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PM_{10} emission efficiency for agricultural soils: Comparing a wind tunnel, a dust generator, and the open-air plot

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ABSTRACT

The PM_{10} emission efficiency of soils has been determined through different methods. Although these methods imply important physical differences, their outputs have never been compared. In the present study the PM_{10} emission efficiency was determined for soils through a wide range of textures, using three typical methodologies: a rotary-chamber dust generator (EDG), a laboratory wind tunnel on a prepared soil bed, and field measurements on an experimental plot. Statistically significant linear correlation was found ($p < 0.05$) between the PM_{10} emission efficiency obtained from the EDG and wind tunnel experiments. A significant linear correlation ($p < 0.05$) was also found between the PM_{10} emission efficiency determined both with the wind tunnel and the EDG, and a soil texture index (%sand + %silt)/(%clay + %organic matter) that reflects the effect of texture on the cohesion of the aggregates. Soils with higher sand content showed proportionally less emission efficiency than fine-textured, aggregated soils. This indicated that both methodologies were able to detect similar trends regarding the correlation between the soil texture and the PM_{10} emission. The trends attributed to soil texture were also verified for two contrasting soils under field conditions. However, differing conditions during the laboratory-scale and the field-scale experiments produced significant differences in the magnitude of the emission efficiency values. The causes of these differences are discussed within the paper. Despite these differences, the results suggest that standardized laboratory and wind tunnel procedures are promissory methods, which could be calibrated in the future to obtain results comparable to field values, essentially through adjusting the simulation time. However, more studies are needed to extrapolate correctly these values to field-scale conditions.

1. Introduction

During the last decades, the study of PM_{10} (particulate matter that is smaller than 10 μm in aerodynamic diameter) emission has taken great relevance. PM_{10} can be emitted from different sources: agricultural soils (López et al., 1998; Sharratt et al., 2007; Li et al., 2015), natural areas (Belnap et al., 2009; Abulaiti et al., 2014), dry lakes (Gillette et al., 1997; Halleaux and Rennó, 2014), sand deserts (Draxler et al., 2001; Hoffmann et al., 2008); and unpaved roads (Goossens and Buck, 2009; Panebianco et al., 2016). PM_{10} emission, suspension, and deposition can have many environmental and socioeconomic impacts (Pope et al., 1995; Seinfeld and Pandis, 1998; Ghio and Devlin, 2001; Herut et al., 2001; McConell, 2007).

A parameter that has been widely used in studies referred to the dynamics of wind-induced PM_{10} emission is the PM_{10} emission efficiency. This parameter is usually calculated as the ratio between the amount of PM_{10} emitted from a surface per unit area per unit time and the corresponding amount of coarser material transported by saltation (the saltation flux). PM_{10} emission efficiency can be used to compare the emission potential of different surfaces. This parameter could also be used within the Aeolian-dust emission modeling context, and also for mapping of potential dust-emission areas or emission inventories. However, the significant differences between the methods that are used to determine the emission efficiencies raise some concerns about the applicability of the results. The PM_{10} emission efficiency has been estimated by different methodologies: field conditions (Gillette et al., 1997; Saxton et al., 2000; Breshears et al., 2003; Rajot et al., 2003; Zobeck and Van Pelt, 2006; Singh et al., 2012; Avecilla et al., 2017), wind tunnels (Marticorena and Bergametti, 1995; Alfaro, 2008; Avecilla et al., 2016; Panebianco et al., 2016) and dust generators (Amante-

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Orozco, 2000; Carvacho et al., 2004; Gill et al., 2006; Mendez et al., 2013, Mendez et al., 2015).

Studies under field conditions usually demand much time and they are expensive. Moreover, field work implies measuring under harsh conditions, multiple variables interaction, and a high variability of many of these variables (Delgado Fernández, 2010). For these reasons, laboratory wind tunnels and dust generators were developed. As compared to field measurements, both of them have advantages about the operation costs, but they also have important constraints concerning the simulation of the physical processes that occur under field conditions. Laboratory wind tunnels are a good alternative for simulating saltation conditions and controlling wind speed and blowing time. However, other variables such as soil moisture and temperature, surface roughness, and the heterogeneity of the soil surface are still difficult to simulate (van Pelt et al., 2010). Therefore, wind laboratory tunnels still have their limitations, especially for evaluating the effects of the surface characteristics and the soil cover. For these reasons portable wind tunnels have been also used (Raupach and Leys, 1990; Pietersma et al., 1996; van Pelt et al., 2010).

