Effects of interrill erosion, soil crusting and soil aggregate breakdown on in situ CO_2 effluxes

S. Bremenfeld [a], P. Fiener [a,b,*], G. Govers [c]

[a] Geographisches Institut, Universität zu Köln, Albertus Magnus Platz, 50923 Köln, Germany
[b] Indo-German Centre of Sustainability, Indian Institute of Technology Madras, Chennai 600036, India
[c] Physical and Regional Geography Research Group, Katholieke Universiteit Leuven, Leuven, Belgium

Keywords:
Soil organic carbon (SOC)
Interrill erosion
Crusting
Soil aggregate breakdown
Soil respiration

A B S T R A C T

Soil and carbon redistribution on arable land and the associated impacts on carbon sequestration and mineralisation may play an important role in the global carbon cycle. While our insight in the process-chain of erosion, transport and deposition has significantly grown over recent years, there are still major gaps in understanding making it difficult to make an overall assessment of erosion processes on carbon exchange between the soil and the atmosphere. One issue is the potential effect soil degradation and erosion processes may have on CO_2 effluxes at eroding sites. The major goal of this study was therefore to analyse and understand the effects of interrill erosion, soil crusting and soil aggregate breakdown on in situ CO_2 effluxes. Therefore a set of rainfall simulations were carried out on bare loess-burden soil with different antecedent soil moisture content. All treatments were compared with controls protected from rain drop impact using a fine-meshed geotextile. As expected, runoff and sediment delivery was significantly larger on bare compared to covered soils, while surface runoff and sediment delivery increased (in most cases) with rising antecedent soil moisture as well as rainfall duration. Crust thickness increased with antecedent soil moisture and rainfall intensity and was in general smaller for the controls. However, variations in crust thickness did not result in significant differences in in situ measured CO_2 effluxes. Also the destruction of the soil crust after six to seven days of measurements did not have a significant effect. This leads to the conclusion that crusting and interrill erosion has no or only a minor effect on in situ CO_2 effluxes. Nevertheless, it should be recognised that topsoil carbon is preferentially removed due to interrill erosion which may result in additional CO_2 release at depositional sites or in stream and/or standing water bodies.

1. Introduction

Globally, soils are the largest terrestrial carbon (C) pool, storing approximately 1500 Gt C (Schlesinger, 2005) in the top 1 m and roughly 2300 Gt C in the uppermost 3 m (Jobbagy and Jackson, 2000). In comparison atmospheric C accounts for only approximately 30% (Schlesinger, 2005) and hence any increase or decrease in soil organic carbon (SOC) content due to land use or climate change will substantially affect the atmospheric CO_2 concentration.

Since end of the 1990s it has been recognised that soil erosion, especially on arable land, may have a substantial effect on the C fluxes between soil and atmosphere (Harden et al., 2008; Jacinthe and Lal, 2001; Lal, 2003; Smith et al., 2001; Stallard, 1998; Van Oost et al., 2007). However, global estimates of erosion-induced C fluxes are contradictory, ranging from roughly an atmosphere to soil flux of 1 Gt C yr^{-1} (e.g. Smith et al., 2001; Stallard, 1998) to a soil C source of the same magnitude (e.g. Lal, 2004). There are several reasons for the large differences or uncertainties which partly result from different global erosion estimates (e.g. Van Oost et al., 2007), but more importantly result from a lack in process understanding regarding the interplay between soil erosion/transport/deposition and soil C sequestration and mineralisation.

In general, three (Van Oost et al., 2009) to five key mechanisms affecting this interaction can be identified: (i) dynamic replacement of C at eroding sites (Harden et al., 1999), (ii) burial of SOC rich topsoil at depositional sites (Stallard, 1998), (iii) mineralisation during transport and shortly after erosion due to aggregate break down exposing previously protected SOC to microbial decomposition, (iv) selective erosion/transport/deposition of SOC leading to an enrichment of C in removed sediment (Quinton et al., 2006), and (v) transport and degradation by microbes in fluvial systems (Cole et al., 2007; Tranvik et al., 2009).

While the first two of these mechanisms can be studied to a certain extend by comparing erosion data with SOC pools (e.g. Van Oost et al., 2007), the latter mechanisms call for a short-term (event-based)

* Corresponding author at: Geographisches Institut, Universität zu Köln, Albertus Magnus Platz, 50923 Köln, Germany. Tel.: +49 221 470 7802; fax: +49 221 470 5124.
E-mail address: peter.fiener@uni-koeln.de (P. Fiener).

monitoring (e.g. Jacinthe et al., 2002; Wang et al., 2010). In the terrestrial system, especially the SOC decomposition during transport and due to aggregate breakdown at erosional as well as depositional sites, which may follow heavy rainfall, is not well understood. While some studies try to estimate the decomposition during transport based on somewhat artificial incubation experiments (e.g. Jacinthe et al., 2002) and others show first attempts to evaluate the short-term effects of deposition on in situ measured CO_2 effluxes (e.g. Van Hemelryck et al., 2010a; Van Hemelryck et al., 2010b), little is known regarding the direct effects of interrill erosion often associated with soil crusting (e.g. Le Bissonnais and Singer, 1992) on CO_2 effluxes (Novara et al., 2012). This is somewhat surprising as especially soil crusting may have multiple effects on these effluxes as it leads on the one hand to aggregate breakdown at the soil surface (Auerswald, 1995; Le Bissonnais and Arrouays, 1997) which exhibits previously encapsulated C to the atmosphere and on the other hand affects the permeability of soils (Assouline and Mualem, 2001) which affects soil moisture regime and reduces gas diffusion potentially lowering C mineralisation by aerobic microbes (Luo and Zhou, 2006).

