge (GWS) treatment for the different sampling dates (T1–T5). Soil dispersivity was calculated either by averaging the dispersivities obtained from the six profile replicates or by fitting the convection–dispersion equation to the average concentration profile. Pore water velocity corresponds to the average value calculated from the six individual core replicates. Error bars indicate ± 1 standard deviation.

Fig. 7. Solute spreading Var_z vs. mean travel depth Z_1 for the different treatments: co-compost of green wastes, wood chips and sewage sludges (GWS), municipal solid waste compost (MSW), biowaste compost (BIO), farmyard manure (FYM), and the control (CONT). Error bars indicate ± 1 standard error of the mean.

Fig. 8. Bromide concentration profiles for the control (CONT), co-compost of green wastes, wood chips and sewage sludges (GWS), and biowaste compost (BIO) treatments predicted by the convection–dispersion equation (CDE) or the stochastic–convective model (SCM) based on dispersivity or flow rate variance, respectively, for treatment dates T1 (13 d after application) to T5 (191 d after application) fitted on the measured profile on T1. Stream-tube dispersivity was fixed at 1 cm.

Table 1. Characteristics of the organic amendments: co-compost of green wastes, wood chips, and sewage sludge (GWS), municipal solid waste compost (MSW), biowaste compost (BIO), and farmyard manure (FYM).

Characteristic†	GWS	MSW	BIO	FYM
pH (in H ₂ O)	7.6 \pm 0.7‡	7.3 \pm 0.4	8.4 \pm 0.4)	9.0 \pm 0.2
OM, g kg ⁻¹ dry wt.	48.8 \pm 5.8	59.3 \pm 11.2	35.1 \pm 5	60.2 \pm 11
Organic C, g kg ⁻¹ dry wt.	267.8 \pm 56.2	317.9 \pm 48.6	184 \pm 25.3	325.5 \pm 52.1
Total N, g kg ⁻¹ dry wt.	23.2 \pm 3.3)	17.4 \pm 2.4	15.5 \pm 3.8	22.8 \pm 2.6
C/N ratio	11.8 \pm 2.8)	18.8 \pm 4.7	12.2 \pm 1.9	14.5 \pm 3.0
Soluble OM, % of OM	45.7 \pm 9.4	42.5 \pm 8.7	44.5 \pm 7.8	39.1 \pm 5.9
Hemicellulose, % of OM	5.1 \pm 2.1	6.4 \pm 2.3	4.2 \pm 2.8	12.0 \pm 2.9

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Cellulose, % of OM	20.8 ± 10.1	35.8 ± 10.3	20.3 ± 2.3	25.5 ± 7.7
Lignin, % of OM	28.5 ± 11.9	15.2 ± 2.4	31.1 ± 8.9	23.4 ± 2.5
CaCO ₃ , g kg ⁻¹ dry wt.	23.3 ± 11.9	62.1 ± 10.3	120.5 ± 51.9	48.1 ± 16.8
P ₂ O ₅ , g kg ⁻¹ dry wt.	27.6 ± 8.5	7.3 ± 1.2	10.6 ± 4.8	12.7 ± 3.0
K, g kg ⁻¹ dry wt.	14.9 ± 5.5	8.8 ± 1.4	20.5 ± 3.5	35.7 ± 3.2
Ca, g kg ⁻¹ dry wt.	34.4 ± 8.5	59.6 ± 4.1	67.3 ± 20.8	30.2 ± 9.3
Mg, g kg ⁻¹ dry wt.	4.7 ± 0.7	10.2 ± 4.1	5.0 ± 0.6	5.7 ± 1.2
Na, g kg ⁻¹ dry wt.	4.8 ± 1.5	15.1 ± 6.6	4.9 ± 1.7	8.1 ± 3.0
Water content, % (w/w)	66.9 ± 19.7	58.6 ± 40.0	52.1 ± 17.2	162.6 ± 65.5
Bulk density, Mg m ⁻³	0.42 ± 0.03	0.29 ± 0.02	0.41 ± 0.03	0.21 ± 0.03

† OM, organic matter content; biochemical fractions according to Association Française de Normalisation (2005).

