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CONTENT

Session I: Climate change, soil organic carbon and soil degradation processes

1.1.O

GREENHOUSE GASES PRODUCTION IN THE SOIL PROFILE OF A CORN FIELD UNDER FLOOD AND SPRINKLER IRRIGATION

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INTRODUCTION

The concentration of the main greenhouse gases (GHG), namely carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), in the soil profile may provide information about the production and transport of these gases throughout the soil. The production and consumption of these gases in the soil are controlled by environmental conditions and microorganism activity (Wang et al, 2013). The concentration of CO_2 in the soil profile is the result of root and microbial respiration (Risk et al, 2002). Nitrous oxide (N_2O) has 265 times the radiative forcing potential of CO_2 (IPCC, 2014) and it is produced mainly by microbial transformations of inorganic N through nitrification and denitrification processes (Deng et al, 2015). Methane (CH_4) has 28 times the radiative forcing potential of CO_2 and it is controlled by the balance between consumption in well aerated zones and production in anoxic zones (Le Mer and Roger, 2001 In Mediterranean agroecosystems, soil GHG concentrations are affected by irrigation due to the impact of water application on soil processes controlling these concentrations (Kallenbach, et al. 2010).

The knowledge of the GHG concentration dynamics in the soil profile and how management practices (e.g. irrigation system) affect vertical GHG concentrations can improve our understanding of GHG exchange between soil and atmosphere. The objective of this study was to assess the concentration of CO_2, CH_4 and N_2O throughout the soil profile under two different irrigation systems (i.e. sprinkler vs. flood irrigation) during the growing season of a corn (*Zea mays* L.) crop.

MATERIAL AND METHODS

This work was performed at the experimental farm of the Estación Experimental de Aula Dei (EEAD-CSIC) in Zaragoza, NE Spain (41º 43’ N and 0º 49’ W), on a tilled corn field. The area has a mean annual precipitation of 343 mm, a mean annual air temperature of 14.8 ºC and a PET of 1100 mm. The soil at the experimental site is a silty loam soil (21% sand, 63% silt, 16% clay), with a basic pH of 8.0 in the upper 0–50 cm soil layer. Gas concentrations of CO_2, CH_4 and N_2O were measured in the soil profile at 10, 20 and 40 cm depth under two different irrigation systems: (i) sprinkler irrigation and (ii) flood irrigation. Irrigation frequency depended on the system type, with two times per week for sprinkler irrigation and every 10 days for flood irrigation. This irrigation frequency was maintained from June to August 2016. Irrigation was applied according to the crop water requirement, with a total amount of water applied of 7070 m^3 ha^{-1} and 8840 m^3 ha^{-1} in sprinkler and flood irrigation, respectively. Corn sowing was carried out on 12 April 2016. The same amount of fertilizer was applied in both irrigation systems. At planting, 800 kg ha^{-1} of a NPK (8-15-15) compound fertilizer was applied. On 13 June 2016, 740 kg ha^{-1} of calcium ammonium nitrate (27% N) was applied as top dressing (V10).

Gas samples were collected along the soil profile using passive soil gas samplers based on the design proposed by Kammann et al (2001). Each sampler consisted of a silicone tube 15 cm long (yielding 106 cm^3) closed with silicone septa at both ends. At one end, a stainless steel tube was inserted to connect the sampler with the soil surface. The end of the steel tube was closed with a three-way stopcock, from which air samples were taken using a 10 mL syringe. Immediately following collection, gas samples were transferred to preevacuated 4.5 mL Exetainer vials (Labco, High Wycombe, UK). From June to September 2016, the gas concentration was measured weekly and with a daily frequency during the five days following N fertilization and flood irrigation events. Concentrations of GHG in the gas samples were determined using a gas chromatograph (Agilent Technologies 7890B) equipped with a flame ionization detector (FID) coupled with a methanizer for CO_2 and CH_4 analysis, and an electron capture detector (ECD) for N_2O analysis. Soil-air sampling started 45 days after the installation of the soil gas samplers to minimize the possible effects of soil probes installation on soil GHG production. Soil temperature and moisture content were also monitored at the same depths every 30 min using 5TM soil moisture and temperature probes (Decagon Devices, Pullman, WA) connected to dataloggers. Water filled pore space (WFPS, %) was calculated from volumetric soil moisture content and soil bulk density measurements, assuming a soil particle density of 2.65 Mg m^{-3}.

RESULTS AND DISCUSSION

In both irrigation systems and for the entire measurement period, the CO_2 concentration increased with soil depth, with mean concentration values ranging from 730 cm^3 m^{-3} at 10 cm depth under flood irrigation to 4186 cm^3 m^{-3} at 40 cm depth under sprinkler irrigation (Table 1). There was a great variability in the daily CO_2 concentration under sprinkler irrigation (Fig. 1a), with the greatest value at 40 cm depth (16593 cm^3 m^{-3}). In flood irrigation, the highest CO_2 concentration value (5083 cm^3 m^{-3}) was observed after the irrigation period and the lowest values in the 24 hours following water applications (Fig. 1b). The lowest concentrations values found in the flood treatment could be explained by the higher values of WFPS in this irrigation system. Other authors (Wunderlich and Borken, 2012; Morstchenbacher et al, 2015) reported a similar behavior under flooded conditions relating the decrease of CO_2 concentration with a decrease in oxygen availability in flood irrigation systems. In contrast, the daily and seasonal concentration of CH_4 showed similar values between the two irrigation systems along the soil profile (Table 1; Fig. 1c, 1d).

Table 1. June to September mean and standard deviation of CO_2, CH_4 and N_2O concentration in cm^3 m^{-3} and water filled pore space (WFPS) in percentage at different soil depths (10, 20 and 40 cm) depending on the type of irrigation system (sprinkler and flood irrigation).

	[CO_2] (cm^3 m^{-3})		[CH_4] (cm^3 m^{-3})	
Depth (cm)	Sprinkler	Flood	Sprinkler	Flood
10	936 ± 469	730 ± 400	2.91 ± 0.77	3.00 ± 0.76
20	1745 ± 2708	1034 ± 828	2.79 ± 0.59	2.95 ± 0.48
40	4186 ± 6363	1530 ± 2205	2.44 ± 0.98	2.87 ± 0.68
	[N_2O] (cm^3 m^{-3})		WFPS (%)	
Depth (cm)	Sprinkler	Flood	Sprinkler	Flood
10	0.23 ± 0.23	1.30 ± 7.04	46.01 ± 8.50	74.37 ± 17.19
20	0.27 ±0.28	2.10 ± 11.68	53.82 ± 7.84	74.86 ± 11.23
40	0.62 ± 1.07	1.05 ± 4.35	59.16 ± 7.97	82.21 ± 10.52

A strong increase in soil N_2O concentration was observed under flood irrigation after N fertilization and irrigation applications, reaching values close to 24 cm^3 m^{-3} at 20 cm depth (Fig. 1f). In the sprinkler irrigation treatment, however, N_2O concentration was lower than 1 cm^3 m^{-3} at the three soil depths measured (Fig. 1e). Differences in N_2O soil concentration between sprinkler and flood irrigation treatments could be related with the different soil water content found between irrigation treatments and the impact of oxygen availability on denitrification processes (Clough et al, 2005).

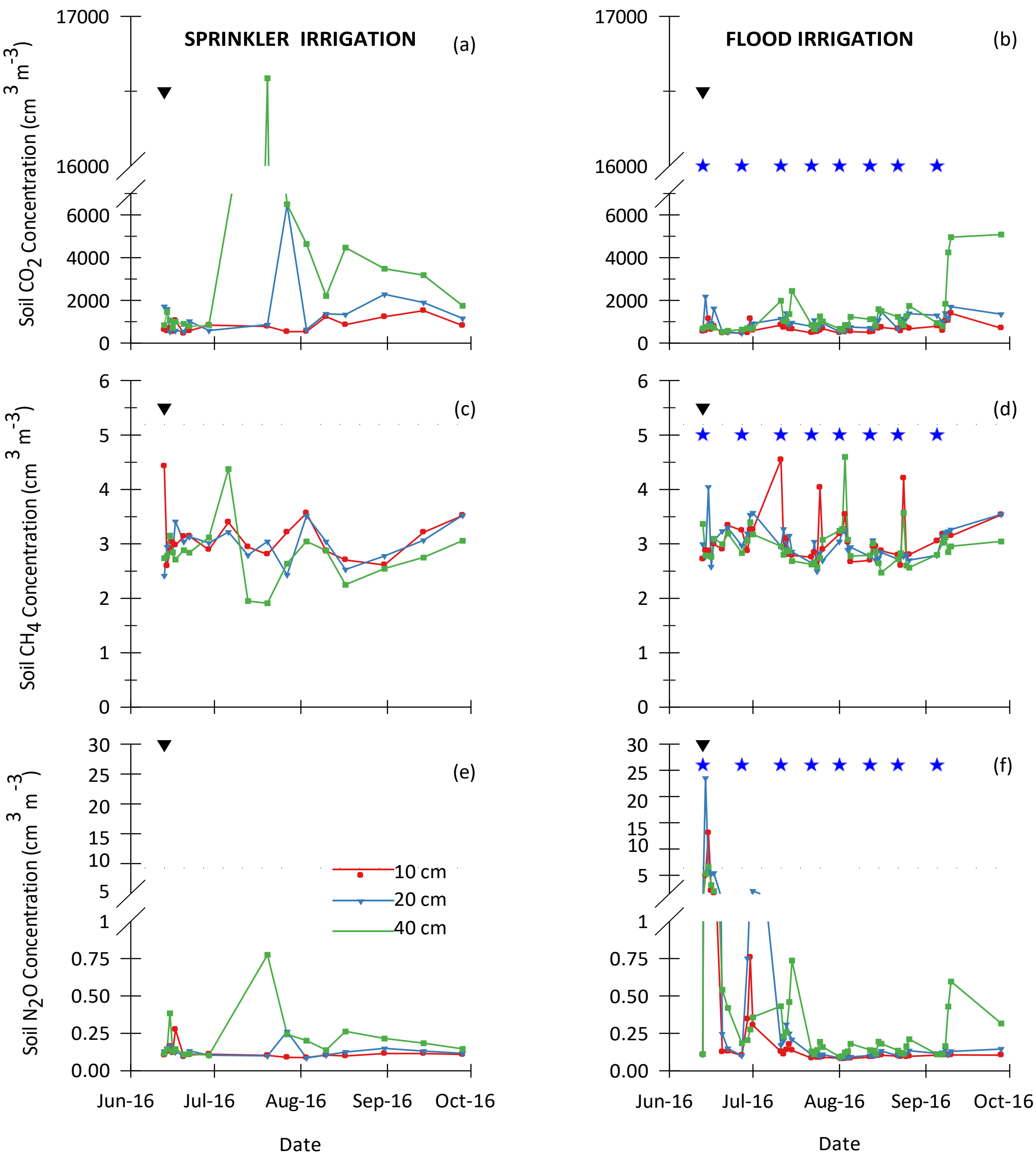

Figure 1. Dynamics of soil CO_2, CH_4 and N_2O concentrations under sprinkler (left graphs) and flood irrigation (right graphs) as affected by soil depth (10, 20 and 40 cm). Black triangles and blue stars indicate N fertilizer application and flood irrigation events, respectively.

CONCLUSION

During the corn growing season, soil CO_2 and N_2O concentrations were affected by the irrigation system and soil depth. Sprinkler irrigation resulted in greater concentration of CO_2 compared with flood irrigation, while the greatest N_2O concentration was observed under flood irrigation, due to the higher WFPS values observed in this treatment. Contrarily, the irrigation system and soil depth did not affect soil CH_4 concentrations. This work highlights the relevance of the irrigation system in the production of GHG in the soil profile and the importance of irrigation management to control soil GHG emissions in Mediterranean agroecosystems.

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1.2.O

ASSESSMENT OF ORGANIC MATTER AS INDICATOR OF SOIL ENRICHMENT IN A MEDITERRANEAN SUBWATERSHED WITH INTENSIVE AGRICULTURAL LAND USES

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INTRODUCTION

Agroecosystems occupy 40% of the global terrestrial surface (FAO 2013). They are both providers and consumers of ecosystem services (ES). In fact, the conversion of natural ecosystems to agroecosystems can reduce the flow of certain ES (Matson et al. 1997), leading to major environmental degradation, including deforestation, soil erosion, nutrient leaching and water abstraction, diversion, and pollution (Simoncini 2009). This is important given that crops usually occupy the most productive lands, like floodplains, one of the most endangered habitats and biodiversity hotspots (Moss and Monstadt 2008). An assessment of ES is a useful tool in order to make effective decisions in the management of agroecosystems (Trabucchi et al. 2014).

Soil enrichment supports other ES (MEA 2005). ES derived from soil enrichment relate to the maintenance of crop productivity on cultivated lands and the integrity and functioning of natural ecosystems (de Groot et al. 2002).

ES provision depends on the studied scale (Power 2010). Land-use patterns affect ES provision (Mitchell et al. 2013). The amount of each ES supplied in a given area depends on both the per hectare provision of service by land-use type and the total amount of each land use found in the study area (Felipe-Lucia et al. 2014).

Most studies use GIS and satellite images, global databases, or models to estimate ES provision. Few studies however, have collected local data across land uses (Raudsepp-Hearne et al. 2010), despite evidence that these data are critical to an accurate assessment of service provision (Eigenbrod et al. 2010). Soil enrichment is a service directly connected to the interaction of rock disintegration and the accretion of animal and plant organic matter (Costanza et al. 1997). Therefore, the organic matter content on topsoil could be used as a suitable indicator of soil enrichment (Lavelle et al. 2006).

METHODS

The study area is a subwatershed (169.5 ha) located in the south of the Flumen River watershed in northeast Spain. Most of it is occupied by intensively irrigated agricultural land use (129 ha). We identified 10 different land use types, 4 natural and semi-natural land uses (hereinafter natural) and 6 agricultural land uses (**Table 1**). For that, we used the Spanish crop and land-use digital map (Ministerio de Medio Ambiente Y Medio Rural y Marino 2009) withArcGIS 10.2.1 (ESRI) and we completed this data with field observations. We assessed the soil enrichment using organic matter (OM) content in topsoil as an indicator. We measured the OM at 0-5 cm and 5-10 cm of depth in a network of 69 plots in the whole subwatershed, and we made 3 replicates in each plot. We checked significant differences between OM content in land uses by an one-way ANOVA ($\alpha=0.05$) followed by multiple pairwise

Table 1. Land uses characteristics in the study area.

Land uses	Total area (ha	Proportion at watershed scale	Soil enrichment at parch scale	Soil enrichment at watershed scale
Riparian forest	2.54	0.15	5.00	0.07
Conifer forest	33.67	0.20	4.57	0.91
Restored riparian forest	4.67	0.03	4.00	0.11
Olive grove	0.30	0.002	3.52	0.01
Corn crop	25.90	0.15	3.22	0.49
Steppe scrubland	6.36	0.04	3.15	0.12
Alfalfa crop	36.04	0.21	3.13	0.67
Barley crop	16.80	0.10	2.92	0.29
Rice crop	14.54	0.09	2.45	0.21
Abandoned crop	28.63	0.17	2.23	0.38
Total	169.47	1		**3.25**

comparisons using Tukey's Honestly Significant Difference (HSD) (Zar 1999), using the statistical software R. We calculated the soil enrichment at patch scale for each land use as the average of the of all the plots of the same land use. Following this, we normalized the results in a 0-5 relative scale using the matrix method (Burkhard et al. 2014). To estimate it at subwatershed scale, we multiplied the value at patch scale of each land use by the proportion it occupies at subwatershed scale. Finally, we simulated four scenarios of land use change: (1) *restoration scenario* (restoration of riparian forests and transformation of abandoned crops into conifer forest and steppe scrubland), (2) *agricultural scenario* (transformation of abandoned crops into alfalfa crops), (3) *light agro-ecological restoration* (restoration of riparian forests and transformation of abandoned crops into alfalfa crops) and (4) *intensive agro-ecological restoration* (restoration of riparian forests; transformation of abandoned crops into alfalfa crops, conifer forest and steppe scrubland; scrubland row plantation between crops; and creation of an artificial small wetland). We used ArcGIS 10.2.1. (ESRI) to calculate the new area of each land-use, and re-estimated the new soil enrichment supply at subwatershed scale.

RESULTS

At both soil depths, natural land use presented a higher content in OM than agricultural land uses, with the exception of scrubland (**Figure 1**). At 0-5 cm of depth, significant differences in OM content were observed between natural land uses and between natural and agricultural land uses, but not between agricultural land uses. In contrast, at 5-10 cm of depth, we found fewer differences between natural and agricultural land uses, but in this depth, there were differences between agricultural land uses. However, the average value of OM of the two soil layers (0-10 cm of depth) shows high differences between natural and agricultural land uses, and also between agricultural land uses and between natural land uses. So, we chose the average of OM of the two soil layers as indicator of soil enrichment.

At patch scale, natural land uses, with the exception of scrubland, supplied more soil enrichment than agricultural land uses **(Table 1)**. However, at subwatershed scale, this ES depends mainly on the surface occupied by each land use in the territory **(Table 1)**. For instance, at patch scale, riparian forest supplied the highest soil enrichment, whereas at subwatershed scale, it is the land use providing the second lowest soil enrichment.

The restoration scenario supplies the highest soil enrichment at subwatershed scale (3.75), whereas the agricultural scenario provides less than any other scenario (3.41). The other two scenarios offer

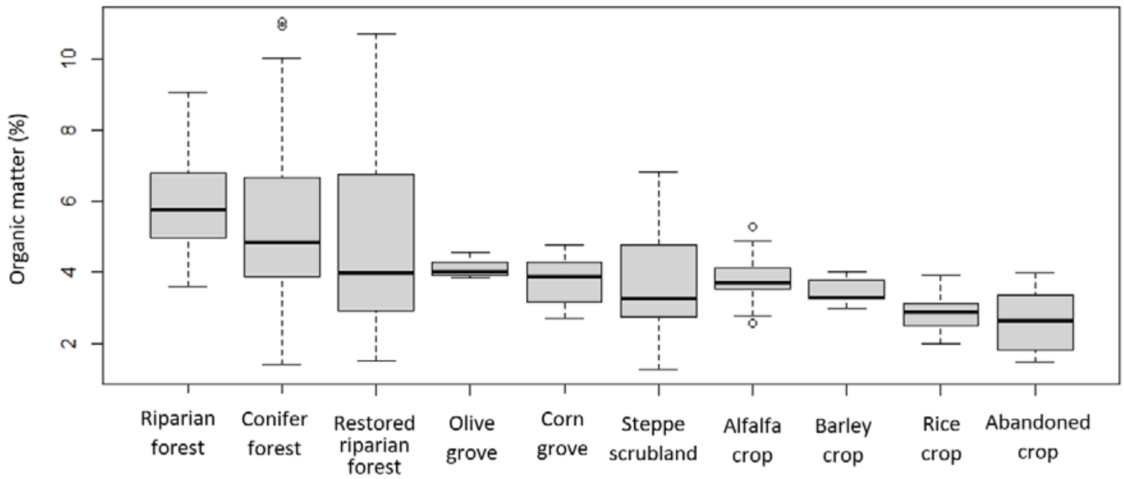

Figure 1. Box splots of the average of the percentage of organic matter at 0-5 cm and 5-10 cm of depth.

intermediate values, being a little greater the intensive agro-ecological restoration (3.59) than the light agro-ecological restoration (3.53). In any case, low differences were observed between the four scenarios, but indicative enough of the potential management of land uses to orientate changes in the provision of ES at subwatershed scale.

CONCLUSIONS

In the southern subwatershed of the Flumen River studied, natural and seminatural land uses provide more soil enrichment at patch scale than agricultural land uses, with the exception of scrubland. At subwatershed scale, this supply depends mainly on the surface occupied by each land use in the territory. Thus, it is critical to pay careful attention to the scale of analysis considered in order to make management decisions. Between different alternatives of simulated scenarios (from agricultural intensification until naturalization through restoration of degraded ecosystems), the restoration of riparian forests and abandoned crops as their potential natural ecosystems (conifer forest and steppe scrubland) would provide more soil enrichment at subwatershed scale. So, it is strongly recommended to incorporate the restoration of degraded ecosystems as a tool for soil enrichment and the sustainability of intensively used agricultural territories.

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1.3.O

BENEFITS OF BIOCHARS FOR SOIL QUALITY, NUTRIENT USE EFFICIENCY AND COWPEA (*VIGNA UNGUICULATA L.*) GROWTH IN ACID ARENOSOL

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1. INTRODUCTION

Acid soils, among them acid Arenosols, are abundant in tropical countries. N, P, and K inputs are required for optimum plant growth in these soils (Martins et al., 2014). Sustainable practices for soil fertility improvement are needed in these poor soils (Marenya et al., 2012). Application of inorganic fertilizer alone in acid arenosols is risky due the low physical fertility of the soil leading to crop failure and decreasing of financial resources to continue purchasing the fertilizer (Agegnehu et al., 2016). Organic amendments are an alternative to optimize the fertilizer use. Biochar as a soil amendment has been shown to be a useful practice to improve the quality of sandy soils (Wen et al., 2016). The extent to what the biochars can reduce the inorganic fertilizer rate without decreasing yield needs to be investigated. This study focuses on understanding the effect of biochar from different feedstocks in combination with NPK fertilizer on: i) soil chemical and biochemical properties; ii) N, P, and K use efficiency by cowpea (*Vigna unguiculata L.*) and iii) cowpea yield in an acid Arenosol.

2. MATERIALS AND METHODS

An albic Arenosol from Mozambique was used in a pot experiment. The following treatments: i. C (Control: without fertilizer or biochar application); ii. NPK (12-24-12) + Urea (46%) at 830 mg pot^{-1} (~ 400 kg ha^{-1}) and 217 mg pot^{-1} (~ 104.3 kg ha^{-1}), respectively; iii. BC1+1/2NPK (baby corn peel biochar + 415 mg pot^{-1} of NPK + 108.5 mg pot^{-1} of Urea); iv. BC1+NPK (baby corn peel biochar + 830 mg pot^{-1} of NPK + 217 mg pot^{-1} of Urea); v. BC2+1/2NPK (branches of tree biochar + 415 mg pot^{-1} of NPK + 108.5 mg pot^{-1} of Urea); vi. BC2+NPK (branches of tree biochar + 830 mg pot^{-1} of NPK + 217 mg pot^{-1} of Urea); vii. BC3+1/2NPK (rice husk biochar + 415 mg pot^{-1} of NPK + 108.5 mg pot^{-1} of Urea); viii. BC3+NPK (rice husk biochar + 830 mg pot^{-1} of NPK + 217 mg pot^{-1} of Urea) were assessed in a completely random design in a greenhouse experiment. All biochars were applied at a rate of 0.9 % (w:w). The experiment data were analysed for analysis of variance (ANOVA). A multiple comparison Duncan test (at 5% significance level) was used to compare the means.

3. RESULTS AND DISCUSSION

3.1. Effect of treatments on soil chemical properties

There was significant (*P < 0.05*) effect of treatments on pH (H_2O and KCl), available P, cation exchange capacity (CEC), exchangeable Ca, Mg, K, and Al, and base exchange saturation (V); however, total carbon (TC), total organic carbon (TOC), soil organic matter (SOM), and exchangeable Na were not significantly affected.

The soil pH increased significantly from 4.27 (in control) to 5.71 (*P<0.001*) for treatments including BC1 or BC2. Cation exchange capacity and available P were highest (*P<0.001*) in treatments including BC1. Soil pH was highest and exchangeable Al was lowest in treatments where BC1 was applied, followed by BC2 treatments. The reduction of acidity was due to the alkalinity of the biochar (pH = 9.63 in BC1) as reported by Zhu et al. (2014) and Solaiman and Anawar (2015). Biochar application improved the chemical properties of this acid sandy soil. Inorganic fertilization did not affect significantly the measured properties. For each type of biochar, there was a not significant difference between full rate of NPK and half rate of the same fertilizer for most soil chemical parameters.

3.2. Effect of treatments on soil enzyme activity

The overall impact of the treatments on soil enzyme activities was limited. There was a significant (*P < 0.05*) effect of treatments on xylosidase (xylo), acid phosphomonoesterase (acP), alkaline phosphomonoesterase (alkP), phosphodiesterase (bisP) and pyrophosphatase-phosphodiesterase (pyro). Biochars 1 and 2 promoted higher activity of pyro (*P<0.05*) and alkP (*P<0.01*), while inhibiting the activity of xylo and acP, probably due pH increase (Table 1). Most of these extracellular enzymes are pH dependent (Lu et al., 2015). In laboratory incubation experiments biochars produced from corn and pearl millet residues increased the activity of alkP in an acid soil (Purakayastha et al., 2015). Improving soil enzyme activities by biochar amended soils has been generally reported (Demisie et al., 2014; Lu et al., 2015; Pandey et al., 2016).

3.3. Effect of treatments on cowpea growth and yield

The effect of treatments was significant (*P < 0.05*) for all cowpea growth and yield parameters (Table 2). All treatments (including NPK) resulted in improved cowpea growth and yield compared to control. The treatments including biochar presented higher dry biomass, pod yield and nodules than the control and NPK alone. Biochar application promoted plant-root interactions with symbiotic bacteria by increasing the root nodulation. Application of BC1+NPK increased by 393 and 76% dry biomass compared with the control and NPK treatments, respectively. In addition, dry pod, root and nodules yields increased by at least 776, 1342, and 344 % in treatments BC1+1/2NPK, BC2+1/2NPK and BC3+NPK compared with the control, respectively. The increase of cowpea yield with the application of biochar is associated with the improvement of soil physical, chemical and biochemical properties. Effects of biochar application on crop yield have been reported previously (Domene et al., 2015).

3.4. Effect of treatments on N, P and K use efficiency by cowpea

The effect of treatments was significant (P < 0.001) for N, P and K agronomic efficiency (AE, g of pod g^{-1} N (or P or K) applied), physiological efficiency (PE, g of pod g^{-1} N (or P and K) uptake), apparent recovery efficiency (ARE, N (or P and K) uptake per cent of P applied), and harvest index

(HI, N, (or P and K) uptake in pod per cent of total N (or P and K) in plant) (with exception of K). Treatments including BC1 presented higher nitrogen AE, PE, and HI. The highest nitrogen ARE was found in treatment BC1+1/2NPK. Treatments BC1, BC2+NPK, BC3+NPK presented highest phosphorus AE, PE, and ARE. The higher AE and PE are attributed to the efficient nutrient uptake and utilization by the cowpea (Singh et al., 2003; Choudhary and Kumar, 2014). Higher N, P, and K uptake by cowpea in these treatments implies that the biochar treated soil maintained higher concentration of these nutrients in soil solution (Agegnehu et al., 2016).

Table 1
Treatments effect on biochemical characteristics of soil after the pot experiment (Numbers in parentheses are the standard deviation (n=3)).

Enzymes	Treatments								ANOVA [a)]
	Control	NPK	BC1+1/2NPK	BC1+NPK	BC2+1/2NPK	BC2+NPK	BC3+1/2NPK	BC3+NPK	
Xylo, nmol g^{-1}	0.46 (0.09) ab [b)]	1.29 (0.20) c	0.66 (0.07) abc	0.6 (0.2) abc	1.53 (1.29) bc	1.16 (0.07) bc	0.4 (0.1) a	1.01 (1.02) abc	*
AcP, nmol g^{-1}	11.66 (2.24) ba	18.25 (2.06) b	6.90 (0.25) a	4.73 (1.68) a	11.96 (6.69) ba	12.80 (1.52) ba	5.70 (2.81) a	13.14 (10.15) ba	*
BisP, nmol g^{-1}	0.49 (0.45) ba	1.67 (0.27) cba	2.44 (0.59) c	1.18 (0.23) cba	2.13 (1.00) cb	2.81 (0.72) c	0.12 (0.20) a	1.72 (2.26) cba	*
Piro, nmol g^{-1}	0.31 (0.39) ba	0.61 (0.23) cb	1.06 (0.45) cb	0.25 (0.14) ba	1.39 (1.32) cb	2.36 (1.10) c	0.03 (0.05) a	1.14 (1.82) cba	*
AlkP, nmol g^{-1}	3.35 (2.61) ad	9.28 (2.28) cba	30.79 (14.21) c	10.86 (2.69) cba	20.26 (9.91) cb	25.78 (8.16) c	1.97 (1.27) d	11.03 (14.85) bcd	**
dsDNA, μg g^{-1}	2.80 (0.34) cba	3.98 (1.39) cb	2.33 (1.26) ba	1.55 (0.55) a	2.06 (0.69) a	1.28 (0.25) a	2.32 (1.27) ba	4.24 (1.23) c	*

[a)] NS not Significant (*P > 0.05*); * Significant at the 0.05 probability level; ** Significant at the 0.01 probability level;

[b)] Means with different lowercase letters under the treatments differed significantly per Duncan multiple mean comparison test at 95% of significance.

Xilo: Xyloxidase; AcP: Acid phosphomonoesterase; BisP: Phosphodiesterase; PiroP: Pirophosphate phosphodiesterase; AlkP: Alkaline phosphomonoesterase;

Table 2
Treatments effect on cowpea growth and yield (Numbers in parentheses are the standard deviation (n=3)).

Cowpea yield	Treatments								ANOVA [a)]
	Control	NPK	BC1+1/2NPK	BC1+NPK	BC2+1/2NPK	BC2+NPK	BC3+1/2NPK	BC3+NPK	
Dry biomass, g pot^{-1}	6.5 (2.1) e [b)]	18.2 (1.5) d	29.47 (6.2) cba	32.07 (2.9) c	26.7 (2.9) cba	29.8 (6.6) cb	22.1 (4.5) ad	23.8 (0.6) abd	***
Dry pods, g pot^{-1}	1.7 (0.5) a	6.8 (3.5) b	15.01 (5.1) c	19.47 (2.2) c	0.3 (0.3) a	16.7 (2.0) c	2.1 (2.7) a	11.4 (2.7) cb	***
Dry roots, g pot^{-1}	0.4 (0.1) a	1.7 (0.4) b	1.92 (0.7) b	1.84 (0.6) b	5.1 (0.2) c	2.6 (2.1) b	4.1 (1.8) bc	2.0 (0.9) b	***
Dry shoot[c)], g pot^{-1}	1.3 (0.3) a	5.8 (2.4) b	6.16 (1.2) bc	7.49 (1.8) bc	11.0 (1.7) c	6.4 (2.6) bc	9.5 (4.1) bc	5.9 (1.3) b	***
Plant height, cm	20.3 (1.3) a	29.7 (5.7) b	28.33 (4.5) b	30.33 (2.5) b	29.7 (3.5) b	29.3 (0.6) b	26.3 (2.5) b	30.7 (3.8) b	*
Diameter of stem, cm	0.5 (0.0) a	0.6 (0.1) b	0.86 (0.1) b	0.86 (0.1) b	0.8 (0.1) b	0.7 (0.1) b	0.7 (0.1) b	0.7 (0.1) b	**
Number of nodules	29.3 (13.2) a	109.7 (49.9) ba	170.67 (40.1) b	200.67 (25.0) b	151.7 (82.8) b	143.8 (8.0) b	132.8 (42.9) b	130.3 (94.5) b	*
Number of leaves	12.0 (0.0) a	20.3 (5.0) b	27.00 (5.2) b	28.67 (4.0) b	21.7 (3.1) b	23.0 (8.2) b	21.7 (1.2) b	23.0 (4.4) b	*

a) * Significant at the 0.05 probability level; ** Significant at the 0.01 probability level; *** Significant at the 0.001 probability level; b) Means with different lowercase letters under the treatments differed significantly per Duncan multiple mean comparison test at 95% of significance; c) Shoot = Stem + Leaves;

4. CONCLUSION

This work supports the idea that application of biochars in combination with NPK fertilizer in acid sandy soil improved soil chemical properties and enzyme activities. The changes in soil properties were accompanied by higher N, P, and K uptake and efficiency in treatments including biochars and, therefore, higher cowpea growth and yield. Among the biochars, baby corn peel biochar (BC1) followed by tree branches biochar (BC2) affected most the soil properties and cowpea yield. Both soil properties and yield parameters in cowpea were not significantly different between full NPK rate and half rate of NPK for each type of biochars. The reduced fertilizer application can reduce the cost of food production, mainly for poor farmers. Further investigation under field conditions is needed to validate these results and quantify the long-term benefit of biochars.

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1.4.O

LAND DEGRADATION BY GULLIES AND LANDSLIDES, SOIL CONSERVATION AND SUSTAINABLE LANDUSE UNDER THE CLIMATE CHANGE PERSPECTIVE, IN A SMALL WATERSHED IN EASTERN ROMANIA

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INTRODUCTION

The application of Law 18 from 1991, regarding the reallotment of the agricultural land to previous landowners, in Romania, had two main negative consequences: a huge fragmentation of the agricultural land and the arrangement of a great number of long and narrow plots, according to local tradition, following the up-and-down hill direction. These two consequences are totally incompatible with a conservative agriculture.

Farmers in Baltati watershed (Eastern Romania) are facing another huge problem, too: land degradation by gullies and massive deep seated landslides.

This paper summarizes the results of studies and inventories of gullies and landslides in Baltati watershed, illustrates the actual structure of agricultural land ownership, including the land consolidation that took place during the last decade, the solutions of land organization, land reclamation, soil conservation, and for sustainable land use, provided on the basis of these analyses and having the climate change perspective.

METHODS

Within the framework of a research project, the Geographical Information System (GIS) of an 8449.6 ha area was realized. The GIS included layers such as: geology, relief, hydrology, vegetation cover, landuse, road network, landslide and gully inventory, and the map of actual farms which resulted after the land consolidation of a huge number of small, long and narrow plots. The actual geomorphological parameters of land degradation features (landslides and gullies) were updated upon the most recent aerial photos and by means of a professional GPS equipment. Based on this information, and also on data regarding climate, hydrogeology, soil map, seismicity, socio-economic data, taken from the field and/or the scientific literature, and according to the actual regulations in Romania, the landslide hazard map of the area was prepared. The entire GIS and specially the mathematical formulae used in the preparation of the landslide hazard map were used to make several designs containing solutions of land organization, land reclamation, soil conservation, and for sustainable land use, having the climatic change perspective.

RESULTS

The surveys regarding gullies and landslides in the studied area, based on maps resulted from different approaches (geomorphological – PhD researches, the most recent cadastral maps available, observations on aerial photographs, and measurements by professional GPS equipment), gave us an image of the amplitude of processes of land degradation: 89 landslides occupy 12.5% of the studied area and 535 wide and shallow gullies occupy 1.1% (Figure 1). The different approaches of the land degradation processes resulted in considerably different data regarding the geomorphological parameters. However, even though the cadastral approach should be, and actually is, the most reliable for practical purposes, the subjective approach of the geomorphologist (Darie, 2013) concerning the "stabilized landslides" is highly consistent with the digital elevation model, the slope map, and the landslide hazard map, leaving the white areas in Figure 1 as the best (most appropriate) available arable land in the area.

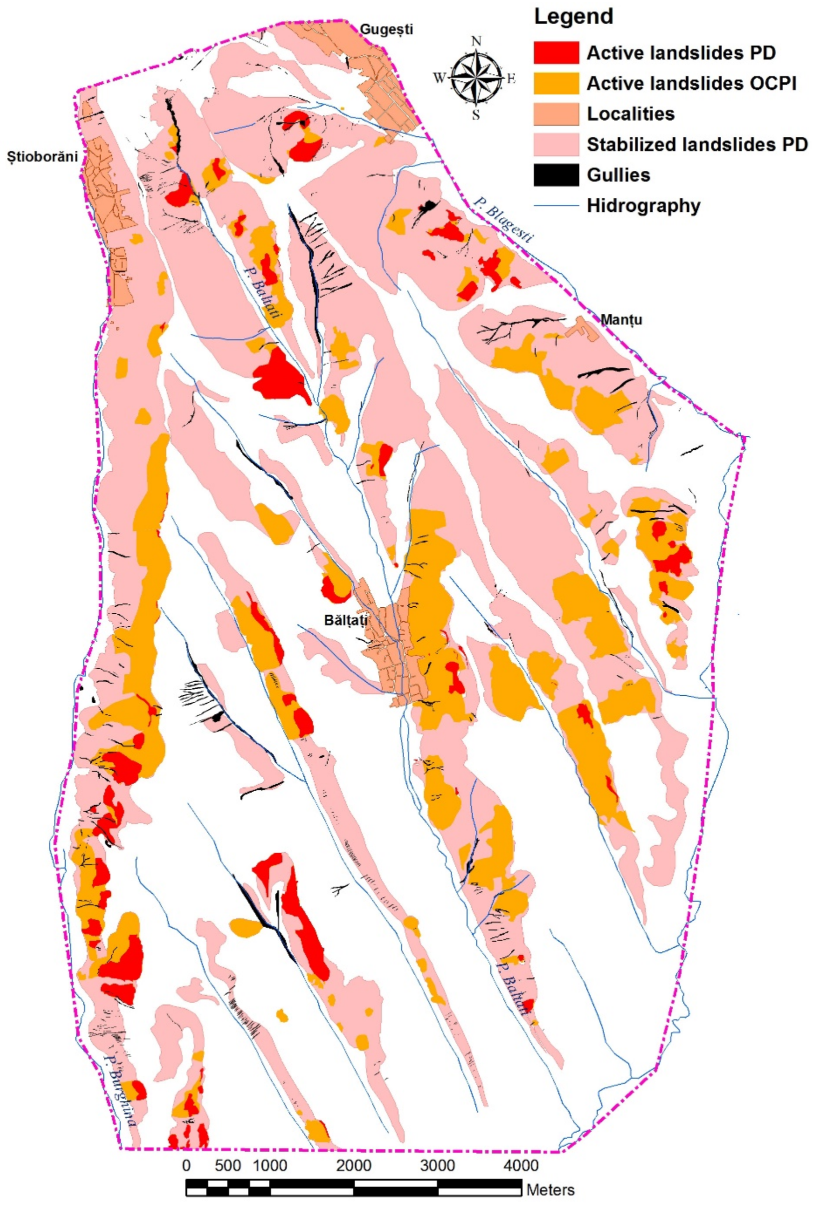

Figure 1. Land degradation by gullies and landslides in Baltati watershed, Eastern Romania. ("stabilized landslides PD" and "active landslides PD" according to Darie, 2013, and "active landslides OCPI" according to OCPI, Vaslui, 1983)

In Table 1 and 2 geomorphological data of gullies and landslides are summarized.

Table 1. Morphometrical data of gullies in Baltati area.

	Area (ha)	Length (m)	Width (m)	Depth (m)	Depth to Width Ratio	Volume (m^3)
Total	89.23					1,824,584.05
Min	0.013	15.34	2.05	1.23	0.48	13.20
Max	7.01	1930.14	21.45	12.61	10.01	117,040.22
Average	0.17	153..61	8.43	1.48	8.54	3,385.13

Table 2. Morphometrical data of landslides in Baltati area.

Cathegory	Count	Total area (ha)	(%) of total area	Min area (ha)	Max area (ha)	Average area (ha)	Perimeter (km)
Active landslides OCPI Vaslui (1983) and RDCSEC Perieni (2016)	89	1077.97	12.5	0.12	119.03	12.11	137.01
Active landslides acc. Darie P. (2013)	71	221.59	2.6	0.01	25.78	3.12	54.41
Stabilized landslides acc. Darie P. (2013)	32	3577.86	42.3	0.03	792.82	111.81	261.90

All these observations and measurements, and specially the Depth-To-Width Ratio, confirm our previous conclusions (Hurjui, 2008) that landslides are largely distributed in areas having a clayey geology (lithology), and gullies in areas with the same clayey lithology are wide and shallow; and conversely, in non-clayey areas, landslides are sparsely distributed and gullies are narrow and deep.

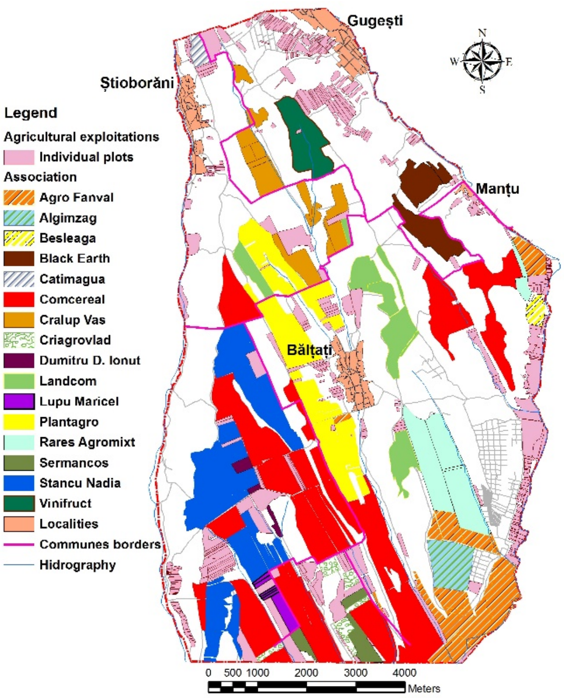

Figure 2. The map of actual consolidate agricultural land in 19 exploitations.

As long as in Romania there is that impression of a huge fragmentation of the agricultural land (48 million individual plots on about 9 million hectares) and an inadequate disposition of the long and narrow individual plots following the up-and-down hill direction, our surveys showed that, at least in this particular area, something very positive happened: 3760.09 ha (77% of the total 4854.97 ha of agricultural land) were consolidated in 19 agricultural commercial exploitations as a matter-of-course, and only 1094.88 ha (23%) belonging to 615 individual land owners remained disassociated, with no chance of future congregation (Figure 2).

This confirms the previous observations regarding the land degradation processes and shows that the private entrepreneurs consolidated the best available agricultural land in their 19 exploitations.

Another interesting result of our field investigations is that almost all those 19 private entrepreneurs are very determined to apply the best conservative practices on their land.

Based on these findings, along with the analyses of relief, actual land use, vegetation cover, hidrology, climate, seismicity, hidrology, anthropic impact, we have provided the appropriate solutions for conservative organization of the agricultural land and sustainable agriculture. The solutions were of two kind: a) structural works, of great amplitude, which necessitate the intervention of the State or some powerful financial private institutions, and b) less costly low amplitude works, which can be easily performed by the land owners, such as: correcting/molding of torrents, rills, or ephemeral gullies, stabilizing of gully banks and thalwegs, tracing of grass strips separating the strip crops, designing the grassed water ways, correcting the technological earthen roads, and so on.

CONCLUSIONS

1. Baltati watershed, having an area of 8449.6 ha is highly affected by landslides (89 areas affected, having a total area of 1077.97 ha – 12.5% from the total area), and by gullies (535 gullies occupying an area of 89.23 ha – 1.1% of the total).
2. The geological structure of the area, predominantly clayey-sandy, determined the occurrence of a small number of landslides, which, in turn, have large dimensions (e. g. 13 km long on the left side slope of Burghina Valley) and a great number of gullies (535), which, in turn, have small dimensions.
3. About 77.4% of the total arable land (4854.97 ha), meaning 3760.09 ha is exploited in a consolidated manner by 19 private entrepreneurs (four such commercial exploitations work over 50% of that area). A number of 615 landowners are exploiting an area of 1094.88 ha (22.6%) with no chance of association in the future.
4. All 19 private entrepreneurs are definitely determined to apply the best conservative practices on their land.
5. When comparing the OCPI maps of 1983 with the actual situation, one may see the following: grassed strips have been destoyed by 71%, the forested area increased by 25%, the area covered by pastures increased by 65%, all orchards and vineyards existing before 1989 have been disbanded, the area of rural localities has doubled.
6. Within the Baltati watershed would be necessary the following: forestation of degraded pastures on an area of 1076.90 ha, plantation of shelter belts (wind breaks) on a length of 27.11 km, 17.34 km of technological earthen roads have to be retraced.

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1.1.O

IMPACT OF CLIMATE CHANGE ON WATER AVAILABILITY IN THE MUGU KARNALI RIVER BASIN OF NEPAL HIMALAYA

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INTRODUCTION

Nepal is situated between China in the north and India in the east, west and south. Mt. Everest (8,848 amsl) - the highest peak in the world is located in Nepal. One of the river basins lying in the Himalayan region of Nepal is Mugu Karnali. Mugu Karnali River basin stretches between 22◦46'40"N -29◦46'30"N latitude and 81◦49'57"E -82◦49◦57"E longitude in the Trans-Himalayan Zone of Nepal extending up to Tibet Autonomous Region of the People`s Republic of China in its` Northern border. The total area of basin is 5,358 sq.km. (Figure-1). The elevation ranges from 1,201 to 6,815 m amsl. The basin experiences temperate to alpine types of climate. It is rich in surface water resources including the Rara Lake (10.80 sq.km.)-the largest lake in the country. The basin offers home to a variety of biodiversity including medicinal herbs and wildlife. Nearly 55,286 human populations with main sources of subsistence agriculture reside in the basin.

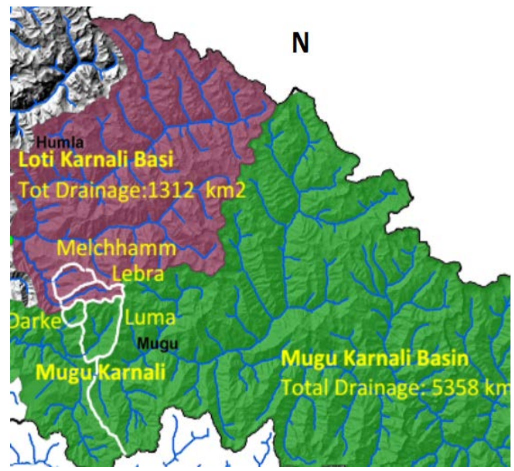

Figure 1: Mugu Karnali Basin

The life of people in the basin is very hard due to remoteness and lack of easy access to outside the world. The diversity of landscape decorates the basin making it exceptionally beautiful. The wilderness features of basin makes it unique. The objective of the paper is to study the impact of climate change on water sources and its effect on availability of water for household and irrigation uses, and livelihoods of the people in the basin.

MATERIALS AND METHODS

Review of Literature

The available pertinent literatures were reviewed and important information was included in the study. Relevant information published in the newspapers was also used. During study preparation, consultations with the researchers involved in similar field of studies were carried out. Such consultations were found useful.

Field Work and Seeking out for Climatological Stations

Researchers visited Mugu Karnali River basin for the purposes of RAP-3 study since 2014. During field visit, consultations and interactions with the local people including observations of the area were carried out on impact of climate change and availability of water. Knowledge earned on this theme has become valuable to carry out present study.

In Nepal, time series climatological data is rarely available. For basin, two climatological stations namely, Gam Shreenagar (Station Index 0306) and Rara (Station Index 0307) were found operational. But available data of precipitation and temperature were not sufficient.

RESULTS AND DISCUSSIONS

Impact of Climate Change on Precipitation and Temperature

The hydrological season in Nepal are classified into i) pre monsoon (March-May), ii) monsoon (June-September) and iii) post-monsoon (October-February). Nepal receives 75-80 percent of its annual rainfall during monsoon season with little rain during post-monsoon and no rain during pre-monsoon. Studies show that due to impact of climate change rainfall pattern and monsoon timing- onset and cessation have become increasingly erratic.

The precipitation trend analysis shows that the annual average precipitation over Nepal is decreasing at the rate of 9.8 mm per decade. The trends in annual discharges of the major river (e.g. Koshi, Gandaki and Karnali) basins indicate that the discharges in these major river basins are decreasing annually (WECS, 2011).

<u>Result of precipitation data analysis</u>

Analysis of precipitation data of the basin showed that volume of precipitation during monsoon decreased by 5.29 mm per year. Similarly, volumes of precipitation during dry/ pre-monsoon and post-monsoon also have declined by 0.75 and 1.73 mm/year respectively (Figures 2, 3, and 4).

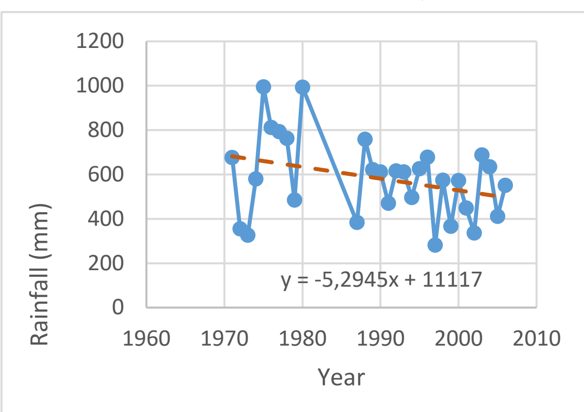

Figure 2: Rainfall in June-September as recorded by Gam-Shreenagar Station

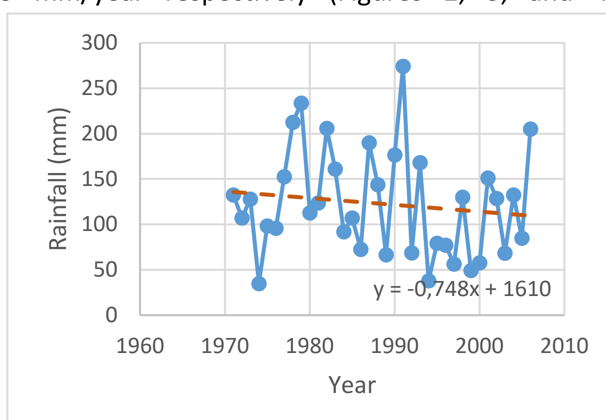

Figure 3: Rainfall in March-May as recorded by Rara Station

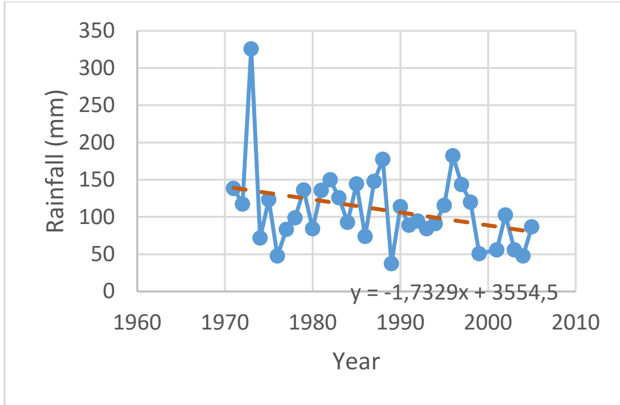

Figure 4: Rainfall in October-February as recorded by Gam-Shreenagar Station

Temperature Increase

Average temperature has increased at an annual rate of 0.06◦C and at a higher rate (0.08◦C) in the Himalayas. Likewise, warming patterns have been most pronounced at higher elevation and also in the western part of the country (Synnott, 2012).

Impact of Climate Change on Water Sources

A large number of local water sources that were supplying water for household consumption as well as irrigation have declined due to erratic nature of rainfall pattern. According to national newspapers, nearly 200 water sources have dried up alone in Mugu district of the basin in 2016. Similarly, water levels of Rara Lake and the Karnali River have also dipped. It has largely increased women workloads as women need to walk for longer distances in search of drinking and household water use.

The findings of a scientific study conducted by Nepali and Spanish team of researchers have claimed that water in the Himalayas is diminishing. The research conducted in the Dudh Koshi basin showed that 30% of water of basin will be decreased by 2100. Further, the study forecasted that more than 50% volume of snow fed rivers will be decreased due to shrinking of volume of ice in the Himalayas by 2100. According to the study, trend of snow melting and expansion of Glacier Lake size is continuing, but eventually snow occurring tendency has declined and rapid melting inclination has increased. The analysis therefore shows that climate change has affected Nepal with less monsoonal rain across the high mountains and more monsoonal rains along the southern hills.

Impact of Water Sources on Agriculture and Medicinal Herbs Production

The increased variability in monsoon onset has delayed the main cropping season and directly affected major agriculture crops productivity. The long dry post-monsoon has prolonged water unavailability and winter agriculture crop cultivation in turn affected their yield and production. Similarly, livestock production has also reduced due to adverse impact on grazing resources. As per the report of local people, prolonged winter drought of 2016 has significantly reduced (about 75%) apple production in its` pocket areas of the basin. Its adverse impact on livelihood of the local people is high. Studies have indicated that such phenomenon may emerge in the other river systems' basins also.

Likewise, production of commonly available high value herbs such as Guchi chayu (*Morchella esculenta* Pers), Yarshagumba (*Ophiocordyceps sinensis* Sacc), Ban lasun (*Allium wallichi* L.) and Katuki (*Neopicrorhiza scrophulariiflora*) has also decreased in its pocket areas due to increased erratic pattern of precipitation in the basin. The overall impact of climate change on water availability and in turn agriculture and medicinal herbs production has affected on food availability and sources of cash earning to the people in the basin.

CONCLUSIONS

Unpredictable rainfall pattern and monsoon timing have adversely affected water availability and in turn livelihoods of the people in the area. It has also increased workload to women. The basin lacks research based information and database for policy formulation and project/programme implementation on climate change adaptation to make it more resilient. Hence, the study may somehow serve to fulfill the gap.

Key words: Climate change, livelihood, impact, water, resilient

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1.6.O

SOIL ACTIVE CARBON AND ORGANIC CARBON STOCK AS AFFECTED BY CULTIVATION AND IRRIGATION WITH TREATED WASTE WATER

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INTRODUCTION

Forest and agricultural soils play a vital role in carbon cycling and storage, contributing greatly to the maximum carbon sink through photosynthesis (Parras-Alcantara *et al*., 2013). However change in land use, management practices and on farm inputs affect stocks and storage of organic carbon in such soils (Sardiana *et al.*, 2014). One essential input is irrigation, however, due to increased drought incidents, availability of high quality water for irrigation in South Africa is becoming increasingly scarce (IPCC, 2014). Some farmers in heavily affected regions have resorted to using wastewater for irrigation. Improvement of soil physical structure and soil microbial activity has been observed in soil irrigated with treated wastewater by AL-Jaboobi *et al.* (2014). However, some studies report that wastewater may negatively affect the soil (Hasan *et al*., 2014) as a result of the different sources such as industrial developments, hospitals and some wastewater sources which may supply newly synthesized substances that could destroy the structure of the soil, degrade microbes (Mouta *et al*., 2008) or interfere with the nutrient flow within soils. Therefore this study was undertaken to investigate the distribution of soil active carbon and organic carbon stocks in the top two depths of cultivated soils irrigated with treated wastewater, when compared to virgin and fallowed soils at the University of Limpopo (UL) experimental farm.

SUMMARY OF METHODS

The study was conducted at UL experimental farm (23°50'80''S; 29°41'29''E), located in Polokwane Municipality, Limpopo Province, South Africa. The farm is found in a semi-arid region with approximately 80 to 87% of the annual rainfalls of 495 mm occurring mostly in the summer months of October to March. Minimum temperatures for the study duration 9.0°C and 16.7°C from April to September 2016 respectively. The maximum temperature ranged between 26.7 and 32.6°C from April to September 2016. Soils of these locations are mainly deep red, sandy clay loam soils classified under South African Binomial Soil Classification System as Hutton (Soil Classification Working Group, 1991), or Rhodic Ferralsol (FAO, 1989) soil form, with an average of ±35% clay and ±65% sand content. Three 15 ha fields were identified: cultivated field (CF) which is currently irrigated with treated waste-water; fallowed field (FF) in its fifth year of fallowing after three year cultivation and irrigation with treated waste-water and virgin field (VF). Fields were divided into 15 equal plots, where soil samples were collected at two top depths (0-20 cm and 20-40 cm) and analysed for particle size distribution, bulk density (BD), soil active carbon (SAC) and soil organic carbon (SOC). The first depth (0-20 cm) of VF was treated as a reference point. Soil organic carbon stocks (SOCS) were calculated for all samples using the equation (Komatsuzaki and Syaib, 2010):

$$\text{SOCS (g. m}^{-2}\text{)} = \text{BD} \times \text{SOC} \times \text{DP} \quad (1)$$

Where BD = soil bulk density ($g.cm^{-3}$); SOC = soil organic carbon content (%); DP = soil depth (cm). Data were subjected to factorial analysis of variance (ANOVA) using Stata 12 software (StataCorp, 2011). A Pearson's correlation was run to assess the relationship among the dependent variables.

RESULTS

Field × depth interaction was highly significant on SOCS and BD with both contributing 2% in total treatment variation (TTV), whereas it was not significant to SAC, SOC, clay and sand. Field was highly significant on SOCS, SOC, BD, clay and sand with 96%, 100%, 95%, 52% and 82% in TTV respectively. Depth was significant on SOCS, SOC, BD and clay but not significant on SAC and sand (Table 1). Relative to the reference point, which was VF, SAC for CF decreased by 39% but increased by 40% in FF (Table 2). Relative to the reference point, SOCS of CF and FF in the first depth decreased with 99.5% and 99% respectively (Table 3). An increase of 33% for SOCS was observed in the second depth of VF. However, for the same depth, 99.5% and 99% reductions were observed in CF and FF respectively. According to Pearson's correlation test, there was a weak positive correlation between SAC and SOCS in both VF($r = 0.02$) and CF($r = 0.32$) (Tables 4 and 5), whereas, for FF a weak negative correlation ($r = –0.16$) was observed (Table 6). Moderate positive correlations were observed between SOCS and SOC in CF and FF while strong positive correlations ($r = 0.99$, $r = 0.84$, and $r= 0.95$) were observed between SOCS and BD in VF, CF and FF respectively.

CONCLUSIONS

High BD and SOC in VF resulted in high soil carbon stocks for the two fields. The results proved a high decline in carbon stocks with cultivation and irrigation with treated waste water. However, fallowing was able to ameliorate the effects as indicated by the increase in both SAC and SOCS in FF.

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Table1. Partitioning sources of variation of soil active carbon (SAC), soil organic carbon stocks (SOCS), soil organic carbon (SOC), bulk density (BD) and particle size, of the three fields of different management systems at University of Limpopo.

SOURCE	DF	SAC		SOCS		SOC		BD		CLAY		SAND	
		MSS	TTV%	MSS	TTV%	MSS	TTV%	MSS	TTV%	MSS	TTV%	MSS	TTV%
REP	14	133830.71	16^{ns}	583.22	0^{ns}	0.20	0^{ns}	0.18	0	299.23	17***	603.67	10***
FIELD (A)	2	509919.98	63**	247870.02	96***	6211.31	100***	72.03	95***	937.95	52***	4839.34	82***
DEPTH (B)	1	142.00	0^{ns}	5042.36	2**	0.80	0**	1.78	2**	336.40	19**	143.89	2^{ns}
A X B	2	40598.94	5^{ns}	5089.99	2***	0.02	0^{ns}	1.68	2***	140.40	8^{ns}	177.89	3^{ns}
ERROR	70	130192.77	16	621.13	0	0.18	0	0.20	0	74.99	4	102.16	2
TOTAL	89	814684.40	100	259206.71	100	6212.51	100	75.86	100	1788.97	100	5866.95	100

TTV% = Total treatment variation = (MSS/TOTAL) × 100; ***highly significant ($p \leq 0.01$); **significant ($p \leq 0.05$); ^{ns}not significant

Table2. Soil active carbon (mg.kg^{-1}) in cultivated and fallowed fields relative to that of virgin field.

Fields	Variable	R.I. (%)
VF	330.20	–
CF	201.41	-39
FF	463.14	40

VF = Virgin Field, CF = Cultivated Field, FF = Fallowed Field;
Relative impact = R.I. (%) = [(Treatment/Standard) – 1] x 100

Table 3. Soil organic carbon stocks (g.m^{-2}) in cultivated and fallowed fields relative to that of virgin field in two depths.

Fields	0-20 cm		20-40 cm	
	SOCS	R.I. (%)	SOCS	R.I. (%)
VF	135.91	–	180.96	33
CF	0.7	–99.5	0.65	–99.5
FF	1.36	–99	1.27	–99

VF = Virgin Field, CF = Cultivated Field, FF = Fallowed Field;
elative impact = R.I. (%) = [(Treatment/Standard) – 1] x 100

Table 4. Pearson's correlation matrix showing relationships among soil active carbon, variables contributing to soil organic carbon stocks and particle size distribution for virgin field.

	SAC	SOCS	SOC	BD	CLAY	SAND
Soil active carbon (SAC)	1.0000					
Soil organic carbon stocks (SOCS)	0.0217	1.0000				
Soil organic carbon (SOC)	–0.1890	0.2625	1.0000			
Bulk density (BD)	0.0314	0.9987	0.2157	1.0000		
CLAY	–0.0528	0.1714	0.1462	0.1686	1.0000	
SAND	0.1440	–0.1112	–0.1769	–0.1057	–0.9365	1.0000

Table 5. Pearson's correlation matrix showing relationships among soil active carbon, variables contributing to soil organic carbon stocks and particle size distribution for cultivated field.

	SAC	SOCS	SOC	BD	CLAY	SAND
Soil active carbon (SAC)	1.0000					
Soil organic carbon stocks (SOCS)	0.3277	1.0000				
Soil organic carbon (SOC)	0.2156	0.629	1.0000			
Bulk density (BD)	0.2629	0.8358	0.1351	1.0000		
CLAY	0.0298	–0.0244	0.1025	–0.041	1.0000	
SAND	–0.1815	0.4103	0.1489	0.4201	-0.3918	1.0000

Table 6. Pearson's correlation matrix showing relationships among soil active carbon, variables contributing to soil organic carbon stocks and particle size distribution for fallowed field.

	SAC	SOCS	SOC	BD	CLAY	SAND
Soil active carbon (SAC)	1.0000					
Soil organic carbon stocks (SOCS)	–0.1556	1.0000				
Soil organic carbon (SOC)	–0.047	0.451	1.0000			
Bulk density (BD)	–0.1827	0.9467	0.1589	1.0000		
CLAY	–0.1044	–0.2609	–0.3072	–0.1456	1.0000	
SAND	0.1108	0.2538	0.2906	0.1516	-0.936	1.0000

1.7.O

CHANGES IN RAINFALL PATTERNS AND IMPACTS ON VEGETATION DYNAMICS, STREAMFLOW AND SEDIMENT YIELD IN A SMALL WATERSHED IN THE MURIAÉ RIVER BASIN, SOUTHEAST BRAZIL

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INTRODUCTION

Due to a worldwide increasing food and biofuel production, the degradation of natural resources is expected to increase, too. In addition, it has been pointed out that higher inter-annual variability of rainfall and intensities will lead to greater rates of erosion because of changes in the vegetation growth dynamics and soil cover. However, there is a lack of studies about the expected impacts of climate change on sediment loads in rivers and streams, particularly in Brazil.

For instances, the Muriaé River basin in SE Brazil has been experiencing an increasing pressure on water resources, due to the population growth of the Rio de Janeiro city area connected with the growth of the industrial and agricultural sector. This leads to water scarcity, riverine forest degradation, soil erosion, land degradation and water quality problems in the river among other impacts. Additionally the region has been suffering from seasonal rainfall variations leading to extreme events such as droughts, floods and landslides. Moreover, the study area belongs to the Brazilian Atlantic Forest Biome, a biodiversity hotspot, highly threatened by climate change and degradation. Therefore, it is crucial to understand how climate affects the interaction between the timing and frequency of rainfall events, hydrological processes, vegetation growth, soil cover and soil erosion. In this context, a fully distributed physically based hydrological and solute transport model can contribute to a better understanding of spatial-temporal process dynamics, at the watershed scale. The Modified Universal Soil Loss Equation – MUSLE (William, 1975) is implemented in the model JAMS/J2K-S to predict the sediment yield of individual storm events, taking into account the antecedent moisture conditions as well as rainfall energy. Since the MUSLE model was produced for specific conditions, in the USA, it is advisable that the equation must be adapted for the local climate and hydrological conditions, namely the empirical factors such as crop management factor (C) and the coefficients “a” and “b” (Chaves, 1996). However, there are few studies about the calibration of such empirical factors in Brasil and its application without calibration for other regions has shown already huge errors (Sadeghi et al. 2014).

Thus, the objectives of this study were to investigate the mentioned empirical factors for local conditions and to assess the impacts of changes in rainfall patterns on vegetation vitality (NDVI), hydrological response and sediment yield, in a small watershed within the Muriaé River basin – the Santa Maria micro watershed.

METHODS

The Santa Maria watershed has an area of 13.56 km^2, it is located in the São José de Ubá Municipality – Rio de Janeiro State, and it is covered by 7% agriculture, 10% forest and 80% pasture for livestock production. Regarding the agricultural fraction, the area is occupied by small farms whose main activity is irrigated tomato in winter (dry season).

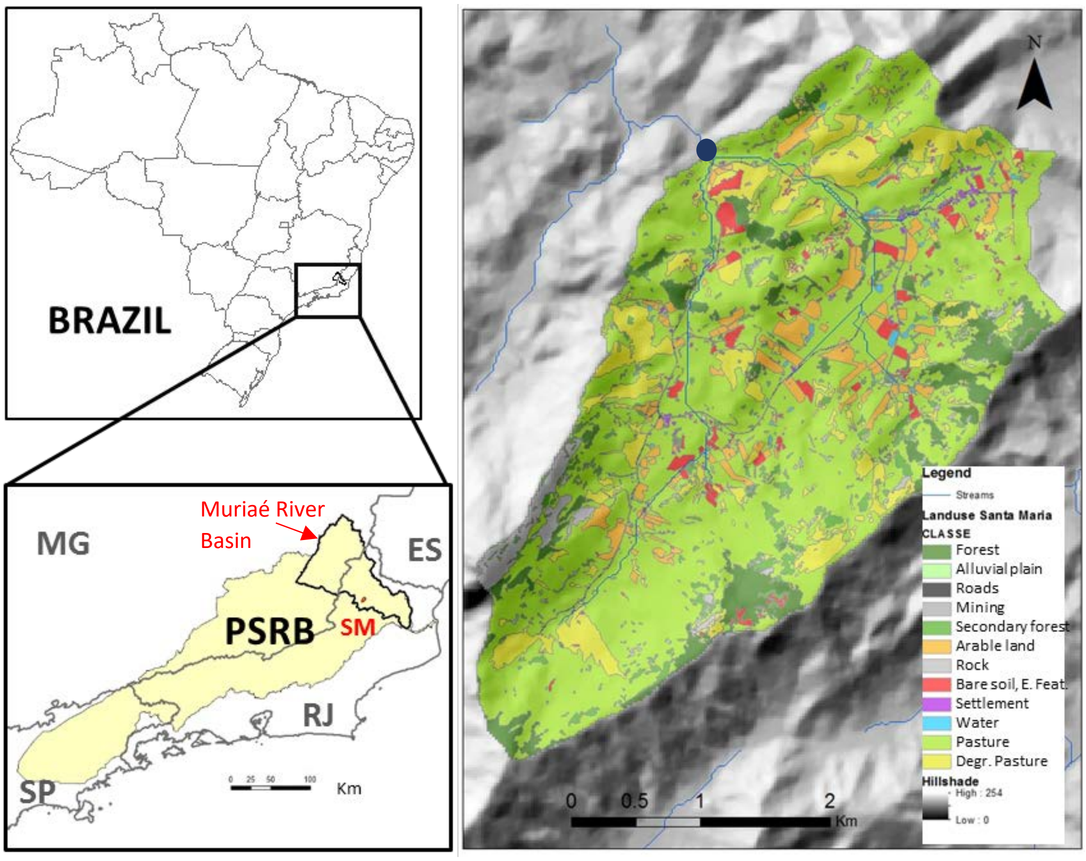

Figure 1. Map of the study area with automatic hydro-climatic and sedimentological station (blue circle - outlet of the Santa Maria micro watershed (SM); ES – Espírito Santo State; MG – Minas Gerais State; RJ – Rio de Janeiro State; SP – São Paulo State; PSRB – Paraíba do Sul River Basin; E. Feat. – Erosion Features (rills, gullies); Source of the Landuse Map:

According to the Köppen Classification, the climate of the study area is Aw (tropical with dry winter). From 2008-2011 the micro watershed was equipped with an automatic hydro-climatic and sedimentological station, which stems from a work conducted by EMBRAPA (Agricultural Research Corporation), SEAPEC - PRR (State Secretariat of Agriculture and Livestock – Rio Rural Project) and INEA (Environmental Agency of Rio de Janeiro State) (Prado et al. 2012). Thus, the analysis presented in this study were done based on observed daily data from the station and a 31-month time series of 16-day MODIS 250 m NDVI data (MOD13Q1). The mean NDVI value (56 in total) for the Santa Maria micro watershed was calculated, and then plotted against the cumulative rainfall, between two NDVI values. The normalized difference vegetation index (NDVI) is one of the main indices used for vegetation monitoring and assessment and it has been used to estimate the crop management factor C (Durigon et al., 2014). Moreover, it is crucial to take into account the soil management practices in the study area and therefore, a crop rotation scheme for pasture and arable land (with the moments of tillage, planting and harvest operations) was generated for the Santa Maria micro watershed.

RESULTS

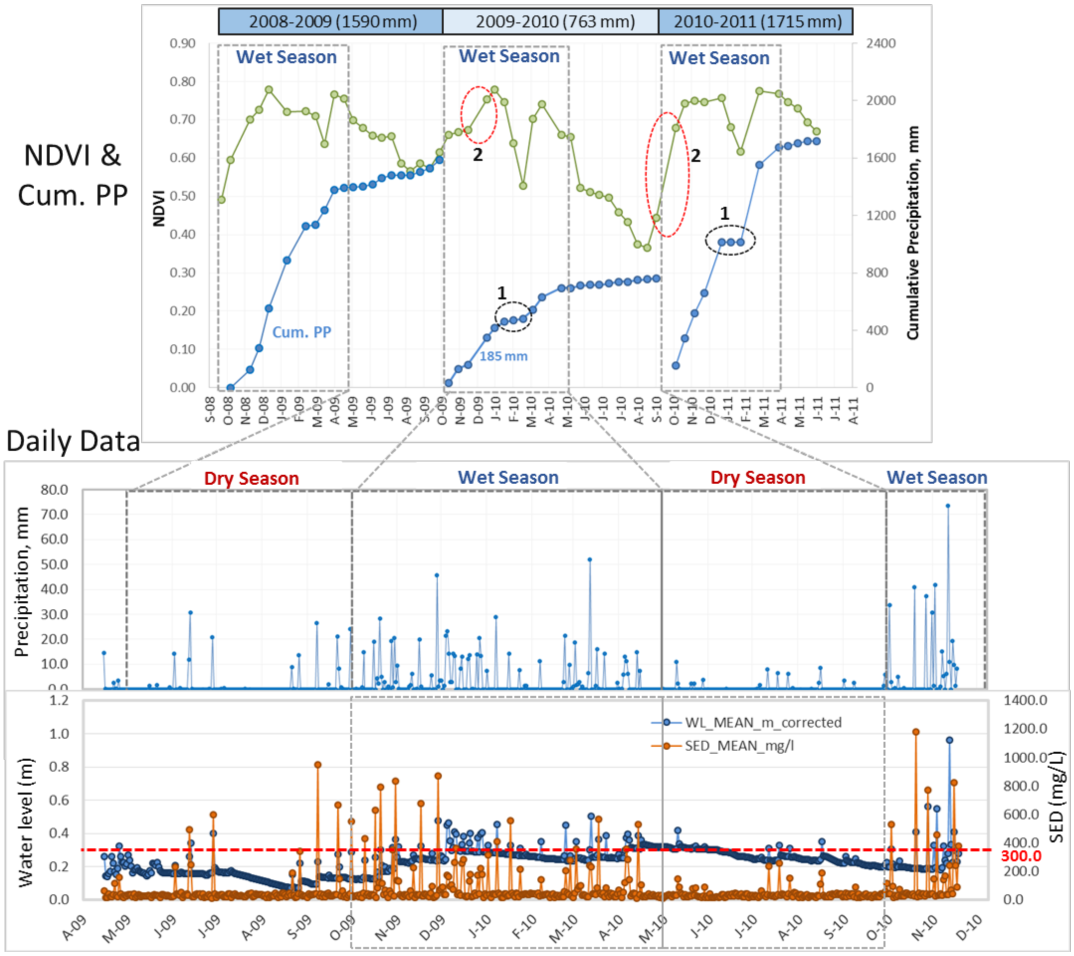

Figure 2. a) NDVI and cumulative rainfall for 3 Hydrological years: 2008-2009, 2009-2010 and 2010-2011, for the Santa Maria watershed, time step - 16 to 32 days, 1 – low cumulative rainfall and correspondent drop on the NDVI, 2 – high NDVI increase; b) daily water level and suspended sediment concentrations (mg/L) from April 2009 to November 2010.

The Figure 2a shows the evolution in time of the NDVI (every 16-32 days) and the correspondent cumulative rainfall. The cumulative rainfall starts in the beginning of each hydrological year (October to September). It is possible to identify differences between the three hydrological years. The hydrological year 2009-2010 is a dry year with a total rainfall of 763 mm, below the long-term mean annual rainfall, and the hydrological year 2008-2009 and 2010-2011 are both wet years, with rainfall above the long-term mean annual rainfall. We found strong seasonal oscillations in the vegetation-growing season (October to March) over the study area, with maximum NDVI observed in December-January. NDVI respond to the rainfall in a different way depending on the timing of occurrence, season, and antecedent conditions. At the end of the dry season (less than 70 mm of rainfall from beginning of May to end of September) of the hydrological year 2009-2010, the NDVI registered the lowest NDVI value comparing with the hydrological year 2008-2009 and 2010-2011, being lower than 0.4. Therefore, the 152.4 mm of cumulative rainfall in the beginning of the hydrological year 2010-2011, and first streamflow events, generated suspended sediment concentrations in the river higher than 300.0 mg/L (Figure 2b), which is considered very high concentrations according to Lima et al. (2003). Very high suspended sediment concentrations were also observed in the beginning of the hydrological year 2009-2010.

CONCLUSIONS

The results from the analysis conducted for the Santa Maria micro watershed are very important to improve the understanding about the link between climate, vegetation, streamflow and sediment yield, at a micro watershed scale, to a better representation of the processes in hydrological and solute transport models. The increasing amounts and frequency of intense rainfall events have great impacts on streamflow generation, and consequently on sediment yield, but the timing of occurrence of those rainfall events, play an important role triggering erosion and therefore, should be consider for adaptation measures and soil protection. Thus, the initial period of the wet season is critical for erosion protection, since the soil is more vulnerable due to low vegetation density.

For instance, Oliveira et al. (2013b) observed that in Brazil high values of annual rainfall do not necessarily produce higher values of erosivity due to variation in rainfall intensity. Also, Carvalho et al. (2014) looking to NDVI images at a watershed located in the south of the Rio de Janeiro State, Brazil, observed the greatest soil loss in October, due to the combination of higher values of rainfall erosivity and lower values of soil cover.

The results also suggest that MODIS NDVI datasets could be used to provide detailed vegetation status to represent a dynamic crop management factor (C) for the different phenological states of the cultures in hydrological and solute transport models.

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1.8.O

EVALUATING THE ROLE OF SOIL AND WATER CONSERVATION ON 'CARBON SEQUESTRATION' FOR REDUCING THE CARBON DIOXIDE (CO_2) EMISSION INTO THE ATMOSPHERE – A CASE STUDY FROM SOUTHERN GUAM

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BACKGROUND INFORMATION

Guam is composed primarily of limestone rock to the north and red volcanic clay soils in the south. The red volcanic soils in the south are severely eroded and locally are referred to as badlands. These barren sites are exposed to overland flow, wind and rain causing severe soil erosion and producing massive amount of sedimentation in the downstream rivers and shorelines of southern Guam. This massive siltation not only damage the marine environment downstream and effect fisheries in the south but also effect fresh flow of rivers which are the main sources of fresh water for residential consumption in southern Guam. Furthermore, soil erosion is the cause of soil fertility depletion, it damages soil structure, and it reduces the 'effective' rooting depth (Lal, 2003). These degradation processes have also produced acidic soils which are high in iron and aluminum oxides that remain in the soil composition therefore making it difficult to sustain crop productivity on these soils.

In addition, the adverse impacts of accelerated soil erosion on carbon (C) dynamics and emission of carbon dioxide (CO_2) into the atmosphere also deserve to be studied (Lal, 2003).

Transfer and redistribution of soil organic carbon (SOC) caused by soil erosion may also have an impact on the overall global C budget (Lal, 2003). In addition, soil disturbances by excessive tillage practices not only are the cause of farmland soil loss by runoff water but also it is a primary cause of loss of soil organic carbon (SOC) (John M. Baker, et all, 2007). Hence, soil organic carbon sequestration may be achieved by changing farming practices from conventional tilling into less intensive farming techniques such as no-tillage and/or other conservation tillage practices. Furthermore, soils are important and active pool of organic carbon thus they play major role in global carbon cycle (Lal, 1997). On the other hand, soil degradation is in fact the major cause of reducing or losing of soil functions hence minimizing the capability of soils for crop production, and poses a threat to agricultural sustainability worldwide (Chen Jie, 2002). Furthermore, soil degradation is also blamed for sedimentation in the waterways and water bodies, watershed behavior, and for global warming (Chen Jie, 2002). Soil erosion is also associated with changes in natural ecosystem and natural habitats leading to loss of genetic stock and biodiversity on the global scale (Chen Jie, 2002). As reported by Oldeman et al. (1991), the major factor of soil degradation is soil erosion and soil loss caused by agricultural mismanagement, overgrazing and deforestation.

The challenge facing soil and agricultural scientists is therefore to develop restoration strategies to improve the current conditions on these soils, and avoid further environmental and financial constraints to the ecosystem that not only has already affected water resources but will have a major impact on already unsustainable agricultural productivity that mostly depend on these lands. Furthermore, the extent of soil erosion, and its impact on soil carbon dynamics, and the potential of erosion management (conservation) for sequestering the 'C' by mitigating the accelerated sedimentation (Lal, 2003) are among the challenges of soil and environmental scientists all around the world.

Toward this end, we have designed an integrated conservation farming technique to study the effect of different farming management on soil quality and productivity while evaluating the techniques on controlling soil erosion and reducing sedimentation. The aforementioned management strategy also includes biochar application for soil quality improvement and for increasing the carbon storage capacity of the soils under study. The later practice (biochar application) is also expected to reduce the CO_2 emission into the atmosphere via carbon sequestration process.

Biochar is a solid substance made from the heating of biomass (e.g., coconut husk, Vetiver grass, tree branches) at high temperatures and low levels of oxygen called 'pyrolysis' (International Biochar Initiative (IBI, 2017). The biochar is basically composed of carbon with some ash that contains N, P, K. Biochar is also known to improve the fertility of soils by raising the soil cation exchange capacity (CEC) while maintaining the carbon storage capacity of the soil. When the CEC is high, the soil is able to hold more cation nutrients, which are otherwise lost via leaching. It is reported (Butnan, et.al. 2015) that liming is also an important co-beneficial aspects of biochar application on similar soils in the tropics. Biochar application effectively reduces the negative effects of Aluminum toxicity and other elements such as Mn (Butnan, et.al. 2015). When added to the soil, biochar may enhance plant growth and improve crop yields and increases production thus ensuring agricultural sustainability especially where soils are depleted with nutrients and have limited organic matter and are suffering from insufficient water resources (US BI, 2017). As reported by the US BI (2017), application of 'biochar' to soils with low carbon content, can increase their potential for higher yield and better crop and improve their overall productivity considerably. The international research effort on 'biochar' continues to indicate that 'biochar' production and its use has the potential to reduce the effect of greenhouse gases (i.e., CO_2) on global warming by holding carbon in soil instead of releasing it into the atmosphere (Small Planet Institute, 2017).

Biochar is also very stable and by adding it to the soils is tantamount to carbon sequestration and helps reduce carbon dioxide emission which is linked to global warming. For this reason, this study is therefore in line with the intention for reducing CO_2 emission as well as conserving energy.

METHODOLOGY

This study include trial plots where growth performance of selected crops (Maize) is being evaluated based on farming method that are practiced on field plots designed for this purpose. The specific farming techniques evaluated in this project are as the following:

- **a**) No-Till (NT) or zero tillage
- b) Reduced tillage (RT)
- c) Conventional Tillage (CT)
- d) Conventional Tillage with Biochar application (CT/BC)

These regimes represent a wide range of practices that are being evaluated as conservation and restoration techniques. These practices are also evaluated based on the amount of crop residue that is left on the soil surface following each harvest. In this experiment, the crop resides on the no-till (NT) plots are bush cut and left on the surface following the harvest. The crop residue is gradually decayed and broken down as organic mulch covering the rows hence protecting the soil surface from the raindrop impact hence eliminating water erosion. The crop residue from the reduced tillage (RT) plots is also treated the same way until just before the next planting phase. At each planting phase, the remaining residues on reduced till (RT) plots are incorporated into the soil surface for bed preparation. Crop residue from both conventional tillage (CT) and conventional with biochar (CT/BC) are removed to the ground and soil surface is left completely bare and exposed to weather conditions until next planting. In addition, soil quality improvement is being evaluated based on the farming techniques practiced in this project. In this regard, specific attention is being given to the plots where biochar is applied. Soil quality improvement is being evaluated based on the amount of biochar added to the soil composition.

In summary, this research experiment was aimed on developing an efficient and feasible conservation program that will improve the quality of the red volcanic soils in southern Guam by using soil enhancer such as biochar, while preserving the natural resources via conservation farming techniques as described above. Also, soil conditions and nutrient deficiencies will be evaluated by running a comprehensive soil testing prior to the application of biochar.

RESULTS

The results from the study are being evaluated based on the following parameters; Soil carbon content; Crop yield; and Carbon dioxide emission from the soils under different farming practices.

Up-to-date (2015/2016) data pertaining to the carbon content of the aforementioned farming practices (NT, RT, and CT, CT/BC) are shown in Figures 1. As shown (Fig. 1), the carbon content was the highest under the No-tillage (NT) practice. It is believed that high carbon content under No-till plot was due to ‘no disturbances’ to the soil surface on these plots during the study period. On the reduced till (RT) plots, the percent carbon content also remained high next to the No-till plots mainly due to the reduced disturbances as compared to conventional tillage practices. On the other hand, as it is shown (Fig. 1), the percent carbon content in the conventional tilled (CT) plots were the lowest for all sampling events while the carbon content of conventional tilled with biochar (CT/BC) application was somewhat higher than conventionally tilled soils mainly due to the carbon effect from the ‘biochar’. Same trend have been observed in the previous years (Golabi, 2015) with NT showing the highest amount of carbon content compared to the other treatments under study. This study showed that the soil organic matters as well as soil organic carbon content are all affected by the tillage treatments applied on these soils (data not shown).

Also, as indicated, the carbon content of the soil was considerably lower in the lower depths regardless of the tillage treatment however; the overall carbon content of the soil under CT is generally lower due to continuous disturbances on the soil surface and within the tillage depth (Golabi, 2015). This could be due to oxidation of soil carbon as the result of exposure to the air following each tillage practice.

Furthermore, the data illustrate that the carbon content of the soil near the surface is higher than 2% for all treatment regardless of the tillage practices. This could be due to higher organic content of the soil (data not shown) as a consequence of plant residues decaying process near the soil surface. On the other hand, the carbon content of the soil is lower at depths below the 8 inch for all treatments. This may indicate that the amount of carbon loss at the deeper depth is the lowest due to less carbon content and more carbon stability conditions at the lower soil profile (Golabi, 2015).

As indicated earlier, applying 'biochar' improved the crop yield as compared to the other treatments (Figure 2). Although, the yield differences are not different statistically, the apparent yield increases due to the 'biochar' application are noticeable (Fig. 2). Crop yield under the no-till (NT) plots have shown to be lower than all other treatments under study. This might be due to compaction hence low aeration. The low yield obtained from the no-till soil plots are also due to low nutrient content and high aluminum saturation (data not shown) of these volcanic soils of southern Guam. This phenomenon is being observed during the last 14 years of conservation tillage practices on these soils, indicating that the no-tillage practices are not adoptable on these volcanic soils of southern Guam. These soils are very high in clay (%60) low pH (5.2 and lower), and are saturated with aluminum (data not shown), and are voided with earthworms and any other microorganisms living in the soil matrix. As shown (Fig. 2) the crop yields from the conventional tilled plots are generally higher than the yield obtained from the no-tilled plots mainly due the aforementioned factors.

Referring to the CO_2 emission, the amount of carbon dioxide released from the 'biochar' treated plots was lower during the active plant growth as compared to the CO_2 emission before planting. This showed an opposite trend as compared to the other treatments where the CO_2 released was higher during the active growing phase than before planting. This could be due to the boost of microbial activities caused by the initial biochar application before planting. However, during the active plant growth, the organic carbon and/or the CO_2 generated by microbial activities might have been tied up as the 'biochar' was incorporated further into the soil during the active growing period. Less carbon dioxide emission from the 'biochar' applied plots could possibly be due to soil carbon being tied up to the 'biochar' as the 'biochar' becomes more activated during the active growing phase.

IMPACTS

This study is intended to evaluate the effect of conservation tillage practices on soil quality improvement on these severely eroded and acidic soils of southern Guam. The study also evaluates the impact of 'biochar' application not only as soil amendment but also as a technique for storing soil carbon content known as 'carbon sequestration'. The conservation practices as evaluated here are also expected to have a major impact on the island's water resources such as fishing and marine life due to soil erosion control techniques implemented in this study. The results of this study also point to the fact that agricultural sustainability thus, food security can be attainable in Guam and other neighboring islands in the western Pacific, provided that the aforementioned techniques such as reduced till and/or conventional tillage practices are combined with 'biochar' application are implemented. Furthermore, the application of 'biochar' as soil amendment is also expected to preserve soil organic carbon thus is reducing the amount of CO_2 release due to the oxidation process. On the other hand the no-tillage practices may prove to be beneficial in northern Guam where the soils are formed from porous lime stone with better nutrient availability and higher porosity. On the other hand, no-tillage farming as practiced in many other regions of the world is not recommended for the soils of southern Guam.

The preliminary results from the study have so far indicated that tillage systems may affect the soil carbon storage capacity and/or the amount of carbon loss due to the level of disturbances and/or amount of residue removal from the soil surface. The study also showed that the conservation farming practices such as no-tillage although not feasible with respect to crop production; they can potentially increase carbon sequestration in the soil as disturbances within the plow layers are reduced to minimum and/or to zero.

The results of this ongoing experiment will provide knowledge based techniques for sustainable agricultural practices applicable to the island's climate conditions and resources available to farmers in this and other islands of Micronesia. The result of this study will also contribute to the overall scientific efforts in understanding the role of agriculture in soil carbon dynamic, and the ways in which this may reduce atmospheric carbon dioxide if the aforementioned techniques are further studied and implemented to a much larger extend.

PUBLIC EDUCATION/OUTREACH

This study will provide information pertaining to soil degradation as relates to the local conditions of the island's tropical climate following the removal of crop residue from the soil surface after the harvest. The results of the study will also provide information pertaining to carbon loss due to disturbances that occur during the tilling processes prior to planting. The study will also provide information pertaining to carbon storage and carbon sequestration potential of the soils under those conservation farming practices that are only adoptable to the local conditions of the island's tropical climate. Application of 'biochar' as soil amendment to these severely eroded soils of southern Guam may prove to be a viable option toward sustainable agricultural practices in this and other islands of Micronesia.

Keywords: Conservation tillage, residue removal, Biochar, Compost, Degraded soils, Badlands, Volcanic clay soils, Pyrolysis.

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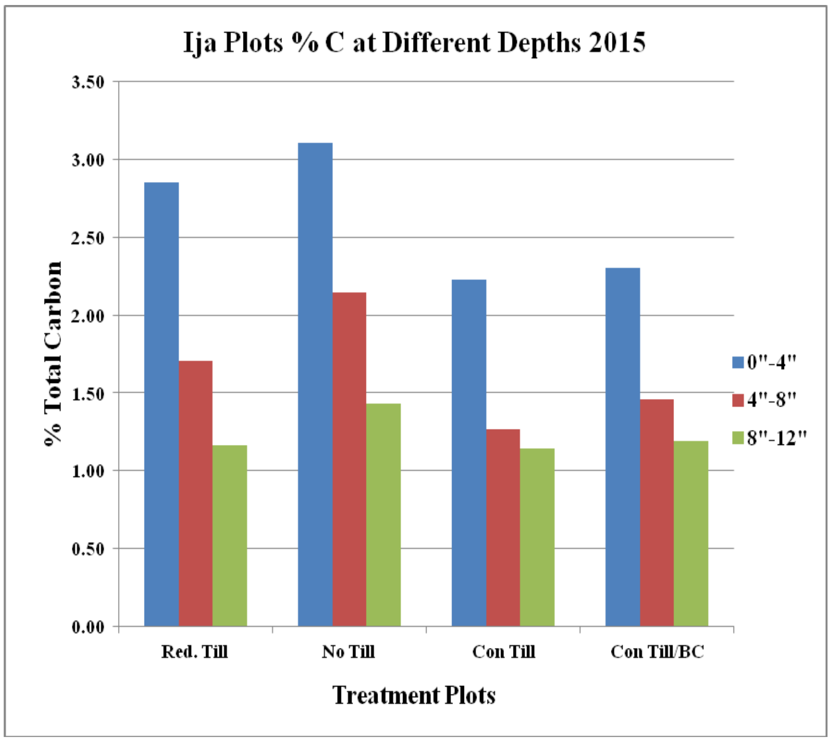

Figure 1.: Showing percent Carbon conten of the soil at diferent depth under diferent treatment (conservation tillage practices including 'biochar' treatment).