Dust generators produce much more controlled conditions that allow conducting research concerning emitted particles at a low operational cost. Depending on the method by which dust aerosols are generated, dust generators can be divided into three classes according to Gill et al. (2006): class "A" (fluidization) in which dust is re-suspended by direct entrainment into air flow in a tube; class "B" (gravitation) in which a source sample falls as a discrete slug through the air into or within an enclosed chamber, from which dust is evacuated; and class "C" (mechanical dispersion or agitation) in which the source material repeatedly falls from top to bottom of a horizontal, rotating cylinder or tube and is entrained into airflow. Generally the results from dust generators depend on the physical strength of the aggregates present in the samples. However, the effect of fundamental physical processes of wind-induced dust emission, such as saltation abrasion and bombardment cannot be measured directly.

One of the great advantages of studies under controlled or laboratory conditions is the standardization of certain parameters such as particle size distribution, wind speed, soil moisture, among others. Standardized procedures allow the experiment to be reproduced under the same conditions at different times and places, unlike experiments conducted with natural soils. Moreover, this procedures allow to reduce the significant variability that it is usually observed at the field scale.

Many dust emission values for soils and other materials have been determined using wind tunnels or chamber-type dust generators. It has been often assumed that these values can be extrapolated to field conditions. However, the emission efficiency can be very sensitive to the measuring circumstances. It is important to determine whether this parameter can be accurately estimated under measurement settings that are dissimilar from the field conditions. To our knowledge, there are so far no direct comparisons between PM_{10} emission efficiency measurements obtained with different methods. The objectives of this work were: to compare the emission efficiencies of PM_{10} of several agricultural soils obtained using standard laboratory methods: wind tunnel and dust generator; and to compare these results with the available measurements made on the same soils under field conditions.

2. Materials and methods

2.1. Soil selection

Six agricultural soils within the central Argentina semi-arid region were selected for their contrasting textures and organic matter content (Table 1). The textural range comprised sand, loamy sand, sandy loam and loam soils (Soil Survey Division Staff, 1993). Soils were ordered according to their growing contents of clay and organic matter (Table 1) considering that these components are the main factors for soil aggregation, a parameter that affects the capacity of the soil to emit PM_{10} (Mirzamostafa et al., 1998; Hagen, 2004).

Soil samples were taken from the first 2.5 cm topsoil. The sampling scheme consisted of four subsamples randomly distributed within a homogeneous area of 100 m^2. The textural composition of each soil was determined by wet sieving and the Robinson pipette method (Gee and Bauder, 1986), which included: destruction of free carbonates (with 6% acetic acid) and of organic matter (with hydrogen peroxide), a dispersion with sodium hexametaphosphate, agitation in water during 30 min at 1500 rpm, and an ultrasound treatment at 35 kHz for 15 min. Organic matter content (OM) was determined by the Walkley and Black method (Walkley and Black, 1934) and the content of free carbonates ($CaCO_3$) by means of the Scheibler calcimeter (Schlichting and Blume, 1966).

2.2. Wind tunnel

The laboratory wind tunnel used for this study is 8 m long. The simulation section is 6 m long, 1 m height and 0.5 m wide. The working section used for placing the soil samples is 4 m long and the clean section is 2 m long. The air is pushed into the wind tunnel by an axial fan located before the clean section. The fan is driven by a Honda GX670 engine. Between the fan and the clean section, there is a flow conditioning section (van Pelt et al., 2010). An abrader hopper was installed at the top of the tunnel, in the middle of the clean section. This device delivers the supply of the saltation fraction of a soil by gravity into the wind tunnel, at a mean flow rate of 0.0055 $kg\,m^{-1}\,s^{-1}$ (van Pelt et al., 2010). Added material (saltation fraction) entered the air flow from the hopper through a tube 1 cm in diameter, at a height of 0.15 m from the wind tunnel floor. More details of the wind tunnel construction and the results of the wind tunnel calibration can be found in Colazo et al. (2016).

The soil samples were air dried, until reaching a humidity moisture content below 2%, and sieved by hand through a 2 mm mesh to be used as the soil bed in the wind tunnel working section. This procedure was made in order to homogenize the aggregate size distribution of the soil bed. The soil bed consisted of a 0.2 m wide and 0.025 m depth tray placed along the wind tunnel working section. The saltation fraction (0.2–0.5 mm, van Pelt et al., 2010) that was used in the abrader hopper of the wind tunnel was separated manually by dry sieving the own soil.

The wind erosion process was simulated under two different conditions: increased saltation (IS), during which the saltation fraction of the soil was released into the air flow prior to the soil bed using an abrader hopper; and no saltation added (NS) during which the soil sample was exposed to the wind stress without the addition of saltating material prior to the soil bed. Simulations lasted 4 min each and were performed at an average friction velocity of 0.21 $m\,s^{-1}$ (standard deviation: 0.05 $m\,s^{-1}$). Each treatment was replicated 4 times.