The general objective of this study is to measure and analyse the effect of interrill erosion, soil crusting and soil aggregate breakdown erosion on in situ CO_2 effluxes. We used a controlled experimental setting allowing us to control the rate of crust formation via simulated rainfall and to investigate the soil response after crust removal and breakdown via in situ soil respiration measurements.

2. Material and methods

2.1. Experimental design

To determine the effects of heavy rainfall and hence crusting and interrill erosion on CO_2 effluxes a series of laboratory rainfall simulations were carried out. The laboratory rainfall simulator that was used is located at the Physical and Regional Geography Research Group at the Katholieke Universiteit Leuven (Belgium) and was a Lechler 460.788 nozzle (Lechler GmbH, Metzingen, GER) based simulator with a falling height of 3.25 m, which provides a homogeneous rainfall on area of roughly 2 m in diameter (Poesen et al., 1990). To test for homogeneity of rainfall an pre-experiment was carried out using four rain gauges (surface 0.01 m^2) within this 2 m diameter rainfall area which resulted in a spatial variability (given as coefficient of variation) of 5.5% for an average rainfall intensity of 38.9 mm h^{-1}. For the experiments a slightly lower rainfall intensity of approximately 30.0 mm h^{-1} was used which was applied for 0.5 h in two experiments and for 1.0 h in a third experiment. Rainfall events of this magnitude would have a return period of 2 yrs and 20 yrs in central Belgium, respectively (Willems, 2011). Demineralised water was applied during all experiments to avoid any change in rainfall erosivity due to solutes in water (Borselli et al., 2001). The rainfall intensity during the different experimental runs was measured using the same four rain gauges placed around the test soil trays.

The soil used for the experiment is a silt loam taken from the plough layer of an arable field close to Leuven, Belgium. It represents a typical soil of the very fertile but also erosion prone Belgium Loess Belt. The soil texture consisted of 6% sand, 79% silt and 15% clay, respectively, while the mean soil organic carbon concentration was 0.79 ($\pm$0.07)%. The soil was stored outside the laboratory in a 1.5 m deep concrete container for approximately two years, whereas a weed cover developed on the top during storage. Before filling the soil into the wooden soil trays used in this study (Fig. 1) it was taken into the laboratory, air-dried for three days and sieved at 0.02 m. During sieving plant residues and roots were handpicked and removed. A soil column of 0.22 m was established in the trays while filling it in approximately 0.03 m layers each compacted with a wooden board to reach a bulk density similar to arable topsoil of approximately $1.35 \cdot 10^3$ kg m^3. The bottom of soil trays was perforated

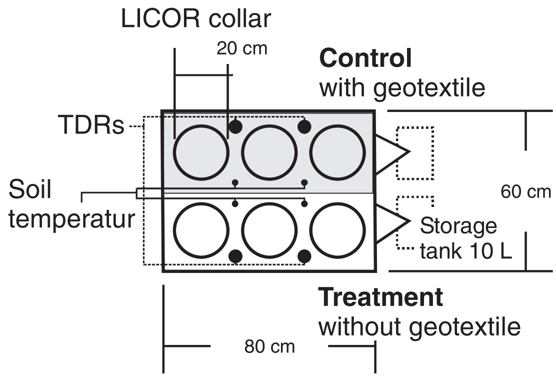

Fig. 1. Top view on the soil containers trays for the rainfall simulations; the LICOR collars to measure soil respiration and the time domain reflectometry (TDR) probes were installed after the rainfall application; soil temperature was measured at slightly changing locations.

and covered with a permeable textile to allow for free drainage. In total four 0.8 m long and 0.6 m wide soil trays with a slope of 3% were filled for the experiment. These soil trays were subdivided in two 0.3 m wide subsections used for a treatment vs. control set-up (Fig. 1).

The effect of interrill erosion, soil crusting and soil aggregate breakdown was evaluated by comparing bare soil in one half of the soil trays with covered, erosion-protected soil in the second half of the soil trays (Fig. 1). The controls were covered by a plastic mesh with a mesh width of 1 mm, tightened 0.02 m above the soil surface, thereby reducing interrill erosion to a minimum (Smets et al., 2007). This treatment vs. control step-up was used under different antecedent soil moisture conditions. In total four different antecedent soil moistures were tested in two consecutive experiments (Fig. 2). The first experiment consisted of two simulations with a rainfall duration of 30 min carried out with a gap of only 30 min. The first simulation had a mean antecedent soil moisture of 4.5% m^3 m^{-3}, while the second simulations had a mean antecedent soil moisture of 10.9% m^3 m^{-3}. As each simulation was carried out with two soil trays the treatments (different soil moistures) and the controls were replicated twice. Subsequently these simulations will be referred as dry and pre-wetted run, respectively. The pre-wetting of the second treatment was carried out with a capillary rainfall simulator mounted 0.5 m above the soil surface. To avoid any direct impact of rainfall during pre-wetting the rainfall intensity was limited to a few mm per hour (rainfall depth approximately 30 mm), and moreover all soil trays were covered with the 1 mm plastic mesh.

The second experiment consisted of one rainfall simulation with a duration of 60 min which was performed on one of the dry and one of the pre-wetted soil trays and their controls (Fig. 2). The mean antecedent soil moisture for this experiment was 13.8 and 18.5% m^3 m^{-3}, respectively. For the second experiment no true replicates of soil trays existed, but as the first set of simulations indicated that the differences in measured CO_2 effluxes (six rings per treatment, see below) within a treatment are relatively small, only one tray was tested per treatment in the third simulation. To indicate that the second experiment is based on the dry and pre-wetted soil trays of the first experiment these are referred to as dry-wet and wet-wet runs.