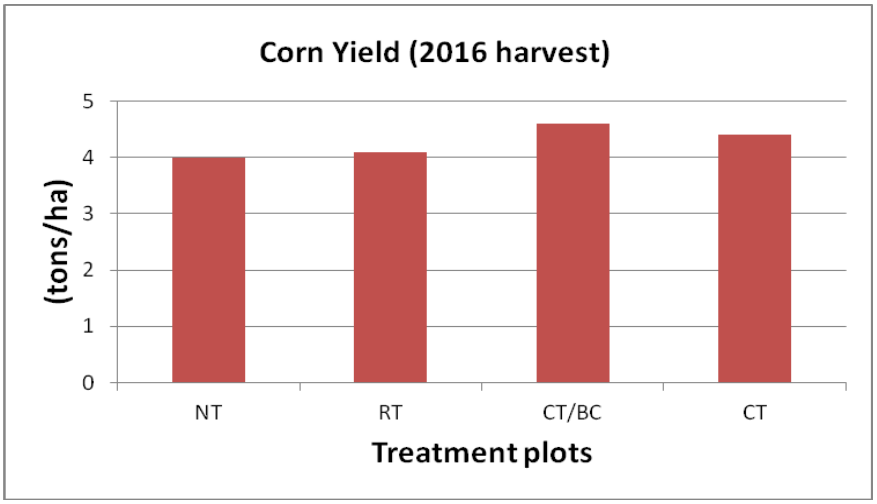

Figure 2.: Showing the corn yield for 2016 harvest under diferent conservation treatments (no-tillage (NT), reduced tillage (RT), conventional tillage with 'biochar' application (CT/BC) and conventional tillage treatment (CT)).

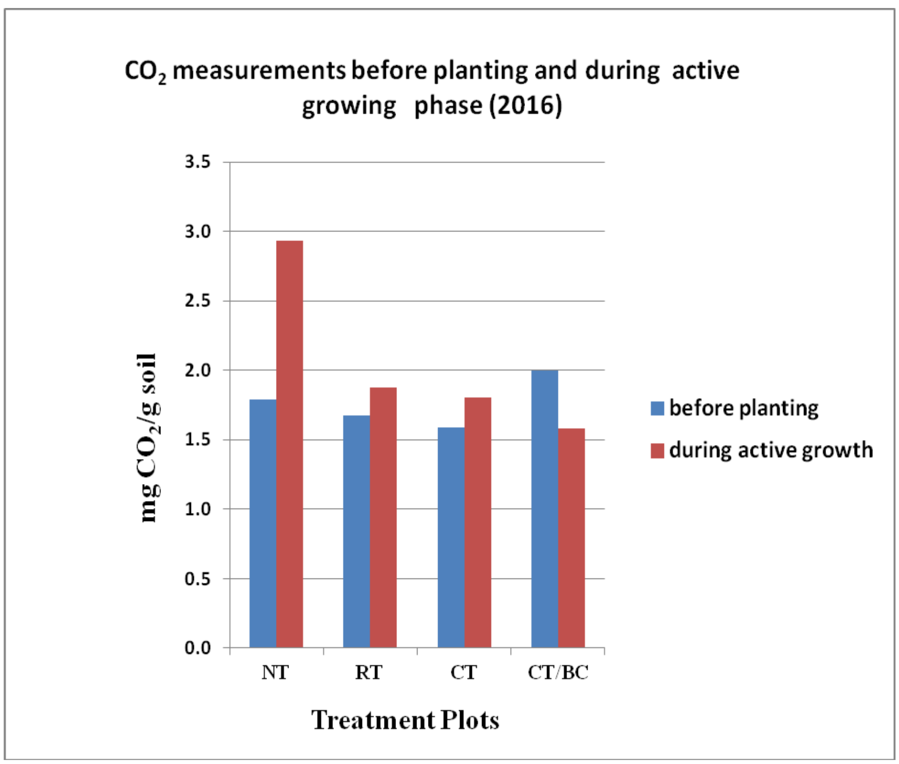

Figure 3.: Showing the amount of CO_2 emision from the treatment plots before and during the active crop growth (2016 growing season).

1.10.O

ASSESSMENT OF CROP DROUGHT TOLERANCE IN BIOCHAR AMENDED SOILS

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ABSTRACT

This research evaluated crop growth response in a biochar amended southern Arizona Aridisol to periodic water stress under greenhouse and field conditions. Three concentrations of biochar (4.3%, 8.8%, and 16.4% by mass) in soil and a soil only control were evaluated in greenhouse experiments with six common or potential southern Arizona crops (alfalfa, wheat, cotton, grain sorghum, sweet sorghum, and switchgrass). Alfalfa height and biomass and sweet sorghum and grain sorghum biomass increased significantly with biochar concentration. There was no significant difference in cotton biomass or height among treatments. Moisture retention curves for biochar and soil are presented and used to analyze crop growth experiments. Sweet sorghum biomass production was evaluated in a field experiment conducted at the University of Arizona Red Rock Agricultural Center (RRAC) with biochar incorporated in the top 20 cm of soil in one treatment. An irrigation simulation was performed using the FAO-CROPWAT computer program to calculate crop water requirements and irrigation requirements based on soil type, plus weather and crop data. The estimated water requirements results in an ETc of 1,033.4 mm and an irrigation requirement of 879.6 mm. According to these results, a net irrigation schedule of 928.4 mm and a gross irrigation schedule of 1326.8 mm were obtained. A total of 555.5 mm were applied to forage sorghum in the field experiment, with a difference of 771.3 mm respect the gross estimated irrigation schedule using CROPWAT.

INTRODUCTION

Biochar is porous charcoal that is produced by heating plant biomass without oxygen in a process known as pyrolysis. Biochar production techniques vary in temperature, time of heating, and chemistry. Fast pyrolysis, which takes a few seconds, is generally used to convert organic waste into a liquid/sludge material that is useful as a fuel in power plants. Byproducts are ash and very small biochar particles; the size of particles and the fractions of oil/sludge, biochar, and ash are dependent on pyrolysis operation parameters. Slow pyrolysis in kilns, which takes a few days, is used to produce charcoal for barbecues, potting soil amendment, water treatment (granular activated carbon), and other purposes.

Use of biochar as a soil amendment results in significant environmental and agronomic benefits. Soil application of biochar is a means of sequestering large amounts of C and may have other greenhouse gas benefits. Preliminary reports of the impact of soil biochar applications on crop yields indicate that biochar quality is very important. Biochar is an effective adsorbent for both nutrients and organic contaminants, hence the presence of biochar in soils has been shown to improve water quality in column leaching and field lysimeters studies and it is anticipated to do the same for agricultural watersheds (Laird et al., 2009).

Application of biochar (charcoal or biomass-derived black carbon (C)) to soil is proposed as a novel approach to establish a significant, long-term, sink for atmospheric carbon dioxide in terrestrial ecosystems. Apart from positive effects in both reducing emissions and increasing the sequestration of greenhouse gases, the production of bio-char and its application to soil will deliver immediate benefits through improved soil fertility and increased crop product (Lehmann et al., 2006).

Cultivated lands in hot, arid and semi-arid environments generally have coarse textures with low water holding capacities and very low organic matter content because organic residues are quickly mineralized. A key potential benefit of soil biochar application to cultivated arid lands is the potential boost in water holding capacity that may increase crop drought tolerance. In addition, biochar represents a durable form or carbon produced in soil environments and thus has the potential to increase soil carbon content substantially in arid systems (Baldock and Smernik, 2002).

This char can be applied to soils as a soil amendment and carbon sequestration agent. The most advantageous use of a given char depends on its physical and chemical characteristics, although the relationship of char properties to these applications is not well understood (Brewer et al., 2009).

Despite the recent interest in biochars as soil amendments for improving soil quality and increasing soil carbon sequestration, there is inadequate knowledge on the soil amendment properties of these materials (Chan et al., 2008).

Charcoal was reported to be responsible for high soil organic matter contents and soil fertility of anthropogenic soils (Terra Preta) found in central Amazonia. Higher nutrient retention and nutrient availability were found after charcoal additions to soil, related to higher exchange capacity, surface area and direct nutrient additions (Glaser et al., 2002).

The present study evaluated the effect of mesquite hardwood biochar produced in traditional kilns (slow pyrolysis) on plant growth and water use for a number of common irrigated arid land crops. The overall hypothesis was that biochar amended soils would improve crop growth in an Arizona Aridisol under drought conditions. Thus, the overall goal of this research was to evaluate whether water holding capacity and resultant crop growth increased in biochar amended soils. This research had three specific objectives:

1. Evaluate crop growth and water uptake for six crops in the greenhouse at four levels of biochar incorporation (0%, 4.3%, 8.8%, and 16.3% by mass).
2. Compare sorghum crop growth in a biochar amended soil treatment and a control (no biochar) in the field.
3. Evaluate moisture retention curves for biochar and soil and use this data to evaluate greenhouse and field experiments.

MATERIALS AND METHODS

This research included a greenhouse experiment, field experiment, and laboratory analysis of biochar/soil characteristics. The greenhouse experiment focused on measurements of water uptake, crop height, fresh and dry biomass, and root development. It was conducted at the College of Agriculture and Life Sciences (CALS) greenhouse facilities on the 6th Street Garage at the University of Arizona. The greenhouse structure horizontal dimensions were 7.3 m x 4.6 m and the vertical height was 4.0 m elevation at the roof peak. The greenhouse had an automated environmental controller, heater, and evaporative cooler (Figure 1). A 35% shade screen was initially on the greenhouse roof and then removed two weeks after germination. Greenhouse temperature was maintained between 20 and 30 ^{0}C during the experiment.

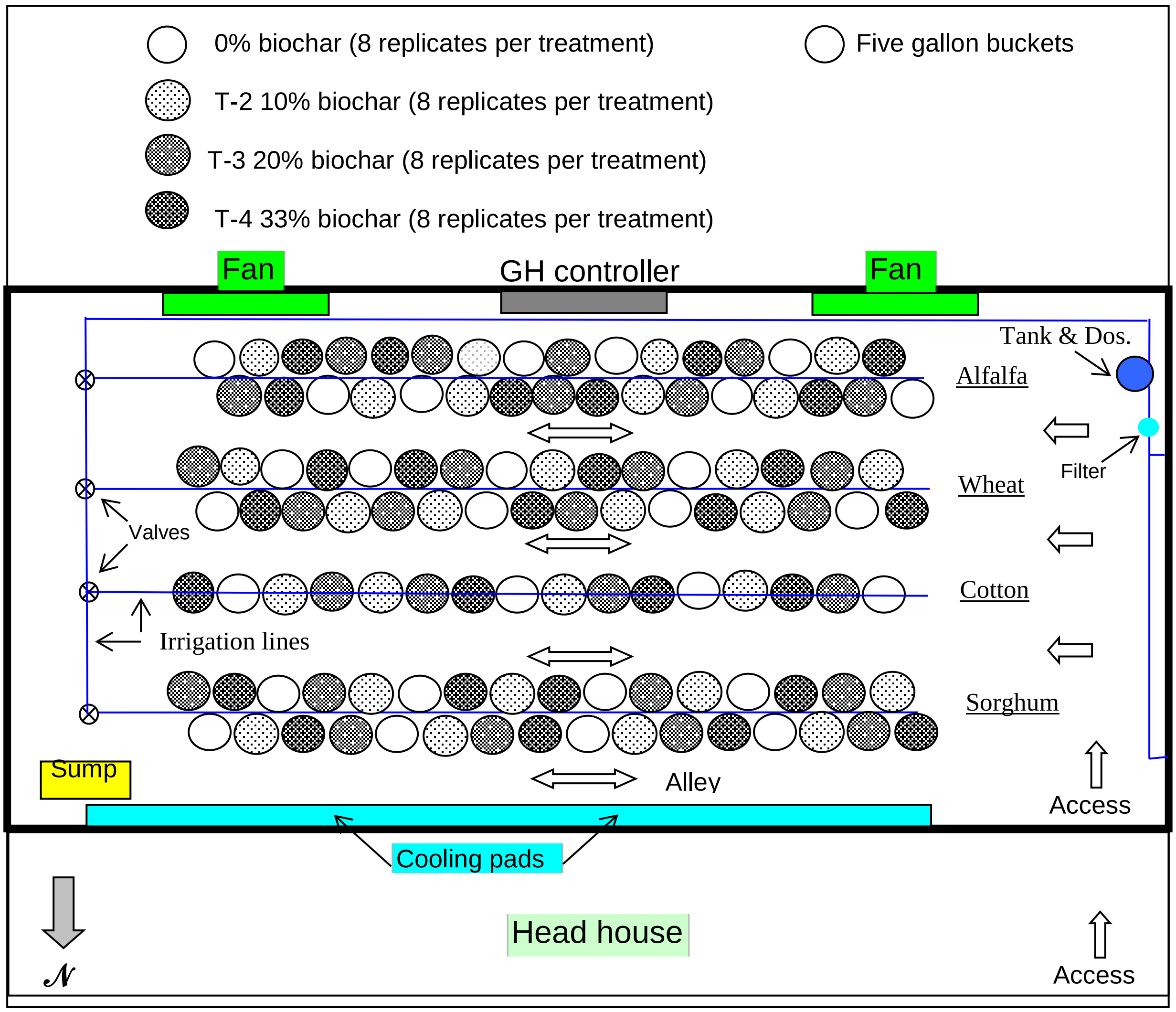

Figure 1. Greenhouse and experimental layout.

The first crops tested in the greenhouse were alfalfa (Medicago sativa), wheat (Triticum durum), cotton (Gossypium, spp) and grain sorghum (Sorghum bicolor). The second group included sweet sorghum (Sorghum bicolor) and switchgrass (Panicum virgatum).

All treatments for each crop received the same amount of water and fertilizer. Table 1 shows the planting dates and total water and fertilizer applied to each crop.

Table 1. Planting dates and total water and fertilizer applied to each crop.

Crop	Planting date	Total irrigation water applied (L)	Fertilizer applied (g) Shultz® (20-20-20)
Alfalfa	2/22/2008	70	4.62
Wheat	2/22/2008	45	2.97
Cotton	2/22/2008	63	4.16
Grain sorghum	2/22/2008	56	3.70
Sweet sorghum	5/26/2008	77	5.08
Switchgrass	7/14/2008	52	3.43

Source: Own elaboration

Crop heights were measured in all treatments before harvest and before each cutting of alfalfa. Alfalfa was cut five times (as with multiple cuttings in normal field practices) before the alfalfa treatments were removed from the experiment. Switchgrass was cut twice. Crop yield and biomass production were measured only at harvest for other crops. Plant biomass from each pot was deposited into a brown paper bag and weighed. Bags were placed in a cool dry room to dry, and were reweighed three to four weeks after harvest to obtain the dry weight. Harvest moisture content was calculated as (fresh weight – dry weight) / fresh weight. Total biomass per pot was reported as fresh biomass and dry biomass. Grain sorghum panicles were harvested from each plant and deposited into brown paper bags, counted, weighed, and then reweighed after drying. Additional plant and root measurements were collected, which are reported in Villarreal-Manzo (2009). Statistical tests for significant differences between treatments were evaluated with two-tailed t-tests, assuming unequal variances.

Moisture retention curves were generated for the following soil/biochar mixes: soil, 80% soil + 20% biochar particles by volume (8.8% biochar by mass), 70% soil + 30% biochar particles (14% biochar by mass), and biochar particles (caught above 1/8” (3.2 mm sieve). Moisture retention curves were measured with Tempe cells, pressure plates, and dew point potentiometers. Details of the procedures are available in Villarreal-Manzo (2009) along with additional moisture retention curves that were generated for biochar dust and other mixes.

The field experiment was conducted at the University of Arizona Red Rock Agricultural Center (RRAC). The particle size distribution was measured for each soil layer, and the “Soil Water Characteristics Calculator” was used to estimate the field capacity and permanent wilting point of the soil layers (Table 2) based on the particle size distribution, a normal level of compaction, and no organic matter.

Table 2. Soil characterization with the calculated parameters field capacity (FC) and permanent wilting point (PMP).

Depth (cm)	Sand (%)	Clay (%)	Loam (%)	Texture	CC (%)	PMP (%)
0 - 4	68	7	24	Sandy loam	12.2	3.9
4 – 12	61	15	24	Sandy loam	18.1	8.9
12 – 62	60	19	21	Sandy loam	20.5	11.3
62 – 81	60	22	18	Sandy clay loam	22.3	13.0
81 – 118	63	20	17	Sandy clay loam	20.5	11.9

A hydrologic balance (Figure 2) and an ombrothermic curve (Figure 3) were conducted and designed considering normal climatologic variables –precipitation, temperature and ETo- for the region and for the 1981-2010 period in order to show water availability and/or water deficits through crop phenological cycle, the same shows a constant water deficit over the year.

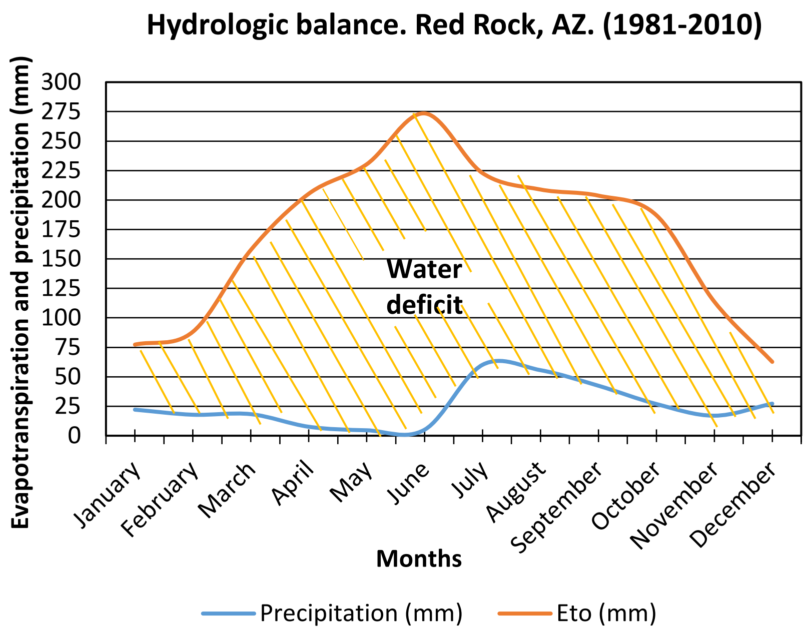

Figure 2. Hydrologic balance. Red Rock, AZ. 1981-2010.

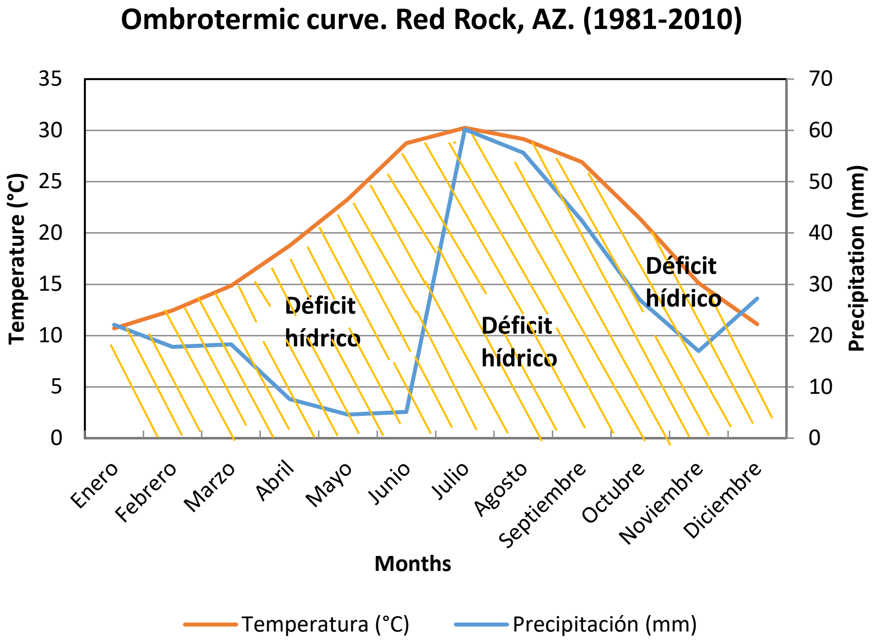

Figure 3. Ombrotermic curve. Red Rock, AZ. 1981-2010

This experiment included two treatments: 9% biochar by mass in the upper 20 cm as in the 9% treatment in the greenhouse, and no biochar. Each treatment had three replicates. The 9% biochar by mass plots were constructed by laying 5.7 cm (5.7 cm * 72% particles = 4 cm) of biochar (Figure 4) over the ground surface and then incorporating the biochar into the soil to a depth of 20 cm with a tractor driven rototiller. Each biochar plot was 8 m by 4 m and the specified 5.7 cm depth of biochar was obtained by using 27 bags of biochar per plot. Each bag weighed 18 kg (Figure 4). The entire field was 20 rows wide (four rows in each plot) and four rows were added as buffers on the east and west sides of the experimental plots. A one-meter buffer was also added at the north and south ends of plots.

Figure 4. Biochar application into the experimental plots,
before its soil incorporation, at the field experiment with forage sorghum at the RRAC

The field was drip irrigated with inline emitter tubing along plant rows. The same Schultz® fertilizer (20-20-20) used in the greenhouse experiment was used in the field experiment. The fertilizer was applied through the drip irrigation system during two injections 15 and 77 days after planting. The total N application during the season was 45 kg/ha Nitrogen (N). This is a recommended application rate for sorghum, which should have no more than 50 kg/ha N in order to discourage rank growth. An irrigation simulation was developed using the FAO-CROPWAT computer program in order to calculate crop water requirements and irrigation requirements based on soil, climate and crop data.
Results of the crop water requirements and the crop irrigation schedule are showed in Tables 3 and 4. The first one shows an ETc of 1,033.4 mm/dec and an irrigation requirement of 879.6 mm, the second shows a net irrigation schedule of 928.4 mm, and a gross irrigation schedule of 1,326.8 mm.

Table 3. Crop Water Requirements. Sweet Sorghum. Red Rock Agricultural Center- U. of A.

Month	Decade	Stage	Kc coeff	ETc mm/day	ETc mm/dec	Eff rain mm/dec	Irr. Req. mm/dec
Jul	2	Init	0.7	6.73	47.1	14.4	36.8
Jul	3	Init	0.7	6.49	71.4	19.4	52
Aug	1	Deve	0.76	6.8	68	17.5	50.5
Aug	2	Deve	0.93	8	80	17.2	62.8
Aug	3	Deve	1.11	9.38	103.2	15.8	87.3
Sep	1	Mid	1.28	10.64	106.4	14.5	91.9
Sep	2	Mid	1.3	10.7	107	13.3	93.7
Sep	3	Mid	1.3	9.99	99.9	11.7	88.2
Oct	1	Mid	1.3	9.23	92.3	10	82.4
Oct	2	Late	1.3	8.53	85.3	8.4	77
Oct	3	Late	1.25	7.44	81.9	7.4	74.4
Nov	1	Late	1.2	6.37	63.7	6	57.7
Nov	2	Late	1.17	5.42	27.1	2.3	24.8
					1033.4	**157.9**	**879.6**

Table4. Crop Irrigation Schedule. Sweet Sorghum. Red Rock Agricultural Center- U. of A.

Date	Day	Stage	Rain mm	Ks fract.	Eta %	Depl %	Net Irr mm	Deficit mm	Loss mm	Gr. Irr mm	Flow l/s/ha
14-jul	1	Init	0	1	100	159	9.8	0	0	14	1.62
15-jul	2	Init	0	1	100	97	7.5	0	0	10.8	1.25
16-jul	3	Init	0	1	100	81	7.5	0	0	10.8	1.25
17-jul	4	Init	11.4	1	100	67	7.3	0	0	10.5	1.21
18-jul	5	Init	0	1	100	58	7.3	0	0	10.5	1.21
19-jul	6	Init	0	1	100	52	7.3	0	0	10.5	1.21
21-jul	8	Init	0	1	100	83	14.4	0	0	20.6	1.19
24-jul	11	Init	0	1	100	63	14	0	0	20	0.77

Date	Day	Stage	Rain mm	Ks fract.	Eta %	Depl %	Net Irr mm	Deficit mm	Loss mm	Gr. Irr mm	Flow l/s/ha
26-jul	13	Init	0	1	100	55	14	0	0	20	1.16
29-jul	16	Init	0	1	100	67	20.2	0	0	28.9	1.11
1 Aug	19	Init	0	1	100	59	20.5	0	0	29.3	1.13
5 Aug	23	Dev	0	1	100	51	20.9	0	0	29.9	0.86
10 Aug	28	Dev	0	1	100	56	27.5	0	0	39.3	0.91
15 Aug	33	Dev	0	1	100	54	31	0	0	44.2	1.02
21 Aug	39	Dev	0	1	100	62	41.4	0	0	59.2	1.14
26 Aug	44	Dev	0	1	100	51	38.3	0	0	54.7	1.27
31 Aug	49	Dev	0	1	100	57	46.9	0	0	67	1.55
05-sep	54	Dev	0	1	100	50	45.4	0	0	64.8	1.5
11-sep	60	Mid	0	1	100	61	56.1	0	0	80.1	1.55
16-sep	65	Mid	0	1	100	50	46.4	0	0	66.3	1.53
21-sep	70	Mid	0	1	100	57	52.8	0	0	75.4	1.75
27-sep	76	Mid	6.2	1	100	52	47.5	0	0	67.8	1.31
02-oct	81	Mid	0	1	100	53	48.4	0	0	69.2	1.6
08-oct	87	Mid	0	1	100	54	50.1	0	0	71.6	1.38
14-oct	93	Mid	0	1	100	52	48.2	0	0	68.9	1.33
20-oct	99	End	0	1	100	51	46.8	0	0	66.9	1.29
28-oct	107	End	0	1	100	56	51.8	0	0	74	1.07
05-nov	115	End	0	1	100	56	51.1	0	0	73	1.06
14-nov	124	End	0	1	100	52	48	0	0	68.6	0.88
15-nov	End	End	0	1	0	0					
							928.4			**1326.8**	

The irrigation schedule applied during the research is shown in Table 5. A total of 555.5 mm was applied during the experiment, a difference of 771.3 mm respect CROPWAT crop gross irrigation schedule. From the total amount of water applied during the experiment, 62.4 mm were applied during August, 248.2 mm were applied during September, and 98.4 mm during October. The crop was intentionally stressed during October, which resulted in the low water application rate. Finally, 146.5 mm was applied during November.

Table 5. Irrigation schedule of the sweet sorghum field experiment.

Date	Irrigation depth applied (mm)	Date	Irrigation depth applied (mm)
08/05/2008	7.8	09/16/2088	47.0
08/08/2008	7.8	09/18/2008	2.1
08/11/2008	7.8	09/19/2008	47.0
08/14/2008	7.8	09/22/2008	42.4
08/18/2008	7.8	09/29/2008	101.9

Date	Irrigation depth applied (mm)	Date	Irrigation depth applied (mm)
08/21/2008	7.8	10/23/2008	25.3
08/25/2008	7.8	10/25/2008	73.1
08/28/2008	7.8	11/01/2008	73.1
09/15/2008	7.8	11/08/2088	73.4
Total			**555.5**

It was observed that weed growth prior to planting was much more rapid in the biochar plots (Figure 5). Weeds were removed prior to planting sorghum.

Figure 5. Weed population increase in biochar plots.

RESULTS AND DISCUSSION

Greenhouse crop height, biomass production, yield, and water consumption are presented for alfalfa, wheat, cotton, grain sorghum, sweet sorghum and switchgrass. Finally, biochar and soil moisture retention curve data are presented and used to assess results in greenhouse and field experiments.

Summary results of average percent differences between the biochar treatments and control are presented here. Treatments that are significantly different from the control are shaded. Box plots and figures with error bars for all treatments can be found in Villarreal-Manzo (2009).

Average crop heights in each treatment prior to cutting are shown in Table 4. In general, crop height increased with biochar concentration in alfalfa but did not increase significantly in other crops based on the two-tailed unequal variance T-Tests. Crop height actually decreased significantly in the 33% biochar switchgrass treatment.

In addition to a statistically significant increase in alfalfa crop height and biomass production, the increase in crop resistance to drought stress was very evident in the alfalfa treatments by visual inspection. During periods of water stress alfalfa in the soil only treatments generally looked stressed and often appeared extremely stressed and wilted while alfalfa in the high biochar treatments generally looked healthy. This was clearly a drought related effect since alfalfa grown in soil without biochar in non-water-stressed conditions generally appears healthy.

There were few consistent trends in plant moisture content (Table 6).

Table 6. Percent differences between biochar treatment average forage moisture content and control. Shading indicates significant difference as measured by t-test at ? = 95%.

Treatment	Biochar – 4% by Mass	Biochar – 9% by Mass	Biochar – 16% by Mass
Alfalfa			
First cutting	24%	16%	26%
Second cutting	5%	6%	6%
Third cutting	– 6%	– 3%	– 2%
Fourth cutting	– 7%	– 7%	– 6%
Fifth cutting	-14%	9%	6%
Wheat	7%	4%	2%
Cotton	– 10%	– 7%	– 11%
Grain sorghum	6%	4%	4%
Sweet sorghum	– 5%	– 10%	– 20%
Switchgrass (1st cut)	– 1%	– 2%	1%

It appears that the only consistent and significant trend is found in the sweet sorghum treatments, with a significant decrease in crop moisture content: 10% and 20% in the 4% and 9% biochar treatments, respectively. This decrease is probably caused by the fact that soil moisture content is a function of plant canopy size in that larger plants require greater moisture. Thus, moisture in the soil may be depleted faster in a treatment with larger plants so resultant plant moisture content at the time of harvest is not necessarily an indication of crop yield or plant health. These results are consistent with the fact that this was also the crop with the largest increase in crop water use with biochar concentration and the greatest depletion of soil water content (Figure 7).

There were increases in forage fresh weights with biochar concentration in most crops although some did not show statistically significant differences (Table 7). All alfalfa biochar fresh weights were significantly greater than the control, except for the first cutting 9% biochar treatment. The wheat 16% biochar treatment was significantly greater than the control, and the cotton 4% biochar treatment was significantly greater than the control. Although grain sorghum biochar treatments were not significantly greater, there was an obvious increase in production with biochar concentration (Table 7). Sweet sorghum 4% and 9% biochar treatments were significantly greater than the control. Switchgrass biochar treatments did not have any significant differences from the control, but there was a downward trend with increased biochar concentration.

Table 7. Percent differences between biochar treatment averages forage fresh weight and control. Shading indicates significant difference as measured by t-test at ? = 95%.

Treatment	Biochar – 4% by Mass	Biochar – 9% by Mass	Biochar – 16% by Mass
Alfalfa			

Treatment	Biochar – 4% by Mass	Biochar – 9% by Mass	Biochar – 16% by Mass
First cutting	35%	25%	30%
Second cutting	22%	26%	28%
Third cutting	29%	14%	51%
Fourth cutting	25%	36%	42%
Fifth cutting	10%	51%	39%
Wheat	8%	– 5%	– 28%
Cotton	53%	36%	53%
Grain sorghum	16%	27%	900%
Sweet sorghum	19%	19%	9%
Switchgrass	12%	– 42%	– 76%

Forage dry weights (Table 8) also showed increased biomass production with biochar concentration in most crops. There were no significant differences in dry weight with biochar for the first two alfalfa cuttings. However, alfalfa in the third, the fourth and the fifth cuttings had significant differences between all biochar treatments and the control except for the fifth cutting in the 4% biochar treatment. Thus, there was increased dry weight production in the alfalfa biochar treatments as the experiment progressed. The reason for the increased response to biochar over time is unknown. Possible changes include improved soil water holding capacity, microbial or rhizobia growth, or soil chemistry. Wheat and cotton 16% biochar treatment dry weights were significantly greater than the control. There were no significant differences in grain sorghum treatments, but all of the sweet sorghum biochar treatment dry weights were significantly greater than the control. The opposite result was seen in switchgrass: the 16% biochar treatment dry weights were significantly lower than the control.

Table 8. Percent differences between biochar treatment averages forage dry weight and control. Shading indicates significant difference as measured by t-test at ? = 95%.

Treatment	Biochar – 4% by Mass	Biochar – 9% by Mass	Biochar – 16% by Mass
Alfalfa			
First cutting	– 4%	16%	26%
Second cutting	6%	14%	15%
Third cutting	46%	48%	60%
Fourth cutting	45%	56%	60%
Fifth cutting	7%	15%	34%
Wheat	– 2%	– 13%	-39%
Cotton	78%	55%	82%
Grain sorghum	6%	19%	1%
Sweet sorghum	24%	30%	28%
Switchgrass	9%	– 43%	– 77%

Sorghum seeds are contained in the panicle. Average crop panicle weights were dramatically greater in biochar treatments than in the control (Table 9) with significant increases in the sweet sorghum 9% and the 16% biochar treatments.

Table 9. Percent differences between biochar treatment average crop panicle weight and control. Shading indicates significant difference as measured by t-test at ⍰ = 95%.

Treatment	**Biochar – 4% by Mass**	**Biochar – 9% by Mass**	**Biochar – 16% by Mass**
Grain sorghum fresh weight	23%	31%	75%
Sweet sorghum fresh weight	62%	64%	72%
Grain sorghum dry weight	13%	18%	54%
Sweet sorghum dry weight	57%	71%	82%

An irrigation water balance was conducted for each greenhouse treatment. The average differences between volumetric water contents before and after irrigation are shown in Figure 6. These values should not be viewed as total available water (TAW) or readily available water (RAW) since the water retained by the pot after irrigation was probably greater than field capacity due to the screen at the bottom of the pot, and the fact that drainage after an irrigation event was into air at the bottom of the pot. The data are useful, however, for comparing water use between treatments. Details on individual pot water contents before and after each irrigation are available in Villarreal-Manzo (2009). Seasonal averages in Figure 7 indicate that sweet sorghum and grain sorghum had increased water extraction with increased biochar concentration with sweet sorghum the highest (0.19). It is hypothesized that the sorghum association with mycorrhizae enabled the increased water extraction. This is discussed in more detail after the water characteristic curves are presented. Averages of soil/biochar treatments across plant types indicate that soil alone (control) had the lowest average extraction rate (0.13) and 9% biochar concentration had the highest average extraction rate (0.15); however, there was not a large difference in average extraction rate with increased biochar concentration in these combined averages, and this difference is greatly overshadowed by differences between plant types.

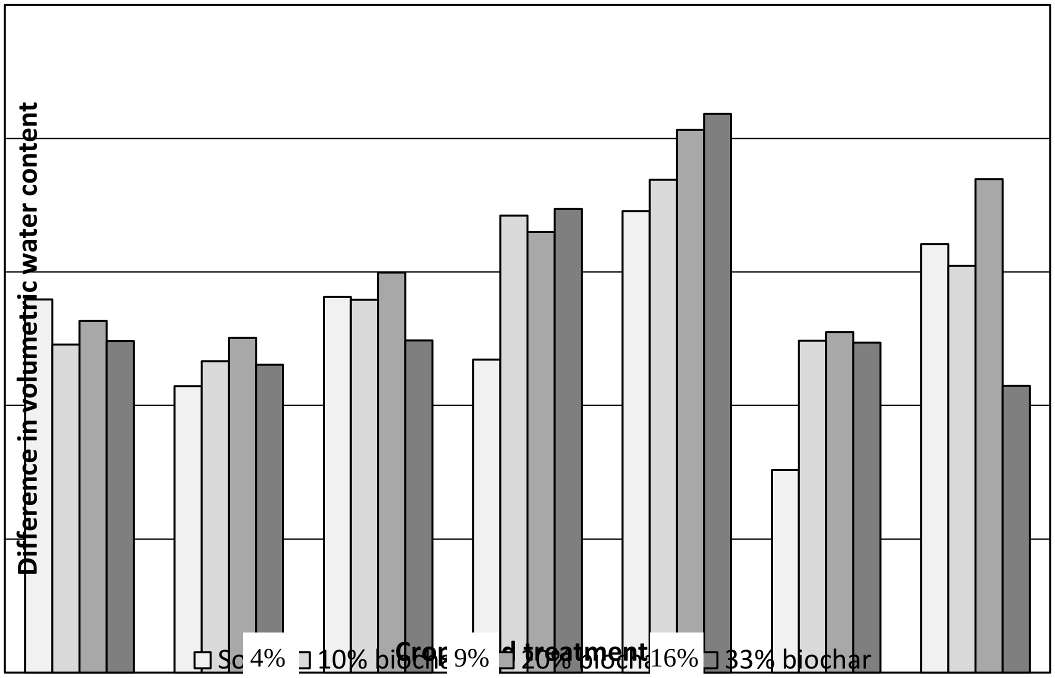

Figure 6. Seasonal average fraction of volumetric water uptake between irrigations.

Observed root patterns help to show the reason for increased water extraction in sorghum treatments, which had roots attached to biochar particles (Figure 7). In contrast, alfalfa did not have roots attached to biochar particles. Attached roots would seem to indicate that mycorrhizae are extracting water from the biochar and providing water to the roots.

In an experiment conducted to determine how different patterns of vertical distribution of charcoal and ectomycorrhizal formation affect the growth of *Larix gmelinii* (Gmelin larch) in post-fire forests, pots with a layer of charcoal in the middle of the soil profile promoted growth of the root system of the seedlings significantly more than did pots with no charcoal or with charcoal scattered throughout the soil. Along with the development of the root system, above-ground biomass and total biomass were also increased (Makoto et al., 2010).

Figure 7. Sweet sorghum roots attached to biochar particles in the greenhouse experiment.

Four mechanisms by which biochar could influence mycorrhizal abundance and/or functioning are (in decreasing order of currently available evidence supporting them): (a) alteration of soil physico-chemical properties; (b) indirect effects on mycorrhizae through effects on other soil microbes; (c) plant–fungus signaling interference and detoxification of allelochemicals on biochar; and (d) provision of refuge from fungal grazers (Warnock et al., 2007).

Further research should be conducted to prove whether or not mycorrhizae obtain water from biochar for sorghum. The following moisture retention curves show that mycorrhizae, which are known to have the ability to extract water to a lower moisture potential (25 bar) than plant roots (15 bar permanent wilting point) could increase the ability of sorghum to extract water.

Pressure plate and dew point potentiometer methods were used to calculate the soil and soil + biochar water retention curves (soil and biochar water holding capacity) using 1/10, 1/3, 1 and 15 bar pressures for 100% soil, 100% small biochar particles, 20% small biochar particles by volume + 80% soil, 30% small biochar particles by volume + 70% soil. Percentages are based on the bulk biochar volume and soil bulk volume prior to mixture.

Pressure plate measurements of water retention curves (1/10, 1/3, and 1, and 15 bar pressure), for soil, 80% soil + 20% small biochar particles (by volume), and 70% soil + 30% small biochar particles (by volume) are shown in Figure 8.

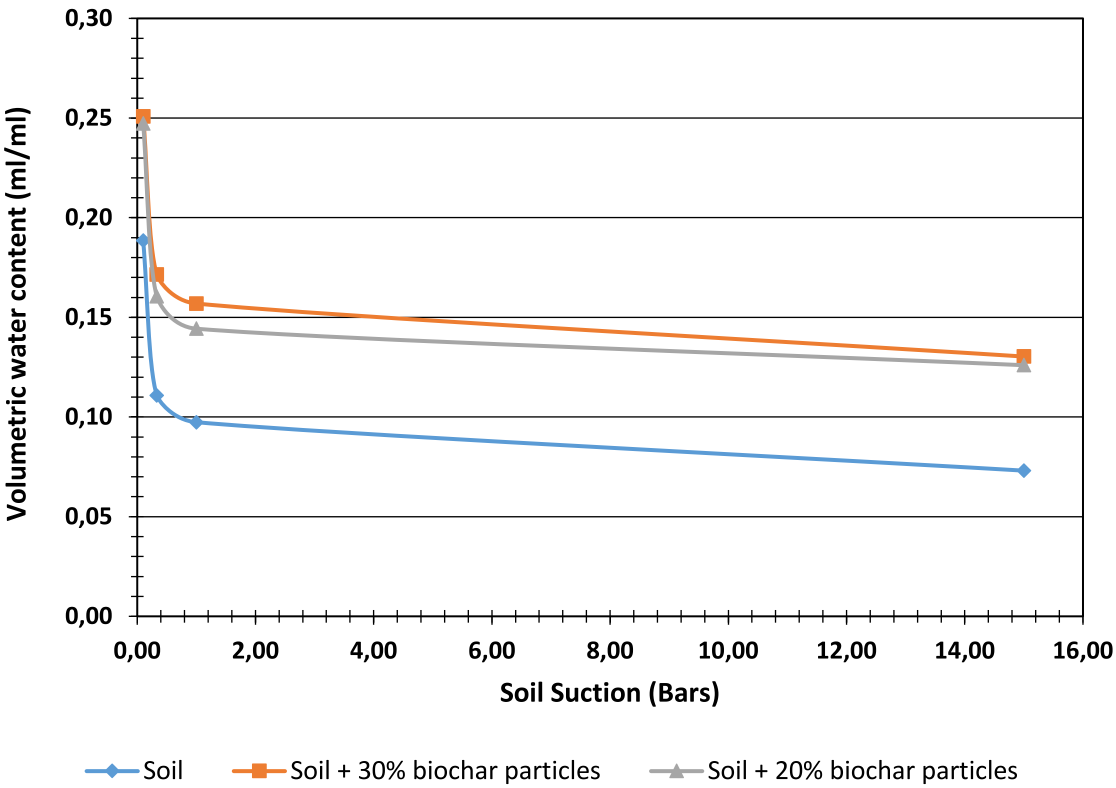

Figure 8. Water characteristic curves for soil and soil + biochar mixes (by volume) between 0.1 and 15 bar measured with the pressure plate.

Because the curves are nearly parallel, but shifted, it is evident that there is little change in total available water (TAW) with biochar addition: the difference between field capacity and permanent wilting point is approximately the same, especially in the normal range of management allowable depletion. Thus, there would seemingly be little change in readily available water with the addition of biochar. However, if a plant was able to access water in the biochar below the normal management allowable depletion or even beyond permanent wilting point, then total available water for these plants would increase with addition of biochar. In order to evaluate this scenario, the water retention curves for biochar particles only and the soil/biochar mix were evaluated at more negative potentials with a dew point potentiometer. The combined water retention curves are shown in Figure 9.

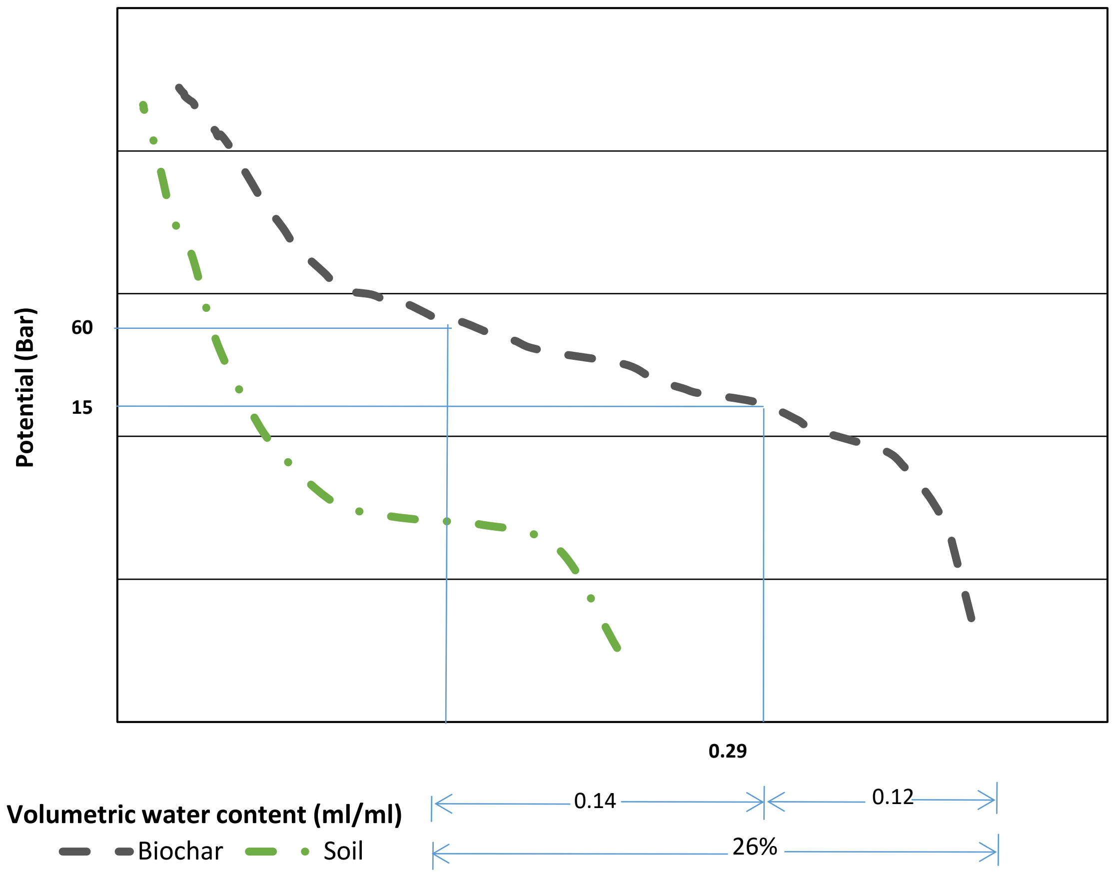

Figure 9. Biochar and biochar/soil (20% by volume) mix water retention curves.

Moisture retention curves (Figure 9) can be used to evaluate the measurements of water extraction in pots in order to estimate the percentage of water removal from biochar. The sorghum treatments, which had roots attached to biochar particles removed approximately 1% extra water by volume from pots for every 7% increase by volume in particle biochar concentration. If all of the water comes from biochar particles, then this indicates that extra 14% water (1% / 7%) is available in biochar particles for sorghum in a sandy loam soil. If this is the case, then sorghum removed the 12% moisture content between field capacity and permanent wilting point (0.29 ml/ml) and then an extra 14% from the biochar (down to 0.15 ml/ml). However, the dew point potentiometer water retention curve tests indicated that the moisture potential at a moisture content of 0.15 in biochar was approximately 60 bar (64 bar, 50 bar, and 70 bar for the three replicates evaluated in the laboratory). Although mycorrhizal fungi are very small and may be able to grow within the small pores of slow pyrolysis biochar, it is generally assumed that they are only able to extract water to 25 bar rather than 60-bar level estimated by the previous calculations. Further experimentation is needed to resolve this apparent contradiction. Nevertheless, the experiment indicates that plants with a strong mycorrhizal association may be able to extract additional water from biochar and in that way improve drought resistance.

Although the improved growth in sorghum can be explained by the increased water removal from biochar-amended soils, this does not explain why alfalfa had improved drought resistance and growth in biochar-amended soils. Alfalfa had no roots attached directly to biochar particles and thus would not have been able to remove water from biochar beyond its normal moisture potential extraction limit. It is possible that chemical or nutrient availability were improved in the biochar-amended soil and that this somehow improved alfalfa drought resistance. It is possible that increased N_2 fixation was enabled in the biochar amended soils (Ogawa, 1994), and that this increased nitrogen availability resulted in increased growth and health of alfalfa in drought conditions. Further research would be needed to evaluate whether this was the cause.

The final phase of this research was the sorghum field experiment. Periodic crop stress was allowed during the growing season. Details on irrigation, rainfall, and climate are available in Villarreal-Manzo (2009). There were no significant differences between treatments in biomass production. Average yields were 49,000 kg/ha in the biochar treatments and 48,000 kg/ha in the no biochar treatments, which are typical sorghum production levels in Arizona. There were also no significant differences in crop moisture content between treatments with 0.47 g/g and 0.45 g/g in the biochar and no biochar treatments respectively.

An FAO 56 dual crop coefficient model was used to model ET and plant stress in the two treatments. In the model, the difference between the two treatments was increased total available water (TAW) in the biochar treatment. Measured and modeled depletion percentages during the mid-season neared 100%, which means that the water content in the root zone was close to the permanent wilting point. Based on the assumed crop water stress index and the modeled moisture contents, the model predicted that the fraction of potential yield would only increase from 73% in the no biochar treatment to 76% in the biochar treatment. Thus, according to the model and as observed in the experiment, there was not enough increase in TAW in the biochar treatment to cause a significant increase in production. It is even possible that biochar in the upper soil layer may have had a detrimental effect: the soil was darker, which increased temperature in midsummer, and additional irrigation or rainwater might be caught in the upper soil layer, only to be evaporated to the atmosphere. Another possible explanation for the lack of response to biochar addition was that mycorrhizal interaction with biochar did not become established because plants were started as seedlings in the greenhouse.

CONCLUSIONS

Biomass production (dry weight) in alfalfa and sorghum increased with biochar concentration. However, wheat and switchgrass biomass production significantly decreased with increased biochar concentration. Other crops generally did not have significant differences between treatments. Alfalfa appeared to have a trend toward increased crop production with biochar concentration as the experiment progressed.

Grain sorghum, barley, and sweet sorghum water extraction increased with biochar concentration. It is hypothesized that the increase in sorghum was caused by the mycorrhizal association with sorghum. This was evidenced by the fact that sorghum roots were attached to biochar. There were significant differences in panicle weight in the sweet sorghum with increased biochar concentration. Although differences were not significant in the grain sorghum, there was a dramatic average percent increase with biochar concentration in the grain sorghum treatments.

Water characteristic curves were derived for soil and biochar mixes. The AWC in both sandy loam soil used in the experiment and the biochar particles was approximately 0.12 ml/ml. The most likely explanation for increased sorghum water extraction was its ability to access water at very negative moisture potential in the biochar with mycorrhizae.

The overall result of this experiment is that the hypothesis is confirmed for some crops: incorporation of biochar in soils improved crop growth and drought resistance. However, not all crops responded positively to biochar. Switchgrass yields decreased with increased biochar. This work underscores the need for continued research over a wide range of soils, plants, ecological zones, and biochar concentrations.

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FILTER SOCKS TO MITIGATE RUNOFF, SOIL AND PHOSPHATE LOSSES FROM ARABLE LANDS UNDER CURRENT AND EXTREME RAINFALL EVENTS

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INTRODUCTION

Filter socks (FS) are widely used in the USA as a best management practice (BMP) to control sediment on construction sites. They are little known within the UK, but with water quality widely declining due to diffuse pollution from arable lands, FS may be one solution to control sediment and phosphate (PO_4^{3-}) inputs to vulnerable watercourses. Controlling these inputs will satisfy water quality targets (prescribed by the EU Water Framework Directive; WFD), decrease sedimentation and mitigate flooding and eutrophication processes. As a result, aquatic ecosystems will be better protected. The main focus of this research is to enhance the efficacy of FS in the control of sediment and PO_4^{3-} using a range of fill media and phosphorus sorbing materials (PSMs).

PSMs are used within the wastewater industry for P adsorption from sewage sludge. They usually consist of P-sorbing resins or membranes, which are rarely used by the agricultural industry due to cost and upscaling constraints. Consequently, there is a need for alternative PSMs that can be used in agricultural settings. Appropriate field placement of FS will allow sediment capture as well as PO_4^{3-} adsorption from the runoff before it enters the drainage network. The PSM initially trialled in this study was chosen from a literature review and project based assessment criteria (ability to reduce PO_4^{3-} to meet the WFD water quality target; 0.05 mg P l^{-1}).

Climate change is projected to increase the number of extreme weather events, including higher rainfall intensities, which will ultimately increase the frequency and magnitude of runoff and soil erosion (IPCC 2014). For the UK, it is been predicted that only a 10% increase in winter rainfall could result in a 150% increase in arable soil erosion (Favis-Mortlock & Boardman 1995), with associated delivery of sediment and PO_4^{3-} load to watercourses. Therefore, it is essential that PSMs and FS fill media are effective in mitigating the effects under current and extreme rainfall events (RE's).

METHODS

The experiment consisted of monitoring soil erosion and runoff in a fully replicated field trial which ran for 36 weeks from July 2015 – March 2016. The field site was located at 52.095354, -2.7217984 (Lat/Long) near Hereford, UK. It was chosen for its high erosion risk, fine silty soils (Cranfield University 2016); commonly steep slope gradients (11-17°); susceptibility to soil compaction and surface capping; relatively high annual rainfall (665 mm yr^{-1}) and increasingly extreme storm events; and erosive land use (maize with rows running up/down slope, perpendicular to the contour). These conditions meant the tenant farmer had regularly experienced soil erosion issues.

Twenty hydrologically isolated erosion plots (1.5 m wide x 15 m long) were set-up, oriented downslope; parallel to the maize rows. The plots allowed the collection and monitoring of runoff and eroded sediment in plastic tanks (capacity 227 litres each), using pre-calibrated Liquid Vertical

Continuous Series (LVCS) sensors located in each of the runoff collection tanks. Each sensor was connected to a data logger (Delta-T DT80/2), and together they monitored the levels of runoff in the tanks, as well as site precipitation, temperature and humidity as recorded by a linked on-site weather station. The data logger began logging immediately after the onset of 0.2 mm of rainfall and continued for 1 hr post-rainfall cessation. Baseline soil sampling was undertaken on each erosion plot, and sampling of the runoff in the tanks took place on 4 occasions (Table 1).

Table 1. Sampling occasions

Sample period	**Date**
1	09/07/15 – 14/10/15
2	15/10/15 – 14/01/16
3	15/01/16 – 17/02/16
4	18/02/16 – 09/03/16

Table 2. Treatment codes

Treatment	**Code**
Control (no FS)	Control
Compost FS	C FS
Woodchip FS	W FS
Compost + 'Product' FS	C+P FS
Woodchip + 'Product' FS	W+P FS

Table 3. Mean runoff volume (l plot^{-1}) and its difference from the control (%), and sediment concentration (g l^{-1}) for each treatment and sample period.

Treatment	**Runoff Volume (l plot^{-1}) and change from the control (%)**					**Sediment concentration (g l^{-1})**			
	Sample period	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**
Control	Mean ±S.E	23.6^{a} ±0.0	59.1^{a} ±6.4	$19.9^{a,b}$ ±4.7	12.6^{b} ±2.2	0.6^{a}	$0.2^{a,b}$	0.6^{a}	0.3^{a}
	Change (%)	-	-	-	-				
C FS	Mean ±S.E	14.8^{a} ±3.8	31.8^{b} ±3.7	11.8^{a} ±1.7	5.9^{a} ±1.2	$0.7^{a,b}$	0.2^{a}	0.4^{a}	0.2^{a}
	Change (%)	-37.5	-46.3	-40.7	-52.9				
W FS	Mean ±S.E	17.0^{a} ±4.1	32.5^{b} ±3.0	$17.0^{a,b}$ ±3.9	5.2^{a} ±0.7	$1.2^{a,b}$	0.2^{a}	0.2^{a}	0.1^{a}
	Change (%)	-28.1	-45.0	-14.8	-58.8				
C+P FS	Mean ±S.E	16.2^{a} ±1.5	$34.7^{a,b}$ ±12.4	27.2^{b} ±5.2	$7.9^{a,b}$ ±2.6	2.3^{b}	$0.3^{a,b}$	0.3^{a}	0.2^{a}
	Change (%)	-31.3	-41.3	+36.3	-37.3				
W+P FS	Mean ±S.E	20.7^{a} ±3.8	$56.1^{a,b}$ ±7.8	$24.8^{a,b}$ ±4.9	13.8^{b} ±2.0	$1.2^{a,b}$	0.3^{b}	0.3^{a}	0.2^{a}
	Change (%)	-12.5	-5.0	+24.4	+9.8				

Sample period 3; plot 14 not included in analysis

Sample period 4 ; plots 1 an
N.B. Within each sample p
different (p≤0.05) following

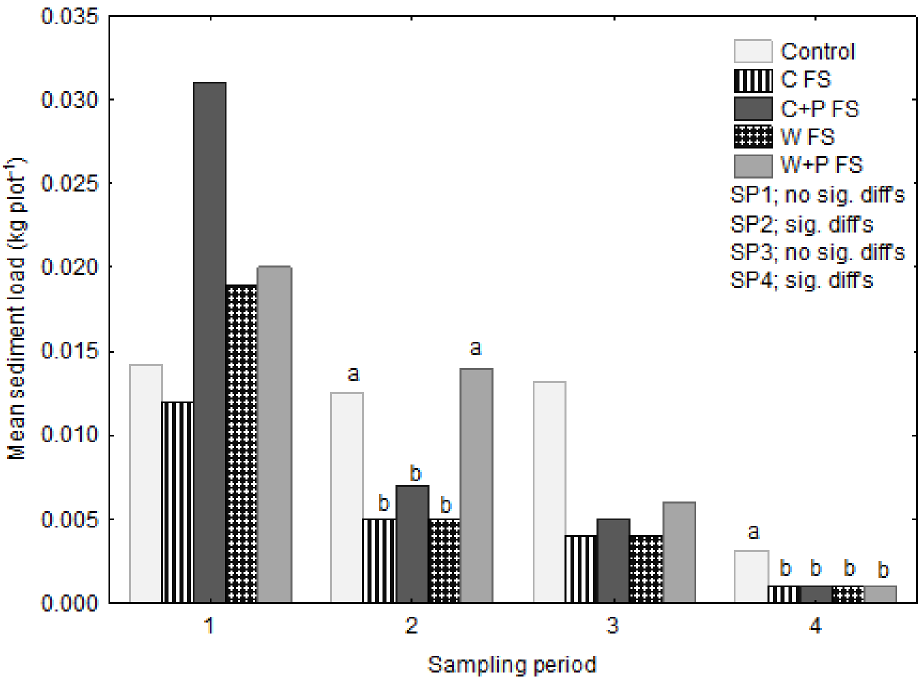

Figure 1. Mean sediment load (kg plot^{-1}) between treatments and sampling periods. Significant results between treatments indicated by different lettering.

The field trial investigated 5 treatments, each with 4 replicates (Table 2) which were randomly distributed across the erosion plots. As the Control did not have a FS treatment it represented 'business-as-usual'. The trials used 8-inch diameter FS with PAS100 compost or fine woodchip used as fill media. This met the FS fill media specifications (Alexander Associates, 2006). A branded proprietary product (P) was used as the PSM, applied at the required application rate of 20 kg m^3 within the C+P FS and W+P FS treatments, providing an industry 'standard' with which to compare alternate PSM treatments. Treatment performance criteria included total runoff volume (l plot^{-1}), runoff sediment concentration (g l^{-1}), sediment load (g plot^{-1}), and runoff PO_4^{3-} concentration (mg P l^{-1}).

Baseline sampling showed plot soil properties (texture, pH, organic matter, Total P) were not significantly different, confirming the trials were a fair test. Weather data indicated 214.8 mm of rain fell during the trial, with a maximum RE intensity of 228 mm^{-1} hr^{-1} over a 7 minute duration (22/08/15), and maximum RE duration of 41 minutes (12 mm^{-1} hr^{-1} intensity, 21/09/15). The majority of REs were 1 in 1 year storms; however a 1 in 30-year and 1 in 400-year RE were also recorded.

The results show an initial release of sediment and nutrients from the FS, and high runoff volume in the early stages (July-October 2015) of FS application. This is consistent with results of other FS and compost studies (Al-Bataina et al. 2016; Glanville et al. 2003; Nguyen & Marschner 2013; Waters 2010). This first 'flush' of runoff is shown in Table 3, where no statistical differences were observed between the FS treatments and the Control. After this initial period, the C FS, W FS, and C+P FS produced statistically less runoff than both the Control and the W+P FS plots. By the end of the trial (Sampling Period 4), the majority of the FS treatments were associated with a consistent reduction in runoff volume, compared to the Control. Regarding sediment concentrations, initially, the FS treatments gave statistically higher values than the Control (Table 3). However a robust trend of lower sediment concentrations was identified by Sampling Periods 3 and 4 suggesting that FS efficiency increases over time. The W+P FS treatment did not perform as efficiently, with runoff and sediment concentrations being no different or even higher than the Control (not significantly) (Table 3). This is due to uncontrollable plot spatial variability (e.g. microtopography, soil compaction levels), and noise in the data. This could have been due to a lack of replicates in Sample Periods 3 and 4, where movement of the plot boundaries only allowed 3 replicates for W+P FS and C+P FS treatments.

Regarding sediment loads (Figure 1), there was a distinct first flush in Sample Period 1 whereby the majority of FS treatments displayed higher sediment loads than the Control, however by Sample Periods 2-4 the FS treatments had consistently lower sediment loads than the Control, although these were not always statistically significant.

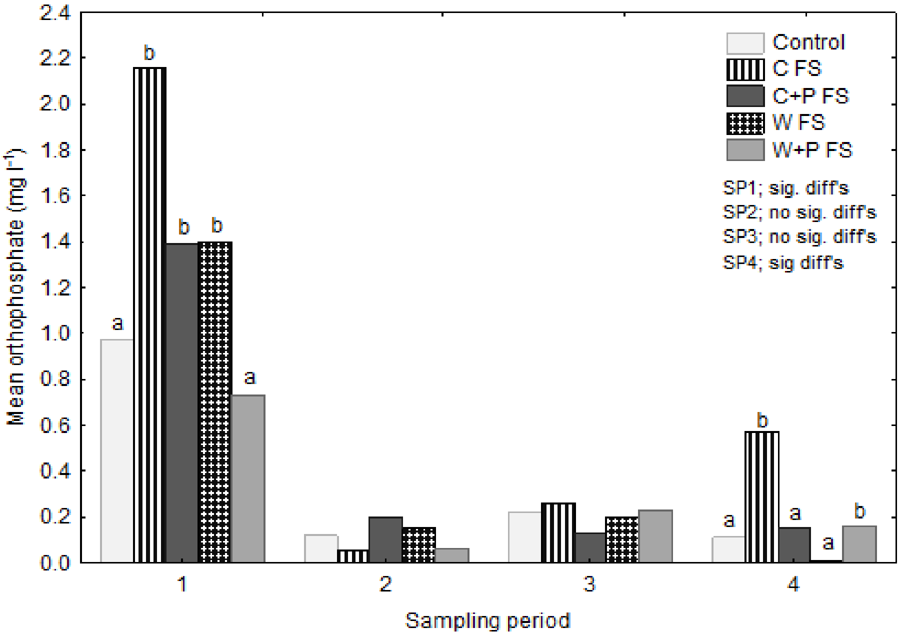

Figure 2. Mean orthophosphate (mg l^{-1}) between treatments and sampling periods. Significant results between treatments indicated by different lettering.

PO_4^{3-} concentrations in the runoff varied significantly between treatments (Figure 2). Runoff from the Control plots had statistically lower PO_4^{3-} concentrations than the FS plots in Sampling Period 1. In Sampling Periods 2 and 3, there were no statistical differences between treatments. A similar result was observed for Sampling Period 4 with the exception of the C FS and W+P FS treatments, which were associated with statistically higher PO_4^{3-}concentrations in the runoff.

It is suggested that the uncontrollable weather conditions impacted on the variability and lack of results between Sample Periods, especially when comparing Sample Period 1 to Sample Periods 2-4. A longer monitoring period might have captured more statistically robust differences between the FS treatments and the Control, as the current results suggest a prolonged first 'flush' period, which has masked treatment efficiency. Variability between same treatment replicates, both within and between sampling periods, highlights the challenges associated with fieldwork, due to temporal and spatial variability in site conditions.

CONCLUSIONS

Initial results show that FS are an effective and under-utilised end-of-pipe solution for the control of soil erosion and runoff. More research is required into the phenomenon of the 'first flush' of runoff, sediment and P and to test appropriate materials which can be used as PSMs. The results will encourage wider uptake of these mitigation measures.

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1.12.O

ENHANCED SOIL ORGANIC MATTER STABILIZATION IN CARBONATED SOILS OF SEMIARID AREAS DOES NOT OCCUR IN LOW ORGANIC MATTER CULTIVATED SOILS

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INTRODUCTION

Soil organic matter (SOM) stocks are determined by the balance between organic matter inputs and decomposition processes. SOM decomposition is driven by a combination of factors and processes. The chemical properties of the organic matter, the physical protection, the structure and activity of the microbial communities are said to be the factors which will define SOM turnover (Condron et al., 2010; von Lützow et al., 2007). In calcareous soils of semi-arid areas carbonates can slow down soil organic matter turnover by interfering with soil aggregation (Bronick and Lal, 2005; Fernández-Ugalde et al., 2014) and by the capacity of Ca^{2+} to bind soil organic matter (Bonneau and Souchier, 1987; Kononova, 1975). Ca salts bonded to clay and organic matter can also lead to increases in aggregate stability (Grant et al., 1992).

In Mediterranean areas calcareous forest soils typically hold high stocks organic matter in contrast arable lands show very low organic matter levels, as low as those of non-calcareous soils (Romanyà and Rovira, 2011). In this context we hypothesize that the stabilization processes that occur in carbonated soils may not be the same in soils with low or high humus content. In order to gain knowledge on the mechanisms stabilizing SOM in soils with carbonates we have studied SOM stability in bulk soils and in particle size fractions with contrasted levels of SOM. We have selected arable, abandoned, and forests soils in six different areas within the Mediterranean region; 3 areas with carbonated soils and 3 areas with non- carbonated soils. Our aims were to determine whether the mechanisms by which carbonates can slow down organic matter are equally important in soils with high humus content than in soils with low humus content. To help interpreting the data we have determined the nature of microbial communities by lipid extraction and PLFA analyses.

MATERIAL AND METHODS

Study site and soil sampling. In the dry Mediterranean area of Central Catalonia six locations were selected; three on carbonated soils and three in non-carbonated soils. Rainfall in the area ranged from 400-620 mm and mean annual temperature from 13-15 ^{o}C. In each location three different land uses were selected (arable land, abandoned fields and mature forests) so we have a range of organic matter content. In each location and land use type we took 3 samples from the upper 10 cm of the mineral and bulked.

Soil analyses. Soils were analysed for texture, pH, organic C and N content and carbonate content (Porta et al., 1986). We also analysed calcium, aluminium and iron by wave-length dispersive X-ray fluorescence spectrometry (WD-XRF, Axios, PANalytical). Soil samples were also fractionated into different particle size fractions: coarse sand (2000-200 µm), fine sand (200-50 µm), coarse silt (50-20 µm) and fine silt+clay (<20
µm) (Rovira et al. 2010).

We determined the PLFA abundance of soils, which are biomarkers for specific groups of microorganisms following the procedures described by Frostegard et al. (2011). To investigate microbial activity, we incubated bulk mineral soils (0-10 cm depth) and soil particle size fractions (coarse sand, fine sand, coarse silt and fine silt+clay) for 119 days at 20 ºC at 60% of WHC and measured soil respiration. Soil respiration was measured by quantifying CO_2 concentration in the headspace within each bottle by a LICOR-840 infra- red gas analyser (IRGA).

RESULTS AND DISCUSSION

Organic carbon in carbonated forest soils was much higher than in non-carbonated forest soils. Abandoned fields also showed larger amounts of organic matter in carbonated soils. This is in agreement with classical soil studies in which calcareous soils hold large amounts of organic matter to form mull humus type (Bonneau and Souchier, 1987). In contrast, arable soils sowed very low levels of organic C irrespective of the presence of carbonates (Table 1).

Table 1. Chemical properties and total PLFA concentration in soils with and without carbonates and with different land use (cultivated field (CFi), abandoned field (AFi) and forest (Fo)). Capital letters show differences between soils with or without carbonates at p<0.05. Values are means ± SE(n=3).

	- carbonates			**+ carbonates**		
	CFi	AFi	Fo	CFi	AFi	Fo
Organic C (%)	1.17±0.31	1.26±0.22 **A**	1.92±0.08 **A**	1.24±0.41	1.93±0.09 **B**	4.49±0.51 **B**
Total N (%)	0.104±0.026	0.142±0.031	0.158±0.019 **A**	0.120±0.030	0.162±0.010	0.279±0.016 **B**
Total PLFA (µmol gC^{-1})	4.62±0.61	8.00±0.09 **A**	4.89±0.22 **A**	8.85±3.33	4.85±0.49 **B**	2.70±0.18 **B**
pH	6.96±0.40 **A**	6.69±0.29 **A**	6.14±0.17 **A**	8.31±0.12 **B**	8.22±0.04 **B**	8.08±0.06 **B**
Carbonates (%)	0.11±0.05 **A**	0.19±0.09 **A**	0.02±0.01 **A**	59.73±3.03 **B**	56.38±3.16 **B**	54.74±5.52 **B**

Organic C in cultivated carbonated soils decomposed more quickly than in non-carbonated soils while in forest soils it showed the opposite trend. No differences between carbonated and non-carbonated soils were observed in abandoned fields (Fig. 1).

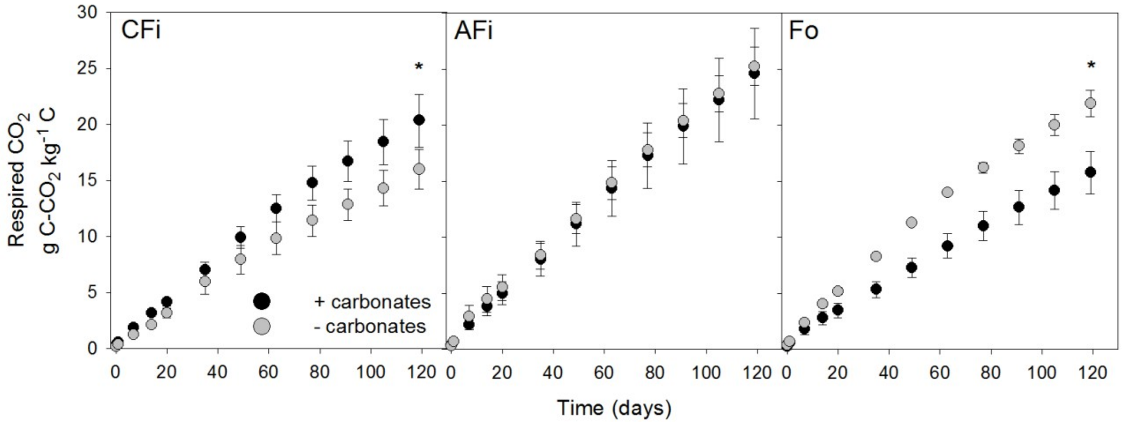

Figure 1. Cumulative carbon (C) mineralization of carbonated and non carbonated soils at three different land use: cultivate field (CFi), abandoned field (AFi) and forest (Fo). Means and standard errors of three plots.

In carbonated forest soils all particle size fractions showed lower microbial activity than non-carbonated soils. Abandoned fields showed a similar trend although the differences between soil types were less or no significant. In contrast, in cultivated soils low microbial activity per unit of C in carbonated soils only occurred in fine silt + clay fraction while coarse silt and fine silt fractions showed increased microbial activity (Fig. 2).

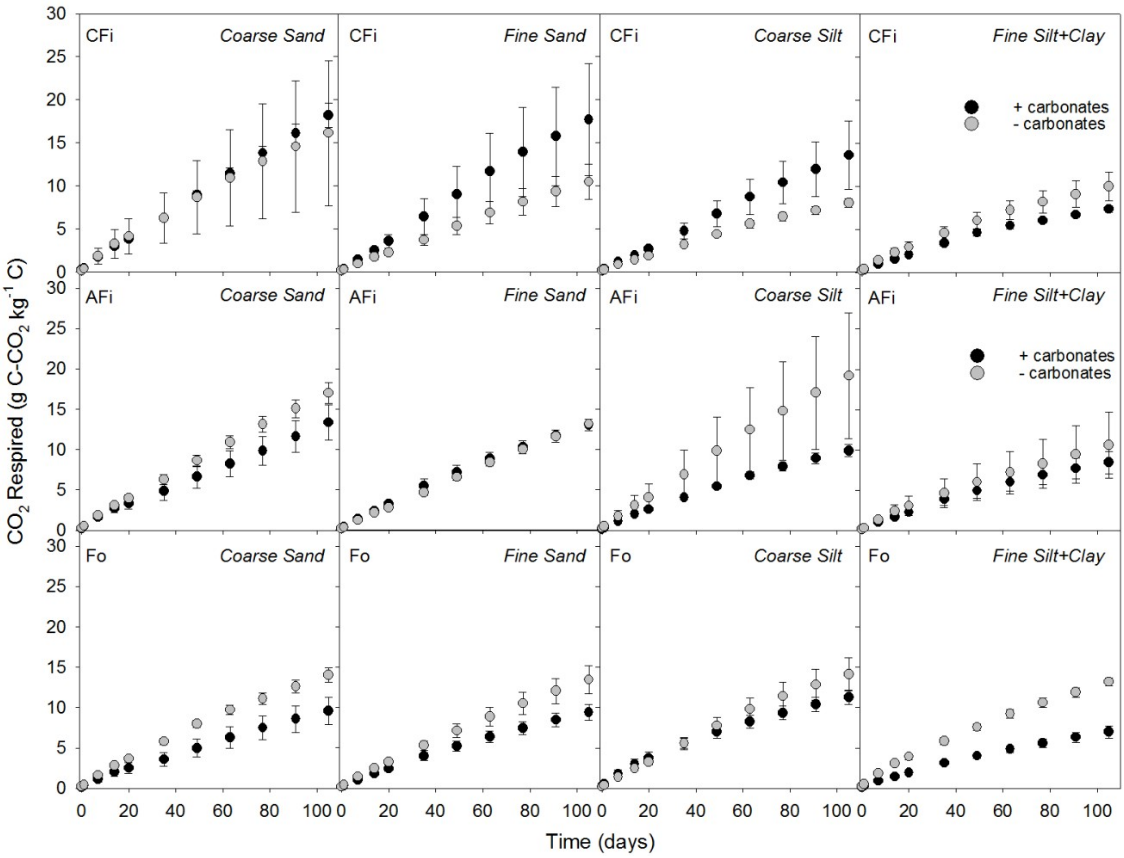

Figure 2. Cumulative carbon (C) mineralization of carbonated and non carbonated soils at three different land use (Cultivate field, Abandoned field and Forest) and for each soil fraction.
Means and standard errors of three plots.

PLFA analysis showed high abundance of bacteria relative to soil C in cultivated carbonated soils as compared to that of forest or abandoned soils. This relative importance of bacteria in cultivated soil has been pointed out in other carbonated soils (Zornoza et al., 2009). In non- carbonated cultivated soils the amount of bacteria relative to of C was similar to that of forest and even decreased in the case gram+ bacteria. Abandoned non cultivated soil showed a large increase of gram- bacteria.

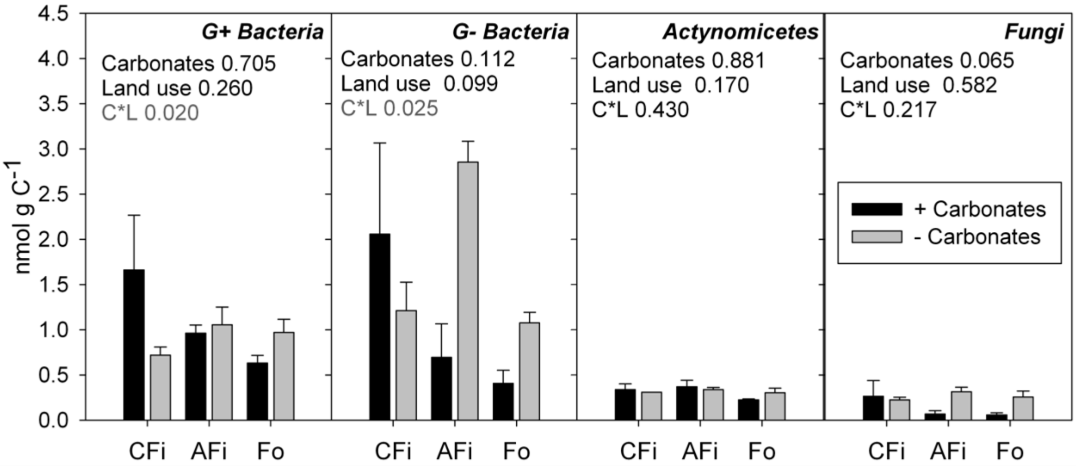

Figure 3. PLFA biomarkers representative for different microbial groups in all land uses (cultivated field (CFi), abandoned field (AFi) and forest (Fo)) and two types of soils (soils with carbonates and without carbonates). Error bars represent standard errors (n=3).

The abundance of gram– bacteria can be related to the abundance of undecomposed organic matter debris (Buyer et al., 2010). In arable lands because of the slow decomposition of newly incorporated organic matter debris accumulate as particulate organic matter (Fliessbach et al., 2000). The high abundance of bacteria, mainly gram – in carbonated soils may be responsible for the higher decomposition rate of the mostly undecomposed particulate organic matter. In high organic matter soils humified and partially decomposed organic matter may distribute across all soil fractions while in low organic matter soils humified organic matter may mostly occur in the fine fraction (sil+clay). In carbonated soils the capacity of Ca^{++} to bind to humified organic matter may slow down organic matter decomposition. However our results show that organic matter respiration in carbonated soils increases with the amount of Ca while it decreases with Fe and Al only carbonated soils (data not shown). On the other hand, high abundance of bacteria (mainly gram -) relative to C in low organic matter soils may explain its high decomposition rate.

CONCLUSIONS

Interactions between polyvalent cations and organic matter can favour soil organic matter accrual when humified organic matter is available. In contrast, soils holding less humified organic matter in coarse fractions does not show this effect as polivalent cations may not bind to fresh organic matter debris.

Relative high abundance of bacteria (mainly gram -) in low organic matter carbonated soils increases fresh organic matter debris decomposition. Increased fresh organic matter decomposition in carbonated arable soils makes difficult soil organic matter accrual in such soils.

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1.1.P

THE INFLUENCE OF THE SOIL MANAGEMENT STRATEGY ON THE SOIL ORGANIC CARBON CONCENTRATION IN MEDITARREAN VINEYARDS

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INTRODUCTION

Mediterranean vineyards are one of the most altered agricultural systems in southern Europe (Rodrigo-Comino et al. 2016). Traditional soil management includes intensive tillage to eliminate water and nutrient competition with weeds. Moreover, the soil is bare the most of the year, vineyards are usually located on slopes (Rodrigo-Comino et al. 2017) and the scarce rainfalls are intense, which encourages soil erosion processes. European countries of Mediterranean region loss an average of 4,31 $Mg{\cdot}ha^{-1}$ of soil by water erosion, reaching 10-20 $Mg{\cdot}ha^{-1}$ of soil loss in agricultural areas under conventional practices (Panagos et al. 2015). Water erosion not only mobilizes sediments, but also certain nutrients and organic matter, driving to a loss of fertility.

Continuous tillage and low annual organic inputs are the main causes of soil organic matter (SOM) depletion and the current low SOM content in Mediterranean vineyards (Panagos et al., 2013). Low SOM content results in a reduction in aggregate stability (Verchot *et al.* 2011), which reduces its resistance to the impact of raindrops. When aggregates are destroyed by erosive rainfalls, the formation of splash crusts occurs. Crusts produce very low infiltration rates, promoting runoff and soil particle detachment.

Therefore, there is a need to perform alternative soil management strategies to encourage soil sustainability. Those managements devoted to enhance SOM contents could increase soil quality because SOM improves physical, chemical and biological soil properties (Six et al., 2000). Groundcovers have been shown as a properly management to reduce soil loss regarding tillage, that in vineyards can reach 100 $Mg{\cdot}ha^{-1}$ (Prosdocimi et al. 2016, García-Díaz et al. 2017). Thus, we propose the use of groundcovers in vineyards to increase SOM content through tillage cessation and higher organic inputs.

This article evaluates the influence of two types of groundcovers (one seeded and other spontaneous) versus conventional tillage (T) along 4 years in vineyards of the Protected Designation of Origin "Wines of Madrid", settled in central Spain.

MATERIAL Y METHODS

Study area

The study was carried out on 4 vineyards located in the Denomination of Origin “Wines of Madrid”, Madrid region, Spain. The average temperature is 14.8 ºC and the average annual rainfall is around 400 mm. Vineyards are located in Belmonte de Tajo (two vineyards), Campo Real and Navalcarnero municipalities. All of them are planted on eroded slopes. The main soil types of the region were represented.

Experimental design, soil measurements and methods

The study is composed by 3 treatments that were performed in all 4 vineyards. In all vineyards, each treatment is performed in three consecutive rows. The treatments were: conventional tillage (T) consisted in 2-3 tillage operations per year (following the conventional soil management of farmers in the area) with chisel and 15-20 cm deepth; a grass, *Brachypodium distachyon* (CB), which was seeded in the central 2 m of the inter-rows in December 2012; and spontaneous vegetation (CS). The vegetation cover treatments were mowed at the height of 12 -15 cm every spring one or two times depending on the cover development. This height allows to CB to auto-seed year after year. Litter was left on the soil surface. Both groundcover were developed from December 2012 to October 2015, were all vineyards and treatments were completely tilled.

From 2013 to 2016, at the beginning of the summer, soil samples from superficial horizon (0 – 10 cm) were collected to analyze soil organic carbon (SOC) and assess its evolution by wet oxidation (Walkley and Black, 1934).

Data Analysis

Data were analyzed using variance analysis (ANOVA) with Statistica 10 software (Statsoft, 2010). In the case of lack of normality variables were transformed. Two types of analyses were performed: statistical differences between years for the same treatment (1) and statistical differences between treatments for the same year (2).

RESULTS

CB treatment did not show statistically significant differences regarding T at any campaign although a continuous increase could be observed (Fig. 1). The increasing SOC rate for CB was 1.01 $g \cdot kg^{-1}$ $year^{-1}$. In 2016 CB scored statistically higher SOC than in 2013. CS was the most effective soil treatment to increase SOC. CS increased SOC from 6.7 $g \cdot kg^{-1}$ in 2013 to 10.7 $g \cdot kg^{-1}$ in 2015 with a rate of 1.33 $g \cdot kg^{-1}$ $year^{-1}$. Statistically higher SOC was observed in 2015 regarding 2013 for CS, while in 2016 (8 months after tilling) the results between SOC in 2016 and SOC at the beggining in 2013 were not significant, due to a dramatical reduction in SOC content.

T treatment showed slight increases along the studied years but not statistically significant, from 6.7 $\cdot kg^{-1}$ in 2013 to 8.6 $\cdot kg^{-1}$ in 2016. In any case, this rate of increase was lower than the rate of the groundcovers.

Between treatments, 2015 was the only year with statistically significant differences with CS differing from T, with almost a 25% more of SOC content.

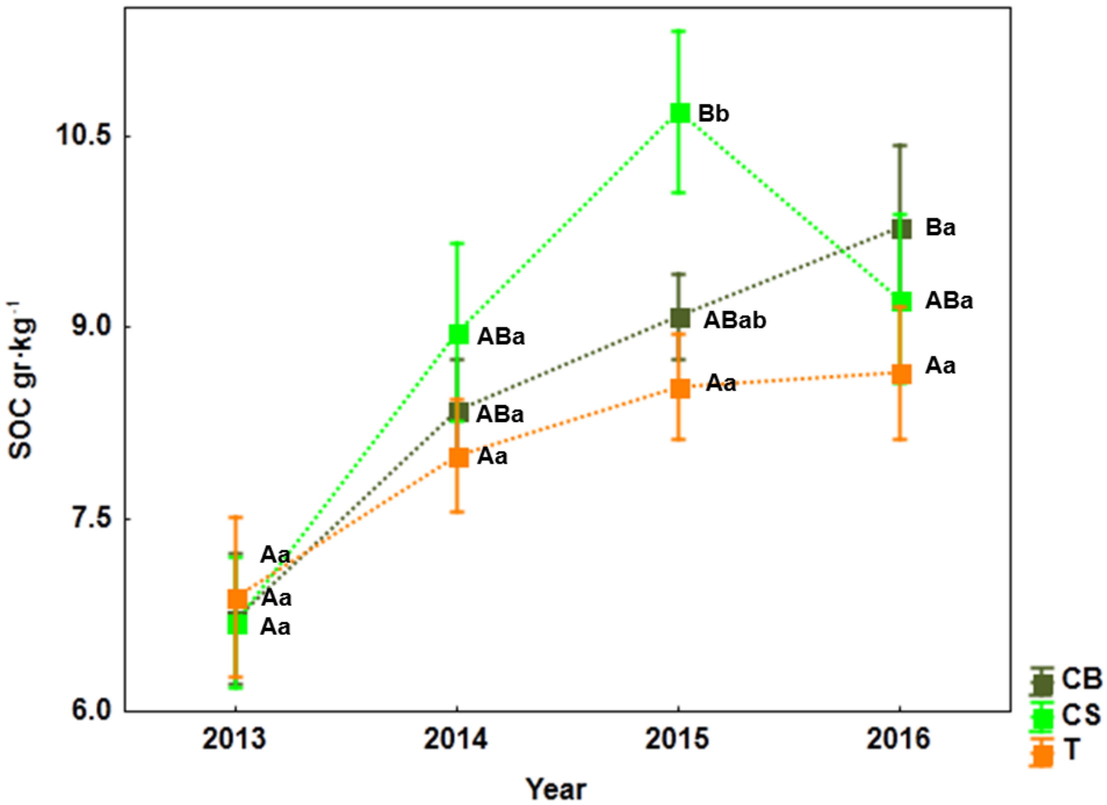

Figure 1. Mean and standard error of SOC from 2013 to 2016. Treatments were: *Brachypodium distachyon* cover crop (CB), spontaneous vegetation cover crop (CS) and conventional tillage (T). Uppercase letters mean differences between years for the same treatment. Lowercase letters mean differences between treatments at the same year. $p<0.05$.

DISCUSSION

CC has been proved to be a useful soil management increasing SOC in Mediterranean semiarid vineyards, as other researchers found (Peregrina et al., 2010; Ruiz-Colmenero et al., 2013). Statistically significant differences were hardly found because of the high variability produced by the four vineyards. CS was the most effective soil management strategy to enhance SOC with an increase of SOC in 4.0 g·kg $^{-1}$ in three years. CB showed lower increases without significant differences regarding T. Nevertheless, both CC showed high values of annual SOC increases with 1.33 and 1.01 g·kg^{-1} year^{-1} for CS and CB, respectively. These rates represent annual increases of 19.9 and 14.9 % (CS and CS respectively) which are above the goal of the 4 per mille initiative (Minasny et al., 2017). This demonstrates the effectiveness of the use of cover crops.

In 2016, with the tillage of the whole trial, statistical differences were not find as it was in 2015 because of a decrease in SOC content of CS. Tillage is, thus, one of the major driving forces of SOC mineralization (Kabiri et al., 2016). CB showed slight enhancement from 2015 to 2016 but these differences were not statistically significant. This decrease in SOC content after tillage was greater in that treatment that presented higher SOC content (Fig. 1).

An increase in SOC content in T along the studied years was observed. This could be attributed to a less intense conventional tillage performed, with less chisel passes, than the typical in the study area.

CONCLUSIONS

CC are proved as an efficient management to increase SOC and promote soil quality improvements in semiarid Mediterranean vineyards from central Spain, but new questions arises about the influence of these practices on grape yields and must quality as well as acceptability by farmers. At this regard, more research should be carried out to increase scientific knowledge about farmer´s economic influence of CC and the suitability of agri-payments to counterbalance possible loss of incomes.

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1.2.P

SOIL/WATER CONSERVATION PRACTICES IN A SEMIARID DEGRADED HILLSIDE BETWEEN THE DRIEST LAND IN THE WORLD (ATACAMA DESERT) AND THE MEDITERRANEAN ZONE OF CHILE.

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INTRODUCTION

Worldwide, arid and semiarid zones cover about 33% of total land surface and are characterised by high ecological vulnerability. CIREN (2010) reports that 49% (~37 million ha) of Chilean territory display some level of soil erosion, with the arid and semiarid zones showing the highest eroded area. Although these zones represent 26% of the total territory, are concentrated between 30°SL (Coquimbo Region) and 35°SL (Maule Region), where 52% of the area has a xeric, hyper-arid or semiarid water regime. The Mediterranean-type ecosystems that cover these zones represent a transition between one of the driest deserts in the world, the Atacama Desert (north of 28°SL) and the mixed deciduous-evergreen temperate forests, which occur south of 36°SL. In this sense, its transitional climate is characterised by frequent droughts and decreasing annual precipitation rates, which often comes in the form of short bursts of high intensity rainfall, causing rapid saturation of an uncovered soil surface and promoting severe erosion processes. Water shortage is a major constraint and constitutes the more common causes for the failure of vegetation cover, due to the combined effect of high evapotranspiration rate, low cumulative annual rainfall (infiltration) and low water holding capacity of the soils. In particular, Coquimbo Region (40,580 km^2), a generally rugged mountainous territory broken at least by three east-west alluvial systems, has a devastated interfluvial landscape occupied mainly by Aridisols (Casanova et al., 2013). Moreover, considering last century, the regional mining industry and the use of non-sustainable cropping systems (poor soil cultivation and excessive grazing by marginal subsistence communities) affected the regional flora composition, particularly shrubs and grasses, causing also an active desertification.

Under the conditions that prevail in the zone, some results are included from several followed strategies (increase vegetal cover and surface roughness, biomass and organic/synthetic emulsions added to soil, reduction both length and runoff process at hillside) as countermeasures to occurring degradation processes, within a demonstrative experimental area.

METHODS

Corresponding to a coastal interfluvial (Choapa-Limarí rivers) rainfed zone, the experimental site (Figure 1) is a hillside (25% slope gradient and equatorial exposure; 200 x 100 m) with severe soil degradation (rills and gullies), located at the semiarid Coquimbo Region, within an Agricultural Community of subsistence (*Angostura de Galvez*, 31°27´ SL; 71°34´ WL).