Except for the cup anemometer, that was placed right before the working section, all the measurements in this work were made at the end of the working section, right before the wind tunnel exhaust, at a fetch distance of 4 m. The wind speed was measured with a pressure anemometer at different heights (0.05, 0.17, 0.315, 0.48 m), obtaining a vertical profile of the wind according to the Law of the wall:

$$U_Z = \frac{u^*}{k}\ln\left(\frac{z}{z_0}\right) \tag{1}$$

where U_Z is the wind speed at height z, u^* is the friction velocity, k is the Von Karman's constant (0.4), and Z_0 is the aerodynamic roughness height. This measurement scheme allows the calculation of several parameters that describe the interaction between the surface and the wind speed (Roney and White, 2006). Both u^* and Z_0 were determined by the slope and the intercept of the logarithmic wind profile respectively (Bagnold, 1941). Speed data for the wind profile were taken within the boundary layer of the wind tunnel (height at which the logarithmic wind speed profile reaches 99% of its maximum value), estimated approximately between 0.4 and 0.6 m (Maurer et al., 2006;

Table 1
Main characteristics of the studied soils (S). Soils were numbered according to a texture index: [(sand + silt)/(clay + OM)].

Soil	Unit	S1	S2	S3	S4	S5	S6
Geographic location		36°33′48″ S 64°18′13″ W	39°23′41″ S 62°37′41″ W	33°40′22″ S 65°22′33″ W	36°34′31″ S 63°59′22″ W	36°32′28″ S 64°17′40″ W	36°35′59″ S 63°57′39″ W
Soil Taxonomy		Tipic Ustipsamment	Ustic Torripsamment	Tipic Ustipsamment	Tipic Ustipsamment	Entic Haplustoll	Entic Haplustoll
Textural class		Sand	Loamy sand	Loamy sand	Loamy sand	Sandy loam	Loam
Clay (< 0.002 mm)	$g\,kg^{-1}$	49.9	92.5	82	74.9	102.3	171.6
Silt (0.002–0.053 mm)		67.3	99.7	124.2	124	186.2	355.5
Total sand (0.053–2 mm)		882.8	807.8	793.8	801.1	711.5	472.9
Very fine sand I (0.053–0.074 mm)		87.2	55.2	230.6	69.6	135.8	129.3
Very fine sand II (0.074–0.105 mm)		176.2	80.7	366.9	191.7	180.8	129.1
Fine sand (0.105–0.250 mm)		543.2	569.4	171.9	287.2	342.1	173
Medium and coarse sand (0.250–2 mm)		76.2	102.5	24.4	252.6	52.8	41.5
GMD	µm	94	74	71	94	72	40
OM	$g\,kg^{-1}$	13.7	20.8	7	18.4	13.1	28.2
$CaCO_3$		5.5	6.5	8.7	5.5	4.3	8.8

OM: organic matter. GMD: geometric mean diameter.

van Pelt et al., 2010). In this work, the boundary layer was about 0.5 m height. In addition, the free stream velocity was measured by a cup anemometer located at the end of the clean section of the wind tunnel, at a height of 0.7 m, out of the boundary layer. The free stream velocity (0.7 m height) was 7.9 $m\,s^{-1}$ (SD = 0.39 for IS and 0.37 $m\,s^{-1}$ for NS), which is above the average threshold wind speed for soils of this region (de Oro and Buschiazzo, 2009).

The horizontal mass flux q ($kg\,m^{-2}$) was measured using BSNE samplers (Fryrear et al., 1998) placed at five different heights (0.05, 0.17, 0.315, 0.48 and 0.75 m) at the center of the wind tunnel exhaust. The collected material was weighed, and the horizontal mass transport Q ($g\,m^{-2}\,s^{-1}$) was determined using the Curve Expert® 1.3 software (Hyams, 2005), by adjusting an exponential function to the mass flux profile across the height, and then integrating it from 0 m to 1 m height:

$$0.2 \times Q = \int_0^1 q^{bz} dz \tag{2}$$

where q is the mass flux at height z in $g\,m^{-2}$, b represents the rate of decay of the horizontal mass flux across the height, and 0.2 is the soil bed width (m).