During all rainfall simulations surface runoff from the soil trays was captured in covered storage tanks (10 L). From the amount of rainfall on the tray surface and the total runoff volume runoff coefficients were calculated. The runoff from the storage tanks was later dried at 105 °C to determine sediment delivery, sediment texture and carbon content.

Shortly after the rainfall simulations (approx 10–20 min), the soil trays were moved outside under a roof to prevent additional rainfall and direct radiation. The geotextiles were removed and the PVC soil

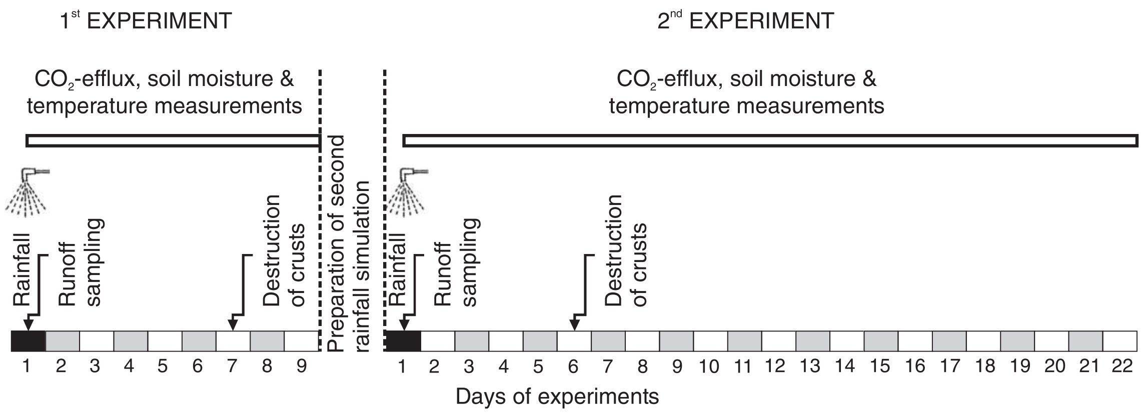

Fig. 2. Schematic sequence of operation of the two consecutive experiments; the first experiment comprises two rainfall simulations on very dry (4.5% m m^{-3}) and pre-wetted (10.9% m m^{-3}), while second experiment comprises one simulation on two soils with a different soil moisture content of 13.8 and 18.5% m m^{-3} (dry/wet and wet/wet runs), respectively.

collars used for CO_2 efflux measurements (Fig. 1) were immediately inserted to the soil. This allowed for first measurements of gas fluxes within 45 min after the rainfall simulation. Before the second experiment started, the soil collars and soil moisture probes were removed and after the rainfall they were installed again.

To test the effect of crust removal we mechanically destroyed the crusts within the collars six days after the first experiment and seven days after the second experiment, respectively (Fig. 2).

2.2. Soil and sediment sampling and analysis

The following soil properties in each tray were determined prior to the experiments based on four soil samples per soil tray taken from the upper 0.03 m using an Eijkelkamp core cutter with a volume of 100 cm^3 (Giesbeek, NL): bulk density, texture, soil organic carbon, nitrogen content and pH. For the sampling around the soil collars at the end of the experiments liner sampler with a diameter of 0.05 m and a length of 0.15 m (Eijkelkamp, Giesbeek, NL) was used.

Before mechanically destroying the crust in the collars to test their effects on CO_2 effluxes the crust thickness was determined. Therefore, the crusts were cut in a cross-section with a sharp knife and the thickness was measured 10 times per collar. The crusts were destroyed using a small rake inserted approximately 10–20 mm into the topsoil.

Except for bulk density which was determined separately by the volumetric core method, all soil and sediment samples were analysed for carbon and nitrogen content, texture as well as pH following the same procedure. Remaining roots as well as recognisable undecomposed plant residues were again hand-picked. Soil samples were then dried at 105 °C for 24 h. Although the loess soils in the area are mostly deeply decalcified, all soil samples were checked for lime ($CaCO_3$) with hydrochloric acid (10%). If any inorganic C content was recognised, it was destroyed using hydrochloric acid. Soil organic carbon (SOC) and N contents were then determined by dry combustion using a CNS elemental analyser (vario EL, Elementar, Germany). Soil texture was determined by a combined sieve-pipette method (Deutsches Institut für Normung, 2002). Soil pH value was measured for 2 min with a Metrohm 827 pH-lab (Metrohm, Runcorn, UK) while using a 1:5 soil solution extract (Deutsches Institut für Normung, 2006).

2.3. Measuring soil respiration, soil temperature and soil moisture

An automated closed dynamic chamber designed for survey measurements (LICOR 8100-103, Lincoln, USA) was used in combination with an infrared gas analyser (LI-8100) to measure soil respiration (Fig. 3). The chamber was closed for 120 s and the linear increase of CO_2 concentration in the chamber was used to estimate soil respiration. As described above the PVC soil collars (∅ 20 cm) used to measure CO_2 effluxes were installed approximately 30 min after each rainfall simulation. Due to the water saturated topsoil no long-term collar settling to prevent side effects typically found if collars are inserted into dry soil was necessary.