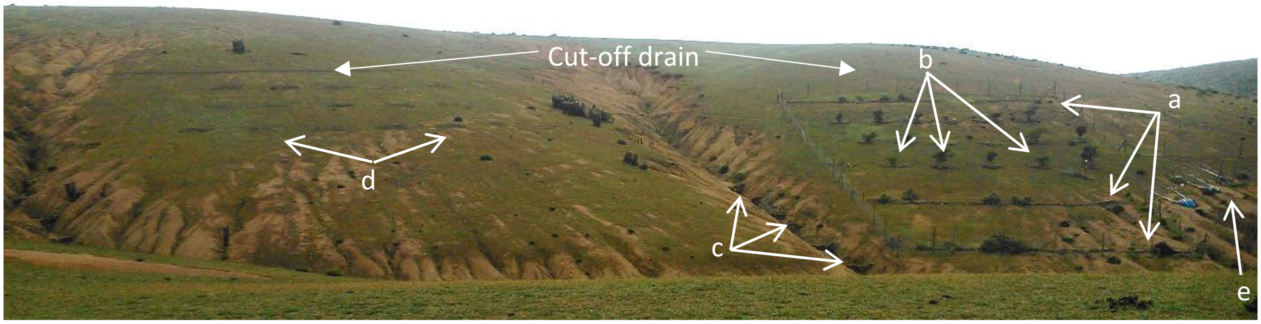

Figure 1. Demonstrative experimental area of soil/water conservation practices. Coquimbo Region, Chile.

Five field essays (Figure 1, a → e) were established to increase vegetal cover (*Atriplex nummularia* and *Acacia saligna*) and surface roughness, to improve surface physical soil condition (goat manure or organic/synthetic emulsions added), and/or to reduce both hillside length and runoff process. According official Chilean methods (Sandoval et al., 2012; Sadzawka et al., 2006) and with three replications, physical (erosion plots/stone-lines) and chemical soil properties (stone-lines/half-moon) were measured. On the other hand, *in situ* soil water contents (FDR probe) were monitored (stone-lines and stony check-dams) and runoff volume and their sediment yields/composition were measured (erosion plots).

RESULTS

The initial characterization of the experimental site (0-20 cm) and the materials applied to the soil highlights a sodic condition (RAS: 13.4) of a low fertility fragile soil (loamy sand), and important organic matter amount, alkalinity and salinity added by the materials. Therefore, sodicity is impairing soil infiltration rates, in addition to raindrops impact, which cause slaking/mechanical breakdown of soil aggregates and dispersion of soil particles, changing pores functionality (sealing) at the soil surface.

a) <u>Three planted (*A. nummularia*) stone-lines, amended with goat manure (+GM)</u>. After four years (2015), this multi-objective system was success in terms of increases vegetal cover (HP and CD), reduces surface roughness/slope length and surface runoff (Table 1). Soil organic matter (SOM) and some total macronutrients (N and K) increased, but also soil salinity with goat manure additions (+GM: 40 Mg/ha) and by abundant dropping leaves to soil (-GM) due to sequesters Na^+ and Cl^- in the plant. In this sense, stone-lines alone are not sufficient to sustain soil productivity if soil and water conservation techniques excludes fertility management practices. Although just trends to improve macro-aggregate stability (Mdv), to increase pores of available water (AW_p) with +GM were observed, with significant increase in fast (Fd_p) and a reduction both in bulk density and slow drainage pores (Sd_p). Considering only average values, water contents monitoring (not included here) during two years shows higher amounts of water in soil depth (W_{30}) than in surface and the effects of *A. nummularia* evapotranspiration. Considering an RI< 1.95 as threshold between wettable and water repellent soils, control (without plants) was hydrophilic and –GM or +GM represents a situation where water infiltration has decreased due to water repellence.

Table 1. Soil properties at stone-lines after four years and organic materials applications.

Treat.	BD	Sd_p	Fd_P	AW_P	SOM	Mdv	RI	HP	CD	W_{10}	W_{20}	W_{30}	Nt	Kt	EC
	Mg/m³	--------------- % ---------------				cm	-	----- m -----		---------- % ----------				mg/kg	dS/m
Control	1.57b	10.0b	15.8b	8.9	1.13	2.10	1,92	-	-	8.65	10.20	15.81	0.07a	199.4a	2.68a
-GM	1.54ab	8.8a	17.6b	5.9	1.53	1.43	2,59	0.81	0.81	10.28	11.06	18.32	0.07ab	199.3a	9.74b
+GM	1.37a	5.4a	22.8a	10.2	2.14	1.15	2,39	0.81	0.91	8.22	7.07	12.01	0.08c	202.7b	8.05b

BD: bulk density; Sd_p and Fd_p: slow (50-10 µm) and fast (>50 µm) drainage pores; AWp: pores of available water (10-0.2 µm); dR: dispersion ratio; Mdv: mean diameter variation; RI: repellence index; HP and CD: height and canopy diameter of plants; W: gravimetric water contents at 10, 20 and 30 cm soil depth. Letters at same column indicate significant differences ($p \leq 0.05$). Standard deviation excluded for space reasons.

(b) <u>Eighteen planted (*A. saligna*) half-moons, amended with liquid emulsions (E) and/or goat (GM) manure</u>. After two years, this multi-objective system was also success increasing vegetal cover, reducing surface roughness/slope length and surface runoff. Table 3 include only chemical soil properties that show significant differences among treatments, evaluated (0-20 cm) during two years.

Particular changes in OM, CEC and pH observed are expressing the characteristics of materials added to soil (Table 2). However, the soil changes were rather temporary in terms of nutrient content, returning quickly to the original levels or even decreasing according the plant uptake. Then, more frequent and regular applications are necessary to increase OM levels in these extreme conditions.

Table 2. Soil chemical properties at planted half-moons amended with organic materials.

Treatment	pH_w		EC		OM		N		CEC	
	2012	2013	2012	2013	2012	2013	2012	2013	2012	2013
	-		----- dS/m -----		----- % -----		---- mg/kg ----		-- $cmol_c$/kg --	
Control	6.3b	7.2a	2.87b	1.42a	0.31b	1.34a	16bc	7a	nd	22b
$E_{100\ L/ha}$	6.5ab	7.2a	1.84ab	1.35a	2.25a	1.43a	7c	5a	nd	36a
$E_{200\ L/ha}$	6.5ab	7.0ab	1.56ab	3.31a	1.21ab	1.19a	18abc	5a	nd	34a
$GM_{15\ Mg/ha}$	6.3b	7.1ab	2.50b	2.77a	1.09ab	1.29a	37ab	11a	nd	27ab
$GM_{30\ Mg/ha}$	6.8a	6.5b	1.06a	2.02a	1.56ab	1.24a	41a	7a	nd	27ab
$E_{100\ L/ha}$+$GM_{15\ Mg/ha}$	6.4b	7.1ab	4.39c	2.55a	1.16ab	1.32a	30abc	9a	nd	37a

pH_w: pH in water; EC: electrical conductivity; OM: organic matter; N: available nitrogen; CEC: cation exchange capacity;. Letters in the same column indicate significant differences ($p \leq 0.05$). Standard deviation excluded for space reasons.

(c) <u>Twelve stony check-dams in gullies filled with graded sediments and amended with goat manure (GM)</u>. This essay was oriented initially to asses soil water dynamics at six depths, when randomized filling (control) is improved with 80 Mg/ha of GM (control+GM) and when materials were deposited in different sequences (C-M-F: coarse-medium-fine; F-M-C: fine-medium-coarse) upstream of stony check-dams (Table 3, Figure 2).

Table 3. Physical properties of materials added upstream of stony check-dams.

Treatment	Depth	Bulk density	Particle density	Fd_P	Sd_p	AW_P	OM
	cm	Mg/m³		---------- % ----------			
Control	0-20	1.65	2.80	23.90	4.76	3.50	0.07
	20-40	1.59	2.83	24.90	4.50	3.20	0.07
	40-60	1.52	2.81	27.00	4.90	0.20	0.06
Control	0-20	1.37	2.50	29.20	3.78	2.10	1.98
+	20-40	1.20	2.57	27.10	2.90	9.60	1.73
GM	40-60	1.48	2.54	27.60	3.30	3.50	1.79
C-M-F	0-20	1.67	2.79	8.90	20.00	2.90	0.04
	20-40	1.54	2.78	8.88	14.10	15.30	0.07
	40-60	1.53	2.60	3.90	9.80	24.90	0.84
F-M-C	0-20	1.57	2.58	6.70	17.70	2.50	0.83

	20-40	1.51		2.68	11.20	13.80	12.90	0.06
	40-60	1.78		2.78	9.22	19.10	4.90	0.06

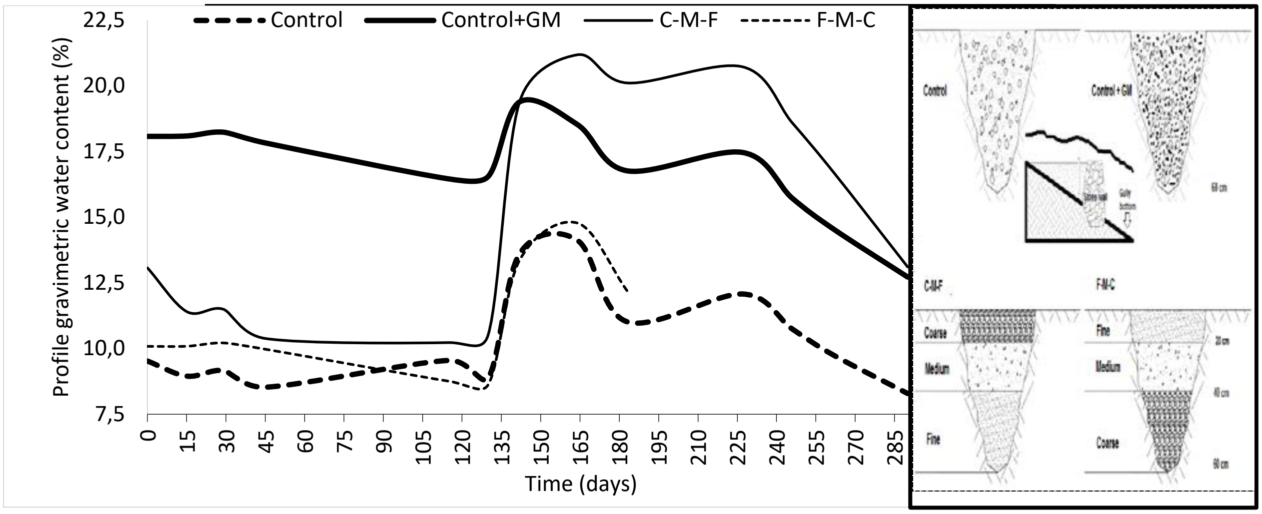

Figure 2. Temporal average water contents in each profile and filling sections sequences (right).

Among others measures, vegetal cover represents an important factor when it is attempt gully control. Our results show that the best and more stable condition to a future vegetal cover establishment was C-M-F, which show more pores of available water and lower amount of fast drainage pores in depth. (d) Fifteen runoff infiltration ditches (3 x 1 m; 0.3 m in depth) amended with emulsion (E) and/or goat manure (GM). Designed to catch all the runoff along hillsides until infiltrate the ground, these retention ditches with earth-bunds in contour are effective only when soil structure at the drainage area is stable. Few intensive rainfall events were enough to fill up with sediments the ditches in our case, highlighting the necessity of complement different soil and water conservation practices. Repellence index (dispersion ratio) values were higher than 1.95 for most conditions and amendment treatments, except inside ditches without amendment (original subsurface soil) and at the earth bounds with GM (40 Mg/ha) added.

(e) Nine erosion plots (4 x 1 m) amended with increased rates of liquid emulsions (E). Although organic amendments have the ability to improve soil physical condition, not significant trend was observed to higher values (runoff and sediment yields), with E applications respect to control (Table 4). At the last year, bulk density increase, micro-aggregates stability show improvement only at highest E rates and the amount of OM increase at the collected sediments. Other measured physical soil properties show erratic trends or not significant (high standard deviation) trends respect to control and are not included here.
Table 4. Particle losses, runoff and surface physical soil properties at erosion plots after four rainfall events.

Rainfall (date)	Treat.	Runoff (L/m²)	Sediment yield (g/m²)	OM in sediments %	Dispersion ratio %	Bulk density Mg/m³
25.6 mm Jun-2013	Control	5.28 ± 0.33a	9.47 ± 0.81a	7.32 ± 0.69a	-	1.89±a
	$E_{100\ L/ha}$	4.01 ± 2.99a	13.69 ± 6.28a	5.89 ± 1.33ab	-	1.96±a
	$E_{200\ L/ha}$	6.23 ± 4.85a	47.64 ± 52.11a	4.23 ± 0.54b	-	1.93±a
46.6 mm Aug-2013	Control	1.10 ± 0.38a	0.22 ± 0.11a	11.35 ± 3.63a	-	1.89±a
	$E_{100\ L/ha}$	1.36 ± 0.46a	0.69 ± 0.46a	5.14 ± 1.64b	-	1.96±a
	$E_{200\ L/ha}$	2.62 ± 2.31a	2.62 ± 2.31a	6.68 ± 0.83ab	-	1.93±a
50.0 mm Jun-2014	Control	4.95 ± 3.75a	4.78 ± 3.82a	4.25 ±0.04a	64.7±ab	1.77±a
	$E_{200\ L/ha}$	11.34 ± 0.69a	26.30 ± 4.29b	5.03 ±0.41a	75.56±a	1.85±b
	$E_{400\ L/ha}$	9.25 ± 6.30a	15.85 ± 11.35ab	4.41 ±0.62a	59.77±b	1.79±ab
93.6 mm Aug-2014	Control	10.00 ± 5.01a	16.10 ± 7.10a	2.84 ± 0.29b	64.7±ab	1.77±a
	$E_{200\ L/ha}$	10.02 ± 5.38a	34.22 ± 17.38a	2.90 ± 0.31ab	75.56±a	1.85±b
	$E_{400\ L/ha}$	19.97 ± 10.42a	18.55 ± 7.96a	5.20 ± 1.55a	59.77±b	1.79±ab

Outside of experimental site, complementing the scarce amounts of available water, a fog-harvesting system was established. In this way, we hope that a more permanent water availability will help to enhance vegetal growth, to improve amendment (calcic) efficiency to resolve sodicity condition and overcome water repellence at studied soils.

CONCLUSIONS

The complex geomorphology, trend towards aridity, high erosivity, high erodibility, high surface runoff and low vegetation cover represent at the site the main factors leaving to an extreme soil degradation.

Despite apparent benefits of organic emulsion adoption to increase organic matter contents, a lack of consistency in the obtained results is observed, when are used in these adverse conditions.

Sodicity and surface water repellence of soils are limiting the most developed soil and water conservation strategies, being success in increase vegetal cover with the resistant plant species, but also in reduce surface roughness/slope length and surface runoff with engineering methods (stone-line/half-moons).

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1.3.P

EFFECTS OF TILLAGE AND SOIL TEXTURE ON SOIL ORGANIC MATTER FRACTIONS UNDER SEMIARID MEDITERRANEAN CONDITIONS

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ABSTRACT

The identification of soil organic matter (SOM) fractions that reflect the management-induced changes in soil organic carbon acquires special interest in semiarid regions where the capacity of soil for agricultural production is limited. The objectives of this study were to evaluate the effect of different tillage and soil management practices on the distribution of C among SOM fractions and determine the influence of soil texture on the protection of soil organic carbon (SOC) in a semiarid Mediterranean region (Aragon, NE Spain). The study was conducted under on-farm conditions where pairs of adjacent fields under long-term no tillage (NT) and conventional tillage (CT) were compared in different cereal production areas. A nearby undisturbed soil under native vegetation (NAT) was also included. Results indicate that clay content was a determinant factor for the easily dispersed mineral-associated OM fraction and the total SOC, indicating that chemical stabilization seems to be a main preservation mechanism in the studied soils. Only the two labile SOM fractions isolated, coarse and fine particulate OM (POM), were sensitive to soil management and their concentrations decreased as soil disturbance increased. The highest differences between NT and CT corresponded to the fine POM in the soil surface where this fraction was 1.2-3 times higher under NT. This relative gain with respect to CT increased in the most arid locations of the region.

INTRODUCTION

Total SOC is not always the best indicator of changes in soil management, especially in semiarid regions where significant increments in SOC are only to be expected after several years. The advances in the separation of SOM into fractions of different composition and stability have allowed differentiate labile and recalcitrant SOC pools and identify those that can serve as early indicators of changes in soil quality (Wander, 2004; von Lützow et al., 2007). However, the literature reveals that the effectiveness of SOC fractions as indicators is variable due to many influential factors and complex stabilization mechanisms (von Lützow et al., 2007; Martín-Lammerding et al., 2013). More research is necessary to develop soil management strategies to increase the quantity and stability of SOM. In Aragon (NE Spain), as in the rest of Spain, the interest of farmers in NT has been increasing (López et al., 2012). However, there is still little available information on SOM fractions in agricultural soils of the region. The objectives of this study were to determine the effect of soil tillage on the distribution of C among SOM fractions and evaluate the influence of soil texture, and other basic soil properties, on SOC protection.

MATERIALS AND METHODS

The study was conducted under on-farm conditions in six different cereal production areas of Aragon where pairs of adjacent fields under long-term NT (9-21 years) and CT were compared. A nearby undisturbed soil under native vegetation (NAT) was also included. Mean annual precipitation in these areas ranging from 350 to 740 mm. All soils were medium-textured soils, alkaline, and generally low in SOC content. Following the procedures of Six et al. (2002) and Plante et al. (2006), soil collected at three depths (0-5, 5-20 and 20-40 cm) was physically fractionated to isolate four SOM fractions: coarse particulate OM fraction (cPOM, >250 µm), fine particulate OM fraction (fPOM, 250-53 µm), easily dispersed mineral-associated OM fraction (d-Min, <53 µm), and mineral-associated fraction occluded within microaggregates (µagg-Min, <53 µm). Organic C content in the four fractions was measured by using a LECO analyser (RC-612 model).

RESULTS AND DISCUSSION

Average total SOC concentration in the 0-40 cm soil layer varied from 10 to 18 g kg^{-1}. The highest contribution to total SOC corresponded to the d-Min-C fraction, accounting for ca. 40-70% of total OC. It was followed by the µagg-Min-C fraction (15-40%) and, finally, by the fPOM-C (6-30%) and cPOM-C (<10%) fractions (Table 1).

The two isolated Min fractions were not consistently affected by soil management. However, soil clay was a determinant factor for d-Min-C and total SOC in both agricultural and NAT soils (Fig. 1). This indicates that chemical stabilization, through clay-organic complexes, seems to be a main OC preservation mechanism in the studied soils. In contrast, the µagg-Min fraction was not correlated with soil texture but it was strongly related with the mass of water-stable microaggregates at either 0-5 or 0-40 cm soil depth (r≈0.900; P<0.0001). This confirms that physical protection provided by microaggregates is also an OC stabilization process in these soils. Considering together all sites and treatments, multiple regression analysis showed that 80-90% of the total variation in the stable microaggregate mass was explained by µagg-Min-C and, to a much lesser extent (10%), by soil clay and $CaCO_3$ contents (Table 2). From these relationships is inferred the importance of SOM, higher than that of clay and $CaCO_3$, as cementing and stabilizing agent of soil aggregates in the study areas.

In contrast to the Min fractions, the POM fractions were influenced by land use and tillage system (Table 1). In the case of the cPOM-C, the percentages were lower (10-63%) and differences between

Table 1. Concentrations of different SOC fractions (g C kg^{-1} soil) at 0-40 cm soil depth as affected by soil management and tillage. In Peñaflor, CC refers to a continuous cereal cropping system and CF to a cereal-fallow rotation. Within the same site and SOC fraction, values followed by the same letter are not significantly different at P<0.05.

		Site					
Soil organic carbon fraction	Treatment	Peñaflor CC	Peñaflor CF	Lanaja	Torres de Alcanadre	Undués de Lerda	Artieda
cPOM-C	CT	0.37 a	0.23 a	0.10 a	0.60 a	0.15 a	1.00 b
	NT	0.44 a	0.27 ab	0.44 b	0.57 a	0.10 a	0.44 a
	NAT	0.65 b	0.65 c	0.53 b	1.64 b	1.19 b	1.31 b
fPOM-C	CT	1.16 a	0.75 a	1.34 a	1.08 a	1.70 a	1.50 a
	NT	1.26 a	0.66 a	1.83 a	1.37 a	1.87 a	1.62 a
	NAT	2.44 b	2.44 b	1.88 a	2.15 b	7.38 b	3.72 b
µagg-Min-C	CT	4.68 b	3.85 a	2.19 a	1.79 a	2.86 a	2.78 a
	NT	3.69 ab	3.90 a	2.19 a	2.36 b	3.34 ab	2.97 a
	NAT	3.25 a	3.25 a	2.15 a	2.59 b	4.07 b	2.45 a
d-Min-C	CT	5.33 a	5.73 a	6.41 a	5.31 a	10.05 a	5.36 a
	NT	5.23 a	5.66 a	6.54 a	5.08 a	9.46 a	5.16 a
	NAT	4.84 a	4.84 a	4.20 b	5.12 a	12.87 a	8.88 b

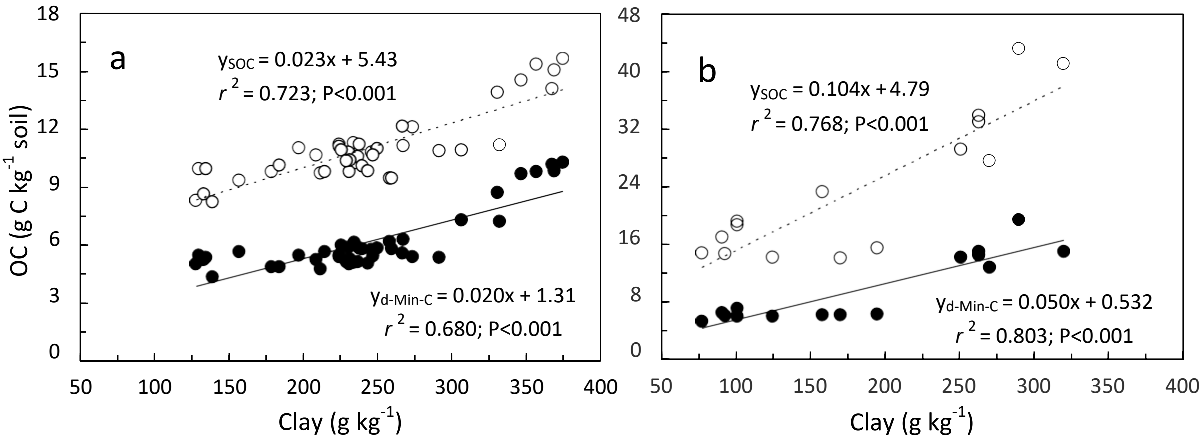

Figure 1. Relationships between the concentrations of total soil organic carbon (SOC, —○—) and easily dispersed mineral-associated organic C (d-Min-C, —●—), and the clay content in agricultural soils at 0-40 cm depth (a) and in natural soils at 0-5 cm depth (b).

agricultural and NAT soils decreased with soil depth, disappearing in the 20-40 cm layer. In the Argentinean pampas, Duval et al. (2013) also found a depletion of fPOM and, especially, of cPOM by cultivation with reductions of 20-70% as compared to NAT soils. In our study, within the cultivated soils, the largest differences among tillage systems were found at the soil surface (0-5 cm) where the fPOM-C content was 1.2-3 times higher under NT than under CT. With the exception of Artieda (farmer removes crop residues from the NT field after harvest), the cPOM fraction followed a similar pattern as fPOM although the tillage effect was less pronounced. In our study conditions, both fPOM and cPOM fractions contributed to the differences among sites based on the relative gain or loss of OC in NT with respect to CT at each site. Figure 2 shows negative and strong relationships between the relative difference in the concentration of both fPOM-C and cPOM-C at 0-5 cm depth and the mean annual precipitation at each site thus indicating that the relative gain of OC with the adoption of NT decreases as precipitation increases. Total SOC was also influenced by precipitation but was much less responsive than POM fractions (r^2=0.588). Our finding on the increase in the relative benefit of NT with reduction in precipitation is supported by Gregorich et al. (2009) and Luo et al. (2010) in Canadian and Australian soils, respectively.

CONCLUSIONS

The d-Min-C fraction provided the highest contribution to total SOC (40-70%) and was not greatly affected by soil management. Soil clay was a determinant factor for this fraction, indicating that chemical stabilization seems to be a main preservation mechanism of OC in the studied soils. In contrast, soil texture did not influence the µagg-Min-C (15-40% of total SOC). The strong relationship

Table 2. Optimum regression equations to estimate the mass of water-stable soil microaggregates (sand-free basis; $Mass_{\mu agg}$, g kg^{-1} soil) as a function of µagg-Min-C (g C kg^{-1} soil) and clay and $CaCO_3$ contents (g kg^{-1}).

Soil depth cm	Equation	r^2	*P*	n
0-5	$Mass_{\mu agg}$= 13 + 56 µagg-Min-C - 9076/clay + 0.121 $CaCO_3$	0.841	<0.0001	55
0-40	$Mass_{\mu agg}$= 43 + 53 µagg-Min-C - 8709/clay + 0.120 $CaCO_3$	0.899	<0.0001	55

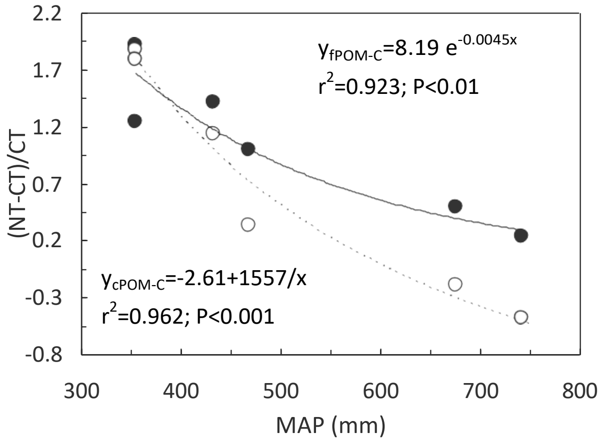

Figure 2. Relative difference between no tillage (NT) and conventional tillage (CT) in fine (fPOM-C, —●—; g C kg^{-1} soil) and coarse particulate organic C (cPOM-C, —○—) at 0-5 cm soil depth as a function of mean annual precipitation (MAP).

found between this fraction and the mass of water-stable microaggregates indicated that physical protection is also an OC stabilization process involved in these soils. Despite their small contributions to total SOC, the two POM fractions isolated were sensitive to soil management and their concentrations decreased as soil disturbance increased (NAT>NT>CT). The highest differences between NT and CT corresponded to fPOM and were generally restricted to the soil surface where this fraction was 1.2-3 times higher under NT. This relative gain with respect to CT increased in the most arid locations of the agroclimatic gradient.

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1.4.P

GREENHOUSE GASES BALANCE IN AN ACRISOL UNDER NO-TILLAGE CROPPING SYSTEMS

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INTRODUCTION

The atmospheric concentrations of the three main greenhouse gases (GHGs) – carbon dioxide (CO2), nitrous oxide (N2O), and methane (CH4) – have increased rapidly in the last few decades, which is a phenomenon associated to anthropogenic activities. Worldwide, it is estimated that agriculture contributes with approximately 22 % to the total CO2 emissions, 80 % to the total N2O and 55 % to the total CH4 emissions (IPCC, 2007).

Most N2O is produced by nitrification and denitrification processes in the soil. Nitrification, which requires aerobic conditions, depends on NH4+ supply and is mediated by autotrophic bacteria, whereas denitrification is performed by anaerobic heterotrophic bacteria, which depend on the availability of labile organic carbon (C) and NO3-. Their activity is intensified in anaerobic environments or soil microsites with low O2 availability (Aita & Giacomini, 2007). In addition to these soil variables, temperature, water-filled pore space (WFPS), and pH have also been identified as N2O production-controlling variables in agricultural soils, since they affect the activity of nitrifying and denitrifying bacteria as well.

Conservation management systems, especially no-tillage (N-T), increase soil C stabilization and may be an important alternative to increase atmospheric CO2 drainage capacity and mitigation of global warming, especially when associated with high-input crop systems of plant residues to the soil (Bayer et al., 2006). The use of legumes in crop systems associated to N-T increases the nitrogen (N) stocks in the soil, due to the higher N contribution by the biological fixation and to the lower mineralization rate of the organic N in this type of system.

Considering that the South of Brazil has a large extension of land destined to intensive agriculture, this study aimed to evaluate the emissions of N2O, CH4 and the global warming potential (GWP) in conservationist systems of soil management, identifying the impact of crop systems on GHG emissions.

METHODS

The study was carried out in a long-term field experiment at the Agronomic Experimental Station of the Federal University of Rio Grande do Sul near Eldorado do Sul in Rio Grande do Sul State, Southern Brazil (30o06'S; 51o4 W, about 46 m altitude). The local climate is subtropical humid (Cfa type, according to

Köppen classification), with annual mean temperature of 19.2oC, mean monthly temperature varying from about 9oC in the winter to 30oC in the summer, and annual mean rainfall of 1440 mm. The soil was classified as an Alumic Acrisol by FAO Legend (FAO, 2006). The soil-particle size distribution was 540 g kg-1 sand, 240 g kg-1 silt, and 220 g kg-1 clay, with the clay fraction composed mainly of kaolinite (720 g kg-1) and iron oxides (109 g kg-1 of Fe2O3).

Two studies were performed in long-term (24 and 26 years) experiments under N-T and without addition of mineral N in an Acrisol, and aimed to evaluate the effect of soil management that include legumes and grasses with maize in succession [oat/maize (O/M), vetch/maize (V/M), oat+vetch/maize (O+V/M), oat+vetch/maize+cowpea (O+V/M+C) and lablab+maize (LL+M)], on GHG emissions in Southern Brazil.

Air samples were collected in aluminum static rectangular chambers (Figure 1) and nitrous oxide (N2O) and methane (CH4) concentrations determined by gas chromatography. Meteorological variables (temperature and moisture) and soil parameters (NO3-, NH4+ and dissolved organic C content) were determined at 10 cm depth. Net global warming potential (GWP) for the management systems was calculated by accounting the annual N2O and CH4 fluxes, annual C retention rates and the C costs of agronomic inputs.

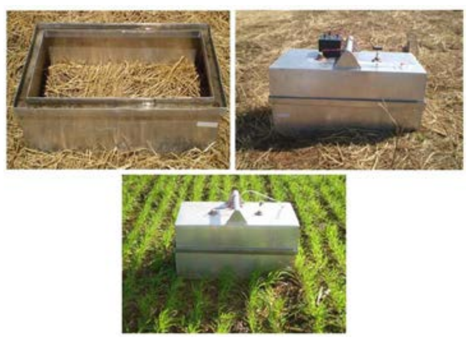

Figure 1. Air samples collected in static rectangular chambers

RESULTS

Study I evaluated the effect of cropping systems on short-term GHG emissions in the period after winter cover crop management (post-management) (2009/10 and 2010/11). Study II evaluated GHG emissions from cropping systems on annual basis and calculated the net global warming potential (GWP). Cumulative N2O emissions (Figure 2) in the post-management period 2009/10 (2,86±0,43 kg N ha-1) were nine times higher than cumulative emission observed in 2010/11 (0,32±0,08 kg N ha-1).

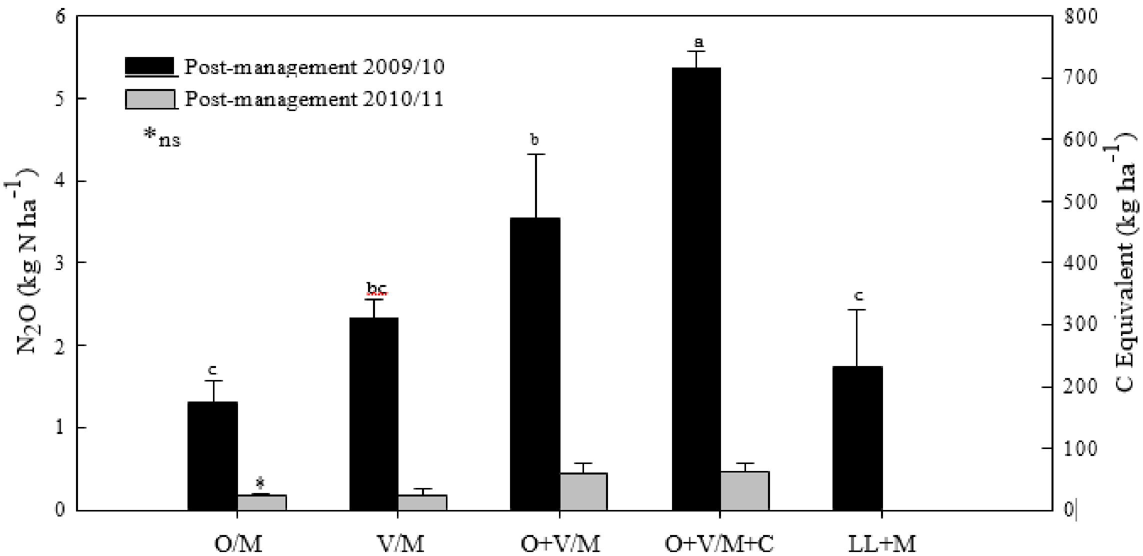

Figure 2. Cumulative emission of N2O (kg ha-1) in the post-management periods 2009/10 and 2010/11 in cover-crop systems (O/M: oats/maize, V/M: vetch/maize, O+V/M: oat+vecth/maize: O+V/M+C: oats+vetch/maize+cowpea, LL+M: lablab+maize) under no- tillage.

The results of the soil N2O emissions obtained in this study were higher than those observed by Gomes et al. (2009) (~ 83.40 µg N-N2O m-2 h-1) and Zanatta (2010) in the same experiment (~ 514, 60 µg N-N2O m-2 h-1), as well as in Japan (~ 567.02 µg N-N2O m-2 h-1), USA (~ 197.08 µg N-N2O m-2 h-1) (Ussiri et al. , 2009) and the United Kingdom (~ 52.10 µg N-N2O m-2 h-1) (Pappa et al., 2011). However, they were similar to those reported for management systems in Brazil (~ 670 µg N-N2O m-2 h-1) (Escobar et al., 2010).

Soil N2O emission in treatments that included vetch (V/M, O+V/M and O+V/M+C) was enhanced by frequent rains after winter cover crop management, which delayed corn sowing in 81 days and maintained higher levels of soil N compared to treatment with oat (O/M). N2O emissions were controlled mainly by water filled pore space (WFPS) and microbial activity (CO2), indicating denitrification as the main process involved in N2O production (Figure 3).

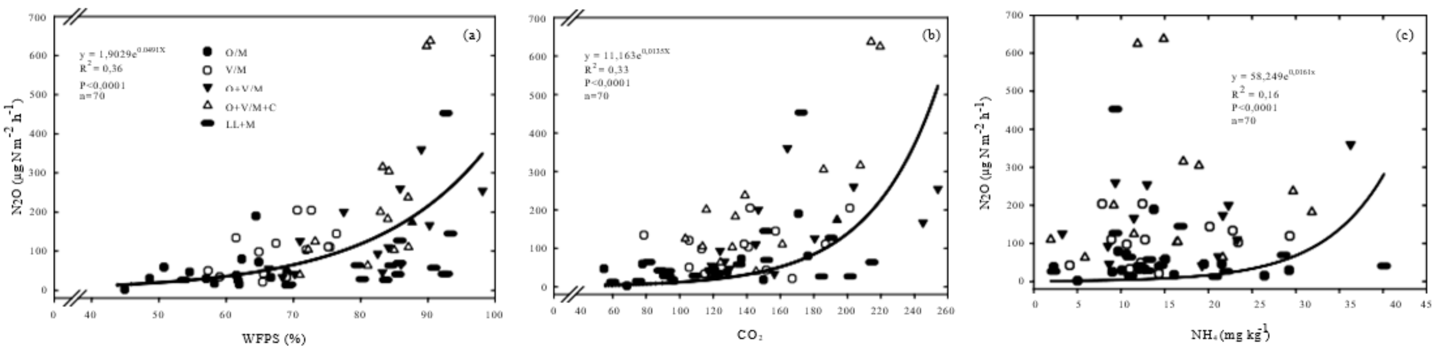

Figure 3. Relationship between the daily emission of N2O (µg m-2 h-1) with (a) the water filled pore space (WFPS), (b) the biological activity (CO2) and (c) ammonium (NH4+) of the soil in the post-management periods 2009/10 and 2010/11 in cover-crop systems (O/M: oats/maize, V/M: vetch/maize, O+V/M: oat+vecth/maize: O+V/M+C: oats+vetch/maize+cowpea, LL+M: lablab+maize) under no-tillage.

The mean annual retention rate of C-CO2 in the soil varied from net emission to the atmosphere of 45.06 ± 33.07 kg ha-1 year-1 to net retention of -1047.22 ± 144.73 kg ha-1 year-1 in soil. The C- CO2 retention rate in soil in crop systems that used legumes (vetch and lablab) as a cover crop, represented an important source of mitigation of global warming. However, the high N2O emissions and the low values of CH4 absorbed by the soil, recorded during the harvest, caused the evaluated crop systems to act as a source of GHG for the atmosphere, as evidenced by the positive GWP, with the exception of the LL + M system (Figure 4). It is necessary to include management strategies to mitigate N2O emissions. High N2O emissions in cropping systems due to prevailing weather conditions during the post-management period 2009/10 had a positive reflection in the GWP.

The relative GWP ranged from -501.22 ± 155.29 kg Ceq ha-1 year-1 (A + V / M + C) to -1700.41 ± 112.12 kg Ceq ha-1 year-1 (LL / M) (Figure 4), values higher than those found by Gomes (2009) for this same experiment (Bayer et al., 2016). The results indicated that agricultural land use under conservation systems, in addition to food production, had a positive impact on the mitigation of the radiative forcing of the atmosphere compared to the traditional system used (PCA / M) in the region

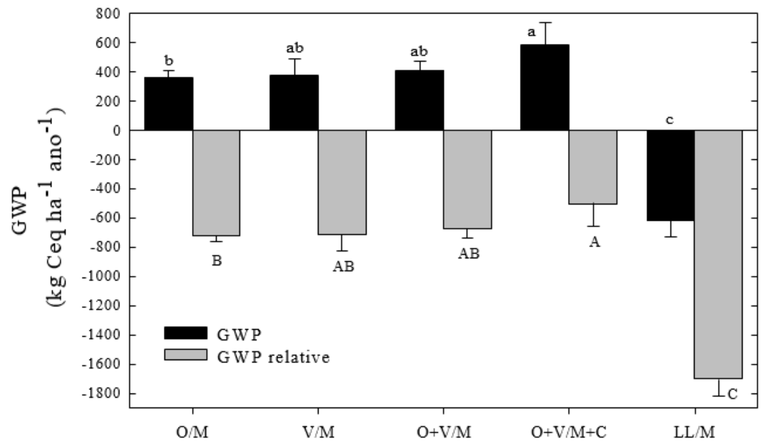

Figure 4. Net global warming potential (GWP) and relative GWP of the soil in the post- management period 2009/10 in cover-crop systems (O/M: oats/maize, V/M: vetch/maize, O+V/M: oat+vecth/maize: O+V/M+C: oats+vetch/maize+cowpea, LL+M: lablab+maize) under no-tillage.

CONCLUSIONS

The dynamics in the emission of nitrous oxide in crop systems is related to the water filled porosity, indicating that the main process involved in the production of this gas in the soil was denitrification.

The use of legumes as a cover crop promoted an increase in the emission of nitrous oxide from the soil in comparison to a system exclusively with grass, which made the global warming potential larger.

The high emissions of nitrous oxide from the soil in cropping systems due to excess rainfall were reflected in the values of positive global warming potential (net emission of GHG to atmosphere).

The soil carbon retention rate in the no-till system of lablab counterbalanced the registered nitrous oxide emissions. The greenhouse gas emission intensity analysis of the management systems indicated that, even in the meteorological conditions in which this study was developed, management systems that involve the use of no-tillage and legumes make it possible to produce food without any contribution to global warming, compared to traditional management systems with conventional tillage.

ACKNOWLEDGEMENTS

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1.6.P

CARBOSOL DATABASE: A RELEVANT TOOL FOR UNDERSTANDING CARBON STOCKS IN SOILS OF SPAIN

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ABSTRACT

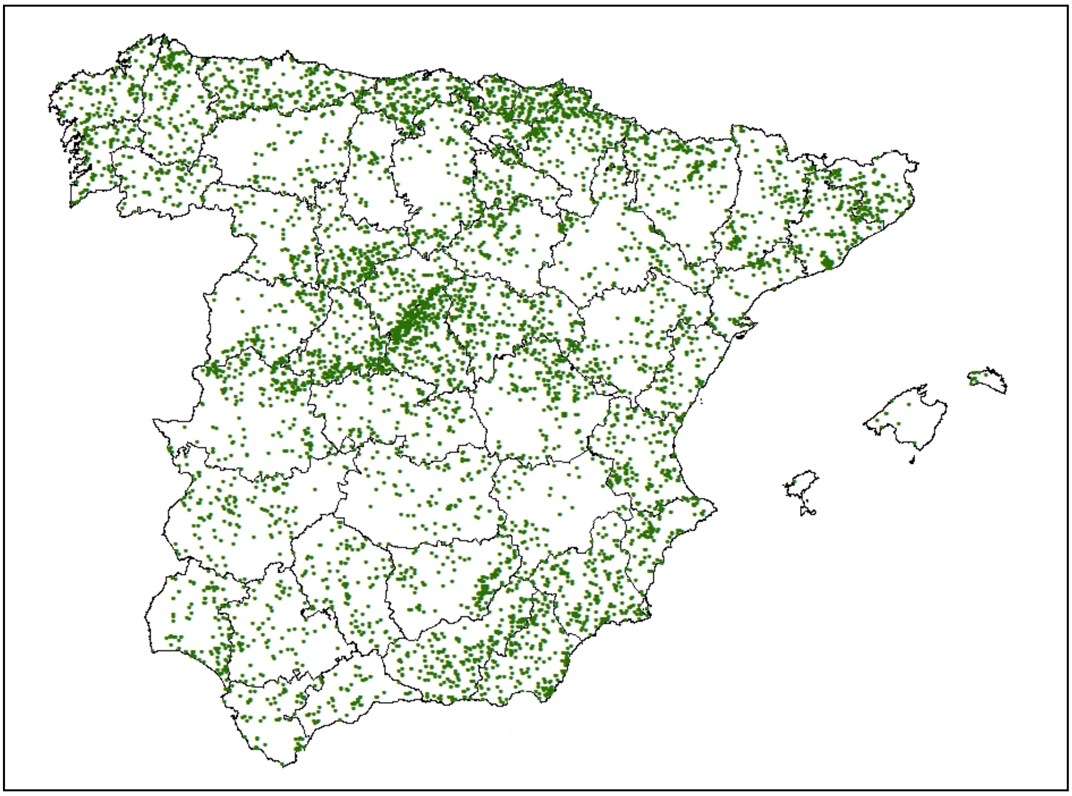
CARBOSOL is a collaborative network of 8 research teams from Spanish Universities and Research Centres focused on the study of soil organic matter and the Global Carbon Cycle. Our aim is to perform an accurate quantitation of Soil Organic Carbon (SOC) stocks in Spain for different land uses, soil types and depths, and assess the environmental drivers of C storage. Furthermore, the compilation and processing of large-scale data sets is crucial for the modelling of C stocks.

Over time, researchers in Spain have generated vast information about soil profiles. However, SOC data often cover limited areas and there are few available studies covering all climatic areas and land uses. Besides, this information is scattered, not easily available and presented in many different formats. The organization of Spanish soil profiles information in a wide, unique and harmonized database is an essential tool for improving our knowledge on Spanish soil properties and dynamics. It is important to point out that the Iberian Peninsula presents a characteristic spatial and temporal variability based upon a diverse geography and a variety of climates. Heterogeneous landscapes offer a natural playground to understand factors affecting SOC, while assessing overall country stocks (Doblas-Miranda et al. 2013). Under the different Mediterranean forest types climate and vegetation control SOC sequestration, while the effect of texture is less pronounced (Chiti et al. 2012).

In collaboration with soil experts, CARBOSOL has compiled soil profile data from 635 sources (published and unpublished studies). Detailed information from 6,610 geo-refereed profiles linked to 22,105 analytical horizons is now available. It represents largest harmonized soil information collection in Spain.

The geographical scope of the database is wide, covering the whole country (Figure 1). Soil carbon stocks vary as a function of soil texture, landscape position, drainage, plant productivity and bulk density, all of which vary spatially, promoting heterogeneity, and hampering the study of temporal changes in soil carbon stocks (Cambardella et al. 1994).

Figure 1. Distribution of soil profiles in Spain. Lines indicate provincial limits.



CARBOSOL Database provides a large amount of information on soil organic matter contents and its associated driver factors: such as soil type, lithology, topography, land use or management. This database is conceived as a public tool for scientific and policy purposes that will allow a comprehensive and more accurate analysis of the current organic C stocks for each soil group in the different land use types in Spain. It will also provide information about the importance of environmental and land-use factors for C sequestration capacity.

Strict criteria have been defined for accepting profiles into the CARBOSOL Database: a) completeness and reliability of data; b) traceability of source of data; c) geo-referencing; d) association of profiles to analytical horizon description, including the measurement of organic carbon of most horizons. The design of CARBOSOL database was governed by the aim of providing easy access to harmonized data. The simplest structure was adopted for the CARBOSOL database. This unsophisticated approach was adopted to encourage the use of the data and facilitate users not trained in database management. As for the main file storage format, the MS-Excel 2016 format was chosen, as this is compatible with most geographic information systems, database management systems, spreadsheets and statistical software packages. A schematic overview of the data model used for CARBOSOL database structure is given in Figure 2.

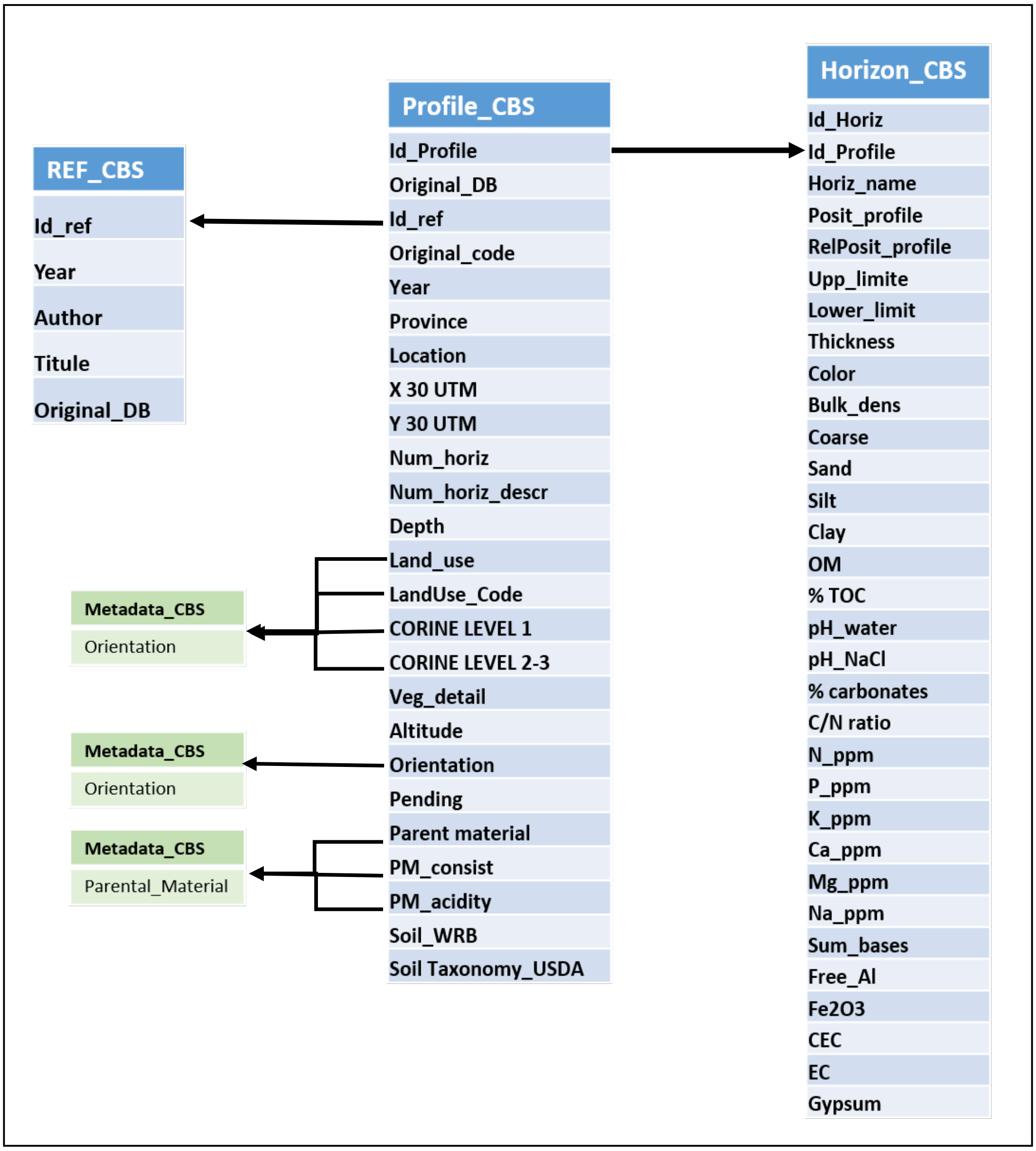


Figure 2. Schematic Data Model for Soil Profile Analytical Database of Spain CARBOSOL.

CONCLUSIONS

CARBOSOL Database seeks to provide a public useful tool to improve knowledge about soils from Span with emphasis in soil carbon stocks and its driver factors.

The first stage of CARBOSOL Network Project highlights the enormous and varied information generated in our country about soils. However, this information is scattered and presented in very variable formats. That's why compiling and harmonizing the soil information is being a laborious and time consuming task. So CARBOSOL Database, in spite of being the most extensive and complete database of soils generated in Spain until now, is now just picking up a part of the total available information. We hope that future projects and further collaborative task will enlarge CARBOSOL Database improving also its quality and usefulness. With this in mind we pay efforts in trying to structure the database in an easily way for handling and understanding using a popular and versatile software. For future work on CARBOSOL Database we recommend focussing attention in the incorporation of information with preference in the proportionally under represented regions and land uses as well as to the incorporation of the most recent information (after 2006). Also, we strongly recommend to stress in revising and providing accurate coordinates in pursuing a robust and quality geo-referenced system of soil profiles that may became an upstanding tool both, for historical studies of biogeochemical changes in soils as well as for the correct management and integration of the information in GIS.

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1.9.P

EFFECT OF BIOCHAR APPLICATION ON SOIL QUALITY AND SOIL CARBON SEQUESTRATION IN ACID SOILS

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ABSTRACT

The quality of soils with biochar application was assessed in acid soil-based areas. The objectives were soil improvement, increasing carbon sequestration and increasing organic crop yields The location was conducted in Lom Kao soil series, Bung Sampan district, Petchaboon province between 2015-2016. Biochar was produced from rice husk and crop yield was organic baby corn. Randomized Complete Block Design (RCB) was applied for 3 replications and 5 treatments. Treatments were control plots (T_1), 3 tons per hectares of biochar application (T_2), 6 ton per hectares of biochar application (T_3), 12 tons per hectares of biochar application (T_4), 18 tons per hectares of biochar application (T_5). Results showed that soil properties had changed as follows: the soil had the mean pH value of 7.1; the amount of organic matter accounted for 2.27%; and the amount of magnesium in the soil accounted for 507.8 milligrams per kilogram. The amount of phosphorus, potassium and calcium increased distinctly with the mean values of 140.2, 287.2 and 7,928 milligrams per kilogram respectively. Moreover, it was found that the amount of actinomycetes and fungi had a tendency to increase as follows: for applying at the different rate of biochar application the amount of actinomycetes bacteria and fungi had changed. After the end of the experiments, it was found that every treatment have an effect on making MWD have a statistical difference. With biochar application for 18 tons per hectare, there was the MWD of 4.11 millimeters. This was followed by biochar application for 12, 6 and 3 tons per hectare whereby there were grain sizes with a MWD accounting for 4.05, 4.02 and 3.53 respectively. For the control plots, there was the smallest MWD with a diameter of 3.06 millimeters. For the percentage of carbon, the highest percentage of carbon in soil equal to 1.47. This was followed by using biochar for 12, 6 and 3 tons per hectare with the percentage of carbon in soil equaling to 1.15, 1.03, and 1.02 respectively. The experiment results also showed that biochar application at different rates for soil amendment had a tendency to increase productivity of baby corn yield whereby using biochar for 18 tons per hectares resulted in highest yields of baby corn equaling to 5,383.9 kilograms per hectares which did not have any statistically significant difference with using biochar for 12 and 6 tons per hectares which yielded 5,457.9 and 5,193.8 kilograms per hectares, respectively.

KEYWORDS: *biochar, acid soils, aggregate size distribution, Soil Carbon Sequestration*

INTRODUCTION

Biological charcoal or biochar is porous charcoal obtained from biomass. To obtain biochar is done by putting this biomass into a octagonal tank that can generate heat with very high temperature (higher than 500 degree Farenheit). Then this biomass goes through biodegradation in presence of chemical heat, which is called pyrolysis. After going through this process within a few hours, this biomass is converted into a coal-like round object, which farmers can use as a soil amendment (Syuhada et al.,, 2005). At present, biochar is an alternative way to solve the problems of the environment such as energy production, food products and global warming reduction. Moreover, this helps for biodegradation increment and carbon sequestration in soils.

To develop charcoal burning is a technology that can be conducted from the level of farmers to the level of industries. Lehmann and Joseph (2009) collected research work related to properties of biochar. It was found that biochar had a property of neutrality to alkalinity. It is porous, can hold water and has a composition of elements such as phosphorus, potassium and calcium etc. while Gul et al., 2015 also collected research work related to biochar. Their findings indicated that biochar can absorb nutrients well, can withstand biological and chemical decomposition and promote activities of microorganism that are useful to soil more. In addition to this, biochar usage is another way to reduce the emission of carbon to the atmosphere, which is the cause of greenhouse effect. Therefore, from the data and the hypothesis as mentioned, biochar is suitable for agricultural application. The objective of this research is to study the effects of biochar on soil properties physically, chemically and biologically in the area of acid soils. To study the amount of carbon in soil from using biochar as soil amendments. The location is Tumbon Nong Jaeng, Bung Sam Pan Amphoe, Phetchabun province.

METHODS

Selected acid soil area in Tumbon Nong Jaeng, Bung Sam Pan district, Phetchabun province which was in the Lom Kao soil series (Lk). They have low fertility and the soil reaction is slightly acid to strongly acid.

Design experimental plot is Randomized Complete Block Design (RCBD) which consists of 5 treatments as following;

1. Control (Conventional Practices, no biochar)
2. Biochar from rice husk 3 tons per hectare
3. Biochar from rice husk 6 tons per hectare
4. Biochar from rice husk 12 tons per hectare
5. Biochar from rice husk 18 tons per hectare

Results

Part I. Plant growth and Soil analysis:

Table 1 Changes of chemical properties of soil at the depth of 0-15 centimeters after the end of the experiments

Treatment	pH	%OM	Avail. P_2O_5	Extr K_2O	Extr. Ca	Extr. Mg
Control	6.97	1.65	85	214	2890	451
Biochar 3 t ha^{-1}	6.97	1.93	97	222	8865	492
Biochar 6 t ha^{-1}	7.71	2.13	149	310	9129	589
Biochar 12 t ha^{-1}	7.00	2.41	151	324	9339	467
Biochar 18 t ha^{-1}	7.2	3.22	219	366	9417	540

Results showed that soil properties had changed as follows: Soil sampling (composite sample) before the experiments was conducted at the depth of 0-15 centimeters. From the results of analyzing chemical properties of soil, it was found that the pH value was 5.9, which showed that soil condition was averagely acidic. The amount of organic matter in the soil was rather low, which accounted for 1.05%. There was an average amount of useful phosphorus and potassium in the soil, accounting for 13 and 60 milligrams per kilogram. For the amount of calcium, it was at the low level of 849 milligrams per kilogram and the amount of exchangeable magnesium was at the average level of 348 milligrams per kilogram. When the first harvest was conducted and at the end of the experiments, it was found that chemical properties of soil had changed. The soil had the mean pH value of 7.1; the amount of organic matter accounted for 2.27%; and the amount of magnesium in the soil accounted for 507.8 milligrams per kilogram. The amount of phosphorus, potassium and calcium increased distinctly with the mean values of 140.2, 287.2 and 7,928 milligrams per kilogram respectively (Table 1).

2. Biological changes of soil

Table 2: Changes of biological properties of soil after harvesting

Treatment	actinomycetes	Bacteria	Fungi
Control	$1.5x10^6$	$1.6x10^5$	$5.5x10^4$
biochar 3 t ha^{-1}	$1.9x10^6$	$2.1x10^5$	$2x10^5$
biochar 6 t ha^{-1}	$1.2x10^6$	$2.8x10^5$	$1.7x10^4$
biochar 12 t ha^{-1}	$6.1x10^5$	$3.4x10^5$	$4.5x10^4$
biochar 18 t ha^{-1}	$1.6x10^6$	$1x10^6$	$2.3x10^4$

Moreover, it was found that the amount of actinomycetes and fungi had a tendency to increase as follows: for biochar application of 3 tons per hectares with the amount of actinomycetes, bacteria and fungi equaling to 1.9×10^6, 2.1×10^5 and $2x10^5$cells per gram of dry soil respectively; for biochar application of 18 tons per hectares, there was the amount of actinomycetes bacteria and fungi equaling to $1.6x10^6$, $1x10^6$ and 2.3×10^6 cells per gram of dry soil (Table 2).

3. Physical properties changes of soil

Table 3: Changes of physical properties of the soil

Treatment	Aggregate size distribution (%)					MWD (mm)
	> 2 mm	1-2 Mm	0.5-1 mm	0.25-0.5 mm	0.105-0.25 mm	
Control	49.46	19.36	12.47	7.99 b	6.66	3.06 b
biochar 3 t ha^{-1}	49.36	19.37	12.09	7.96 b	6.69	3.53 b
biochar 6 t ha^{-1}	49.67	19.28	12.19	8.13 b	6.71	4.02 a

Treatment	Aggregate size distribution (%) > 2 mm	1-2 Mm	0.5-1 mm	0.25-0.5 mm	0.105-0.25 mm	MWD (mm)
biochar 12 t ha^{-1}	50.26	19.48	12.48	8.26 ab	6.79	4.05 a
biochar 18 t ha^{-1}	50.19	19.97	12.91	8.58 a	6.80	4.11 a
p<0.05	ns	ns	ns	*	ns	*
CV (%)	2.21	2.63	2.60	2.65	2.52	12.82

After the end of the experiments, it was found that aggregate size distribution (%) is increased after the rate of biochar increased. Also the treatment have an effect on making MWD have a statistical difference. With biochar application for 18 tons per hectare, there was the MWD of 4.11 millimeters. This was followed by biochar application for 12, 6 and 3 tons per hectare whereby there were grain sizes with a MWD accounting for 4.05, 4.02 and 3.53 respectively. For the control plots, there was the smallest MWD with a diameter of 3.06 millimeters (Table 3).

4. Carbon sequestration (% of carbon in the soil)

Table 4: Percentage of carbon in the soil

Treatment	Soil carbon (%)		
	crop 1	crop 2	average
Control	0.65	0.96 a	0.80 a
biochar 3 t ha^{-1}	0.92	1.12 a	1.02 a
biochar 6 t ha^{-1}	0.83	1.24 b	1.03 a
biochar 12 t ha^{-1}	0.90	1.40 b	1.15 ab
biochar 18 t ha^{-1}	1.07	1.87 b	1.47 b
p<0.05	ns	*	*

Note The average that is followed by the same letter means that it is not significantly different at the 95% confidence level by using DMRT method.

Before the experiments, soil carbon was obtained from composite sample, which accounted for 0.65%. After the end of the experiments, it was found that the percentage of soil carbon tended to increase in every treatment with different rates of biochar application. For the biochar application of 18 tons per hectare, there was the highest average soil carbon, accounting for 1.47% (Table 4).

5. Yield of corn (kilograms per hectare)

Table 5: Yield of corn (kilograms per hectare)

Treatment	yield (kilograms per hectare)		
	crop 1	crop 2	average
Control	4,840.6	5,055.6 a	4,948.1 a
biochar 3 t ha^{-1}	4,994.7	5,066.8 a	5,030.8 a
biochar 6 t ha^{-1}	5,177.7	5,210.2 ab	5,193.9 a
biochar 12 t ha^{-1}	5,333.6	5,582.3 b	5,457.9 b
biochar 18 t ha^{-1}	5,262.3	5,605.2 b	5,383.8 b
$p<0.05$	ns	*	*
CV (%)	10.47	9.91	10.87

Note The average that is followed by the same letter means that it is not significantly different at the 95% confidence level by using DMRT method.

From the study on the average productivity of baby corn twice, it was found that biochar application at different rates for acid soil amendment tended to make the productivity of baby corn different whereby the treatment with biochar application for 12 tons per hectare yielded the most average productivity of baby corn, accounting for 5,457.9 kilograms per hectare. This was followed by biochar application for 6 and 18 tons per hectare, giving similar average yields of 5,193.9 and 5,383.8 kilograms per hectare respectively. And non-biochar application gave the lowest yield of 4,948.1 kilograms per hectare (Table 5).

CONCLUSION

Applying biochar for agricultural area can be benefit to land due to its properties. It can change chemical biological and also physical properties in soil. For carbon sequestration in the soil at the end of the experiments, it was found that biochar application at different rates tended to increase carbon sequestration in the soil whereby it varied according to the amount of biochar application. Biochar application at different rates in soil amendment of acid soil tended to increase the productivity of baby corn whereby with biochar application for 12 tons per hectare, baby corn yielded the most, accounting for 5,457.9 kilograms per hectare. This was not significantly different with biochar application for 18 and 6 tons per hectare, yielding the productivity of 5,193.9 and 5.383.8 kilograms per hectare respectively. Therefore, to grow organic baby corn in the area of acid soils in Lom Khao soils series Petchaboon, apply the biochar at the rate of 12 tons per hectare would be recommeneded.

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1.10.P

ADAPTATION STRATEGIES TO CLIMATE CHANGE FOR DELIVERING SOIL SERVICES IN MODEL AREAS OF AFRICA

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INTRODUCTION

Africa, due to physical and socioeconomic reasons, shows a high vulnerability to climate change. The CLIMAFRICA Project (EU, 2010-14)[1] aimed at analyzing the impacts of climate change in Sub-Saharan Africa (SSA) through different perspectives and across multiple scales. These included climate forecasting and impacts on water resources and agriculture, socio-economic analysis of such impacts and suggesting measures for climate change adaptation planning. Case studies represented the most detailed scale studies of the Project. The objectives were to make a diagnosis of the soil quality (physical and chemical), to predict its evolution under a downscaled climate forecast, and to propose adaptation measures for soil conservation. The approach involved collecting field data on ecological and socio-economic aspects at local level. These data were used to empirically assess local-scale vulnerability to climate change in seven study areas of Sub-Saharan Africa using a multidisciplinary case study approach.

Materials and methods

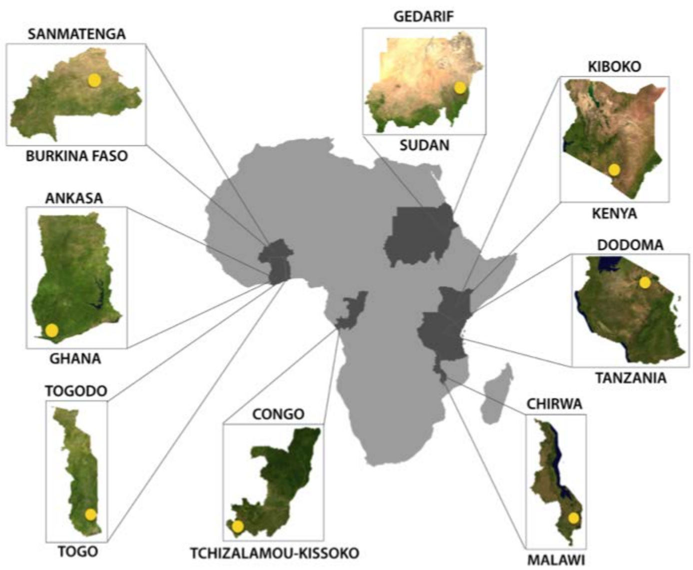

Fig 1. Location of the case studies

Soil quality parameters from seven case studies in Africa (Burkina Faso, Congo, Kenya, Ghana, Malawi, Sudan, Tanzania and Togo (Fig 1)), with annual rainfall from 400 to 2000 mm, were determined using a standardized sampling and survey framework of CLIMAFRICA EU project. Soil samples from 0-10 and 10-30 cm depth from land-use and agricultural practices were sampled. Soil physical quality parameters determined were texture, bulk density and stone content. The soil chemical parameters were pH, EC, Organic Carbon, Nitrogen, Phosphorous, Cation Exchange Capacity and Exchangeable cations.

[1]http://www.climafrica.net/index_en.jsp

RESULTS

The soils were neutral to acidic, with low base saturation and low P (except those from the Sudan site, calcareous and with high clay content), Organic Carbon (OC) contents around 1%, low available water capacities and local compaction problems (Burkina Faso). Soil organic carbon stocks (SOCS) are low, ranging between 1 and 2 kg OC $m^{-2}$for the 10 cm soil depth, and between 2 and almost 6 kg OC m^{-2} for the 30 cm soil depth. The highest values –also the most variable- correspond to the case study of Kenya, particularly soils under natural savannah or grassland (Fig. 2).

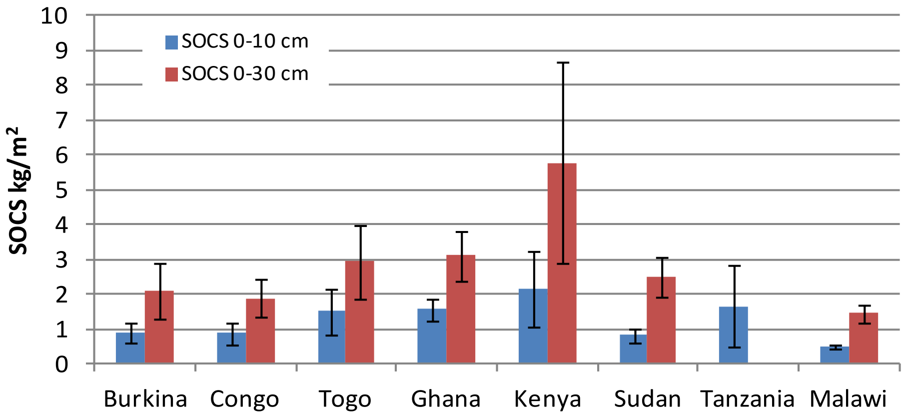

Fig. 2. Soil Organic Carbon Stocks (SOCS) of the soils in the study areas.

The results of SOCS and land-use for the Togo (West Africa), Kenya (East Africa) and Sudan (North Africa) sites are presented in Table 1. The results show SOCS from 0 to 10 cm and 0 to 30 cm depth grouped according to different land-uses. SOCS were higher in forest than in savannah or agricultural land-use in Togo sites. The two latter land-uses yielded soil carbon stocks that were not significantly different at neither 10 nor 30 cm depth.

Table 1. Soil Organic Carbon Stocks (SOCS) of selected sites. Significance: 10%

			SOCS0-10 cm (kg C m^{-2})		SOCS0-30 cm (kg C m^{-2})	
		n samples	Average	Std. Error	Average	Std. Error
Kenya	Savannah	6	2.25 a	0.41	5.44 a	0.63
	Grassland	6	1.62 ab	0.74	4.80 ab	2.21
	Shrubland	6	0.60 ab	0.39	1.60 ab	1.07
	Crops	6	0.22 b	0.14	0.68 b	0.43
Sudan	Savannah	6	0.37b	0.17	1.17b	0.53
	Rangeland	6	0.65ab	0.23	1.88ab	0.64
	Fallow	7	0.86a	0.07	2.56a	0.16
	Crops	31	0.79a	0.04	2.44a	0.11
	Other	20	0.84 a	0.04	2.53 a	0.12
Togo	Fallow and cereal	8	0.82 b	0.24	1.90 b	0.51
	Deciduous forest	8	1.92 a	0.20	3.38 a	0.33
	Savannah	8	0.31 b	0.21	0.61 b	0.45

Soils of Sudan savanna had the lowest SOCS values, while the rest of land-uses showed no significant differences. Contrarily, the studied Kenyan savannahs had higher SOCS than soils used for crop production.

The Congo data (Fig. 3) showed a strong and significant relation between land-use and SOCS. Soils in secondary forest contain more soil organic carbon, especially in the top 60 cm than soils in rangeland and in Eucalyptus plantations. Eucalyptus plantations had the lowest SOCS values.

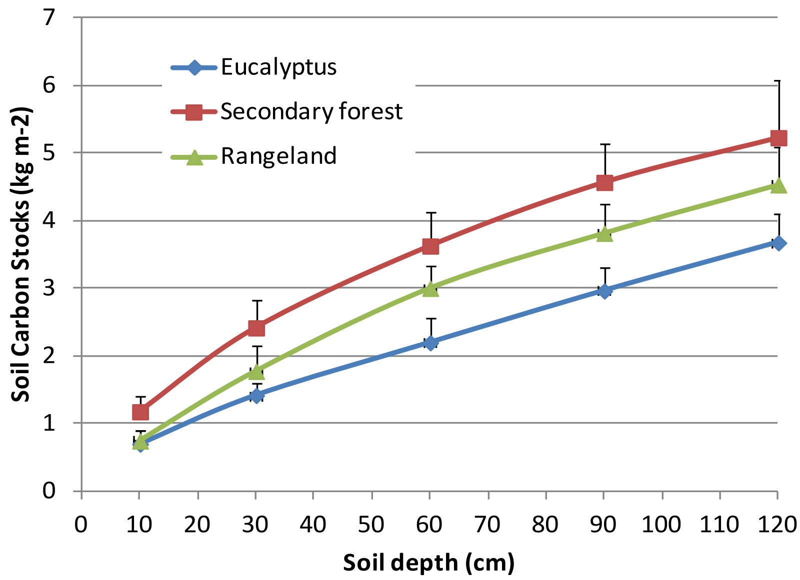

Fig 3. Soil Organic Carbon Stocks under three land-use systemsin Congo. The bars indicate the standard deviation.

Downscaling of climate predictions (Coop 2014) proved insight into how evolution of the soil properties affect soil quality (Table 2).The common trend is a site-dependent decrease in soil organic matter caused by increased mineralisation rates. Other effects are lower available water contents for plants, and increased erosion risks. These effects affect organic matter content of the soils

Table 2. Effects ofdownscaling projections for mid-21st century on soil properties

Burkina Faso	Higher crop water requirements, moderate loss of OM by mineralisation, high erosion risk in case of removal of plant cover
Congo	Higher soil erosion (sandy soils, OM poor), less acidification
Ghana	Higher crop water requirements, moderate loss of OM by mineralisation
Kenya	Loss of OM by mineralisation, moderate erosion risk on slopes
Malawi	Less acidification, loss of OM by mineralisation, higher erosion
Sudan	Soil water imbalance, higher erosion risk at the end of rainy season
Togo	Loss of OM by mineralisation, lower water availability to plants

OM: Organic Matter

Since lower available water content for plants and erosion risk effects also depend on organic matter, they should always be taken into account in organic matter management. In Kenya, Sudan, Togo and Congo there is a relationship between SOC and land-use. Therefore land cover management might be expected to have an effect on quality of soil in these sites.Other locations

(Ghana, Malawi, Burkina) did not show any significant relationship, however proper fertilisation and use of specific amendments would be useful as adaptation measures. Conservation measures on sloping soils would have to be carefully implemented.

Adaptation strategies and best management practices

Table 3 presents soil management practices to reduce climate change effects at the study sites. These were based on the intrinsic limitations of the quality of the studied soils and on the qualitative evaluation of their evolution under a climate change.

Table 3. Summary of the soil management measures or pratices in order to mitigate the effects of climate change.

	Soil management to maintain or increase SOM	Tillage techniques to improve Available Water	Erosion control measures on sloping soils	Observed relation between land cover and SOCS
Burkina Faso	**		*	No
Congo	***	**	*	Yes
Ghana	**	**	*	n.t.
Kenya	*		*	Yes
Malawi	**	*	*	No
Sudan	*	*	**	Yes
Togo	**	*		Yes

Expected effects: *slight, ** moderate, *** strong; n.t. not tested

CONCLUSIONS AND RECOMMENDATION

The following adaptation strategies to climate change for delivering soil services in the study areas were made:

a) Changes in soil quality parameters in ClimAfrica sites under a climate change were mainly a loss of OM, less available water capacity and higher water erosion risk. All these properties depend directly or indirectly on SOM making SOM management paramount.
b) West African sites experience a strong acidification, low water holding capacities, low chemical fertility and low organic matter contents. Climate change minimization on crop production should be related to land-use and organic matter stocks.
c) Soils from Kenya and Sudan have higher chemical and physical quality, and it is possible that they can buffer better future changes. Efforts should therefore be directed to conserve the quality of the medium- and high-quality soils in each site, in order to increase the efficiency of the management measures.

Recommendation: Given the very poor chemical quality of most of the soils (acidity and low nutrient content), frequent fertilisation and liming could compensate the degradation processes caused by climate change, by promoting a higher biomass production.

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1.11.P

DETERMINING THE STABILIZATION OF SUGARCANE FILTERCAKE BIOCHAR IN SOIL ENVIRONMENTS WITH CONTRASTING LEVELS OF ORGANIC MATTER

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1. INTRODUCTION

Soil organic matter (SOM) is essential in providing structure and nutrients to soils. Contributing to 80% of the terrestrial carbon (C) pool, SOM is considered an important C sink for mitigating the greenhouse effect (Leifeld and Kögel-Knabner, 2005). However, it can also be a source, as C is lost to the atmosphere in the form of carbon dioxide (CO_2) depending on land use and soil, water, and vegetation management (Lal, 2009). Adding organic matter (OM) to the soil is considered key in building up SOM, particularly for agricultural soils which are generally C-depleted. However, not all OM is retained in the soil, as it depends on the soil's ability to physically stabilize OM and on OM quality; more labile sources decompose more quickly than more recalcitrant sources (Angers et al. 2010).

There are several types of organic amendments that can be added to soil including compost, green manure, and mulch. A more recalcitrant soil amendment that has received attention in the last few years is biochar. Biochar is charcoal derived from waste biomass by pyrolysis and is considered efficient and stable in the long-term (Lehmann, 2007; Clough and Condron, 2010). It has been shown to improve soil fertility and C sequestration (Lehmann and Joseph, 2009). Yet, where in the soil matrix the biochar C stabilizes and whether its stabilization can be driven by microbial or physico- chemical processes is less understood. The objectives of this study were: 1) to determine the degree of physical stabilization of biochar C in soil fractions of soils with different OM contents and 2) to determine whether biochar has synergistic or antagonistic effects on microbial activities by measuring respiration rates.

2. MATERIALS AND METHODS

2.1. Soil selection and biochar production

Three soils (Inceptisols, USDA classification) were collected from an agricultural field and a holm oak (*Quercus ilex* L.)-cork oak (*Quercus suber* L.) mixed forest adjacent to the field, located in the municipality of Sant Celoni (41°41'N 2°29'E), province of Barcelona, about 60km northeast of the city of Barcelona, Spain. The topsoil of the agricultural field and the upper mineral layer of the forest soil were chosen to represent two soils with low OM content, while soil collected from the forest Oa horizon represented an organic soil.

Biochar was commercially produced (SPPT Ltda., Mogi Morim, São Paulo, Brazil) from sugarcane (a C4 plant) filtercake. The filtercake (a waste residue leftover after sugarcane distillation) was pyrolyzed at two different temperatures, 400°C and 600°C, to produce two biochar types.

2.2. Sample preparation and laboratory incubation

Plastic sample cups (100 mL) were filled with 40 g of fresh soils. Filtercake biochar (0.5 g, equivalent to 6 t C ha^{-1}) was mixed into each cup except the control soils which received no biochar. All substrates were wet to 60% field capacity. Every cup was placed opened in 1 L Mason™ jars containing 20 mL of distilled water to maintain a moist atmosphere, together with a second plastic cup containing 15 mL of 0.5M sodium hydroxide (NaOH) solution to trap CO_2. This made for a total of 40 Mason jars: 3 substrates x 3 biochar treatments (control soil, biochar 400°C, and biochar 600°C) x 4 replicates = 36 plus 4 blanks. The Mason jars were sealed and incubated in a dark room at 33°C for 91 days.

2.3. CO_2 analysis

The NaOH solutions were removed for CO_2 analysis and immediately replaced with new cups on days 8, 15, 22, 36, 43, 50, 71, and 91. At each time, dissolved inorganic C in the NaOH solution was determined performing hydrochloric acid (0.5M HCl) titration.

2.4. Soil fractionation and isotope analysis

Soil physical fractionation of each treatment, pre- and post-incubation, was carried out by ultrasonic dispersion of soil-water suspensions (Edwards and Bremner, 1967). The soil suspensions were then sifted through two sieves of different mesh sizes: > 0.05 mm for sand, > 0.02 mm for coarse silt, and < 0.02 mm for fine silt+clay. Aluminum potassium sulfate ($AlK(SO_4)_2$) was added to the fine silt+clay suspensions to cause the fine silt and clay to precipitate by flocculation. Fractions were then placed in an oven to dry at 60°C. Once dried, all fractions were finely ground, with the >0.05 mm representing the *sand* fraction, the 0.02 – 0.05 mm the *silt* fraction and the < 0.02 mm the *clay* fraction. Whole soil samples and soil fractions of pre- and post-incubated soils were analyzed for $\delta^{13}C$ on an elemental analysis–isotope-ratio mass spectrometer (Flash 2000 HT, Thermo Fisher Scientific, Bremen, Germany) in order to trace the fate of biochar C in the soils.

2.5. Calculations and statistical analysis

The proportional contribution of biochar C in the soil-biochar mixtures and their fractions was calculated by mass balance, taking the $\delta^{13}C$ of the biochar and soils as endmember values. The total amount of biochar C was then determined from their proportional contributions to the total C multiplied by the total C in the sample.

The effect of soil and biochar types on biochar C content in pre- and post-incubation soils was determined by univariate analysis of variance (ANOVA) using IBM® SPSS® Statistics (Version 23, SPSS. Inc., Chicago, USA). Differences between pre- and post-incubation soils were determined by repeated measures ANOVA, with "soils" and “biochar type” as the between-subjects factors and "time" as the within-subjects factor. The effect of soil and biochar types on cumulative soil respiration was also evaluated by univariate ANOVA, and their effect over time by repeated measures ANOVA. Where differences were significant, a post-hoc Tukey test ($P < 0.05$) was used to compare means. Values presented in graphs are means ±1 standard error.

3. RESULTS

The amount of biochar C in the biochar-soil treatments did not vary significantly by biochar type in both pre- and post-incubation soils (Figure 1). The forest organic soil contained higher mean biochar C than the field and forest mineral soils, but was only significantly ($P < 0.05$) higher in soils post-incubation. The repeated measures ANOVA showed that overall biochar C did not change significantly during the incubation period.

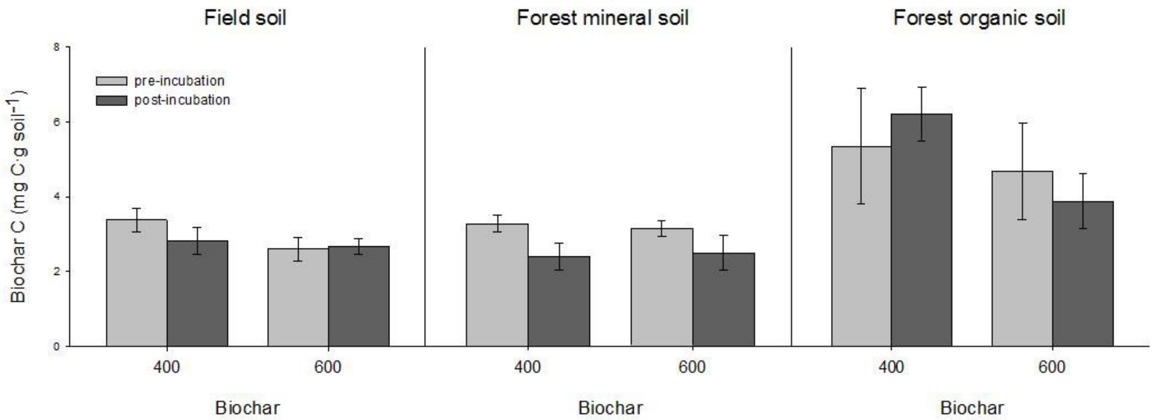

Figure 1. Mean biochar C (mg biochar C · g soil^{-1}) in soils pre- and post-incubation.

Amount of biochar C in the soil fractions did not differ between the soils, nor were there differences between the biochar C in the fractions for each soil (Figure 2). In addition, only the biochar C 400°C in the clay fraction of the forest organic soil increased during the incubation while the biochar C in the other fractions for the three soils remained similar pre- and post-incubation (Figure2). However, in the clay fraction of the field soil and in both the silt and clay fractions of the forest mineral soil, biochar C from soils with biochar 400°C were higher compared to soils with biochar 600°C (Figure 2).

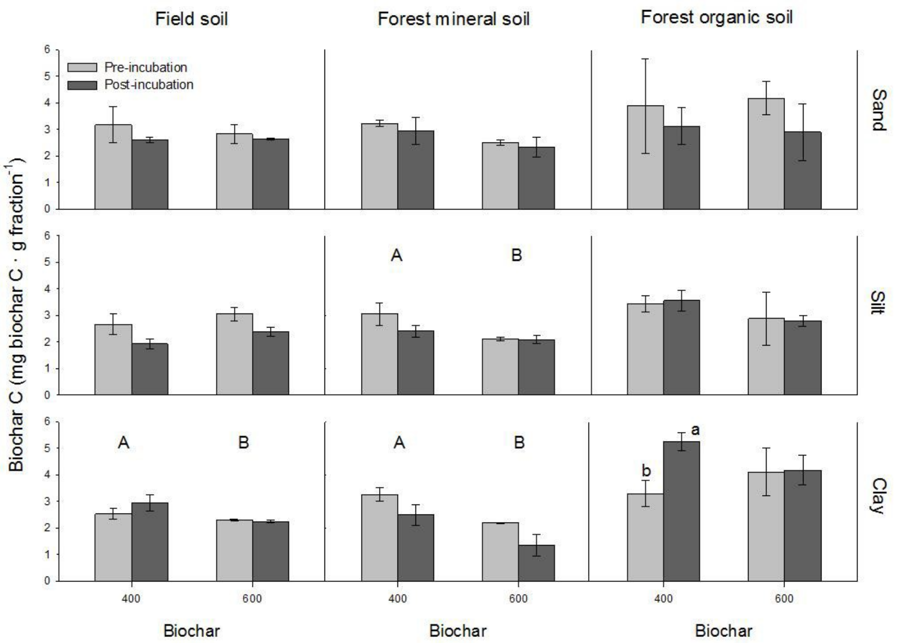

Figure 2. Mean biochar C in each fraction (mg biochar C · g fraction^{-1}) for each soil pre- and post-incubation. Different capital letters indicate significant differences between the biochars (400 and 600°C) and lowercase letters indicate significant differences between pre- and post-incubation within a biochar type (Tukey test, P < 0.05).

The 3 soils incubated differed significantly in their cumulative CO_2 efflux following the order forest organic soil> forest mineral soil>field soil. Over the incubation period, respiration rates from forest organic soil treatments started higher than the other treatments and dropped quickly over time. Respiration rates in forest mineral soils started high and dropped gradually over time, while field soils had the least pronounced drop (Figure 3A).

Biochar type also had a significant (P < 0.05) effect on soil respiration rates, where soils with biochar 400°C had higher cumulative CO_2 emissions than biochar 600°C in the field and forest mineral soils, but not in the forest organic soil (Figure 3B). Respiration rates increased in soils amended with biochar 400°C compared to controls, except for the forest mineral soils. By contrast, the amendment with biochar 600°C only caused a significant increase of soil respiration in the field soils (Figure 3B).

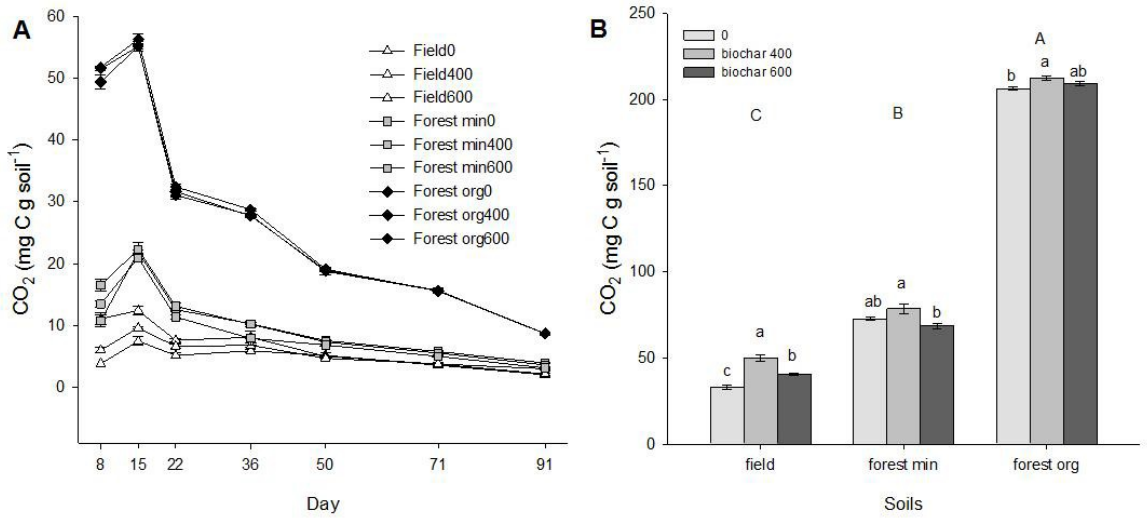

Figure 3. A) Mean CO_2 (mg C g soil^{-1}) per treatment per sampling day. B) Mean cumulative CO_2 (mg C g^{-1} soil) over a 91-day incubation. Capital letters indicate significant differences between the soil types (post-hoc Tukey test, $P < 0.05$) and lowercase letters indicate significant differences between biochars (400 and 600) and the control (0) (Tukey test, $P < 0.05$).

4. CONCLUSIONS

These results suggest that biochar C remained physically stable in soil fractions over a short-term period. However, the increase in biochar 400°C in the clay fraction of the forest organic soil suggests that there is potentially more biochar C stabilization within clay aggregates for soils with higher OM content. Biochar distribution among fractions was similar in all soil types irrespective of its OM richness. In the silt and clay fractions, biochar 400°C contributed more biochar C than biochar 600°C in the field and forest mineral soils. This may be due to differences in particle size between the biochar types. Higher soil respiration rates after adding biochar 400° C may have primed SOM decomposition, particularly in low-OM soils, while using a more recalcitrant, higher temperature biochar (e.g. 600°C) may help prevent microbial stimulation.

5. ACKNOWLEDGEMENTS

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1.13.P

EFFECT OF NITROGEN AND STOVER MANAGEMENT ON GREENHOUSE GAS EMISSIONS FROM MAIZE

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1. INTRODUCTION

The agricultural greenhouse gases (GHG) emissions account for 10% of total global anthropogenic emissions (Eurostat, 2015). However, agriculture owns significant climate change mitigation potential (Bellarby et al., 2008). In this sense, mineral and organic nitrogen fertilizers are known to be important variables in the regulation of trace gas emissions from agricultural soils (IPCC, 2007; Abalos et al., 2013).

The application of organic amendments to improve soil fertility and to sequester carbon has been gaining attention. According to Wilhelm et al. (2004) the application or incorporation of organic amendments increases soil productivity. The incorporation of straw provides an opportunity for climate change mitigation (Abalos et al., 2013). The main benefits of the incorporation of maize stover include the maintenance of soil organic matter (SOC), the prevention of soil erosion, the supply of macronutrients and micronutrients and of energy for soil biota (Mubarak et al., 2002; Lal, 2005). Agricultural crop residues with C/N ratios greater than 20, such as maize straw, are difficult to decompose (Snyder et al. 2009). However, adding a complementary source of N (organic or mineral) with the crop residues incorporated into the soil could stimulate straw mineralization increasing N- use efficiency and producing higher yields (Garcia-Ruiz and Baggs, 2007; Abalos et al., 2013). Straw or stover addition to soils can affect N_2O emissions in different ways (Malhi and Lemke, 2007) since they affect several factors regulating both nitrification and denitrification (Kurganova and Lopes de Gerenyu, 2010). The overall response of soil N_2O emissions to organic amendments remains uncertain (Lu-Jun Li et al., 2013). In the same way, although, interactions between stover management and N fertilisation have been reported by several authors (Maskina et al., 1993; Power et al., 1998; Karlen et al., 2011), there is little information about the interaction between N fertilisation doses and stover management in irrigated, high-yielding corn crops such as those in the Ebro Valley (NE Spain; Biau et al., 2013).

The objective of this study was to compare the effect of different doses of mineral nitrogen fertiliser combined with two different maize stover managements (incorporation or removal), in a soil with a high mineral N content, on the greenhouse gas (GHG) emissions, in order to improve the sustainability of the maize production system.

2. MATERIAL AND METHODS

2.1. Site description

The study was conducted at a commercial field in Almacelles (NE Sain, 41º43'N, 0º26'E) under sprinkler irrigation. The field localisation is characterized by a semiarid climate with high temperatures (19.1ºC) and low precipitation (192 mm) during the growing period of maize. Soil is classified as a Typic Calcixerept (SSS, 2003). Some of the physicochemical properties of the top 0-22 cm of the soil layer are shown in Table 1.