The PM_{10} concentration was measured using a Kanomax digital dust monitor (model 3443). The Kanomax 3443 is a light scattering digital dust monitor with a particle range from 0.1 to 10 µm. It measures PM_{10} in a concentration range between 0.001 and 10000 $mg\,m^{-3}$ with an intake flux of 1 $l\,min^{-1}$ (more details of the dust monitor can be found in www.kanomax-usa.com). The PM_{10} vertical flux in $g\,m^{-2}\,s^{-1}$ (F_VPM_{10}) was calculated with the equation proposed by Gillette (1977), which was widely used in many studies (Kjelgaard et al., 2004; Singh et al., 2012):

$$F_VPM_{10} = \frac{k \cdot u^* \cdot (C_1 - C_2)}{\ln(Z_2/Z_1)} \tag{3}$$

where k is the Von Karman constant (0.4); u^* it is the friction velocity in $m\,s^{-1}$; C_1 and C_2 are the PM_{10} concentration values (in $g\,m^{-3}$) at the heights Z_1 and Z_2, (0.05 and 0.75 m respectively). The PM_{10} emission efficiency (dimensionless), the potential of a soil to emit PM_{10} during the saltation process, was defined as the F_VPM_{10} to Q ratio (Alfaro, 2008).

Details on the experimental design and methodology of the wind-tunnel study can be found in Avecilla et al. (2015, 2016).

2.3. Easy dust generator (EDG)

The EDG (Mendez et al., 2013) is a "Class C" dust generator (Gill et al., 2006). According to Gill et al. (2006), these kind of dust generators are based on mechanical dispersion or agitation (rotating drums and similar techniques). The mechanical dispersion or agitation generator has been used to estimate the PM_{10} emission by wind erosion, tillage operations and traffic on unpaved roads. This kind of device transfers mechanical or kinetic energy to dust source materials, creating aerosols from the abrasion or fracture caused when grains of the source material collide with each other and/or the dust generator. The EDG is composed by two parts: a dust generating chamber and a concentration chamber.

The generating chamber, where the soil sample is placed, consists of an 116 mm wide and 200 mm tall plastic bottle. Inside the bottle, four 50 × 200 mm plastic blades are attached to the bottle walls to mix the soil sample during bottle rotation. A 21 mm diameter orifice on one extreme of the allows the free entrance of air and it is also used to introduce the soil sample. On the other side of the bottle, an 8 mm diameter plastic tube is inserted through a rubber stopper which is placed in a 21 mm diameter orifice for avoiding air losses. The portion of the plastic tube outside the generating chamber is coupled to a glass tube connected to the concentration chamber.

The concentration chamber is an 116 mm wide and 250 mm long plastic bottle. A 9 mm diameter and 150 mm long plastic tube is placed in the middle of the concentration chamber, perpendicularly to its main axis. This plastic tube connects the concentration chamber to the dust monitor. A vacuum source working at 5 $l\,min^{-1}$. (5 l: 0.005 m^3) is connected to the distal extreme of the concentration chamber through a 10 mm diameter pipe. More details on the construction and calibration of the EDG can be found in Mendez et al. (2013).

Inside the rotating bottle, 1.5 g of soil sieved by 2 mm were placed. The duration of the simulations was 5 min, with measurement frequency programmed every 6 s (dust monitor) and replications were made 5 times for each soil.

The total PM_{10} emission, expressed in micrograms of PM_{10} per gram of soil ($\mu g\,g^{-1}$) was calculated with the following equation:

$$TE_{PM_{10}} = \frac{PM_{10}\ ave \times VARB \times DE}{PMS} \tag{4}$$

where TE_{PM10} is the total PM_{10} emission in $\mu g\,g^{-1}$ of soil, $PM_{10}\ ave$ is the averaged PM_{10} concentration along the experiment in $mg\,m^{-3}$ of air, VARB is the air volume removed by the vacuum pump in $m^3\,min^{-1}$, DE is the experiment duration in minutes, and PMS is the soil sample weight in g (Mendez et al., 2013). In this work, PM_{10} emission efficiency was considered as the PM_{10} emission per gram of soil introduced into the bottle.

2.4. Field measurements

Two soils were selected: loamy sand (Tipic Ustipsamment) and loam soil (Entic Haplustoll). The loamy sand soil is soil n°4 (S4) and the loam is soil n°6 (S6) in Table 1. Both soils are placed in the Anguil Experimental Station of the National Institute for Agricultural Technology (INTA, 36°32′27″S, 63°59′29″W), Argentina.

Measurements were made during two years on an 80 m wide and 200 m long experimental plot, prepared parallel to the prevailing wind direction. The plot was tilled regularly (every fifteen days) with a disc plow in order to maintain the surface with minimal roughness and vegetation cover. Measurements were made only during high wind-speed events. The duration of an erosive event was defined as the period during which the wind speed exceeded $5\,m\,s^{-1}$ at 2 m height. This wind speed has been considered a the average threshold for wind erosion (Fryrear et al., 1998). The measurement periods did not exceeded 24 h, in order to avoid changes of the wind direction.