‡ Means ± standard errors of the six organic amendments applied since 1998, except for bulk density whose values correspond to the amendments applied in 2006.

Table 2. Main soil characteristics of the plowed layer (0–28 cm) measured in September 2007 for the control, co-compost of green wastes, wood chips, and sewage sludge (GWS), municipal solid waste compost (MSW), biowaste compost (BIO), and farmyard manure (FYM) treatments.

Treatment	Bulk density Mg.m ⁻³	Organic matter %	Organic C — g kg ⁻¹ dry wt. —	Total N	Organic C/N ratio	pH (H ₂ O)	Cation exchange capacity cmol kg ⁻¹	Mi bi mg C k
Control	1.33 ± 0.05 ab†	1.8 ± 0.1 a	10.5 ± 0.3 a	1.1 ± 0.0 a	9.9 ± 0.4 a	6.8 ± 0.1 a	8.53 ± 0.76 a	244.0 ±
GWS	1.35 ± 0.03 b	2.4 ± 0.1 c	14.0 ± 0.4 c	1.4 ± 0.1 c	10.0 ± 0.2 a	6.9 ± 0.1 a	9.19 ± 0.54 ab	278.0 ±
MSW	1.28 ± 0.06 ab	2.1 ± 0.0 ab	12.0 ± 0.2 ab	1.2 ± 0.0 b	9.9 ± 0.2 a	7.4 ± 0.1 b	9.34 ± 0.53 ab	291.0 ±
BIO	1.24 ± 0.02 ab	2.5 ± 0.2 c	14.3 ± 1.4 c	1.4 ± 0.1 c	10.5 ± 0.3 a	7.7 ± 0.1 c	10.34 ± 0.56 b	304.0 ±
FYM	1.28 ± 0.05 ab	2.4 ± 0.2 bc	13.8 ± 0.9 bc	1.3 ± 0.1 bc	10.4 ± 0.3 a	7.2 ± 0.1 b	9.80 ± 0.45 ab	287.1 ±

† Means of four replicates ± standard errors. Means followed by different letters in a column indicate significant differences at the 5% level.

Table 3. Bromide concentration profile spatial moments for the co-compost of green wastes, wood chips, and sewage sludge (GWS), municipal solid waste compost (MSW), farmyard manure (FYM), biowaste compost (BIO), and control (CONT) treatments at T1, T2, T3, T4, and T5 sampling dates, including the number of days after Br⁻ application (DAA), mean Br⁻ recovery (m_r), the first normalized spatial moment or center of mass (Z_1), spreading around the center of mass (Var_z), and soil dispersivity (λ), and values in parentheses are standard deviations.

Sampling date	Cumulative rainfall mm	DAA d	m_r %	GWS				MSW				FYM	
				Z_1 cm	Var_z cm ²	λ cm	m_r %	Z_1 cm	Var_z cm ²	λ cm	m_r %	Z_1 cm	V cm ²
T1	57	15	82 ± 16†	16 ± 3	88 ± 38	4 ± 3	90 ± 1	17 ± 2	92 ± 23	4 ± 1	86 ± 5	16 ± 2	99 ±
T2	97	55	71 ± 14	26 ± 5	196 ± 93	5 ± 3	80 ± 7	27 ± 3	198 ± 46	5 ± 1	76 ± 4	24 ± 2	200
T3	150	101	68 ± 24	29 ± 5	300 ± 115	7 ± 3	80 ± 16	35 ± 5	283 ± 50	5 ± 1	94 ± 12	31 ± 4	342
T4	226	126	71 ± 27	45 ± 6	503 ± 132	10 ± 5	69 ± 11	44 ± 4	345 ± 22	5 ± 1	90 ± 18	40 ± 2	495
T5	266	190	90 ± 9	51 ± 5	505 ± 98	13 ± 7	80 ± 9	51 ± 8	608 ± 12	8 ± 2	100 ± 10	44 ± 1	725

† Means ± standard deviations.