Table 1. Main chemical and physical soil properties at the experimental site

Soil properties	
Depth (cm)	0-22
Clay (%)	25
Silt (%)	33
Sand (%)	42
pH	8.2
Organic matter (%)	3.30
EC (dS m^{-1})	0.19
P (Olsen) (mg kg^{-1})	90
K (NH_4Ac) (mg kg^{-1})	383
Mineral N (kg N ha^{-1} yr^{-1}; spring 2010)	300

2.2. Experimental design

This study was conducted during three cereal growing seasons (2013, 2014 and 2015). Maize (*Zea mays* L.) was sown in early April at a rate of 90,000 to 95,000 plants ha^{-1}, and the distance between rows was 71 cm.

The experimental plots were established and arranged in a split plot design with four replicates. The applied crop stover management treatments were: i) stover incorporation (+R) with conventional tillage (by disk plowing) to a depth of 25 to 30 cm and ii) stover removal (-R) from the field after each year's maize harvest, using commercial machinery. The applied N treatments were: i) N0 (no N application); ii) N200 (200 kg N ha^{-1}) and iii) N300 (300 kg N ha^{-1}). As previously mentioned, these three treatments were applied with both (+R and -R) crop stover treatments. The N fertilizer (ammonium nitrate [AN]: 33.5% N) was applied in two side topdressings using a small drop-type hand driven fertilizer spreader; 50% at growth stage V3-V4 and 50% at V5-V6.

2.3. Gas sampling and quantification

The soil samples were collected weekly throughout the maize crop season. Undisturbed soil cores were taken with PVC cylinders (16.5 cm height x 7 cm Ø) and were transported to the laboratory. The greenhouse gas (N_2O, CO_2 and CH_4) emission fluxes were measured sampling air from semi-static closed chambers (Smith and Arah, 1992) and analysing the GHG concentration through a photoacoustic analyser (INNOVA 1412). The concentration of GHG from the soil samples was measured at 0, 20 and 40 minutes after closing the chambers.

The soil samples were dried to quantify their water content. The cumulative GHG emissions together with the yields allowed to calculate the Global Warming Potential (GWP) and Greenhouse Gas Intensity (GHGI) parameters.

3. RESULTS

During the experiment (from 2013 to 2015) the daily fluxes of N_2O were higher at the beginning of the trial, in the moment of N fertilisation application. Their values diminished until the end of trial. All treatments presented the similar behaviour for the cumulative emissions. For all treatments, the daily fluxes of CO_2 were higher at the beginning of the trial and tended to zero at the end of it. Generally, the treatments with R- presented higher cumulative emissions than the R+ treatments. The daily fluxes of CH_4 were negative. The cumulative emissions for CH_4 were different for each year and treatment. In 2013 the CH_4 emissions from the control treatment with R- were higher than from the control treatment with R+ while those from the rest of the treatments were similar. In 2014 the control treatment with R- had lower CH_4 emissions than the control treatment with R+. In 2015 all the treatments with R- had lower CH_4 emissions than the treatments with R+.

The GWP was positive for all the treatments in 2013 and negative for the same treatments in 2014 and 2015. The GHGI was positive in 2013 and negative in 2014 and 2015. These results allow us to say that the soil acted as a source of GHG in 2013 and as a sink in 2014 and 2015.

4. CONCLUSIONS

The daily fluxes of N_2O can be related to the application of nitrogen fertilizer. The daily fluxes of CO_2 were high at the beginning of the trial and diminished until zero at the end of it. The daily fluxes of CH_4 were negative. For all gases, in general, the treatments with R- presented higher emissions than the treatments with R+.

The GWP and GHGI parameters are related. The GHGI results show that the studied soil acted as a source of GHG in 2013 and as a sink in 2014 and 2015.

Considering the "Climate Smart Agriculture" objective of maintaining a high yield together with mitigating GHG emissions, applying a medium N dose together with stover incorporation could be a good practice in order to maintain sustainability of these highly intensive maize systems.

5. ACKNOWLEDGEMENTS

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1.14.P

CARBON AND NITROGEN DYNAMICS IN PRUNINGS DECOMPOSITION OF SUBTROPICAL CROPS

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INTRODUCTION

Traditional agriculture tends to significantly reduce the content of soil organic carbon (SOC), although organic matter is key in soil quality and fertility, improving soil physical properties and serving as a source of nutrients and energy for populations of microorganisms, fungi, invertebrates and earthworms (Birkhofer et al., 2008). Soil managements such as the application of crop residues can increase SOC. These pruning remains are burned in traditional agriculture, thus removing, nutrients from the environment and increasing C emissions. The effect of the addition of pruning remains on C fixation in soils has been studied in several crops. Thus, in olive groves Nieto et al. (2010) reported an increase in SOC content with the addition of pruning remains by 4-6 times compared to nude soils during 6 and 10 year periods of continuous management. Litter decomposition rates can be explained by climatic variables and litter characteristics such as nitrogen content (Sariyildiz et al., 2003), as well as by fauna activity (Zhang et al., 2015). In addtion, the percentage of decomposition of plant matter in soils depends on its form and composition (Paul & van Veen, 1978).

The aim of this work is to determine the C and N dynamics over time in the decomposition process of pruning remains added to the soil surface under subtropical crops.

MATERIALS AND METHODS

The study area is located at the experimental farm El Zahorí (36º45'54.2"N, 3º39'55.0"W, 209 m a.s.l.) in Almuñécar, Granada, S Spain (Figure 1). Soils are Eutric Anthrosols (IUSS Working Group WRB, 2015) and the predominant climate is Subtropical Mediterranean.

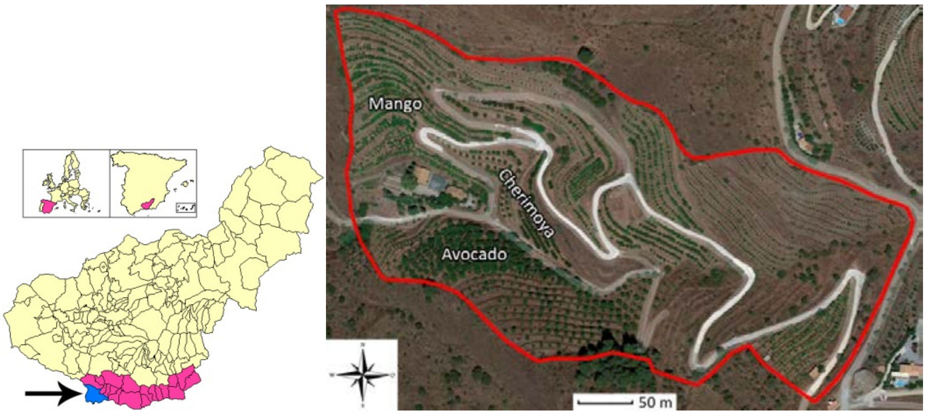

Figure 1. Study areaand crops location.

The experiment was carried out with three crops: avocado (*Persea americana Mill.*), cherimoya (*Annona cherimola Mill.*) and mango (*Mangifera indica L.*). 135 mesh bags (26 x 26 cm^2) were made with PVC mesh (2 mm mesh) and sewn with nylon. They were filled with 100 grams of shredded pruning remains with a crusher to create litterbags. Three treatments were applied: prunings from each crop (avocado, cherimoya and mango) were placed on the soil surface underneath their correspondent trees. A total of 45 bags were placed underneath each crop. The litterbags (9 per period) were removed from the soil after 3, 6, 12, 18 and 24 months. In laboratory the foreign material was removed with a brush and weighed. All samples were dried for 72 h in a forced-air oven at 65 °C and then weighed again. The dried samples were finely grounded and C and N contents were determined with a CN element analyzer (LECO TruSpec CN). Carbon and nitrogen released from decomposing litter was obtained as (Lado-Monserrat et al., 2015):

$N_t = C_0 - [(1 - W)C_t]$

where N_t is the amount of nutrient released (positive values) or absorbed (negative values) at time t, C_0 is the initial nutrient litter concentration, C_t is the nutrient litter concentration at time t (all three expressed in percent) and W is weightloss at time t (%/100). Differences in the C and N dynamics and in the weightloss were tested for each date separately by one-way ANOVAs using Tukey's HSD test, and statistical significance was defined as p < 0:05.

RESULTS AND DISCUSSION

There were no significant differences in weightloss between avocado and cherimoya crops for each period, but there were significant differences between the mango and the other prunings after the third month (Figure 2). In fact, the most intense weightloss was observed in the first three months. This is in agreement with Flores-Sanchez et al. (2016), who showed that the weightloss by decomposition was more intense in the first 4 months. Although cherimoya prunings had the highest weightloss after 24 months, losing 55,94 % of its weight, there were no statistically significant differences with the other prunings during this period. Other studies demonstrated that after long periods, the proportion of retained plant material was very similar in different soils, using different plant material and different quantities of plant material added to the soil (Paul & van Veen, 1978).

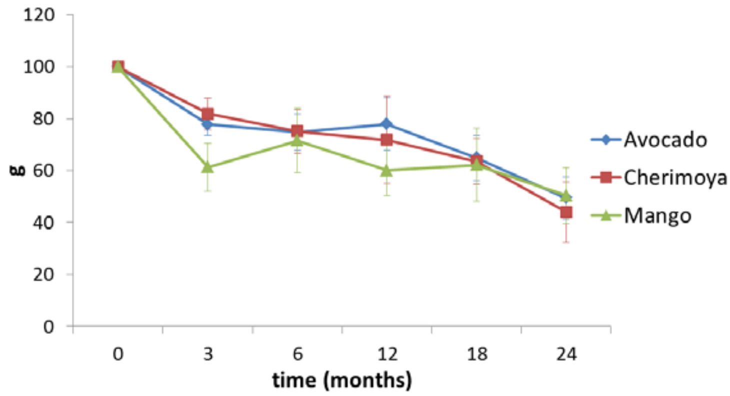

Figure 2. Weightloss of prunings remains over time for each crop.

As Figure 3 (a) shows, there is a decrease in C ratios over time, as other authors have described (Bravo-Oviedo et al., 2017), suggesting an intense decomposition. The lowest rate of C release in the pruning remains of all species occured at 24 months. The highest C release occurred after 3 months in cherimoya prunings, where 40,72 % (with respect to the initial concentration)was released. Samples in avocado and cherimoya showed significant differences between months 3 and 24. Carbon release in mango prunings had small fluctuations over time, however there were significant differences between 6 and 24 months.

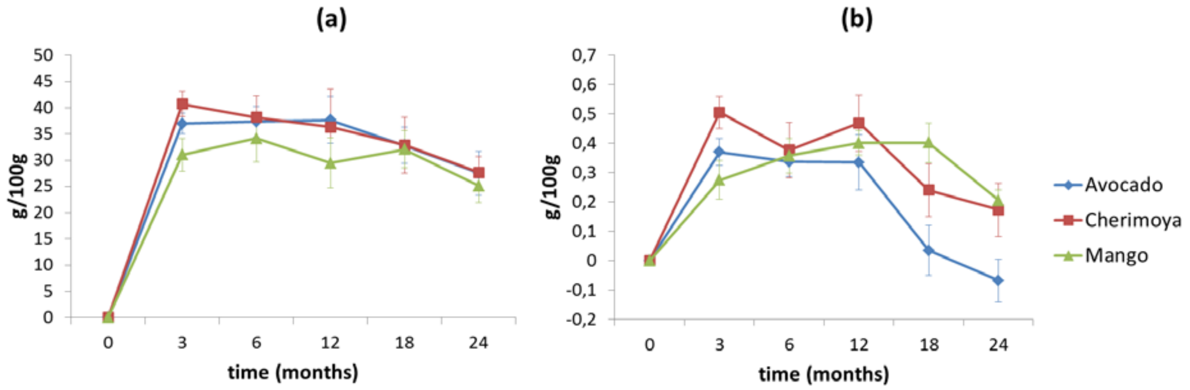

Figure 3. C (a) and N (b) released or absorbed (positive or negative values respectively) in the decomposing pruning remains for each crop.

Regarding N content (Figure 3 b), a great release occurred in all prunings in the first 3 months of sampling. Subsequently, at 6 and 12 months the release ratios kept constant with slight differences between pruning remains. N release over time in avocado and cherimoya prunings in samplings at 18 and 24 months tended to decrease, as showed by other authors with several vegetal matters (Berg & Staaf, 1981; Flores-Sanchez et al., 2016). At the end of the experiment the values for N in avocado decreased even to negative values, suggesting N absorption. Thus, N concentration in avocado prunings at 3 months was 0,71 % and 1,19 % after 24 months. This nitrogen increase in avocado is probably due to the increase in microbial activity by the addition of organic matter to the soil surface, thus obtaining microbial nitrogen in addition to the nitrogen present in the prunings. As Schmidt et al. (2016) showed, microbial activity plays an important role in litter decomposition under aerobic conditions. In addition, according to Delgado-Baquerizo et al. (2015), irrespective of litter quality, soil characteristics such as microbial biomass may increase C and N availability during litter decomposition.

CONCLUSIONS

In order to analyze the decomposition of pruning remains over time it is important to carry out sampling during the first 3 months, because the intense decomposition in these first months causes a significant loss of weight as well as a greater release of C and N with respect to the subsequent samplings. The release of C and N decreased after the first 3 months, even showing negative values for N in the last samples of avocado prunings. The addition of pruning remains to the soil surface and its subsequent degradation release C and N into the soil, contributing to nutrient recycling.

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KEYWORDS

Crop residues, litterbags, decomposition rates, C release, N release, mulching, subtropical crops, pruning, avocado, mango, cherimoya.

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1.15.P

LONG-TERM NO-TILL EFFECTS ON GREENHOUSE GAS EMISSIONS AND SOIL ORGANIC CARBON

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INTRODUCTION

Agriculture is believed to be responsible for the losses of more than 50 Pg C mostly as CO_2 emission, about 50% of global CH_4 emission, and up to 80% of anthropogenic N_2O emissions (EPA, 2009). Therefore, knowledge of the trade-offs between soil GHG emissions, yields and agricultural practices is essential for the development of sustainable agriculture and climate mitigation.

In this study field data and simulated data are presented to estimate the effect of tillage on crop yield and greenhouse gas emissions.

MATERIALS AND METHODS

Experiment 1

A field study was initiated in 1996 at the Tennessee Valley Research and Extension Center, Belle Mina, AL (34o 41' N, 86o 52'W) on a Decatur silt loam (clayey, kaolinitic thermic, Typic Paleudults) soil. Crop rotation pattern starting in 1996 has been two continuous years of cotton followed by one year corn (cotton-cotton-corn). Treatments included three tillage systems: conventional tillage (CT), mulch tillage (MT), and no tillage (NT); two cropping systems: cotton in the summer and fallow in the winter (CF) and cotton in the summer and cereal rye cover crop in winter (CR); two sources of nitrogen: ammonium nitrate @ 100 kg N ha^{-1} and poultry litter @ 100 and 200 kg N ha^{-1}, a control treatment with no N application was also included. A bare fallow treatment was also maintained without any crop, tillage and fertilizer application. The experimental design was a randomized complete block with an incomplete factorial treatment arrangement. The winter rye cover crop (cv. Elbon) was seeded at 60 kg ha^{-1} in fall with a no-till grain drill and killed with glyphosphate herbicide about 7 d after flowering in the spring of every year. The time between killing of winter rye and cotton planting was about 4 wk each year to allow for total drying of residues. No fertilizer was applied to the cover crop.

Experiment 2

The second experiment was done in a cornfield from 2012-2015 with two tillage treatments (conventional tillage-CT; no-tillage-NT), along with some other agricultural practices at Agricultural Research and Education Center, Tennessee State University, Nashville, TN and a nearby commercial field as described by Deng et al, 2016.

We simulated soil N_2O emissions under conventional tillage and no-tillage treatments and some other agricultural practices using the DeNitrification-DeComposition (DNDC model), Seasonal dynamics and annual N_2O emissions were estimated under different agricultural practices. The effects of tillage on GHG emission is quantified using the natural log of the response ratio (RR):

$$RR = \ln(X_t/X_c) = \ln(X_t) - \ln(X_c)$$

where RR is the ratio of the mean value of the soil CO_2, CH_4, or N_2O flux in the treatment group (X_t) to that in the control group (X_c), an index of the effect of the experimental treatment on the target variable. A weighted RR is computed from individual RR by giving greater weight to studies whose estimates have greater precision (lower variance) [24 – 27]. The treatment effect of tillage is considered to be significant if the 95% confidence interval (CI) of RR does not overlap with zero [27]. RR can be converted back and presented as percentage changes.

RESULTS AND DISCUSSION

Experiment 1

Soil erosion estimates in cotton plots under conventional tillage system with winter rye cover cropping declined by 36% from 8.0 Mg ha^{-1} $year^{-1}$ in 1997 to 5.1 Mg ha^{-1} $year^{-1}$ in 2004 . This result was largely attributed to cumulative effect of surface residue cover which increased by 17%, from 20% in 1997 to 37% in 2004. In conventional tillage without winter rye cover cropping, soil erosion estimates were 11.0 Mg ha^{-1} $year^{-1}$ in 1997 and increased to 12.0 Mg ha^{-1} $year^{-1}$ in 2004. In no-till system, soil erosion estimates generally remained stable over the study period, averaging 0.5 and 1.3 Mg ha^{-1} year $^{-1}$ with and without winter rye cover cropping, respectively (Nyakatawa et al , 2007).

Soil CO_2 efflux was measured throughout the cotton growing season from 2003 to 2006. In all years interactions for tillage x N source and tillage x cropping system were found significant. On average, NT with ammonium nitrate at 100 kg N ha^{-1} (100 ANN) recorded the lowest CO_2 efflux (1.96 µmol m^{-2} s^{-1}) and CT with poultry litter at 100 kg N ha^{-1} (100 PLN) showed the highest CO_2 efflux (3.60 µmol m^{-2} s^{-1}) (Table 1). Averaged over the three years CT and MT with 100 PLN released 37 and 25% higher CO_2 into the atmosphere compared to NT with 100 PLN; details are previously published (Roberson et al, 2008).

Experiment 2

Results indicate that no-till significantly increased corn yield by 28% over the conventional tillage due to soil water conservation. The management practices significantly altered soil N2O emission, with the highest in the conventional till with urea ammonium nitrate application and the least in the no-till with denitrification inhibitor (Deng et al., 2015). DNDC model Simulations indicated that N_2O emission and corn yield increases due to tillage compared to no-till (Table 2). N_2O emission is sensitive to precipitation, temperature, soil organic carbon, and the amount of total N fertilizer applied (Deng et al., 2016).

CONCLUSIONS AND RECOMMENDATIONS

In our long term study, all three tillage practices; conventional tillage (CT), mulch tillage (MT) and no-tillage (NT) had similar cotton lint yields with application of ammonium nitrate (AN) at 100 kg N ha^{-1}. No-tillage with poultry litter @ 100 kg N ha^{-1} reduced soil CO_2 emissions by 37 and 25%, respectively compared to conventional and mulch tillage during a cotton growing season of about 165 days. Further, NT system recorded significantly lower soil erosions (0.4 to 2.0 Mg ha^{-1}) compared to CT (5.1-12 Mg ha^{-1}).

These data were also reinforced in the second study where no-till increased corn yield and reduced N2O emissions.

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Table 1. Interaction effect of tillage and nitrogen sources (poultry litter-pl and ammonium nitrate-an) on soil co_2 efflux, belle mina, al 2003, 2004 and 2006

Tillage	Nitrogen Source							
	2003		2004		2006		Average	
	100PLN	100AN	100PLN	100AN	100PLN	100AN	100PLN	100AN
	----------------------μmol m^{-2} s^{-1}----------------------							
CT‡	4.39a†	3.65b	3.00a	2.74ab	3.40a	2.98ab	3.60	3.12
MT	4.17a	3.09c	2.90ab	2.95a	3.39a	2.70bc	3.49	2.91
NT	2.84c	2.25d	2.57ab	2.04b	2.47c	1.58d	2.63	1.96

† Treatment means in each year followed by the same lowercase letter are not significantly different from each other at P ≤ 0.05. ‡ CT= Conventional Tillage, MT= Mulch Tillage, NT= No Tillage, 100 PLN= 100 kg N ha^{-1} as poultry litter, 100 ANN= 100 kg N ha^{-1} as ammonium nitrate.

Table 2. Simulated corn yield and annual cumulative N_2O emissions over three years under different tillage (conventional till-CT and No-till-NT) and fertilizer management practices in a cornfield using the DNDC model

Variable	Year	NT-URAN	NT-inhibitor	NT-manure	NT-spilt	CT-URAN
Corn yield	2012	4285	4517	3938	4478	3348
(kg C ha^{-1})	2013	4573	4833	4238	4770	4037
	2014	4506	4653	4331	4601	3846
N_2O emission	2012	1.94	1.32	2.03	2.01	2.87
(kg N ha^{-1})	2013	2.34	1.84	2.23	2.35	2.91
	2014	2.12	1.67	2.20	1.92	2.82

Session II: Soil and water conservation practices

Session VIII: Salinization and contamination of soils and waters

Session IX: Socio-economical factors in soil and water conservation

2.2.O

STRIP-TILL FOR EROSION REDUCTION IN MAIZE

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INTRODUCTION

Erosion by water is a frequently occurring soil degradation process in the Belgian loess belt. Therefore, the Flemish government has recently forced farmers to combat soil erosion on field parcels with high or very high erosion risks through cross compliance of the European Common Agricultural Policy (CAP). The government puts its focus on on-site erosion measures, such as non-inversion tillage in combination with cover crops, but even with non-inversion tillage, maize fields remain vulnerable to erosion. Therefore, the objective of the Gomeros-project is to investigate the effect of alternative conservation practices on soil erosion by water and crop yield. In this article, we present the results of two field trials in which the effect of non-inversion tillage and strip-tillage on maize growth and soil erosion was studied.

SUMMARY OF METHODS

In 2016, two field trials, i.e., 'Maarkedal' and 'Bierbeek' (named after the municipalities they were located in) were established comparing conventional mouldboard ploughing with non-inversion tillage and strip-till in maize. Both field parcels are classified by the Flemish Government as being highly sensitive to erosion, with slopes in the area of the erosion simulations of 5.4-8.2% in the 'Maarkedal' trial and 8.3-11.1% in the 'Bierbeek' trial. According to the Belgian soil map (1:20.000) and the derived World Reference Base soil units (1:40.000), both field parcels are well-drained and classified as Nudiargic Luvisols. The soil texture of the Maarkedal trial is silt to silt loam and of the Bierbeek trial loam to silt loam (USDA soil texture classification). The organic carbon content of the top soil (0-5 cm) was 1.08% at the Maarkedal trial and 0.87-1.74% at the Bierbeek trial (measured at the time of rainfall simulations). The nutritional status (pH, P, K, Mg, Ca) was within or above the recommended range for the region. The growing season of 2016 was very wet end of May and in June (2-2.5 times the normal precipitation) and very dry at the end of the summer and autumn (70-90% less precipitation than normal).

The Maarkedal field parcel is normally conventionally tilled. After the harvest of winter wheat in 2015, the soil was non-inversely tilled to a depth of 30 cm with a Kuhn cultisol DC 301 and in the same tillage operation the cover crop, being a mixture of several crops, was sown. Japanese oat and white mustard were predominantly present and well developed. The Japanese oat was not completely frozen to death and, therefore, the cover crop was destroyed with a glyphosate based herbicide. The Bierbeek field parcel was not tilled with a mouldboard plough since 8 years. The main tillage operations are executed with an Agrisem cultiplow until a depth of 30-35 cm. In 2015, grain maize was grown after which the field was immediately superficially cultivated twice with a disc harrow. Soil penetration resistance measurements until a depth of 80 cm before the start of the experiments, did not reveal a plough pan at both sites. In most parts of the field parcels the penetration resistance remained below 3MPa but the arable layer (0-30 cm) was clearly less loosened in the Bierbeek trial.

The Maarkedal trial consisted of one treatment with conventional mouldboard tillage (CT), two treatments with non-inversion tillage (Kuhn cultisol DC301 (RT1) or Carré Neolab eco (RT2)) until a depth of 30 cm and four strip-till treatments (S), differing in tillage depth (16, 21 or 23) and/or teeth. The cover crop was superficially incorporated with a disc harrow in the CT, RT1 and RT2 treatments and shred in the S treatments. In all treatments, urea was applied before the main tillage operations and mineral fertilizer in the row during sowing. The slurry ($35 m^3/ha$) was injected in the row during the strip-till operations, 8-10 cm less deep than the tillage depth, and applied with line spreading booms in the other treatments just before the main tillage operations. All treatments were sown the same day with GPS-RTK (May 7 2016).

The Bierbeek field trial consisted of one treatment with conventional mouldboard ploughing (CT), one treatment with non-inversion tillage (Agrisem cultiplow) until a depth of 30-35 cm (RT) and four strip-till treatments (S). Three S treatments differed in tillage depth, i.e., 16, 21 or 25 cm. A fourth S treatment was tilled twice, the first time at a depth of 23 cm and a few weeks later, together with the other S treatments, at a depth of 13 cm. All treatments received broadcasted mineral fertilizer just after the main tillage operations and mineral fertilizer in the row during sowing (May 13 2016).

Soil surface cover was determined by taking orthogonal pictures with a $1x1m^2$ frame and counting the number of crosses of a $5x5cm^2$ grid with crop residues, weeds or crop. Crop yield was determined by harvesting corn cobs by hand at 4 locations in each treatment. At each location 2 or 4 rows of 5 m length were harvested. The cobs were treshed with a field trial tresher and the dry matter was determined by drying 1 kg of grains at 70°C for 72h.

At the Maarkedal trial rainfall simulations were conducted on June 28-29 2016, using a nozzle-type field rainfall simulator with the nozzle (Lechler 460 788) suspended at 3 m height (Poesen et al., 1995; Leys et al., 2007). Three simulations were conducted in the CT treatment and 2 in the RT2 and S16 treatments. The plots were ca. $0.8x0.8m^2$ large, containing 2 maize rows. All simulations were outside wheel tracks. The plots were rained with rainwater for 60 minutes with an average intensity of 36 $l/m^2.h$. Run-off and sediment was collected, the time-runoff relationship was recorded and the sediment concentration was determined by taking three homogenized subsamples (ca. 1 l) at the end of the simulation and drying the samples at 105°C.

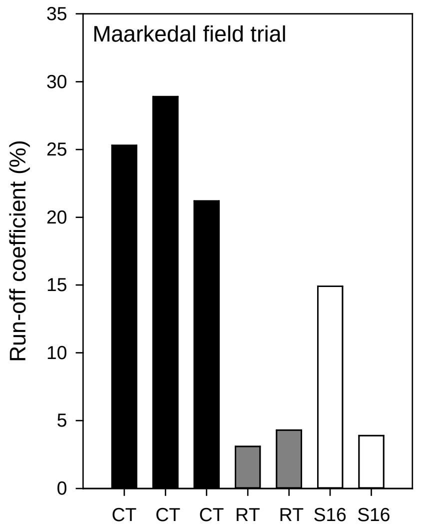
At the Bierbeek trial rainfall simulations were conducted on July 7-8 2016, using 6 sprinklers (TeeJet TG SS 14 W) suspended at 1.8 m height (Vermang, 2012). Two simulations were conducted in the CT, RT and S21 treatments. The plots were 5m long and 2 m width, containing 3 maize rows. Each plot was divided in two subplots ($5x1m^2$) to distinguish between the effect in and outside the wheel track of the sowing machine. The plots were rained with rainwater for 20 minutes with an average intensity of 157 $l/m^2.h$. Run-off and sediment was collected, the time-runoff relationship was recorded and the sediment concentration was determined at regular intervals.

RESULTS

Soil surface cover in the Maarkedal field trial just after sowing was considerably higher in the S treatments (13.0-29.8%) compared to RT (4.7-5.4%). The percentage of soil covered with grain maize residues of the previous year was nearly equal in the Bierbeek field trial for the S (8.6%) and NT (6.3%) treatments, but total soil surface cover was much higher for the S treatment due to the prevalence of weeds (17.3% total cover). The soil surface cover in the CT treatments was in both field trials very low (<0.3%).

No significant differences in grain yield were detected in the Maarkedal site. At the Bierbeek trial, yield reductions compared to CT were 10.6% for RT and 17.8% to 27.9% for S. Yields in the S treatments were positively correlated with tillage depth and the highest yield was obtained in the treatment that received two tillage operations. There are several explanations for the yield results. A first important factor are the initial soil conditions. In the Maarkedal trial, the soil was well loosened after wheat harvest and before cover crop sowing which enabled the roots also to grow between the tilled strips of the S treatment. In the Bierbeek trial, no tillage had occurred in the autumn of 2015 and no cover crop was grown, leaving a harder soil. As a result, roots were not able to penetrate the soil between the tilled strips of the S treatments (visual assessment). As the roots only had the tilled strip to explore, higher tillage depths resulted in larger rootable soil volumes and, as a consequence, also higher crop yields. A second factor was the fertilization strategy. Relatively more fertilizer was injected in the tilled row of the S treatments in the Maarkedal trial. In both trials, mineral fertilizer was provided in the row during sowing, but in Maarkedal also slurry was applied in the row during strip-tillage. In another field trial, not presented here, we observed higher yields for the S treatments when slurry was provided in the tilled row compared to the S treatments where slurry was broadcasted. A third factor was snail damage in the strip-till treatments of the Bierbeek trial.

Runoff and sediment production was seriously reduced in both field trials in the non-inversion tillage and strip-till treatments (Figure 1 and 2). Compared with CT, sediment production was reduced with 51 to 97% in the RT treatments and with 83 to 100% in the S treatments (Figure 2a). Sediment production inside the wheel tracks of the sowing machine was for CT 62 to 287 % higher than for CT outside the wheel tracks. Comparing all treatments inside wheel tracks, RT could seriously reduce sediment production compared to CT and this reduction was even higher for S (Figure 2b). Based on the measurement of rills induced by natural rainfall at the Bierbeek trial we can conclude that RT and S can reduce 70-100% of the rill erosion. Rill erosion occurred for CT both in the wheel tracks of the sowing machine and in the crop row, but only in the crop row for S and only in the wheel tracks for RT. This illustrates the importance of crop residues which were almost absent in the crop rows of S and less abundant between the rows of RT than of S.



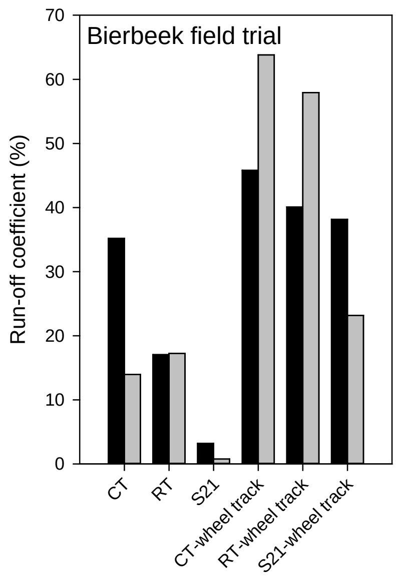


Figure 1: Runoff coefficients measured during rainfall simulations; CT= conventional tillage, RT = non-inversion tillage; S=strip-tillage at a depth of 16 cm (S16) or 21cm (S21)

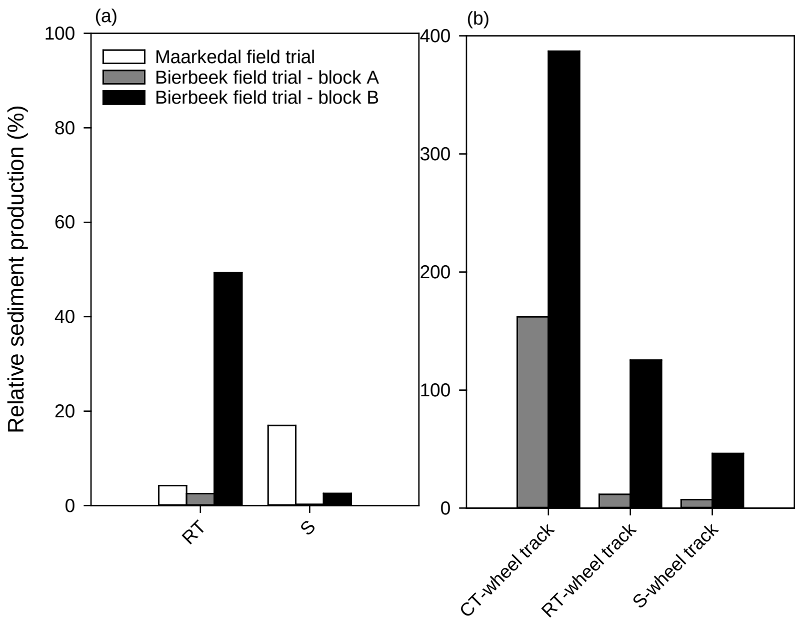


Figure 2: Sediment production relative to the conventional tillage treatment (CT) outside a wheel track (=100%) measured during rainfall simulations for (a) non-inversion tillage (RT) and strip-tillage (S) outside wheel tracks and (b) CT, RT and S inside wheel tracks of the sowing machine. (a) shows data for the Maarkedal and Bierbeek field trial. The results of the Bierbeek field trial are shown for the two replications (block A and B) separately due to block effects. (b) shows data for the Bierbeek field trial only.

CONCLUSIONS

We conclude that good physical soil conditions at the start and applying fertilizers in the row during tillage are crucial for the successful implementation of strip-till. Both non-inversion tillage and strip-till seriously reduced soil losses by erosion, but strip-till did not systematically reduce erosion more than non-inversion tillage. There is therefore at present no evidence to promote strip-till over non-inversion tillage in erosion-prone areas.

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2.4.O

INTERACTION BETWEEN COVER CROPS AND SOIL MICROBIOLOGY DURING MAIZE CULTIVATION

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INTRODUCTION

Conventional maize-after-maize systems in the UK are frequently associated with on-site soil degradation and off-site environmental pollution (Boardman, 2013). This is due to a combination of factors including late autumn harvest (often in wet soil conditions) and post-harvest management that leaves minimal soil protection from stubble over autumn and winter. Typically these actions lead to soil compaction, surface runoff and soil erosion (Posthumus et al., 2011). Cover crops (CCs) can be used as a soil erosion control measure (Langdale et al., 1991) and potentially offer additional soil health benefits through their interaction with the soil microbial community and by influencing soil physical and chemical processes (Brennan and Boyd, 2012). This research aimed to study the effect of CCs (under-sown with maize) on the soil microbial community, and infer whether these changes reduced soil erodibility and improved sustainable maize cultivation.

METHODS

The field trial ran from August 2015 to March 2016 in the Lugg sub-catchment (UK) (OS grid references: 344505 E, 257646 N). Annual average rainfall for the area is 794mm yr^{-1} (Met Office, 2016). The site was a sandy loam soil, and had field slopes of 3° to 4°. Experimental variables comprised four treatments: i) Italian ryegrass (*Lolium multiflorum*), ii) Italian ryegrass and hairy vetch (*Vicia villosa*), iii) Italian ryegrass and berseem clover (*Trifolium alexandrinum*), iv) No CCs (post-harvest wintered maize stubbles) replicated five times and arranged randomly across the slope. Microbiological analyses comprised microbial biomass, phospholipids fatty acid analysis (PLFA) and multiple substrates inducted respiration. Microbial biomass was determined using the fumigation extraction method (Anderson and Domsch, 1978) using a K_{ec} of 0.45 (Vance et al., 1987). The phenotypic profile of the microbial community was determined by PLFA (Pawlett et al., 2013). Additionally, the fatty acid 18:2ω6,9 was used as indicator of the incidence of ectomycorrhizal fungi, and the sum of the mol% of the fatty acids 15:0a, 15:0ai, 15:0, 16:0i, 16:1ωc, cyc17:0 isomer, and 17:0ai as an indicator of total bacteria (Frostegard, 1996). The functional profile of the microbial community was determined by MSIR (Multiple Substrate Induced Respiration) using RABIT (Rapid Automated Bacterial Impedance Technique, Don Whitley, UK) (Pawlett et al., 2013). Substrates used were D-(+)-glucose (75mM), L- arginine (15mM), L-glutamine (15mM), citric acid (100mM), alpha-Ketoglutaric acid (100mM), and deionised water for basal respiration measurement. Aggregate stability was determined using a gravity-fed rainfall tower to simulate a natural rainfall event (Cooper, 2007). During the test, two rainfall intensities were used: 86.8 mm h^{-1} and 47.7 mm h^{-1}, corresponding respectively to 1-50 years storm in and 1-10 years return storms for the River Lugg catchment. Statistical analysis was performed using the software STATISTICA 12 (Dell Inc. 2015). One-way ANOVA followed by post-hoc Fisher LSD comparison was performed to find homogenous groups. Significant difference was set at $p<0.05$. Principle Component Analysis (PCA) was used to analyse the multivariate PLFA and MSIR data. Resultant PCA factors scores were analysed for significant differences using ANOVA.

RESULTS

There were no significant differences in microbial biomass between the treatments either in November 2015 or March 2016 (p>0.05), Table 1. In addition, there was no significant treatment effect on aggregate stability for either of the two rainfall intensities used (p>0.05), Table 1.

Table 1: Microbial biomass (µg C g^{-1}) and unstable aggregates (% weight g).

	Microbial biomass (µg C g^{-1}) ± SE		Unstable aggregates (% g) ± SE	
	November 2015	March 2016	1-50-yr storm	1-20-yr storm
Control, no CCs	226.1 ± 10.1	166.5 ± 9.7	50.2 ± 5.8	36.8 ± 10.5
L. multiflorum	232.3 ± 10.5	189.9 ± 10.8	47.9 ± 2.1	55.2 ± 5.7
L. multiflorum & *T. alexandrinum*	204.7 ± 15.3	171.2 ± 8.6	48.3 ± 3.8	41.3 ± 8.6
L. multiflorum & *V. villosa*	237.0 ± 10.9	191.9 ± 4.7	44.6 ± 1.6	27 ± 7.1

CCs cover crops, SE standard error

Cover crop choice affected the MSIR functional profile of the microbial community (p<0.01). In March 2016, the treatment *L. multiflorum* & *V. villosa* was significantly different from the other treatments on PC4, Figure 1a. This was mostly due to the basal respiration rate (loading >0.6). The first four principle components (PC) accounted for 86% of the total variation (PC1 31%, PC2 22%, PC3 19%, and PC4 15%). Cover crop choice also affected the phenotypic (PLFA) profile of the microbial community (p<0.001). In March 2016, the treatment *L. multiflorum* & *T. alexandrinum* had a significantly different profile on PC1 compared to the other treatments, Figure 1b. On PC1, fatty acids with loadings >=0.8 included: 14:0, 15:0i, 15:0ai, 15:0, 16:0i, 16:1ω7c, and Me 17:0 isomer; 18:1ω9c had a loadings <=-0.8. PC1 to PC3 accounted for 59% of the total variation (PC1 29%, PC2 19%, PC3 10%).

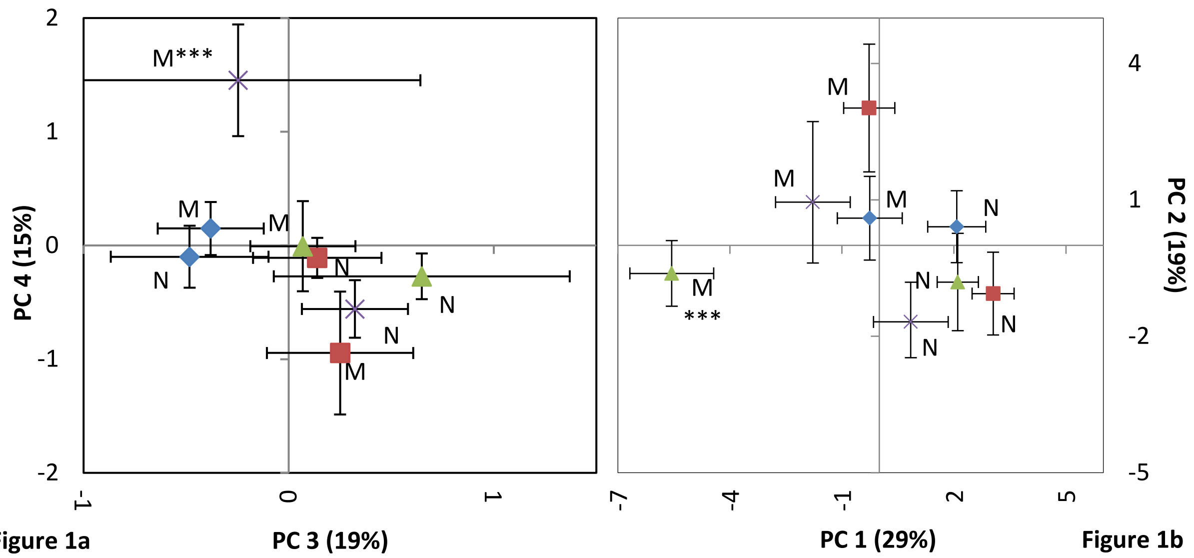

Figure 1: PCA score plots of the means (n=5 ± SE). Figure 1a: MSIR profile, PC 3 and 4. Figure 1b: PLFA profile PC 1 and 2. Diamonds are control, squares *L. multiflorum*, triangles *L. multiflorum* & *T. alexandrinum*, exes *L. multiflorum* & *V. villosa*. N=November 2015, M=March 2016. ***=significant differences (p<0.05).

In March 2016, relative quantities of the ectomycorrhizal fungi (indicated by the %mol of the fatty acid 18:2ω6,9) were significantly greater (+40%) in the presence of CCs compared to the control, Table 2. Total bacteria (%mol) were significantly reduced (-21%) where *L. multiflorum* & *T. alexandrinum* were grown compared to other treatments. The fungi/bacteria ratio was 45% lower in the control compared to CCs treatments, Table 2.

Table 2: Fatty acids biomarkers (%mol) for fungi, total bacteria and fungi/bacteria ratio for the samples collected in March 2016.

	Fungi (18:2ω6,9)	Total bacteria	Fungi/bacteria ratio
Control, no CCs	2.5 ± 0.2^{a}	18.8 ± 2.1^{a}	0.13 ± 0.01^{a}
L. multiflorum	4 ± 0.3^{b,c}	19.1 ± 0.9^{a}	0.21 ± 0.02^{b}
L. multiflorum & *T. alexandrinum*	4 ± 0.2^{b}	16.2 ± 2.3^{b}	0.25 ± 0.02^{b}
L. multiflorum & *V. villosa*	4.6 ± 0.2^{c}	18.5 ± 2.6^{a}	0.25 ± 0.02^{b}

CCs cover crops, SE standard error, and letters represent homogenous groups

CONCLUSIONS

The absence of an effect of CCs on microbial biomass could be attributed to the short time (eight months) of the experiment. The majority of studies which reported an increase in microbial biomass were conducted over multiple growing seasons (Dinesh et al., 2006; Sainju et al., 2007). Therefore, if the experiment was conducted for longer, effects may become more pronounced. The lack of a significant effect on aggregate stability was attributed to the brief permanence of the CCs in the soil when the soil samples were taken (three months); other time-limited studies have come to similar conclusions (Welch et al., 2016). Studies reporting an increase of stable aggregates after CC implementation typically collect samples after longer periods, e.g. after CC termination (Hermawan and Bomke, 1997; García-Gonzáez et al., 2016). The absence of differences in the first three PC of the MSIR profile could indicate that other soil proprieties, such as pH or total organic carbon, were the main drivers of the functional profile of the microbial community (Creamer et al., 2016), or that one season of CC implementation was too short to see an effect. However, the presence of a significant difference in PC4 for the treatment *L. multiflorum* & *V. villosa* in March 2016 could be an indication of an early-stage differentiation of the functional profile due to the CCs. It is also noticeable that the control treatment in November 2015 and the control in March 2016 were similar; while the functional profiles of the CCs in March 2016 appear to be diverging from November 2015, suggesting that a differentiation could be evident if the experiment was conducted for longer. The alteration of the relative abundance of the ectomycorrhizal fungi (18:2ω6,9) and total bacteria in the presence of CCs in March 2016 demonstrate a microbial community shift towards fungal dominance. Similar conclusions were found in other studies (Nakamoto et al., 2012). However, this shift was captured by the PCA of the PLFA profiles only for *L. multiflorum* & *T. Alexandrinum*, indicating an early stage of differentiation. The noticeable divergence of the CCs in the PCA score plot from November 2015 to March 2016 and the lack of separation between the controls support our hypothesis of a microbial community shift caused by the presence of CCs. Additional ongoing analysis of ergosterol and easily extractible glomalin may allow improved data interpretation. We concluded that the implementation CCs shifted the microbial community towards greater fungal dominance after one season, which in the long term could result in an increase of aggregate stability and therefore a decrease of soil erodibility and could improve sustainable maize cultivation. The lack of significant effects on microbial biomass and aggregate stability, and the small effects on the functional profile of the microbial community are to be attributed to the short observed time. This research suggests that it may be necessary to implement conservation practices over a longer term in order to see changes in soil health.

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2.5.O

EFFECTS OF DIFFERENT AGRICULTURAL MANAGEMENT PRACTICES ON THE DISSIPATION OF TWO HERBICIDES AND ON SOIL MICROBIAL COMMUNITIES

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1. ABSTRACT

The application of organic residues as soil amendments modifies the pesticide behaviour and both organic residues and pesticides can change the function, abundance and diversity of soil microbial communities. In this sense, a field study was conducted in order to assess the dissipation of two herbicides (triasulfuron and prosulfocarb) after repeated application in unamended soil and soil amended with two different green composts (GC-1 and GC-2), and under two irrigation regimes. The effect of both herbicides and composts on several biochemical parameters of soil such as microbial biomass, respiration and dehydrogenase activity was evaluated at different times during the experiment. Six treatments were studied in 9-m^2 experimental plots by triplicate. Six plots were amended with GC-1 and other six with GC-2 at a rate of 120 and 180 t/ha, respectively. Six more plots were used as controls and did not receive herbicide application. Triasulfuron and prosulfocarb were applied as a commercial formulation to the plots at doses of 250 g a.i./ha and 11.25 kg a.i./ha, respectively. Once the herbicides half-lives (DT_{50}) were achieved in all plots (after 68 days), the herbicides were applied again at the same doses. For each soil treatment, three plots did not receive irrigation while other three plots received weekly 2.5 mm. Soil samples from 0 to 10 cm were collected to determine herbicide dissipation and biochemical parameters at different sampling times. The determination of herbicide concentrations in soil extracts was performed by HPLC-MS. The dissipation of triasulfuron was slower than that of prosulfocarb. The soil amended with GC-2 showed the highest DT_{50} values for the dissipation of both herbicides. Their dissipation was faster after the second herbicide application and in irrigated soils. The application of herbicides, organic amendments and irrigation had different effects on the biochemical parameters studied.

2. INTRODUCTION

Biodegradable organic residues such as vegetal compost (GC) from pruning of plants in urban gardens and parks are applied as amendments to increase soil organic matter (OM) content. This agricultural practice can influence the physicochemical behavior of herbicides applied to soils modifying their persistence, mobility and/or degradation. However, the studies on the fate of pesticides in amended soils under field conditions are limited (Herrero-Hernández et al., 2015).

Triasulfuron and prosulfocarb are two herbicides used as pre- and post-emergence in winter cereals. Triasulfuron is a mobile herbicide in soil due to its high water solubility and low hydrophobicity (EC

2000). Prosulfocarb is a hydrophobic herbicide and presents a high adsorption, a low mobility and a moderate persistence in soil (EFSA, 2007).

The objective of this work was to assess the dissipation of triasulfuron and prosulfocarb after repeated application in unamended soil and soil amended with two GC (GC-1 and GC-2) under two irrigation regimes and field conditions. The effect of both herbicides and GC-1 or GC-2 on soil microbial biomass, respiration and dehydrogenase activity was also evaluated at different times.

3. METHODS

3.1. *Herbicides, amendments and soil*

A commercial formulation of triasulfuron (20% p/p) and prosulfocarb (80% p/v) (Auros Plus, Syngenta Agro S.A., Madrid) was used for the field study. The analytical standards of both herbicides (purity > 98.9%) were supplied by Sigma Aldrich Química S.A. (Madrid). Water solubility values are 815 and 13.0 mg L^{-1} and log Kow are -0.59 and 4.48 for triasulfuron and prosulfocarb, respectively.

The green composts used, GC-1 and GC-2, were generated from pruning of plants in gardens and parks of Salamanca (Spain). They had organic carbon (OC) contents of 9.8% and 24.1%, respectively.

The characteristics of unamended and amended soils were determined (Table 1). This soil presents a sandy clay loam texture (25.1% clay, 17.0% silt and 57.9% sand).

Table 1. Characteristics of unamended and amended soils.

	pH	OC (%)	DOC (%)	N (%)	C/N
Soil	7.35	1.30	0.006	0.12	10.8
Soil+GC-1	7.77	1.98	0.007	0.19	10.6
Soil+GC-2	7.30	4.66	0.027	0.42	11.0

3.2. *Field experiment, extraction and analysis of herbicides and data analysis*

An experimental layout of randomized complete blocks was designed with six treatments and three replicates per treatment (18 plots of 9 m^2). Unamended soil and soil amended with GC-1 or GC-2 at rates of 120 and 180 t ha^{-1} on dry weight basis, respectively, were prepared in March 2016. For each soil treatment, three plots did not receive irrigation while other three plots received weekly 2.5 mm. Six more control plots (two by treatment) did not received herbicide application, but three of them received irrigation. The commercial formulation of triasulfuron and prosulfocarb was applied to the experimental plots at doses of 250 g a.i. ha^{-1} and 11.25 kg a.i. ha^{-1}, respectively. Once the herbicides half-lives (DT_{50}) were achieved in all plots (after 68 days), the herbicides were applied again at the same doses. Soil samples from 0 to 10 cm were collected to determine herbicide dissipation at different sampling times up to 215 days (October 2016) after the first application.

Soil samples were sieved (<2 mm) and moisture content of the bulk sample was determined. Duplicate sub-samples of moist soil (6 g) were extracted with methanol (12 mL) by shaking and sonication. The determination of the herbicide concentrations in the soil extracts was performed by

HPLC-MS (Waters Assoc., Milford, USA). The molecular ions [m/z] 402.8 (triasulfuron) and 252.4 (prosulfocarb) were monitored and the retention times were 6.1 min and 14.1 min, respectively.

The dissipation kinetics was fitted to the single first-order (SFO) and first-order multicompartment (FOMC) models and dissipation parameters were estimated using the Excel solver add-in package (FOCUS, 2006).

Soil biomass, respiration and dehydrogenase activity (DHA) were determined at 0, 28, 69, 97,124 and 215 days after the initial application of the herbicides (García et al., 2003).

4. RESULTS

The dissipation kinetics of herbicides after the first and second application fitted the SFO model for most of the treatments, except in five plots where the dissipation kinetics fitted the FOMC better. The DT_{50} values for the first and second application of herbicides are included in Table 2.

Table 2. Dissipation half-lives of triasulfuron and prosulfocarb after repeated application (DT_{50}(1) and DT_{50}(2)) in unamended or amended soils under non-irrigated or irrigated conditions calculated by fitting the SFO or FOMC(*) models.

		Soil		Soil+GC-1		Soil+GC-2	
	Plot	DT_{50}(1) (days)	DT_{50}(2) (days)	DT_{50}(1) (days)	DT_{50}(2) (days)	DT_{50}(1) (days)	DT_{50}(2) (days)
Triasulfuron/	A	30.3	15.5	30.0	12.1*	146.7	24.1
Non-irrigated	B	34.1	15.0	36.0	23.5	235.5	34.9
	C	27.2	16.6	13.3*	23.3	67.5	29.7
Triasulfuron/	A	26.2	17.6	28.6	20.7	87.0	32.5
Irrigated	B	23.3	14.9	30.1	23.1	141.5	21.8
	C	26.8	16.3	14.7*	21.6	72.7	26.4
Prosulfocarb/	A	8.2	10.3	12.5	13.6	21.0	15.3
Non-Irrigated	B	11.8	11.8	11.5	9.6*	18.6	23.6
	C	8.6	10.2	9.1	10.6*	19.2	33.3
Prosulfocarb/	A	9.9	6.5	10.8	9.7	21.2	11.0
Irrigated	B	9.1	7.7	9.8	9.6	17.5	11.2
	C	9.0	7.2	8.7	11.6	18.2	13.3

The dissipation of prosulfocarb was faster than that of triasulfuron for all the treatments studied. In general, the DT_{50} values for the dissipation of triasulfuron and prosulfocarb in the soils followed the order: soil ~ soil+GC-1 < soil+GC-2. However, the DT_{50} values after the second application of herbicides in soils under irrigation were lower in unamended soil than in amended soils. These results could be related with the higher OC content of GC-2, which could contribute to the increase of the herbicide adsorption by soil and formation of bound residues and a decrease in its bioavailability and biodegradation (Gennari et al., 2002; Said-Pullicino et al., 2004). The dissipation rate of both herbicides was faster in irrigated soils, indicating the possible herbicide leaching through soil profile due to irrigation. After the second application of triasulfuron to soil its dissipation rates increased while those of prosulfocarb were similar or increased slightly.

Soil biomass and DHA increased up to 69 days and after this time these parameters decreased which could be related with the soil moisture content and precipitations registered. Soil respiration also increased up to 28 days and decreased after. In general, the biochemical parameters were higher in soil amended with GC-1 and GC-2. Soil biomass increased in unamended soil or decreased in the amended soils treated with herbicides. Soil respiration increased after the first application of herbicides. However, DHA decreased in the soils treated with the herbicides. Irrigation affected in a different way to the biochemical parameters in unamended (decreased) or amended soils (increased or similar values) with regard to non-irrigated soils.

5. CONCLUSIONS

The dissipation of triasulfuron and prosulfocarb in an agricultural soil under field conditions was influenced by the type of GC applied to soil and by the irrigation regime. The dissipation of prosulfocarb in all soils was faster due possibly to its higher retention. Soil biochemical parameters were affected by application of organic amendments, herbicides and irrigation.

6. ACKNOWLEDGEMENTS

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2.6.O

EXPERIENCES IN SCALING UP SOIL AND WATER CONSERVATION MEASURES TO LANDSCAPE LEVEL IN AFRICA

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INTRODUCTION

Soil is a nonrenewable resource. Landscape restoration, the rehabilitation and protection of productive soils and more efficient use of rainwater are essential conditions for food security for a growing world population. They are a pathway to adapt agriculture and ecosystems to climate change particularly in developing countries with high exposure to climate change. However, to gain significant impacts, large-scale implementation is needed. What principles apply to transform small programme initiatives working with some producer groups or villages into programmes covering hundreds of villages?

What are results and lessons learnt with regard to yields and ecosystem services, technical, organizational and managerial principles and institutional frameworks to be successful?

METHODS

During the last 25 years, German Development Cooperation has improved 2.2 million hectares (56 % Africa, 41 % Asia and 47 % fields, 34 % forest, 11 % grazing lands, 8 % irrigated area), equal to a belt around the globe of 500 m width.

Table 1: Areas rehabilitated and improved during the last 25 years (Lehmann, 2016)

Region	ha fields (incl. agroforestry)	ha grazing area	ha small-scale irrigation	ha forested area	Total ha	%
Africa	905.859	172.467	134.915	18.005	1.231.246	56%
Asia	103.827	69.071	15.888	713.147	901.934	41%
Latin America and Caribbean	27.579	0	17.520	7.725	52.824	2%
Total ha	1.037.265	241.538	168.323	738.877	2.186.004	100%
%	47%	11%	8%	34%		

Another 180.000 hectares in five countries are currently under improvement since 2014, by the Programme Soil Protection and Rehabilitation for Food Security within the BMZ Special Initiative ONE WORLD – No Hunger. The Programme has currently a budget of 72.1 million EUR for the period 2015-2021 with the following measures:

- Ethiopia: (i) Semi-arid Afar Lowland Region: Water spreading weirs, land use planning for the protection of micro-watersheds and rotation of grazing areas, soil and water conservation to manage spring floods and improve land for grazing and agricultural usage, field trials for adapted varieties, etc. (ii) Ethiopian highland areas: integrated soil fertility management with liming, targeted organic and mineral fertilizer, intercropping, development of adapted fertilizer recommendations (organic and mineral).

- Benin: Soil and water conservation with biological and physical methods, soil fertility management, compost, agroforestry and leguminous trees, farmer business schools with integration of soil aspects as part to the business model.
- Burkina Faso: Soil and water conservation with biological and physical rehabilitation methods, agroforestry, improved framework conditions for soil protection through improved access to land and investment incentives.
- India: Integrated soil fertility management, soil health cards, targeted fertilization, improved agronomic practices (row seeding, weeding), ICT for agro-meteorology and soil data combination and improved agricultural advisory.
- Kenya: Watershed approach with physical and biological rehabilitation methods, conservation agriculture, liming, soil fertility management (compost).

Partners are government services, national and international research and NGOs.

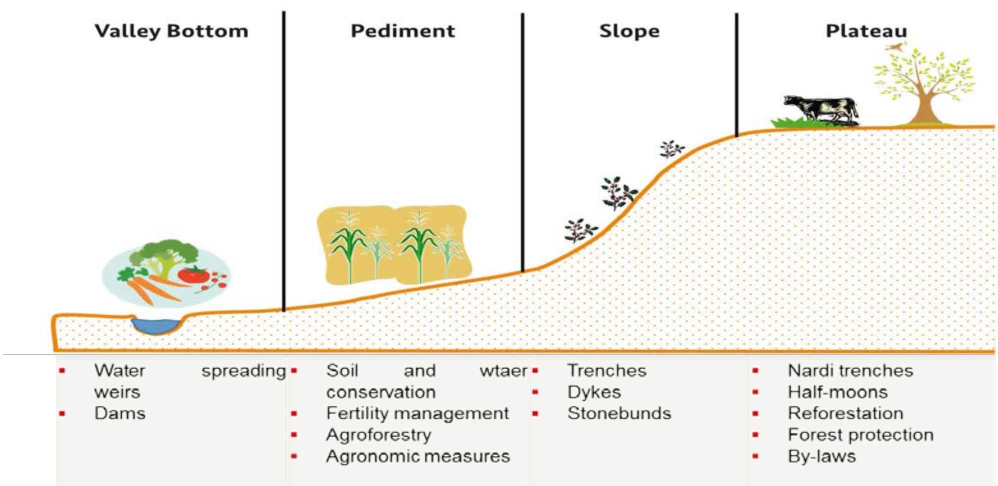
The following working steps turned out to be essential in such landscape approaches:

a) Participatory land-use planning at community, watershed and communal level and establishment of restoration and conservation plans.
b) Organization of resource user / community groups and provision of organizational and technical trainings.
c) User agreements on regulation and management of common grazing and forested lands (by-laws, joint forest management).
d) Clarification on resource and land-use rights i.e. by-laws, (provisional) land titles.
e) Joint implementation of restoration and conservation plans with substantial user contribution (labor, local materials) with specific technology packages for different landscape units.
f) Establishment of market linkages.

Figure 1 gives a schematic overview of a typical approach applied in West Africa according to the landscape units plateaus, slopes, hill foots (pediments) and valley bottoms:

- Plateaus with grazing areas: erosion control, pasture improvement (seeding), zero grazing areas and seed banks, cut and carry systems, grazing and use regulations, tree protection and planting.

Figure 1: Specific technology packages for different landscape units



- Plateaus and slopes with forested areas: Erosion control, reforestation, forest protection, regulated use of wood, fodder and non-timber forest products, pruning techniques.
- Foot hills with fields: Soil- and water conservation, grass strips, live hedges, terracing, protection of natural tree regrowth/agroforestry, soil fertility management and agronomic measures (Dorlöchter-Sulser and Nill, 2012).
- Valley bottoms with fields and gardens: construction of micro dams, water-spreading weirs (GIZ/KfW, 2012) and small-scale irrigation systems for production of staple crops and vegetables.

RESULTS

The implementation of such programmes has a number of economic, ecological and social impacts:

- Yield increases in rain-fed crops e.g. Niger: millet grain yield x2.2 (182 kg/ha to 392 kg/ha) and irrigated crops e.g. millet grain yield irrigated x2.0, millet production x6 (Mamadou et al., 2015).
- Improved availability of fodder, e.g. herbaceous biomass on plateaus x14 (57 kg/ha to 778 kg/ha).
- More availability of fire wood / timber and non-timber forest products.
- Increased production through reintegration of degraded soils into agricultural use.
- Improved and diversified food and incomes (Table 2).

Table 2: Gross margins (EUR per hectare and cropping cycle) from 1 hectare without and with SLM practices in Ethiopia (Deichert, 2016)

Crop	Non-SLM micro watershed	SLM micro watershed	Difference in EUR	Factor increase
Tef	57	211	154	x3,7
Maize	55	103	48	x1,9
Wheat	37	211	175	x5,7
Millet	-47	-6	40	x7,8

- Livestock carrying capacity is increased due to higher fodder availability.
- More climate resilience of environment and households.
- Higher biodiversity (wild life, natural species, agro-biodiversity).
- Renewed ecosystem services (bees for pollination, flood control, groundwater availability, micro-climate).
- Better trained farmers, workers and craftsmen and more jobs especially for young people.
- Stronger participation of women in decision taking positions and higher esteem.
- Often higher social coherence and transparency in resource use in villages.

CHALLENGES

What are success factors in programs so far? The following challenges need to be tackled in order to arrive at large-scale implementation and large-scale impacts (Deichert et al., 2014):

- Strong ownership and leadership of Government (integration into national policies), providing responsive services and orientation.

- The programmes need long duration (15 to 25 years) to transform economies, management practices, attitudes and institutions. Adapted approaches need time to develop and to be translated into efficient implementation. Works in one watershed take 5 to 7 years in order to arrive at significant area coverage.
- A long-term funding mechanism has to be established to ensure reliable incentives for sustainable management (e.g. multi-donor funds, basket funds, payment for environmental services schemes).
- Stakeholder communities need to be actively involved in all stages of planning, implementation and management of watershed development activities and contribute to the works to create ownership also for later maintenance.
- Women involvement in watershed-development planning, implementation and management is key.
- Planning for watershed management activities should be realistic, based upon local capacity, locally available resources and other forms of government and partner support.
- Watershed development activities should provide tangible and quick benefits to households.
- Watersheds sizes must be reasonable and clustered for efficient management. Feasible sizes were in the range of 500 ha.

CONCLUSIONS

Large-scale landscape rehabilitation and protection programs allow significant and long-term contributions to food security and enable rural populations to invest hundred thousands of labour days in soil fertility restoration. They improve income and employment opportunities and strengthen climate resilience of rural populations. They contribute to gender balance and social well-being and increase the attractiveness of rural areas. They reestablish eco-system services and improve biodiversity.

Implementing them follows certain rules i.e. strong long-term political will and funding by national governments and international donor community, engagement and participation of the local populations and site-specific technical solutions.

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2.7.O

ADOPTION OF SOIL CONSERVATION PRACTICES IN MEXICAN AGROECOSYSTEMS

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INTRODUCTION

The deterioration of soil quality is one of the most severe problems that have affected the Mexican countryside, limiting food sovereignty and the ability of agroecosystems to adapt to climate variability. Poor soil conditions may explain why, in Mexico, 48.6% of agricultural units encounter agricultural production problems (INEGI, 2012), hence contributing to poverty and rural migration. The deterioration of soil quality may also have been one of the causes of the abandonment of 1.4 million agricultural production units in the country.

Farmers and livestock breeders working in different contexts have developed innovative strategies to improve soil quality and deal with climate variability. In this respect, the restoration of traditional management practices can provide elements to develop adaptive strategies to respond to climate change (Astier et al. 2012).

Despite decades of government soil conservation programs, the adoption of such practices by Mexican farmers—a basic premise for soil conservation—remains low.

Because of rural Mexico's topographical, ecological, social and institutional diversity, a strategy that works at a given site may be totally inadequate at another site. Among the questions that still need to be answered are: How is soil conservation done in Mexican agroecosystems? Is the choice of soil conservation practices affected by the environmental, social and institutional conditions specific to each site? What are the main obstacles to carrying out these practices, and how are they overcome?

The main objectives of this study were to identify the work being carried out by peasants, farmers, livestock breeders and social organizations to adapt traditional soil conservation practices to different environmental, social and institutional conditions; to understand their motivations and expectations in undertaking this work; and to see how the obstacles encountered are overcome.

METHODS

To gather the necessary information while taking Mexico's heterogeneity into account, a survey was designed and implemented that covered the following four main topics: (i) the most common soil conservation practices in each type of agroecosystem; (ii) the main socio-environmental characteristics of the sites where these practices are carried out; (iii) the methods for disseminating and assessing these practices; and (iv) the main obstacles encountered, and means of overcoming them.

The survey included 42 open- and closed-ended questions and was conducted by three means: (i) the website *encuestasdinámicas.com*; (ii) email; and (iii) face to face, when applied to peasants without access to electronic media. The survey described the soil conditions at one site; the same person or organization could thus answer more than one survey. The survey was conducted for a period of 6 months.

RESULTS

A total of 65 responses were obtained, which included soil information for 91 sites distributed in 20 Mexican states (Figure 1). Over 39% of the returned surveys were from the dry tropics; 30% from regions with temperate climate; 27% from the humid tropics; and 3% from regions with arid climate.

In most cases (70%), soil conservation practices were carried out on soils that were already degraded, with the aim of recovering their properties and functions and improving yields. These practices were thus used as corrective rather than preventive measures for soil erosion.

A large part of the assessed agricultural systems were established on communal lands (*ejidos*; 50%) or the land of communities (28%), with an area ranging from 1 to 3 ha. The area of privately owned lands, in contrast, could exceed 20 ha.

Figure 1. Sites where the survey was conducted and their production system

Agricultural systems were primarily for self-subsistence in conjunction with the sale of surplus products, except for one case where production was destined for domestic and international markets. The production of silvopastoral systems, for its part, was mainly destined for the regional market.

Sixty-nine percent (69%) of the reported soil conservation practices were already known by land owners, or designed for the site's specific socio-environmental conditions by experts from civil society organizations. In agricultural systems, the most widely accepted practices were crop rotation, intercropping, the application of organic material and, less importantly, the construction

of terraces. In silvopastoral systems, the most common practices were the establishment of living fences, the control of animal load, pasture rotation and the planting of shrubs, along with mechanical practices for water storage.
After five years of soil conservation, the most obvious benefits reported were from, on the one hand, the income generated by increased yield and crop diversification, which allowed sporadic but constant sales throughout the year; and on the other hand, improved soil quality through decreased soil erosion and increased organic material and infiltration rate.

Regardless of land tenure, 97% of respondents identified social organization—in the form of associations, groups of neighbors, *ejidos* and cooperatives—as a basic condition for carrying out soil conservation actions. Indeed, organized work allowed to reduce costs, share knowledge, expand collaborative networks and share risks. All these forms of organization had to be strengthened, however, so that they could operate with clear rules.

In both agricultural and silvopastoral systems, the main obstacle to developing soil conservation practices was economic (Figure 2), making it highly dependent on government grants and highlighting the need for alternative funding sources. This situation, added to cultural factors such as the fear of change, lack of social cohesion and absence of technical support, led to a difficult and slow acceptance of soil conservation practices locally.

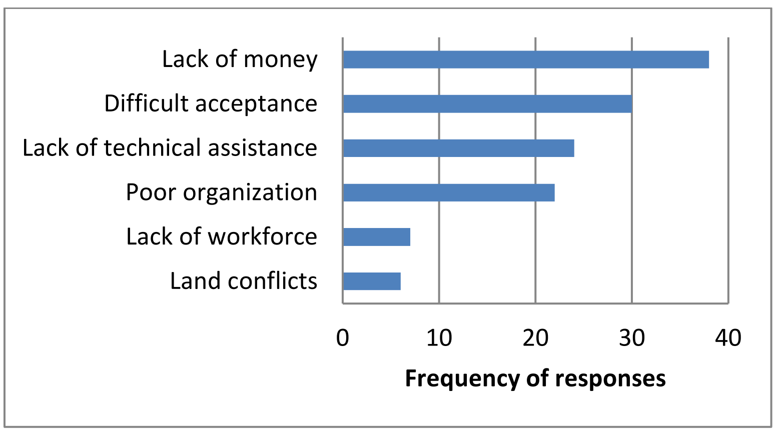

Figure 2. Main obstacles to the implementation of soil conservation practices in Mexican agroecosystems

Different alternatives had been built to overcome these obstacles. Partnerships with governmental and non-governmental organizations were particularly important in this regard for providing both technical assistance and training through courses, workshops and the exchange of experience between peasants, as well as ensuring ongoing assessment of the environmental and economic results achieved.

CONCLUSIONS

This study showed that the process of adopting soil conservation practices is site specific, the attitudes and personal incentives of land owners playing an important role in it. The survey exposed a variety of soil conservation strategies carried out under different land tenure conditions, on different plot sizes, for different product destinations and under different climates (Cotler and Cuevas 2017).

In the cases surveyed, soil conservation practices were implemented in a context of scarce human and financial resources. The long-term viability of these practices largely depended on their results, which could be assessed taking into account yields, the possibility of diversifying products and markets, as well as the improvement of soil quality.

The soil conservation practices reported in the surveyed agricultural and livestock systems mainly used agronomic techniques and vegetation, respectively, occasionally completed by hydraulic infrastructure works (e.g., dams, gabions and levees). These results showed that the practices actually adopted largely differed from the practices supported and implemented by government soil conservation programs, which mostly consist of mechanical actions and do not consider the site's biophysical conditions (Cotler *et al.* 2013).

The main motivation of land owners for practicing soil conservation was economic. In this regard, the relationship between soil quality and yield is clearly established. It was also recognized that soil conservation generates benefits from increased infiltration rate, organic material and agrobiodiversity. Less visible but equally important were the social benefits from implementing these practices. A number of respondents, indeed, reported that their social status was improved because they were perceived as inventive people, with an ability to teach their family and unite it around an innovative project.

The results obtained here call for a reassessment of how public policies regarding soil conservation are designed, with a view to make them preventive rather than corrective and better reflect Mexico's diverse environmental, social and institutional conditions.

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2.9.O

CONSERVATION AGRICULTURE – PROBLEMS, PROSPECTS AND POLICY ISSUES IN INDIAN CONTEXT

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ABSTRACT

India has the unenviable task of feeding a global population of 17.5% with only 4% of the water of 2.3% of the Land Resources at its disposal. It is also been estimated that 120.7 million hectare of the land is effected by various[1] degrees of soil and water degradation to ensure food nutritional and livelihood security to the second largest population in the world. The Government of India has initiated many programmes in many of which are Conservation Agriculture (CA).

CA technologies involve minimum soil disturbances, permanent soil cover through crop residues or cover crops, and crop rotations for achieving higher productivity. In India, effort to develop, refine and disseminate conservation-based agricultural technologies have been underway for nearly two decades and made significant progress since then even though there are several constraints that affect adoption of CA. Particularly, tremendous efforts have been made on no-till in wheat under a rice-wheat rotation in the Indo-gangetic plains. There are more payoffs than tradeoffs for adoption of CA but the equilibrium among the two was understood by both adopters and promoters. The technologies of CA provide opportunities to reduce the cost of production, save water and nutrients, increase yields, increase crop diversification, improve efficient use of resources, and benefit the environment. However, there are still constraints for promotion of CA technologies, such as lack of appropriate seeders especially for small and medium scale farmers, competition of crop residues between CA use and livestock feeding, burning of crop residues, availability of skilled and scientific manpower an overcoming the bias or mindset about tillage. The need to develop the policy frame and strategies is urgent to promote CA in the region. This article reviews the emerging concerns due to continuous adoption of conventional agriculture systems, and analyses the constraints, prospects, policy issues and research needs for conservation agriculture in India.

KEY WORDS:

Conservation agriculture; Conventional agriculture; Prospects and policy of CA adoption; Resource use efficiency; Zero tillage

1 INTRODUCTION

In order to meet food and energy of a growing population and under the current scenario of depleting natural resources, adverse impacts of climatic change, continuous change in cost of inputs and liable to change rapidly in food prices are the major challenges before the Asian countries. In addition to these challenges, the principal indicators includes: soil erosion, soil organic matter decline, salinization. These are caused mainly by: (i) intensive tillage induced soil organic matter decline, soil structural degradation, water and wind erosion, reduced water infiltration rates, surface sealing and crusting, soil compaction, (ii) insufficient return of organic material, and (iii) monocropping. Therefore, a paradigm shift in farming practices through eliminating unsustainable parts of conventional agriculture (ploughing/tilling the soil, removing all organic material, monoculture) is crucial for future productivity gains while sustaining the natural resources. Conservation agriculture (CA), a concept evolved as a response to concerns of sustainability of agriculture globally, has steadily increased worldwide to cover about ~8% of the world arable land (124.8 M ha) (FAO, 2012). CA is a resource-saving agricultural production system that aims to achieve production intensification and high yields while enhancing the natural resource base through compliance with three interrelated principles, along with other good production practices of plant nutrition and pest management (Abrol and Sangar, 2006).

Traditional agriculture, based on tillage and being highly mechanized, has been cause of being responsible for soil erosion problems, surface and underground water pollution, and more water consumption (Wolff and Stein, 1998). Moreover, it is implicated in land resource degradation, wildlife and biodiversity reduction, low energy efficiency and contribution to global warming problems (Boatmann et al., 1999). Hence, conservation agriculture (CA) is a way to cultivate annual and perennial crops, based on no vertical perturbation of soil (zero and conservation tillage), with crop residue management and cover crops, in order to offer a permanent soil cover and a natural increase of organic matter content in surface horizons. The main environmental consequences of this method have been investigated worldwide with the objective of presenting a synthesis of the available studies and documents to the farmers and scientific communities. It stresses the very beneficial impacts of a conservative way of cultivation on the global environment (soil, air, water and biodiversity), compared to traditional agriculture (Derpsch et al., 2010; Derpsch et al., 2011). Further, it also presents the actual gaps or uncertainties concerning the scientists' positions on these environmental aspects.

Conservation agriculture is a management system that maintains a soil cover through surface retention of crop residues with no till/zero and reduced tillage. CA is described by FAO (http://www.fao.org.ag/ca) as a concept for resource saving agricultural crop production which is based on enhancing the natural and biological processes above and below the ground. As per Dumanski et al. (2006) conservation agriculture (CA) is not "business as usual", based on maximizing yields while exploiting the soil and agro-ecosystem resources. Rather, CA is based on optimizing yields and profits, to achieve a balance of agricultural, economic and environmental benefits. It advocates that the combined social and economic benefits gained from combining production and
Protecting the environment, including reduced input and labor costs, is greater than those from production alone.

As per FAO definition CA is to i) achieve acceptable profits, ii) high and sustained production levels, and iii) conserve the environment. It aims at reversing the process of degradation inherent to the conventional agricultural practices like intensive agriculture, burning/removal of crop residues. Hence, it aims to conserve, improve and make more efficient use of natural resources through

integrated management of available soil, water and biological resources combined with external inputs. It can also be referred to as resource efficient or resource effective agriculture.
Conservation agriculture systems require a total paradigm shift from conventional agriculture with regard to management of crops, soil, water, nutrients, weeds, and farm machinery (Table 1).

Table 1. Some distinguishing features of conventional and conservation agriculture systems

Conventional agriculture	Conservation agriculture
● Cultivating land, using science and technology to dominate nature	● Least interference with natural processes
● Excessive mechanical tillage and soil erosion	● No-till or drastically reduced tillage (biological tillage)
● High wind and soil erosion	● Low wind and soil erosion
● Residue burning or removal (bare surface	● Surface retention of residues (permanently covered)
● Water infiltration is low	● Infiltration rate of water is high
● Use of *ex-situ* FYM/composts	● Use of *in-situ* organics/composts
● Green manuring (incorporated)	● Brown manuring/cover crops (surface retention)
● Kills established weeds but also stimulates more weed seeds to germinate	● Weeds are a problem in the early stages of adoption but decrease with time
● Free-wheeling of farm machinery, increased soil compaction	● Controlled traffic, compaction in tramline, no compaction in crop area
● Mono cropping/culture, less efficient rotations	● Diversified and more efficient rotations
● Heavy reliance on manual labor, uncertainty of operations	● Mechanized operations, ensure timeliness of operations
● Poor adaptation to stresses, yield losses greater under stress conditions	● More resilience to stresses, yield losses are less under stress conditions
● Productivity gains in long-run are in declining order	● Productivity gains in long-run are in incremental order

2. PRINCIPLES OF CONSERVATION AGRICULTURE

Conservation agriculture practices perused in many parts of the world are built on ecological principles making land use more sustainable (Wassmann, 2009; Behera et al., 2010; Lal, 2013). Adoption of CA for enhancing Resource use efficiency (RUE) and crop productivity is the need of the hour as a powerful tool for management of natural resources and to achieve sustainability in agriculture. Conservation agriculture basically relies on 3 principles, which are linked and must be considered together for appropriate design, planning and implementation processes. These are:

2.1 *Minimal mechanical soil disturbance*

The soil biological activity produces very stable soil aggregates as well as various sizes of pores, allowing air and water infiltration. This process can be called “biological tillage” and it is not compatible with mechanical tillage. With mechanical soil disturbance, the biological soil structuring processes will disappear. Minimum soil disturbance provides/maintains optimum proportions of respiration gases in the rooting-zone, moderate organic matter oxidation, porosity for water movement, retention and release and limits the re-exposure of weed seeds and their germination (Kassam and Friedrich, 2009).

2.2 *Permanent organic soil cover*
A permanent soil cover is important to protect the soil against the deleterious effects of exposure to rain and sun; to provide the micro and macro organisms in the soil with a constant supply of “food”; and alter the microclimate in the soil for optimal growth and development of soil organisms, including plant roots. In turn it improves soil aggregation, soil biological activity and soil biodiversity and carbon sequestration (Ghosh et al., 2010).

2.3 *Diversified crop rotations*
The rotation of crops is not only necessary to offer a diverse “diet” to the soil micro organisms, but also for exploring different soil layers for nutrients that have been leached to deeper layers that can be “recycled” by the crops in rotation. Furthermore, a diversity of crops in rotation leads to a diverse soil flora and fauna. Cropping sequence and rotations involving legumes helps in minimal rates of build-up of population of pest species, through life cycle disruption, biological nitrogen fixation, control of off-site pollution and enhancing biodiversity (Kassam and Friedrich, 2009; Dumanski et al., 2006).

3. Status of conservation agriculture in India and abroad

Globally, CA is being practiced on about 125 M ha (Table 2). The major CA practicing countries are USA (26.5 M ha), Brazil (25.5 M ha), Argentina (25.5 M ha), Canada (13.5 M ha) and Australia (17.0 M ha). In India, CA adoption is still in the initial phases. Over the past few years, adoption of zero tillage and CA has expanded to cover about 1.5 million hectares (Jat et al., 2012; www.fao.org/ag/ca/6c.html). The major CA based technologies being adopted is zero-till (ZT) wheat in the rice-wheat (RW) system of the Indo-Gangetic plains (IGP). In other crops and cropping systems, the conventional agriculture based crop management systems are gradually undergoing a paradigm shift from intensive tillage to reduced/zero-tillage operations. In addition to ZT, other concept of CA need to be infused in the system to further enhance and sustain the productivity as well as to tap new sources of growth in agricultural productivity. The CA adoption also offers avenues for much needed diversification through crop intensification, relay cropping of sugarcane, pulses, vegetables etc. as intercrop with wheat and maize and to intensify and diversify the RW system. The CA based resource conservation technologies (RCTs) also help in integrating crop, livestock, land and water management research in both low and high-potential environments.

Table 2. Global adoption of conservation agriculture systems

Country	Area (M ha)	% of Global Area
USA	26.5	21.2
Brazil	25.5	20.4
Argentina	25.5	20.4
Australia	17.0	13.6
Canada	13.5	10.8
Russian Federation	4.5	3.6
China	3.1	2.5
Paraguay	2.4	1.9
Kazakhstan	1.6	1.3
Others	5.3	4.2
Total	124.8	100.0

In India, efforts to adopt and promote conservation agriculture technologies have been underway for nearly a decade but it is only in the last 8 – 10 years that the technologies are finding rapid acceptance by farmers. Efforts to develop and spread conservation agriculture have been made through the combined efforts of several State Agricultural Universities, ICAR institutes and the Rice-Wheat Consortium for the Indo-Gangetic Plains. The spread of technologies is taking place in India in the irrigated regions in the Indo-Gangetic plains where
Rice-wheat cropping systems dominate. Conservation agriculture systems have not been tried or promoted in other major agro-ecoregions like rainfed semi-arid tropics and the arid regions of the mountain agro-ecosystems. Spread of these technologies is taking place in the irrigated regions of the Indo-Gangetic plains where the rice-wheat cropping system dominates. The focus of developing and promoting conservation technologies has been on zero-till seed-cum fertilizer drill for sowing of wheat in rice-wheat system. Other interventions include raised-bed planting systems, laser equipment aided land leveling, residue management practices, alternatives to the rice-wheat system etc. It has been reported that the area planted with wheat adopting the zero-till drill has been increasing rapidly (Sangar et al., 2005), and presently 25% – 30% of wheat is zero-tilled in rice-wheat growing areas of the Indo-Gangetic plains of India. In addition, raised-bed planting and laser land levelling are also being increasingly adopted by the farmers of the north-western region.

4. POTENTIAL BENEFITS OF CA

Adoption and spread of ZT wheat has been a success story in North-western parts of India due to (1) reduction in cost of production by Rs 2,000 to 3,000 ha-1 ($ 33 to 50) (Malik et al., 2005; RWC-CIMMYT, 2005); (2) enhancement of soil quality, i.e. soil physical, chemical and biological conditions (Jat et al., 2009a; Gathala et al., 2011b); (3) enhancement, in the long term C sequestration and build-up in soil organic matter constitute a practical strategy to mitigate Green House Gas emissions and impart greater resilience to production systems to climate change related aberrations (Saharawat et al., 2012); (4) reduction of the incidence of weeds, such as *Phalaris minor* in wheat (Malik et al., 2005); (5) enhancement of water and nutrient use efficiency (Jat et al., 2012; Saharawat et al., 2012); (6) enhancement of production and productivity (4% – 10%) (Gathala et al., 2011a); (7) advanced sowing date (Malik et al., 2005); (8) reduction in greenhouse gas emission and improved environmental sustainability (Pathak et al., 2011); (9) avoiding crop residue burning reduces loss of nutrients, and environmental pollution, which reduces a serious health hazard (Sidhu et al., 2007);

(10) providing opportunities for crop diversification and intensification-for example in sugarcane based systems, mustard, chickpea, pigeonpea etc. (Jat et al., 2005); (11) improvement of resource use efficiency through residue decomposition, soil structural improvement, increased recycling and availability of plant nutrients (Jat et al., 2009a); and (12) use surface residues as mulch to control weeds, moderate soil temperature, reduce evaporation, and improve biological activity (Jat et al., 2009b; Gathala et al., 2011b). Because of the ZT wheat benefits, the CA based crop management technologies have been tried in other cropping systems in India (Jat et al., 2011), but there are large knowledge gaps in CA based technologies which indicates there is a need to develop, refine, popularize and disseminate these technologies on a large scale. Zero tillage is a technology where the crop is sown in a single tractor operation using a specially designed seed-cum-fertilizer drill without any field preparation in the absence of anchored residue at optimum to slightly wetter soil moisture regimes. Experiences from several locations in the Indo-Gangetic plains showed that with zero tillage technology farmers were able to save on land preparation costs by about Rs. 2,500 ($41.7) per ha and reduce diesel consumption by 50 – 60 litres per ha (Sharma et al., 2005). Zero tillage allows timely sowing of wheat, enables uniform drilling of seed, improves fertilizer use-efficiency, saves water and increases yield up to 20%. Success has also been achieved in bed planting of wheat, cotton and rice. This has resulted in savings in irrigation water, improved fertilizer use and reduced soil crusting.

5. PROSPECTS OF CONSERVATION AGRICULTURE

The direction that Asian countries take to meet their food and energy needs during the coming decades will have profound impacts on natural resource bases, global climate change and energy security for India, Asia and the world. These challenges draw attention to the need and urgency to address options by which threats to Indian/Asian agriculture due to natural resource degradation, escalating production costs and climate change can be met successfully. A shift to no-till conservation agriculture is perceived to be of much fundamental value in meeting these challenges. Asian farmers/researchers will continue to need assistance to reorient their agriculture and practices for producing more with less cost through adoption of less vulnerable choices and pathways. Therefore, business as usual with conventional agriculture practices does not seem a sustainable option for sustainable gains in food-grain production, and hence CA-based crop management solutions adapted to local needs will have to play a critical role in most ecological and socio-economic settings of Asian agriculture. The promotion of CA under Indian/Asian context has the following prospects:

(i) Reduction in cost of production – This is a key factor contributing to rapid adoption of zero-till technology. Most studies showed that the cost of wheat production is reduced by Rs. 2,000 to 3,000 ($ 33 to 50) per hectare (Malik et al., 2005; RWC-CIMMYT, 2005). Cost reduction is attributed to savings on account of diesel, labour and input costs, particularly herbicides.

(ii) Reduced incidence of weeds – Most studies tend to indicate reduced incidence of *Phalaris minor*, a major weed in wheat, when zero-tillage is adopted resulting in reduced in use of herbicides.

(iii) Saving in water and nutrients – Limited experimental results and farmers experience indicate that considerable saving in water (up to 20% – 30%) and nutrients are achieved with zero-till planting and particularly in laser leveled and bed planted crops. De Vita et al. (2007) stated that higher soil water content under no-till than under conventional tillage indicated the reduced water evaporation during the preceding period. They also found that across growing seasons, soil water content under no-till was about 20% greater than under conventional tillage.

(iv) Increased yields – In properly managed zero-till planted wheat, yields were invariably higher compared to traditionally prepared fields for comparable planting dates. CA has been reported to enhance the yield level of crops due to associated effects like prevention of soil degradation, improved soil fertility, improved soil moisture regime (due to increased rain water infiltration, water holding capacity and reduced evaporation loss) and crop rotational benefits. Yield increases as high as 200 – 500 kg ha-1 are found with no-till wheat compared to conventional wheat under a rice-wheat system in the Indo-Gangetic plains (Hobbs and Gupta, 2004). Review of the available literature on CA provides mixed indications of the effects of CA on crop productivity. While some studies claim that CA results in higher and more stable crop yields (African Conservation Tillage Network, 2011), on the other hand there are also numerous examples of no yield benefits and even yield reductions particularly during the initial years of CA adoption.

(v) Environmental benefits – Conservation agriculture involving zero-till and surface managed crop residue systems are an excellent opportunity to eliminate burning of crop residue which contribute to large amounts of greenhouse gases like CO_2, CH_4 and N_2O. Burning of crop residues, also contribute to considerable loss of plant nutrients, which could be recycled when properly managed. Large scale burning of crop residues is also a serious health hazard.

(vi) Crop diversification opportunities – Adopting Conservation Agriculture systems offers opportunities for crop diversification. Cropping sequences/rotations and agroforestry systems when adopted in appropriate spatial and temporal patterns can further enhance natural ecological processes. Limited studies indicate that a variety of crops like mustard, chickpea, pigeonpea, sugarcane, etc., could be well adapted to the new systems.

(vii) Resource improvement – No tillage when combined with surface management of crop residues begins the processes whereby slow decomposition of residues results in soil structural improvement and increased recycling and availability of plant nutrients. Surface residues acting as mulch, moderate soil temperatures, reduce evaporation, and improve biological activity.

6. LIMITATION/CONSTRAINTS FOR ADOPTION OF CONSERVATION AGRICULTURE

A mental change of farmers, technicians, extensionists and researchers away from soil degrading tillage operations towards sustainable production systems like no tillage is necessary to obtain changes in attitudes of farmers (Derpsch, 2001). Hobbs and Govaerts (2010) however, noted that probably the most important factor in the adoption of CA is overcoming the bias or mindset about tillage. It is argued that convincing the farmers that successful cultivation is possible even with reduced tillage or without tillage is a major hurdle in promoting CA on a large scale. Spread of conservation agriculture, therefore, will call for scientific research linked with development efforts. The following are a few important constraints which impede broad scale adoption of CA.

(i) Lack of appropriate seeders especially for small and medium scale farmers: Although significant efforts have been made in developing and promoting machinery for seeding wheat in no till systems, successful adoption will call for accelerated effort in developing, standardizing and promoting quality machinery aimed at a range of crop and cropping sequences. These would include the development of permanent bed and furrow planting systems and harvest operations to manage crop residues.

(ii) The wide spread use of crop residues for livestock feed and fuel: Specially under rainfed situations, farmers face a scarcity of crop residues due to less biomass production of different crops. There is competition between CA practice and livestock feeding for crop residue. This is a major constraint for promotion of CA under rainfed situations.

(iii) Burning of crop residues: For timely sowing of the next crop and without machinery for sowing under CA systems, farmers prefer to sow the crop in time by burning the residue. This has become a common feature in the rice-wheat system in north India. This creates environmental problems for the region.

(iv) Lack of knowledge about the potential of CA to agriculture leaders, extension agents and farmers: This implies that the whole range of practices in conservation agriculture, including planting and harvesting, water and nutrient management, diseases and pest control etc. need to be evolved, evaluated and matched in the context of new systems.

(v) Skilled and scientific manpower: Managing conservation agriculture systems, will call for enhanced capacity of scientists to address problems from a systems perspective and to be able to work in close partnerships with farmers and other stakeholders. Strengthened knowledge and information sharing mechanisms are needed.