On each plot, horizontal mass flow was measured by BSNE collectors placed on a mast at different heights (0.135, 0.3, 0.5 and 1 m) and then the horizontal mass transport was calculated using equation 2 (Eq. (2)). The BSNE collectors used to calculate Q was located at the windward edge of the plot, together with the dust monitors.

PM_{10} concentration emitted from the plot was measured using two Kanomax dust monitors (model 3443). The dust monitors were placed at 1.8 and 3.5 m above the soil surface. Measurements were made at a frequency of 60 s. The PM_{10} vertical flux (F_VPM_{10}) was calculated according to equation 3 (Eq. (3)).

Ten events were recorded on S4 and 15 events on S6. The PM_{10} emission efficiency was calculated as an average for each erosive event. More details on the experimental design used for the field study can be found in Avecilla et al. (2017). As for the wind tunnel, the PM_{10} emission efficiency was defined as the ratio between F_VPM_{10} and Q.

3. Results and discussion

3.1. Sensitivity of the dust generator and the wind tunnel to the soil texture

A significant linear correlation ($p < 0.05$) was found between the PM_{10} emission efficiencies determined using the wind tunnel and the EDG, and the soil texture index (sand + silt)/(clay + OM) (Fig. 1). Soils with higher sand contents were less efficient than the fine-textured soils. Previous studies under wind tunnel and field conditions also showed the high capability to emit PM_{10} of the fine-textured soils (Alfaro, 2008; Li et al., 2015; Panebianco et al., 2016). The slope of the best-fit correlation lines indicate that the PM_{10} emission obtained with both methodologies followed similar trends. The soil texture index was highly correlated to the emission efficiency values, regardless of the methodology. These results are in agreement with those found in previous studies made in wind tunnels by Bullard et al. (2004) and Gill et al. (2006), and with dust generators comparable to the EDG (Amante-Orozco, 2000; Carvacho et al., 2004; Gill et al., 2006).

For the coarser soils, the PM_{10} emission efficiency values obtained with the wind tunnel were similar than the ones obtained with the EDG. On the contrary, for the fine-textured soils, the highest efficiency values were found with the EDG. This difference can be attributed to the interaction between particles and aggregates, and the energy provided during each method. Under wind tunnel, PM_{10} emission is mainly originated in the wind energy-driven saltation process. The saltation process involves abrasion, bombardment and fragmentation of the particles moving at high speeds. In the EGD, the rotating movement provides less amount of energy for abrasion; hence fragmentation predominates due to the mixing effect of the blades. On samples composed predominantly by individual mineral particles, and to a lesser extent by aggregates, the EDG underestimates the PM_{10} emissions because the dimensions of the dust chamber does not provide enough energy for the simulation of the abrasion and bombardment produced by saltation particles under wind tunnel. On the contrary, on samples from aggregated soils, the EDG overestimates the PM_{10} emission due to the excessive fragmentation of the aggregates rotating inside the chamber. Mendez et al. (2015) observed that for the more aggregated fractions of the soil, the PM_{10} emission increases with the simulation time, whereas on less aggregated fractions the emission decreases rapidly with the simulation time due to the absence of successive fragmentation.