Fig. 1.

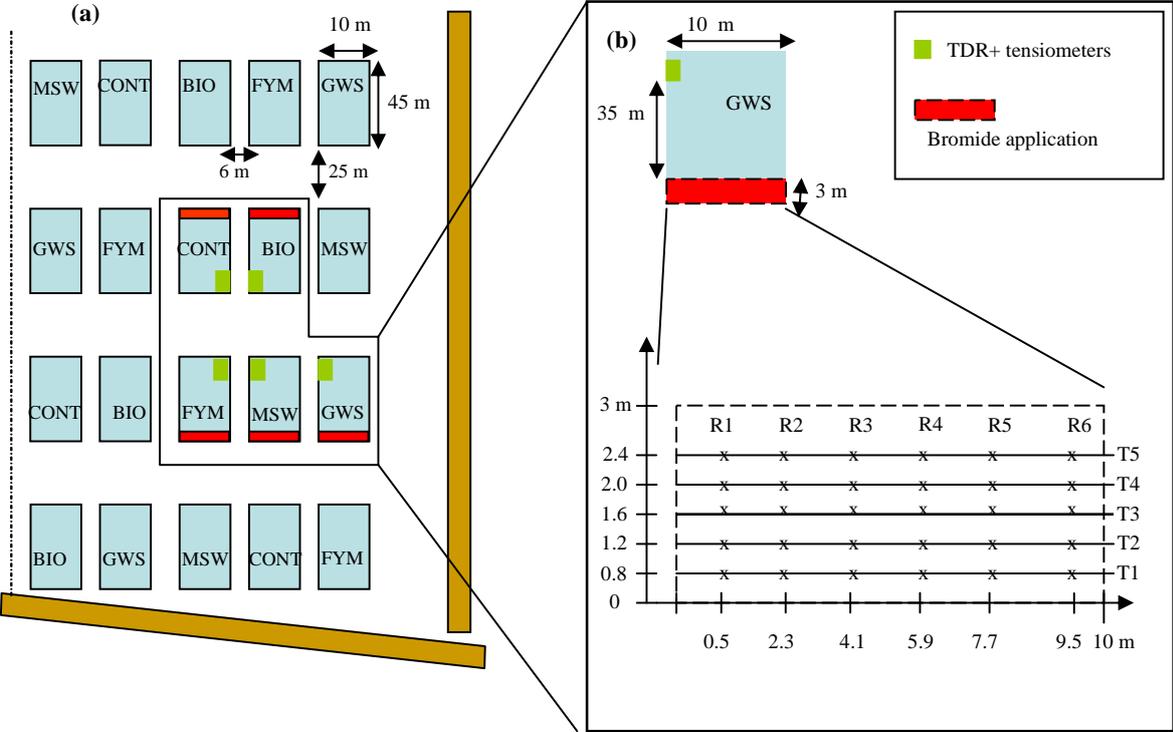


Fig. 2.