7. CHALLENGES IN CONSERVATION AGRICULTURE

Conservation agriculture as an upcoming paradigm for raising crops will require an innovative system perspective to deal with diverse, flexible and context specific needs of technologies and their management. Conservation agriculture R&D (Research and Development), thus will call for several innovative features to address the challenge. Some of these are:

(a) Understanding the system – Conservation agriculture systems are much more complex than conventional systems. Site specific knowledge has been the main limitation to the spread of CA system (Derpsch, 2001). Managing these systems efficiently will be highly demanding in terms of understanding of basic processes and component interactions, which determine the whole system performance. For example, surface maintained crop residues act as mulch and therefore reduce soil water losses through evaporation and maintain a moderate soil temperature regime (Gupta and Jat, 2010). However, at the same time crop residues offer an easily decomposable source of organic matter and could harbour undesirable pest populations or alter the system ecology in some other way. No-tillage systems will influence depth of penetration and distribution of the root system which, in turn, will influence water and nutrient uptake and mineral cycling. Thus the need is to recognize conservation agriculture as a system and develop management strategies.

(b) Building a system and farming system perspective – A system perspective is built working in partnership with farmers. A core group of scientists, farmers, extension workers and other stakeholders working in partnership mode will therefore be critical in developing and promoting new technologies. This is somewhat different than in conventional agricultural R&D, the system is to set research priorities and allocate resources within a framework, and little attention is given to build relationships and seek linkages with partners working in complementary fields.

(c) Technological challenges **–** While the basic principles which form the foundation of conservation agriculture practices, that is, no tillage and surface managed crop residues are well understood, adoption of these practices under varying farming situations is the key challenge. These challenges relate to development, standardization and adoption of farm machinery for seeding with minimum soil disturbance, developing crop harvesting and management systems.

(d) Site specificity **–** Adapting strategies for conservation agriculture systems will be highly site specific, yet learning across the sites will be a powerful way in understanding why certain technologies or practices are effective in a set of situations and not effective in another set. This learning process will accelerate building a knowledge base for sustainable resource management.

(e) Long-term research perspective – Conservation agriculture practices, e.g. no-tillage and surface maintained crop residues result in resource improvement only gradually, and benefits come about only with time. Indeed in many situations, benefits in terms of yield increase may not come in the early years of evaluating the impact of conservation agriculture practices. Understanding the dynamics of changes and interactions among physical, chemical and biological processes is basic to developing improved soil-water and nutrient management strategies (Abrol and Sangar, 2006). Therefore, research in conservation agriculture must have longer term perspectives.

8. POLICY ISSUES

Conservation agriculture implies a radical change from traditional agriculture. There is need for policy analysis to understand how CA technologies integrate with other technologies, and how policy instruments and institutional arrangements promote or deter CA (Raina et al., 2005). CA offers an opportunity for arresting and reversing the downward spiral of resource degradation, diminishing factor productivity, decreasing cultivation costs and making agriculture more resource – use-efficient, competitive and sustainable. While R&D efforts over the past decade have contributed to increasing farmer acceptance of zero tillage for wheat in rice-wheat cropping systems, this has raised a number of institutional, technological, and policy related issues which must be addressed if CA practices are to be adopted in large scale in the region on a sustained basis. The following are some of the important policy considerations for promotion of CA.

(i) Scaling up conservation agriculture practices: Efforts to adapt the CA principles and technological aspects to suit various agro-ecological, socio-economic and farming systems in the region started a few decades ago. Greater support from stakeholders including policy and decision makers at the local, national and regional levels will facilitate expansion of CA and help farmers to reap more benefits from the technology. In India much research work on CA has been conducted for more than a decade, mostly at the *Indian Agricultural Research Institute.* However, its percolation to farmers is very limited. There is a need to think about the problems faced at the implementing level and devise a strategy involving all who are concerned. Most cases, where changes in favour of CA have occurred, are limited in success. FAO (2001) have reported that this is partly because policy environments are not favorable. One of the reasons for poor percolation of the technology to the farmers was the past bias or mindset about tillage by the majority of farmers (Hobbs and Govaerts, 2010). Under such situations, farmers participatory on-farm research to evaluate/refine the technology in initial years followed by large scale demonstration in subsequent years is needed. In India, efforts are being initiated through a network research project for on-farm evaluation and demonstration of CA technology for its promotion.

(ii) Shift in focus from food security to livelihood security: Myopic “food security” policy based on cereal production must now replace a well-articulated policy goal for livelihood security. This will help the diversification of dominant rice-wheat cropping systems (occupying about 10.5 million ha) in the Indo-Gangetic Plains, the cultivation of which in conventional tillage practice has overexploited thenatural resources in the region. The nature of cropping patterns and the extent of crop diversification are influenced by policy interventions. The government policies that directly or indirectly affect crop diversification are: pricing policy, tax and tariff policies, trade policies and policies on public expenditure and agrarian reforms (Behera et al., 2007).

(iii) CA offers opportunities for diversified cropping systems in different agro-ecoregions. Developing, improving, standardizing equipment for seeding, fertilizer placement and harvesting ensuring minimum soil disturbance in residue management for different edaphic conditions will be key to success of CA. For many situations for example, in hilly tracts, for small land holder's bullock drawn equipment will have greater relevance. Ensuring quality and availability of equipment through appropriate incentives will be important. In these situations, the subsidy support from national or local government to firms for developing low cost machines will help in the promotion of CA technologies.

(iv) CA technologies bring about significant changes in the plant growing micro-environment. These include changes in moisture regimes, root environment, emergence of new pathogens and shift in insect-pest scenario etc. The requirement of plant types suited to the new environment, and to meet specific mechanization needs could be different. There is a need to develop complementary crop improvement programmes, aimed at developing cultivars which are better suitable to new systems. Farmers-participatory research would appear promising for identifying and developing crop varieties suiting to a particular environment or locations.

(v) There is a need for generating a good resource database with agencies involved complementing each others' work. Besides resources, systematic monitoring of the socio-economic, environmental and institutional changes should become an integral part of the major projects on CA.

(vi) Policy support for capacity building by organizing training on CA is needed. Availability of trained human resources at ground level is one of the major limiting factors in adoption of CA. Training on CA should be supported at all levels. Efforts to adequately train all new and existing agricultural extension personnel on CA should be made in relevant departments. Consideration of extension approaches such as the 'Lead Farmer Approach' should also be made as a way to mitigate extension shortages at the local level. In the long term, CA should be included in curricula from primary school to university levels, including agricultural colleges. Inclusion of conservation and sustainability concepts in the course curricula with a suitable blend of biophysical and social sciences would be important for sustainable resource management.

(vii) Institutionalize CA: CA has to be mainstreamed in relevant ministries, departments or institutions and supported by adequate provision of material, human and financial resources to ensure that farmers receive effective and timely support from well trained and motivated extension staff. Key local, regional and national institutions should have dedicated CA champions among their staff who will help to ensure that relevant plans, programmes and policies embrace CA. In the short to medium term, policy makers could support activities of national and regional CA working groups to ensure that relevant thematic (research, technical, extension, training, education, input and output markets, policy) areas are covered by various CA programmes. Institutionalizing CA into relevant government ministries and departments and regional institutions is required for sustainability of the technology. Local, national and regional policy and decision makers could spearhead and support the formulation and development of strategies and mechanisms for scaling up the technology. CA could be integrated into interventions such as seed, fertilizer and tillage and draft power support programmes as a way of further enhancing productivity.

(viii) Support for the adaptation and validation of CA technologies in local environments: Adaptive research is required to tailor CA principles and practices to local conditions. This should be done in collaboration with local communities and other stakeholders. Issues that should be addressed include crop species, selection and management of crop and cover crop and rotations, maintenance of soil cover and CA equipment. The resource poor and small holder farmers in India do not have economic access to new seeds, herbicides and seeding machineries etc. (Sharma et al., 2012). This calls for policy frame work to make easily available critical inputs.

(ix) Support the development of CA equipment and ensure its availability: While some countries produce CA equipment, most of the available implements and equipment are imported. In the short term, consideration could be made on removing or reducing tariffs on imported CA equipment and implements to encourage and promote their availability. In the medium to long run, local manufacture of these will increase availability, ensure that equipment is adapted to local conditions, increase employment opportunities and reduce costs. The larger and more complex equipment is expensive and users may have to hire it. There is an opportunity to develop a local hire service industry by providing equipment, and training on machine maintenance and business skills. Where governments support land preparation schemes using ploughs, there is scope to change the equipment to rippers or direct seeders to reduce the cost and align the schemes to CA approaches. In India, significant efforts have been made in developing, refining and promoting the second generation zero-till multi-crop planters, but quality control assurance on standards and their availability at the local level with after-sale services and spare parts is still an issue. The new machineries, viz. happy seeder, turbo seeder, laser land leveler etc. Are found useful for CA practices, but these machines are more suitable for rich and medium to large farmers groups. These machines need more horse power (>50) for smooth functioning in field conditions. Small and marginal farmers having small holdings and economic limitations are unable to afford for such heavy machines. They need smaller versions of these machines which needs policy support for manufacturing at the local level.

(x) Promote payments for environmental services (PES) and fines for faulty practices: Adopters of CA improve the environment through carbon sequestration, prevention of soil erosion or the encouragement of groundwater recharge. It provides ecosystem services, thus, farmers could be rewarded for such services, which have a great impact on the quality of life for all. Continuous rice-wheat (RW) cropping in an area of 13.5 million ha with intensive tillage has resulted in over exploitation of resources, a decline of productivity, loss of soil fertility and biodiversity, and a decline of resource use efficiency in the Indo-Gangetic plains of South Asia. This has led to un-sustainability of agriculture in the region. Additionally, burning of huge quantities of crop residues has adverse environmental impacts. In a prosperous state like Punjab in India, 81% of the rice straw (crop residue) is burnt, leading to the loss of a huge quantity of nutrients and pollution of the environment (Yadvinder-Singh et al., 2005). Incorporation of crop residues is being considered as an alternative to burning and alters the soil environment, which in turn influences the microbial population and activity in the soil and subsequent nutrient transformations (Yadvinder-Singh et al., 2005). There is a need for a strong policy intervention for prohibiting such an unscientific practice by imposing a fine.

(xi) Building partnership: CA systems are very complex and their efficient management needs understanding of basic processes and component interactions which determine the system performance. A system perspective is the best to build working in partnership with farmers, who are at the core of farming systems and best understand this system. Scientists, farmers, extension agents, policy makers and other stakeholders in the private sector working in partnership mode will be important in developing and promoting new technologies. As FAO (2005) reported, the challenge is for would-be advisers to develop a sense of partnership with farmers, participating with them in defining and solving problems rather than only expecting them to participate in implementing projects prepared from outside. Instead of using a top-down approach where the extension agent places CA demonstrations in farmer fields and expects the farmer to adopt, a more participatory system is required where the farmers are enabled through provision of equipment and training to experiment with the technology and find out for themselves whether it works and what fine-tuning is needed to make it successful on their land.

(xii) Credit and subsidy: The other important thing for successful adoption of CA is the need to provide credit to farmers to buy the equipment, machinery, and inputs through banks and credit agencies at reasonable interest rates. At the same time government need to provide a subsidy for the purchase of such equipment by farmers. For example, the Chinese government in recent years adopted a series of policy and economic measures to push CA techniques in the Yellow River Basin and is providing a subsidy on CA machinery and imparting effective training to farmers (Yan et al., 2009). This resulted in a considerable increase in area under CA. Currently in Shanxi, Shandong and Henan provinces over 80% area under maize cultivation depends on no till seeder.

9. CONCLUSION

Conservation agriculture offers a new paradigm shift for agricultural research and development different from the conventional one, which mainly aimed at achieving specific food grains production targets in India. A shift in paradigm has become a necessity in view of widespread problems of resource degradation, which accompanied the past strategies to enhance production with little concern for resource integrity. Integrating concerns of productivity, resource conservation and soil quality and the environment is now fundamental to sustained productivity growth. Developing and promoting CA systems will be highly demanding in terms of the knowledge base. This will call for greatly enhanced capacity of scientists to address problems from a systems perspective; be able to work in close partnerships with farmers and other stakeholders and strengthened knowledge and information-sharing mechanisms. Conservation agriculture offers an opportunity for arresting and reversing the downward spiral of resource degradation, decreasing cultivation costs and making agriculture more resource – use-efficient, competitive and sustainable. “Conserving resources – enhancing productivity” has to be the new mission.

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2.16.O

ASSESSING THE EFFECT OF RAINWATER HARVESTING AND CONVERSATION ON SELECTED SOIL PHYSICO-CHEMICAL PROPERTIES AND CROP YIELDS IN COMPARISON WITH TRADITIONAL FARMING PRACTICES

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INTRODUCTION

Poverty is the main challenge that South Africa's population is facing. About half of the population in South Africa is classified as living in poverty while 25 % is categorised as extremely poor (Botha *et al*, 2015). This situation is even worse in rural areas. To address these food security challenges, the agriculture sector plays a vital role in providing food and income for the majority of the population (Lema and Majule, 2009). However, most communal farmers in South Africa rely on rainfed agriculture. Moreover, the country is experiencing the reduction in rainfall events as well as low soil fertility. These are major limiting factors to food production, since South Africa is dominated by semi-arid climate. It is therefore important for rural resource poor farmers to optimal utilise their limited water reserves. Rainwater harvesting (RWH) technologies are amongst other possible alternatives to maximise agricultural production. Therefore, the aim of this research was to assess the impact of contour ridge as a RWH technique on plant yields and soil productivity in comparison with the traditional farmer practice.

AIM

- Assess the effect of micro-catchment rainwater harvesting (contour bunds) with organic mulch on top of the bunds on selected soil physico-chemical properties and crop yields.

MATERIAL AND METHODS

- The study was conducted at a homestead in Quvile and Madosini in the Eastern Cape Province close to Tsolo during the 2013/14 and 2014/15 growing seasons, respectively.
- Treatments used were:
  - ✓ Contour bunds with organic mulch on top of the bunds with a bare runoff area (RWH&C). Rows of maize were planted on both sides of the contour bund.
  - ✓ Traditional farming practice where farmers were allowed to use their preferred farming practice where in both sites they have considered broadcasting of maize seeds (Control).
- Soil samples were collected at 0 - 10, 10 - 20 and 20 - 30 cm depths and subjected to chemical analysis (exchangeable bases, micro-nutrients, pH) and physical analysis (bulk density and aggregates stability) expressed as mean weight diameter (MWD).
- Gravimetric soil moisture content (GMC) was measured at different stages of maize growth (planting, tasselling and harvesting).
- Above-ground biomass and grain yield were measured.

RESULTS AND DISCUSSION

Aggregate Stability And Bulk Density

Table 1 below represents the aggregate stability and bulk density at the study sites in Quvile and Madosini. RWH&C improved aggregate stability compared to the control treatment. However, it decreases with increasing soil depth in both treatments.The control had higher bulk density compared to the RWH&C treatment. RWH&C therefore directly improves soil infiltration status and decreases soil compaction.

Table 1 Aggregate stability and bulk density at the study sites in Quvile and Madosini

Growing season	**Site**	**Treatment**	**Soil depth** (cm)	**Aggregate stability** MWD (mm)	**Bulk density** (g cm^{-3})
2013/14	Quvile	Control	0 - 10	3.7	1.20
			10 - 20	3.7	*
			20 - 30	3.6	*
		RWH&C	0 - 10	4.4	1.11
			10 - 20	4.4	*
			20 - 30	4.2	*
LSD p<0.05 Treatment*site*depth				**0.49**	**0.42**
2014/15	Madosini	Control	0 - 10	1.7	1.28
			10 - 20	1.6	*
			20 - 30	1.6	*
		RWH&C	0 - 10	2.7	1.16
			10 - 20	2.6	*
			20 - 30	2.5	*
LSD p<0.05 Treatment*site*depth				**0.46**	**0.32**

** Bulk density was only analysed at 0 - 10 cm soil depth*

Gravimetric Moisture Content

In figure 1 and 2 below, gravimetric moisture content at Quvile and Madosini is presented. Gravimetric moisture contents were generally higher under RWH&C than the control.RWH&C improves soil moisture content, which directly improves plant nutrient uptake. Similar trends were observed both sites.

❖ **Quvile site**

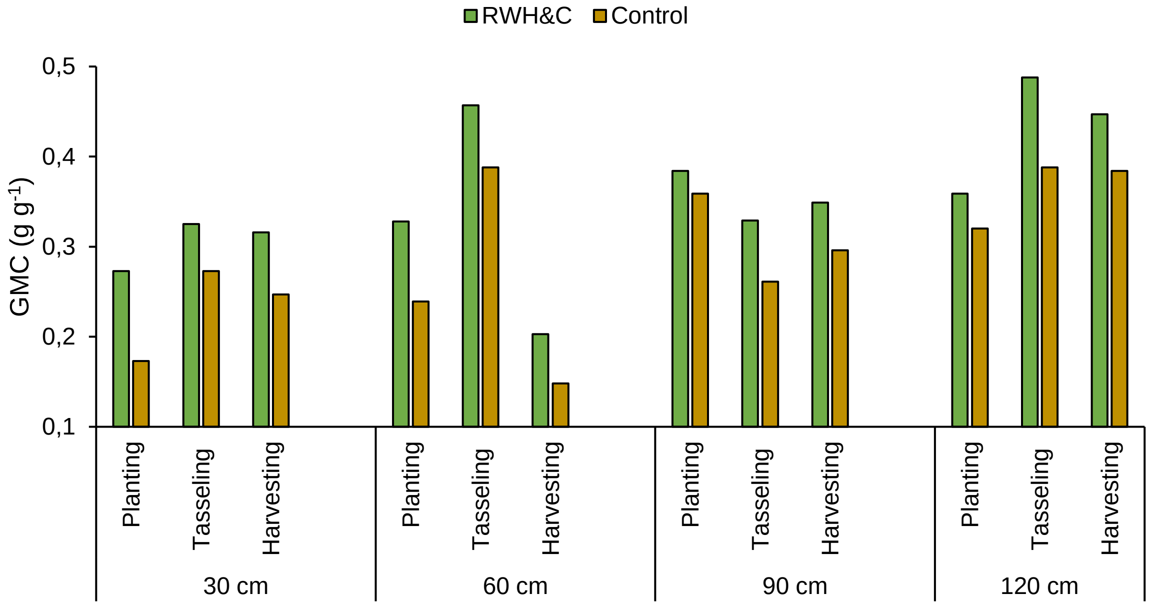

Figure 1 **Gravimetric moisture contents at various soil depths at three growth stages in the RWH&C and the control treatments at Quvile during the 2013/14 growing season.**

❖ **Madosini site**

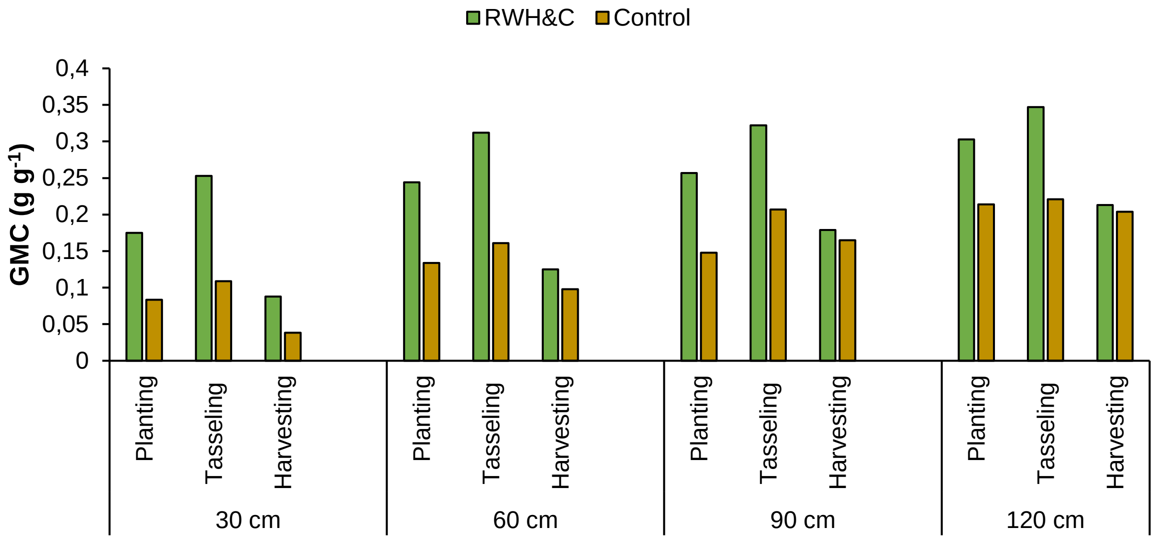

Figure 2 Gravimetric moisture contents at various soil depths at three growth stages in the RWH&C and the control treatments at Madosini during the 2014/15 growing season.

Soil Chemical Properties

Table 2 below represents soil chemical properties at 0-30 cm soil layer at homestead Quvile and Madosini. Rainwater harvesting was found to control the soil pH towards neutral at both homesteads. However, there was no significant difference in soil pH between the control and

RWH&C (p>0.05). Micro nutrients and exchangeable bases were not influenced by the treatments over the short term (p>0.05)

Table 2. **Average soil pH, micronutrients contents and exchangeable bases for the 0-30 cm soil layer at the homestead in Quvile (2013/14 season) and Madosini (2014/15 season)**

Homestead	Treatment	pH (H_2O)	Micro-nutrients(mg/L)				Exchangeable bases (mg/L)			
			Cu	Zn	Mn	Fe	Ca	Mg	Na	K
Quvile	Control	8.37	9.60	32.87	2.67	1.16	120	824	33	30
	RWH&C	7.78	5.91	39.64	0.75	0.38	130	750	26	107
LSD (p<0.05)		4.51	1.05	20.74	3.61	20.68	315.3	895.1	8.1	153.9
Madosini	Control	7.53	18.88	28.30	6.60	0.85	263	1130	11	241
	RWH&C	7.49	19.45	28.02	6.02	1.40	549	1389	24	253
LSD p<0.05 Treatment*site*growing season		0.39	31.40	0.774	4.343	0.774	193	895.1	9.227	214.9

The figure below (figure 4) shows the grain and biomas yield at Quvile and Madosini in 2013/14 and 2014/15 growing seasons. RWH&C was found to improve both grain yield and biomass yield at both sites.

Grain And Biomass Yield

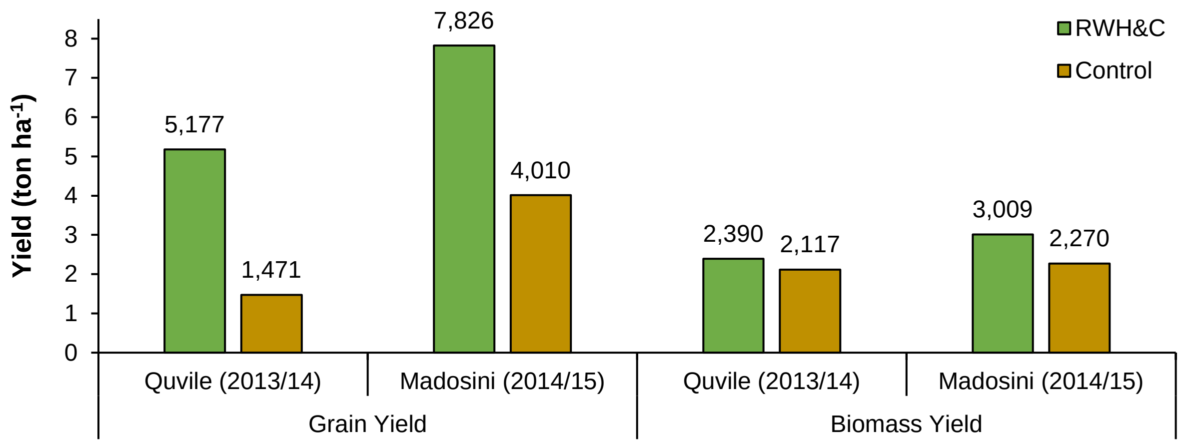

Figure 4 Grain yield and above ground biomass of RWH&C and the control at the sites in Quvile and Madosini during the 2013/14 and 2014/15 growing seasons, respectively.

CONCLUSIONS AND RECOMMENDATION

- RWH&C improves some soil physico-chemical properties.
- RWH&C has a positive effect on grain and biomass yields.
- It is recommended that farmers who rely on rainfed agriculture should consider implementation of RWH&C as it improves soil health while improving crop yields.

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2.17.O

PEANUT POD YIELD AND SOIL COMPACTION IN CONSERVATION AGRICULTURE SYSTEMS

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1. INTRODUCTION

In Brazil the cultivated area with peanut is approximately 120.000 hectares and approximately 80% are concentrated in Sao Paulo State, mainly as crop rotation to renew sugarcane crop. Recently, there has been increasing the adoption of green harvested sugarcane system, which represents almost 75% of 9,2 million hectares. After harvesting a great amount of straw (average 15 Mg of dry matter per hectare) is deposited over soil surface, thus the adoption of conservation agriculture principles (legume breaks, minimum tillage, trash retention and when it is possible to use controlled traffic) are desirable in order to reduce the costs, to diminish the soil erosion and to maintain the good stalk yields (Bolonhezi et al., 2007). Peanut farmers are resistant to adopt the no-till under sugarcane straw, although there are a lot of scientific results (that show many benefits in terms of yield, weed control, pest control, soil water availability and profitability). On the other hand, there are questions to answer in a commercial scale, such as digging and soil compaction.

2. MATERIAL AND METHODS

In this context, two trials were carried out in a commercial fields located at Pitangueiras city (Red Latosol with high clay content) and Planalto city (Red-Yellow Latosol with high sand content), both in Sao Paulo State, Brazil. According to randomized block experimental design, three treatments were installed; conventional tillage (moldboard plowing followed by two applications of disk harrowing), reduced tillage (Rip Strip® after spraying the area with 3.6 kg a.i. ha^{-1} of glyphosate), and no-tillage (crop residues on the soil surface after spraying of glyphosate). In Planalto city, peanut variety IAC-503 was sown in 12th November 2015 using a vacuum planter calibrated to establish 166.000 plants ha^{-1} in paired rows 90 cm apart with a large coulter followed by a double disc opener. In Pitangueiras city, the peanut genotype IAC-OL3 was planted in 28th November 2016 a vacuum planter with the same characteristics. Agronomic characteristics were evaluated from 30 days after planting to the harvest at the beginning of April. The soil cone index was evaluated before installing the tillage and after harvesting of peanut using a digital penetrometer model PNT-2000® (DLG Company). Samples of pods were harvested in a grid of seven rows by seven columns, with distal points of its neighbourhood at intervals of 20 meters, in order to quantify the spatial variability of yield for each tillage system. After digging and mechanical harvesting, the same procedure was done to evaluate the pod loss in 2 m^2 of soil surface.

3. RESULTS

For clay soil conditions (Table 1), It was verified that the pod yield in no-tillage was in average 4000 and 2500 kg ha^{-1} lower than conventional and minimum tillage, respectively. On the other hand, for sand soil, no significant difference between conventional and reduced tillage was observed for pod yield (Figure 1). It's interesting to emphasize that the pod yield determined by samples has showed higher production in no-tillage for sand soil, but was lower for total area harvested. It's happened

Table 1. Agronomic characteristics of peanut cv. IAC-OL3, in different soil management in Pitangueiras city, Sao Paulo State, 2016. Different letters compare means (Tukey 5%) between soil management treatments.

Soil Management (M)	# plants initial	# plants final	Pod Yield	Kernel/Pod Porcentage	Impurity	Aflatoxin	Pod Loss
	plants m^{-1}		kg ha^{-1}	%	%	ppb	kg ha^{-1}
Conventional	16.3 a	16.4 a	6046 a	76.1 a	11.3 b	0.96 a	761 a
Rip Strip	16.3 a	14.2 a	4514 b	75.5 a	14.2 b	0.20 a	775 a
No-tillage	11.2 b	11.0 b	2054 c	70.2 b	20.2 a	1.78 b	543 a
F-probability	9.6 **	13.2 **	52.7 **	12.3 **	8.9 **	1.41 ns	1.14 ns
L.S.D. (Tukey 5%)	3.6	2.8	1046	3.5	5.7	2.52	439
CV%	17,2	13,2	17,5	3,3	26,3	180,0	55,6

because in no-tillage under sugarcane straw the number of gaps (part of the rows without plants) is almost 10 and 100 times higher than conventional and reduced tillage, respectively.

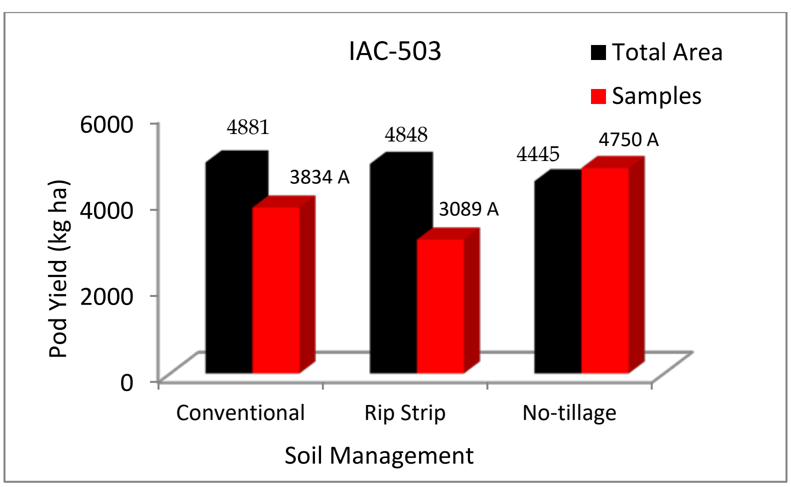


Figure 1. Pod yield (kg ha^{-1}) of peanut cv. IAC-503 planted in different soil managements in sand soil located in Planalto city, Sao Paulo State, Brazil. 2016

According to Siri-Prieto et al. (2009), in general, peanut response to conservation tillage has been variable in the last 20 yr due to issues such as weed control, diseases, hardpan, pod digging, and weather conditions. Maybe for the present research, the most important factor was the reduction in the plant population, due to the great amount of straw the seedling needs more time to emergence. The other important factor is the soil compaction, specially in rotation with sugarcane, in which the traffic during the harvest is intensive.

When it comes to soil compaction, in the Figure 1 and 2 are showed the soil index cone for both soils in two different measurements. In sand soil, when the measurement was done with good soil water content, the value of soil index cone is high in the no-tillage, but the opposite result it was verified during a dry period (Figure 1). For clay soil, despite of the higher soil moisture, the soil strength is higher in the conservation tillage in comparison with conventional (Figure 2). The literature review showed that the majority of crops have problem with vegetative and root growth when soil strength is higher than 2,5 MPa or a soil bulk density is higher than 1,4 g dm^{-3} (Canarache, 1990). The use of Rip Strip for reduced tillage, it seem to be an acceptable option to reduce the soil compaction in the peanut row and to keep the straw in a good level to protect against soil erosion .

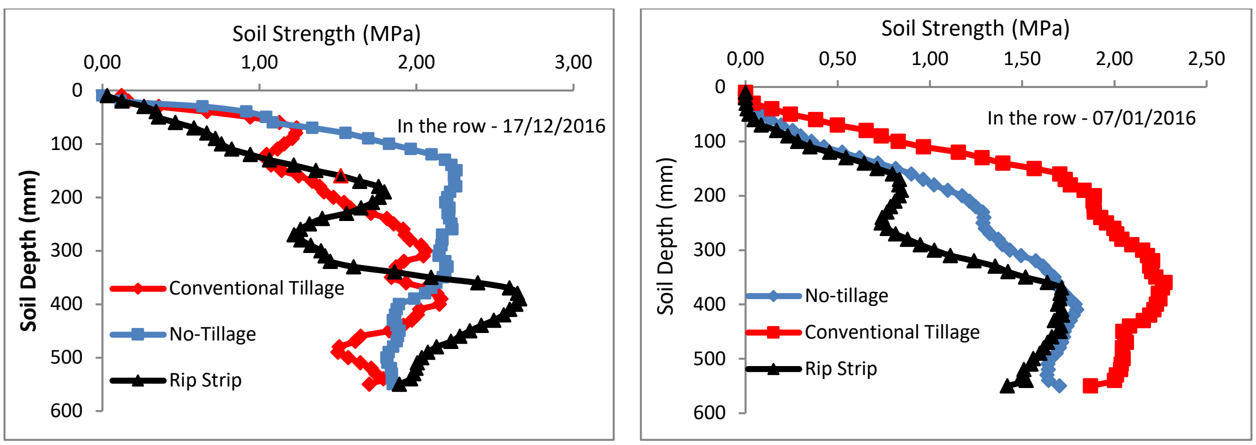


Figure 2. Soil strength (MPa) measured in a sand soil at the beginning and in the middle of cycle of peanut crop, cultivated in Planalto city, Sao Paulo State, Brazil. Means come from 30 replications.

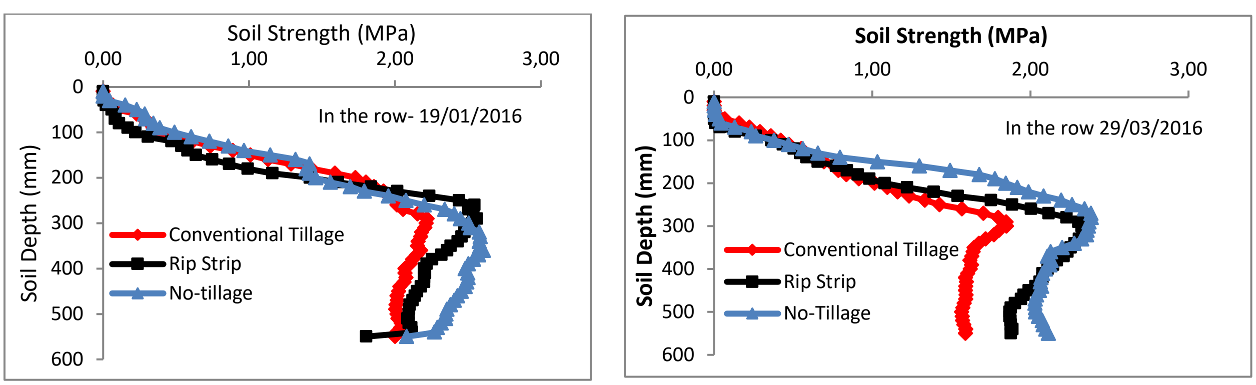
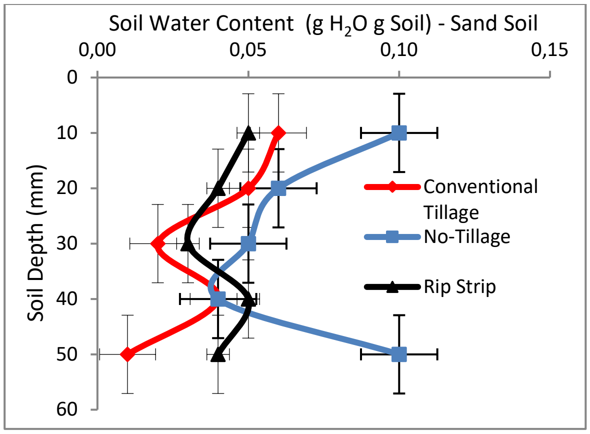


Figure 3. Soil strength (MPa) measured in a clay soil at the beginning and next to digging of peanut crop, cultivated in Pitangueiras city, Sao Paulo State, Brazil. Means come from 60 replications.



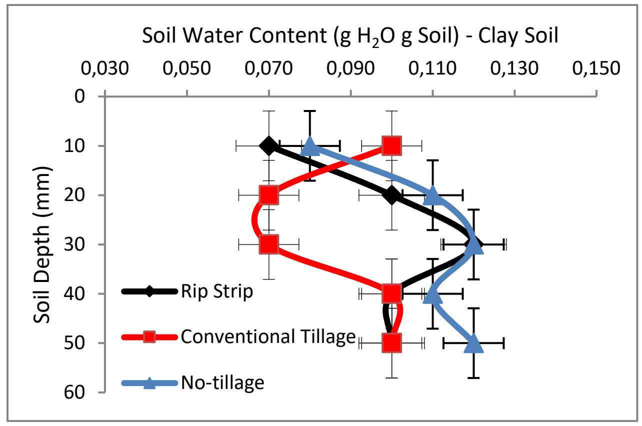


Figure 4. Soil water content(g H_2O g soil) measured in a sand soil in the middle of cycle of peanut crop (left) and a clay soil (right) in different soil managements. Means come from 10 replications.

4. CONCLUSIONS

The pod yield was three times and 26% lower in the no-tillage and Rip Strip, respectively, for clay soil, but no difference was observed in the Rip Strip in comparison with conventional for sand soil. On the other hand, the pod loss after digging was 7 % lower in conservation agriculture. The plant failure (gaps) in the rows was 0,18 %; 2,52% and 21,29 % in the conventional, Rip Strip and No-tillage, respectively.

The highest soil strength for both soils was verified from 0,10 to 0,30 m depth. During dry period, the value of soil strength achieved 2,5 MPa, mainly for sand soil in the conventional tillage. Rip Strip it seem to be a good strategy to diminish the soil compaction in the row of peanut crop and can reduce the variability of seedling emergence, consequently a good pattern of plant population.

5. ACKNOWLEDMENTS

The authors wish to thank the AGRISUS Foundation for supporting this research project (grant # PA 1694/15). Also we appreciated the CNPQ (National Council for Scientific and Technological Development) for scholarships and the farmers of Valochi and Ulian families for all dedication to carrie out this research.

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2.18.O

CONSERVATION AGRICULTURE PRINCIPLES APPLIED FOR SUGARCANE CROP PROPAGATED BY PRE-SPROUTED BUDS SYSTEM IN BRAZIL

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1. INTRODUCTION

In Brazil the sugarcane crop is cultivated in almost 10 million hectares, which 60% of plantations are concentrated in Sao Paulo state. The government of Sao Paulo State has mandated (Act # 47,700 of March 1st, 2003, regulates the law # 11,241 of September 19th, 2002) that in flat (less than 12% of slope) sugarcane production areas feasible for mechanical harvest, the use of fire to reduce biomass before harvesting must be banished completely up to 2017. Due to this social pressure, nowadays the majority of fields are harvested without burning (around 80 %), then a great amount of straw (average 15 Mg of dry matter per hectare each year) is deposited over the soil surface.

The sugarcane straw provide significant changes on the crop system, such as; to control the soil erosion (Prove et al., 1995), to control of weeds (Cristoffoleti et al., 2007), to reduce the greenhouse gases emissions from the soil (La Scala et al., 2006), to increase the soil organic matter and to increase the soil moisture (Bolonhezi et al., 2010). On the other hand, the great amount of residues increase the cost with tillage in almost 30%, consequently is desirable the adoption of conservation soil management to cultivate leguminous used as crop rotation and to grow sugarcane. But there are few scientific results about no-tillage or conservation agriculture principles applied for sugarcane crop, specially when the crop is installed by pre-sprouted bud, popularly known as bud chips.

In Brazil, these systems are known since 1980 decade, but recently the use of pre-sprouting system for propagation of sugarcane is increasing in Brazil. According to Abd El Mawla et al. (2014), these systems reduce the amount of seed cane from 10 to 2 Mg ha^{-1}, save 2000 m^3 of water for irrigation, save money and increase the tillering. Considering that “seed cane” used for replanting accounts for over 20 percent of the total cost of production, the objectives of this research were; to study the vegetative and root growth of two varieties of sugarcane in different soil managements after green manure and to quantify the changes on agronomic and technological characteristics, as well as to understand the changes on the soil carbon stock.

2. MATERIAL AND METHODS

This long-term experiment was started in 2003 in an eutrophic Clayed Rhodic Hapludox (Oxisol) located at Experimental Station of APTA/IAC, Ribeirao Preto city, Brazil. It was adopted a randomized complete block design in a split-plot scheme, with four replications. The main plots are consisted by three soil managements; conventional tillage (moldboard plowing, 30 cm depth followed by two applications of offset disk harrow), reduced tillage (use of Rip Strip® before planting) and no-tillage (crop residues left on its surface after spraying the area with 3.6 kg ha^{-1} a.i. of glyphosate).

After twelve years with cultivation of grain crops, two sugarcane varieties (IACS95-5000 and an energy cane, used for biomass) were planted as a subplot, in rotation with green manure (*Mucuna aterrima* L.). Samples of plant, roots and soil were collected during fifteen months (from planting to harvesting) in order to evaluated vegetative and roots growth, agronomic and technological characteristics, as well as some physics (soil strength) and chemical attributes (soil carbon stock). Some of these preliminary results are presented in this paper.

3. RESULTS

There were verified significant differences between soil management for the plant cane harvest (year 2016) in terms of agronomic characteristics (Table 1). The stalk yield was **10,8 e 18,4 Mg ha^{-1}** higher in no-tillage and reduced tillage, in comparison with conventional tillage. Maybe the higher soil water content in conservation agriculture had helped to increase the numbers of tiller and consequently it was verified gains in the productivity. No differences were observed between the sugarcane varieties. Bolonhezi et al. (2014) verified gains of 10 Mg ha^{-1} of stalk yield in no-tillage, but for conventional method of propagation.

When it comes to soil compaction, it’s verified in the Figure 1, the index cone measured in February

Table 1. Stalk yield and agronomics characteristics for plant cane (first harvest) of two sugarcane varieties (IACSP 95-5000 and Energy Cane) in different soil managements (**CT**-conventional tillage; **RS** – Rip Strip e **NT**-no-tillage). Ribeirao Preto, Sao Paulo State, Brazil, 2016. Means come from 4 replications.

Soil Management	Stalk Yield estimated Mg ha^{-1}	Stalk Yield measured Mg ha^{-1}	# of Stalk estimated Un. ha^{-1}	# of Stalk measured Un. ha^{-1}	Stalk Biomass kg Un.$^{-1}$
Conventional	133,70	93,53 b	139722	157500	1,03
Reduced Tillage	145,16	111,89 a	158945	198333	0,98
No-tillage	119,24	104,33 ab	139139	190833	0,97
F probability	2,09 ns	5,82*	2,21 ns	0,75 ns	1,04 ns
L.S.D.(Tukey 5%)	39,01	16,60	32893	80632	0,13
Varieties (V)					
IACSP 95-5000	132,50	103,98	102870 b	122778 b	1,29 a
Enery Cane	132,90	102,51	189000 a	241667 a	0,70 b
F probability	0,00 ns	0,14 ns	141,48*	78,51**	30,47**
L.S.D.(Tukey5%)	32,57	8,92	16380	30354	0,24
Interaction M x V	0,17 ns	0,65 ns	0,23 ns	0,64 ns	0,33 ns
CV(%) tillage	19,2	10,5	14,7	28,8	8,8
CV(%) varieties	26,6	9,4	12,2	18,0	26,6

2016 (soil with good level of water). The highest value of soil strength was observed in the conventional tillage at depth of 0,14 m, followed by no-tillage (2,92 MPa at 0,15 m) and reduced tillage (2,28 MPa at 0,30 m). The same trend was verified during the winter season (low soil water content). This result is important for new method of sugarcane propagation (pre-sprouted bud) because the sugarcane seedlings root system are very sensitive in terms of soil compaction. Furthermore, its import to mention that in average 60% of sugarcane biomass are concentrated at depth of 0,30 m. Rip strip it seems to be a good equipment to reduce the soil compaction in the row of sugarcane and could be important to diminish the impact of traffic in mechanized fields.

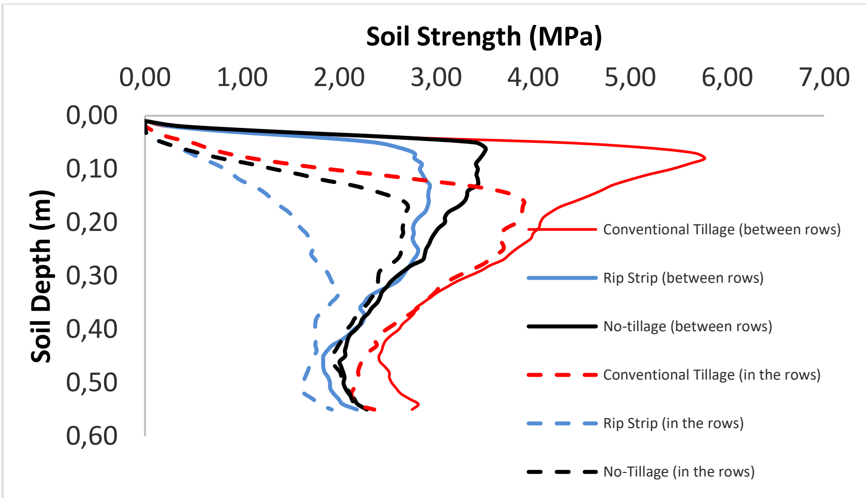

Figure 1. Soil strength (MPa) in different soil managements (**CT**-conventional tillage; **RS** – Rip Strip e **NT**-no-tillage) and sugarcane varieties (IACSP 95-5000 and Energy Cane). Ribeirao Preto, Sao Paulo State, Brazil, February, 2016. Means come from 40 replications.

The interesting result was the gains in terms of soil carbon stock (Figure 2) for conservation agriculture systems. After twelve years of conventional tillage, there was a reduction of 16,4 Mg ha^{-1} on the soil carbon stock, in comparison with no-tillage and reduced tillage. Segnini et al. (2013) in long-term research carried out at the same place, found soil carbon stock of **CS=120±5 Mg ha^{-1}** for the conventional tillage and **CS=127±4 Mg ha^{-1}** for no-tillage after seven years with harvest without burning associated with conservation agriculture. These results represent gains of **1,0 Mg C ha^{-1} year^{-1}** for conservation agriculture. Regarding to this present research include the sugarcane crop two years ago, it could be inferred that the great amount of biomass will help to increase the soil carbon sequestration. The literature review have indicated that the tilling soil increases soil organic matter decomposition and reduces soil C stocks, because the disruption of soil aggregates and exposure to microbial activity. It's important to emphasize that for sugarcane crop system, according to Carvalho et al. (2016) that the aboveground sugarcane straw is the main source of C to the soil. Then, the indiscriminate removal of crop residue to produce cellulosic biofuels can reduce soil carbon stocks. However, the contribution of belowground biomass (roots) is not yet completely understood for sugarcane, mainly for varieties known as energy cane (*Saccharum spontaneum* L.).

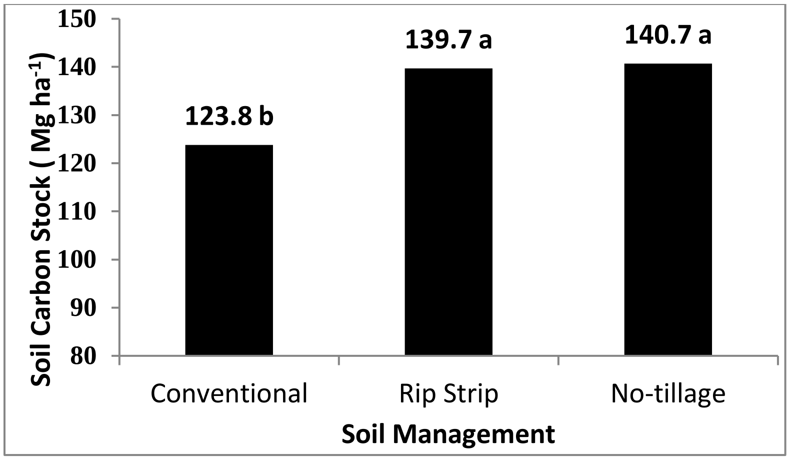

Figure 2. Soil carbon stock (Mg ha^{-1}) in different soil managements (**CT**-conventional tillage; **RS** – Rip Strip e **NT**-no-tillage) and sugarcane varieties (IACSP 95-5000 and Energy Cane). Ribeirao Preto, Sao Paulo State, Brazil, February, 2016. Means come from 12 replications.

4. CONCLUSIONS

It could be conclude for both sugarcane varieties planted by pre-sprouted bud that the adoption of conservation agriculture principles, increase the stalk yield up to 18, 4 Mg ha^{-1} and the soil carbon stock up to 16,4 Mg ha^{-1}. On the other hand, the conventional tillage showed the highest soil strength at 0,14 m depth.

5. ACKNOWLEDMENTS

The authors wish to thank the AGRISUS Foundation for supporting this research project (grant # PA 1494/15). Also we appreciated the CNPQ (National Council for Scientific and Technological Development) for scholarship and the CAPES for supporting the graduate student.

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2.1.P

JET FUEL FEEDSTOCKS IMPROVE UTILIZATION OF PRECIPITATION IN SEMI-ARID ENVIRONMENTS

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INTRODUCTION

The traditional practice of summer fallow in alternating years in dryland spring wheat cropping systems leads to numerous environmental challenges including soil C depletion, reduced soil water holding capacity, and inefficient precipitation storage during periods of fallow. Renewable jet fuel feedstocks grown in place of fallow can potentially offset the demand for petroleum-based transportation resources, diversify cropping systems, and increase ecosystem services. However, identifying suitable jet fuel feedstocks that best utilize limited participation in semi-arid environments remains a primary constraint to widespread adoption.

METHODS

A 4-yr, multi-site experiment initiated in fall 2012 investigated the precipitation use efficiency (PUE) of six winter- and twelve spring-types of cool-season oilseed feedstocks. Sidney, MT site was one among eight sites in the western USA. Other sites that conducted the same study included Akron Colorado, Ames Iowa, Mandan North Dakota, Morris Minnesota, Moscow Idaho, Pendleton Oregon, and Temple Texas. Winter types of *Camelina sativa* (1), *Brassica napus* (4), and *B. rapa* (1) were planted in September, while spring types of *Camelina sativa* (1), *B. napus* (4), *B. rapa* (1), *B. juncea* (2), *B. carinata* (2), and *Sinapis alba* (2) were planted in April for the growing seasons of 2013-2016.

RESULTS

‘Joelle’ camelina showed excellent cold tolerance and was the only winter type that survived the cold winters (-32°C). Not a single plant from other winter types survived during the four years of this study. Adverse weather in the form of hail stones severely damaged oilseeds during the 2013 and 2015 cropping season, thus only data from 2014 and 2016 are presented. Monthly weather during the growing seasons of those two years is shown in Table 1.

Table 1. Average air temperature and precipitation near Froid, MT USA during 2014 and 2016.

Month	Average air temp, °C		Precipitation, mm	
	2014	**2016**	**2014**	**2016**
Feb	-12.0	-0.7	1.5	8.4
Mar	-3.3	2.9	9.4	11.9
Apr	5.0	6.8	31.8	82.6
May	12.6	13.8	89.4	37.3
Jun	16.0	19.1	32.3	33.8
Jul	20.4	20.7	12.7	55.9

Seed yield ranged from 0 to 2130 kg ha^{-1} and means for 2014 and 2016 are shown in Figure 1. Spring types *B. napus*, *Camelina sativa*, and *B. carinata* showed the greatest yield potential. *B. carinata* seed yields in 2016 were impacted by herbicide damage, thus reducing the average yield shown in Figure 1.

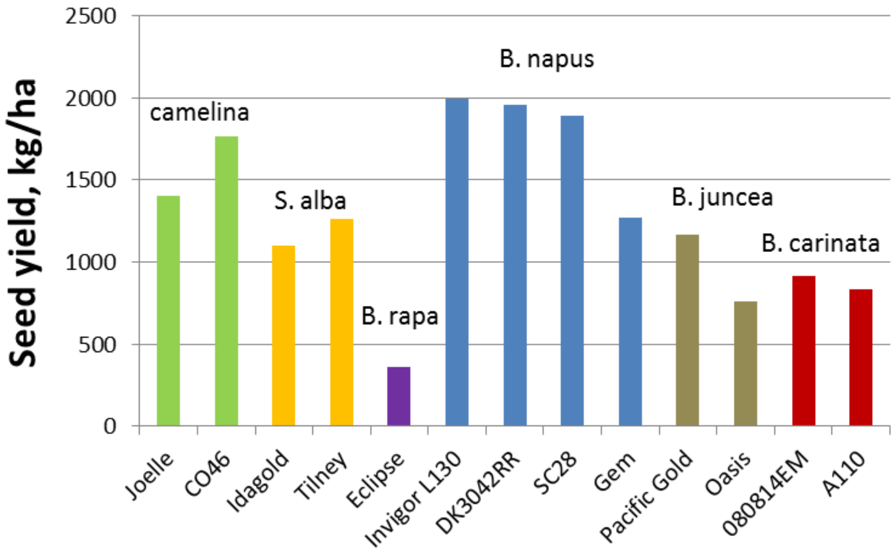

Figure 1. Mean seed yield from 2014 and 2016 oilseed feedstocks grown near Froid, MT USA.

Seed oil ranged from 0 to 878 kg ha^{-1} and means for 2014 and 2016 are shown in Figure 2. *B. napus*, *Camelina sativa*, and *B. carinata* showed the greatest oil yield potential. *B. carinata* in 2016 was impacted by herbicide damage that likely reduced its seed yield and oil %, and thus reduced its average oil yield shown in Figure 2.

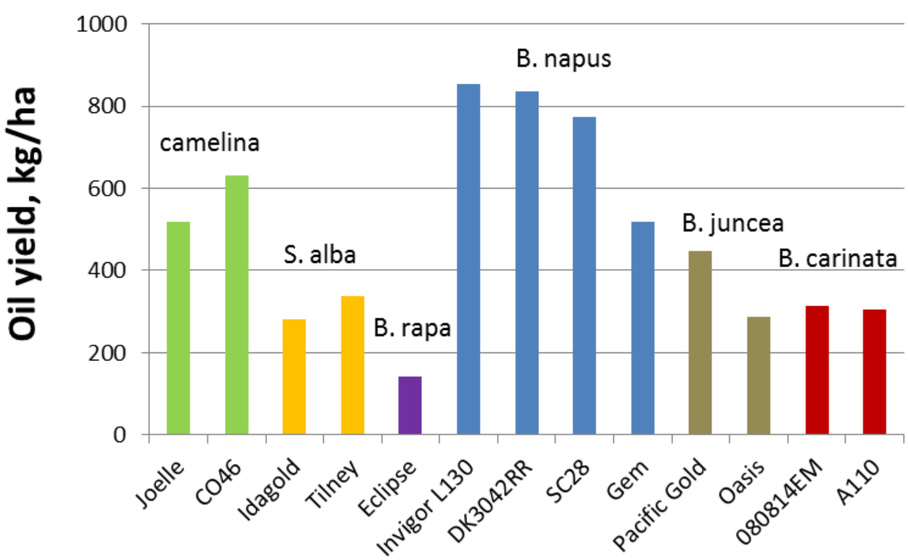

Figure 2. Mean seed oil yield from 2014 and 2016 oilseed feedstocks grown near Froid, MT USA. Oil yield = seed yield x seed oil concentration.

Growing season precipitation used to calculate precipitation use efficiency (PUE) ranged from 148 to 213 mm. The PUE for seed production ranged from 0 to 12.9 kg ha^{-1} mm^{-1} and means for 2014 and 2016 are shown in Figure 3. *B. napus*, *Camelina sativa*, and *B. carinata* showed the greatest PUE potential for seed yield. *B. carinata* in 2016 was impacted by herbicide damage that likely reduced its seed yield, and thus reduced its average precipitation use efficiency use for seed production shown in Figure 3.

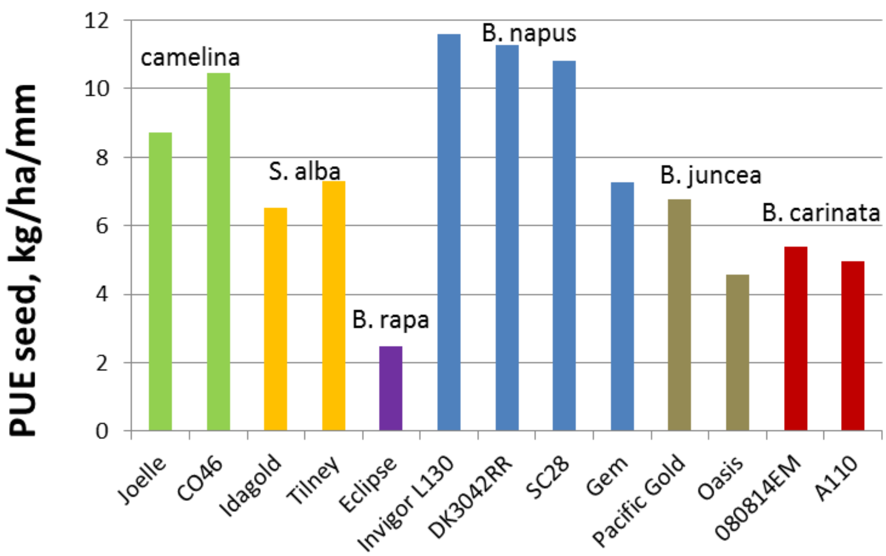

Figure 3. Mean precipitation use efficiency for seed production from 2014 and 2016 oilseed feedstocks grown near Froid, MT USA. PUE = seed yield÷growing season precip.

Precipitation use efficiency for oil ranged from 0 to 5.9 kg ha^{-1} mm^{-1} and means for 2014 and 2016 are shown in Figure 4. *B. napus*, *Camelina sativa*, and *B. carinata* showed the greatest potential for precipitation use efficiency for oil. *B. carinata* impacted by herbicide damage in 2016 likely reduced seed yield, reducing its average precipitation use efficiency use for oil production shown in Figure 4.

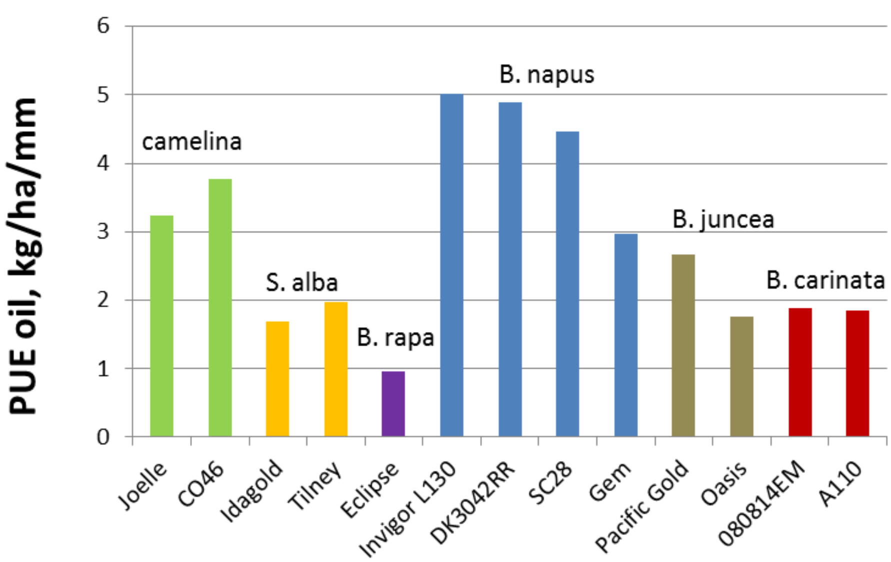

Figure 4. Mean precipitation use efficiency for oil from 2014 and 2016 oilseed feedstocks grown near Froid, MT USA. PUE = oil yield÷growing season precip.

Other measurements taken, but not reported included crop phenology, canopy spectral reflectance, leaf area, canopy temperature, water use, crop biomass, yield components, seed oil%, seed fatty acid composition, and drought resistance.

CONCLUSIONS

Overall, spring and winter camelina and spring *B. napus* and *B. carinata* were suitable alternatives to summer fallow that could improve utilization of precipitation and provide sustainable jet fuel feedstocks in the semi-arid northern Great Plains, USA.

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2.3.P

LEACHING OF DISSOLVED ORGANIC NITROGEN AND CARBON IN A COVER CROP-MAIZE ROTATION

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INTRODUCTION

Dissolved inorganic nitrogen (DIN) forms, such as NO_3-N and ammonium (NH_4-N), have been pointed out as the main sources of N leaching in agricultural systems, and have been focused of attention in the last decades. However, recent studies reported that dissolved organic N (DON) leaching can be another important N loss pathway from agroecosystems that promote eutrophication of water bodies (Abaas et al., 2014; Scott and Rothstein, 2014). In addition, DON is directly associated with dissolved organic carbon (DOC). On the one hand, the inclusion of cover crops during the intercropping period of maize has been proposed to counteract the negative impacts of N diffuse pollution from irrigated maize fields (Gabriel and Quemada, 2011). On the other hand, it is not well understood what is the impact of different crop rotations in the DON and DOC leaching. The main aim of this study was to investigate the leaching of DON and DOC in cover a crop-maize rotation.

STUDY SITES AND METHODS

This study was carried out in a temperature-controlled glasshouse with undisturbed soil columns packed in PVC tubes (0.2 m diameter, 0.5 m long) in August 2015. The columns were taken from the Antumapu Experimental Farm (Santiago, Chile) in a soil classified as Entic Haploxerolls, with loam to loamy sand textural class, pH_{water} = 8.6 and total organic C content of 0.65%. A combination of 13 treatments (Table 1) with four replications were evaluated considering combination of: N doses of 250 or 400 kg N ha^{-1} for maize (*Zea mays*) or 0 or 150 kg N ha^{-1} for cover crops, crop rotation (maize-fallow, maize-cover crop, cover crop or fallow) and cover crop species (*Lolium multriflorum* or *Trifolium repens*). For maize a dose of 250 kg N ha^{-1} corresponds to an optimum N dose, whereas 400 kg N ha^{-1} represent an excessive N dose usually applied by farmers in Central Chile. Treatments with fallow had bare soil during the autumn-winter period (April to September), while maize is cultivated during spring summer (October to March). The first season with maize was set from October 2015 to March 2016, but samples were not taken because the soil columns were in a setting stage. The maize was harvested in March 2016. From April 2016 to January 2017, water that percolated from the columns after irrigation events was measured and a subsample saved for analyses of DOC and total dissolved N (TDN) using a Shimadzu TOC-N analyser; and DON was calculated as the difference between TDN measurements and DIN concentrations using colorimetric methods. Dissolved organic N and DOC loads were calculated as the volume of percolated water times the DON and DOC concentrations and converted to kg DON ha^{-1} and kg DOC ha^{-1} , respectively, considering the column area.

RESULTS

During the autum-winter period, the highest DON loads were found in treatments with permanent *L. multiflorum* (Lm (0N) and Lm (150N)) with values ranging from 39 to 75 kg DON ha^{-1}; whereas the treatments with fallow and with the highest N fertilization (400 kg N ha^{-1}) showed the lowest DON loads with values ranging from 0 to 4 kg DON ha^{-1}. During spring-summer period there were not significant differences in DON loads among the treatments (Table 1).

Table 1. Mean dissolved organic nitrogen (DON) loads in the different treatments

Treatment[1]	Season[2]			
	Autumn-winter		Spring-summer	
	-------------------- kg DON ha^{-1} --------------------			
F	0.09 ± 0.09	e	4.70 ± 2.92	a
Lm (0 N)	39.43 ± 8.28	ab	5.44 ± 2.92	a
Lm (150 N)	75.17 ± 16.84	a	4.23 ± 3.06	a
Tr (0 N)	5.04 ± 3.05	de	6.49 ± 3.06	a
Tr (150 N)	4.13 ± 3.05	de	5.81 ± 2.92	a
Zm–F (250 N)	1.16 ± 1.58	e	3.51 ± 3.06	a
Zm–F (400 N)	3.55 ± 3.05	e	4.46 ± 2.92	a
Zm–Lm (250 N)	10.87 ± 1.37	cd	8.46 ± 3.06	a
Zm–Lm (400 N)	3.18 ± 3.05	e	5.21 ± 2.92	a
Zm–Tr (250 N)	6.21 ± 4.99	cde	7.38 ± 2.92	a
Zm–Tr (400 N)	2.79 ± 3.05	e	3.89 ± 2.92	a
Zm–Lm+Tr (250 N)	1.89 ± 1.37	e	8.25 ± 2.92	a
Zm–Lm+Tr (400 N)	29.92 ± 10.96	bc	12.00 ± 3.06	a

[1] F = fallow; Lm = *Lolium multiflorum*; Tr = *Trifolium repens*; Zm = *Zea mays*.

[2] Means ± standard errors (autumn-winter, n = 12; spring-summer, n = 8) with different letters within a column are significantly different (ANOVA, $p<0.05$).

In general, treatments with permanent *L. multiflorum* showed the highest DOC loads with values ranging from 7 to 15 kg DOC ha^{-1} and from 22 to 49 kg DOC ha^{-1} during spring-summer and autumn-winter periods, respectively; while the treatments with *T. repens* showed the lowest DOC loads during the study period with values ranging from 7 to 9 kg DOC ha^{-1} and from 29 to 37 kg DOC ha^{-1} during spring-summer and autumn-winter periods, respectively.

The inclusion of *L. multiflorum* could enhance soil organic C pools and microbial activity, and in consequence increased the amount of DON and DOC susceptible to leaching. In permanent *L. multiflorum* treatments was found that DON and DOC loads were higher during the autumn-winter period than during the spring-summer. This can be by the higher water percolation during the autumn-winter compared to the spring-summer period, with transpiration of *L. multiflorum* reducing percolation during the spring-summer period.

Table 2. Mean dissolved organic carbon (DOC) loads in the different treatments

Treatment[1]	Season[2]			
	Autumn-winter		Spring-summer	
	-------------------- kg DOC ha^{-1} --------------------			
F	39.67 ± 7.21	abc	10.80 ± 2.28	abc
Lm (0 N)	22.20 ± 4.32	de	14.51 ± 2.28	a
Lm (150 N)	49.50 ± 10.19	a	7.31 ± 2.43	bc
Tr (0 N)	29.05 ± 5.17	abcde	6.49 ± 2.43	c
Tr (150 N)	31.22 ± 5.50	abcde	6.54 ± 2.28	c
Zm–F (250 N)	46.60 ± 9.18	ab	6.74 ± 2.28	c
Zm–F (400 N)	35.14 ± 6.21	abcd	8.80 ± 2.28	abc
Zm–Lm (250 N)	27.31 ± 4.92	bcde	13.18 ± 2.28	ab
Zm–Lm (400 N)	25.70 ± 5.04	cde	14.71 ± 2.28	a
Zm–Tr (250 N)	37.62 ± 6.73	abc	8.66 ± 2.28	abc
Zm–Tr (400 N)	30.64 ± 5.41	abcde	6.19 ± 2.28	c
Zm–Lm+Tr (250 N)	20.18 ± 4.12	e	8.39 ± 2.28	bc
Zm–Lm+Tr (400 N)	22.05 ± 4.88	de	11.61 ± 2.28	abc

[1] F = fallow; Lm = *Lolium multiflorum*; Tr = *Trifolium repens*; Zm = *Zea mays*.

[2] Means ± standard errors (autumn-winter, n = 12; spring-summer, n = 8) within columns with different letters are significantly different (ANOVA, p<0.05).

CONCLUSIONS

Higher DON and DOC loads were lost by leaching under permanent *L. multiflorum* than under a fallow soil. In permanent *L. multiflorum* treatments DON and DOC loads were higher during the autumn-winter period than during the spring-summer period.

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2.5.P

TWO TERBUTHYLAZINE MANAGEMENTS FOR SOIL AND WATER PROTECTION

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INTRODUCTION

The negative environmental impact of pesticides remains a topic of interest, especially when rivers, reservoirs and ground waters are affected and it largely influences human health. One of the areas most affected by this type of contamination is that surrounded by olive orchards in Mediterranean areas as Southern Spain, which has recently been linked to surface and ground water pesticide pollution episodes (Hermosín et al, 2013¸ Masiá et al., 2013). The research works have directed all their efforts to minimize this contamination risk by using different strategies (Reichenberger et al., 2007). In this sense, two main strategies have been selected to verify its effectiveness: a culture practice as cover crops (CC); and the application of herbicide as slow release formulation (SRF). In relation to the first one, few works have studied the effect of cover crops, as culture management, for reducing the leaching of pesticides, especially in the case of olive orchards. Regarding the use of SRFs, which the active ingredient of pesticide is supported in an adsorbent or carrier, has been widely studied (Carrizosa et al., 2000; Cabrera et al., 2015), mainly in order to obtain modified standard clays in laboratory. However, scarce works are based on natural, local and low values clays, and commercial organoclays, as carriers for SRFs. Thus, our objective was to compare the behaviour of the herbicide terbuthylazine in soil leaching processes at lab scale, under two types of soil and herbicide managements, and asses the possible advantage of each strategies to minimize herbicide water contamination

MATERIALS AND METHOD

Soils, carriers and formulations: Two soils devoted to olive orchards were used for each leaching experiment: S1, with and without cover crop and S2 for SRFs. The soil S1 was maintained with a cover crop for 8 years (CC). To enable comparison of the results, the soil S1 without cover crops also was used. This soil was kept bare by means of conventional tillage (CT) consisting of regular chisel plow passes. The main physicochemical properties of the soils S1 (CC and CT) and S2 are summarized in Table 1.

Table 1: Physicochemical properties of soils S1 and S2

Soils	Management	Sand (%)	Silt (%)	Clay (%)	pH	$CaCO_3$ (%)	O.M. (%)
S1	CT	56	29	14	8.53	27.20	1.16
	CC	61	22	16	8.58	23.80	1.94
S2	SRF	64	17	18	7.87	9.80	1.57

The carriers used for SRFs study were: a natural and low value clay (CTI) and the commercial organoclay Cloisite 30B® (Cl-30). Two SRFs at different herbicide rates were prepared with these two clays: CTI-terbuthylazine at 4% (w/w) and Cl-30-terbuthylazine at 20% (w/w). Both SRFs were prepared as strong complex adding 5 ml of methanol to a mixture of terbuthylazine/clay (20mg/480mg for CTI, and 100mg/400mg for Cl-30), shaking 24 h, and also grinding after methanol evaporation. Commercial terbuthylazine formulation (Cuña® 50% w/w) was used in both leaching experiments.

Leaching experiment in soil S1: Three replicates of hand-packed glass soil columns were filled with 140 g of soil S1 (CC and CT). Prior to herbicide application, soil columns were water saturated and let drained for 24h. In both cases, commercial terbuthylazine was applied on the top of the columns at field dose (1 kg/ha). 15 ml of water were daily applied to the columns and the leachates were analysed by HPLC in order to know the amount of herbicide leached. After total leaching of herbicide, the soil of the columns was divided in four sections and extracted each one with 100 ml of methanol, shaking during 24 h and centrifuging 10 min at 8.000 r.p.m. The supernatants were filtered (0.45 µm) and analysed by HPLC to obtain the amount of terbuthylazine retained in soil.

Leaching experiment in soil S2: Three replicates of soil hand-packed columns were filled with 160 g of soil S2 for each treatment (commercial, CTI-terbuthylazine and Cl-30-terbuthylazine formulations). Soil and leachates analysis were done as described above.

Herbicide analysis: Terbuthylazine analysis in leachates and soil extracts was carried out by HPLC in a 1525 Waters (Milford, MA) system under the follow conditions: Mobile phase of 50% acetonitrile: 50% water; UV detection at 220 nm wavelength; flow of 1ml/min and 25 µl injection volume. The limit of detection (LOD) and limit of quantification (LOQ) for terbuthylazine was 0.01 and 0.05 µg/ml, respectively

RESULTS AND DISCUSSION

The results obtained in the leaching experiment of terbuthylazine in soil S1 with cover crop (CC) and with conventional tillage (CT) are shown in Figure 1. These results were represented as breakthrough curves (BTCs) and cumulative leaching curves after a total of 800 ml of water added. The maximum peak concentration of terbuthylazine shifted to the right in the case of CC versus CT, indicating herbicide retention under CC. Also, the maximum concentration of terbuthylazine found in leachates decreased from 0.3 mg/l in CT to 0.1 mg/l in CC. This effect was more noticeable in cumulative curves, where the total amount of terbuthylazine leached during the experiment was higher in CT (68%) than in CC (20%). Under CC management, an evident increase in soil O.M. (1.94%) was observed compared to CT management (1.16%), as well as was found by other authors (Dabney et al., 2001). This soil O.M. increase had a direct effect on herbicide soil sorption, and thus, minimizing the risk of water contamination by leaching of terbuthylazine. Even, the additional effect of CC as barriers to control soil erosion could reduce the losses of pesticides by runoff.

Results from leaching experiment with terbuthylazine in soil S2 in different types of formulations (commercial and SRFs) are represented In Figure 2, as BTCs and cumulative curves. Maximum leachate concentrations decreased from 0.35 mg/l for commercial terbuthylazine to 0.18 mg/l for CTI-terbuthylazine formulation. In the case of Cl-30-terbuthylazine formulation, the maximum concentration of terbuthylazine was in the range of 0.05-0.1 mg/l during the whole experiment, until complete disappearance of the herbicide. The shape of this curve (Cl-30) was particular to a pesticide released very slowly from the sorbent, probably due to a strong sorption process. The nanohybride (organic-inorganic) character of this clay can contribute to this high sorption. The sorption of terbuthylazine to Cl-30 organoclay is established through hydrophobic interactions to the organic fraction favouring the releasing of small amount of pesticides with the continuous water applications on the top of the columns (Hermosín et al., 2006). This effect was not so noticeable in CTI where only electrostatic interactions can take place between the herbicide and the adsorbent. In any case, the total amount of terbuthylazine leached was clearly reduced from commercial terbuthylazine (90%) to CTI (58%), and Cl-30 (20%) (Figure 2, right).

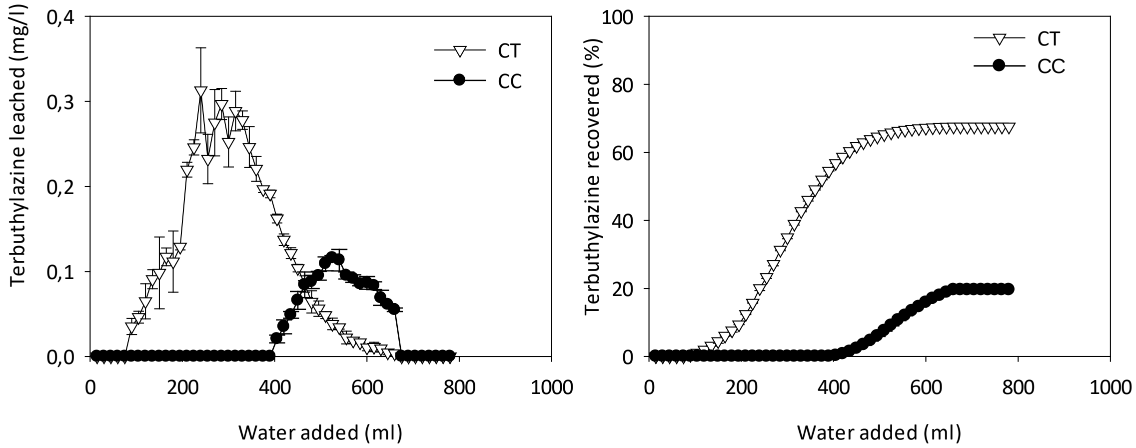

Figure 1: Terbuthylazine BTCs (left) and cumulative leaching curves (right) in soil S1 under CT and CC.

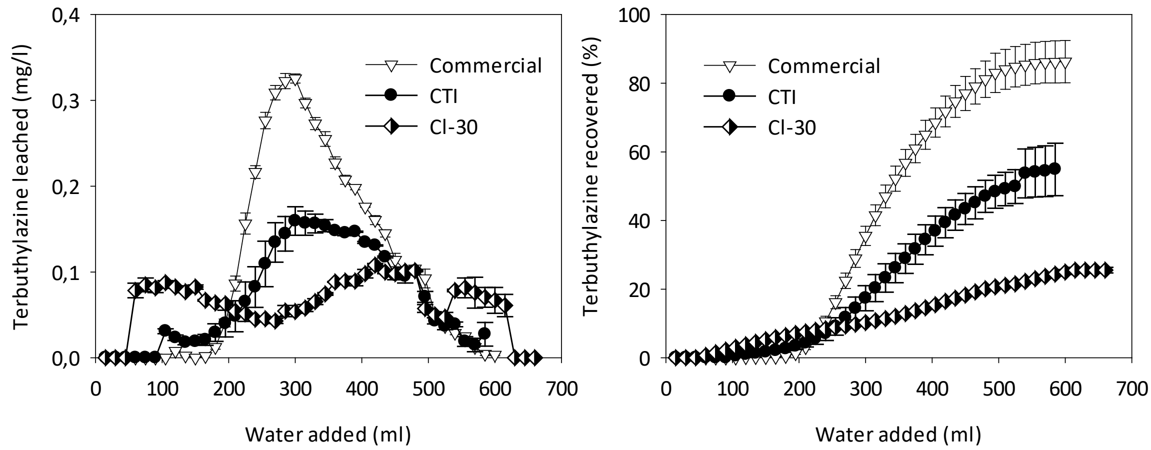

Figure 2: Terbuthylazine BTCs (left) and cumulative leaching curves (right) in soil S2 as commercial formulation and as SRFs (CTI and Cl-30).

After complete leaching of terbuthylazine from the hand-packed soil columns, soil was extracted with methanol in order to know the herbicide retained. In Table 2 are summarized the results of this extraction together with the total terbuthylazine leached and no recovered from both leaching experiments. The lower amounts of terbuthylazine extracted from soil columns corresponded to soil S1 (0% for CT and 7% for CC) in comparison with those found in soil S2 (8% for commercial, 10% for CTI and 25% for Cl-30). When a balance between terbuthylazine applied and recovered (leached and extracted) was made, we obtained the amount of terbuthylazine no recovered in both experiments, which followed the trend: CC>Cl-30>CTI>TT>commercial. The amounts of herbicide no recovered were related with strong sorption and /or biodegradation processes. These processes gained importance under CC management, due to the increase in soil O.M. content, and under SRF management with commercial organoclay (Cl-30), because the interaction between pesticide and the organic fraction of the clay.

Table 2: Terbuthylazine recovered (leaching and soil extraction) and no recovered, as percentage, in leaching experiments with soils S1 and S2.

	Soil managements (S1)		Herbicide formulations (S2)		
	CT	CC	Commercial	CTI	Cl-30
Leached	68	20	86	55	25
Extracted	0	7	8	10	20
No recovered	32	73	6	35	55

CONCLUSION

The results show that both strategies, soil management with CC and use of SRFs with low value clays and commercial organoclays, are very promising for water contamination protection and as sustainable use of the herbicide terbuthylazine for healthy crop production.

ACKNOWLEDGEMENTS

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2.7.P

CONTROL DE MALEZAS MEDIANTE USO DE COBERTURAS LEGUMINOSAS EN PLANTACIONES DE TECA EN COSTA RICA.

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ABSTRACT

International forest certification has increasingly forced us to seek more environmentally correct solutions to weed control that should be applied to forest plantations, which aim at exporting certified wood. Traditionally, in Costa Rica, weed control in forest plantations is deficient. Some mathematical models have shown significant losses due to the presence of weeds, without seeking compatible alternatives for weed control. Within an ITCR research project and in collaboration with a reforestation company (LTC) a weed control trial was established in the Southern Zone of the country on a newly established *Tectona grandis* plantation. An experimental design of random complete blocks was established, using three cover species, *Canavalia ensiformis*, *Crotalaria juncea* and *Vigna radiata*, compared to three management treatments, chemical control, manual weeding and a control. Preliminary results show that weed levels in the coverages are smaller, as expected; there have been increases in nitrogen levels in the soil and leaf level of Teak. The results indicate the most appropriate species to weed control weed, allowing at the same time to reduce the input of nitrogen fertilizers.

INTRODUCCIÓN

El control de malezas es una actividad imprescindible para el manejo de plantaciones forestales. Se encuentra bien documentado que las arvenses o malezas afectan el crecimiento y disminuyen la productividad de las plantaciones en sus estadios iniciales (Adams et al 2003, Garau et al 2009, Ladrach, 2010).

A pesar de que las malezas se pueden controlar de forma manual, mecánica o química; su alto costo, posibilidad de compactar el suelo (control mecanizado) y en algunos casos su poco efecto en el tiempo (control manual) limitan mucho la cantidad y calidad de las operaciones. Aunado a lo anterior, las actuales restricciones de organizaciones como la OMS y el FSC para utilizar productos químicos reducen aún más las posibilidades de empresas reforestadoras y pequeños productores (Jabran et al 2015, FSC 2015).

A nivel mundial y principalmente en la agricultura, los cultivos de cobertura se han convertido en uno de los métodos alternativos de control de malezas y de remplazo de herbicidas más importantes. Entre sus beneficios se reconocen la supresión de malezas ya sea por competencia de luz, agua y nutrientes o por liberación de sustancias alelopáticas de tejidos vivos o en descomposición (Burst et al 2014). Paralelamente, los cultivos de cobertura proporcionan otros tipos de beneficios como enriquecer el suelo mediante la fijación de nitrógeno, mejorar su estructura y controlar poblaciones de herbívoros.

En el campo forestal son pocas las experiencias desarrolladas a nivel centroamericano. Destacan las tesis de Arias (1998) y Sima (2010) donde evalúan el porcentaje de cobertura en el suelo y el efecto de mezclar algunas especies leguminosas con métodos tradicionales de control de malezas. En Costa Rica no se reportan estudios científicos de uso de cultivos de cobertura en plantaciones forestales por lo que existe un vacío importante de información que es importante llenar. Ante la necesidad por parte del sector reforestador de Costa Rica de buscar nuevas alternativas para el control de malezas que sean menos costosas, más eficientes en el tiempo y que puedan traer algún beneficio al cultivo. El objetivo del presente trabajo fue evaluar distintos cultivos de cobertura como alternativa de solución al control de malezas, y como efecto maximizador del crecimiento en plantaciones recién establecidas.

MATERIALES Y MÉTODOS

Área de estudio: El ensayo se realizó en la localidad de Salamá, distrito de Piedras Blancas, cantón de Osa, provincia de Puntarenas, Costa Rica (8°48'41.54" Lat Norte y 83°17'37.39" Long Oeste); a una altitud de 20 m, con una temperatura promedio de 26,8° C y una precipitación de 4450 mm. El suelo del sitio pertenece al suborden Inceptisol (Atlas Digital 2014).

Diseño experimental: Se utilizó un diseño experimental de bloques completos al azar, que consta de 7 tratamientos en total con 3 repeticiones cada uno. Los tratamientos a aplicar son: T0: Testigo, T1: Cobertura vegetal *Canavalia ensiformis*, T2: *Vigna radiata*, T3: Cobertura vegetal *Crotalaria sp.*, T4: Mezcla entre Vigna *radiata* y Pueraria phaseloides, T5: Control de malezas manual y T6: Control de malezas químico

Cada unidad experimental o parcela cuenta de 64 árboles incluyendo los bordes y de 36 árboles para cada parcela útil.

Preparación del terreno y siembra de coberturas: Antes de establecer los sitios de ensayo, se mecanizó el terreno utilizando subsolador y rastra. Posteriormente se establecieron los camellones. Estas áreas estuvieron libres de arvenses al momento de la siembra.

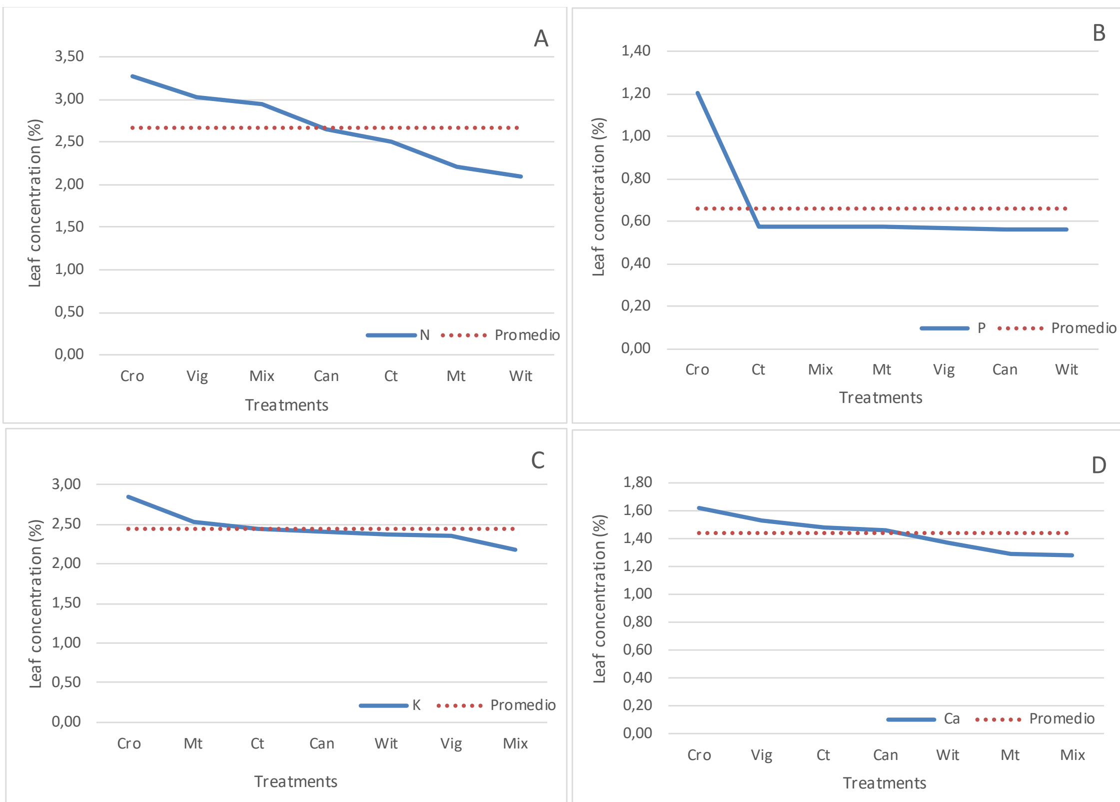
Los cultivos de cobertura fueron sembrados al pie de cada árbol y a lo largo del surco. Estas fueron sembradas al momento del establecimiento de la plantación.

Variables a evaluar: Para evaluar el establecimiento y desarrollo de las coberturas se utilizó la variable porcentaje de cobertura. Los muestreos para la obtención del porcentaje de cobertura fueron a las 2, 4, 8 y 16 semanas de establecido el ensayo.

Muestreo foliar: Para el muestreo de análisis foliar se aplicó la metodología modificada de Murillo et al. (2014). Las hojas fueron almacenadas en bolsas negras, colocando una etiqueta con el respectivo tratamiento; posteriormente fueron colocadas en hieleras para evitar la degradación de las mismas durante el traslado del material desde el sitio del ensayo hacia el campus. Las muestras fueron enviadas al laboratorio.

RESULTADOS Y DISCUSIÓN

La concentración promedio de Nitrógeno para todas los cultivos de cobertura estuvo dentro de los niveles adecuados, pues éstos se encuentran dentro de los límites establecidos por Alvarado y Raigosa (2012) (Figura 1. A.). El tratamiento correspondiente a Cobertura Crotalaria presentó los mayores niveles promedio de concentración en los elementos N, P, K, Ca (Figura 1. A.B.C.D) respectivamente, esto puede deberse a que anteriormente se hizo una corta completa debido a la altura de la cobertura y posiblemente parte de la materia orgánica ya se había incorporado al suelo; en cuánto al Mg, Crotalaria no obtuvo la mayor concentración, sin embargo, se mantuvo por encima del promedio. A pesar de que Crotalaria presenta buenos niveles nutricionales no se recomienda para plantaciones recién establecidas, pues es muy agresiva en cuanto a su crecimiento, superando a los 2 m de altura, lo cual genera sombra a los arboles de teca, ésta cobertura podría ser una buena herramienta para el control de malezas en plantaciones ya establecidas.



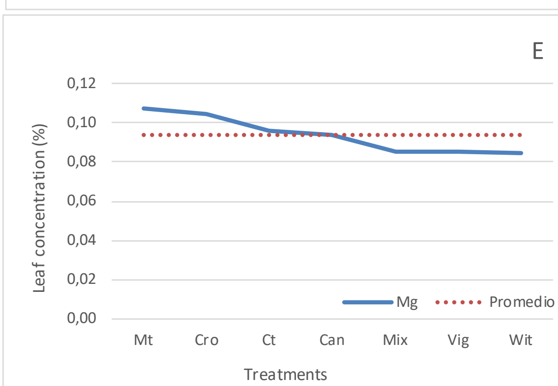


Figura 1. Concentración foliar de Nitrógeno (A), Fósforo (B), Potasio (C), Calcio (Ca), Magnesio (E), para siete tratamientos: Cobertura Crotalaria, Cro; Cobertura Vigna, Vig, Cobertura Canavalia, Can; Control Manual, Mt; Control Químico, Ct; Mezcla, Mix; Testigo, Wit, establecidos en una plantación de teca recién establecida ubicada en Salamá, Puntarenas, Costa Rica.

Las cultivos de cobertura con especies leguminosas son una opción viable para el aumento de Nitrógeno a nivel del suelo como a nivel foliar, ya que éstas tienen una gran capacidad de absorción de nutrientes en las capas más profundas del suelo, aumentando la concentración de los mismos en capas más superficiales; así como el beneficio brindado al cultivo principal por la fijación biológica de éste elemento (Frageria et al 2005).