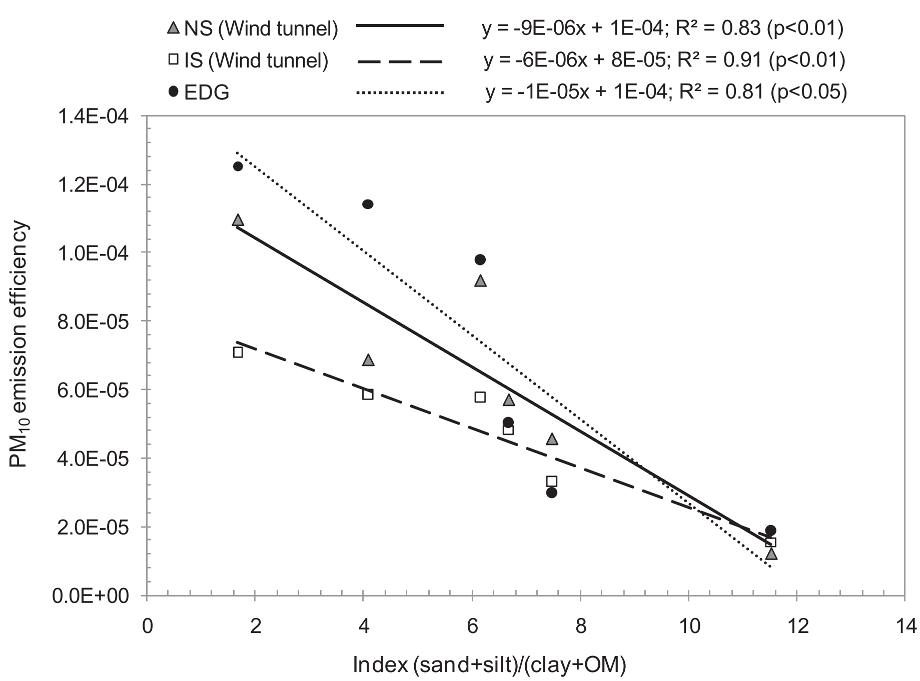

Fig. 1. Correlation between PM_{10} emission efficiencies and the soil texture index.

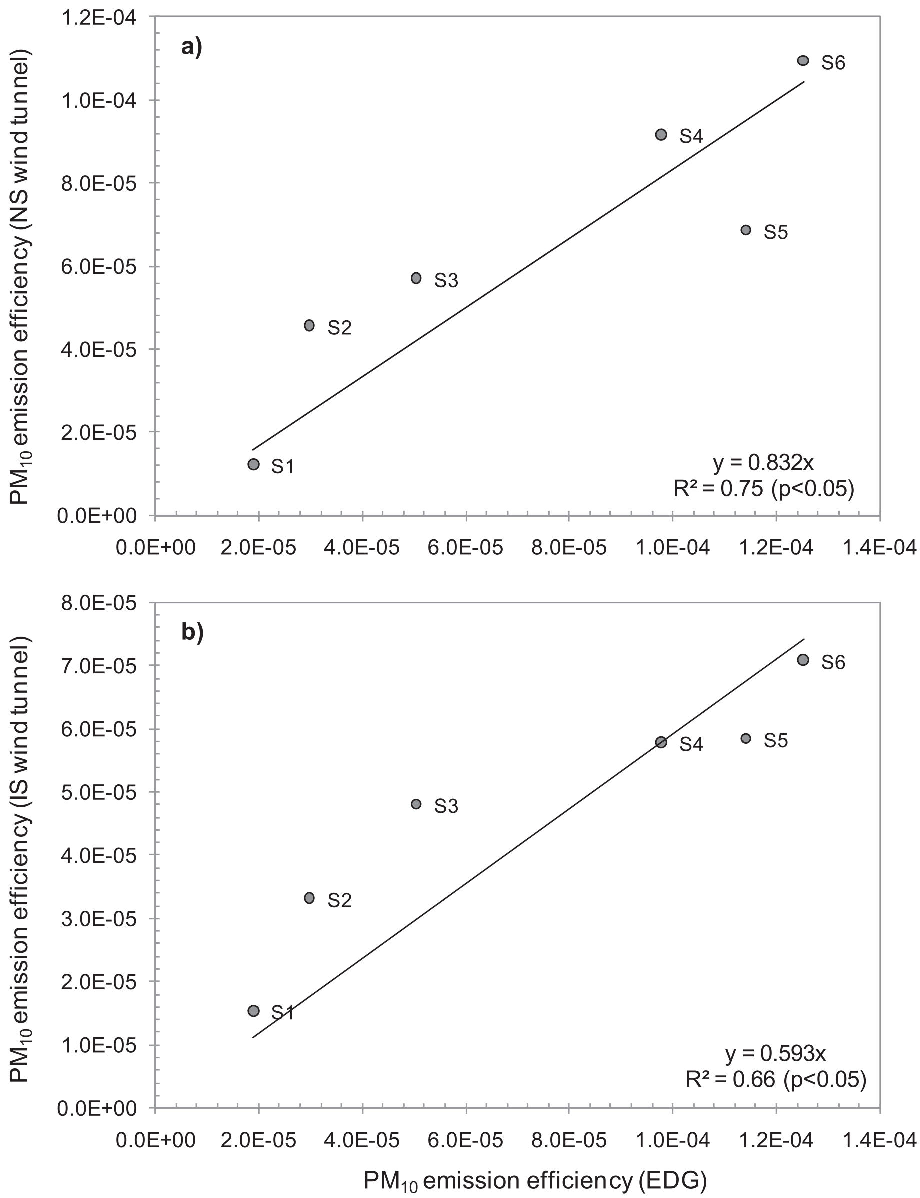

Fig. 2. Correlation between the PM_{10} emission efficiency of the EDG and the wind tunnel for different treatments (NS and IS).