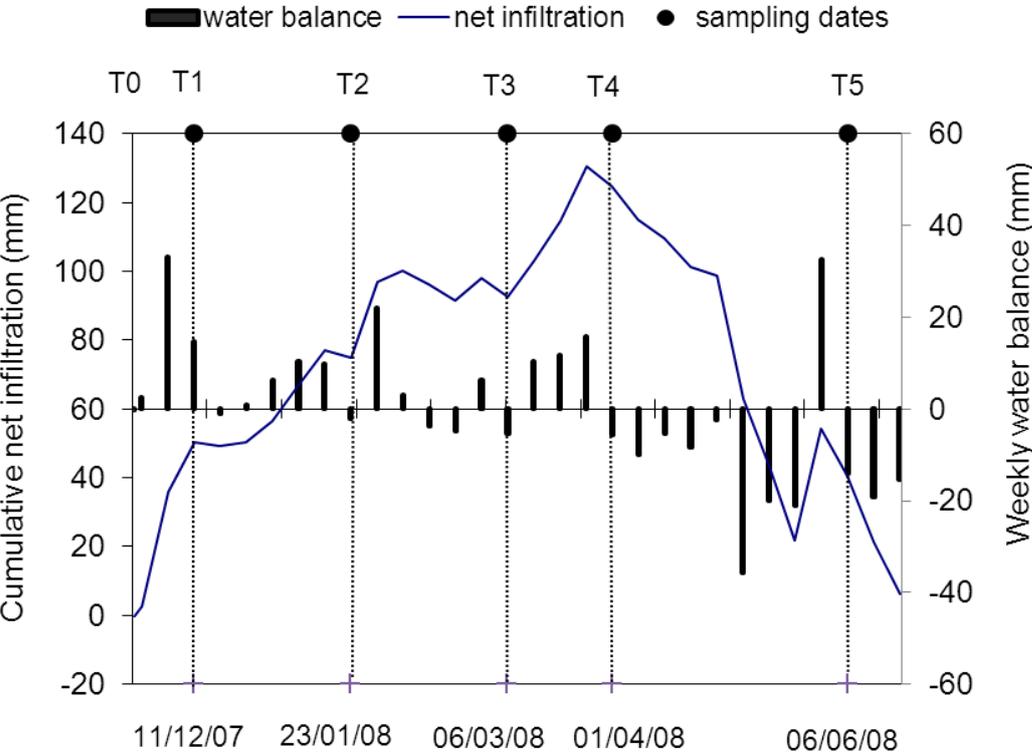


Fig. 3.