In addition to the three collars per treatment (and control) two frequency domain reflectrometry (FDR) probes per treatment (0.14 m long; Tektronix 1502B Soil Moisture Measurement System and a CR10 data logger from Campbell Scientific; Logan; UT; USA) were installed to monitor soil moisture changes. Moreover, soil temperature (upper 0.05 m) was determined using a thermocouple soil temperature probe connected to the Licor LI-8100 system. Due to some technical problems this temperature data could not be used continuously for all efflux measurements. Hence, we estimated soil temperature from the LI-8100 chamber temperature which showed a close relation to soil temperature in case of a similar temperature and missing direct radiation in an earlier study ($R^2 = 0.82$, $n = 539$; Fiener et al., 2012). However, differences in soil temperature between different treatments within one set of CO_2 efflux measurements, which took approximately 90 min in case of the 24 soil collars measured after the first experiment and 45 min in case of the 12 collars after the second experiment, should be very small. Hence, no correction for soil temperature was carried out which may call for more precise soil temperature measurements. Moreover, the measuring order at the different collars during each set of measurements was chosen randomly to avoid any systematic bias due to a potential increase or decrease of soil temperature during the 90 and 45 min, respectively. After the first experiment four measurements per collar and day were carried out, while eight measurements per collar and day were performed during the first seven days after the second experiment. Following the fourteenth day measurements were reduced to four per collar and day.

As the study was focused on a comparison between different treatments we calculated a virtual mean daily efflux per collar from the different sets of measurements. These virtual daily measurements per collar (CO_2 efflux plus soil temperature) are the basis to calculate a mean daily efflux/temperature per treatment (subsequently referred to as daily mean). In case of soil moisture only one measurement at the beginning and the end of a daily campaign was carried out and the daily mean soil moisture is calculated from these two measurements.

Apart from the daily means we also calculated mean virtual CO_2 effluxes per collar to analyse correlations between flux and crust thickness. All statistical analyses were carried out using the GNU R version 2.6 (R Development Core Team, 2009). When possible mean values are given with the standard deviation (Mean ± SD) which is also indicated in error bars within figures. Significance levels are calculated using Student *t*-test.

Fig. 3. Soil containers and CO_2 measurement equipment (LICOR 8100-103 survey chamber and LICOR 8100 infrared gas analyser in foreground) located outside the laboratory under a roof to prevent additional rainfall and direct radiation.

3. Results and discussion

3.1. Crusting, runoff and sediment delivery

As expected runoff coefficients and sediment delivery of controls were much smaller than those of the respective treatments (Fig. 4). This indicates that the geotextile on the one hand successfully reduced the crusting effect which directly reduced surface runoff and on the other hand reduced sediment transport via a reduction in surface runoff amount and turbulence introduced by rain drop impact.

Comparing the behaviour of the different treatments the first experiment showed the expected results that dry soils produced less runoff than pre-wetted soils. Sediment delivery was also reduced when the soil was initially dry. For the second experiment the picture is less clear. Treatment and control in case of the soil trays with a smaller antecedent moisture content produced higher runoff coefficients than their counterparts with an approximately 3 to 4% m m^{-3} higher antecedent moisture content (Fig. 4). This may be partly caused by the high roughness following the crust destruction at the end of the first experiment, which introduced some random effect in building up slightly concentrated surface runoff in some trays. Also, initially dry loess soils are known to be susceptible to slaking, which may lead to aggregate breakdown and crusting even in the absence of raindrop impact (Le Bissonnais and Singer, 1992).

During the first experiment, where the soil surface was relatively smooth before the rainfall was initiated, a homogeneous crust developed with an increasing thickness (by 55% on average for treatments and controls) from the up-slope to the down-slope end of the trays. This indicates that the surface runoff produced could not transport all eroded material to the outlet of the soil trays. The mean crust thickness of the treatments (Mean 2.4 ± 1.5 mm) after the first experiment was significantly larger ($p < 0.05$) than that of the respective crusts of the controls (Mean 1.4 ± 1.0 mm). In the second experiment a different picture arose where the dry/wet run had a significantly thinner ($p < 0.05$) mean crust than its control while the opposite was true for the wet/wet run and its control. This may have resulted from the higher surface runoff in case of the dry/wet run (Fig. 4) and potentially resulted in significant crust erosion.

Due to the small amount of delivered sediment and some problems with the C measurements it was not possible to reliably determine C enrichment compared to parent soil for all treatments. However, the average enrichment ratio measured during the first experiment was much more pronounced (average value of 3.1 for dry and pre-wetted control and pre-wetted treatment) than the enrichment ratio observed during the second experiment (average of 1.7 for dry/wet and wet/wet control). A similar decreasing trend was observed by Polyakov and Lal (2004). This could have two explanations (i) the light C was already removed during the first run or (ii) there was less selective removal due to the higher runoff amounts in the second run. However, as the crusts were destroyed before the second experiment the second reason seemed to be more probable.

3.2. Soil moisture and CO_2 efflux

Soil moisture and CO_2 efflux measurements were performed over 30 days: The first 9 days represent results of the first experiment and the following 21 days represent the second experiment (Fig. 2). The mean daily CO_2 effluxes calculated for these 30 days are based on 1972 individual measurements at the different soil collars. The measured CO_2 effluxes range from 0.14 to 1.41 µmol m^{-2} s^{-1} with a mean of 0.52 ± 0.16 µmol m^{-2} s^{-1}.

As the measurement of CO_2 effluxes started roughly 45 min after the end of the rainfall simulations it was not possible to capture the direct degassing of soil air as initiated by infiltrating water. Our first measurements (Figs. 5 and 6) started when in all cases the soil surface was water saturated and hence, as expected, small CO_2 effluxes were measured during the first day (mean 0.29 ± 0.11 µmol m^{-2} s^{-1}), independent of treatment. The only exception to this was the dry control where the relatively small rainfall amount seemed to infiltrate more rapidly due to the limited crust development and the overall soil moisture remained relatively low (Fig. 5). It is worth noting that the visually obvious saturation of the topsoil could not be detected by the 0.14 m long TDR probes averaging the moisture in the saturated topsoil and the unsaturated soil below.