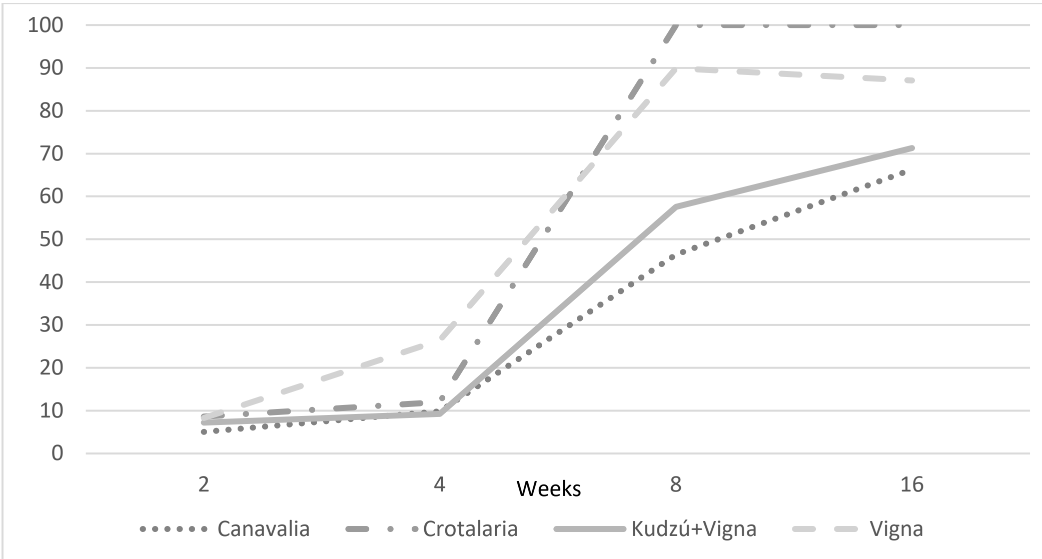


Figura 2. Porcentaje de cobertura de distintas especies leguminosas utilizadas como control de malezas en plantaciones recién establecidas de *Tectona grandis.*

Como se puede observar en la figura 2 las especies seleccionadas empiezan a mostrar su potencial para el control de arvenses a partir de las 4 semanas. Las especies Crotalaria y Vigna mostraron mayor porcentaje de cobertura del área, logrando un 100% a las ocho semanas de establecido el ensayo. Sin embargo, Crotalaria no se recomienda para plantaciones recién establecidas ya que triplicó la altura de las plántulas de teca. A pesar de este aspecto, esta especie puede tener un gran potencial para el control de malezas en plantaciones mayores a 1 año.

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2.8.P

WATER USE AND WATER USE EFFICIENCY OF OILSEED CROPS IN NORTHEASTERN MONTANA

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INTRODUCTION

Oilseed crops have been increasingly grown under dryland in the northern Great Plains region in the US because of their profitability and other agricultural benefits (Johnson et al., 2002). Oilseed crops are the most promising feedstock for biodiesel production. Little is known about oilseed crops' water use characteristics in dryland environments, thus, knowledge of crop water use (WU) in dryland semiarid environments is important for crop water use efficiency (WUE) and optimizing crop production systems (Kar et al., 2007). The objective was to evaluate the 2014 WU and WUE of 12 spring varieties and one winter variety of cool-season oil-seed crops in a field study under dryland conditions.

METHODS

The study was conducted at a USDA-ARS research site located about 8 km northwest of Sidney, Montana, USA (47°46'N, 104°16'W; elevation 690 m). Soil was mapped as a Williams loam (fine-loamy, mixed, superactive, frigid Typic Argiustoll).

Soil at the 0- to 30-cm depth had 350 g kg^{-1} sand, 325 g kg^{-1} silt, 325 g kg^{-1} clay; pH of 6.1and organic matter of18 g kg^{-1}. Long-term mean annual precipitation at the research site was 357 mm, with about 77% occurring from April through September. Monthly average, maximum, minimum, air temperature, and monthly total precipitation for 2013-14 growing seasons were collected from a weather station at the study site (Table 1).

These 13 varieties are grouped under six species (camelina sativa, Sinapis alba, Brassica rapa*,* Brassica napus, Brassica juncea, and Brassica carinata). All varieties were direct-seeded into plots each 3 m × 9 m under no tillage conditions. Treatments were replicated four times in a randomized block design. Seasonal WU was estimated using a soil water budget equation. Water use efficiencies for all 13 varieties were evaluated in terms of above ground biomass, seed yield and seed oil yield.

Table 1. Monthly average, maximum, and minimum air temperature and total precipitation during 2013-2014 growing seasons at the study site.

Month	Air temperature, °C Average	Maximum	Minimum	Precipitation, mm	
January	-8.2	9.4	-28.8	3.0	
February	-12.0	7.7	-26.0	1.5	
March	-3.3	16.6	-29.9	9.4	
April	5.0	22.6	-14.2	31.8	
May	12.6	30.7	-2.9	89.4	
June	16.0	26.6	4.2	32.3	
July	20.4	34.0	7.9	12.7	
August	19.5	34.0	4.5	101.9	
Winter camelina 2013-2014 cumulative growing season precipitation					336.8
Oilseed crops 2014 cumulative growing season precipitation					236.2

Two plant samples were hand harvested and yield data (seed and oil) are based on two locations of 1 m × 3 rows with 20 cm distance between rows in the center of each plot.

Actual crop water use (WU) is the sum of evaporation from the soil surface and transpiration from plant (mm). Crop water use was calculated as:

$$WU = R - (\theta_v post - \theta_v pre) \qquad [1]$$

where R is the amount of seasonal precipitation (mm) from Table 1, $\theta_v post$ is volumetric soil water content at post-harvest, $\theta_v pre$ is volumetric soil water content at pre-planting (the change in stored water content of the 100 cm soil profile during the growing period (mm)). Calculations of WU are based on the assumption that runoff and drainage from the plots were negligible.

Volumetric soil water content (θ_v) in the soil profile (0 -100 cm depth) at pre-plant and post-harvest (mm) for each growing season was calculated using a gravimetric method.

Crop water use efficiency, *WUE* (kg ha^{-1} mm^{-1} or kg m^{-3}) is mathematically defined as:

$$WUE = \frac{Y}{WU} \qquad [2]$$

Where Y is the yield (seed, and oil) in kg ha^{-1} and *WUE* the amount of water consumed by the crop during the growing season.

RESULTS:

Crop WU averaged across species was 314, 269, 221, 258, 268, and 290 mm for camelina, S. alba, B. rapa, B. napus, B. juncea, and B. carinata, respectively. Water use results showed that different crops consumed different amounts of water during the growing season because all crops were grown under same soil and microclimatic conditions.

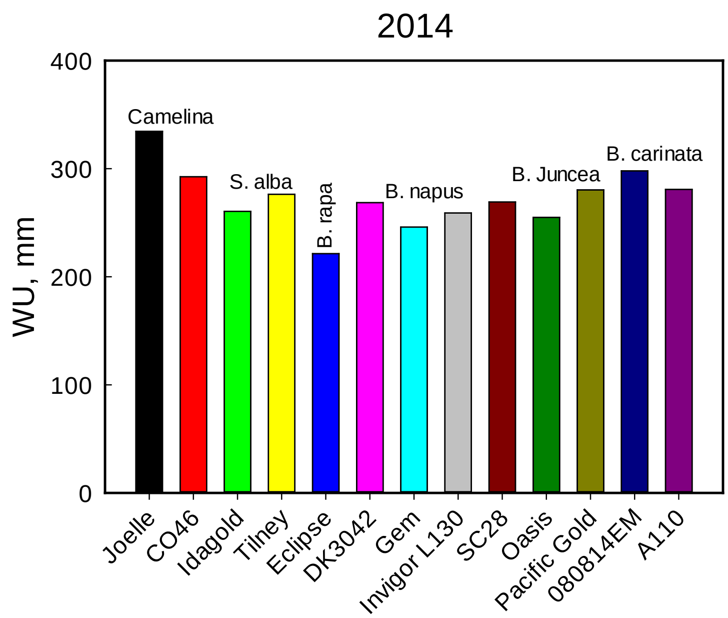
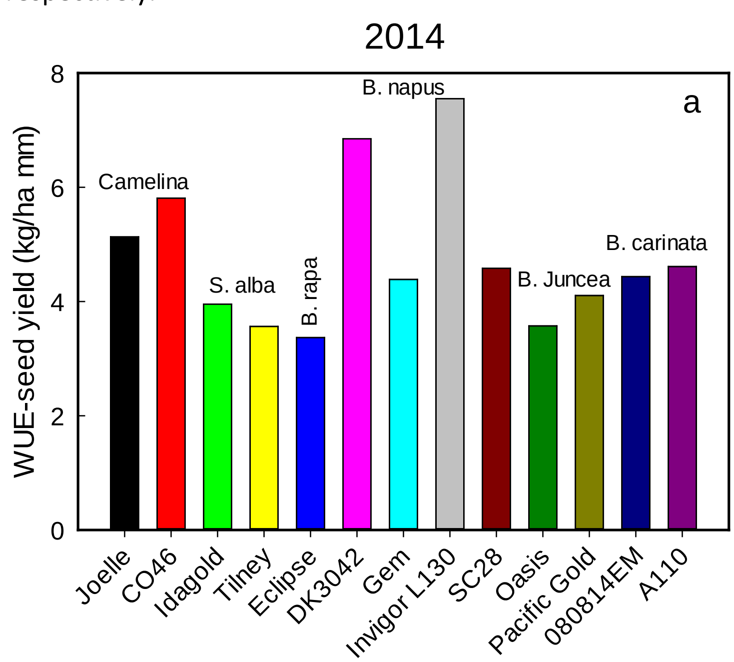


Fig.1. Water use during the 2013-2014 growing season.

Average WUE in terms of seed yield and oil yield was 5.47, 2.16; 5.47, 2.16; 3.37, 1.32; 6.26, 2.85; 4.09, 1.76; and 4.53, 1.77 kg ha^{-1} mm^{-1}, for camelina, S. alba, B. rapa, B. napus, B. juncea, and carinata, respectively.



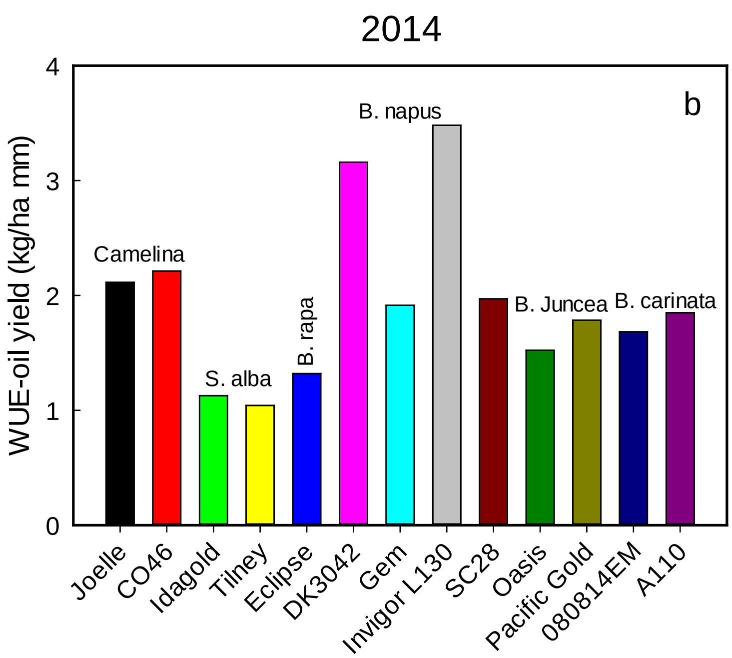


Figure 2. Water use efficiencies of (a) seed yield and (b) oil yield.

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2.9.P

SLOW-FORMING TERRACE SYSTEMS –BLESSING OR CURSE FOR SMALL-HOLDER FARMERS IN THE ANDES?

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INTRODUCTION

Soil erosion has been linked to the destruction of several civilizations (e.g. Diamond 2005, Lowdermilk 1950, Mieth & Bork 2003) and still forms a worldwide persistent threat to food security (Pimentel & Burgess 2013). One of the hotspots of soil erosion are the Andes, where as in many developing countries marginal land on slopes is cultivated (Lal 2004). In these areas labour or machinery and related financing for implementing soil conservation measures, such as common bench terracing, are limited. Slow-forming terraces have therefore widely been promoted by governmental and non-governmental organisations. However, there have been concerns about the sustainability of this method:

- The intrinsic characteristic of this method is the redistribution of topsoil from the upper to the lower part of the terrace by tillage erosion. Especially in shallow soils like in the Andes, the infertile subsoil is soon reached in the upper part of the terrace and likewise spread on it. This can lead to a gradient in soil properties within the terrace (Dercon 2003)
- Slow-forming terrace systems use barrier stripes consisting of different grasses and/or bushes and trees. On the edge of the terraced field these plants in the barrier stripes can compete with the crops for nutrients and/ or water resources (Dercon et al. 2006, Pansak et. al. 2007)

Both concerns can lead to reduced yields. Such a yield reduction can lead to a rejection of the conservation method by the farmers, especially in combination with the increased effort or difficulty of field preparation in these systems. These concerns have been raised from short-term analyses which have been undertaken 2-4 years after the initiation of the conservation measure. A long-term evaluation of the effectiveness of slow-forming terraces is missing up to date and is the objective of this study.

MATERIALS AND METHODS

The terraces selected in this study, located at 2770 m a.s.l. in Gima, Azuay province, Ecuador, have been analysed 18 years ago by Dercon et al. (2003 & 2006). In this study, concerns about the sustainability of the conservation method have been raised, due to an observed gradient in both the soil properties as well as the plant characteristics. It was suggested, that the development of the gradient was caused by redistribution of the soil through tillage, leading to the exposure of infertile subsoil in the upper part of the terrace as well as by competition with the plants of the barrier stripe in the lower part. To investigate the long-term development of the slow-forming terraces, soil properties as well as yield distribution and plant traits are assessed and compared to the results of

the short-term analysis. The selected terraces are managed by local farmers using chicken manure and animal-powered tillage and thus give a good insight in their performance and development under real farmers' conditions in the Ecuadorian Andes.
Soil samples have been collected using a systematic grid and are currently being analysed for physical and chemical soil properties like bulk density, macro nutrients, pH and others. First results of bulk density, pH and soil organic carbon are available.

Furthermore, plant traits including height, diameter, yield (as dry weight of kernels) and ^{13}C discrimination in leaves and kernels have been measured for seven randomly chosen samples within each meter width of the terrace. The plant density was observed to be partly heterogeneous within the fields. Therefore the distance to the nearest plant was also noted for each plant sampled. Since it was observed, that some ears have not been fully pollinated, the length of the ears was also measured as an additional indicator of the potential yield with respect to complete pollination. Since some plants had more than one ear, both the length of the first ear as well as the cumulative length of the ears of one plant were assessed

PRELIMINARY RESULT

The analysis is currently ongoing and some results are still pending. First preliminary results show considerable improvements in soil properties of the slow-forming terrace systems in general and furthermore a reduction of the markedness of the gradient within the terraces. Till date the main observed characteristics are:

Location of MIN and MAX values within the terrace (Table 1):

MIN and MAX location of analysed plant traits: short time analysis by Dercon et al (2006) showed a very clear pattern of MAX values at the lower end of the alleys (normalized positions between 0,89 and 1,0), while the MIN values were always observed at the upper part of the terrace (normalized positions between 0,08 and 0,00).
This situation has changed and the MIN and MAX values of several observed plant characteristics do not show any such clear characteristic trend for their position. Interestingly the MAX of the yield was now even observed in the upper part of the terrace -between width positions 0,2 (1st meter) and 0,6 (3rd meter)- in comparison to the results from the short-term observation, which were 0,8 and 1,0 (4th and 5th meter) -the lower part of the terrace. Only one characteristic was consistently observed at the same normalized width position in all terraces: the minimum height of the plants was always observed at position 0,8 (4th meter), while the maximum plant heights were observed in the middle of the field at width positions 0,4 or 0,6 (2nd or 3rd meter).
Both MIN and MAX values of the distance to the closest neighbouring plant were measured in the upper part of the terrace (width positions 0,2 and 0,4 – 1st and 2nd meter)). The values varied between 2 cm and 95 cm.

Table 1: Comparison of minimum and maximum observations of plant traits

Observation 22 years after initiation								Observation 4 years after initiation	
MAX	height	diameter	closest plant	length of ear1	cum. Length of ears	dry weight	corr. dry weight	length of wheat stalk	dry weight grains
TL1	0.40	0.40	0.40	0.40	0.20	0.60	0.60	-	-
TL2	0.60	0.60	0.20	0.50	1.00	0.20	0.20	0.94	1.00
TL3	0.60	1.00	0.20	1.00	0.60	0.40	0.40	0.95	1.00
TR1	0.60	1.00	0.20	1.00	0.60	0.40	0.40	0.89	0.92
TR2	-	-	-	-	-	-	-	1.00	1.00
average	**0.55**	**0.75**	**0.25**	**0.73**	**0.60**	**0.40**	**0.40**	**0.95**	**0.98**
stdt dev	**0.10**	**0.30**	**0.10**	**0.32**	**0.33**	**0.16**	**0.16**	**0.05**	**0.04**
MIN	height	diameter	closest plant	length of ear1	cum. Length of ears	dry weight	corr. dry weight	length of wheat stalk	dry weight grains
TL1	0.80	0.60	0.20	1.00	0.80	0.20	0.20	-	-
TL2	0.80	0.20	0.40	0.60	0.60	0.40	0.40	0.00	0.00
TL3	0.80	0.20	0.20	0.20	0.20	1.00	1.00	0.00	0.08
TR1	0.80	0.20	0.20	0.20	0.20	1.00	1.00	0.00	0.00
TR2	-	-	-	-	-	-	-	0.00	0.00
average	**0.80**	**0.33**	**0.27**	**0.60**	**0.53**	**0.53**	**0.53**	**0.00**	**0.02**
stdt dev	**0.00**	**0.23**	**0.12**	**0.40**	**0.31**	**0.42**	**0.42**	**0.00**	**0.04**

Impact of width position on plant traits

A three-way analysis of variance (ANOVA) was carried out to see the dependency of the observed plant traits to the terrace, width position (w) and intraspecific competition (Table 3). The values were normalized for each terrace in order to be able to compare the spatial distribution within the terraces. The index of intraspecific competition was categorized from the measured distance to the nearest corn plant.

The width position, which is the most important with respect to the evaluation of the terrace system development, showed a significant impact on the vegetative traits. Post hoc TukeyHSD shows significant differences of the vegetative traits –height, diameter and stem volume – of the 4th meter width to the 1st, 2nd and 3rd meter width p=0.01 level. The performed post hoc TukeyHSD shows furthermore significant differences of the diameter of the stems between no competition and high competition at the 0.05 level.

However, none of the tested parameters had a significant influence on the yield, which is the most important measure for the farmer.

Table 2: ANOVA results showing p-values of significance; p-values < 0.05 are highlighted

	Terrace	w	Competition	Terrace X w	w X Competition	Terr X Competition	Terr X w X Comp
Height	0.75903	0.00040	0.53537	0.88589	0.43578	0.21451	0.52250
Diameter	0.05960	0.02750	0.10630	0.04030	0.90250	0.97060	0.40970
Stem vol.	0.02855	0.00977	0.27815	0.19764	0.77633	0.89091	0.51401
Yield	0.85700	0.55300	0.62430	0.83470	0.46700	0.14270	0.08350
Length ears	0.10500	0.18600	0.11200	0.92800	0.24200	0.20200	0.11200
Length ear1	0.00002	0.08050	0.32510	0.49020	0.86700	0.51100	0.55900

Vegetative plant characteristics like height and circumference of the stem show a slight decrease in the lower part of the terrace as shown in Figure 1a) and 1b). Interestingly this trend is not significantly represented in the reproductive traits such as yield and length of the ears.

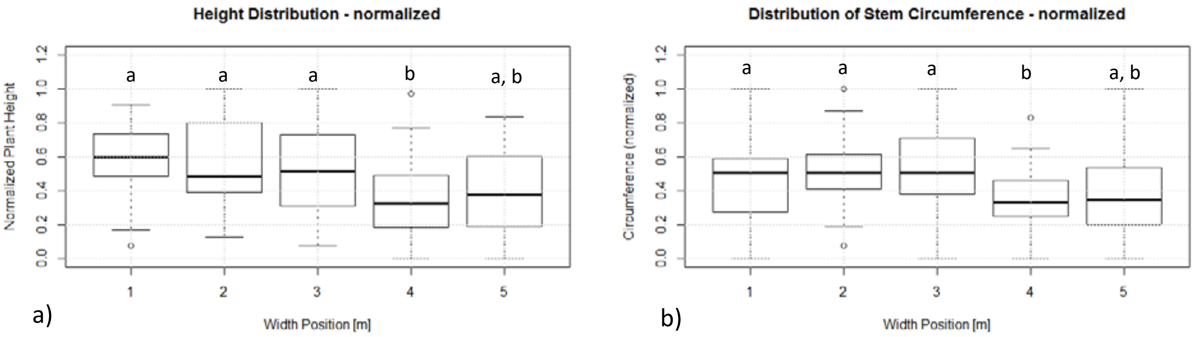

Figure 1: Box-Whisker-Plots of the Distribution of Different Plant Characteristics (identical letters show no significant difference): a) Plant Height, b) Stem Circumference

Pearson correlation was performed and is shown in Table 3. Plant height showed a significant dependency to the width position w, but with a low correlation factor of -0.36. Further results of the soil analysis are awaited to continue the analysis.

*Table 3: Pearson Correlations of Preliminary Results (Normalized); significance indication: *** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$*

	w	SOM	pH	bulk_d	field_moisture	height	circumference	stem.vol	distance.to.plant	length.of.ear	added.length.of.ears
w											
SOM	0.10										
pH	0.23*	-0.50***									
bulk_d	-0.01	0.00	-0.26**								
field_moisture	-0.27**	-0.01	0.00	0.44***							
height	-0.36***	0.05	-0.19	0.08	0.16						
circumference	-0.09	-0.30**	0.27**	-0.03	0.11	0.36***					
stem.vol	-0.15	-0.24*	0.14	0.04	0.15	0.62***	0.91***				
distance.to.plant	-0.10	-0.10	0.03	-0.09	0.02	0.03	0.10	0.13			
length.of.ear	0.06	-0.05	0.14	-0.16	-0.14	0.12	0.16	0.11	-0.04		
added.length.of.ears	0.01	-0.01	0.01	-0.03	-0.03	0.19	0.28**	0.32**	0.33***	0.29**	
corrected.dry.weight	-0.12	0.02	-0.07	0.03	-0.01	0.25**	0.34***	0.36***	0.12	0.21*	0.67***

First results of the soil analysis

Till date only the results of bulk density, soil organic carbon and pH are ready. All of these characteristics improved compared to the measurements performed by Dercon et al. (2003) 18 years ago. Especially the pH was a matter of concern, because the values measured in the short-term assessment were mainly between 4 and 5 and the related potential Aluminium toxicity in this pH range inhibits root growth and nutrient and water uptake (Panda et.al, 2009). The pH ranges now between 5.9 and 7.5 with an average of 6.8 and the risk of Aluminium toxicity seems eliminated. This can be explained by the common practice of adding chicken manure, which has a liming effect like shown by Jobe et a. (2007).

CONCLUSION

The preliminary results suggest that the conditions in the slow-forming terrace systems have strongly improved in the long-term compared to the short-term evaluation performed by Dercon et al. (2003 & 2006). The most important parameter for the farmer, the yield, is not significantly influenced by the width position of the plant within the terrace and therefore, the observed significant bias in the spatial distribution seems to have diminished to a degree that it is negligible. To draw concrete conclusions, the analyses of the soil and plant traits need to be completed. If the improvement gets further confirmed, it would show that the disadvantages of the conservation method can be overcome under common real-life farmer management and that slow-forming terraces are a sustainable soil conservation method against erosion. An important aspect of the farmer management is the application of poultry manure, which due to its high liming effect helped to mediate the acidic soil successfully. Future research should include surveys on the acceptance of the method depending on the management practices and feedback from the farmers. Furthermore, the large gap of 18 years between the research needs to be considered, since long-term sustainability might be outweighed by short- to mid-term profitability and its perception the farmers. Furthermore, research applying a constant monitoring on a slow-forming terrace system and analysis of the effect of different management practices should be conducted in order to be able to effectively judge the suitability and sustainability of the conservation method.

ACKNOWLEDGEMENTS:
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2.13.P

EFFECT OF SOIL COVERAGE BY CROP RESIDUES USED AS A CONSERVATION PRACTICE ON SOIL TEMPERATURE REGIME

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INTRODUCTION

Temperature is a critical factor for biotic and abiotic processes in the soil and on its surface. For instance, crop germination and emergence are processes highly dependent on soil temperatures (ST) (Angus et al., 1980). Crop emergence is a crucial step in crop development because it determines the time when plants begin to intercept solar radiation and, with this, to produce biomass.

Crop residues help to create permanent soil cover conditions, one of the principles of conservation agriculture (CA) (FAO, 2017). Crop residues reduce erosion and promote water infiltration and organic matter formation (Verhulst et al., 2010). However, residues may influence the ST regime when covering the soil surface, or if incorporated into the most superficial layer, by reducing the heat flow into the soil (Sauer and Horton, 2005). Low ST is a major drawback for CA adoption in irrigated maize-based systems as farmers aim at the earliest possible sowing in the spring to extend the cycle and avoid late-season rainfalls. Farmers could remove part of the residue from the field to increase ST while maintaining the protecting benefits. On the other hand, and depending on the crop, growing phase, and agricultural management, crop residues on the soil surface could help to avoid damaging extreme ST.

This study aims at evaluating the ST regime as affected by the percentage of soil cover by crop residues, as well as the effect of such a regime on the seedling emergence, and propose alternative residue management options for farmers.

METHODS

The experiment was conducted in a fallowed plot at the IAS-CSIC, in Cordoba, Spain (37° 51’ N, 4° 47’ W, 110 m a.s.l.), in 2011. We evaluated 4 levels of soil cover with maize residues (0-30-60-100%) and 3 replications, with plots of 2.25 by 2 m. The residue was placed on top of ploughed soil trying to reach the targeted cover of each plot by visual estimation. Then plots were photographed, and the cover was measured using an image analysis software (ENVI 4.7, Exelis Visual Information Solutions, CO, USA). When required, soil cover was modified adding or removing residue until reaching the targeted percentage of cover. 50 seeds of maize and cotton were sown in each plot at 3 cm depth on 10-Mar and 4-May, respectively. ST was measured at the center of every plot at 3 and 7 cm depth using type-k thermocouples connected to a datalogger. ST was measured every 30 minutes from 2-Mar to 26-May. Day-degree units (DD) above 9.8 and 15.5 °C for maize and cotton, respectively (Angus et. al. 1981), were calculated for each plot using ST recorded at 3 cm depth. Daily DD were summed to calculate the total DD required to fulfill the sowing-emergence phase (DDSE). Plots were considered as emerged when 50% of plants did. Globally averaged values of DDSE for both crops were used to estimate the emergence date simulating sowing every day from 2-Mar 2 to 25-May.

RESULTS AND DISCUSSION

Maximum daily soil temperatures (STMAX) decreased as cover increased (Fig. 1 A and C). There were also differences in minimum daily soil temperature (STMIN) (Fig. 1 B and D). The daily thermal amplitude was higher as ground cover decreased mainly due to the increase of STMAX. Differences in ST between treatments decreased with depth and over time as solar radiation increased, and were negligible in cloudy days (e.g. 3-Apr and 11-May). The influence that the mulch has on the soil thermal regime can be derived the differences in thermal conductivity between crop residue and soil [estimated as 0.038 W m^{-l} K^{-1} by Azooz et al. (1997) and ranging from 0.73 to 1.42 W m^{-l} K^{-1} (Villalobos and Fereres, 2016), respectively].

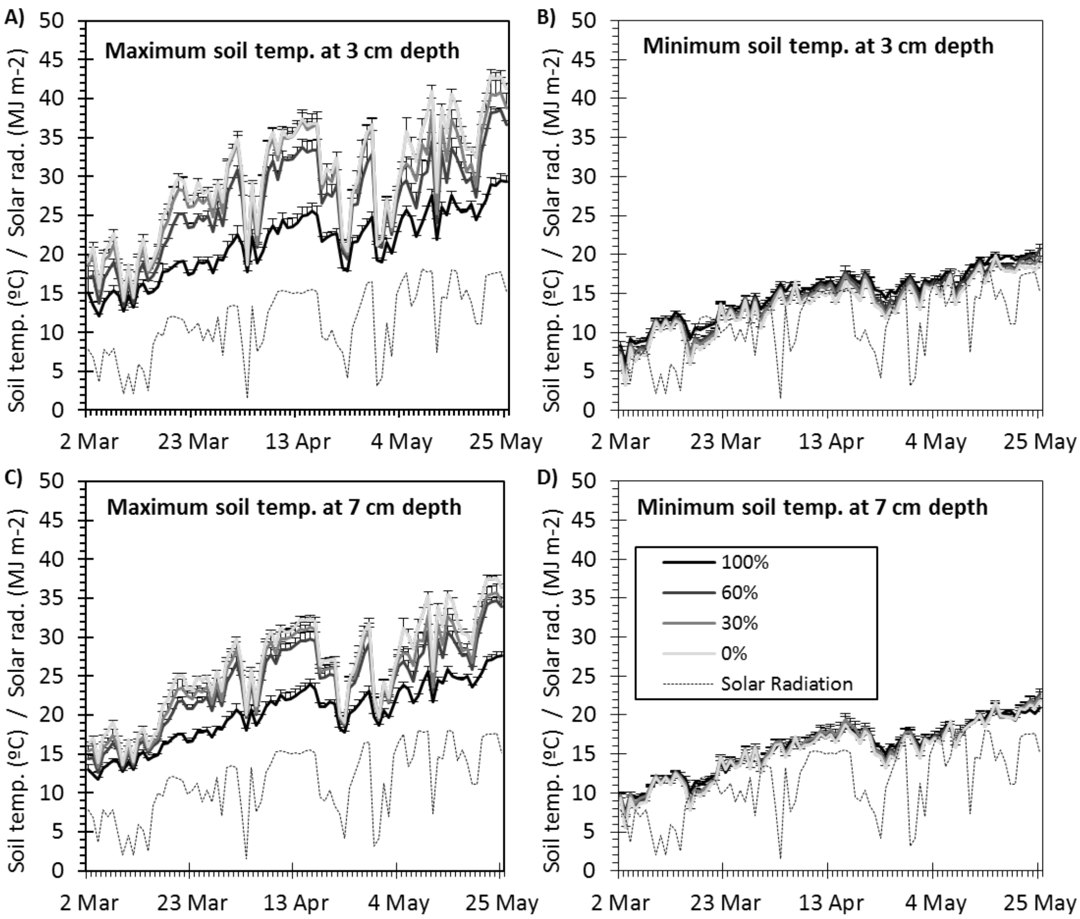

Figure 1. Daily maximum and minimum soil temperature for each treatment at 3 and 7 cm depth and daily solar radiation. The bar indicates the standard deviation value for each treatment and day.

Calculated DDSE were 51 and 81 DD for maize and cotton, respectively. Estimated number of days to emergence for every day during the study are shown in Figure 2. The greater the cover and the earlier the sowing, the greater the time required to achieve crop emergence. The number of days from sowing to emergence (Fig. 2 A-B) and the difference between treatments in such a number (Fig. 2 C-D) decreased over time, except for rainy periods with relatively low solar radiation (mid-April and early-May).

For what could be taken as a traditional sowing window for the Cordoba province (light-grey areas in Fig. 2), 30% of cover caused a maximum of 0.4- and 1.1-day delay (maize and cotton) compared to 0%, i.e. bare soil, while 60% of cover caused a maximum of 1- and 1.9-day delays (maize and cotton) compared 0%. For a targeted earlier sowing (2-15 March), the emergence of maize plants will take from 0.9 to 1.8 days more with 60% cover compared to the emergence in bare soil, therefore, maintaining up to 60% cover is probably not conflicting with a good plant stand establishment (dark-grey areas in Fig. 2). In the case of cotton, however, having soil covered if sowing takes place in late-April will result in delays that lead to an unacceptable time for plant emergence.

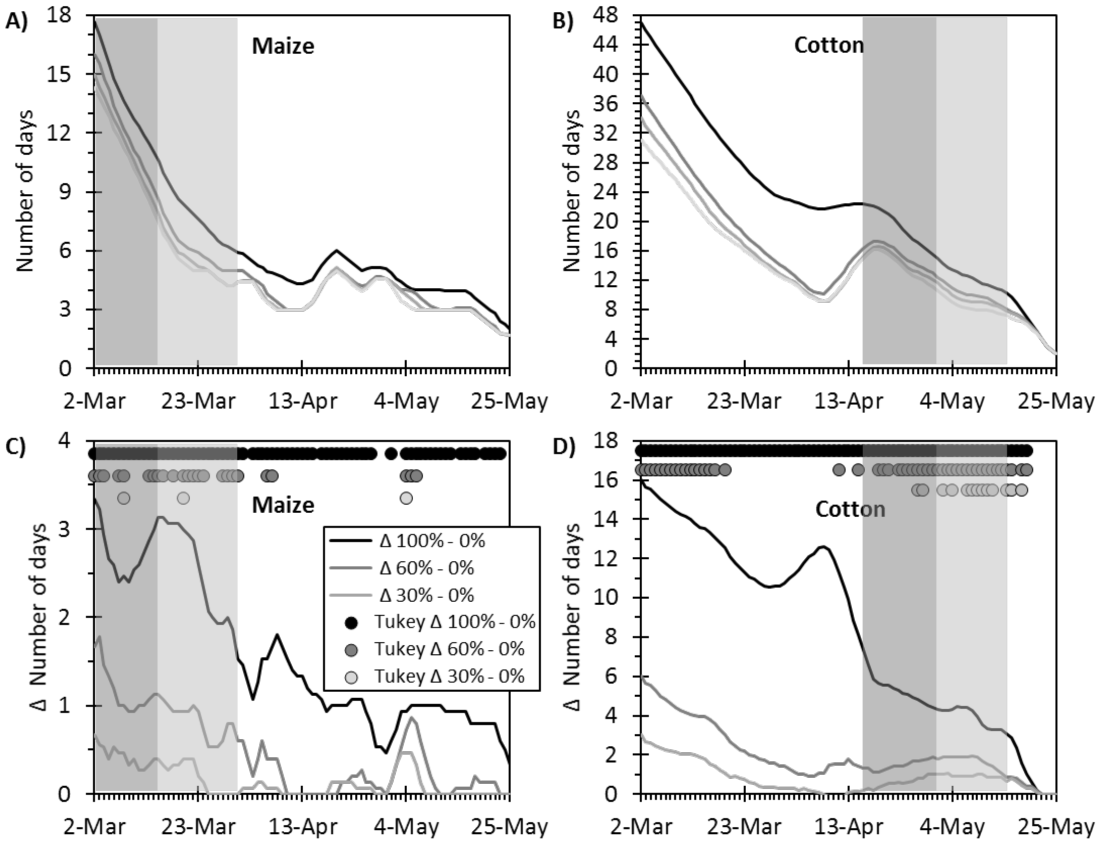

Figure 2. Time from sowing to emergence (A-B) and difference in time from sowing to emergence between treatments with soil cover (100, 60, and 30%) and bare soil (0%) (C-D). Circles in C and D indicate that the daily difference in the number of days is significantly greater than zero (Tukey HSD 5%).

CONCLUSIONS

Farming practices such as crop residue management can influence soil thermal regime. Compared to bare soil, covering the soil with crop residues up to 60% did not delay plant emergence or lead to a maximum of 1.9-day delays in conventional and early planting periods for the area. In the case of earlier sowing dates for maize (early-March) and cotton (late-April), delays were of up to 1.8 and 1.9, respectively. The planting date for a given crop should be scheduled after considering some factors of cropping systems such as the species and its thermal demand (e.g. from sowing to emergence), seasonal and current weather conditions, and the crop residue accumulation and its management.

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2.14.P

DO RAINFALL HARVESTING PITS AS RESTORATION TECHNIQUE FAVOUR INFILTRATION IN ARID ECOSYSTEMS?

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INTRODUCTION

Water harvesting increases water availability for plants during dry periods and decreases plant mortality. The annual rainfall in Almería (SE Spain) is about 235 mm and the amount of evaporation largely exceeds the amount of rainfall. Water scarcity demands the maximum use of every drop of rainfall particularly in arid and semi-arid regions where collected soil-water is subjected to high evaporation losses before plants can use it. Water harvesting can be achieved using pre-treated catchment and microcatchment areas to increase the efficiency of runoff and maximize the amount of collected rainfall (Oweis and Taimeh, 1996; Fleskens et al. 2005). Some of these methods, however, require continuous maintenance over the years, and are not suitable for mining sites with mild slopes. The efficiency of traditional techniques for water harvesting is also limited by soil infiltration and climatic conditions (Renner and Fraiser, 1995).

In mining degraded areas on calcaric sandstones and marls in SE Spain, soils are predominantly silty and have very low infiltration rates (Luna et al., 2016). In such cases, the depth of water infiltration is reduced and water may remain in the upper soil horizon. Under high winds and evaporation, collected water is lost to the atmosphere rapidly and is unavailable for plants (Abu-Zreig and Tamimi, 2011).

In a calcareous quarry in Almería (SE Spain), we tested a relatively new technique of water harvesting consisting in infiltration pits drilled into the soil, and filled with the same three components to facilitate water and nutrient resources to planted native vegetation. The objective of the present study is to compare the effect of different depths and densities of pits on soil moisture and vegetation under semiarid-arid climate.

METHODS

The experimental area is a calcareous quarry located in the Gádor Mountains (Almería, SE Spain, 36º55'20''N, 2°30'29''W), in the boundary between arid and semiarid Mediterranean climate. The mean annual rainfall of the area, 242 mm at the closest meteorological station (Alhama de Almería, AL003, Junta de Andalucía) has a high interannual variability and very dry summers. Via an in-situ automatic meteorological station the annual precipitation during the study period (2015) was 135 mm, a particularly dry year. Mean annual temperature is 17.6°C, mean maximum temperature for the hottest month (August) is 31°C, and the mean minimum temperature of the coldest month (January) is 8°C. Mean potential evapotranspiration is 1225 mm year-1 (Lázaro et al., 2004).

Seven plots of approximately 130 m^2 (A, B, C, D, E, F, G) where used for the experimental design, 6 of them with pits at different depths and separation, and one control (G) without pits (Fig. 1). A, B and C plots had a density of 2500 pits ha^{-1} (pits 4 m apart), and plots D, E, F a density of 625 pits ha^{-1} (pits 2 m apart). Plots A and D had 1 m deep pits, B and E, 1.5 m deep pits, and C and F, 2 m deep pits. All infiltration pits had 10 cm diameter. Each pit was filled in its lower third by zeolite, middle third by compost made from plant residues from greenhouses and upper third by gravel (Fig. 2), with the purpose to be a reservoir for water and nutrients. Independently of the pits and their separation, three native plant species (*Pinus halepensis*, *Rosmarinus officinalis* and *Macrochloa tenacissima*) were planted with a planting pattern of 1 x 1 m (Figs. 1 and 3). Soil moisture was automatically recorded in every plot, every hour at a depth of 0.15 m. Plant survival and cover were measured 12 months after plantation.

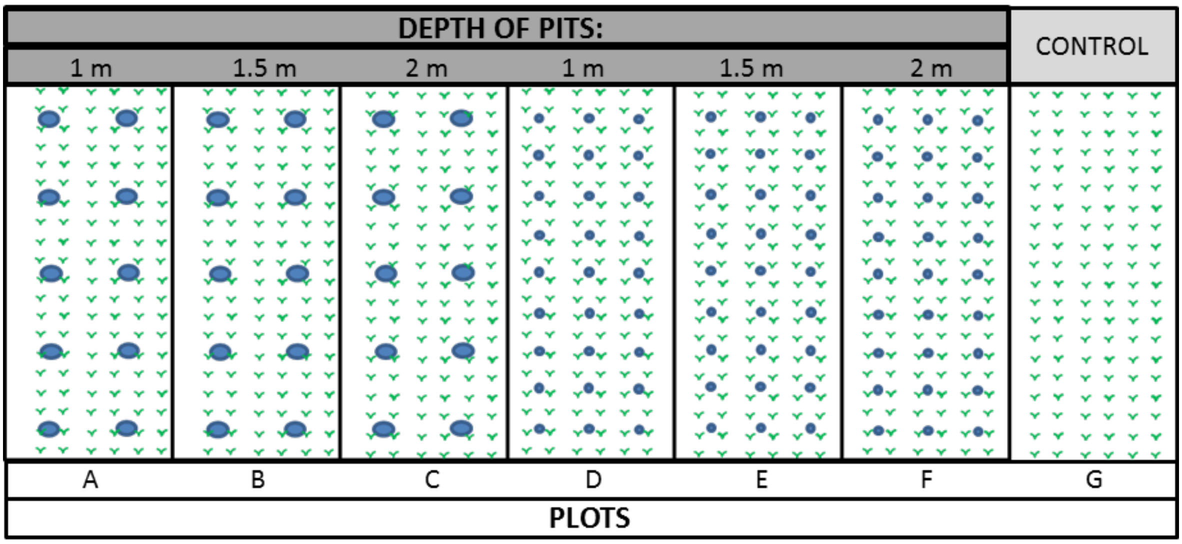

Fig. 1- Scheme of the experimental area (six plots with infiltration pits and one control plot without). The blue dots are the 10 cm diameter pits which depths are indicated; large dots (⬤) represent pits separated 4 x 4 m and small ones (•), 2 x 2 m. The green marks (✓) correspond to plants which plantation pattern, 1 x 1 m, is the same in all plots. The minimum distance of plants to infiltration pits is 50 cm. The control plot at the right has no infiltration pits.

Fig. 2. Scheme of a pit. Fig. 3. Experimental area with pits and vegetation.

RESULTS

Preliminary results did not show significant differences between treatments and control (without pits), probably because rainfall during the tested period was very scarce. Figure 4 shows soil moisture evolution during the 12 months of the experiment and Figure 5 the averages of every plot for the whole period: no significant differences are observed between treatments. Plant survival and cover were also similar under the different treatments and control. Probably, the effect of pits on vegetation might be observed after a few years when the plant root system becomes larger.

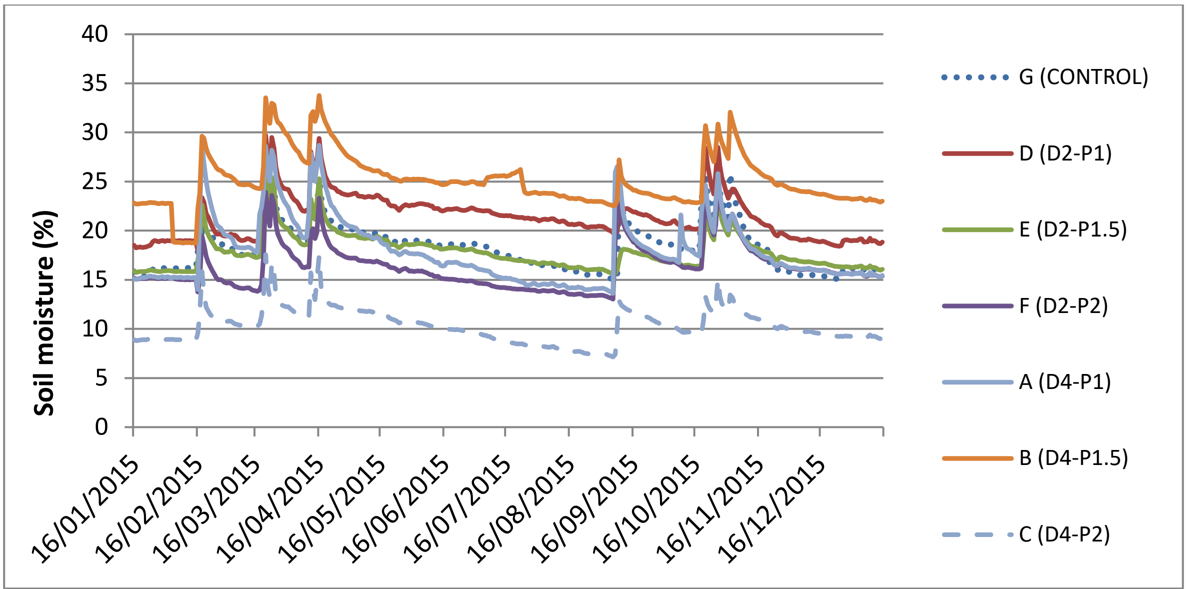

Fig 4. Evolution of the soil moisture during the tested period where the control without pits has intermediate values with regards the plots with infiltration pits.

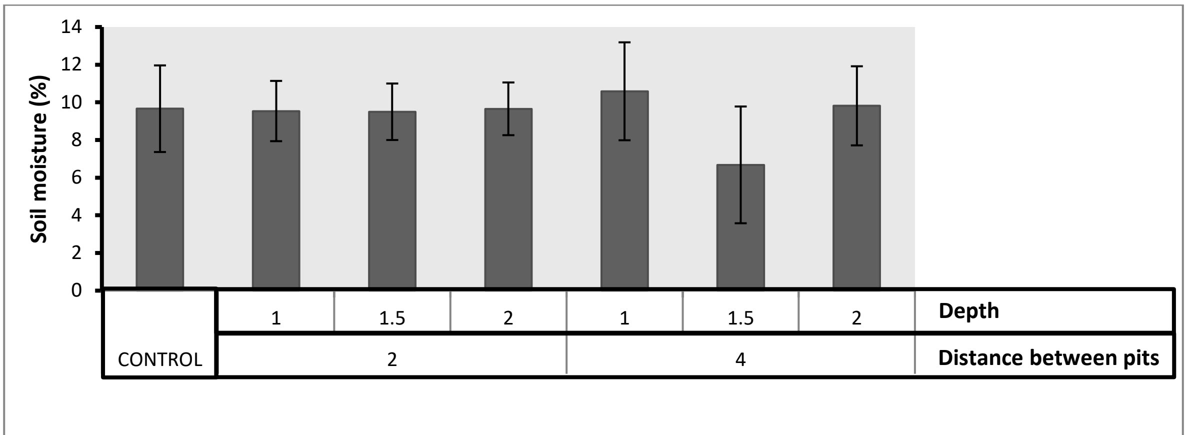

Fig. 5. Soil moisture at 3 depths (1, 1.5 and 2 m) and at 2 and 4 m, and a control plots without pits.

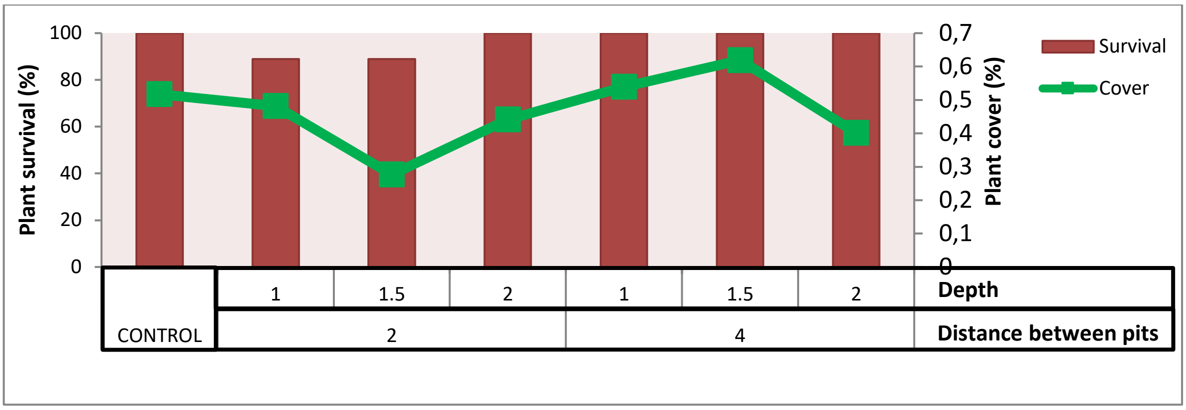

Fig. 6. Plant survival and cover in the different treatments.

CONCLUSIONS

During the first year, we did not find significant results of different depths and densities of pits on soil moisture and vegetation. Probably, the effect of pits on vegetation may be observed after a few years when the root system of the plants is larger. Long-term studies are required to better assess effects on the establishment of plants.

ACKNOWLEDGEMENTS

The Cement Factory of Gádor (Almería, Spain) and the Andalusian Regional Government provided essential financial aid for the “Experimental soil restoration in calcareous quarries in the Gador Range” (RNM-5887 and ARESASA Projects). The first author also acknowledges a Ph.D. research grant from the Andalusian Regional Government (2012-2016).

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2.15.P

ASSESSING THE POTENTIAL OF NO TILLAGE FARMING ACROSS EUROPE

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INTRODUCTION

Agriculture is facing the challenges of feeding a growing population under a changing climate and the need to reduce environmental impacts. This tensions demand sustainable approaches to farming, considering not only yield and productivity, but also environmental and social welfare. For that matter, conventional tillage (CT), has shown its agronomic effectiveness in reducing weeds and obtaining a uniform bed for root growth, but with high environmental costs such as erosion and reduction of infiltration leading to increasing floods. These consequences of tillage make the practice unsustainable, and the development of alternatives in conservation agriculture such as no tillage (NT). In this practice, 'crops are sown without any prior loosening of the soil by cultivation other than the very shallow disturbance (<5 cm) which may arise by the passage of the drill coulters and after which usually 30-100% of the Surface remains covered with plant residues' (Soane *et al.*, 2012). However, when considering alternative practices, these are expected to be suitable for the local land capability and for the wider context of the farmer´s livelihood.

Despite environmental benefits of no tillage, its spread in the World is geographically diverse. While it has been adopted in the USA in 25.5% of its arable land (Derpsch *et al.*, 2010), in Europe the percentage is only 3% (Eurostat, 2010). Reasoning about adoption rates in Europe arise questions about the existence of unbeatable constraints or latent possibilities for more sustainable agricultural practices.

This PhD (2016-2020) will assess the potential of adopting direct sowing in different regions in Europe, taking an interdisciplinary approach to study which factors prevail when deciding about no tillage adoption. It will take into account two traditionally diverse research perspectives, namely land use suitability from the agri-environmental disciplines and the socio-economic and socio-cultural analysis in its broadest meaning, including farmers' values, attitudes and identities. Thus, it will bridge the scientists' and farmers' perception about the soil, by measuring soil quality indicators at farms and compare the results with farmers' expectations.

METHODS

To give coherence to the research, a theoretical approach has been chosen to be used as a lens to understand no tillage and conventional tillage practices, and all the factors involved. The selected approach is based on co-creation of innovation with an actor-network theory (ANT) perspective, following (Schneider *et al.*, 2012) application of ANT to no tillage adoption in Switzerland. It considers farmers as active participants in innovation, valuing farmers' site-specific knowledge, necessary to adapt scientific knowledge to local conditions (Burgess, Clark and Harrison, 2000).

ANT was developed in the early 1980's by Michel Callon, Bruno Latour and John Law (Gray and Gibson, 2013) bridging together the social with their material environment. In accordance to ANT, non-human actors also have agency, understood as the ability to intervene in the world, connect things, and make differences (Schneider *et al.*, 2012). Therefore, because of the refusal of the dualism nature-society, as well as the refusal of the dualism science-culture, local-global and expert-lay knowledge, ANT is a powerful tool for environmental questions (Burgess, Clark and Harrison, 2000). At the same time, it makes it possible to study an agricultural network and identify all the involved actors and their relations.

Geographical variability will be included selecting case studies, based on the identification, through GIS, of biogeographical and socio-cultural diverse regions. The nature of the data sets for the GIS analysis are socio-environmental potentially relevant variables such as no tillage adoption rates, biogeographic regions, soil classification, farm size, managers gender, age and educational level.

In each region, physical and social factors from conventional and no tillage farms will be compared. In the pilot study, preliminary information will be collected through participant observation, visiting farms and attending farmers' workshops, as well as from focus groups. This will be used to refine the selection of topics to discuss during semi-structured interviews with farmers in each geographical location (5 conventional, 5 no tillage). The aims of these interviews are, on one hand, to identify the actors (human and non-human) which are limiting or enabling no tillage adoption; and on the other hand, to understand farmers' expectations about the effects of their chosen tillage management practices on their soils' quality. These expectations will be compared to scientific measurements of soil quality, whose results will be discussed as well with the farmers.

Farm measurements will include soil physical, chemical and biological indicators to assess soil quality. Soil physical properties will be assessed on-site with the Visual Evaluation of soil Structure (VESS) method (Askari, Cui and Holden, 2013). Water stable aggregates will be assessed off-site, at the laboratory with the wet sieving procedure (Elliott and Cambardella, 1991). Penetration resistance will be measured with a hand penetrometer at field. Bulk density will be measured through the mass – volume relationship from cylinder samples taken in field. Particle size distribution (soil texture) will be measured (method to be confirmed). Soil biological quality will be assessed through earthworm abundance and diversity analysis (method to be confirmed). Finally, soil chemical quality will be assessed measuring physical fractions of soil organic matter, named, total organic carbon by loss-on-ignition and following the fractionation method of (Cambardella and Elliott, 1993), dissolved organic carbon, free particulate organic matter and mineral associated organic matter. Available phosphorus and available nitrogen will also be analysed, as well as pH and electric conductivity.

This research work will be accomplished during the next three years, the following section are the results of preliminary data collected in the UK during autumn (soil sampling) and winter (participant observation) 2016.

PRELIMINARY DATA COLLECTION

Soil samples were collected during October and November 2016 in three cereal farms in UK (A: 53.74303, -0.92685; B: 52.67414, -1.87926 and C: 53.55283, -1.18400) with soils classified as freely draining slightly acid sandy soils, freely draining slightly acid loamy soils and freely draining lime-rich loamy soils, respectively. In each location NT and CT fields were sampled. NT had been used for 1, 6 and 8 years respectively. In each location five pits were opened in no tillage fields and three in tillage

fields. In each pit soil samples were taken in layers of 10 cm depth until 60 cm. Additional cylinder samples to measure bulk density were taken in the same 10 cm layers, until 30 cm depth. VESS analysis were performed at each pit. Nitrates, ammonium, ortophosphates and potassium were analysed for the first 50 cm with an Ion Cromatography analyser. Wet aggregate stability was analysed using wet sieving for samples until 40 cm depth. Loss of ignition was used to measure soil organic matter and soil organic carbon (SOC) was calculated for all samples until 60 cm depth.
Attendance to farmers' workshops in Scotland during February and March permitted to collect 15 questionnaires and general interests and attitudes.

PRELIMINARY RESULTS

No significant differences were found between NT and CT VESS scores. However, during visual inspection, stratification was obvious in the first cm of topsoil in NT soil blocks whereas CT were homogeneous. No significant differences were found between NT and CT in BD. When present, significant differences in the smallest category (0.065 mm) of water stable aggregates were due to higher values in CT fields. Aggregates in the categories 0.065 – 0.250 mm and 0.250 – 1.0 mm were dominant in all samples, with the exception of location C. No significant differences in MWD neither for TOC between NT and CT at any depth. No increased stratification of TOC at surfaces layers as discussed by (Hernanz *et al.*, 2002; Blanco-Canqui and Lal, 2007) has been found in NT (example from location A in *Figure 1*). When analysing SOC per aggreagate size, the trend in all farms, for all depths and for both tillage management was a significantly higher percentage in 0-0.063 mm falling to minimum in the categories 0.065 – 0.250 mm and 0.250 – 1.0 mm and increasing again in bigger aggregates. Higher concentrations in fractions below 0.063 mm corroborate higher C stabilisation in free microaggregates (Six *et al.*, 2002). Significant differences between NT and CT were found only in location A, only at 20-30 cm depth and in the category 0.25 – 1.0 mm where SOC% is minimal; here NT had higher SOC value.
For nitrates, significant differences were found in Location 1 for at depths 10-20 cm and 20-30 cm with higher values for CT. Potassium was higher at surface 0-10 cm for NT in Location 1 and 2. Differences in ortophosphates were found in Location 2 from surface to 30cm depth.

During attendance at the farmers' workshops, it was possible to observe an overall interest about

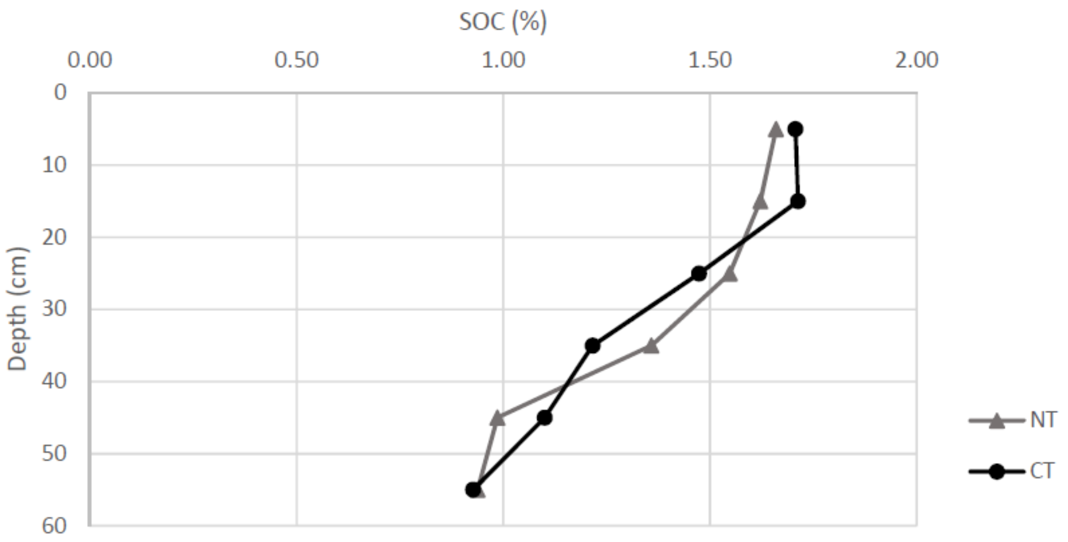

Figure 1. SOC% with depth in Location A.

soil health (workshops' topic). However, at the first event one farmer commented "I don't understand the soil and I never will" and several other farmers' agreed. This attitude contrasted with the trust in soil scientists criteria, exposed during presentations. Farmers partnered with regional institutions to be examples of good practices of soil conservation, and also with other research programmes. Moreover, farmers travelled to meet conservation specialists in USA and learn from them. These has a connotation of active roles in innovation. Other personal interests of farmers were about machinery collection and new technologies such as drones for precision agriculture. Towards NT the general attitude was that it is not suitable for Scotland, although the reasons vary.

Questionnaire responses about reasons for non-adoption of NT in Scotland include repeatedly weed control, price and availability of equipment, inexperience and lack of confidence. Other responses were slug risk, yield loss, good seed bed preparation with ploughing, weather and related compaction due to wet season and short growing season for cover crops, no suitable crops, no suitable soils or lack of knowledge about suitability, organic matter incorporation and overall reliability of ploughing.

Reasons for adoption include better soils, reduce inputs, costs cover and environmental benefits.

CONCLUSIONS

No significant differences were between NT and CT fields, neither improving soil quality neither deteriorating it. Further research will compare this scenario with other climatic regions and soil types. Farmers' reasoning about NT adoption will be analysed in depth, as well as their expectations and how they respond to soil's health changes.

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2.16.P

PROMOTION OF BETTER MANAGEMENT PRACTICES TO CONSERVE MOISTURE AND NUTRIENTS IN THE CONTAINER PRODUCTION OF NURSERY CROPS IN SE USA

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INTRODUCTION

Better management practices in soilless cultures of container production could pave the way for an efficient utilization of resources including moisture and nutrients and higher profit margins. Eco-friendly production and management practices will lead to decreased use of chemical inputs (fertilizers and pesticides) and reduce NO_3 and PO_4 leachates.

Currently, pine bark is widely used as soilless substrate in container production (Warren et al., 2009). Pine bark is porous, inexpensive, stable, and free from pathogens and has low water-holding capacity and low bulk density (Bilderback et al., 2005). It also has low cation exchange capacity, 40-75 meq/L and low anion exchange capacity, close to zero value meq/L. However, pine bark requires frequent and extended periods of watering either using overhead sprinklers or drip systems to accelerate growth and prevent moisture stress. Excessive use of water soluble and slow release fertilizers used in these soilless container production results in significant runoff and leaching of nutrients such as nitrate (NO_3) and phosphate (PO_4). Most of the commercial nurseries grow several different species and varieties of ornamentals and fruit trees at different stages of growth and development in containers of varying volumes. All these varying types of plants and containers in the entire field receive the same formulation of fertilizers; source, rate, timing and application method. This could result in poor nutrient uptake, non-uniform growth, biotic and abiotic stress and ultimately crop and monetary losses. Also, high alkaline local water contributes to nitrification of NH_4 to NO_3 resulting in micronutrient deficiencies and increased pH, which further reduces the plant growth.

Substrate components for container production

Container grown plants require significantly more water supply than field grown plants even during wet season because the root growth and development is restricted to the container volume irrespective of the shoot growth. Additional factors of concern include wind direction in the case of overhead sprinkler system and downward vertical movement of water in porous substrates such as pine bark. Appropriate irrigation systems and field layout of different size and shape plants would help deliver water precisely rather than over or under irrigation. Currently, there are no standard protocols to deliver water to plants in container production systems. At times, water has to be supplied twice/thrice or even four times per day during vigorous plant growth period due to low water holding capacity and high porosity nature of substrates used in container production. In most of the container production systems, a significant percentage ($\leq$ 80%) of irrigated water using the sprinkler system is lost (Fig 1). It is partly due to improper organization and management of container plants by not taking into consideration of the form of the plant canopy, size, leaf orientation, pot size, type of plant (deciduous versus evergreen), growth habit, etc. in spacing and arranging plants.

Figure 1 Inefficient water management in soilless pine bark substrate affecting plant growth. Root growth occupies <40% of the container volume.

Nitrogen form and Nitrification

In container production, the total elimination of soil from the soilless substrate mix has been causing loss of water and nutrients. The addition of 5% -10% of clay in potting media could provide solution to leachate runoff and reduce the excessive water use. During liming ($CaCO_3$ or $CaMg(CO_3)_2$) operation, the addition of weak bases and their solubility is the function of substrate pH adjustments. The increase in pH could have significant effect on NO_3 leaching, because nitrification process is pH dependent and liming has significant influence on pH dynamics (Agner, 2003).

NH_4-Nitrogen form → liming → ↑ pH → Nitrification →NO_3-Nitrogen form (leached due low AEC)

Liming material source, particle size and rate of application is critical in raising pine bark substrate pH. It could contribute to excessive above normal pH resulting in NO_3 leaching plus micronutrient deficiency (Sharma et al. 2008). Normally, controlled-release fertilizers are used in container production and it is formulated with urea as N source, which readily hydrolyzes to NH_4. The risk of loss of NH_4 to NO_3 or NH_3 depends on the pH factor. The other contributing factor in raising the pH is the alkaline ground water used in irrigation. Under this condition, a) it is important to monitor and manage pH, b) substitute gypsum ($CaSO_4$) for lime that does not raise pH but provide Ca and S nutrients or c) acidify the alkaline water to minimize nutrient loss (See Fig 2).

MATERIALS AND METHODS

To improve the container grown plant's uptake of N efficiency is by organizing and arranging the container grown plants in the field by similar preference for inorganic N i.e. NH_4 or NO_3-N forms/source over the other (Brian et al. 1988), adjusting the pH accordingly, water requirement (High/ Moderate/low), by family, genus, or species association with symbiotic microbes (Rhizobium/Mycorrhizae), (See Table 1) and slow growing species versus vigorous growing species. The availability of water is limited in container grown plants compared to field grown plants. The percentage of available water is less than 22% of container capacity and it declines overtime with the decomposition of substrate. Unwanted nutrient and water loss is of environmental concern for the nursery industry

Water conservation, capturing runoff, and recycling water will promote and strengthen the sustainability of nursery industry in the Southeastern United States. Dynamic research and outreach program by reaching out to the nursery growers on the impact and strategies of minimizing NO_3 and PO_4 leachate run-off, updates on the cultural practices specific to each species could reduce inputs and conserve water.

This outreach study-Conservation Innovation Grant-involves comparing several substrate and fertilizer formulations for conserving moisture and nutrients in container production based on existing research data and revised set of nursery-specific and site specific findings to minimize/ eliminate NO_3 and PO_4 leachate runoff in nursery operations and management. A set of treatments have been established at the Agricultural Research and Education Center, Tennessee State University for training purposes.

CONCLUSIONS

The efficient container production technologies demonstration will have a direct impact on the nursery crop production both in terms of moisture and nutrient conservation and profitability in Tennessee and other SE states of the USA.

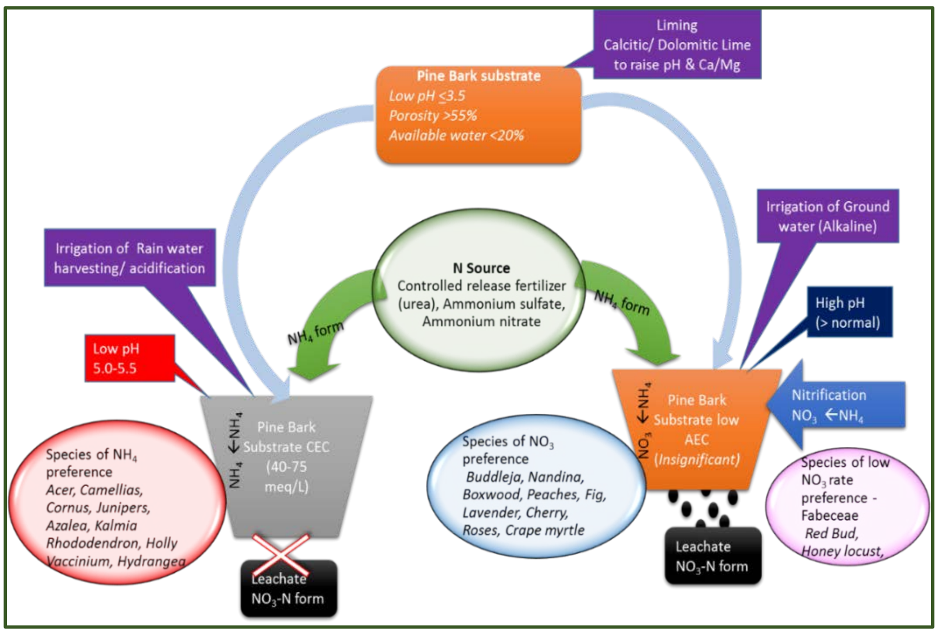

Figure 2. Flow chart of nitrification and NO_3 leachate runoff in pine bark soilless substrate container production due to poor liming rate, pH and alkalinity factors in species preference for inorganic N forms.

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Table 1. Species specific requirement of N form, pH, Ca source and Irrigation water quality.

Selected Species	Nitrogen form/ source	Preferred pH acidic/basic	Liming source $CaCO_3$ or $CaMg(CO_3)_2$	Irrigation water quality
Species preference for NH_4-N. 1. Acer spps (*Acer rubrum, Acer palmatum),* 2. Blueberry (*Vaccinium corymbosum*) 3. Camellias (*C. japonica*, *C. sasasqua*), 4. Dogwood (*Cornus alba*), 5. Hydragea (*Hydrangea paniculata),* 6. Junipers *(Juniperus chinensis),* 7. Kalmia *(Kalmia latifolia),* 8.*Rhododendron spp.*	$\geq$ 75% NH_4 or Urea & $\leq$ 25% NO_3	5.0 – 5.5	$CaSO_4$	Acidify / rain water
Species preference for NO_3- N. *1. Butterfly bush (Buddleia davidii),* *2. Nandina (Nandina domestica),* *3. Boxwood (Buxus sempervirens),* *4.Peaches (Prunus persica)* *5. Fig (Ficus carica) 6. Crape Myrtle (Lagerstroemia Hybrid*)	$\geq$ 75% NO_3 & < 25% NH_4 or Urea	5.8 – 6.8	Calcitic ($CaCO_3$) or Dolomitic lime $CaMg(CO_3)_2$	Neutral/Alkaline water or rain water. Do not acidify the water.
Species preference for low N for symbiotic microbes (Rhizobium) 1. Redbud (*Cercis canadensis),* 2. Honeylocust *(Gleditsia triacanthos),*	75% NO_3 25% NH_4/Urea Mix 5 -10% of soil to soilless substrate for rhizobium/ Mycorrhizae	5.8 – 6.8	Calcitic ($CaCO_3$) or Dolomitic lime $CaMg(CO_3)_2$	Neutral/Alkaline water or rain water. Do not acidify the water.

2.18.P

WATER EROSION AND SOIL CONSERVATION PRACTICES IN "EL ARENAL" WATERSHED, MEXICO

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INTRODUCTION

The productive capacity of ecosystems in Mexico is being lost considerably due to overuse of resources. The soil and water management programs with watershed approach are essential for the sustainable management of resources (Loredo *et al.,* 2007). The objectives of this study were to characterize the small watershed "El Arenal" resources, assess the potential and actual erosion risk, propose and implement a program of management and soil conservation. This small watershed is located in an arid area of the municipality of Ahualulco in the state of San Luis Potosi, Mexico, has 254 inhabitants and a surface of 5864 ha.

METHODS

The water erosion risk was estimated, using a GIS and the prediction model: USLE (Wischmeier y Smith, 1978). Five categories were considered: no apparent erosion risk (soil loss less than 2.2 t ha^{-1} yr^{-1}), slight erosion risk (2.2-10 t ha^{-1}), moderate (10-50 t ha^{-1}), high (50-200 t ha^{-1}) and very high (> 200 t ha^{-1}). A database was performed; the available technology for resource management was analyzed and management practices more appropriate for conservation and restoration land were identified. Integrating efforts of several institutions currently "El Arenal" has actions of soil and water management and conservation in 225 ha, which include rangeland rehabilitation, reseeding with grasses and reforestation. Since 2004, a working group was organize with "El Arenal" members, which received support from the federal and state governments to perform these actions, under the State Program of micro-watershed, to achieve integrated and sustainable use of natural resources.

RESULTS

The mean annual temperature in "El Arenal" is 19.3°C; mean annual precipitation: 370 mm. The altitude varies from 1869 to 2015 meters above sea level (masl). The main types of soil are leptosol: 75%; leptosol in association with calcareous fluvisol: 2.3%; and, fluvisol 23.7% (Figure 1).

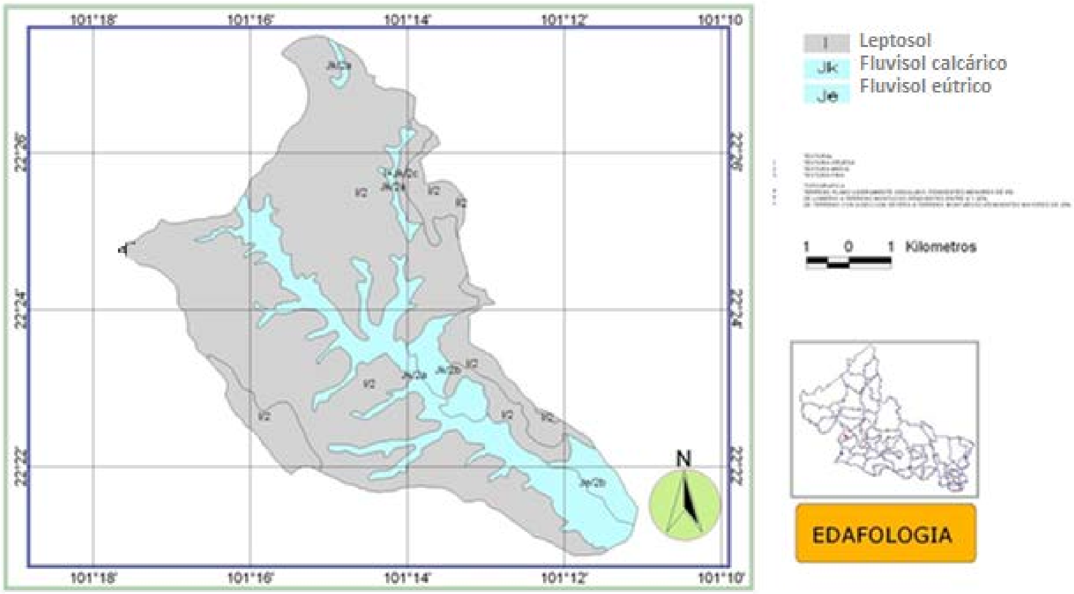

Figure 1. Soil types of the "Arena" micro-watershed. Digitalization of the letter F14A-73 soil esc 1:50 000 INEGI

The actual soil use presents the following distribution: 89 % are special associations of vegetation (evergreen subdesert shrublands associated with grassland and thorny scrubland) and 11% are agricultural use land. The principal use land is the livestock (Figure 2).

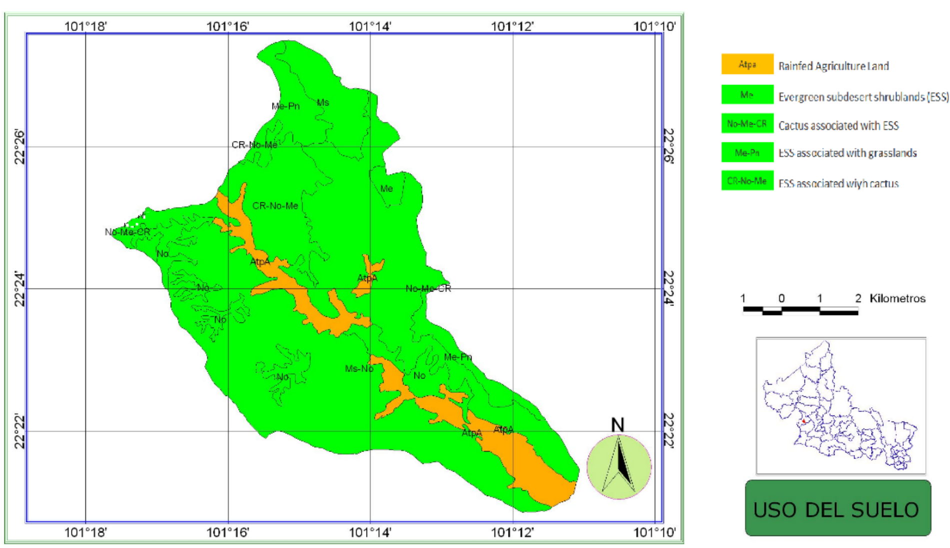

Figure 2. Distribution of vegetation type and current land use in the "Arenal" micro-watershed watershed. Digitalization of the letter F14A-73 vegetation esc 1:50 000 INEGI.

Classes of potential land use are distributed as follows: Class VIII covers 77.58% of the area, the class IV/c 17.87%, the 4.55% is a Class VI/sc; the main restrictive factors are the climate, topography and the soil depth.

The surface of the watershed affected by slight erosion risk corresponds to 43%; 41% for a moderate erosion; 15% corresponds to high erosion and very high erosion covers 1% of watershed surface (Figure 3).

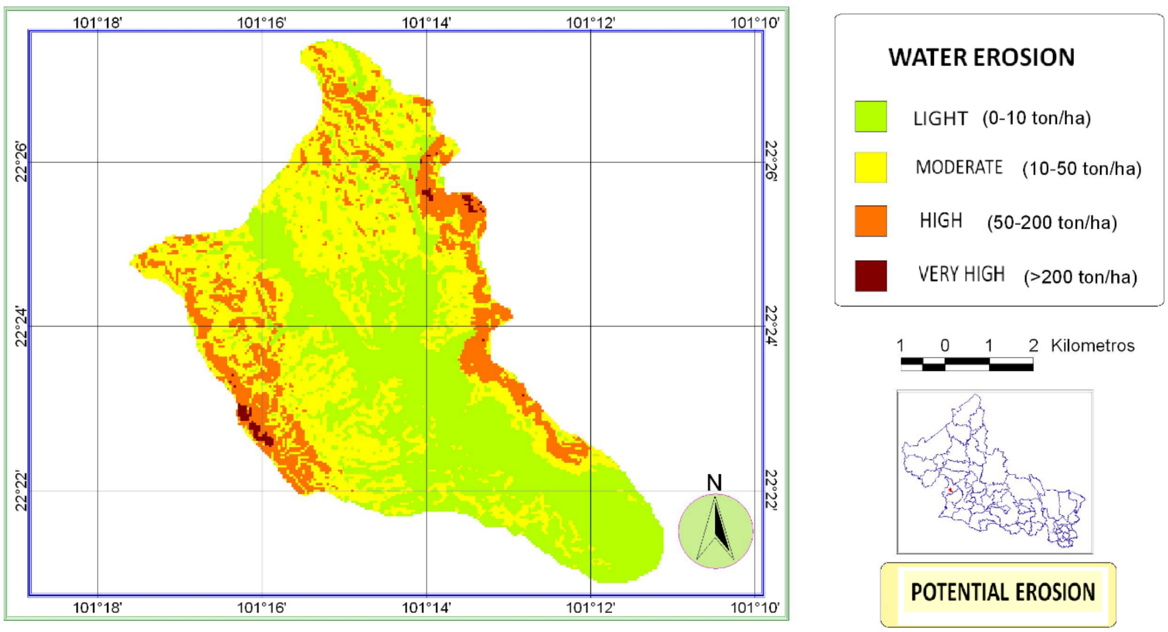

Figure 3. Potential risk of water erosion on the surface of the "El Arenal" micro-watershed.

In Mexico, the rangelands are the ecosystems most deteriorated due overgrazing has damaged 60 million hectares. Secondly, damage are forest areas and thirdly rainfed agriculture, which has identified 21 million hectares with problems of water and wind erosion (Ortiz *et al.,* 1994). According to the National Commission for the Determination of the Coefficients of Rangeland Use (COTECOCA, 1974), the micro watershed "The Arenal" has a production of annual forage of 425 kg MS/ha/year and 20.64 ha are required for one United Animal unit (cow of 450 kg with its calf). With the active participation of producers was possible to define priority actions for the management of water, soil and vegetation resources. The management practices (vegetative and mechanical) proposals were stoking control (respecting the 20 hectares per animal unit), rangeland revegetation, reforestation, to increase the vegetal cover; settles trench, sediment control structures for gully control erosion. With the application of these practices, the grade of actual soil erosion will be reduced to allowable limits. This information was integrated into the Master Plan of the Watershed “El Arenal”, where there have been mechanical and coverage vegetal management with support from federal and state government practices.

CONCLUSIONS

The surfaces with potential water erosion risk and the expected erosion and mechanical cover management practices in the “El Arenal” micro-watershed were identified with the USLE model. With the active participation of producers was possible to define priority actions for the management of water, soil and vegetation resources. This information was integrated into the Master Plan of the watershed “El Arenal”, where there have been mechanical and coverage management with support from federal and state government practices. Lack assesses the impact of such actions and that this actions to be continued.

ACKNOWLEDGEMENTS

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2.19.P

UTILIZATION OF SEMI-DETAILED SOIL MAP FOR IRRIGATION PURPOSE AT A FARM SCALE: A CASE IN BURIRAM PROVINCE, THAILAND

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ABSTRACT
Due to the drought is threatening an agricultural production in Thailand, recently. The irrigation management is now playing an important role, term to irrigated agriculture. Therefore, this study is to determine the available water capacity (AWC) extracted from soil map and their implications on the irrigation management in Buriram province. The study focused on utilization a group of soil series map (1: 25,000), where the specific properties are contained, the term of irrigation purpose. The GIS tool was used to extract AWC map cover whole of the country to creating secondary data maps. The soil particle size classes were mapped and related the AWC data. Then CROPWAT simulation model was used to simulate the irrigation scheduling at a farm scale, based on soil and other required data. Sixty two group of soil series were examined the relationship between soil water content on soil and plant utilization to determine the schedule duration and the amount of water the plants. The data were put in the CROPWAT and mapping water plants utilize were created by GIS technique

The results showed that the fine silty particle has available water plant utilization at maximum average of 20.55%, while the sand particle size class has the minimum average of 4.27%, the relationship between soil texture and the amount of water that plants use both two cases showed a trend in the same direction. It was concluded that available water correlated to soil particle size.

For the irrigation scheduling and amount of water were conducted at -33 kPa in Buriram province. CropWat model was used to soils irrigation timing and among of water for a sugarcane Planting season in two site study, in Nang Rong and Nonsuwan districts. Two soil series was found and studied as Ko Rat series (Kt) and Kabin Buri (Kb). The best result show that Kt soils were 11 times with 127.62 cubic meters per hectare, while in Kb soils were 13 times/crop of 135.07 cubic meters per hectare are recommended.

KEYWORDS: Soil map, Available water capacity, GIS, CROPWAT, Irrigation scheduling

1. INTRODUCTION

In Thailand, soils are an important basis of Thailand's economy, so that soil survey, primarily for agriculture, has a long history in the country. Soil survey and mapping projects were initiated about 78 years ago. By Land Development Department (LDD), Ministry of Agriculture & Cooperation, the project has been executed for almost 50 years. Soil resources have been considered as resources for agricultural production. With the increasing pressure on land and new demands of soil information for irrigation management, LDD is facing new challenges due to drought and flood hazards.

Within the 6th national economy and social development plan (1987 – 1991), soil survey & classification division initiated semi-detailed scale soil mapping project (provincial soil map updating project/land use planning for economic agriculture re-planting project) for district and sub-district level using. Soil map in each province was produce to a group of soil series at a scale of 1:25,000. The produced maps used group of soil series system (62 soil units system) representing in soil map to simplify information for classification (Office of Soil Resource Survey and Research, 2007). Distinct soil characteristics and properties for economic crops, table of soil suitability rating, limitation, and recommendation for soil management were also included in the reports. Semi-detailed survey in some districts were conducted and soil maps at 1:25,000 were published for district development planning, target to economic plant management.

In the current situation, it has been recognized that the drought hazard has occur as a worldwide problem also in Thailand. Since 2012, Thailand has faced to the drought area effecting to agricultural production. The AWC was descripted in whole country, where relation of the AWC was related with 12 soil texture classes, thus it can be applied directly in agricultural purpose. Sugarcane is the major economic plant in Thailand. In this study, the sugarcane was selected to determine the irrigation requirement for two soils in Buriram province. The irrigation management is required for agricultural production (Haomyamyen, 2009 and Nilapunt, 2007). Thus, Thai government considered to the water resource management, especially for agricultural purpose. Therefore, this study realizes that irrigation management must be determined and implemented at farm scale term to irrigation requirement and management.

2. METHODS

2.1 Study areas and soils

LDD has been released semi-detailed soil map at a scale of 1:25,000 containing 62 groups of soil series. Thus existing soil property such as available water capacity can be extracted for irrigation purpose. The study along recorded 2009 to 2012, location of two study areas for irrigation scheduling at farm scale in Buriram province were Area A: Hau Thanon subdistrict, Nang Rong, 48P 247274E 1628736N and Area B: Krok Kaew sub-district, Non Suwan district, 48P 295467E 1648912N, Burirum province.

2.2 Climate

Climate data was characterized by a tropical monsoon climate with three seasons. The mean monthly temperature varies between 36.30°C in April to 30.30°C in December and the average annual rainfall is 1,282.60 mm and effective rainfall is 921.30 mm The wet and dry period are shown in Figure 1.

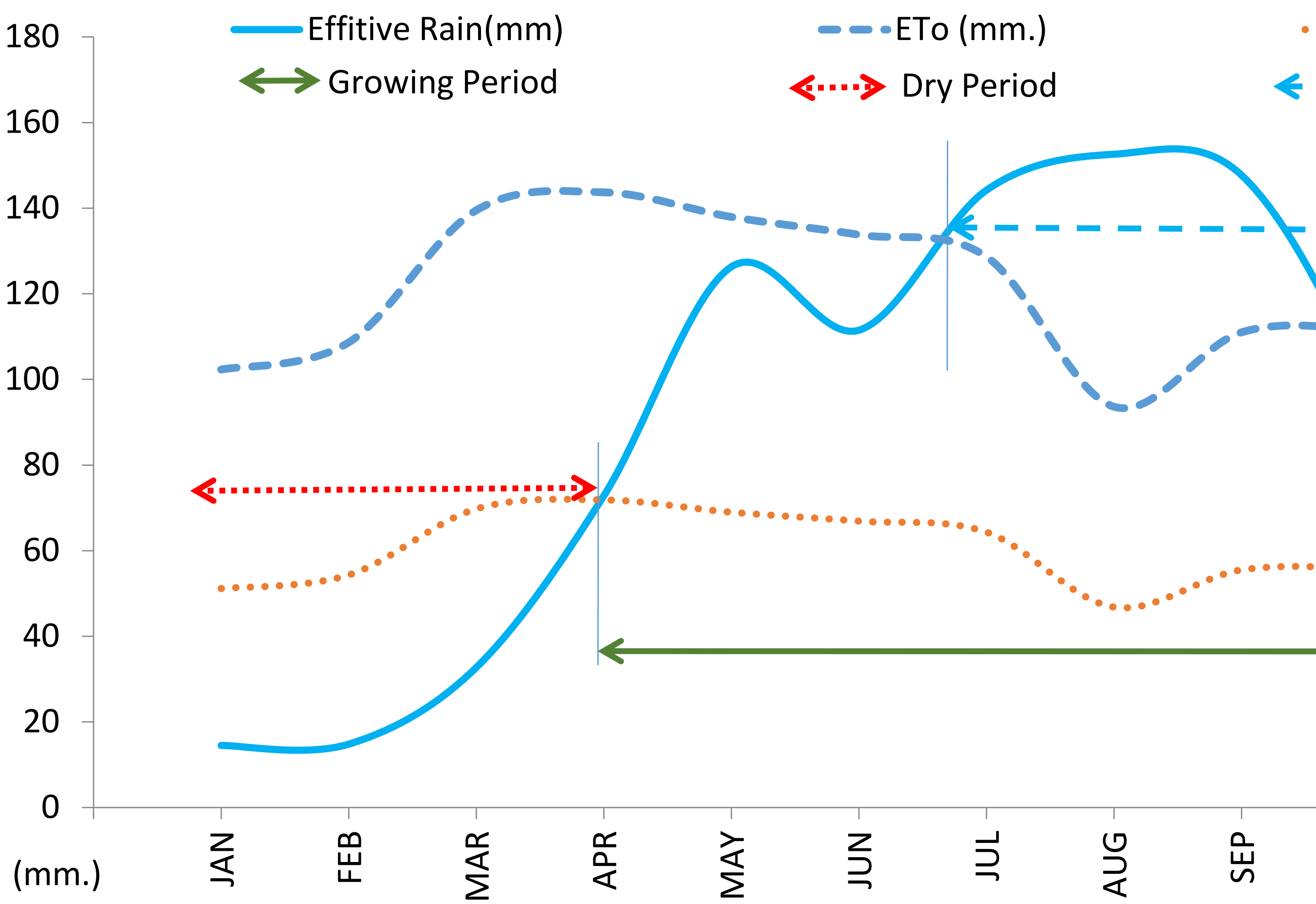

Figure 1: Monthly rainfall and water balance of Buriram province (2002-2011)

2.3 Method and data collection

2.3.1 The AWC map

A group of soil series map (1:25,000) was used to extract and mapped AWC, then correlation of the AWC and soil particle size were described with 62 groups.

2.3.2 Data collection (with a return period of 10 years)

Climate data was collected from Buriram meteorology station (The Thai Meteorological Department, 2014). In sugarcane plantation areas, two locations with different soil series (Kt and Kb) were selected for the irrigation scheduling using CropWat simulation model. Where some required soil properties such as texture (0-30, 30-50 and 50-100 cm), bulk density, Ksat, water retention (-.33 and -1500 kPa) were collected during a field work in September 2009 to September 2011.

2.3.3 CROPWAT for the irrigation management

CropWat 4.0 was used to simulate a crop water requirement (CWR) and crop irrigation requirement (Clarke, 1998). Two Planting date were compared the irrigation requirement (plot 1 =1 March and plot 2=20 May)

3. RESULTS AND DISCUSSION

3.1 Soil available water capacity

Figure 2-a shows AWC derived from group of soil series map using geographic information system (GIS). The AWC was mapped and correlated to soil particle size.

Figure 2-b, the results show that the AWC is correlated to particle size classes from high to low correlation as fine-silty, 20.55, very fine clayey (19.99), fine clayey (18.84), loamy skeletal (16.47), clayey skeletal (16.22), fine loamy (15.20), coarse loamy (9.79) and sandy (4.27), respectively.

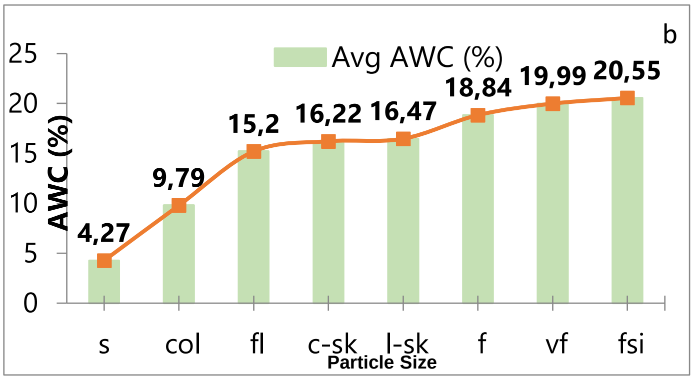

Figure 2: a) map of AWC and b) correlation of Average AWC and soil particle size classes

3.3 Reference crop evapotranspiration (ETo)

The results obtained from the 10 year climatic data was used in the CROPWAT 4.0 to determine the reference crop ETo for the study area varied from 3.02 mm/day in August to 4.79 mm/day in April. The results show that ETo was lowest during the rainy season to highest during the summer season.