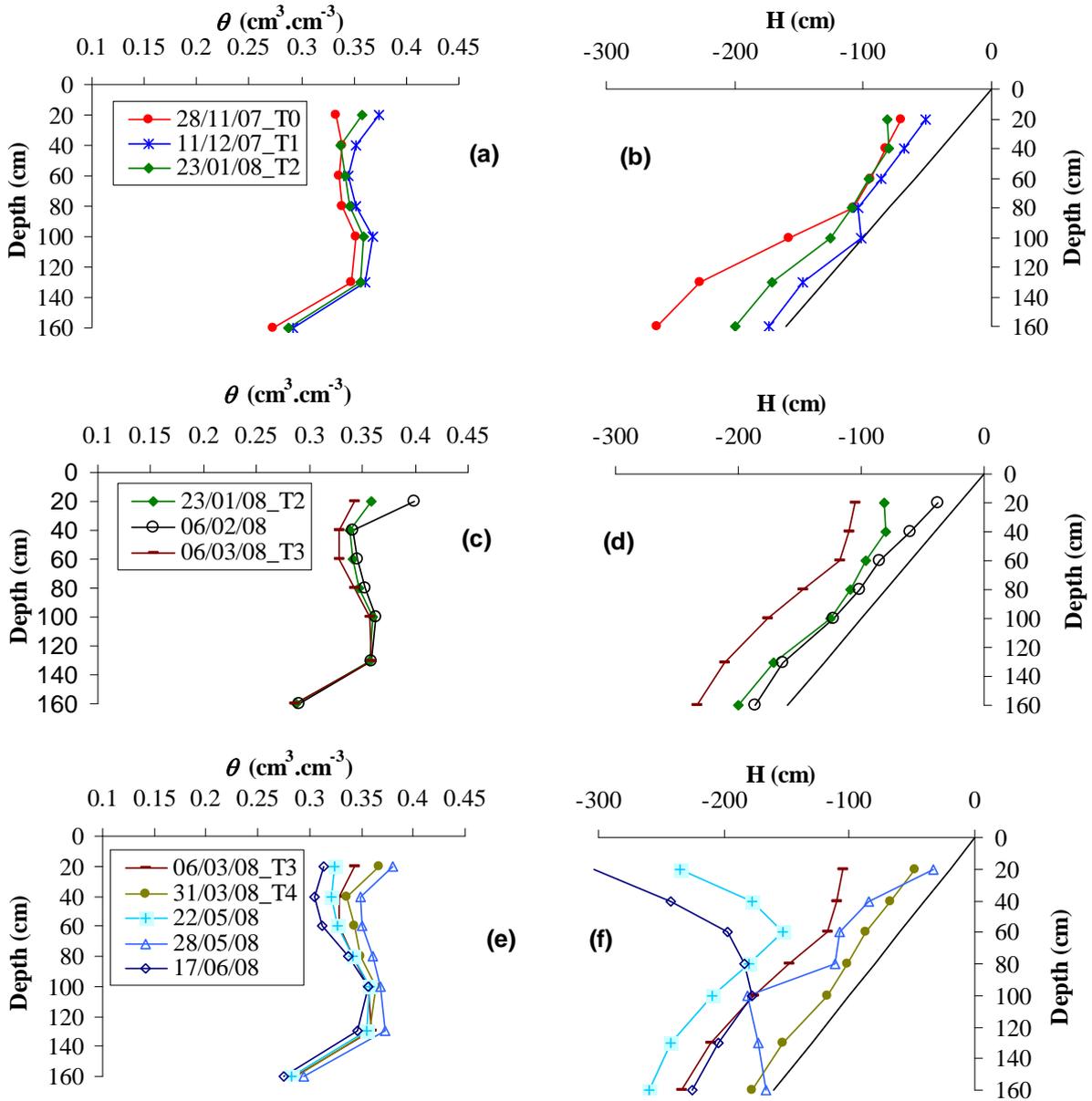


Fig. 4.