At the second day all treatments and their controls showed a steep increase of CO_2 effluxes even if measured soil water content and temperature (except of pre-wetted run) stay more or less constant. This is probably due to a degassing of CO_2 from soil layers below the saturated topsoil layer after the larger soil pores are drained again. Except for the wet/wet run no significant ($p < 0.05$) difference between treatment and control could be detected on the second day.

In the following three to four days no significant differences ($p < 0.05$) were detected between the treatments and the treatments and their controls (except dry/wet vs. dry/wet control; Fig. 6). In case of all experiments the soil crusts were destroyed before the seventh and the sixed day (Figs. 5 and 6) to test the hypothesis that soil crusts hinder CO_2 diffusion and their destruction leads to an increase in CO_2 efflux. However, no significant differences between treatments and controls were measured and overall no effect of scratching the upper 10–20 mm of the topsoil could be detected. The slight increase and decrease in CO_2 effluxes after the crust destruction for the first (Fig. 5) and the second experiment (Fig. 6), respectively, appeared to be more related to changes in soil temperature.

Following the crust destruction the CO_2 effluxes are mainly governed by changes in soil temperature while the decline in soil moisture has no clear effect. These findings correspond with those reported from similar experiments: variations in soil moisture are not important as long as soil moisture is in the mid range, providing

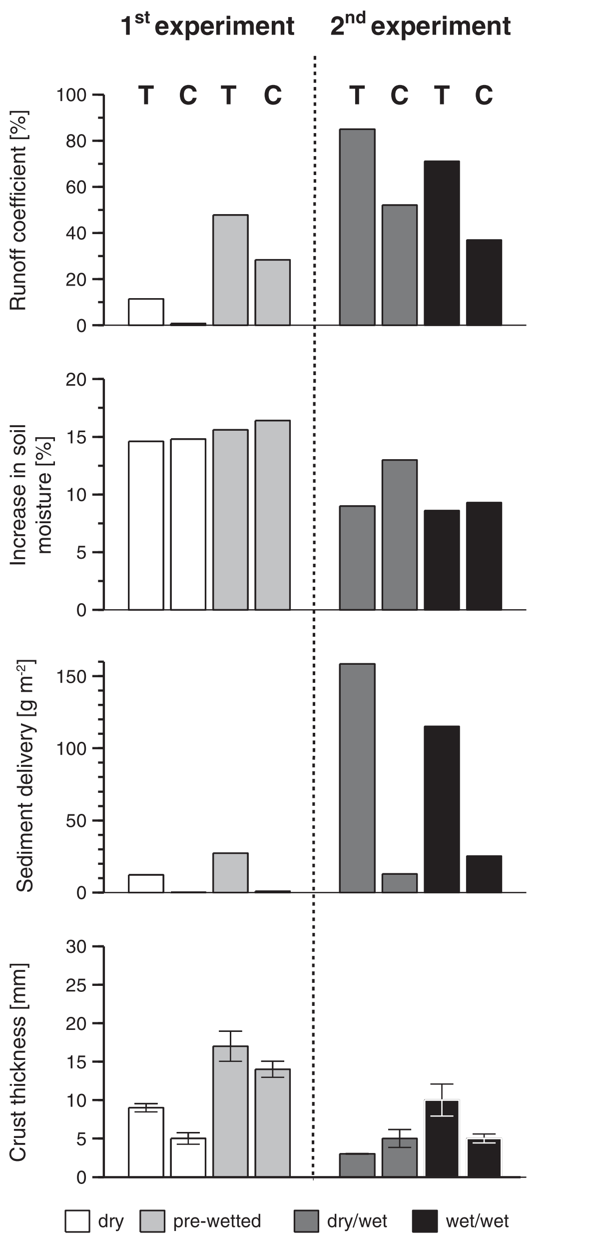

Fig. 4. Mean runoff coefficients, mean increase in soil moisture content due to the different rainfall simulations, mean sediment delivery, and mean crust thickness (inclusive standard deviation) of the different treatments (T) and their controls (C).

both sufficient oxygen and water to soil micro-organisms (Muhr et al., 2008). More importantly, no significant differences between different treatments as well as treatments vs. controls were found.

Calculating a mean flux per treatment for the first as well as the second experiment we found no significant ($p>0.5$) difference between the treatments and their respective controls (Fig. 7) indicating that interrill erosion and crusting have no obvious effect on following CO_2 effluxes. The absence of a strong, clear crust effect on CO_2 effluxes is underlined when comparing the mean daily flux per treatment during the first days of the experiments (before crust destruction) with its respective mean crust thickness per treatment. A linear regression between crust thickness and CO_2 effluxes indicates a slight negative relation ($R^2=0.3$) which is not significant on a $p<0.1$ level.

The lack of a clear crusting effect might result from different counteracting processes. On the one hand crusting reduces gas diffusion and air permeability, as it reduces porosity and air-filled pore connectivity, which both should reduce gas effluxes (e.g. Deepagoda et al., 2011; Kawamoto et al., 2006a, 2006b) and interrill erosion preferentially removes (labile) carbon that may be more prone to mineralisation (Kuhn et al., 2010). Both processes should lead to a decrease in CO_2 effluxes. On the other hand, we hypothesized that the effect of aggregate breakdown by intense drop impact (Le Bissonnais and Arrouays, 1997; Legout et al., 2005) might lead to the release of significant amounts of CO_2 by exposing protected soil organic carbon to oxygen, so that, overall, crust formation would lead to a net carbon loss on eroding sites.