3.2. Overall efficiencies: dust generation in a rotary chamber vs. wind-tunnel saltation

Fig. 2 shows the correlation between the PM_{10} emission efficiencies obtained with the EDG and the wind tunnel (Fig. 2a and b). Despite the significant linear correlations ($p < 0.05$), differences were observed between methodologies concerning the efficiency values. The slope of the correlation lines indicate that emission efficiencies measured with the EDG were 17% and 41% higher than those measured during NS and IS treatment respectively. In NS the flux of saltating particles was considerably lower than in IS, therefore the emission values were more similar to the ones found with the EDG.

As discussed in the previous section, the differences observed can be attributed to the preponderance of the fragmentation of the aggregates of the fine-textured soils inside the rotating chamber. This effect is emphasized due to the overtime of the experiment in the EDG (Mendez et al., 2016).

3.3. Comparing between the EDG, the wind tunnel and the field measurement

Fig. 3 shows the PM_{10} emission efficiency values determined with the EDG, the wind tunnel and with light scattering devices in open-air field conditions for sandy (S4) and loam (S6) soils. Unfortunately, field measurements for the rest of the soils are not available. In every case, S6 showed higher emission efficiencies than S4. Under field conditions, the average PM_{10} emission efficiency for S6 was 2.5 times higher than the one from S4. With the EDG the efficiency for S6 was 1.28 times higher than for S4, while in the wind tunnel it was 1.20, and 1.22 times greater for NS and IS respectively.

Results found herein are consistent with other studies. The PM_{10} emission efficiency values obtained under field conditions are generally between 10^{-5} and 10^{-2} (Gillette et al., 1997; Gomes et al., 2003a,b; Rajot et al., 2003; Kok et al., 2012). Roney and White (2006), in a study developed in wind tunnel with soils of similar characteristics to those used in the present work (sand, loamy sand and sandy loam) found that the emission efficiency was higher in finer textures, in ranges of similar friction velocity (0.3–0.4 m s^{-1}) to those used in this work. In addition,

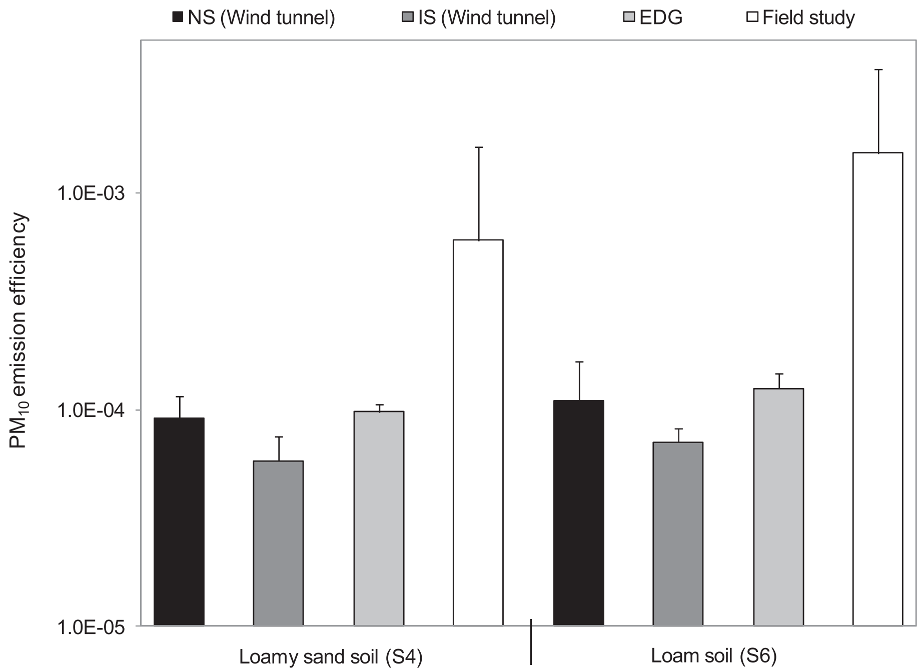

Fig. 3. PM_{10} emission efficiencies obtained by different methods for a sandy (S4) and for a loam (S6) soil. Error bars represent standard deviations.

these authors compared their results with those obtained by other authors in field studies on a loam and sandy soil, finding similar trends. Despite the differences in the experimental conditions between these studies, the results indicate that the methodologies can reflect the effect of the soil texture on the PM_{10} emission efficiency.