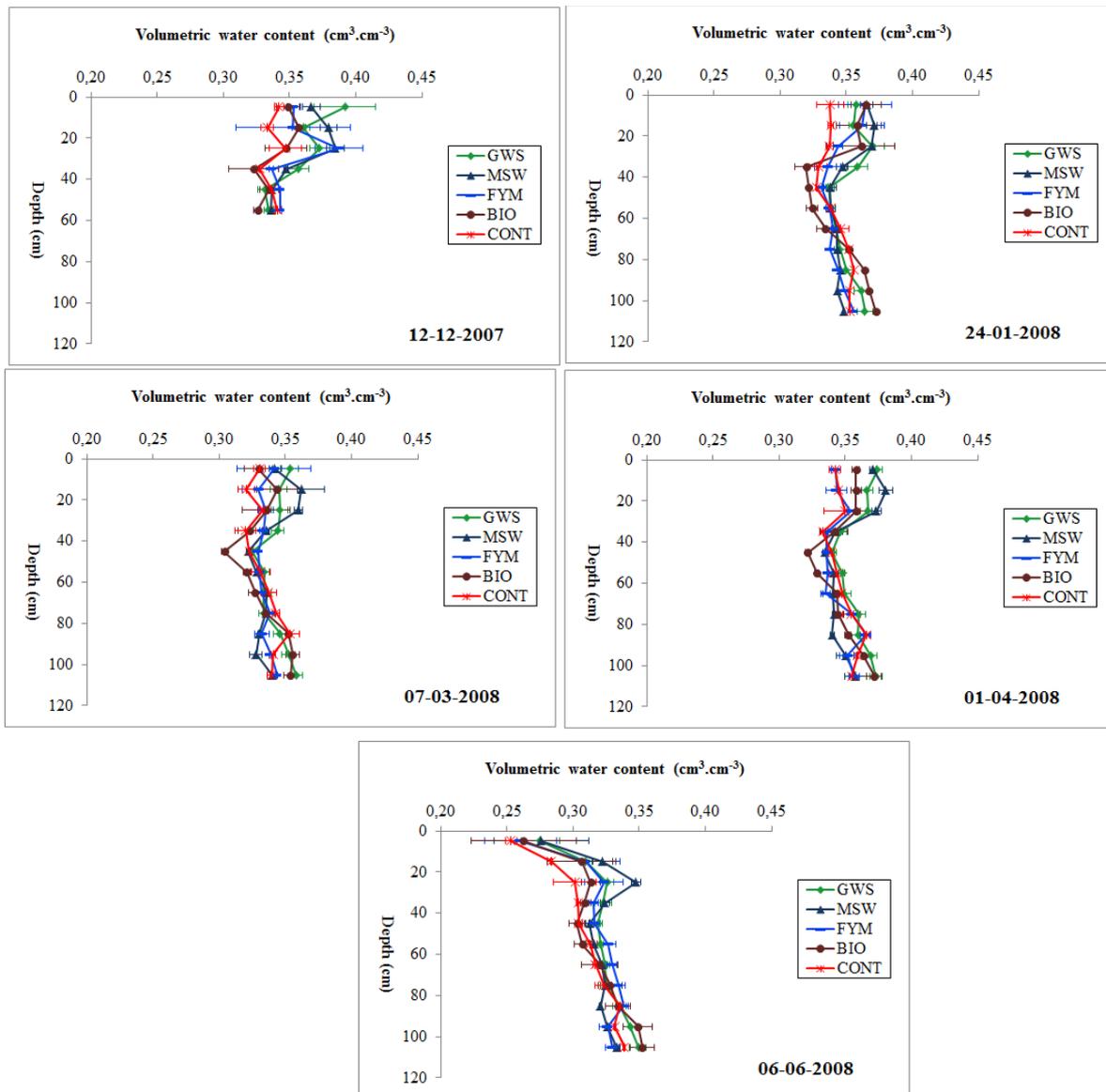


Fig. 5.