Our data indicate that, at least for the conditions tested, this does not appear to be the case and suggest that additional CO_2 release due to aggregate breakdown and crust formation is limited, if at all present. Our data do not allow us to quantitatively isolate the effects of aggregate breakdown and reduced CO_2 efflux due to the presence of a crust, but it can be confidently stated that both are relatively small. If the presence of crust would induce a major blockage effect, one would expect (i) significant differences between treatments and (ii) a significant release of CO_2 from the soil at the moment that crusts are destroyed. However, none of both was measured. If aggregate breakdown and crusting would lead to significant additional SOC mineralization, differences between treatments would be expected as well as a significant decline of CO_2 efflux over time after rainfall application and/or a peak in efflux immediately after crust destruction. It is also unlikely that these effects were partially missed due to the delay between the end of rainfall application and the first measurement of CO_2 efflux. Indeed, the effect of crusting on gas diffusivity and air permeability should be small in the beginning of the experiments as the topsoil is saturated. Under such saturated conditions only minor gas fluxes can be expected (e.g. Arthur et al., 2012) independent from crust formation which was confirmed by the first measurements after the rainfall simulations.

In general our results indicate that interrill erosion, crusting and associated soil aggregate breakdown have no prominent effect on in situ soil respiration, at least for the soils studied here. This does not

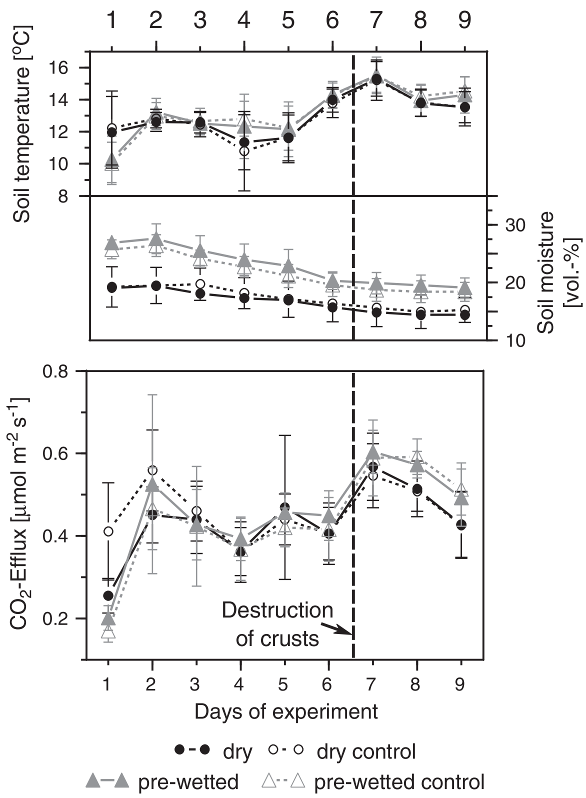

Fig. 5. Mean daily soil temperature, soil moisture and CO_2 effluxes calculated from all measurements taken in the different treatments per day during the first experiment; error bars give standard deviations of these daily means.

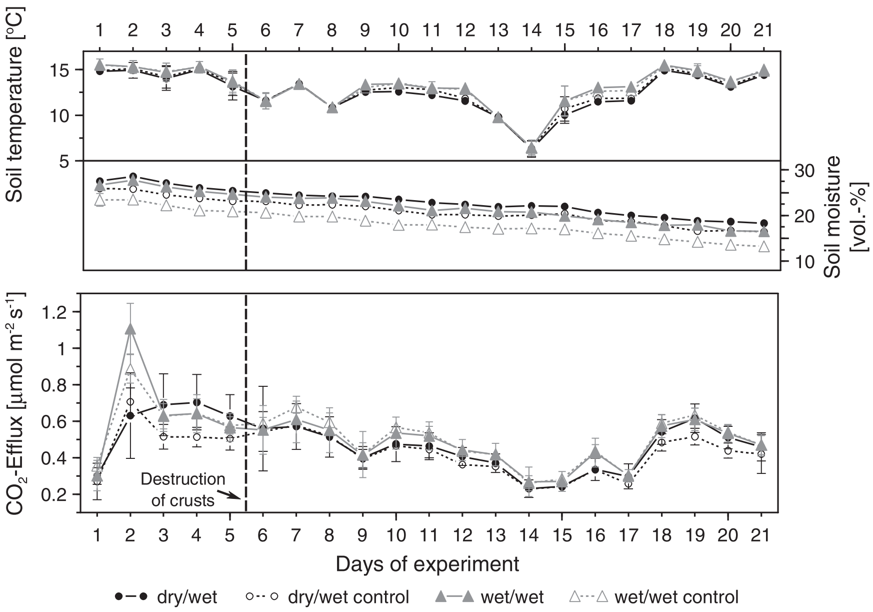

Fig. 6. Mean daily soil temperature, soil moisture and CO_2 effluxes calculated from all measurements taken in the different treatments per day during the second experiment; error bars give standard deviations of these daily means.

imply that the effects of interrill erosion on soil organic carbon can be neglected: interrill erosion leads to a preferential loss of topsoil SOC. Part of this eroded SOC might be mineralized after deposition within the terrestrial realm (Fiener et al., 2012; Van Hemelryck et al., 2010a) while another part enters rivers and lakes where it may foster CO_2 effluxes from aquatic ecosystems (Battin et al., 2009).