The higher PM_{10} emission efficiencies measured in the field as compared to those obtained with the wind tunnel and the EDG can be attributed to several scale-related issues. In the first place, for the wind tunnel and the EGD the duration of the experiments was 4 and 5 min, while under field conditions the average duration of the sampling time was 360 min. Moreover, in the wind tunnel, stable and high wind velocities and the extra injection of saltation produced increased values of Q, which reduces the PM_{10} emission efficiency. In the field studies, the saltation occurs intermittently (Stout and Zobeck, 1997), while dust emission occurs more frequently during erosive events because of direct suspension (Sharratt et al., 2007) and also particles moving slowly by creep, below the sampling heights, can contribute to dust emission. But the PM_{10} particles, once ejected over the saltation layer, keep moving by suspension at higher speeds than the saltating particles, following the wind profile. Therefore, they were consistently sampled by the dust monitors at the end of the field, while saltating particles were sampled at lower rates. In addition, in the field there are many conditions regarding the size and variability of the surface distribution of roughness elements, the presence of crusting, plants and micro topography changes that can have slowed down the movement of the coarser, saltating particles, affecting the sampling rate. Hence, the mass transport per unit of time averaged for the duration of a complete erosion event is less on the field than in a wind tunnel, where the saltation flux can be sampled almost completely.

It is widely known that the PM_{10} emission by wind erosion is mainly a produced by saltation (Shao et al., 1993; Houser and Nickling, 2001). This mechanism releases the PM_{10} attached both to the saltating particles and to the particles that get hit by others on the surface of the soil. The efficiency is given by the relation between the energy of the saltating particles (which depends on the speed, size and density of the particle), the cohesion forces that holds the PM_{10} attached to the particles, and to the total content of PM_{10} in the soil (Alfaro, 2008; Panebianco et al., 2016). Thus, keeping the last two variables constant, the emission efficiency depends only on the energy applied to the dynamic particle-surface system. In the case of the tunnel this depends on the wind speed and time of saltation, whereas in the case of the dust generator the total energy applied to the system depends on the time of simulation, given that the diameter of the generating chamber, the revolutions per unit time and the particle size for a given soil are constant. Hence, in this study, the energy applied to each soil with the dust generator was higher, considering that PM_{10} emission efficiency measured with the dust generator was 17% and 41% higher than those measured during NS and IS treatment in wind tunnel.

Panebianco et al. (2016) found that even under wind tunnel conditions, the PM_{10} emission efficiency increases along the duration of the erosion event because mass transport (Q) decreases more rapidly with time than the PM_{10} emission, particularly on fine-textured soils. Considering that the experiments performed in both the wind tunnel and the EDG were of short duration, as mentioned above, it could be expected that the emission efficiency in the laboratory studies will increase with time, increasing the average efficiency of the entire experiment. Thus, one of the possible ways to obtain efficiency values of comparable magnitudes between field and laboratory studies would be to increase the duration of the latter, although a considerable amount of work is still needed to test this.

Despite the differences, the general trends of the results for both soils were maintained when comparing the results found with the three methodologies. Taking into account the significant amount of resources needed for field studies, methods based on dust generators and wind tunnels are promising for obtaining information on the emission efficiency at lower costs. Nevertheless, further studies are needed to accurately correlate the results obtained at the laboratory with the corresponding field measurements.

4. Conclusions

Three methodologies frequently used for the estimation of the PM_{10} emission efficiency were compared: dust generator (EDG, Easy Dust Generator), wind tunnel and field measurements.

The results of this work indicate that both the wind tunnel and the EDG can be used to study the effect of soil texture on PM_{10} emission efficiencies. Results found with of both methods showed a linear and significant correlation. Wind tunnel values were between 0.66 and 0.75

times the EDG values, depending on the saltation rate. According to linear correlations between a soil texture index for cohesion strength and PM_{10} emission values, both methods were able to reflect the effect of soil texture. Results point out the importance of setting adequately the duration of the measurement time in both wind tunnel and dust generator in order to obtain comparable absolute values of PM_{10} emission efficiencies.

When compared with the field measurements, the difference of the working scales in space and time, created differences on the emission efficiency values by around 3 orders of magnitude. The PM_{10} emission efficiency measured under field conditions was higher than in the wind tunnel and the EDG because at the field scale the amount of soil removed per unit time (the mass transport, Q) that can be adequately sampled is relatively low as compared to the simulated, laboratory scales, while the amount of PM_{10} that can be sampled at the end of a plot remains relatively high. Moreover, duration of wind storms at the field were much higher than the duration of the laboratory experiments, and PM_{10} emission efficiency (PM_{10}/Q) increases along the duration of the erosion event, as the mass transport (Q) runs out.

The laboratory methods are promising for determining field emission efficiencies, but it is necessary to make further studies in order to be able to extrapolate the results to the open-air conditions.

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