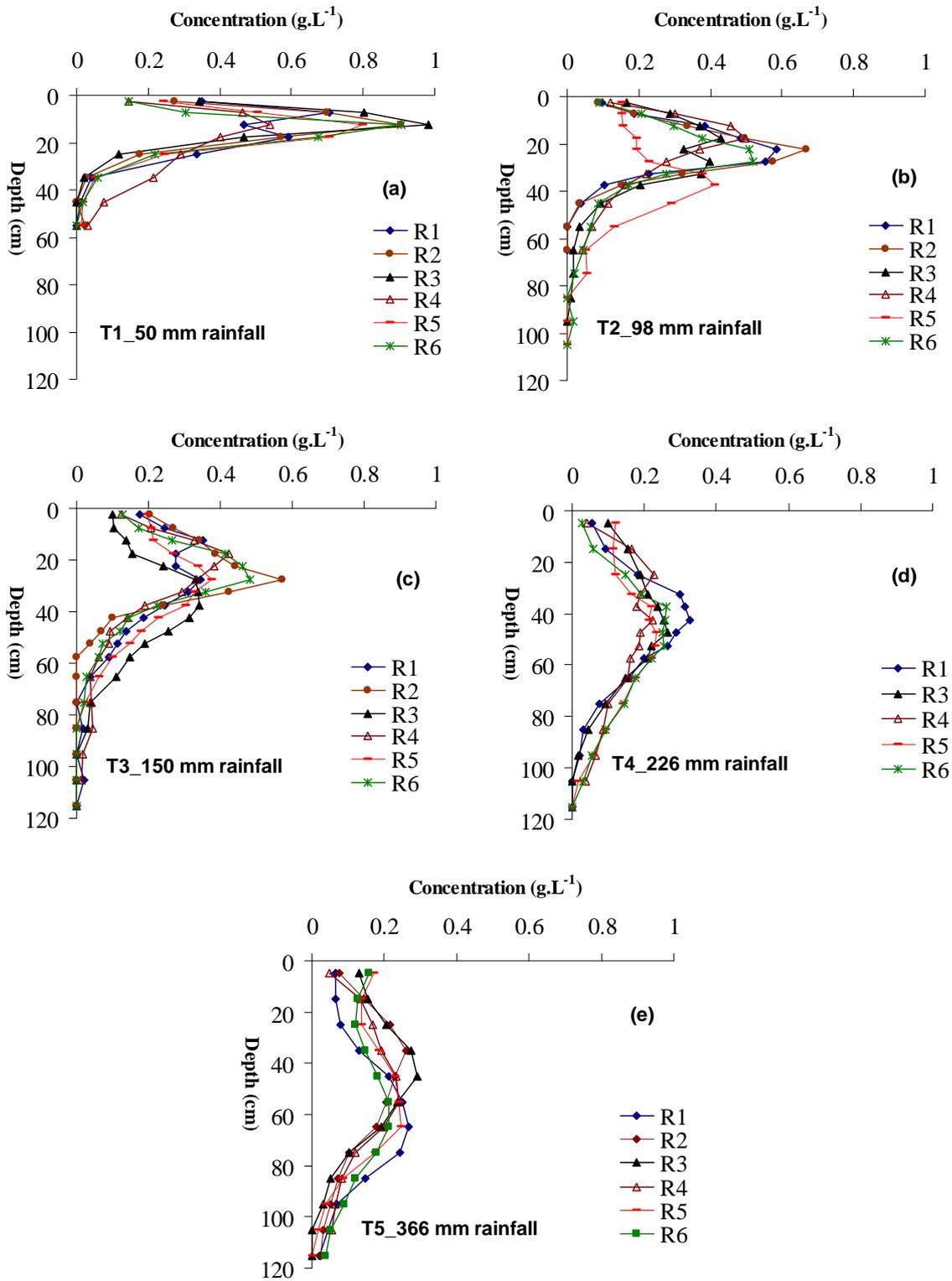


Fig.6.

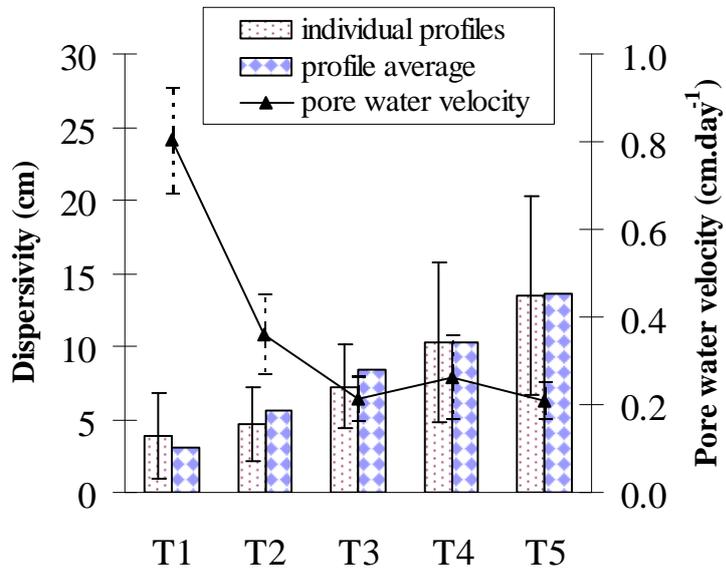


Fig. 7.

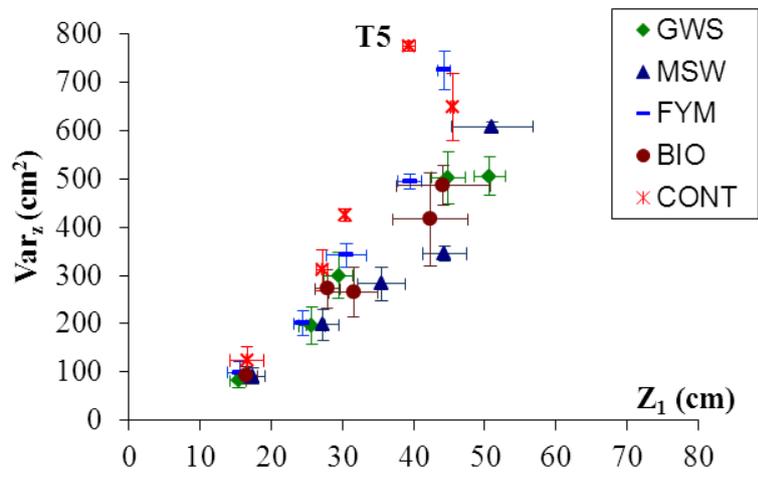


Fig.8.