4. Conclusions

In this study the effects of interrill erosion, soil crusting and soil aggregate breakdown on in situ CO_2 effluxes were evaluated using rainfall simulations. As expected rain drop impact and hence interrill erosion on bare soil was significantly larger than on soil covered by a geo-textile (control), and surface runoff and sediment delivery generally increased with increasing antecedent soil moisture as well as rainfall duration. The same behaviour (with one exception) could be observed regarding soil crust development whereas crust thickness increased with antecedent soil moisture and rainfall intensity and was in general smaller for the controls.

However, neither the different crust thickness resulted in significant differences in in situ measured CO_2 effluxes nor an effect of crust destruction after six to seven days of measurements could be determined. This leads to the conclusion that crusting has no or only a minor effect on in situ CO_2 effluxes. The reduction of effluxes shortly after the rainfall simulations is most likely related to a limitation of gas diffusion and air permeability due to water saturation in topsoil. Moreover, our data did not support the hypothesis that rainfall induced aggregate breakdown increases CO_2 effluxes. However, it should be recognised that topsoil carbon is preferentially removed due to interrill erosion which may results in additional CO_2 release at depositional sites or in stream and/or standing water bodies.

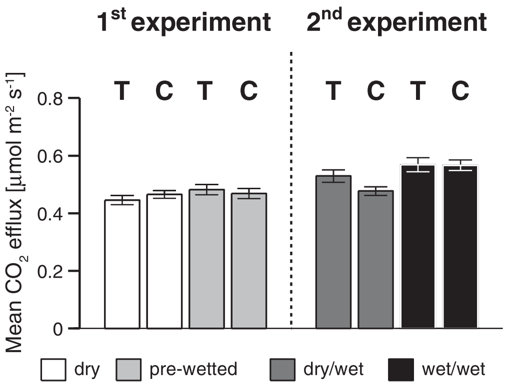

Fig. 7. Mean CO_2 effluxes from the different treatments (T) and their controls (C); error-bars indicate 95% confidence intervals.

Acknowledgements

This study carried out in cooperation between the University of Cologne and the Katholieke Universiteit Leuven (KUL). It was partly funded in the framework of the Deutsche Forschungsgemeinschaft (DFG) project “Soil redistribution in agricultural landscapes — source or sink of CO_2?” (FI 1216/4-1). We gratefully acknowledge the technical support during the rainfall simulations by Valentijn Tuts and Christoph Langhans and the support during the CO_2 efflux measurements by Sabrina Vandevelde.

References

Arthur, E., Moldrup, P., Schjonning, P., De Jonge, L.W., 2012. Linking particle and pore size distribution parameters to soil gas transport properties. Soil Science Society of America Journal 76, 18–27.

Assouline, S., Mualem, Y., 2001. Soil seal formation and its effect on infiltration: uniform versus nonuniform seal approximation. Water Resources Research 37, 297–305.

Auerswald, K., 1995. Percolation stability of aggregates from arable topsoils. Soil Science 159, 142–148.

Battin, T.J., Luyssaert, S., Kaplan, L.A., Aufdenkampe, A.K., Richter, A., Tranvik, L.J., 2009. The boundless carbon cycle. Nature Geoscience 2, 598–601.

Borselli, L., Torri, D., Poesen, J., Sanchis, P.S., 2001. Effects of water quality on infiltration, runoff and interrill erosion processes during simulated rainfall. Earth Surface Processes and Landforms 26, 329–342.

Cole, J.J., Prairie, Y.T., Caraco, N.F., McDowell, W.H., Tranvik, L.J., Striegl, R.G., Duarte, C.M., Kortelainen, P., Downing, J.A., Middelburg, J.J., Melack, J., 2007. Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. Ecosystems 10, 171–184.

Deepagoda, T.K.K.C., Moldrup, P., Schjonning, P., de Jonge Kawamoto, L.W., Kawamoto, K., Komatsu, T., 2011. Density-corrected models for gas diffusivity and air permeability in unsaturated soil. Vadose Zone Journal 10, 226–238.

Deutsches Institut für Normung, 2002. DIN ISO 11277: 2002–08 Bodenbeschaffenheit - Bestimmung der Partikelgrößenverteilung in Mineralböden - Verfahren mittels Siebung und Sedimentation. Beuth Verlag, Berlin.