3.4 Irrigation requirement in area *A* and area *B*

A soil in area *A*, Korat series (Kt), deep soils, are predominantly fine-loamy textured sandy loam textured ranging from sandy loam to sandy clay loam in the surface and sandy clay loam in the sub surface. The available water capacity and saturated hydraulic conductivity, which soil permeability is moderate, moderate to high available water capacity (7.5%) and crop coefficient (Kc) is 455.29 mm/day**.** The area *A* + plot 1 (Planting date on 1st March), irrigation requirement in area *A* with effective rainfall is 1277.10 mm, irrigation depth is 498.50 mm and number of irrigation are 11 times, where volume of irrigation is 127.62 m^3/ha (Table 1). The area *A* + plot 2 (Planting date on 20th May), irrigation requirement in area *A* with effective rainfall is 889.50 mm, irrigation depth is 604.40 mm and number of irrigation are 13 times, where volume of irrigation is 154.73 m^3/ha.

Table 1: Irrigation requirement under irrigated condition for sugarcane in area *A*, the1st of March (Plot 1)

No.	Date	TAM (mm)	RAM (mm)	Rain (mm) Total	Rain (mm) Efct.	Etc (mm)	SMD (mm)	Interv (Days)	Net Irr (mm)	Vol. Irr (m^3/ha)
Plant	1-Mar	37.9	22.7	0	0	1.7	20.6			
1	3-Mar	39.6	23.8	0	0	1.7	24	2	24	6.14
2	12-Jun	75.8	45.5	0	0	5.6	49	101	49	12.54
3	13-Nov	75.8	45.5	0	0	4.2	49.6	154	49.6	12.70
4	26-Nov	75.8	45.5	0	0	4.2	48.1	13	48.1	12.31
5	8-Dec	75.8	45.5	0	0	3.9	48.3	12	48.3	12.36
6	20-Dec	75.8	45.5	0	0	3.7	45.6	12	45.6	11.67

7	2-Jan	75.8	45.5	0	0	3.5	46.6	13	46.6	11.93
8	15-Jan	75.8	45.5	0	0	3.5	45.8	13	45.8	11.728
9	29-Jan	75.8	45.5	0	0	3.4	46.9	14	46.9	12.01
10	12-Feb	75.8	45.5	0	0	3.3	47.2	14	47.2	12.08
11	27-Feb	75.8	45.5	0	0	3.1	47.4	15	47.4	12.13
Harvest	28-Feb	75.8	45.5	3.3	0	3.1	3.1			
	Total			1,277.10	1,012.10	1,494.49			498.50	127.62

Remark: **RAM** = Readily Available Moisture = TAM * P [mm]. **SMD** = Soil Moisture Deficit [mm].
TAM = Total Available Moisture = (FC% - WP%)* Root Depth [mm]. **Net Irr.** = irrigation depth (mm).
Irr Interv = Irrigation interval (days) **Vol. Irr.** = Volume irrigation (m^3/ hectare)

A soil in area *B*, Kabin Buri series (Kb) are deep gravelly soils, predominantly clayey skeleton textured ranging from gravely clay loam to very gravely clay in the surface and gravelly clay in the sub surface. The available water capacity and saturated hydraulic conductivity, which soil permeability is moderate, moderate to high available water capacity (6 .7 %) and crop coefficient (Kc) is 4 7 .0 2 mm/day**.** The area *B* + plot 1 (Planting date on 1st March), irrigation requirement in area *B* with effective rainfall is 971.60 mm, irrigation depth is 527.60 mm and number of irrigation are 13 times, where volume of irrigation is 135.07 m^3/ha. The area *B* + plot 2 (Planting date on 20th May), irrigation requirement in area *B* with effective rainfall is 869.60 mm, irrigation depth is 622.50 mm and number of irrigation are 15 times, where volume of irrigation is 159.36 m^3/ha.

4. CONCLUSIONS

- The AWC map of Thailand can be extracted from a semi-detailed soil map at scale 1: 25,000, can be applicable in agricultural purpose.
- The AWC data and other covariable data are important for irrigation management at farm or sub-watershed.
- The AWC correlate to soil textures and particle size classes.
- The irrigation simulation results suggest that in both area A and B, the recommended method, plot 1 (planting on the 1st of March) is recommended for sugarcane irrigation scheduling.

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Session III: Hydrological processes on soil and water conservation

3.3.O

HISTORICAL TORRENTIAL FLOODS IN WATERSHEDS OF THE DRINA RIVER BASIN IN SERBIA

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INTRODUCTION

The aim of this work is to characterize the phenomenon of torrential floods in the Drina river basin according to dataset of the Inventory of torrential floods in Serbia (made by authors of this work) and analysis of recent torrential flood event. The Drina is an international river with river basin shared between territories of three countries – Serbia (western and south-western part of the territory), Montenegro and Bosnia and Herzegovina. Torrential floods have had severe consequences for the society in the past so that there is a need to define the character of this natural hazard in the Drina river basin, occupying considerable part of the territory of Serbia (ca. 20%).

The most recent and representative examples are torrential floods of three local torrents which attacked municipality of Krupanj and surrounding settlements on May 15th 2014, causing the death of two people and material damage estimated to more than 30 million €.

Summary of Methods

Torrent	• Štira (Drina)
Date of event	•07/05/1987
Affected locations and settlements	•Loznica
Number of casualties	•2
Damage description	•According to the information from the spot of happening, ca. 2,000 households and 10,000 ha of arrable land are flooded. Production in industry is stopped for a while.
Event description	•According to meteorological data, amount of rainfall for 24 hours is 90 l/m^2 - an extreme precipitation episode which make serious flood.
Source of informaton	•"Politika"

Figure 1. Minimum data for registering torrential flood event

Method for building the dataset of torrential flood events for the Drina river basin consists of five important steps: (1) defining spatial and time framework and needed parameters after the insights into data availability, (2) data collection, (3) organizing data, (4) data analysis, (5) data publishing, distribution and use (Petrovid et al., 2014). Collection of minimum data (Figure 1) facilitated analysis of temporal distribution of torrential flood events in the Drina river basin in order to show trends of frequency of their occurrence.

MAIN RESULTS

Historical dataset shows two main results: 1) When observing the historical torrential floods events in Drina river basin in Serbia distributed per months, the highest peaks are shown in May and June in summer season (which is in accordance with monthly distribution of rainfall), then in November (Figure 2). 2) When observing the torrential flood events in Drina river basin in Serbia distributed per year during observed period (1931-2014) (Figure 3), trend of linear increase of torrential flood events is notable, which is also shown in their distribution in three periods, showing that the last one has an annual average fourfold when compared with the first period (Table 1). The first data on torrential floods in the Drina river basin we found was from 1931, although we observed period from the beginning of the 20th century. For the time period of 84 years, there are 72 registered torrential flood events with 9 deaths as a consequence of this natural hazard.

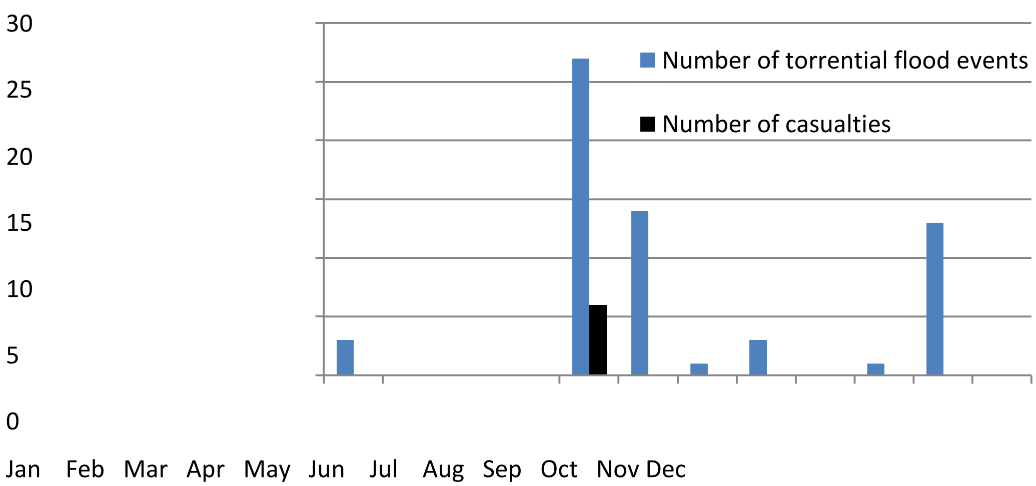

Figure 2. Historical torrential flood events in the Drina river basin distributed per month

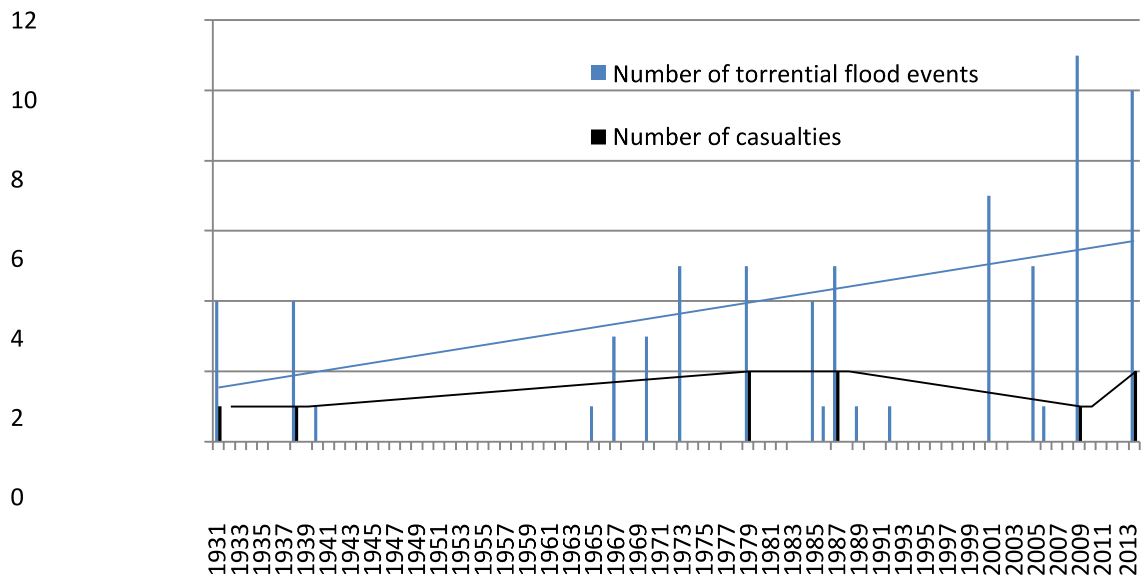

Figure 3. Historical torrential flood events in the Drina river basin during observed period

Period	Number of torrential flood events	Annual average	Number of casualties
1931-1960	9	0.3	2
1961-1990	28	0.93	4
1991-2014	35	1.46	3
Total	72		9

Table 1. Historical torrential flood events in the Drina river basin distributed per period

Short analysis of extreme torrential flood in Krupanj in 2014

After rainy April and first half of May 2014, especially western part of Serbia was a victim of a tree – day rainfall episode (May 14, 15 and 16). It was a perfect occasion for occurrence of torrential flood wave of three small torrents (Čađavica, Kržava and Brštica) that meet in Krupanj making the Likodra river which also has a torrential character. It was a real local calamity with an image of destroyed house and households, arable land covered with erosion sediment, broken bridges and flooded roads. As a consequence of long-lasting rainfalls, many landslides and rock-falls were activated and consequently made an additional material damage in traffic infrastructure and households, etc.

The fact is that the torrent channels in Krupanj were regulated by stone masonry. At first glance, there is a question how such a large discharges and disastrous flood wave occurred? Terrain observation after this event was included natural and especially anthropogenic factors. Natural conditions such as type of soil, shallowness and previous water saturation of soil and amount of precipitation were pretty beneficial for flood occurrence. In these three days, rainfall amount was 428 mm/m^2 or ca. half of total annual precipitation, so that already saturated soil and vegetation cover and torrent control works in Krupanj (designed for lower maximal discharges and only on a short stretch) were not helpful. Other reasons for catastrophic consequences for the local population are: the urban chaos due to houses and other household units built close to river channel, and on the other side a lack of biotechnical torrent control works (afforestation, terracing, grassing, check dams) in upper part of these three watersheds. These works could have make benefits for hydrological regimes of local torrents so that peaks of torrential flood could have been much lower and consequently, damages as well. According to study Kostadinov et al. (2014), return period of flood wave of the torrent Čađavica was 6000 years.

CONCLUSIONS

If there is an access to data about certain problem, it is easier to deal with it, and likewise the results of this work could be of essential importance for improving the system of prevention and prediction measures and instruments for mitigation of torrential flood consequences to an acceptable level in the Drina river basin. The main results regarding the dataset related to the Drina river basin are in accordance with results and conclusions on the level of the Inventory of torrential flood events in Serbia. The analyses presented here should be supported to have continuity for the forthcoming torrential flood events in order to have a detailed inventory for this type of natural hazards.

ACKNOWLEDGEMENTS

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3.4.O

OVERLAND FLOW HYDRAULICS IN NO-TILL SYSTEMS

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INTRODUCTION

No-till (NT) has been widely implemented to reduce erosion problems in Brazil, and currently occupies about 30 Mha (FEBRAPDP, 2013). However, changes in its implementation, such as forgoing crop rotation, removal of terraces, down-slope cultivation and soil compaction (Dindone et al., 2014), have led to a resurgence of erosion in NT areas in southern Brazil. On areas cultivated downhill and without adequate crop residues, shallow flow generated during severe storms can acquire sufficient hydraulic power to remove mulch, provoking rill erosion even under NT.

The concept of hydraulic power is used in fluvial mechanics to quantify sediment transport in rivers (Yang, 1972). Moore & Burch (1986), for example, using the Unit Stream Power (ω) parameter, obtained good results estimating sediment transport under shallow flow. Conceptually, ω (eq. 1) is defined as the time rate of potential energy per unit weight of water (Yang & Stall, 1974).

$$\omega = V * S \text{ [m s}^{-1}\text{]} \quad \text{(eq.1)}$$

Where: ω is the Unit Stream Power (m s^{-1}), *V* is the overland flow velocity (m s^{-1}) and *S* is the average slope (m m^{-1}). The Manning equation (eq. 2) and Darcy-Weisbach equation (eq. 3) are used to estimate flow velocity (V).

$$V = \frac{1}{n} * Rh^{\frac{2}{3}} * S^{\frac{1}{2}} \quad \text{(eq.2)} \qquad\qquad f = \frac{8*g*h*S}{V^2} \quad \text{(eq.3).}$$

Where: n is the Manning coefficient of roughness (dimensionless), Rh is the hydraulic radius (m), *f* is the Darcy- Weisbach coefficient of roughness (dimensionless), g is the gravity acceleration (m s^{-2}) and h is flow depth (m).

Unit Stream Power (ω) can also be applied to conservationist practices designed to prevent mulch failure and rill erosion in NT systems. For example, ω can be used to estimate the critical slope-length for terraces or to estimate the mulch rate (in a crop sequence) necessary to avoid mulch failure and consequent erosion (Foster et al 1982; Gilley et al. 1994). Before using the ω equation, flow velocity must be estimated (using eq. 2 or 3), where the friction factor is the most important hydraulic parameter (n-Manning or *f*-Darcy-Weisbach). These factors, however, are site-specific to different soil types and crop residues and are rarely found in the literature. Using data generated by Morais & Cogo (2001), this work aims to derive the friction factors n and *f* for soybeans-straw, cornstalk and black oats-straw under the NT system. In addition, this work explores the relationships between ω and hydraulic parameters, such as Reynold's number, that characterize shallow flow.

MATERIAL AND METHODS

Experimental Procedures

In order to derive the friction parameters *f*-Darcy-Weisbach and n-Manning, the unit flow rate (q) and flow velocity (V) obtained by Morais and Cogo (2001) were used. These authors measured flow velocity using dye (methylene blue) over the course of 6 m starting at the top of the upper plot (3.5 m x 11 m) to investigate mulch failure in NT for different treatments, as described in Table 1.

Table 1. Treatments and experimental procedure used by Morais & Cogo (2001) to evaluate conditions necessary for residue removal (under NT) on a sandy loam Ultisol in south Brazil.

Treatments	Slope ($m\ m^{-1}$)	Simulated rain ($mm\ h^{-1}$)	Unit flow-q ($m^2\ s^{-1}$)
T1 = No-till - Corn anchored residue 7600 kg ha^{-1} dry mass	0.097	58.3	27,41,61,77,103,128, 132, 143, 187, 199
T2 = No-till - Corn anchored residue 6200 kg ha^{-1} dry mass	0.098	58.3	25, 35, 54, 74, 106, 131, 135, 145, 184, 208
T3 = No-till - Black Oat anchored residue 5600 kg ha^{-1} dry mass	0.111	60.6	29, 37, 64, 82, 103, 119, 139, 157, 176, 192
T4 = No-till - Black Oat anchored residue 3400 kg ha^{-1} dry mass	0.114	60.6	25, 36, 61, 80, 103, 120, 146, 164, 170, 184
T5 = No-till - Soybean unanchored residue 4700 kg ha^{-1} dry mass	0.097	55.4	50, 61, 79, 93, 114, 132, 149, 157, 176, 195
T6 = No-till - Bare soil	0.098	55.4	49, 65, 73, 87, 109, 120, 144, 163, 183, 185
T7 = No-till - Soybean anchored residue 5100 kg ha^{-1} dry mass	0.111	55.2	13, 50, 69, 82, 107, 115, 131, 160, 168, 177
T8 = No-till - Soybean anchored residue 3400 kg ha^{-1} dry mass	0.114	55.2	13, 50, 66, 82, 98, 110, 133, 139, 160, 174
T9 = No-till - Soybean unanchored residue 3950 kg ha^{-1} dry mass	0.097	60.3	21, 47, 62, 89, 104, 124, 138, 146, 164, 194
T10 = No-till - Bare soil	0.098	60.3	18, 47, 66, 96, 102, 111, 136, 153, 193, 212
T11 = No-till - Soybean anchored residue 4500 kg ha^{-1} dry mass	0.111	58.7	14, 29, 57, 74, 96, 11, 127, 144, 172, 175
T12 = No-till - Soybean anchored residue 2900 kg ha^{-1} dry mass	0.114	58.7	10, 41, 66, 83, 106, 127, 135, 161, 186, 185

The two equations used to estimate the hydraulic parameters are described below.

$$F = \frac{V}{\sqrt{g*h}} \qquad \text{(eq. 4)}$$

Where F is the Froude Number (dimensionless), h is the flow depth (m);

$$Re = \frac{V*h}{\upsilon} \qquad \text{(eq.5)}$$

Re is the Reynolds number (dimensionless), ν is the kinematic viscosity ($m^2\ s^{-1}$);

RESULTS AND DISCUSSION

Fig. 1 (a) shows the shallow flow regime according to the Reynolds (Re) and Froude numbers observed in the twelve treatments carried out by Morais & Cogo (2001). According to this figure, the predominant shallow flow regime was supercritical/transition and subcritical/transition. Supercritical shallow flows are considered potentially erosive.

Table 2 presents the n-Manning and *f*-Darcy-Weisbach values derived for cornstalk, black oats-residue, soybeans-residue and bare soil. The highest friction coefficients were observed with black oats-residue, demonstrating the capacity of this crop (biomass> 3 Mg ha^{-1}) to reduce shallow flow velocity. For bare soil, the n-Manning and *f*-Darcy-Weisbach values were, respectively, 5 and 27 times lower than those observed for treatments with crop residues.

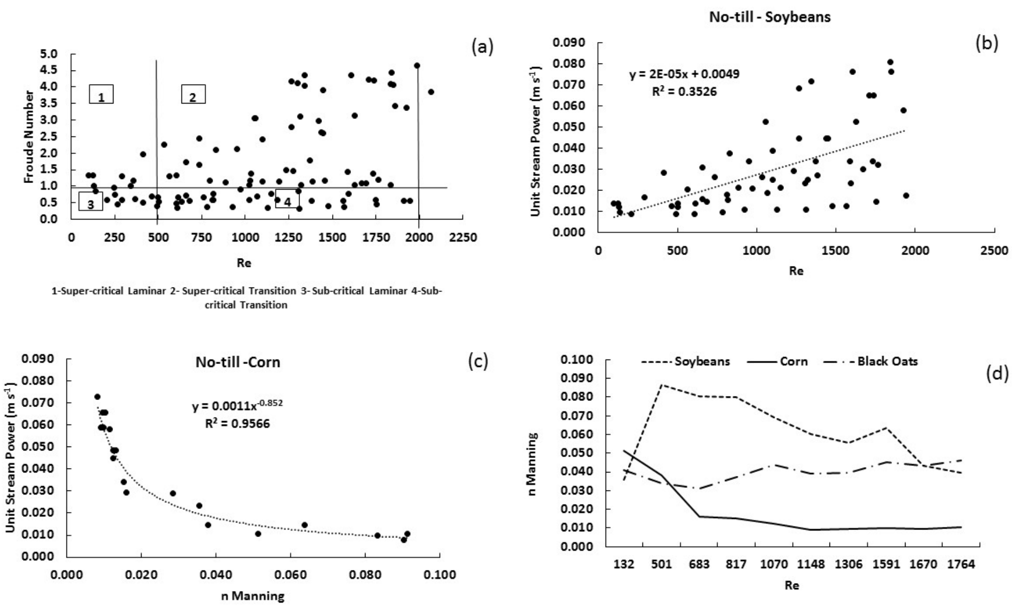

Fig 1. (a) Hydraulic flow regime for Morais & Cogo (2001) experiment data; (b) Re vs ω for soybean-residue in NT system; (c) ω vs n-Manning for cornstalk in NT system; (d) Re vs n-Manning for cornstalk, black oats-residue and soybean-residue in NT system.

Table 2. Derived n-Manning and *f*-Darcy-Weisbach friction factors for cornstalk, black oats residue and soybeans residue cultivated downhill using NT system.

Crop residue and biomass[1]	**Friction factor[2]**	
	n-Manning	***f*- Darcy-Weisbach**
Cornstalk (7.6-6.2 Mg ha^{-1})	0.031	0.82
Black Oat-residue (5.6-3.4 Mg ha^{-1})	0.062	1.70
Soybeans-residue (5.1-3.4 Mg ha^{-1})	0.050	1.54
Bare soil	0.009	0.04

[1]Dry biomass, [2]Average of data set considering different shallow flow rates (Table1).

The main observations verified when ω was compared with shallow-flow hydraulic characteristics were as follows:

- Increased flow turbulence (represented by the Reynolds number value) resulted in a linear increase of ω (Fig 1 (b));

- ω was reduced with increased n-Manning friction factor values (Fig. 1 (c));

- n-Manning values are dependent on Re number, and its friction effect decreases as flow turbulence increases due to immersion of rough elements at higher flow depths (Fig. 1 (d)).

CONCLUSIONS

The presence of crop residues in NT systems increased the friction factor, represented by the n-Manning and *f*-Darcy-Weisbach values, between 5 and 27 times. Among the different crop residues, black oats-residues showed the highest friction coefficient, demonstrating the importance of this crop in reducing shallow flow hydraulic power when included in a crop sequence. The ω increased linearly with increasing Re, but it was reduced by higher n-Manning values.

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3.6.O

ESTIMATION OF RAINFALL EROSIVITY: A CASE STUDY OF RIMAC RIVER BASIN

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INTRODUCTION

Peru exhibits a soil erosion spatial variability due to particular topographic and climate conditions, which are influenced by the tropical Andes and El Niño events. This phenomenon triggers convective storms in arid zones, mostly coastal areas (Rau et al., 2016). The kinetic energy of these raindrops that contributes to the detachment of soil particles is called rainfall erosivity.

Rainfall erosivity plays a key role in land use planning because it is one of the most widely used indicators to determine the potential risk of water erosion. Nowadays, Peru has scarce studies of rainfall erosivity estimation (Romero et al., 2007; Rosas and Gutierrez, 2016). Thus, in recent research, equations from other geographical contexts have been used to estimate the R factor. However, it has been shown that these relations tend to overestimate R factors and are not capable of representing its spatial variability (Naipal et al., 2015).This is mainly because some regions have complex rainfall patterns due to the presence of mountains and high local rainfall intensities (Rau et al., 2016).

On the other hand, some projections based on global models have indicated that Peru will face variations in rainfall patterns due to the process of global warming (Vuille et al., 2008). According to the UN Post- 2005 Development Agenda entitled *Transforming Our World: the 2030 Agenda for Sustainable Development*, governments such as Peru have adopted new global sustainable development goals. The fifteenth goal found in this document stipulate to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss. In this context, this study aims to estimate rainfall erosivity from daily precipitation records whose case study is the Rimac River basin (United Nations, 2015).

MATERIALS AND METHODS

The study area is located within the provinces of Lima, Huarochiri and Callao, in which different activities such as mining and agriculture are developed in the highest area, and urban development in the lower area of the basin. Rimac river basin covers an area of 3503.95 km^2 and represents 0.24% of the total area of Peru, approximately (ANA, 2010). Moreover, rainfall patterns show marked variations, for instance in the lower area of the basin rarely they exceed 50 mm/yr, and while in the highest area rainfall exceed 1000 mm/yr.

The methodology used to estimate rainfall erosivity can be summarized in the following four processes:
(1) an analysis of consistency and homogeneity of rainfall data was performed based on the Regional Vector Method (RVM), (2) daily rainfall erosivity were calculated by Wischmeier and Smith (1978) approach from RUSLE model for 4 stations with 2-years high resolution data, (3) rainfall erosivity estimates

from daily rainfall intensity data were determined by Yu and Rosewell (1996) model, and (4) a rainfall erosivity map was generated from the interpolation of the calculated R-values.

Rainfall data was collected from 13 weather stations, of which only 4 had 2-years high resolution data and the others only a daily resolution level (Figure 1). This meteorological stations are part of the national rain gauge network managed by the SENAMHI (National Meteorological and Hydrological Service of Peru). The database consisted of rainfall series for the period 9 September 1995 to 31 December 2015.

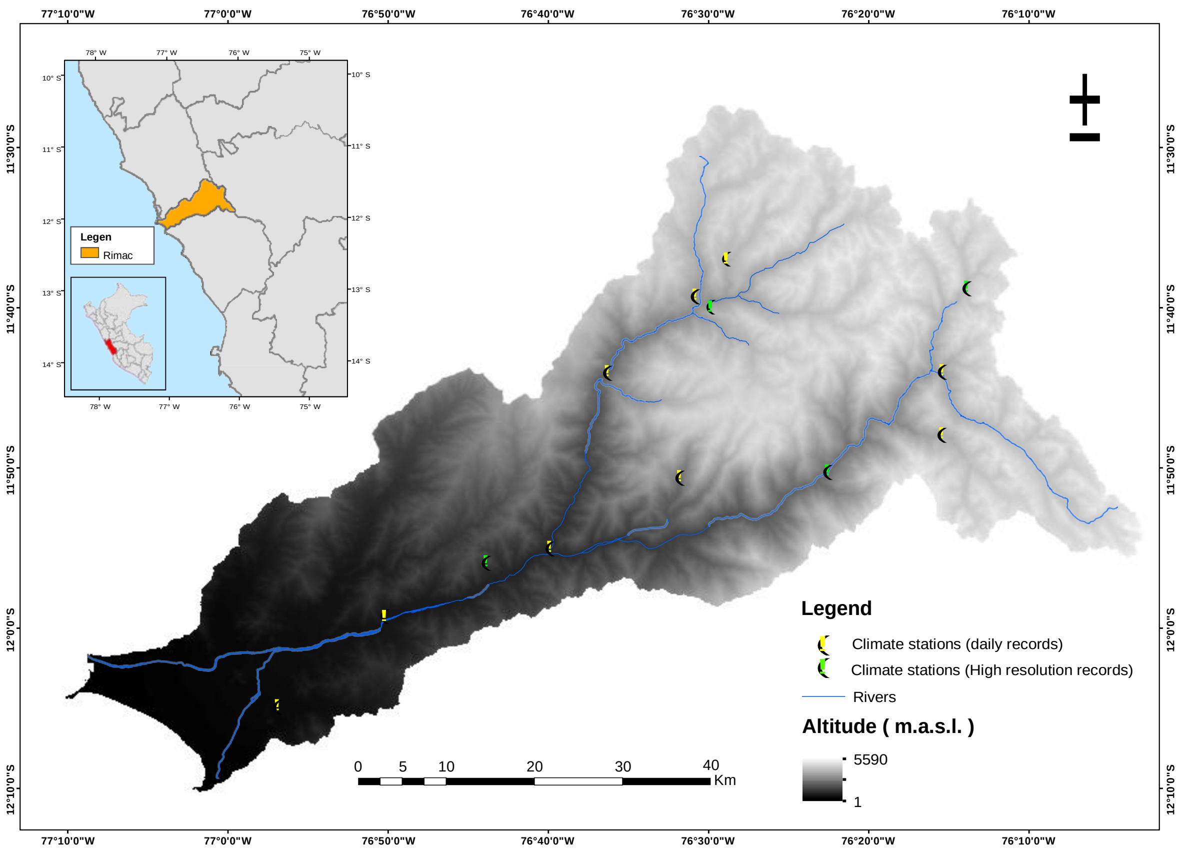

Figure 1: Location of the study area and the weather stations used to estimate R-factors

According to the first step, rainfall series were subjected to an analysis of consistency and homogeneity based on RVM. For that, the study area was divided into four zones with similar rainfall behaviors. This implied replacing and completing records with values corresponding to nearby stations, as a result of which it was possible to create a continuous daily rainfall database. As a second step, daily rainfall values for 4 stations with 2-years high resolution data were calculated based on the methodology described by Wischmeier and Smith (1978) and Renard et al. (1997). R factor is calculated by the average annual sum of the product of a storm's kinetic energy and its maximum 30-min intensity:

$$R = \frac{1}{n}\sum_{j=1}^{n}[\sum_{k=1}^{m}(E)_k(I_{30})_k]_j \tag{1}$$

where R is the average annual rainfall and runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), E is the total storm kinetic energy (MJ ha^{-1}), I30 is the maximum 30-min rainfall intensity (mm h^{-1}), j is an index of the number of years used to produce the average, k is an index of the number of storms in each year, n is the number of years used to obtain the average R, and m is the number of storms in each year.

The next step was to estimate rainfall erosivity from daily rainfall intensity data for the same 4 weather stations but with 20-years records by Yu and Rosewell model. This equation is based on the Richardson et al. (1983) method and uses a sinusoidal function to describe the seasonal variation of their coefficients. The model can be written in the following form (Yu and Rosewell, 1996):

$$EI = \alpha\left[1 + \eta\cos(2\pi f_j - \omega)\right] \sum_{j=1}^{N} P_k^{\beta}, \qquad P_k > P_0 \tag{2}$$

where P_k is the daily rainfall amount, P_0 is the threshold rainfall amount which was considered as zero, N is the number of rain days with rainfall amount in excess of P_0 in the month, and α, β, η and ω are model parameters. These parameters were estimated by using a iterative algorithm minimizing the sum of squared errors.

Finally, a rainfall erosivity map was generated from the interpolation of the calculated R-values. For that, the estimated R-values were extrapolated for the remaining 9 weather stations with the associated precipitation data. Therefore, rainfall erosivity points estimated in the 13 stations were interpolated by kriging and a linear semi-variogram model. Consequently a rainfall erosivity map was generated in order to obtain a spatial distribution of R factor.

RESULTS AND DISCUSSION

Mean annual precipitation map (MAP) was generated (Figure 2) from 20-years records, in which the highest rainfall is located in the upper area of the basin at Tingo station (835.8mm/yr), and the lowest rainfall is located in the lowest area of the basin recorded at the Ñaña station (3.3mm/yr). Furthermore, to estimate R values for 2-years high resolution data at Casapalca, Chosica, Matucana and Sheque weather stations we analyzed 494 storms.

Rainfall erosivity map was produced for 20 years of period from 1995 to 2015 (Figure 2). The results show that Tingo presents the highest erosivity values (512.6 MJ mm ha^{-1} h^{-1} yr^{-1}). Conversely, Ñaña obtained the lowest erosivity values (2.24 MJ mm ha^{-1} h^{-1} yr^{-1}). Carampoma, Casapalca, Rio Blanco, San Jose de Parac and Tingo exhibit the highest R-factor values because this area shows the highest rainfall and are located close to the Andean region. Von Humboldt and Ñaña present the lowest R-factor values because in this region rainfall is scarce.

CONCLUSION

This study presents a rainfall erosivity map of the Rimac River basin, which is very relevant because the most important city of the country is located here. This map is an indispensable tool in order to evaluate the soil erosion risk to be able to mitigate its effects with the application of soil conservation measures. It is important to mention that methodology used to estimate erosivity values represents a great opportunity to apply it to areas where updated information is limited. Likewise, this study is also the first step toward achieving the objective fifteenth of the Sustainable Development Goals.

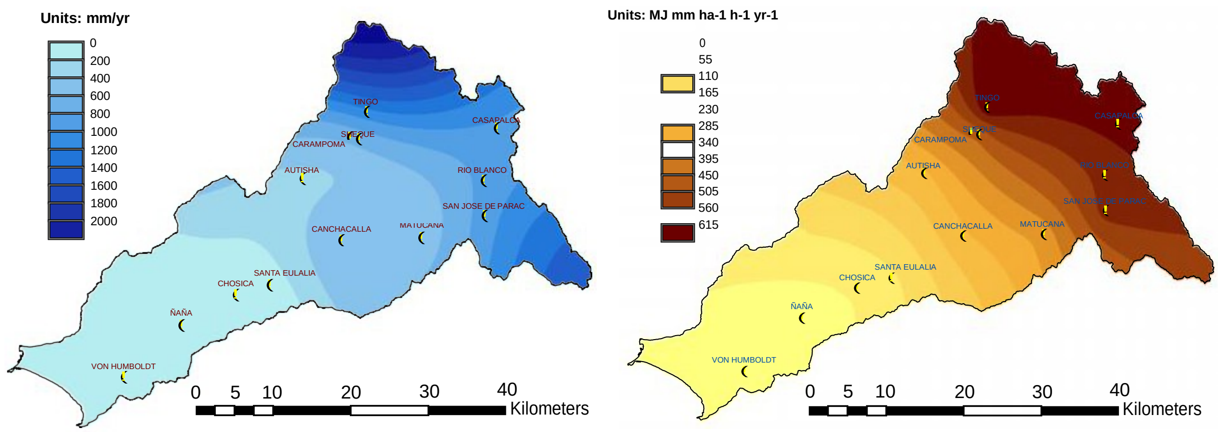

Figure 2: Mean annual precipitation (left) and the annual rainfall erosivity (right) maps for the Rimac River basin.

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3.7.O

WATER BALANCE IN COLOMBIAN ANDEAN SOILS WITH CONVENTIONAL AND ORGANIC PASSIONFRUIT CROPS UNDER THE "*EL NIÑO*" PHENOMENON.

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INTRODUCTION

The effects of climate instability have been steadily increasing in recent years in Colombia, resulting in a rise in the economic and social vulnerability of small and medium-sized farmers. The adaptability of agriculture in regions affected by the negative effects of climate change depends to a large extent on the intensity, frequency and duration of extreme events. In this sense, the development of strategies should include not only the study of alternative crops that are more resistant to the new conditions, but also the implementation of technology in order to improve the resilience of the traditional ones. Either way, it is essential to advance the knowledge of soil-plant-climate relationships, since water needs and availability are the main factors in determining the adaptive capacity of any crop. This study shows an approximation to the water balance in two passionfruit orchards.

METHODS

Commercial plots were on volcanic soils (Ando soils) and located between the towns of Manizales and Chinchiná (Caldas, Colombia). Full-productive three-year passionfruit (*Passiflora edulis*; local name *maracuyá*) was cultivated in both orchards, one of them with conventional methods and the other with organic ones. In each one of them, a G3 Gee Passive Capillary lysimeter and two 10HS moisture sensors (Decagon Devices, Pullman, WA, USA) were installed in order to monitor the evolution of soil moisture and drainage every hour. Moisture sensors were placed at 40 and 80 cm depth. The lysimeter, connected to an Em50G data-logger (Decagon Devices, Pullman, WA, USA), was used for hourly direct monitoring of drainage water below the root zone at a depth of 1 m, as well as for monthly extraction of drainage water when drainage occurred for leachate analysis. The volcanic texture allowed the lysimeter to be installed using the ''intact soil monolith'' installation method (Decagon Devices, 2014), which leaves part of the soil profile undisturbed. A weather station (with relative humidity, temperature, wind speed, precipitation and radiation sensors) (Decagon Devices, Pullman, WA, USA) was installed for ET_0 calculation through the Penman-Monteith method (Allen et al., 1998), which allowed estimating ET_C. Both fields were managed as usual by the farmer.

The daily soil water balance, is described by (Allen et al., 1998):

$$P + I = ET_C + D + \Delta H$$

where P is the precipitation (mm); I stands for irrigation (mm) ET_C is the crop evapotranspiration (mm); D is the drainage (mm); ΔH is the variation in soil water storage (mm). Runoff was not included in the soil water balance equation, because there were no runoff events because these soils had good infiltration (>100 mm h^{-1}).

Precipitation values (P) were obtained from the meteorological station, while in the crops considered there was no irrigation (I = 0). The volume of drained water (D) was obtained from the lysimeters, and the variation in soil water content (ΔH) was calculated by difference.

RESULTS AND DISCUSSION

Meteorological condition and drainage were monitored for the period September 2015 - August 2016. The first remarkable point is that, during this period, rainfall was 46% lower than the average of the location (Guzmán-Martínez and Baldión-Rincón, 2003), with eleven out of the twelve months registering lower precipitation values (Table 1). Indeed, the "El Niño Southern Oscillation" (ENSO) phenomenon in these months has been thoroughly reported to be one of the most severe drought episodes in recent times in Colombia, causing several problems related to water availability and agricultural productivity.

Table 1. Drainage, rain, average rain at the location (historical means from Guzmán-Martínez and Baldión-Rincón, 2003), ET_C and water balances. All values are in mm.

	Drainage		Rain	Rain average	ETc	Water balance	
	Conventional	Organic				Conventional	Organic
sep15	0	0	53	191	94	-41	-41
oct15	86	93	210	306	100	24	18
nov15	26	53	158	260	86	45	19
dec15	0	0	3	185	97	-93	-93
jan16	0	0	51	147	109	-57	-57
feb16	0	0	48	147	99	-51	-51
mar16	0	0	87	201	111	-24	-24
apr16	132	81	324	279	88	104	155
may16	90	64	199	280	90	20	46
jun16	46	52	109	191	89	-25	-31
jul16	0	24	74	155	92	-18	-42
aug16	0	0	35	168	106	-72	-72
Total	380	367	1351	2510	1161		

Attending to the results, this episode was especially severe from December 2015 to March 2016, resulting in four months with continuous negative balance. This four-month low-rainfall period led to a continuous decrease in soil volumetric water content at 80 cm, as shown in Figure 1. During this time, rain events resulted in small increases in water content at 40 cm, but water did not reach the lower depth, showing that water was either consumed by the plants or retained in the soil profile at upper levels.

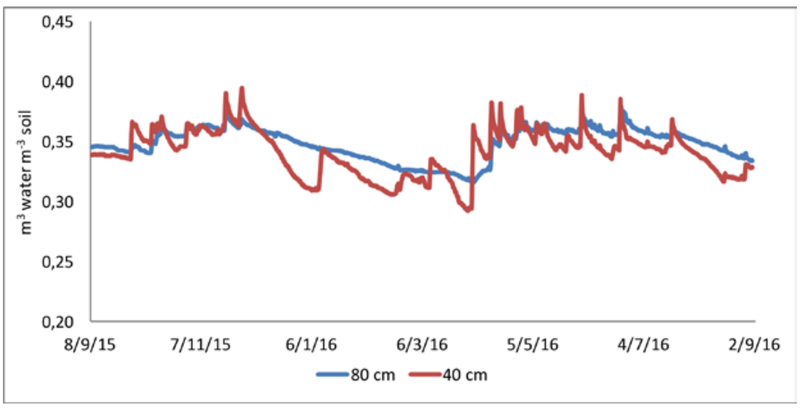

Figure 1. Soil volumetric water content at two depths (40 and 80 cm) in the conventionally-grown passionfruit plot.

Passionfruit produces a continuous crop, with flowering and fruit set happening at the same time as the first fruit ripens. This extends the harvest season of passionfruit over the entire year. Due to the lack of irrigation, yield must have been affected by such negative balances. According to Menzel et al. (1986), even mild soil moisture stress can severely limit vegetative growth and potential yield in passionfruit, especially during flowering. Consequently, passionfruit productivity may have been impaired up to six months after the stress finished. In this preliminary study, no remarkable differences were found between the organic and the conventionally-grown orchards.

CONCLUSIONS

As an important fact to highlight, the influence of the *El Niño* phenomenon was remarkable, and it subjected the crop to some periods of water stress during 8 out of the 12 months, 4 of which continuously. With this kind of climatic scenario getting more likely, it is necessary to raise the need of implementing support irrigation systems, selecting drought-resistant cultivars and including better soil and water management and conservation techniques.

ACKNOWLEDGEMENTS

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3.8.O

DEFINING DRAINAGE CLASSES FROM A CLOSED-FORM PARAMETER FOR PREDICTING THE HYDRAULIC CONDUCTIVITY.

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INTRODUCTION

The Less Favoured Area (LFA) scheme is a major element of the EU Rural Development Policy, aimed at supporting farming in areas with natural handicaps or low soil productivity. In Ireland, soil drainage class is considered to have a predominant influence on soil processes. Under Article 32 of EU Regulation 1305/2013 (the Rural Development Regulation) each Member State (MS) must designate areas eligible for payments under the Areas of Natural Constraints (ANC) scheme. From 2018, eligible areas must instead be designated using a set list of bio-physical criteria such as low temperature, dryness, excess soil moisture, limited soil drainage, unfavourable texture and stoniness, shallow rooting depth, poor chemical properties, steep slope. The mapping of soil drainage classes is necessary however it often becomes laborious and expensive using classical soil mapping methods because of the intensive sampling that it's required over large areas.

Soil drainage classes refer to the frequency and duration of periods of water saturation or partial saturation and refer to the average soil moisture regimes. There are two terms to be considered in soil drainage: Infiltration capacity and permeability. Soil water infiltration is the downward entry of water into the soil. Therefore, it is the maximum rate at which soils and rocks can absorb rainfall, whereas the soil permeability is the ability of water to move through depth. Soil drainage depends on the physical and chemical properties of the soil; notably particle size distribution (the range of particle sizes present), pore space, pore size and the continuity of the spaces. Soil drainage is otherwise referred to as hydraulic conductivity (K). There are two main types of hydraulic conductivity, surface infiltration rate and saturated hydraulic conductivity (Ksat). Ksat is the rate at which a soil that is already saturated with water will conduct water away from the source. It is one of the most important soil physical properties for determining infiltration rate and other hydrological processes (Gülser and Candemir, 2008).

Obtaining accurate measurements of hydraulic conductivity is very difficult, mainly because of the wide variability of soils and the presence or absence of pores, cracks, worm holes etc. The direct methods of measuring this property, either in situ or in the lab, are time consuming, expensive and impractical across large scales. Pedotransfer functions (PTFs), which relate readily collected data to hydraulic properties, can be used to overcome these problems (Hulme et al., 2001). Some PTFs have been incorporated into standalone computer programs like RETC or Rosetta to predict water retention parameters, Ksat and unsaturated hydraulic conductivity. In Ireland, with the use of these PTFs and with the completion of the 3rd edition Irish soil map, the opportunity to map soil drainage classes for agricultural areas on a national scale exists for the first time.

The objectives of the current study were: to define the drainage classes using predictive hydraulic conductivity (Ksat) for a real set of soil profiles and to derive general Ksat values for each drainage class. This will then facilitate the mapping of the spatial distribution of drainage classes using the National soils map of Ireland applicable for a more localised level to satisfy the necessary level of detail required for soil input data for delineation of ANC.

MATERIAL AND METHODS

Soil Sampling

A dataset, containing 801 profiles (3953 soil horizons samples), was gathered from An Foras Talúntais soil survey (AFT) (legacy data) and from Irish Soil Information System (SIS) database. In both soil survey, soil profiles were sampled and analysed for each soil horizon to a depth of 1 metre for a range of measurements, including soil organic carbon, texture, cation exchange capacity, pH. However, in An Foras Talúntais soil survey (AFT) in most cases no bulk density (BD) measurements were taken. These horizons required BD predictions using specific PTF. Reidy et al. (2016) describes in detail the methodology applied in the calculation of PTFs for soil bulk density at the soil horizon level for the legacy data and Irish SIS data.

Use of PTF for Ksat prediction

Indirect methods (PTF) have been developed to estimate Ksat from other available properties, such as sand, silt, and clay contents, organic matter, bulk density, and particle-size distribution. Some PTFs have been incorporated into standalone computer programs like RETC or Rosetta (Schaap et al., 2001). Both models are able to estimate van Genuchten (1980) water retention parameters and Ksat, as well as unsaturated hydraulic conductivity parameters based on Mualem-Van Genuchten equation (Mualem, 1976, van Genuchten 1980) and pore-size model (Schaap et al., 2001).

The RETC model was applied to 192 modal profiles (about 716 horizons), however Rosetta model was used to obtained the same properties for 609 non-modal profiles, up to approximately 2043 soil horizons. A linear correlation confirmed that the data results were similar using either of the two models. On this basis, the Rosetta model was used instead of RETC for predicting Ksat due to the increased time requirement to use RETC.

However, for both model, some profiles were excluded where the models did not work properly because these models do not sufficiently predict Ksat (cm/hour) for a set of soil properties (Fenton et al., submitted):

- Organic horizons: SOC (%) > 12
- Humic horizons: SOC (%) = [5-12] & BD = [0.5-1.2]
- Sandy horizons: SOC (%) < 5%; BD = [0-1] & Sand > 70%
- Alluvial soils
- Groundwater Gleys

(Soil Organic Carbon (SOC); Soil bulk density (BD); soil textural class affected for sand content > 70%: sandy soils, loamy sand or sandy loam soils.)

Diagnostic features and drainage class designation

The new soil classification system includes Soil Subgroups. The 62 Soil Subgroups existing, are classified within a Great Soil Group by using 9 diagnostic features described in the Irish Soil Classification. The diagnostic features are referred to major processes affecting the top 40 cm in soils and therefore useful in understanding soil management and helping to distinguish the next level of detail in the soil classification. The key diagnostic features relating to drainage status (redoximorphic features) in Irish soils are 'gleyic' and 'stagnic' (Schulte et al., 2015).

Following the methodology developed by Simo et al. (2015b) in tandem with expert knowledge, drainage classes were assigned at soil subgroup level.

Table 1. Categorisation of taxonomic soil subgroups into drainage classes, based on the diagnostic criteria listed based on Schulte et al. (2015) and Simo et al. (2015a).

Drainage category	Diagnostic rules
Excessive	Dominance of sandy loam and sandy textural classes within association
Well	No mottling, no full argic/spodic horizon present
Moderate	Mottling 40 cm to 80 cm with no organic matter accumulation, but argic or spodic may be present
Imperfect	Mottling 40 cm to 80 cm AND some organic matter accumulation and argic/spodic horizon present (at least a score of 1 in either category)
Poor	Mottling within 40 cm argic /spodic horizon causing stagnation

In cases where evidence of saturation was found within 40 cm of the surface horizon the diagnostic feature was recorded as '1'. These soils were designated as poorly-drained. Where saturation occurred between 40 cm and the lower profile boundary (80 cm), the diagnostic feature was recorded as '0.5'. These soils required additional diagnostic features to allow for designation to either the moderately-drained or imperfectly-drained category. Imperfectly drained soils required the diagnostic feature of organic matter accumulation to score '0.5' or greater and scores of at least '1' for either the argic or spodic features. By contrast, designation of moderately drained soils did not require any organic matter accumulation, but may score '0' or '0.5' for the spodic or argic features. Where no evidence of waterlogging occurs, the feature is recorded as '0'. Subsequently, soils with a score of '0.5' or less for organic matter, spodic or argillic features were designated as well-drained. Finally, excessively drained soils were considered those associations that are dominated by soils that have a loamy sand or sand texture (Schulte et al., 2015). Based upon this criterion, that includes the relevant diagnostic features (wetness, organic carbon, argillicity and spodicity) for each subgroup (62 soil subgroups in the Irish Soil Classification System), a query to find subset of subgroups with similar diagnostic features criterion was applied. This resulted in 26 drainage group models with similar diagnostic features.

Estimation of Ksat for drainage classes

A selection of 564 soil profiles (1938 horizons) from the Irish SIS soil profiles were used for the calculation of the ranges of Ksat for drainage classes. A representative equation for each drainage group model was calculated.

In order to justify the tendency or pattern of the different drainage classes taking into account permeability and infiltration capacity and diagnostic features, a second model of the multiple linear regressions was applied. The aim of this approach was to ascertain how many drainage classes exist based upon the Ksat. The multiple linear regressions were applied between Ksat equations for each drainage group model. The slope of the Ksat equations shows the effect of depth on the hydraulic conductivity representing the permeability while the intercept show the effect at the surface thereby representing the infiltration capacity.

A similar approach was adopted whereby diagnostic features were applied, where scores used were 1, 0.5 or 0. Multiple linear regression was correlated the slope or intercept with the scores 0, 0.5 or 1 for each of the diagnostic features (wetness (W), organic carbon (OC), argillicity (A) and spodicity (S)) depending on the relation to drainage status. The square of the diagnostic features (W2, OC2, A2, S2) and some intersects between diagnostic features (OMxS, OMxA, WxA, WxS, AxS, WxOM, AxS) were considered and to be integrated in the multiple linear regression model.

With the equations product from MLR, the Slope equation and Intercept equation were obtained, which were used to calculate the Slope and Intercept indexes. The continuous soil drainage index was calculated for each subgroup combination by Intercept index divided by slope index.

Accuracy assessment of the drainage equations

Correlation coefficients (r2) and mean squared error (MSE) for the drainage group models (amalgamation of subgroups according to diagnostic features) were calculated between the reference (Ksat from PTF) and the drainage equation of the multiple linear regression models. This data was used to assess the quality of the drainage equations. Equations with relatively low MPE and high correlation coefficient were considered good equations. Coefficient of determination (r2) and root mean squared predicted error (RMSPE) were used to assess the quality of the curve fitting.

Mapping drainage classes

The soil map of Ireland at subgroup scale defines the major pedogenic properties influencing soil management (Simo et al., 2015b). This map provides a simplified version of the national soil series association map, but is designed using the same approach with a soil subgroup leading and many soil subgroups component. Simo et al. (2015a) describe in detail the methodology applied for the validation of the soil subgroup map product. This subgroup map is still a map of soil associations, but the number of subgroups found within each association is more limited than soil series, however of the relative importance (%) of each subgroup component in the association have been obtained.

The use of subgroups allows the indicative assignment of drainage classes to the most dominant drainage class in the association based upon its proportionality.

RESULTS

Correlation Rosetta with RETC model

PTF developed Ksat prediction models, RETC and Rosetta were, both used, with both achieving good Ksat results. In the absence of measured values for a particular soil type, the hierarchical level 5 of Rosetta yielded the best results with the measured input data (bulk density and soil texture) which aligns with Alvarez-Acosta et al. (2012).

The correlation data between both models (Rosetta or RETC) is almost 100%. This finding endorsed the use of the faster Rosetta model, as both yielded very equivalent results.

Drainage group model designation and Soil drainage equations

Soil Subgroups of the Irish Soil Classification System (Simo et al. 2014) were amalgamated according to the diagnostic features (W, OM, A, S) criteria (0, 0.5, 1) in order to be able to calculate the best regression correlation for a set of soil profiles between Ksat (cm/hour) and depth (cm). A total of 15 drainage group models were studied with their drainage equations (Table 2).

These equations explain how water is moving in the soil. Multiple linear regressions between the slope or the intercept of the ksat equations and the diagnostic features showed the tendency or pattern of the different drainage classes (well drain, moderately drain, imperfectly drain, poorly drain).

Table 2. Ksat linear equations and validation of the equation using MPE, SDPE, RMSPE and r2 for each drainage group model studied.

Drainage Group model	Ln (Ksat) Equations (Ksat*) (cm/hour)	Validation			
		MPE	SDPE	RMSPE	r^2
Group 2	Ksat* = -0.0105*depth + 1.6573 R^2 = 0.9126	-0.56	2.51	2.53	0.15
Group 3	Ksat*= -0.0084*depth + 2.0538 R^2 = 0.7412	0.27	2.24	2.19	0.32
Group 5	Ksat* = -0.0014*depth + 1.9657 R^2 = 0.3506	0.21	5.14	5.07	-
Group 7	Ksat* = -0.0069*depth + 1.9819 R^2 = 0.4441	-0.17	3.13	3.1	0.05
Group 8	Ksat* = -0.0178*depth + 2.1039 R^2 = 0.8399	-0.42	1.77	1.64	0.91
Group 9	Ksat* = -0.0172*depth + 1.9657 R^2 = 0.9562	1.08	4.87	4.64	0.41
Group 10	Ksat*= -0.0069*depth + 2.2264 R^2 = 0.9119	-0.65	4.55	4.38	0.29
Group 11	Ksat* = -0.0073*depth + 2.2471 R^2 = 0.63	-0.11	2.88	2.74	0.27
Group 12	Ksat* = -0.0206*depth + 1.9663 R^2 = 0.9391	-0.03	1.09	1.05	0.69
Group 13	K sat*= -0.0097*depth + 2.7266 R^2 = 0.8499	2.17	3.56	3.97	0.27
Group 17	Ksat* = -0.0143*depth + 1.9297 R^2 = 0.4952	-0.46	3.8	3.75	0.63
Group 18	Ksat* = -0.0084*depth + 1.8071 R^2 = 0.9687	-1.23	3.42	2.96	0.37
Group 21	Ksat* = -0.0222*depth + 2.566 R^2 = 0.962	2.56	3.47	3.01	0.15
Group 23	y = -0.0277*depth + 2.303 R^2 = 0.8105	-	-	-	-
Group 25	Ksat* = -0.0067*depth + 1.4721 R^2 = 0.5915	1.07	1.13	1.48	0.64

The results of the multiple linear regressions (Table 3) show that when equation slopes are considered, wetness was obtained as the only significant diagnostic feature correlated. Positively, wetness is scored using 0, 0.5 and 1, resulting in three significant slopes in the model that represent permeability, which match with three drainage classes (well, moderately and poorly drained). This indicates that the imperfectly drained class should not be considered. Conversely, applying multiple linear regression with the intercepts of equations versus the diagnostic features scored with 0, 0.5 or

1 of (wetness, organic carbon, argillicity and spodicity), the results showed that spodicity, the interaction of OCxA and the interaction of WxOC, were significant in the model, therefore infiltration capacity can be represented by several diagnostic features.

The only drainage group model that was not clearly showing a direct drainage class was group 2. This drainage group was mostly formed by Luvisols, the soil drainage index obtained is limiting between well drained soil or moderately drained soils. Under expert knowledge Luvisols or soils with clay increase in depth have low permeability. This is similar to other authors (Hulme et al., 2001) who found that soil with high clay increases at depth have problems of vertical water flow because clay is strongly influenced by low permeability layers within it. Thus, drainage group 2 was considered moderately drained.

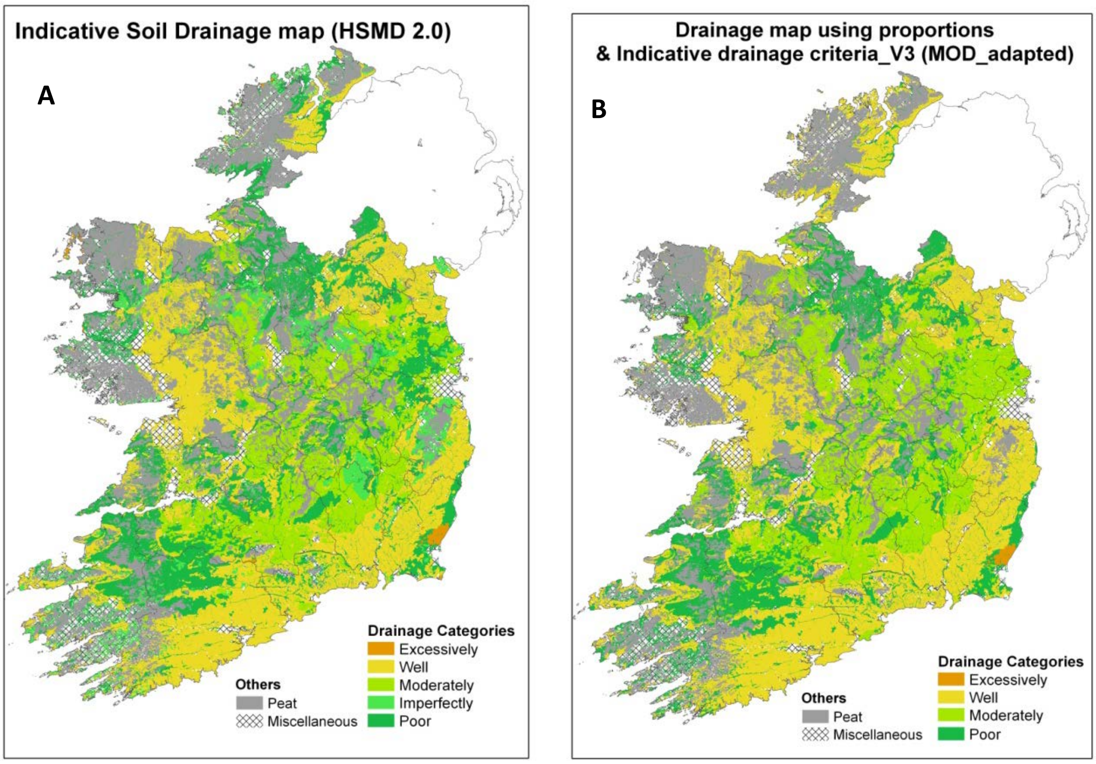
Table 3. Slope and Intercept equation were products of the multiple linear regressions. Equations used for obtaining the continuous soil drainage index.

Combination	Slope =-0,008785-0,01122*W	Intercept =1,827163+0,777767*S-2,176*OMxA+2,572498*WxOM	Index Value (I/S)	Index ranges	Final Drainage class
Group 3	-0.0088	2.2160	-252.2534	< -200	Well
Group 5	-0.0088	1.827163	-207.9867		
Group 7	-0.0088	1.8272	-207.9867		
Group 11	-0.0088	2.2160	-252.2534		
Group 13	-0.0088	2.6049	-296.5202		
Group 2	-0.0088	1.8272	-207.9867	-200 to -100	Moderately
Group 8	-0.0144	2.2160	-153.9456		
Group 12	-0.0144	1.8272	-126.9304		
Group 18	-0.0144	1.8272	-126.9304		
Group 21	-0.0144	2.4703	-171.6073		
Group 25	-0.0144	1.8272	-126.9304		
Group 23	-0.0144	1.3823	-96.0255	> -100	Poorly
Group 17	-0.02	1.99	-99.5		
Group 9	-0.0200	1.8272	-91.3353		

Updated drainage map for Ireland

The drainage map shows the spatial distribution and geographical variation of drainage classes for Ireland, this soil drainage map will underpin the SMD map for the assessment of two of the bio-physical criteria considered under the ANC: excess soil moisture and limited soil drainage.

Figure 1: Evolution of the Indicative Soil Drainage Map. (a) Indicative Soil Drainage map (HSMD 2.0) (2014) (b) The new map is based on the diagnostic features and proportionalities relative proportions to derive the dominant drainage class in the association.



CONCLUSIONS

The application of the multiple linear regressions reached positive results on permeability and infiltration capacity, as a result of this three drainage classes (well, moderately and poorly drained) were applied. Notably the soil drainage index that was generated aligned well with expert drainage knowledge, conceptual soil–landscape patterns and existing soil profiles. This finding highlighted the importance of the three discrete soil drainage classes related to the continuous soil drainage index; well drain (< - 200); moderately drain (-200 to -100); poorly drain (> -100). The excessively drained soils were those that have a loamy sand or sand texture.

The generated drainage index surfaces were found to be an important input parameter for the operational drainage map. The approach proved a viable technique for spatial drainage class predictions within available project resources, for operational digital soil assessment. The drainage map shows the spatial distribution and geographical variation of drainage classes for Ireland, this soil drainage map will underpin the SMD map for the assessment of two of the bio-physical criteria considered under the ANC: excess soil moisture and limited soil drainage.

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3.9.O

WILTING-POINT AND HYPER-DRY WATER CONTENTS WERE INDEXES OF SOIL-WATER REPELLENCY VARIATIONS ACROSS A COARSE SANDY FIELD.

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SUMMARY

Soil water repellency (SWR) can reduce root-water uptake and increase the risk for groundwater contamination due to accelerated pesticide leaching via preferential flow paths. The objective of this study was to predict the moisture dependent behavior of the SWR(w) relation (where w is gravimetric water content, g g^{-1}) across a coarse sandy field by using indexes of hyper-dry water contents and the permanent plant wilting point (PWP) water content (corresponding to a soil-water matric potential of pF 4.2). The SWR of 88 soils samples were measured at target moisture contents by the molarity of an ethanol droplet (MED) test and the corresponding pF value at each moisture content were measured using a dew-point potentiometer. We found maximum values of SWR closely around the PWP water content across the entire field. At the PWP, both capillary and adsorptive forces act inherently, retaining water in different soil compartments. Whereas capillary water is held in medium-sized pores and curvatures, adsorbed water is retained as moisture films and patches around soil particles. To account for this water retention behavior in the dry and hyper-dry range between pF 3 and oven dryness (pF 6.9), we established a simple soil-water retention w(pF) model with three fitting parameters. The new water retention model quantifies total, adsorptive and capillary water content and well represent w(pF) within the dry and hyper-dry regions where SWR starts and vanishes. The new w(pF model) was inserted into a recently developed SWR(w) model to accurately describe the entire SWR-pF relation. Both the maximum SWR (given as apparent contact angle), and the total SWR strength (SWR.area, area under the SWR-w curve) were well related to PWP and selected hyper-dry water contents, as well as to the soil organic carbon (SOC) content. In conclusion, PWP and selected hyper-dry water contents seem promising as field-scale predictors of areas with higher SWR and thus reduced soil hydrological and plant-support functions.

INTRODUCTION

Soil science increased attention in soil water repellency (SWR) through last decades (Dekker et al., 2005). SWR is a widespread property of soils with important hydrologic implications. For instance, it is known the feasibility of SWR to create unstable wetting fronts in soils (Bachmann et al., 2007; Deurer and Bachmann, 2007), compromising its filtration efficiency (Müller et al., 2014). Additionally, SWR can constrain the infiltration capacity, as compared to wettable soils, with consequently greater run off and erosion (Buczko et al., 2007).
The effect of SWR is related to its severity (degree) and persistence (Chau et al., 2014). Severity is associated to the interfacial energies of the solid-vapor, solid-liquid and liquid-vapor interfaces, defined in Young's equation. Severity is either measured by water drop contact angles (CAs), (Chau et al., 2014) or by liquid surface tension (ST, mN m^{-1}) of aqueous ethanol droplets infiltrating standardized soils (e.g. MED test), (Roy and McGill, 2002). MED is a ninety degree surface tension equivalent test inferring "apparent-CAs", sensitive in the range between 90° to 109° (Roy and McGill, 2002). Soils with contact angles below 90° (CAs<90°), exhibit subcritical SWR, in turn CAs>90° suggest strong SWR. Persistence is considered as a time-dependent contact angle CA(t), and can be closely related to the water content and the wetting history of soils (Bachmann et al., 2007).
Global change in clime is expected to increase affected areas by drought, inevitably leading to both reduced soil moisture and more frequent and prolonged droughts (IPCC, 2008). Scientist found that the severity of SWR is maximum near the permanent wilting point (PWP), (e.g. Regalado and Ritter, 2009; Rodríguez-Alleres and Benito, 2011), corresponding to most common water potential status in early summer or under tropical and subtropical clime regimes.
The pronounced seasonal variability of the severity and persistence of SWR (Buczko et al., 2007), driven by its nonlinear soil moisture dependency, results in the definition of "critical" soil water contents. For example, θ_{crit} (g g^{-1}) as a function of soil organic matter content (SOM) for prediction of the wettable and non-wettable soil regions (Taumer et al., 2005); θ_{FC} for soil water content of the fully wettable region around field capacity (Regalado and Ritter, 2005); $\theta_{g\text{-}max}$ or $\theta_{WR\text{-}max}$ the soil water content (kg kg^{-1}, m^3 m^{-3} respectively) at the maximum SWR and θ_g or $\theta_{non\text{-}WR}$ the soil water content at which SWR ceases (Regalado and Ritter, 2005; Karunarathna et al., 2010). Remarkable recent progress in modeling SWR has been achieved based on soil water content variations (e.g. Bachmann et al., 2007; Karunarathna et al., 2010; Regalado and Ritter, 2009). However, to the better known of the authors, little focus has been directed towards linking soil matric potentials to the severity of SWR, though it is well documented maximum effects of SWR near PWP (i.e. pF 4.2, where pF = log (|-15.000 cm H_2O|)).
The objective of this study is to present a new, simple but accurate function to represents the dry-end soil water retention (SWC), targeting the moisture region where SWR starts and vanishes. To obtain hyper dry soil water contents using the new model and try to explain SWR indices across the field. Additionally, we want to introduce a new w(pF) model for the SWR curve fitted to data, to describe SWR as liquid surface tension (ST) versus pF values (SWR-pF curve). Finally, we attempt to re-capture the measured variations of SWR.area across field conditions, via linear correlation between hyper dry-end soil water contents.

METHODS

A total of 88 soil samples were extracted from a barley field (0-20 cm depth) on a 15 × 15 m grid (Fig. 1), in spring 2012, prior to annual ploughing (22-23 cm depth), in Jyndevad, southern Denmark (54 53.620 N, 09 07.208 E). The samples 2-mm sieved in air-dry state were homogenized prior to analysis. The soil texture was determined with a combination of wet-sieving and hydrometer methods (Gee and Or, 2002). Bulk samples were ball-milled to determine the total organic carbon content (TOC, kg kg^{-1}) as well as CO_3-C with a FLASH 2000 organic elemental analyzer coupled to a thermal conductivity detector (Thermo Fisher Sci, Waltham, MA, USA). To fit the new dry-end soil water retention model, sets of nine 10-g air-dry subsamples were prepared from the 88 samples. Water was added while continuously mixing the subsamples to attain target moisture contents ranging from 0.01 to 0.08 kg kg^{-1} (Tuller and Or, 2005). Then the subsamples were stored in Ziploc bags for 3 weeks at 5°C to allow equilibrium. Following this period, the subsamples were transferred into a WP4-T Dewpoint Potentiameter (Decagon Devices Inc., Pullman, WA, USA), based on the chilled-mirror dew point technique (Gee et al., 1992). Two additional water contents (w) were determined by oven-drying samples at 60°C and 105°C, 48 h. Duplicate soil water potential (pF) measurements were taken for each subsample (20°C). The exact w at each pF value was found by oven drying at 105°C for 24h. The molarity (M, mol L^{-1}) of an ethanol droplet test (MED), (Watson and Letey, 1970; King, 1981) was undertaken for the subsamples used in pF measurements. A series of aqueous ethanol solutions from 0.01 to 0.40 m^3 m^{-3} (in 0.01 m^3 m^{-3} steps), were used to lower the ST of deionized water (δw = 72.1 mN m^{-1} at 20°C), and to calculated the severity of SWR. Higher the molarity of ethanol of a 60 µL droplet that remains on a plane soil surface for at least 5 seconds indicate stronger SWR (de Jonge et al., 2007). The relationship between molarity and corrected ST was determined by ST = 61.05 – 14.75 ln(M+0.5) according to Roy and McGill (2002). We calculated SWR.area as an estimation of the trapezoidal area below the SWR-w curve (ST vs w). Additional SWR indices (e.g. w.α.max and w.non.WR) were calculated following Karunarathna et al., 2010.

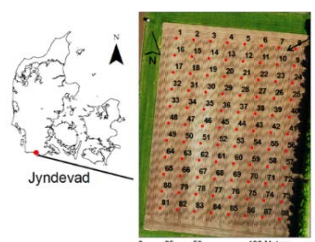

Figure 1. Sampling site field of the 88 soil samples and the 15 x 15m grid arrangement.

RESULTS

Table 1 shows the average and range of the parameters fitted to data of the 88 soil samples, using equation 1. A new, simple but accurate function for dry-end water retention is suggested to obtain water contents at given pF values (see Fig 2a). The black line represents the total water content (w, kg kg^{-1}), the red and blue lines correspond to the left and right side of the equation, respectively. In appearance the lines from Fig 2a display comparable trends to the lines representing adsorptive and capillary retention forces labelled in prior studies (e.g. Or and Tuller 1999).

Table 1. Distribution of the three fitting parameters of the new w(pF) model (SWC).

Parameter	Average	Minimum	Maximum
H	8.19 x 10^{-3}	3.42 x 10^{-3}	1.22 x 10^{-2}
B	3.75 x 10^{-3}	1.02 x 10^{-5}	2.53 x 10^{-3}
C	4.45	3	6

$$w = H(6.9 - pF) + B(6.9 - pF)^C \quad \textbf{(Eq. 1)}$$

From figure 2b we can see the utility of Eq. 2 (derived from Eq.1) used to describe ST versus pF. It is expected that this model (Eq.2) can achieve an optimal description of SWR in terms of CAs versus pF. The pF where SWR ceases is important for example in optimal irrigation planning and thereby is one of the key indices of the SWR-pF curve. As can be seen from Table 2, compared with the generally used proxy of SOC to predict water repellency, hyper dry-end water contents derived from the new model (Eq. 1) were preforming at the same level or better to predict several key SWR indices derived from the SWR-w curve (e.g. SWR.area, A1 and A2 not shown).

$$pF = 6.9 - 10^{\left(\frac{\log\left(\frac{w}{B}\right)}{\log\left(\frac{H}{B}\right)+C}\right)} \quad \textbf{(Eq. 2)}$$

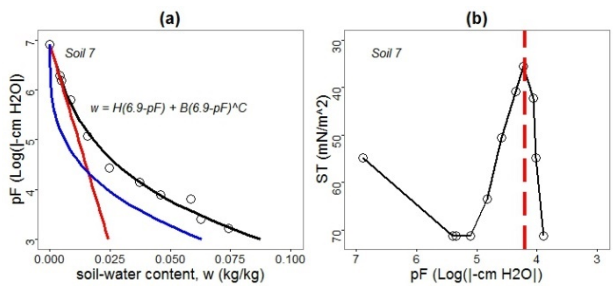

Figure 2. a) A new SWC model fitted from oven dryness to pF 3.0, b) A modeled SWR-pF curve.

Table 2. Correlation matrix of Pearson R coefficients for key SWR indices and hyper-dry water contents.

	SOC	w.pF3*	w.pF4.2	w.pF6.0	SWR.area	w.α.max	w.non.WR
SOC	1	0.74	0.58	0.48	0.43	0.57	0.56
w.pF3*		1	0.68	0.62	0.51	0.68	0.64
w.pF4.2			1	0.70	**0.54**	**0.71**	0.58
w.pF6.0				1	0.39	0.63	0.48
SWR.area					1	0.68	0.76
w.α.max						1	0.85
w.non.WR							1

(*) observed data, non-fitted, SWR*: trapezoidal approximation of the area below the SWR-w curve.

CONCLUSIONS

Permanent wilting-point water content (w.pF4.2) seemed the best proxy for SWR.area but also hyper dry-end water contents performed better than SOC for most SWR indices examined. Relations between hyper dry-end water contents and SWR indices were strong enough to use for example the wilting-point water content to re-capture the measured variation pattern across field conditions.

ACKNOWLEDGEMENTS

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3.10.O

EFFECT OF DRIP WATERING REGIMES ON THE GROWTH PERFORMANCE, YIELD, AND WATER USE EFFICIENCY OF SORGHUM (*SORGHUM BICOLOR*) IN SEMI -ARID ENVIRONMENT OF TANZANIA

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INTRODUCTION

The problem of water scarcity as one of the results of the climate change and its variability is expected to worsen more on agriculture and therefore food security and livelihood in arid and semi-arid areas (ASALs) of the world. This is partly because ASALs` dwellers rely on rainfed agriculture, which is highly sensitive to climate variability. According to IPCC, 2014, there will be increasingly failure of crops in the ASALs which dominate most countries of Sub-Saharan Africa. Since agricultural activities in this region rely on rainfed condition, any adverse effect on climate change would have a devastating effect on crop production and subsequently livelihood of the majority.

Like in other Sub-Saharan Africa countries, ASALs of Tanzania are mainly characterized by low rainfall, which is unreliable for food and commercial crop production. The areas are prone to harsh drought condition almost throughout the year. Farmers have been applying different soil and water conservation measures to improve crop production. Pits and traditional terraces have been used as a traditional sowing method to collect water in situ during the early growth stages of the crops in the (Shemdoe, 2011). However, due to the high rate of evapotranspiration demand and the presence of surface sealing and crusting; these techniques have been not promising.

Dodoma region in Tanzania is the representative in terms of the climate of many parts of arid and semi-arid areas of the Sub-Saharan Africa and the world in general, where sorghum crop is mostly grown. The area receives the very low amount of rainfall (250-550 mm), which is erratic and unpredictable to meet crop water requirement of the major crops grown. There has been persistent crop failure even for sorghum (one of the main staple food and cash crop), which was believed to be the drought tolerant crop. The breeding program has led to release drought tolerant and high yielding sorghum varieties, whose genetic yield potential under limited soil moisture can never be realized unless supplemented with water in dry spells or purely irrigated in dry seasons. Although the area experiences water scarcity, it still has an opportunity of utilizing underground water resources, intermittent rivers and rainwater harvesting techniques to increase crop yield production through efficiency means of drip irrigation in the dry season and dry spell periods. However, there was no scientific information on the effectiveness of the different drip watering regimes on enhancing sorghum growth performance that optimizes grain yield and thereby contributes to stabilization of food security and livelihood, while sustaining scarce water resource and safeguards the environment in ASALs.

METHODS

Field trials were conducted in the semi-arid area of central Tanzania, Dodoma, at Makutupora Agricultural Research Institute, with the aim of assessing the effectiveness of three drip watering regimes on the growth parameters, biomass, WUE and grain yield of sorghum. The irrigation regimes treatments were: Watering early in the morning (EM), late in the evening (LE) and both early in the morning and late in the evening (ELE). Treatments were replicated three times in a randomized complete block design for the two seasons (Dry season and dry-wet season). Seven mm of water was supplied to every plant on daily basis (McWilliams, 2003). Some adjustments were made depending on the sorghum growth stage, weather and soil moisture condition (FAO, 2013). Tensiometer and gravimetric water content methods were used to monitor the amount of water in the sorghum root zone.

Plant height, stem thickness, leaf width, leaf length and leaf numbers were measured in every phenological stage under different irrigation regimes. Leaf width and Length were lately used to compute leaf area index (LAI).

LAI = LA/GA ………………………(i), where; LAI: Leaf Area Index; LA: Leaf Area; GA: Ground Area.

Water use efficiency was determined by dividing the total amount of dry matter (Kg) by the total amount of water used by the crop in each season (mm).

$WUE = DM / ET_{cr}$ *........... (ii)*, where; WUE: Sorghum Water Use Efficiency; DM: Dry Biomass (Kg)

ET_{cr}: Amount of water used by Sorghum in a season (mm).

At physiological maturity, the harvested grains were sun dried and its moisture content was determined using digital grain moisture meter SINAR[R] AP (6060 Moisture analyzer). The grains were then weighed with electronic weighing balance in order to correlate the yield response of sorghum grown under different drip irrigation regimes.

RESULTS

The result showed that sorghum which was irrigated twice a day (ELE) had a higher plant height (P<.001%) than those irrigated once a day. The tallest plant height (138 cm) was measured at soft dough stage in plots that were irrigated twice a day in the dry-wet season, and the shortest plant height (36.6 cm) was recorded at vegetative stage in sorghum irrigated late in the evening in the dry season. When leaf numbers were counted, there was a significant difference (P<.001) in the two seasons at vegetative, soft dough and hard dough stages between the treatments. In both seasons, sorghum which was irrigated twice a day appeared to have the low number of leaves. The performance of sorghum stem thickness across the seasons appeared to be significantly different (P<.001) at vegetative, flag leaf, soft dough, and hard dough stage. The difference was brought by the favourable climatic condition in the dry-wet season. Within the season, the sorghum thickness in the plot irrigated early in the morning appeared to be thicker at vegetative (2.02 cm) and flag leaf stage (2.74 cm) compared to the sorghum irrigated late in the evening. This phenomenon can be explained by the fact that, during the daytime, the rate of evapotranspiration is very high. At this time stomata pores are open and more water is taken up by the plant for its biophysiochemical activities including photosynthesis and transpiration. In the dry season, the stem thickness was observed to increase up to flowering stage and then decreased at soft dough stage.

The effect of the three irrigation regimes on leaf area index (LAI) in the two seasons appeared to be significantly different (P<.001) at vegetative and soft dough stages. There was no significant difference (P>0.05) of LAI at flag leaf, flowering and hard dough stage between treatments. The LAI of the sorghum irrigated twice a day appeared to outperform other irrigation regimes in almost all the phenological stages of the Sorghum crop.

In evaluating the response of sorghum biomass and water use efficiency (WUE) under the three irrigation regimes and the two seasons, the result showed that sorghum irrigated twice a day had appreciably used the small amount of water to produced more biomass. The highest biomass (24910 Kg/ha) and WUE (4.53) were recorded from sorghum which was irrigated twice a day (ELE) in dry wet season, while the lowest biomass (10850 Kg/ha) and WUE (1.973) were registered in the dry season when sorghum was irrigated late in the evening (LE). This proves that biomass accumulation is not only a function of water but also the time of irrigation. Watering late in the evening produced less biomass

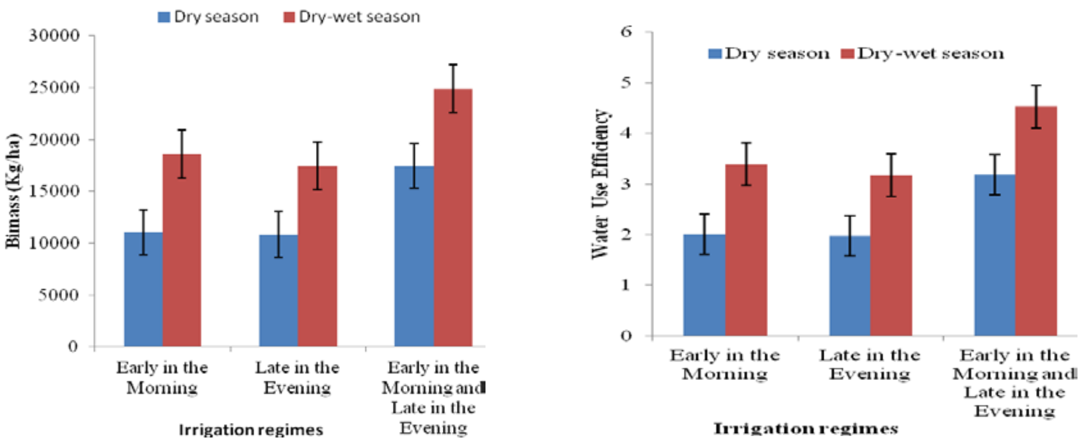

Fig 1: (a) Effect of irrigation regimes on biomass (b) Effect of irrigation regimes on WUE

The response of sorghum grain yield to drip irrigation under different irrigation regimes appeared to be significantly higher at P<.001 by two-fold when the crop was irrigated twice a day (ELE) as compared to when it was irrigated once a day. The highest grain yield (13.21 tons/ha) was harvested in the sorghum that was irrigated twice a day, and the lowest yield (6.82 ton/ha) was recorded from the sorghum irrigated late in the evening (LE) in the dry season.

There was no significant difference (P>5%) for the sorghum`s plots which were irrigated once a day within the season. The performance of yield in the dry-wet season appeared to be higher than the dry season's yield by 16.4% and 18.6% for EM and LE irrigation regimes respectively. This increment might be due to the influence of supplemental drip irrigation that was being carried out during the dry-wet season. It might have also been influenced by precipitation, which was mostly rained during the night hours. Contrary to the dry season, the performance of yield for the ELE sorghum in the dry season was higher than dry-wet season by 0.509 t/ha. This additional yield might have been due to a higher rate of photosynthesis triggered by long sun hours in the dry season as compared to the dry-wet season.

Application of water twice a day maintained additional available soil moisture to crop water requirement that created a favourable microclimatic condition for sorghum growth. The result suggested that irrigating once a day might cause partial root drying, and as a result, the leaf area is reduced and the overall photosynthetic activities of the plant are affected. The low grain yield in sorghum, which irrigated once a day in the morning (Figure 2) might be due to partial root drying caused by a higher rate of evaporation and soil moisture deficit occurring during daytime. The result also suggested that irrigating only late in the evening might result into hidden water stress that reduces yield significantly.

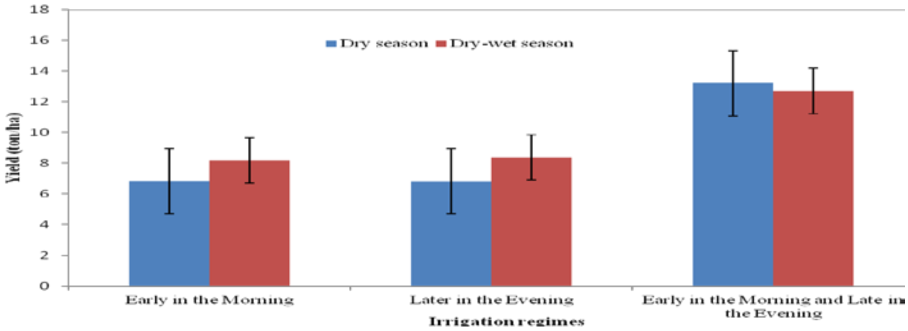

Figure2: Effect of different irrigation regimes on sorghum's grain yield production

The grain yields obtained from this study shows that a farmer could get 7.58, 7.59 and 14.68 ton/ha; times more by irrigating LE, EM, and ELE, respectively during the dry season. It also shows that if sorghum would have been supplemented with drip irrigation in LE, EM or ELE during the dry-wet season; a yield of 9.08, 9.31 and 14.11 ton/ha times more would have been obtained respectively.

CONCLUSION

In ASALs where water is physically not available, matching the crop varieties with effective drip watering regimes optimizes crop growth performance, WUE, and the sorghum`s grain yield. This sustains the scarce water resources and the environment while contributing to enhancing food security to feed the ever-increasing population under the challenging climate change and weather variability. By irrigating twice a day and not once, a farmer is likely to double the grain yield. The result clearly indicates that irrigating twice a day in dry season results in more grain yield than supplementing. However, to ensure maximum grain yield production the study recommends practicing both drip pure irrigation and supplementation dry-wet season. This will increase resilience to drought, stabilise food security and livelihood; and mitigate the climate change in the long run.

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3.1.P

FLOW VELOCITY OVER FROZEN AND NON-FROZEN SLOPES OF LOESS AND STONE MIXTURE

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SUMMARY

Soils in high altitude or latitude cold regions are commonly rich in stone fragments. Hydrodynamics of water flow over frozen and thawed stony slopes are the most essential parameters related to soil erosion and sediment transportation. This study used laboratory experiments to measure velocity of water flow over frozen slopes by electrolyte trace method under different hydrological conditions as determined by slope gradient and flow discharge rate. The experiments involved four flow rates of 1, 2, 4 ,8 L/min, four slope gradients of 5°, 10°, 15°, and 20°, and four stone contents on mass basis of 0%, 10%, 20% and 50%. Nine sensors were used to make measurements of flow velocity by tracing the solute transport process at the positions of 10, 100, 200, 300, 400, 500, 600, 700, and 800 cm from the upper stream where water flow was introduced into the rill. The measured flow velocities indicated that they were strongly correlated to stone content, flow rate, and slope gradient. Flow velocity increased with slope gradient and flow rate. The effect of stone content on flow velocity became stronger with the increase in slope gradient. The single-valued maximum flow velocity appeared at the stone content of 14.92% over gentle slope gradient of 5° and flow rate of 1 L/min over other slopes. Higher stone contents caused fast decline of flow velocity. Flow velocity over frozen slope was approximately 1.2 times of those over non-frozen slope. When the stone content increased from 0 to 20%. The flow velocities over frozen and non-frozen soil became gradually about the same with stone content of 50%. This study will help research and further understanding of the hydrodynamics and soil erosion as well as sediment transport behaviors of frozen hillslopes of different stone contents.

KEYWORDS: Flow velocity; Electrolyte tracer method; Frozen-stony soil; Stone content; Slope gradient; Flow rate

3.2.P

INNOVATION OF PERMEABLE GROUNDSILLS FOR LOCAL SCOUR CONTROL

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INTRODUCTION

Groundsill is one of the common practices implemented for torrent control. It serves the purposes of controlling longitudinal scour in stream channels as well as checking channel gradients. Nevertheless, groundsills often dissect aquatic habitat, and the unavoidable local scour (Lenzi et al., 2002; Guan et al., 2014; Ota and Sato, 2014) at the groundsill toe every so often endangers the safety of groundsill structure. Therefore, the objective of this study is to explore the feasibility of permeable groundsills in a laboratory setting under different hydraulics conditions while varying the structure permeability.

METHODS

This study was conducted in a 4-m long, 0.3-m deep, and 0.9-m wide semi-circular flume. A pair of revetment with sideslope of 1:0.3 (V:H) was installed in the flume to form a 0.3-m bottom width trapezoidal channel. Froude Number Similarity was followed while considering the availability of experiment sediment sizes and pump capacity so that 1:50 scale was therefore selected.

Groundsill models tested in this study included traditional and permeable design. Traditional groundsill model was made of solid plywood slab; whereas, permeable groundsill model was constructed by trimming the top rim section of a traditional groundsill then replacing with a permeable structure to form a Type A and Type B model (Fig. 1). Permeable Type A groundsill was made of beam frame filled with crashed stones in the frame cavity, which resulted an average 7% permeability by area. Type B groundsill was made of PE corrugated board to mimic a box culvert array, which resulted a constant 68% permeability by area.

Groundsill models were then installed in the flume with the interval of 0.2 m (Fig. 2) equivalent to 10-m prototype interval that commonly implemented for torrent control (Wu et al., 2002). Three channel slopes (4, 6, 8%), three flow rates (Model: 4.5×10^{-4}, 6.7×10^{-4}, 8.5×10^{-4} cms; Prototype: 7.95, 11.84, 15.05 cms), and three sediment sizes (Model: 0.85, 1.05, 1.45 mm; Prototype: 42.5, 52.5, 72.5 mm) that forming the channel bed were selected for the hydraulic experiments. Laser profiler was used before and after each experiment to scan the channel topography within the measurement section, and the channel bed elevation data were used afterwards to acquire the maximum scour depth, extent of the scour as well as the location where maximum scour occurred. Flow depths at various locations were also measured using linear displacement transducer during the experiment.

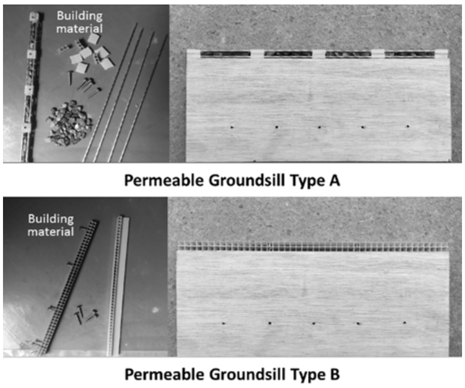


Fig. 1 Permeable groundsill models.

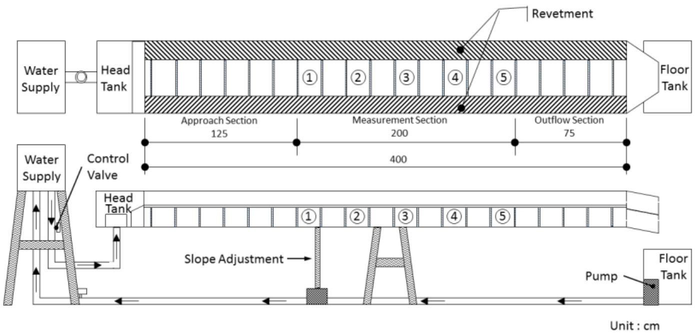


Fig. 2 Schematic drawing of experiment setup.

RESULTS

Main mechanics that causing local scour at the toe of groundsill was due to plunging flow that overflowing from the groundsill. Through flume experiments we found that the characteristics of local scour was controlled by (1) flow rate, (2) channel gradient, (3) sediment size which made up the channel bed, and (4) permeability of the groundsill protruding from the channel bed.

Hydraulic experiment results indicated that the maximum scour depth (y_m) increased as flow rates and /or channel gradient increased; whereas it decreased as sediment sizes increased. Similar results were also observed for average scour length that measured from the toe of groundsill to the downstream rim of the local scour. However, the flow regimes were different when permeable groundsills were involved.

When channel flow overflows from a groundsill, the plunging flow immediately scours the channel bed and two rotational flow circulating in opposite directions are formed inside the scour hole; one circulates toward upstream direction and the other circulates toward downstream direction (Bennett et al., 2000). The sediment transported by downstream rotational current will be either transported further downstream or deposited before the next groundsill and gradually forms a deposition dune (D'Agostino, 1994; Wu et al., 2014).

As protruding portion of the groundsill becomes permeable, the permeability of the structure dominates the local scour development until portion of permeable voids become partially or fully clogged by sediment. Experiment results also showed that permeable groundsills could effectively control the scour depth and the extent of scour. However, sediment from upstream channel was still trapped by the permeable structures but with less uniformity. Non-uniform deposition on the backside of the permeable groundsills provides a gateway for stream flow to penetrate, which changes the flow depths as well as affects the scour mechanics. Fig. 3 illustrated the diverse channel bed features under traditional, Type A, and Type B groundsills.