Deutsches Institut für Normung, 2006. DIN ISO 10390: 2005 Bodenbeschaffenheit - Bestimmung des pH-Wertes. Beuth Verlag, Berlin.
Fiener, P., Dlugoß, V., Korres, W., Schneider, K., 2012. Spatial variability of soil respiration in a small agricultural watershed — are patterns of soil redistribution important? Catena 94, 3–16.
Harden, J.W., Sharpe, J.M., Parton, W.J., Ojima, D.S., Fries, T.L., Huntington, T.G., Dabney, S.M., 1999. Dynamic replacement and loss of soil carbon by eroding cropland. Global Biogeochemical Cycles 13, 885–901.
Harden, J.W., Berhe, A.A., Torn, M.S., Harte, J., Liu, S., Stallard, R., 2008. Soil erosion: data say C sink. Science 320, 178–179.
Jacinthe, P.A., Lal, R., 2001. A mass balance approach to assess carbon dioxide evolution during erosional events. Land Degradation and Development 12, 329–339.
Jacinthe, P.A., Lal, R., Kimble, J.M., 2002. Carbon dioxide evolution in runoff from simulated rainfall on long-term no-till and plowed soils in southwestern Ohio. Soil Tillage Research 66, 23–33.
Jobbagy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10, 423–436.
Kawamoto, K., Moldrup, P., Schjönning, P., Iversen, B.V., Komatsu, T., Rolston, D.E., 2006a. Gas transport parameters in the vadose zone: development and tests of power-law models for air permeability. Vadose Zone Journal 5, 1205–1215.
Kawamoto, K., Moldrup, P., Schjönning, P., Iversen, B.V., Rolston, D.E., Komatsu, T., 2006b. Gas transport parameters in the vadose zone: gas diffusivity in field and lysimeter soil profiles. Vadose Zone Journal 5, 1194–1204.
Kuhn, N.J., Armstrong, E.K., Ling, A.C., Connolly, K.L., Heckrath, G., 2010. Interrill erosion of carbon and phosphorus from conventionally and organically farmed Devon silt soils. Catena 91, 94–103.
Lal, R., 2003. Soil erosion and the global carbon budget. Environment International 29, 437–450.
Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.
Le Bissonnais, Y., Arrouays, D., 1997. Aggregate stability and assessment of soil crustability and erodibility: II. Application to humic loamy soils with various organic carbon contents. European Journal of Soil Science 48, 39–48.
Le Bissonnais, Y., Singer, M.J., 1992. Crusting, runoff, and erosion response to soil water content and successive rainfalls. Soil Science Society of America Journal 56, 1898–1903.
Legout, C., Leguédois, S., Le Bissonnais, Y., 2005. Aggregate breakdown dynamics under rainfall compared with aggregate stability measurements. European Journal of Soil Science 56, 225–237.
Luo, Y., Zhou, X., 2006. Soil Respiration and the Environment. Elsevier, Amsterdam.
Muhr, J., Goldberg, S.D., Borken, W., Gebauer, G., 2008. Repeated drying-rewetting cycles and their effects on the emission of CO_2, N_2O, NO, and CH_4 in a forest soil. Journal of Plant Nutrition and Soil Science 171, 719–728.
Novara, A., Armstrong, A., Gristina, L., Semple, K.T., Quinton, J.N., 2012. Effects of soil compaction, rain exposure and their interaction on soil carbon dioxide emission. Earth Surface Processes and Landforms 37, 994–999.
Poesen, J., Ingelmo-Sanchez, F., Mücher, H., 1990. The hydrological response of soil surfaces to rainfall as affected by cover and position of rock fragments in the top layer. Earth Surface Processes and Landforms 15, 653–671.
Polyakov, V.O., Lal, R., 2004. Soil erosion and carbon dynamics under simulated rainfall. Soil Science 169, 590–599.
Quinton, J.N., Catt, J.A., Wood, G.A., Steer, J., 2006. Soil carbon losses by water erosion: experimentation and modeling at field and national scales in the UK. Agric. Ecosystems and Environment 112, 87–102.
R Development Core Team, 2009. R: A language and environment for statistical computing. http://www.R-project.org.
Schlesinger, W.H., 2005. The global carbon cycle and climate change. Adv. Econ. Environ. Resour. 5, 31–53.
Smets, T., Poesen, J., Fullen, M.A., Booth, C.A., 2007. Effectiveness of palm and simulated geotextiles in reducing run-off and inter-rill erosion on medium and steep slopes. Soil Use and Management 23, 306–316.
Smith, S.V., Renwick, W.H., Buddemeier, R.W., Crossland, C.J., 2001. Budgets of soil erosion and deposition for sediments and sedimentary organic carbon across the conterminous United States. Global Biogeochemical Cycles 15, 697–707.
Stallard, R., 1998. Terrestrial sedimentation and the carbon cycle: coupling weathering and erosion to carbon burial. Global Biogeochemical Cycles 12, 231–257.
Tranvik, L.J., Downing, J.A., Cotner, J.B., Loiselle, S.A., Striegl, R.G., Ballatore, T.J., Dillon, P., Finlay, K., Fortino, K., Knoll, L.B., Kortelainen, P.L., Kutser, T., Larsen, S., Laurion, I., Leech, D.M., Leigh McCallister, S., McKnight, D.M., Melack, J.M., Overholt, E., Porter, J.A., Prairie, Y., Renwick, W.H., Roland, F., Sherman, B.S., Schindler, D.W., Sobek, S., Tremblay, A., Vanni, M.J., Verschoor, A.M., Von Wachenfeldt, E., Weyhenmeyer, G.A., 2009. Lakes and reservoirs as regulators of carbon cycling and climate. Limnology and Oceanography 54, 2298–2314.
Van Hemelryck, H., Fiener, P., Van Oost, K., Govers, G., Merckx, R., 2010a. The effect of soil redistribution on soil organic carbon: an experimental study. Biogeosciences 7, 3971–3986.
Van Hemelryck, H., Govers, G., Van Oost, K., Merckx, R., 2010b. Evaluating the impact of soil redistribution on the in situ mineralization of soil organic carbon. Earth Surface Processes and Landforms 36, 427–438.
Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., Marques da Silva, J.R., Merckx, R., 2007. The impact of agricultural soil erosion on the global carbon cycle. Science 318, 626–629.
Van Oost, K., Van Hemelryck, H., Harden, J.W., 2009. Erosion of soil organic carbon: implications for carbon sequestration. In: McPherson, B.J., Sundquist, E.T. (Eds.), Carbon Sequestration and Its Role in the Global Carbon Cycle. American Geophysical Union, Washington DC, pp. 189–202.
Wang, Z., Govers, G., Steegen, A., Clymans, W., Van den Putte, A., Langhans, C., Merckx, R., Van Oost, K., 2010. Catchment-scale carbon redistribution and delivery by water erosion in an intensively cultivated area. Geomorphology 124, 65–74.
Willems, P., 2011. Evaluatie en actualisatie van de IDF-neerslagstatistieken te Ukkel. Faculteit Ingenieurswetenschappen, Departement Burgerlijke Bouwkunde, Afdeling Hydraulic, Leuven.