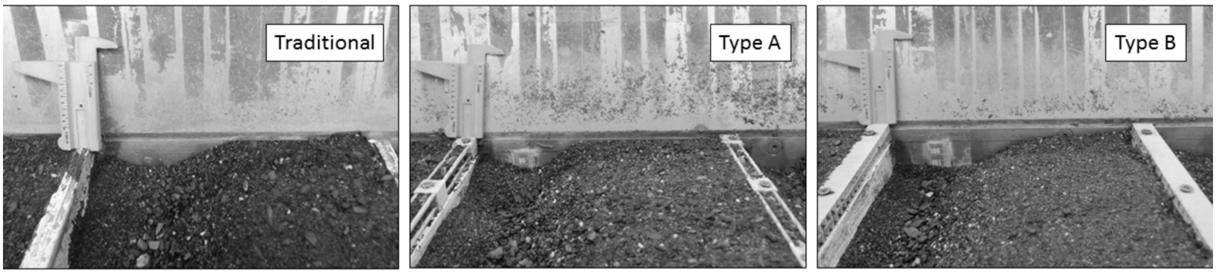
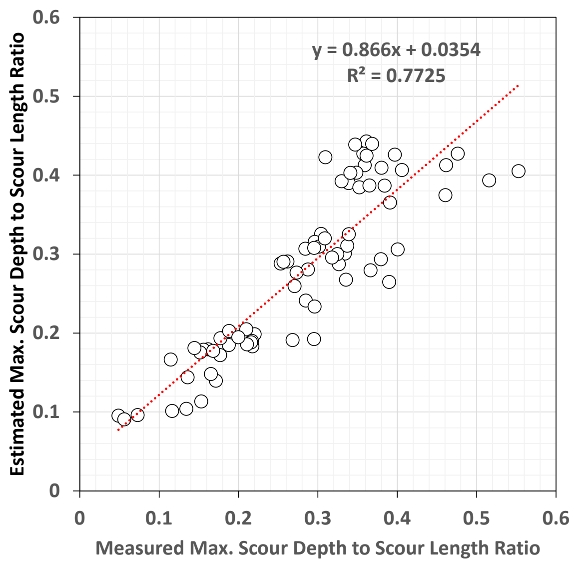


Fig. 3. Local scour at the toe of traditional, permeable Type A, and permeable Type B groundsills.

After eliminating outliers from the dataset, we first conducted log transformation to the remaining dataset, followed by multiple regression analysis on dimensionless Π terms, we finally achieved a dimensionless regression equation, which could be used to estimate the maximum scour depth to average scour length ratio,

$$\frac{y_m}{\lambda_s} = 4.95\frac{B_k^{0.56} \cdot S^{1.12} \cdot F_r^{0.11}}{R_R^{0.18}} \quad ; \quad \text{Adjusted } R^2 = 0.819 \qquad [1]$$

; in which, (y_m/λ_s) is the maximum scour depth to average scour length ratio; B_k is the residual groundsill impermeability (defined as 1.0 for traditional groundsill); S is channel gradient, F_r is Froude number at the rim of groundsill; R_R is the ratio of sediment size to the flow depth measured at the groundsill rim, which can be interpolated as resisting force of the sediment resting on unit channel bed area against the driving force of the flow that acting on unit channel bed area.



Measured maximum scour depth to average scour length ratio (y_m/λ_s) is plotted against the estimated ratio that calculated from Eq. [1] (Fig. 4). A linear regression line is then fitted and the coefficient of determination (R^2) reaches 0.7725, which implies that Eq. [1] provides adequate estimations of maximum scour depth to average scour length ratio that can be used to estimate how deep a groundsill has to be embedded into the channel bed.

Fig. 4. Estimated vs. measured scour depth-length ratio.

CONCLUSIONS

Within the scope of this study, we found that permeable groundsills – groundsills consist of permeable portion that protruding from channel bed, are capable of easing the scour extent and maximum scour depth, and they are capable of altering streamflow, which in turns produce diverse channel bed features.

Groundsills with high permeability, such as that made of array of box culverts, are not suitable to use in high gradient torrents because the mechanics of local scour in channel bed is changed from plunging impact to shooting flow. Low permeable groundsills are capable of reducing local scour depth and overflow depth at low flow rates and mild channel gradients until voids are gradually clogged by sediment. When clogging occurs, low permeable groundsills tend to behave similar to traditional groundsills except the local scour induced by plunging flow being less severe.

Permeability of the groundsill does affect the extent of local scour. However, we believe the characteristics of the permeable section, which affecting the throughflow, the clogging, and the damming of streamflow after clogging will play even influential role in checking the local scour.

ACKNOWLEDGEMENTS

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3.3.P

FILLING IN DAILY RAIN SERIES FAILURES BY THE USE OF MARKOV AND GAMMA DISTRIBUTION STOCHASTIC MODELING: A CASE STUDY FOR THE MIRIN LAKE BASIN / RS / BRAZIL

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INTRODUCTION

The Lagoa Mirim Watershed has a prominent role in terms of water resources management, since it is a cross-border basin between Brazil and Uruguay. Despite its importance, water-flow information are scarce, which complicates the design of hydraulic structures as spillways, drainage canals, as well as those related to soil and water conservation, among others (Teixeira et al., 2011). To get around this situation, hydrological models have been applied for rainfall-runoff transformation (Fiorio et al., 2012; Lalozaee et al., 2013). Among them, the Soil Conservation Service model (SCS, 1972) makes use of intensity-duration-frequency (IDF) relationships to estimate hyetograms and unit histogram for routing surface runoff.

Long series of rainfall data are available for the Lagoa Mirim watershed but with missing records, hindering estimation of the IDF relationships. As it is an extreme phenomenon, intense rainfall ought to be characterized by continuous and representative data series. This way, the longer the series used in the IDF equation, the greater the chance of contemplating extreme phenomena. In order to fill the missing values of daily rainfall and therefore increasing series size, Markov and probabilistic models can be used to generate synthetic series. The Markov process considers a minimum of two transition states, rainfall presence or absence and, which is quantified by a parametric distribution, such as Gamma or Exponential (Stern and Coe, 1984). The knowledge on rain behavior over time can be enabled by mixing observed data and those estimated by the above-mentioned stochastic process. This study aimed to check information gains when estimating rainfall intensity-duration-frequency relationships through the Markov chain and Gamma distribution, by comparing them with a daily rainfall disaggregation method. For that purpose, we used data gathered during N + M years, in which a two-state Markov chain is fit to N continuous years of daily data and used to fill M years of missing data for a few cities within the Lagoa Mirim Watershed.

MATERIAL AND METHODS

Daily rainfall data were taken from seven rain gauge stations located inside the Lagoa Mirim Watershed (88), which is inserted into the South Atlantic basin (8). The study area is located between parallels 31° 30' and 34° 30' Southern latitude and 52° 00' and 56° 00' Western longitude, with an area of around 62,250 km^2, of which 29,250 km^2 (47%) in Brazil and 33,000 km^2 (53%) in Uruguay. Table 1 shows information on the rain gauge stations provided by the *Agência Nacional de Águas - ANA* (National Water Agency), which comprises code, name and location, as well as geographic coordinates, altitude and data gathering period.

Table 1. Rain gauge stations used in the study.

Code	Name	Location	Latitude (S)	Longitude (O)	Altitude (m)	Period
3152003	Canguçu	Canguçu	31º24'16''	52º40'24''	400	1943- 2012
3152004	Cascata	Pelotas	31º28'00''	52º31'00''	224	1967-1981
3152005	Vila Freire	Pedro Osório	31º40'10''	52º46'22''	250	1977-2012
3152008	Granja São Pedro	Pelotas	31º40'08''	52º10'50''	3	1967-2012
3152010	Morro Redondo	Morro Redondo	31º38'00''	52º39'00''	245	1966-1981
3252003	Estação do Curtume	Rio Grande	32º26'00''	52º36'00''	4	1964-1976
3252008	Granja Santa Maria	Rio Grande	32º24'16''	52º33'21''	12	1966-2006

Watershed rainfall modelling by a two-state Markov chain, which defines the probability of one day being dry or rainy, depends singly on the previous day condition, wherein days are considered dry (0) or rainy (1) (Stern and Coe, 1984). Daily rainfall below 1 mm was regarded as dry day (Minuzzi and Lopez, 2014). Transition probabilities between dry and rainy conditions, i.e. P(0,0), P(0,1), P(1,0) and P(1,1) (BAÚ et al., 2013), were determined for an annual series, without considering monthly stationarity, since this study aimed to fulfill daily gaps and subsequently set an annual maximum daily rainfall series.

After estimating transition probabilities, 100 sequences of dry/rainy days were generated for the entire period where gaps occurred. From these sequences, we could estimate precipitation amounts on the rainy days using a two-parameter Gamma distribution (Detzel and Mine, 2011), whose parameters were set by moment method. Once missing data were fulfilled, continuous series of annual maximum daily rainfall data were established for each of the seven stations. For this purpose, the Gumbel's theoretical probability distribution was used in association with return periods of 5, 10, 20, 50 and 100 years (Quadros et al., 2011). Then, this annual maximum daily rainfall was disaggregated into periods of 10, 20, 30, 40, 50, 60, 120, 360, 720 and 1440 min by relation methods (Teixeira et al., 2011). From the disaggregated data transformed into average maximum rainfall intensity, IDF relationship parameters were adjusted (Michele et al., 2011), minimizing the objective function $(fobs - fmod)^2$ using the Excel function Solver with a non- linear optimization code, known as Generalized Reduced Gradient.

The results were validated by following these items: a) N-year IDF relationships of disaggregated daily rainfall in hourly and sub-hourly durations were compared with those of PRUSKI et al. (2006), considering their closeness (Ludwig et al., 2013). The null hypothesis stated that maximum obtained values do not have significant differences at a 5% level. Therefore, we used the Student's t test with n- k degrees of freedom, in which n is the sample size and k is the number of explanatory variables for linear (β_0) and angular (β_1) coefficients. The hypothesis is accepted when the Student's t test is less than its critical value. Moreover, standard error was estimated for maximum values within established return and duration periods; Moreover, standard error was estimated for maximum values within established return and duration periods; b) N-year IDF relationships of disaggregated daily rainfall in hourly and sub-hourly durations were compared with those obtained within N+M years. Likewise, t test and estimate standard error were used; c) The method was considered valid for low standard errors when comparing maximum rainfall obtained from a series of N+M years with those of N years. It is noteworthy highlighting that in an item, we singly validated the daily rainfall disaggregation method for IDF relationship estimation; whilst in b, besides that, we also validated the use of Markov chain and Gamma distribution.

RESULTS AND DISCUSSION

Table 2 presents the transition probabilities values P (0,0), P(0,1), P(1,0) and P(1,1) of Markov chain used in simulation sequences for the seven stations in the Lagoa Mirim Watershed, Rio Grande do Sul, Brazil. P (0,0) ranged from 0.76 (3154004) to 0.84 (3152008 and 3252003), while for P(1,1) the amplitude was between 0.36 (3152008) and 0.54 (3152003 and 3152004). The highest values of P (1,1) were registered in 3152003, 3152004, 3152005 and 3152010 gauge stations, which are at a higher altitude (400, 224, 250 and 245 m, respectively). Conversely, the lowest values were found in stations at lower altitudes as 3152008 at 3 m, 3252003 at 4 m, and 3252008 at 12 m. Mean values of annual rainfall in Pelotas and Rio Grande (RS), Brazil, which are located in the coastal plain, were 1379 and 1207 mm, respectively; while in the other locations within the southeastern mountain slope, these annual values were 1552 mm (Canguçu), 1480 mm (Morro Redondo) and 1562 mm (Pedro Osório).

Table 2. Transition probabilities P(0,0), P(0,1), P(1,0) and P(1,1) of Markov chain and Gamma distributions parameters of shape (α) and scale (β), from N years, for the seven stations in the Lagoa Mirim watershed, Rio Grande do Sul, Brazil.

Station	Transition probabilities				Gamma distribution parameters	
	P(0,0)	P(0,1)	P(1,0)	P(1,1)	α	β
3152003	0,79	0,21	0,46	0,54	0,39	8,36
3152004	0,76	0,24	0,46	0,54	0,89	9,34
3152005	0,80	0,20	0,49	0,51	1,09	9,30
3152008	0,84	0,16	0,64	0,36	1,38	8,45
3152010	0,79	0,21	0,52	0,48	0,88	9,06
3252003	0,84	0,16	0,57	0,43	1,21	7,13
3252008	0,83	0,17	0,55	0,45	1,53	8,77

Larger rainfall volumes occurred at higher altitudes. Back et al. (2012) determined relationships among rainfalls of different time lengths in Santa Catarina state (Brazil) and noted a relief contribution on their distribution in different areas of the state, being most abundant in areas near mountain slopes, since moist and warm air raise favors large rainfall volumes. Gamma distribution parameters of shape (α) and scale (β) used to estimate daily precipitated water depth is shown in Table 2.

CONCLUSIONS

Stochastic modeling using a homogeneous first order Markov chain showed to be adequate to estimate sequences of dry and rainy days. The statistics values of observed daily rainfall series were preserved when it was used the Gamma probability distribution to simulate the amount of rainfall. The daily rainfall disaggregation technique presented a good performance, composing a feasible alternative for estimations of rainfall intensity-duration-frequency relationships. Mostly important is that there was information gain on intensity-duration-frequency relationships by using N+M years of disaggregated rainfall daily data compared to N years.

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3.5.P

RESPONSES OF INFILTRATION UNDER STRAW-MAT MULCH AND CARPET GRASS COVER

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INTRODUCTION

Surface mulch and ground vegetation have been commonly implemented to prevent soil erosion and control soil loss owing to the surface protection as well as runoff and sediment retention they provide. Past researches have tried to cast light to the effect of ground cover and residue mulch on infiltration processes (Mupangwa et al., 2007; Chakraborty et al., 2008; Jordán et al., 2010; Wu et al., 2011), and physical processes-based simulation models have also tried to estimate surface runoff by various means. Taking Soil and Water Assessment Tool (SWAT) as an example, SCS Curve Number and Green and Ampt Equation were used to estimate the volume of surface runoff (Neitsch et al, 2011). Due to complexity of the infiltration process and that of the surface boundary conditions when surface treatment is involved, responses of soil water content under either practice still require further investigation.

METHODS

Total of three standard USLE (Universal Soil-Loss Equation) runoff plots were prepared for this study. Each runoff plot measured 22-m long, 3-m wide and was manually levelled to achieve a uniform slope of 9%. Preceding vegetation and residue was first removed prior to plot preparation. Soil crumps were then smashed, sieved, and inter-exchanged between plots to minimize the initial difference caused by prior land use.

Fig. 1. Layout of the study runoff plots.

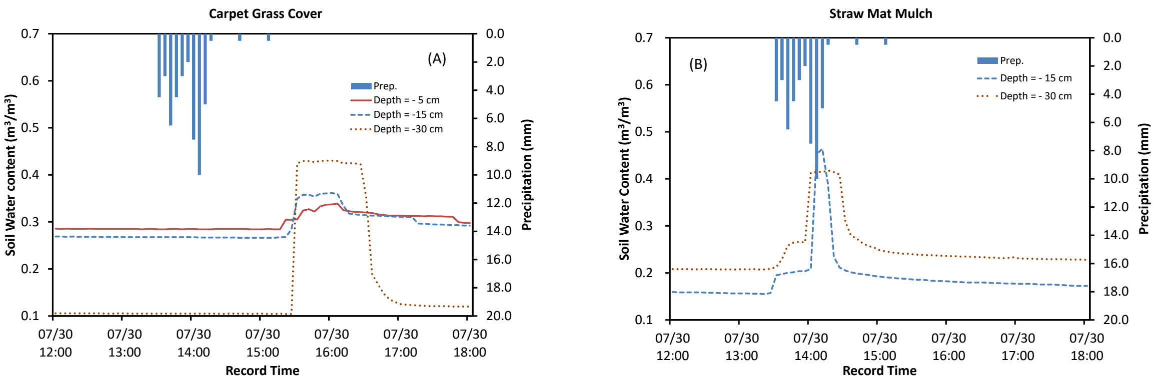
Carpet grass (*Axonopus affinis*) and straw mats were selected along with a bare runoff plot as the control. Carpet grass was transplanted from plant nursery; whereas straw mats with the specification of 350 g/m^2, 1-m wide x 100-m long/role were purchased from factory then installed onsite.

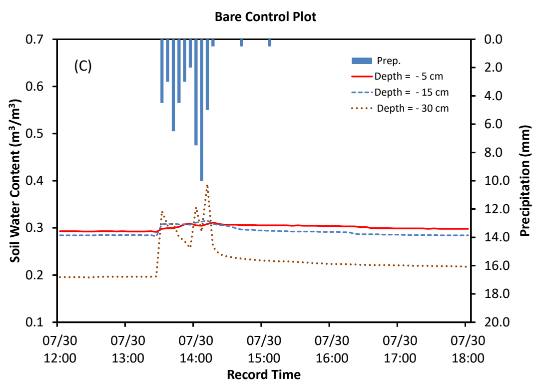
Soil water content at the depth of 5, 15, and 30 cm was monitored continuously using soil moisture sensor. Rain gauge with 0.2-mm resolution was used to document the rainfall events. Data loggers were attached to soil moisture sensors and rain gauge respectively. The study period covered the entire rain season in 2016, during which straw mats were replaced entirely once on July 22, weeds were manually removed constantly, and othorphotos were taken every 7 days to chronicle the surface coverages.

RESULTS

We finished runoff plot preparation on January 11, started monitoring soil water contents on April 29, and terminated field monitoring on the last day of 2016. Total of 35 rainfall events were recorded, in which, 26 of them (intermittent storms) produced single peak in soil water histograms with apparent delays as compared to rainfall hyetographs. Rest of the rainfall events (continuous storms) produced multiple peaks in soil water histograms that almost synchronized with rainfall hyetographs.

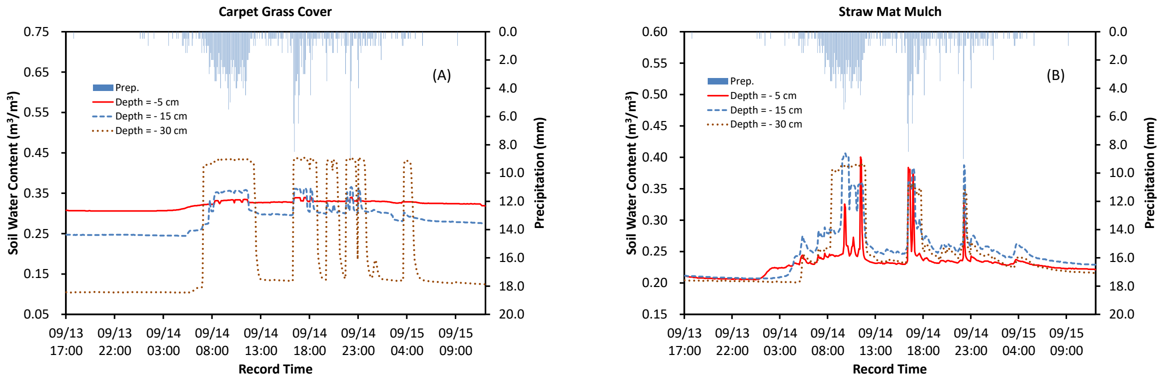
Time-to-peak of soil water content, defined as the time between the beginning of rainfall event and peak soil water content, as well as peak soil water content were found to be affected by surface treatments and time intermission between current and antecedent rainfall event. Field observations indicated that time-to-leak of soil water content at all monitoring depths under Carpet grass treatment tended to delay the most as compared to that under straw-mat protection and bare control plot when intermittent storm occurred (Fig. 2). Shorter the intermission between storms, faster the soil water response time that Carpet grass reacts.

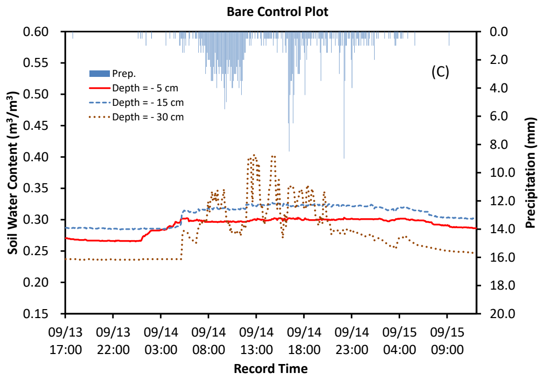




When intermittent storms occur, Carpet grass tends to intercept rainwater until its retention capacity is filled. Straw-mat mulch behaves similarly but with less retention capacity, and its rainwater retention capacity decreases as degradation process proceeds.

Fig. 2. Responses of soil water content under intermittent storm.





When continuous storm occurs (Fig. 3), bare control plot still leads in soil water response, followed by straw-mat mulch and Carpet grass cover. Soil water contents at depths of 5 and 15 cm (Fig. 3(C)) recorded at bare control plot remain steady regardless of rainfall intensities.

Fig. 3. Responses of soil water content under continuous storm.

Soil water responses under straw-mat mulch behave differently (Fig. 3(B)) as if soil water contents at three monitoring depths synchronize with each other, particularly during the course of continuous storms if and only if the storms provide ample rainwater to fulfill retention capacities of the straw-mat mulch as well as that of the top 5 cm of the soil matrix. However, an apparent delay in soil water content was found between -5 cm and -15 cm in the beginning of the storm, and this time lag again decreases as degradation of straw-mat proceeds.

Soil water content recorded from all 35 rainfall events also indicated that Carpet grass cover provided the highest soil water content among all surface treatments at the depth of 5 cm. Furthermore, when accumulated precipitation exceeded 7.5 mm and 5-min precipitation exceeded 3.0 mm, soil water content at the depth of 15 cm quickly surpassed that at the depth of 5 cm but immediately dropped when rainfall ceased. On the other hand, soil water content at the depth of 15 cm remained higher than that at the depth of 5 cm for straw-mat mulch and bare control plot.

Desorption rates of soil water content were also investigated, and we found that desorption rates tended to be slow for bare soil when the intermission time exceeded 17 hr and 50 min. We believe this was mainly due to the ‘ink-bottle’ effect (Hillel, 1982). Field observations also indicated that soil water desorption rates at all depths for straw-mat mulch was affected by 10-min accumulated precipitation before the arrival of peak soil water content. Less the accumulated precipitation, slower the desorption rate appears when soil is protected by straw-mat mulch.

CONCLUSIONS

Findings from this study provide the evidences that soil water response is not only control by the soil matrix, but also affected by the characteristics of surface treatments, length of intermission duration, and storm patterns – intermittent versus continuous. Different surface treatments have different rainwater retention capacities. Additional soil water passages produced by vegetation can also change soil water responses. Over simplifications of such complex phenomenon may affect the estimation of runoff volume as well as the runoff occurrence time, which may not be accurately conveyed by either SCS Curve Number, Green and Ampt Equation or other infiltration theories.

Further researches are still needed to address the dynamic contributions of live vegetation and bio-residue to soil water responses; particularly surface treatments' rainwater retention capacities as well as the interactions between retention capacities and ground coverages that change within the course of growth or decomposition of the vegetation cover and residue mulch.

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3.6.P

HOW ADD-INFLOW AND SUBSURFACE DRAINAGE AFFECTING GULLY EVOLVEMENT

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INTRODUCTION

Processes involved in gully evolvement include downward scour in gully channel, headward retreat at gully headwall, and lateral expansion of the gully channel. These processes have often been related to threshold velocity and threshold shear stress of the flow, which also have been correlated with drainage areas and drainage slopes (Montgomery and Dietrich, 1994; Vandaele et al., 1996). Other processes such as soil piping (Valentin et al., 2005; Moges and Holden, 2008) and subsurface seepage (Scholten, 1997) have been addressed as conditional processes when the desirable environment was met. As rills gradually develop into ephemeral gullies then into permanent gullies, different erosion processes take place (Laflen et al. 1985). The objective of this study is to inspect the evolvement of a pre-existing gully in a laboratory settings and to identify the contributions induced by add-inflow and subsurface drainage.

METHODS

Laboratory experiments were conducted in a 2.2-m long, 0.6-m wide, and 0.16-m deep soil box (Fig. 1) packed with loamy soil. A trapezoidal trench with the same channel gradient as the soil box but top trench width taken up 50% of width of the soil box was inscribed into the soil box before the experiment to mimic a pre-existing gully. Light sprinkle at 6.3 mm/hr was applied continuously to gently wet the soil surface but not forming indentations and micro-rills yielded from drop impact.

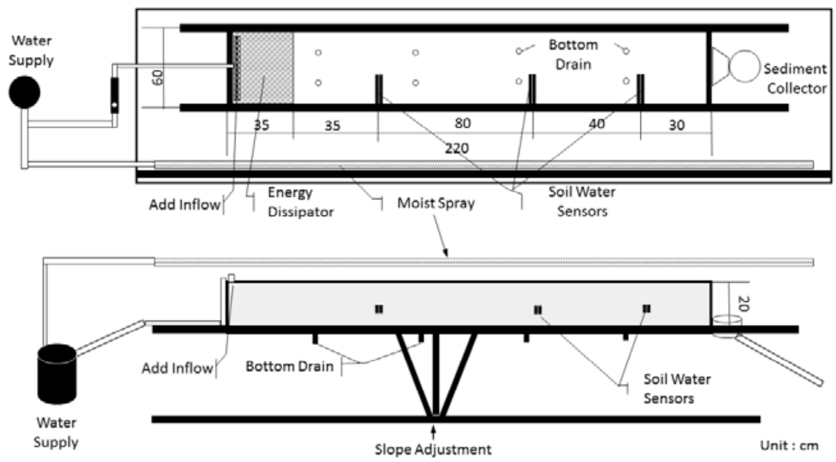


Fig. 1. Schematic drawing of experiment apparatus layout.

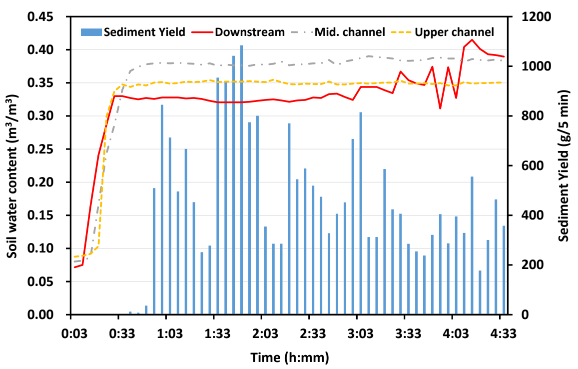
Three add-inflow rates (0.0, 0.5, and 1.0 L/min), three slope gradients (5°, 7.5°, and 10°), and bottom drains in either close or open positions constitute the combinations of experimental conditions.

Soil was first air dried, sieved, and packed into the soil box by layers to achieve an average bulk density of 1,300 kg/m^3 and 1,500 kg/m^3 for the top 11 cm and rest of the 5 cm soil depth respectively. Sensors were embedded during the packing process to continuously monitor volumetric soil water contents at upper-, mid-, and lower section of the pre-existing gully channel, and the insertion depth was 4.5 and 12 cm from the bottom of the gully channel and soil surface respectively. Add-inflow was added to the soil box for add-inflow runs when volumetric soil water content, which was approximately 33% for the test soil and 40 to 50 minutes into the runs, became steady.

Surface profiles were scanned across the soil box prior to, every 40 minutes into, and at the end of experiment using a 75µm-resolution laser scanner at the pre-selected cross sections. Digital elevation models (DEM) were constructed to obtain the rates of gully headwall retreat as well as that of sidewall expansion. Sediment sample at the outlet of the gully was taken continuously in 5-min interval. Locations as well as the extents of sidewall slump were also documented during the experiment.

RESULTS

In spite of experiment conditions, laboratory observations indicated that slump along gully channel sidewalls dominated gully lateral expansion in the initial evolvement stage, and the occurrence of sidewall slump was found in this study to be related to the volumetric soil water content when water content reached 62.1% ~ 98.7% and 65.7% ~ 99.3% of the maxima values under drain-disabled and drain-enabled condition respectively. Sediment as a result constituted a major portion of the total sediment yield as illustrated in Fig. 2 for channel gradient at 10°, drain-disable, and no inflow.



During the initial stage of runoff occurrence, headcut appeared on the gully bed and slowly migrated upstream. Video footages suggested that shearing action of the runoff dominated the headcut process while cohesive soil on the channel bed gradually losing its strength.

Fig. 2. Sediment yield from gully outlet.

Since soil maintained its cohesive strength, therefore, headcut process on the gully bed halted until either soil lost its strength or fine-grain fraction of the soil reached liquid limit and forced soil skeleton to crumble. In result, headcuts on the gully bed occurred in patches.

Soil mass resulted from sidewall slump often partially blocked the passage of flow when runoff became concentrated in the gully channel, that not only altered the flow path but also changed the flow transport capacity. Some sediment was therefore deposited while some of the soil mass from sidewall slump that previous dammed in the gully channel was either dislodged as non-cohesive aggregates or eroded away yet remaining the integrity. The result of selective deposition and erosion altered the flow path and gradually created a meandering pattern in the gully channel. Meander of the concentrated flow cut the outer bank toe and further initiated the expansion of sidewall slump. Similar processes can be found in rill development (Wu et al., 2011). Fig. 3 illustrated the evolvement of a pre-existing gully when channel gradient was set at 5° with 0.5 L/min add-inflow and subsurface drain disabled.

Scour caused by plunging flow did not occur at the gully headwall in this study regardless the conditions of add-inflow. However, shearing action of the surface runoff flowing over the gully headwall appeared to dominate process during headwall retreat. Piping due to seepage occurred only once in the study as that shown in Fig. 4. The center of the circle in Fig. 4 was the place where seepage emerged.

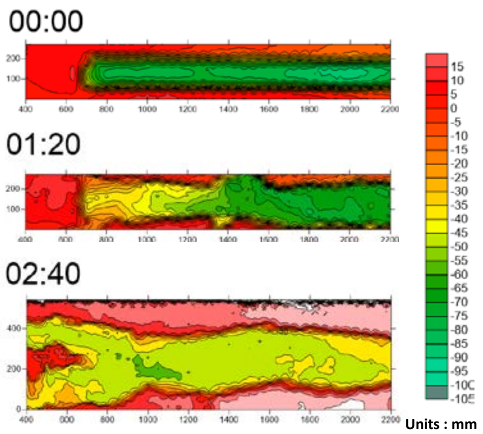


Fig. 3. Evolvement of gully channel.

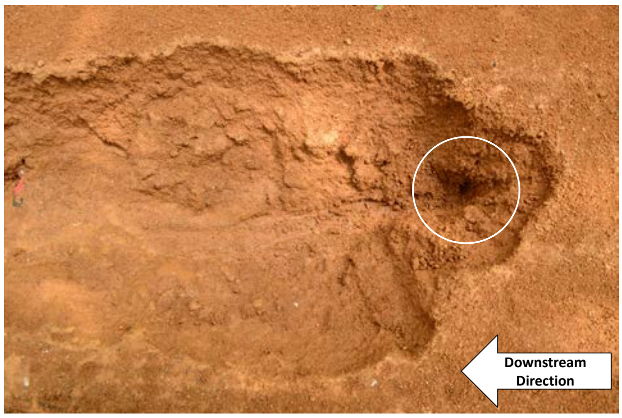


Fig. 4. Seepage (center of the circle) causes erosion.

Preliminary results from this study did not provide enough evidence to support the contribution to gully evolvement caused by subsurface drainage. The main reasons were due to limited raise of saturation zone as well as domination of gully sidewall slump during the course of experiment. Additional investigations are needed to further examine the rates of gully expansion and gully incision through detailed analysis of the DEM data generated in the experiments.

Add-inflow, on the other hand, accelerated gully evolvement by providing additional shear forces to undercut gully sidewalls and channel bed. A train of small headcuts was found in gully channel when channel bed was submerged in the concentrated flow. However, the frequency of headcut occurrence, for the cases of 7.5^o channel gradient, revealed higher frequency of 0.569 at upper gully channel, followed by 0.316 at mid-section, and 0.114 at lower gully channel. The rate of retreat could reach as high as 0.30 cm/s and as low as 0.18 cm/s. The mechanics involved in the channel bed headcut retreat is still under investigation.

CONCLUSIONS

This study was conducted in a laboratory setting with a trench inscribed into the soil bed to mimic a pre-existing ephemeral gully. Plunging-flow scour did not occur at the toe of the gully headwall, but sidewall slump along the gully as well as hydraulic erosion by concentrated flow dominated the gully evolvement.

The rates of add-inflow accelerated gully lateral expansion with the assistance from channel gradients. The addition of add-inflow helped accelerate hydraulic shearing action at the gully headwall but not as much at channel incision processes. It also created additional seepage to degrade gully development. However, subsurface drainage condition played less evident role in this study.

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3.7.P

CONSERVATION AGRICULTURE IMPLICATIONS FOR SOIL WATER BALANCE. A MODELLING APPROACH

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INTRODUCTION

Conservation agriculture (CA), defined as the integration of minimum soil disturbance, permanent mulch ground cover and crop rotation, offers opportunities to reducing the risk of soil erosion. On the other hand, soil compaction and the presence of a mulch layer, which reduces soil temperature, may limit crop establishment, particularly in Mediterranean spring crops. In general, CA improves water infiltration into the soil but the benefits on water conservation and on crop performance, particularly in irrigated Mediterranean conditions, are not clear. Processes involved in the water balance under CA include residues capacity to intercept water and the evaporation of this water, soil water infiltration, soil water storage capacity, and soil water percolation, evaporation and runoff. The aim of this study is to review the scientific literature for deepening on how CA affects these processes in irrigated annual crops-based systems and how three major crop models capture these effects.

METHODS

We have analysed three major crop growth models (CropSyst, DSSAT-CSM and STICS) for which the effect of different CA management practices can be simulated, via the effect of tillage and/or of maintaining residues on the soil surface. In particular, we have studied how these practices affect the water balance and how relevant mechanisms are handled. Main studied mechanisms were residue water interception, residue evaporation and subsequent effect on soil evaporation, residue effect on runoff and tillage effects on soil physical properties. We did not include water infiltration for comparison as the three models use the same cascade approach. Subsequently, we conducted a literature review of the application of these models in which CA practices were fully or partially used, to outline strengths and limitations for each of the models to simulate CA effects.

RESULTS

In the three models, water intercepted by residues is a function of their biomass, although for CropSyst, water holding capacity or wettability is assumed to be 4 kg of H_2O kg residue^{-1}, while for DSSAT-CSM and STICS it is a parameter that can be input being similar the maximum values.

The three models use Ritchie's two-stage evaporation approach (1972) and routines related to residues influence in soil evaporation are similar. The water that is intercepted by crop residues is prone to evaporate at the potential rate, thus it is subtracted from the potential soil evaporation. For CropSyst, water evaporates at the potential rate as long as residue water content is higher than its maximum water holding capacity; below that threshold, evaporation is proportional to the ratio of residue water content to residue water holding capacity. For STICS, residue water evaporates at the potential rate as long as the residue water content is higher than the maximum water holding capacity of the crop mulch, which is dependent on the amount of residue; below that threshold, residue water does not evaporate. DSSAT-CSM uses a similar approach to STICS for calculating potential residue evaporation, but limits maximum residue water that can be evaporated to 85% of the total mulch water stored on any given day (Table 1). After mulch evaporation is subtracted from the potential soil evaporation, the second evaporative stage (limited stage, independent of mulch) takes place for computing daily soil evaporation.

Table 1. Factors that affect the mulch evaporation (EM) mechanism

CropSyst	DSSAT-CSM	STICS
- Reference potential ET (ETs) - Mulch cover (Mc) - Mulch quantity (Qm) - Conditions ▪ If mulch water (θm) > Max. water holding capacity of mulch (θmSat) → EM ▪ If θm < θmSat → EM proportional to θm/θmSat	- ETs - Mc - Qm - Conditions ▪ EM up to 0.85 x θm limit	- ETs - Mc - Qm - Conditions ▪ EM up to θmSat limit

CropSyst and DSSAT-CSM use the SCS Curve Number approach (USDA-SCS, 1988) to account for runoff. Cropsyst does not consider residue mulch for runoff beyond how crop management affects the surface retention factor (the threshold of maximum soil moisture retention before runoff begins). Additionally, CropSyst takes residue effect into account for its soil erosion routine, the crop management factor for the Revised Universal Soil Loss Equation (RUSLE). DSSAT-CSM modifies the procedure by making the initial abstraction ratio within the runoff equation a variable dependent on mulch characteristics. STICS has its own algorithm to calculate surface runoff, although uses a few parameters based on the SCS CN approach. Mulch comes into play after runoff is activated by passing the runoff threshold, then its quantity will determine the degree of runoff, depending on another mulch quantity threshold to cancel runoff that ultimately depends on mulch characteristics (Table 2).

Table 2. Factors that affect the surface runoff (R) mechanism

CropSyst	DSSAT-CSM	STICS
- SCS Curve Number approach (SCS CN) - Precipitation (P) - Surface retention factor (S) - Initial abstraction ratio (rIA) fixed as 0.2	- SCS CN - P - S - rIA variable: ▪ Cumulative to previous values ▪ Maximum and minimum values ▪ Mulch cover (Mc) ▪ Mulch quantity (Qm)	1st process related to S: - If P – stem flow (SF) > S → R 2nd process related to Qm: - If Qm < 0.1 t ha^{-1} → R as in bare soil (SCS CN) - If 0.1 < Qm < Maximum quantity of mulch to nullify R (Qm0) → R proportional to 0.33 x (Qm0 – Qm) - If Qm > Qm0 → R = 0

For the three models, tillage mainly affects surface residue redistribution within the tillage depth, where it mixes with the current soil organic matter pools and alters residue decomposition rates due to changes in moisture and temperature. As for physical properties, for CropSyst, tillage affects some parameters that govern RUSLE, as with surface residue effects. STICS has some routines in development that can potentially alter water infiltration and redistribution, such as changes in structure, compaction via machinery passes and surface fragmentation. DSSAT-CSM is the only model for which tillage alters physical properties such as bulk density, soil layer thickness and water holding capacity.

Few studies have attempted to validate CropSyst performance on different residue rates on relevant water balance parameters. Díaz-Ambrona et al. (2005) and Monzon et al. (2006) found that simulated yield, soil moisture content and other water balance outputs were in agreement with monitored values. Pannkuk et al. (1998) found water balance variables were well simulated under CA, but in terms of yield and evapotranspiration only found a good agreement conventional tillage and no residue removal and no-tillage and residue removal. Sommer et al. (2007) found that the best model calibration for conservation tillage led to problems concerning surface runoff being higher with no tillage and residues, although modelled soil water content and soil evaporation were accurate in conservation agriculture, whereas Cantero et al. (2016) found that when residues were incorporated, cumulative soil evaporation was overestimated.

Performance of DSSAT-CSM has proved to be reasonably accurate in regards to water balance and yield predictions under conservation agriculture (Devkota et al., 2015, Corbeels et al., 2016), although sometimes yield and soil moisture content were overpredicted (Liu, 2013). In general, not many studies evaluate conservation agriculture as there is a lack of residue rate experiments. There are, however, many tillage experiments conducted for which DSSAT-CSM performed accurately in terms of yield and water balance components.

STICS has been extensively applied on commercial farms for evaluations and simulations, however, there are few examples in which STICS details conducted experiments that concern residues or tillage management practices. Scopel et al. (2004) developed the mulch module for STICS and evaluated it for two field experiments in Mexico and Brazil, with maize and millet crops. STICS correctly modelled aboveground biomass and yield, although water balance yielded mixed results, overestimating water conservation with mulch and also mulch water evaporation. In general, STICS performs well in modelling the water balance, yield and yield components on field-scale conditions and fallows, but lacks in-depth studies on residue rates and tillage regimes.

CONCLUSIONS

Mixed results concerning water balance parameter evaluation for CropSyst might denote a performance issue when both tillage and residue management interact with each other. The runoff approach in CropSyst is limited for CA and so far the water balance is unaffected by tillage effects, which also happens for STICS, although tillage mechanics are being improved for the latter. This implies there is room for improvement in these two models regarding the pointed out issues. DSSAT-CSM strength lies in its robust management module that handles tillage along with residue management and other components, that interacts quite well with the crop module, making it robust in yield prediction. Unfortunately there are very few in-depth residue studies to point out its actual limitations, and STICS seems to follow that trend too, despite of being quite robust at field-scale. This means that DSSAT-CSM and STICS need to be further evaluated for conservation agriculture in their in-depth water balance mechanics.

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3.8.P

ORIBATID MITES IN DRYLAND SYSTEMS: EFFECTS OF FERTILIZATION PRACTICES

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INTRODUCTION

Fertilization of crops with animal residues, widely used in dryland Mediterranean agriculture, affects soil health. Soil health is defined by Doran and Zeiss (2000) as the capacity to function as a vital living system, within ecosystem and land-use boundaries, to sustain plant and animal productivity, to maintain or to enhance water and air quality, and to promote plant and animal health.

To evaluate the effects and sustainability of agricultural practices we must determine soil health using indicators of soil quality. Soil organisms and biotic parameters are useful indicators of soil quality because they respond sensitively to land management practices and weather conditions: in this context, soil invertebrates such as the microarthropods are studied (Doran and Zeiss, 2000; Kautz, et al., 2006; Bosch-Serra et al., 2014)

Soil invertebrates, including oribatids (microarthropods), have direct and indirect effects in the physical, chemical and biological soil processes such as nutrient cycling, carbon transformation, formation of biostructures and soil humus, pore networks and fragmentation of organic materials to increase their availability for microorganisms (Coineau, 1974).

Oribatid mites depend on the input of crop and root residues or organic manure for their food, i.e. they are sensitive to the amount and the quality of the organic input (Kautz, et al., 2006).

The application of animal residues as organic fertilizers and the tillage system used has an impact on composition and abundance of oribatid species (Bosch-Serra et al., 2014). However, Andrés et al. (2011) reported that freshly digested sludge decreased the numbers of oribatid mites.

Oribatid mites can be used as biological indicators of soil disturbances associated with agricultural management including fertilization practices. In semiarid environments, there is a lack of available information on the use of Oribatida as bioindicators especially in land management that includes pig slurry and sewage sludge as the main organic fertilizers.

The objective of this study was to determine the impact of long-term organic fertilization on diversity and abundance of oribatid mites.

METHODS

Description of the study area and experimental design

The experiment was established in a dryland agricultural system in the 1997/98 cropping season in Agramunt (Lleida, Spain). Barley and wheat were the main winter cereal crops.
The soil of the site was classified Typic Xerorthent(Soil Survey Staff, 2014), with a loam texture (clay: 18%, silt: 44% and sand: 38%) in the surface layer.
The annual average temperature is 14.0°C and the annual average precipitation is 563 mm. The weather conditions during the crop season (2015/16) when the samplings were done, are shown in Fig. 1.

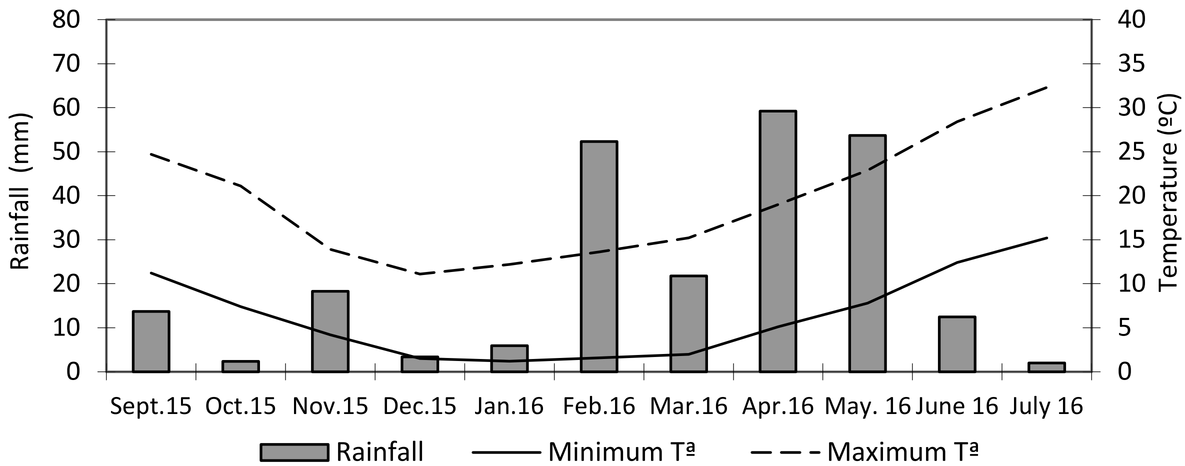

Figure 1. Monthly rainfall and minimum and maximum temperature during the 2015-2016 cropping season, from an automatic meteorological station located 12km from the experimental field (Climate station: Tornabous). Source: www.Ruralcat.net

Five fertilization treatments were distributed in three randomized blocks. The treatments were applied annually, just before sowing. They were established according to the N dose: mineral (MI) 80 kg N ha^{-1}, pig slurry (PS) 125 kg N ha^{-1}, sewage sludge 1 (SS1) 125 kg N ha^{-1}, sewage sludge 2 (SS2) 250 kg N ha^{-1} and control (CO) with no N.

Sampling, extraction and taxonomic identification of mites

Oribatids were sampled from the first 6 cm soil depth, in a total volume of ~400 cm^3 per plot. Samplings were done three times during the cropping season: November 2015, January 2016 and April 2016. In the laboratory, the oribatids were extracted with a modified Berlese-Tullgren funnel and stored in 70% ethanol. The adults were identified at species level using several taxonomic keys.

RESULTS

The maximum average of individuals was found in January (28,108 individuals m^{-2}) which was double the number for April.

Fertilization with sludges reduced their numbers when compared with PS and CO. Sixteen oribatid species were observed. Plots fertilized with PS achieved the greatest diversity. The main species present were *Acrotritia hyeroglyphica* (Figure 2a), *Scutovertex sculptus* (Figure 2b) and *Passalozetes* (P.) *africanus*, the latter being the dominant species for all plots and sampling times (Figure 2c).

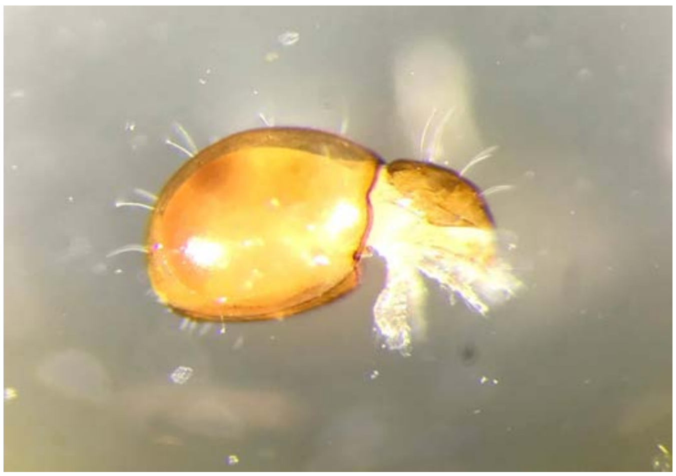

Figure 2a. *Acrotritia hyeroglyphica* (Berlese, 1916), Oribatid species mainly present in plots fertilized with sewage sludge. Average length of aspis: 260 µm.

Figure 2b. *Scutovertex sculptus* (Michael, 1879). Oribatid species mainly present in plots fertilized with pig slurry. Size: 486-660 µm x 345-410 µm, (Pérez-Iñigo, 1993)

Figure 2c. *Passalozetes (P.) africanus* (Grandjean, 1932) Dominant species. Size: 230-285 µm x 115-135 µm (Pérez-Iñigo, 1993)

CONCLUSIONS

Long-term fertilization with PS and sludges affects the diversity and abundance of the oribatid community in a dryland system. The main species present indicate that the system is under great risk of desertification.

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3.9.P

EFFECT OF SOME POLYMERIC MATERIALS ON RUNOFF AND SEDIMENT QUANTITY GENERATED FROM TYPIC XEROCHREPT DEPENDING ON INITIAL AGGREGATE SIZE UNDER SEQUENTIAL SIMULATED RAINFALL

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INTRODUCTION

Earth is under global change. Nowadays, scientific studies in attempt to protect and manage of natural sources, are planning and implementing around the understanding of processing mechanisms of events that takes part in this change. Thus, studies about mechanism of soil erosion are of great importance. At the beginning of the factors affecting water erosion is the erodibility of the soil as well as the rainfall erosivity (Morgan, 1996). For this reason, integrated studies in which soil conservation practices are based on the erosivities of precipitation are more meaningful in combat to water erosion. Polyacrylamide (PAM) and polyvinyl alcohol (PVA) have been used for erosion studies since 1950s (Blanco-Canqui and Lal, 2008). It is emphasized as a general result that even when the polymers are applied to the soil surface at low doses they may have significant positive effects on the improvement of aggregate stability and structural integrity (Gabriels, 1990; 1994; Sojka and Lentz, 1994; Amezketa, 1999; Imbufe et al., 2005). In order to obtain the most effective result from polymer application the most appropriate polymer type, application form, and dosage must be determined for each soil. It should be kept in mind, however, that the effectiveness in soil of the polymer applied to combat erosion is closely related to the dynamics of aggregate disintegration, which is highly influenced by the physical properties of erosive force. The objective of this study was to investigate the effects of polyacrylamide (PAM) and polyvinyl alcohol (PVA) applied to aggregates with different sizes of Typic Xerochrept soil on the runoff and soil loss under the sequential simulated rainfalls.

METHODS

In this study, different sized aggregates (< 1, 1-2, 2-4, 4-6.4, >6.4 mm and mixed all aggregates) of an Inceptisol were used. Some properties of Typic Xerochrept were shown in Table 1. Aggregates with diverse sizes of the soil were put in small erosion pans. Polymers were applied on these air dry aggregates with 6.25 kg ha^{-1} dose as solutions. After 24 hours waiting time, the pans were placed on 15 % slope, and simulated rainstorms with 36 cm h^{-1} intensity were applied by using Eijkelkamp mini rainfall simulator for a duration of 12 minutes. This intensity and shower duration were selected for the best representing of kinetic energy per unit per unit drop impact area of natural rains. Many researchers used Eijkelkamp mini rainfall simulator have used the same intensity and different rain duration values (Martin et al., 2010; Nciizah and Wakindiki, 2014). Runoff starting time (RST), runoff quantity (RQ), sediment quantity transported by runoff (SQTR), and transported soils to two sides of pan by splash (SLS) were measured as variables. Two days after first rainfall application, sequential simulates rainfalls were applied on same pans. Findings obtained from second sequential rainfall were thought as a different experiment. Experiments were planned at factorial design of randomized plots.

Table 1. Some properties of experiment soil

pH	$EC_{25ºC}$ dS m^{-1}	$CaCO_3$ g kg^{-1}	SOC g kg^{-1}	Clay g kg^{-1}	*Silt g kg^{-1}	Sand g kg^{-1}
8.3	2.24	117	12	393	231	376

*According to USDA. SOC: Soil organic carbon, EC: Electrical conductivity

RESULTS

Measuring values of variables under first and sequential rainfall were given in Table 2 & 3, respectively. All polymer applications delayed the RST. Under first rainfall, minimum and maximum RQs were measured as 12.6 mm (for PAM + >6.4 mm) and 46.4 mm (for no polymer + < 1mm), respectively. These values were recorded for same pans as 18.7 and 61.1 mm, respectively under second rainfall. Under first rainfall, SQTR and SLS values were changed between 53.3 (for PVA + >6.4 mm)-2925.6 (for no polymer + <1 mm) and 2.0 (for PAM + >6.4 mm)-18.1 g m^{-2} (for no polymer + <1 mm), respectively. These values were measured as 71.8 (for PVA + >6.4 mm)-4699.3 (for no polymer + <1 mm) and 5.5 (for PAM + >6.4 mm)-21.1 g m^{-2} (for no polymer + <1 mm) under sequential rainfall. According to ANOVA results, RQ from pans under first rainfall affected statistically by applications ($P<0.01$) and initial aggregates sizes ($P<0.001$). Also under second rainfall, subjects effected on RQ ($P<0.001$). The effects of applications and initial aggregate sizes on SQTR were statistically significant under both rainfalls ($P<0.001$). While effect of applications on SLS was statistically no significant that of initial aggregate size was significant at level of $P<0.01$, under first rainfall. Under sequential rainfall, effects of applications ($P<0.05$) and initial aggregate sizes ($P<0.001$) on SLS were found significant. Duncan comparison test results through the applications and initial aggregate sizes were given in Table 4 & 5, respectively.

Table 2. Soil and water loss from Typic Xerochrept under first simulated rainfall

Application	Initial aggregate size (mm)	RST (sec)	RQ (mm)	SQTR (g oven dry soil m^{-2})	SLS (g oven dry soil m^{-2})
No polymer	<1	35	46.4	2925.6	18.1
	1-2	80	38.8	332.5	16.2
	2-4	144	32.0	231.0	9.8
	4-6.4	159	26.2	143.4	8.2
	>6.4	223	23.6	110.1	6.3
	All	175	34.1	196.5	8.6
PAM	<1	123	45.1	2071.0	18.0
	1-2	139	34.7	307.6	3.8
	2-4	164	21.5	131.2	3.6
	4-6.4	259	18.3	116.2	2.1
	>6.4	338	12.6	86.5	2.0
	All	247	24.8	156.5	4.0
PVA	<1	84	42.4	2310.2	13.8
	1-2	124	34.9	176.4	7.9
	2-4	156	29.1	107.5	7.4
	4-6.4	168	21.9	69.1	4.0
	>6.4	303	14.3	53.3	3.2
	All	190	34.4	112.6	6.4

RST: Runoff starting time, RQ: Runoff quantity, SQTR: Sediment quantity transported by runoff, SLS: Transported soils to two sides of pan by splash

Table 3. Soil and water loss from Typic Xerochrept under sequential simulated rainfall

Application	Initial aggregate size (mm)	RST (sec)	RQ (mm)	SQTR (g oven dry soil m^{-2})	SLS (g oven dry soil m^{-2})
No polymer	<1	25	61.1	4699.3	21.1
	1-2	75	48.1	794.2	20.3
	2-4	112	43.6	314.5	12.0
	4-6.4	114	37.3	156.0	10.7
	>6.4	126	32.2	137.9	12.3
	Tüm	89	47.9	477.3	13.2
PAM	<1	37	55.0	2428.0	19.6
	1-2	96	44.6	330.8	13.8
	2-4	122	28.9	186.6	10.3
	4-6.4	133	26.5	126.7	6.4
	>6.4	179	18.7	89.1	5.5
	Tüm	111	36.9	160.1	12.3
PVA	<1	47	52.0	2844.2	18.7
	1-2	105	44.8	304.8	12.0
	2-4	137	40.5	138.8	9.1
	4-6.4	142	36.5	117.0	6.8
	>6.4	160	25.6	71.8	6.3
	Tüm	94	39.0	145.3	11.4

RST: Runoff starting time, RQ: Runoff quantity, SQTR: Sediment quantity transported by runoff, SLS: Transported soils to two sides of pan by splash

CONCLUSIONS

As a conclusion, PAM and PVA applied a Typic Xerochrept reduced soil and water losses. Their effectiveness varied depending on initial aggregate size. Total soil-water losses were different for first and sequential rainfalls. Obtained findings showed that the precipitation pattern of the region and the periodic dominant aggregate size group in the soil should be taken into account in order to effectively combat erosion.

Table 4. Duncan test results through the applications

Variables	Application	N	Sub groups	Application	Sub groups
			First rainfall		Sequential rainfall
RQ	PAM	12	26.1667b	PAM	35.0750c
	PVA	12	29.5000ab	PVA	39.7333b
	No polymer	12	33.5083a	No polymer	45.0500a
SQTR	PVA	12	471.4750b	PAM	553.5500b
	PAM	12	478.0083b	PVA	603.6500b
	No polymer	12	656.5167a	No polymer	1096.5250a
SLS	PAM	12	No significant	PVA	10.7000b
	PVA	12	No significant	PAM	11.3333b
	No polymer	12	No significant	No polymer	14.9417a

SQ: Runoff quantity, SQTR: Sediment quantity transported by runoff, SLS: Transported soils to two sides of pan by splash

Table 5. Duncan test results through the initial aggregate sizes

Variables	Aggregate size	N	Sub groups	Aggregate size	Sub groups
			First rainfall		Sequential rainfall
RQ	>6.4	6	16.8333e	>6.4	25.5000f
	4-6.4	6	22.1333de	4-6.4	33.4333e
	2-4	6	27.5667cd	2-4	37.6667d
	Tüm	6	31.1000bc	Tüm	41.2500c
	1-2	6	36.1167b	1-2	45.8333b
	<1	6	44.6000a	<1	56.0333a
SQTR	>6.4	6	83.3000c	>6.4	99.5833c
	4-6.4	6	109.5500c	4-6.4	133.2167bc
	Tüm	6	155.2167c	2-4	213.2667bc
	2-4	6	156.5833c	Tüm	260.8833bc
	1-2	6	272.1667b	1-2	476.6333b
	<1	6	2435.1833a	<1	3323.8667a
SLS	>6.4	6	3.8000b	4-6.4	7.9500c
	4-6.4	6	4.7667b	>6.4	8.0167c
	Tüm	6	6.3500b	2-4	10.4333bc
	2-4	6	6.9167b	Tüm	12.3333bc
	1-2	6	9.3000b	1-2	15.3833ab
	<1	6	16.6333a	<1	19.8333a

RQ: Runoff quantity, SQTR: Sediment quantity transported by runoff,
SLS: Transported soils to two sides of pan by splash

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Session IV: Evaluation and modeling soil and water degradation processes

4.1.O

USING A CONTINUOUS MODEL FOR REFINEMENT OF NUTRIENT RISK ASSESSMENT TOOLS

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INTRODUCTION

Implementation of nutrient management plans should protect the environment, maintain crop productivity, and increase profitability. Nutrient management planning is a complex process requiring planners knowing what resources are available and what needs are to be met. The Natural Resources Conservation Service (NRCS) 590 standard sets the national minimum standards for nutrient management in the U.S. (USDA-NRCS, 2012). The phosphorus (P) index (PI) is one of the management tools that can be used to identify agricultural fields with a high potential for runoff P losses. The PI is a systematic method for integrating a wide range of field characteristics into a prediction of the potential for P loss from the field. A revision of the 590 standard, suggested using the Agricultural Policy/Environmental eXtender (APEX) model as the new risk assessment tool to target critical source areas and practices to reduce agricultural non-point sources losses of sediments, P and nitrogen (N) (Sharpley et al., 2011).

APEX is a semi-process-based, distributed hydrologic and water quality model that operates at continuous daily time step (Radcliffe et al., 2015). This model has been used to evaluate various land management strategies considering sustainability, erosion (wind, sheet and channel), economics, water supply and quality, soil quality, plant competition, weather and pests (Steglich and Williams, 2013). However, APEX has been reported to have limitations due to model structure to accurately predict objective parameters (i.e. flow, sediments and nutrient losses) under different circumstances (e.g. Sen et al. (2012) and Collick et al. (2016)). Concerned that APEX could not adequately capture P losses, members of the USDA Southern Extension and Research Activity 17 (SERA-17) developed a white paper stating the need to compare Indices and water quality model performance, using edge-of-field based P runoff data (Sharpley et al., 2011).

In order to determine if APEX could be used to refine or replace P risk assessment tools in the southern region of the U.S., a study was performed to compare predictions from edge of field models, including

APEX, against measured P loss data to determine if models could be used for refinement or replacement of P Indices in the southern U.S.

METHODS

Uncalibrated and calibrated APEX model predictions were compared against measured water quality data from twenty different scenarios including row crops and pasture fields in four southern states of the United States. The general characteristics of the field sites can be found in Table 1. A detailed description of the water quality datasets used to evaluate the model can be found in Bolster et al. (2017) and Ramirez-Avila et al. (2017).

The information about soils, weather, agricultural operations (e.g. operation type and date, application rates) and soil properties in the fields (e.g. pH, soil test P) used to setup the model were supplied by researchers of the corresponding study sites and/or obtained from publications (Pierson et al., 2001; Yuan et al., 2013; Larsen et al., 2014; Edgell, 2015). The datasets for runoff depth and quality for the fields used to tests the model were directly supplied by the corresponding researchers.

Table 1. General characteristics of the sites

	Arkansas	Georgia	Mississippi	North Carolina
# Fields (Area)	Seven (0.4-ha)	Six (0.75-ha)	Two (11 & 13-ha)	Twenty (0.017-ha)
Annual Rainfall (mm)	1,215	1,000 - 1,120	990 - 1,300	1,020 - 1,195
Crop	Fescue (Hay-and-Grazing)	Bermuda grass/Fescue	Soybean/winter wheat Cotton/winter wheat	Sweet corn/winter wheat
Soils	silt loam	fine sandy loam & sandy loam	silty clay loam, loam, very fine sandy loam & clay	silt loam
Slope (%)	2.0	6.0 to 8.0	0 to 6.5	3.5 to 4.3
Mehlich-3 P (mg-P kg^{-1})	91 – 183	22 to 53	38 and 50	27 to 81
Field and nutrient management	Hay No-P Hay org.-P (broadc.) Hay org.-P (injected) Cont. grazing org.-P Rotat. grazing org.-P	Organic-P	Red. Tillage inorganic-P	Conv. Till. Inorg.-P Conv.Till. org.-P No Till. Inorg.-P No Till org.-P
Applied P rate (kg ha^{-1})	0 to 80	215 to 327	9.5	0 to 114

Model performance for event-based runoff, sediment and P loads predictions was evaluated using the Nash-Sutcliffe efficiency (NSE) and percent bias (PBIAS) with critical values of NSE ≥ 0.30 and absolute value of PBIAS < 0.35, 0.6, 0.7, and 0.7 for runoff, sediment, dissolved P (DP) and total P (TP), respectively. Comparisons were made on an event basis.

RESULTS

Overall, uncalibrated and calibrated APEX models predicted runoff depths that met the performance criteria for the event-based predictions for Georgia, North Carolina, and Mississippi (Table 2). Runoff depths were highly overestimated in Arkansas as the weather time series used to setup the model affected its performance. Overall, neither the uncalibrated nor the calibrated model could accurately predict sediment, DP, or TP losses. Satisfactory performance for calibration of sediment loss was observed only from predictions on the Mississippi fields. TP loads in Arkansas met the performance criteria. However, since the runoff and sediment predictions were unsuccessful, the P predictions at this site are questionable. APEX is not sensitive to predict small concentrations and loads of nutrients (Francesoni et al., 2014), which could cause inaccurate predictions of TP and DP loads at the evaluated sites. Unsuccessful performance and underestimations in surface applied organic P sites could be caused because APEX lacks a routine for predicting manure-P processes on the soil surface. APEX also overestimated the P loss carried by irrigation-runoff events in Mississippi, caused by the overestimation in sediment loss from irrigation-runoff events.

Table 2. APEX model performance estimates for uncalibrated and calibrated predictions in Arkansas, Georgia, North Carolina and Mississippi.

	Uncalibrated				Calibrated			
	AR	GA	NC	MS	AR	GA	NC	MS
Runoff								
NSE	-17.08	0.58	0.21	0.67	-0.10	0.70	0.47	0.72
PBIAS (%)	-12.15	17.08	-3.42	47.22	-5.38	2.59	-2.50	19.65
Sediment								
NSE	-321534	-	-160	0.34	-0.28	-	0.02	0.48
PBIAS (%)	-13057.9	-	-752	49.77	30.65	-	47.55	21.74
Total P								
NSE	-118.27	-0.10	-0.34	-0.51	-0.34	-0.34	-0.27	-0.79
PBIAS (%)	-296.65	-11.10	78.05	74.17	77.43	86.37	86.21	53.8
Dissolved P								
NSE	-1.39	0.04	-0.05	-0.61	-0.46	-0.27	-0.15	-1.79
PBIAS (%)	46.62	5.39	84.27	89.04	77.06	91.36	95.96	6.79

CONCLUSIONS

Based on the analysis of key details about the observational data and model characteristics, it was concluded that the capability of APEX to predict P losses is limited and consequently, cannot be used to refine or replace P indices in the southern U.S. Results identified a critical need for reviewing and updating APEX routines to better represent the effects of the nutrient management factors immersed on the PI that influence potential P movement to surface waters.

More detailed and extended analysis for the study is presented in Ramirez-Avila et al. (2017)

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4.2.O

IMPACT OF RAINFALL PATTERN ON INTERRILL EROSION PROCESS

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Rainfall is an essential driving force in governing interrill runoff and erosion processes. The impacts of rainfall characteristics, such as rainfall intensity (*I*), duration, interval, kinetic energy flux (KE_r), and raindrop impact incidence have been extensively studied to date, and remain the focus of soil erosion mechanistic research (Poesen and Nearing, 1993; Erpul et al., 2004; Kinnell, 2005). Among these characteristics, KE_r and *I* are the dominant factors affecting interrill erosion and surface hydrologic responses. KE_r plays an important role in interrill erosion and sediment transport by enhancing transport capacity in raindrop-induced flows (Zhang et al., 1998; Kinnell, 2005; Bako et al., 2016). Rainfall intensity exerts inconsistent effects on runoff and soil losses that likely result from seal formation and the varied erodibility that occurs within a storm (Poesen and Nearing, 1993; Huang et al., 2001). The empirical relationship between interrill erosion rate (D_i in g/m^2·s) and effective rainfall intensity (I_e in mm/s) (Meyer, 1981), and a more process-based concept related to kinetic energy (*KE* in J/m^2) (Morgan *et al.*, 1998) represent significant milestones in improving soil erosion prediction models.

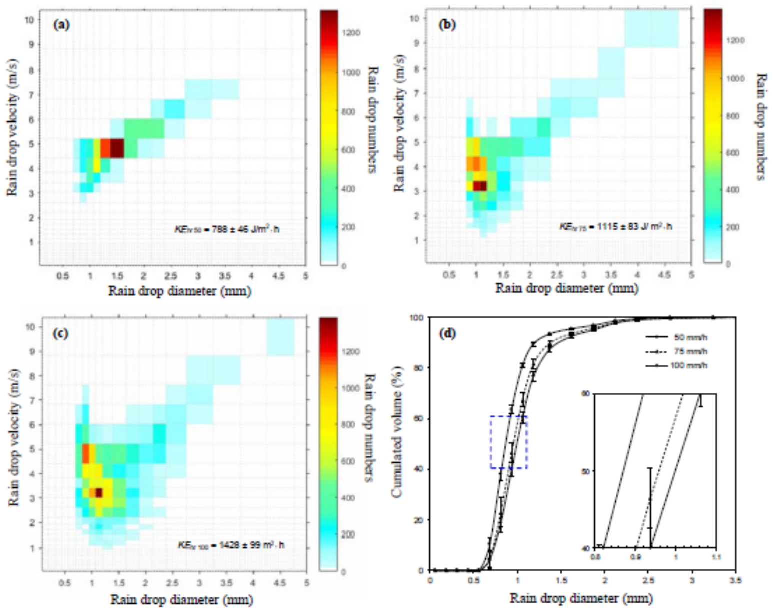

Figure 1. Spectrum view of raindrop velocity /diameter combinations with drop counts in 1-min measuring interval for simulated rainfall intensities 50 mm/h (a), 75 mm/h (b) and 100 mm/h (c); and for raindrop size distributions (d).

These concepts have been widely accepted by the research community and form the basic structure of the interrill erosion component in most models. Based on these approaches, however, it’s easy to come with the assumption that a given *I* or *KE* value for a specific soil and rainfall duration result in the same amount of soil loss. The ability to quantify the impact of rainfall patterns will enhance our understanding of soil detachment and sediment transport processes. The primary objectives of this study were: 1) To quantify the impact of rainfall patterns on interrill runoff and soil losses, 2) to investigate the interactions of varied intensity, rain stage, intensity sequence, total *KE*, and cumulative *KE* on soil erosion, and 3) to analyze the dynamic mechanism under different rainfall patterns.

Systematic rainfall simulation experiments involving various rainfall intensities, stages, intensity sequences, and surface cover conditions were conducted in this study to investigate their effects on interrill erosion process. Each rainfall KE_r was approximately 788, 1115, and 1428 $J/m^2 \cdot h$ accordingly, thus attaining about 78%, 75% and 70% of natural rainfall energy (Salles et al., 2002). Raindrop diameter ranged from 0.43 to 5.50 mm, and the D_{50} (median diameter) was 0.87, 0.96, and 1.00 mm corresponding to 50, 75, and 100 mm/h, respectively (Figure 1). Five rainfall patterns designed with the same total kinetic energy/precipitation (increasing, decreasing, rising-falling, falling-rising and constant patterns) were randomly delivered to a pre-wet clay loam soil surface at a 10° slope gradient.

RUNOFF AND SOIL LOSS

No significant differences in total runoff were observed among the different rainfall patterns, suggesting that rainfall patterns did not influence the total runoff during an event (Table 1). With the same total 75 mm precipitation, the average runoff decreased from 65.62 to 60.43 mm when raindrop impact was absent. The impacts of rainfall patterns and surface cover conditions were significant in regards to soil loss, however: Soil losses were reduced about 82% when raindrop *KE* was eliminated by the surface cover. Additionally, remarkable differences were observed among runs in which rainfall intensities varied or remained constant. The CST pattern produced the lowest sediment yield at around 61.8% and 36.4% of the average soil loss for the ICR pattern with/without raindrop impact, respectively. For both surface cover conditions, soil loss ranked similarly as: ICR > FR > DCR > RF > CST.

Table 1 Runoff, soil loss, runoff peak and erosion peak for different rainfall patterns with two surface cover treatments (with/without raindrop impact).

Treatment	Rainfall patterns	Runoff (mm)		Soil loss (kg/m^2)		Runoff peak (L/min)		Erosion peak (g/ min)	
with raindrop impact	ICR	65.95 ± 0.05	b	0.55 ± 0.01	a	2.98 ± 0.05	a	36.80 ± 0.42	a
	DCR	65.81 ± 0.67	b	0.44 ± 0.03	bc	2.95 ± 0.31	a	29.43 ± 5.24	b
	RF	65.98 ± 0.41	b	0.41 ± 0.03	c	3.02 ± 0.02	a	27.23 ± 0.32	b
	FR	65.01 ± 0.37	b	0.45 ± 0.00	b	2.92 ± 0.03	a	28.10 ± 1.84	b
	CST (P)[†]	65.35 ± 0.39	b	0.32 ± 0.01	d	2.30 ± 0.13	b	12.65 ± 0.28	c
without raindrop impact	ICR	59.15 ± 0.27	b	0.11 ± 0.01	a	3.20 ± 0.08	a	8.85 ± 0.78	a
	DCR	59.08 ± 3.33	b	0.08 ± 0.00	c	2.54 ± 0.24	b	6.92 ± 0.39	b
	RF	61.66 ± 2.05	ab	0.07 ± 0.00	d	3.10 ± 0.21	a	5.18 ± 0.11	c
	FR	62.67 ± 1.05	ab	0.09 ± 0.00	b	3.07 ± 0.22	a	7.50 ± 0.28	b
	CST (P)[†]	59.62 ± 0.62	ab	0.03 ± 0.00	e	2.14 ± 0.03	c	1.55 ± 0.17	d

Note: Mean values followed by different letters (i.e. a, b and c) means significantly different at 0.05 level according to the LSD test; [†]the rainstorm event suffered a total precipitation value 75 mm the same as the other four rain pattern.

RUNOFF AND SOIL LOSS PROCESS

Runoff rate showed a rapid and obvious response to rainfall intensity, as we had expected. Runoff rates ranged from 1.08 to 1.64 L/min for 50 mm/h rainfall intensity, 1.87 to 2.42 L/min for 75 mm/h, and 2.19 to 3.00 L/min for 100 mm/h with raindrop impact, respectively. A similar range of runoff rates was found in storms without raindrop impact, except that there was less uncertainty than those simulations with raindrop impact, most likely due to the near-saturation soil surface before each runs (Parsons and Stone, 2006).

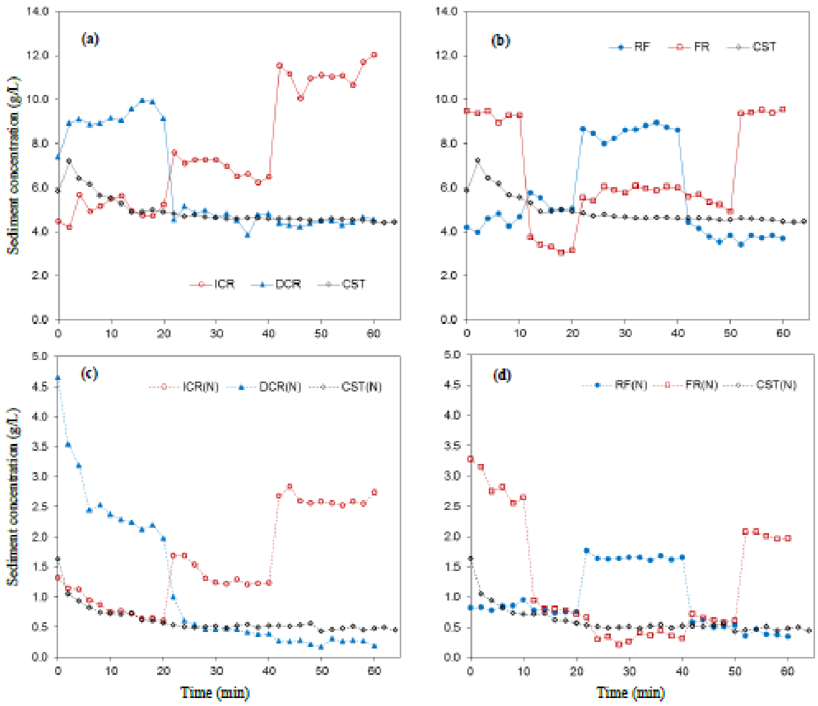

Figure 2. Temporal variations of sediment concentration for different rain patterns with raindrop impact (a, b) and without raindrop impact (c, d)

The storms with raindrop impact resulted in an obviously higher sediment concentration than those without raindrop impact. Complex phenomena were detected among different rainfall patterns and rain stages, however (Figure 2). When raindrop impact was present, sediment concentration for the CST pattern reached a peak value of 7.2 g/L in the beginning and then decreased gradually to a relatively low value after the first rainfall stage. Sediment concentration fluctuated for the other four rainfall patterns and reached peaks at different stages, implying that dominant erosion regimes varied with rainfall patterns. Furthermore, temporal variations of sediment concentration for DCR (Figure 2a) and FR (Figure 2b) patterns suggested that prior high rainfall intensity caused relatively low and stable sediment concentration; this stable phase of sediment concentration was unbroken until another highly intense rainfall was delivered. Sediment concentration was reduced by more than 60% on average when raindrop impact was eliminated, but showed similar trends as in the storms with raindrop impact (Figure 2c, 2d). That said, sediment concentration decreased sharply during the very first 5-10 min for all patterns when raindrop impact was eliminated.

RAINFALL PATTERN IMPACT ON DYNAMIC MECHANISM

We anticipated that rainfall patterns would have a profound effect on raindrop-induced sediment transport processes. Results showed that the erosion peak decreased as runoff peak increased for CST with raindrop impact (Figure 3), implying that the raindrop-associated transport process controlled the rate at which loose particles moved across the soil surface. Similar results were reported by Fox and Bryan (1999), Martínez-Mena et al. (2002), and Hammad et al. (2006), all of whom observed that factors like crusting can reduce the amount of available sediment for transport under experimental conditions with high and constant intensity (60-90 mm/h). We also observed an obvious positive relationship between erosion peak and runoff peak, however, when as rainfall with varying intensity was delivered to the soil surface. There was a transition in the erosion regime as the rainfall pattern shifted, which supports our second hypothesis regarding the impact of rainfall pattern. Mayer and Harmon (1989), Fox and Bryan (1999), Hammad *et al*. (2006) and Ran *et al*. (2012) also observed erosion regime transitions, but mainly focused on the different conditions of slope gradients and rain intensities. Furthermore, no evident trend was found between the erosion and runoff peaks without raindrop impact, further supporting our hypothesis that rainfall pattern effects are facilitated by KE_r.

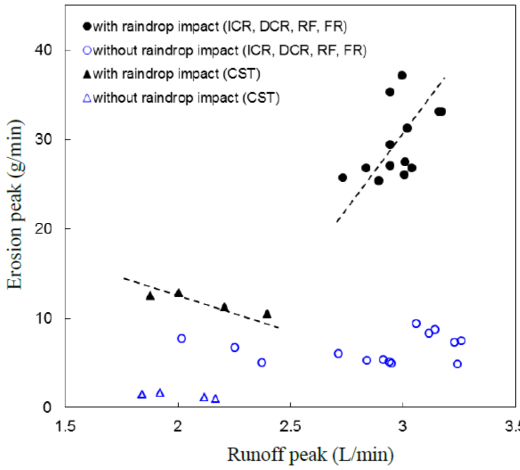

Figure 3. Relationship between runoff and erosion peaks for five rainfall patterns

CONCLUSION

Kinetic energy flux (KE_r) was a governing factor for interrill erosion. Varied-intensity patterns had a profound effect on raindrop-induced sediment transport processes; path analysis results indicated that said effect was complex, interactive and intensity-dependent. Low hydraulic parameter thresholds further indicated that KE_r was the dominant factor in detaching soil particles, while overland flow mainly contributed to transporting the pre-detached particles. This study not only sheds light on the mechanism of interrill sediment transport capacity and detachability, but also may provide a useful database for developing event-based interrill erosion prediction models.

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4.3.O

SOIL EROSION IN SUBTROPICAL FORESTS - EFFECTS OF SPECIES DIVERSITY, SPECIES IDENTITY, FUNCTIONAL TRAITS AND SOIL FAUNA

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INTRODUCTION

Soil erosion is a serious environmental problem especially in regions with high anthropogenic influences on the ecosystems (Morgan, 2005) such as the PR China (Guo et al., 2015). Vegetation covers are a key factor for the occurrence of soil erosion. Forests provide a multi-storey canopy layer that largely influences rain throughfall patterns and leads to the capture and storage of raindrops (Puigdefábregas, 2005). At the same time, species-specific effects on throughfall kinetic energy (TKE) are evoked by individual tree structures and traits, such as leaf area index (Geißler et al., 2013).
Nevertheless, large drops can be formed at leaf apexes (Geißler et al., 2012) and thus may increase TKE in older forest stands (Nanko et al., 2008). This may lead to considerable soil loss, if the forest floor is uncovered, which can be the case if protecting layers diminish (Onda et al., 2010). A leaf litter layer on the forest floor protects the soil from direct raindrop impact and modifies the water flow and storage (Kim et al., 2014). Leaf litter provides habitats for soil fauna and in addition is an important food source for physical litter transformers (Sayer, 2006).

The biological diversity of trees (tree species richness) as well as functional diversity of tree communities can play a critical role in improving ecosystem services (Scherer-Lorenzen, 2014). Although positive effects of mixed-species tree stands, such as increasing productivity, were demonstrated (Bauhus and Schmerbeck, 2010), the effects of higher species mixtures on erosion control are still unclear. Nevertheless, there is growing evidence that higher species richness can generally reduce soil erosion (Körner and Spehn, 2002), e.g. a positive diversity effect on TKE was observed in a subtropical secondary forest (Geißler et al., 2013). Conceivable mechanisms underlying positive species richness effects on soil erosion are that vegetation covers with a high number of species include a high number of plant functional groups, which complement one another (Kelty, 2006). Thus, they are more effective in controlling erosion processes than vegetative covers with few species (Pohl et al., 2012). TKE reacts strongly to individual tree characteristics and may affect sediment discharge as a consequence (Geißler et al., 2013). In addition, a highly diverse structure within the leaf litter layer on the forest floor can improve its protecting effect (Martin et al., 2010).

In this work, mechanisms of soil erosion in a subtropical forest were investigated and effects of species diversity, tree species identity, functional traits and soil fauna were observed. We focussed on changes in TKE during the canopy passage of raindrops (I) as well as interrill soil erosion rates determined by micro-scale runoff plots under natural (II) and simulated rainfall (III).

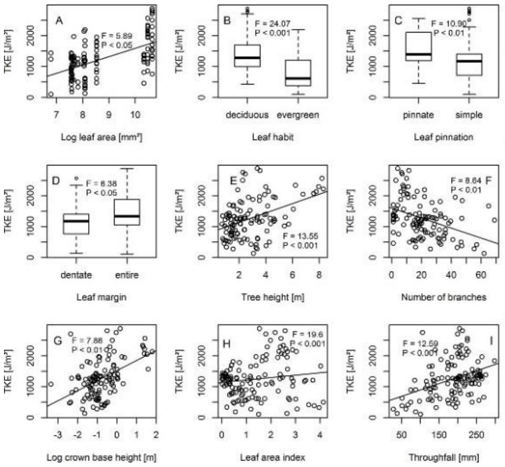
METHODS

This work was conducted in the Biodiversity and Ecosystem Functioning China (BEF China, www.bef-china.de) experiment (Bruelheide et al., 2014) in Xingangshan, PR China. The area comprises a mountainous landscape with a mean slope of 29 ° (Scholten et al., 2017). The climate in Xingangshan is characteristic for subtropical summer monsoon regions (Goebes et al., 2015b). The experiment was clear-cut and replanted in 2009-2010 following a plot-based design (Yang et al., 2013). In total, 566 experimental plots (25.8 m × 25.8 m) were established using a pool of 40 native tree species planted randomly in seven species richness levels.

Measurements of throughfall kinetic energy (I) were carried out on 40 plots (Goebes et al., 2015a; Goebes et al., 2015b) with Tübingen splash cups (T-cups, Scholten et al., 2011). Therefore, 24 different tree species and six levels of tree species richness (17 × div1, 10 × div2, 6 × div4, 4 × div8, 1 × div16, 2 × div24) were used. The measurements were located in a central area with 6 × 6 tree individuals per plot. For each tree, nine different splash cup positions were chosen to investigate different tree traits. In total, 1800 splash cups were measured in 2013 during five rainfall events. Furthermore, measurements of sediment discharge under natural rainfall (II) were conducted with micro-scale ROPs (0.4 m × 0.4 m) on a selection of 34 plots (Seitz et al., 2016). The selected set comprised a bare ground feature (4 × div0) and four levels of tree species richness (20 × div1, 4 × div8, 4 × div16 and 2 × div24) with a total of 22 tree species, two of which only appeared in mixtures. Plots were equipped with five ROPs each, resulting in a total number of 170 ROPs. ROPs were placed randomly in selected areas, which were representative for the range of surface properties of the study area and were operated in May and June 2013 during the rainy season. At last, measurements of sediment discharge under simulated rainfall (III) were conducted with micro-scale ROPs modified with a fauna exclusion feature (Seitz et al., 2015). Leaf litter of seven domestic tree species was used in one-, two- and four-leaf species mixtures with a total number of 96 ROPs. Rainfall was simulated for 20 minutes at each ROP with the Tübingen rainfall simulator (Iserloh et al., 2013). Measurements were carried out when ROPs were fully covered (May 2012) and after decomposition had led to reduced leaf litter coverage (September 2012).

RESULTS

(I) TKE was not influenced by tree species richness at the plot level although monocultures showed slightly lower TKE (-6 %) than species-mixtures. This is likely due to the young age of the experiment, where a dense and high tree canopy has not yet been developed. At the time of measurements, tree characteristics, such as crown area, were evolved to approximately 10 % compared to mature trees in the study region (Bruelheide et al., 2014). It is likely that tree species richness effects on TKE develop over time, which was shown in grassland biodiversity experiments (Reich et al., 2012). However, TKE was positively influenced by neighbourhood tree species richness indicating that tree species richness only affected TKE on a small spatial scale within the direct neighbourhood. Thus, tree species richness in young forests seems not to be beneficial to ecosystem functioning due to enhanced soil erosion potential. TKE showed distinct spatial variability ($p < 0.1$). Directly below the first branch of the tree individuals TKE was lowest (430 J m^{-2}) while it was highest in the middle of four tree individuals (556 J m^{-2}). Mean FKE was 480 J m^{-2} and TKE was only higher than FKE below trees exceeding 3.3 m. Results further showed that TKE was species-specific in this young successional stage as three out of 11 species showed distinct differences. Highest TKE was found below *Choreospondias axillaris* and *Sapindus saponaria*, while *Schima superba* showed lowest TKE. *Schima superba* is well-known to show high values of canopy interception during rainfall (Guo et al., 2006), which is as much more relevant as *Schima superba* represents one of the dominant tree species in the regional species pool (Bruelheide et al., 2011). Species-specific effects were mediated by leaf habit, leaf area, leaf pinnation, leaf margin, stem diameter, crown height, tree height, number of branches, LAI and throughfall. Among these, leaf area, leaf habit and tree height showed the highest effect sizes on TKE and can be considered as major drivers. TKE was lower in evergreen, simple leaved and dentate leaved than in deciduous, pinnated or entire leaved species (Figure 1).



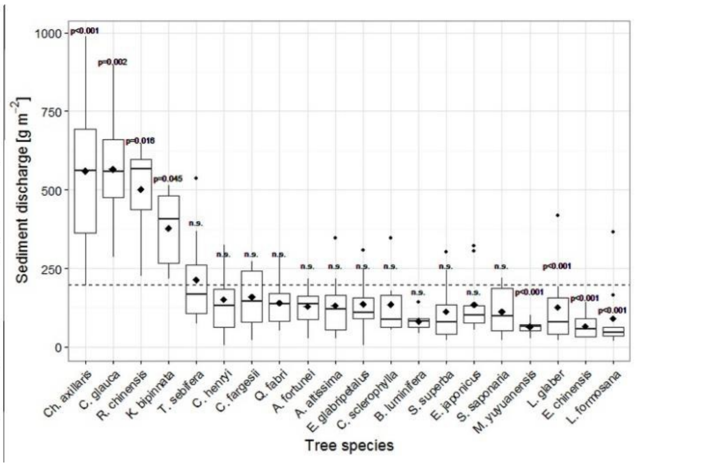


Figure 1 (left): Throughfall kinetic energy (TKE) versus leaf traits (A-E), tree architectural traits (F-H) and abiotic covariates (I). Black solid lines indicate linear trend (from Goebes et al. 2015b)

Figure 2 (right): Sediment discharge under 20 tree species in monocultures in Xingangshan, PR China in 2013. Dashed line indicates mean sediment discharge of all 20 species. Horizontal lines represent median and diamonds represent means (from Seitz et al., 2016)

(II) Although a negative trend in sediment discharge was visible from level 1 to 8 and mixed stands showed a more balanced and homogenous vegetation development than monocultures (cf. Kelty, 2006), higher tree species richness did not mitigate soil erosion. The absence of a species richness effect on soil loss was likely attributable to the early successional stage of the forest experiment with low tree ages (see above). Moreover, this study provided evidence that different tree species affect interrill erosion processes as several species showed significant variation from mean sediment discharge (199 g m^{-2}). Thus, it could be shown that different tree morphologies have to be considered, regarding erosion in young forest ecosystems. *Chorespondeas axillaris*, *Cyclobalanopsis glauca*, *Rhus chinensis* and *Koelreuteria bipinnata* were related to increasing soil erosion rates, whereas *Magnolia yuyuanensis*, *Lithocarpus glaber*, *Elaeocarpus chinensis* and *Liquidambar formosana* mitigated soil erosion in young forest stands (Figure 2). Thus, we can confirm a species-specific effect on sediment discharge for this subtropical experimental area. Furthermore, it appears that the appropriate choice of tree species for afforestation against soil erosion becomes already important in an early successional stage. Species-specific effects can result due to different throughfall kinetic energy and were ascribed to different tree architectural characteristics, leaf traits and site characteristics. Low tree stands with high canopy cover are effectively counteracting soil loss in this initial forest ecosystem. Even if a leaf litter cover was not present in this experiment, soil surface cover by stones and biological soil crusts was the most important driver for soil erosion control. Furthermore, soil organic matter had a decreasing influence on sediment discharge.

(III) Results confirmed that a leaf litter cover protects soil surfaces, as sediment discharge on leaf covered plots was reduced by 82 % compared to bare plots. Neither leaf species diversity nor functional diversity influenced sediment discharge, leaf litter cover and thus soil erosion. Hence, better overlap and gap-filling or different decomposition rates in highly diverse litter mixtures seem not to be considerable parameters for soil erosion control. Functional diversity of leaf traits did not influence erosion parameters but notwithstanding, single leaf species in monocultures showed rather different impacts on sediment discharge and thus erosion control. In this experiment, runoff plots with leaf litter from *Machilus thunbergii* showed the highest sediment discharge (68.0 g m^{-2}) whereas plots with *Cyclobalanopsis glauca* showed the smallest rates (7.9 g m^{-2}). This can be related to variable leaf habitus, different decomposition rates and food preferences of litter decomposing fauna. Further, different leaf species showed rather different decomposition rates and thus developments of surface litter cover. This variability leads to positive and negative feedbacks of leaf litter species on soil erosion, but within leaf mixtures those differences are levelled (cf. Hättenschwiler et al., 2005). The protective effect of a leaf litter cover was influenced by the presence of soil mesofauna (Figure 3). Fauna presence increased soil erosion rates significantly by 58 % and this effect was slightly decreasing from spring to autumn. It is assumed that the activity of soil-dwelling and surface-active mesofaunal organisms led to the loosening and translocation of soil particles within the top soil. Furthermore, soil mesofauna and macrofauna are a prominent factor in litter fragmentation and decomposition and thus the reduction of protecting litter covers.

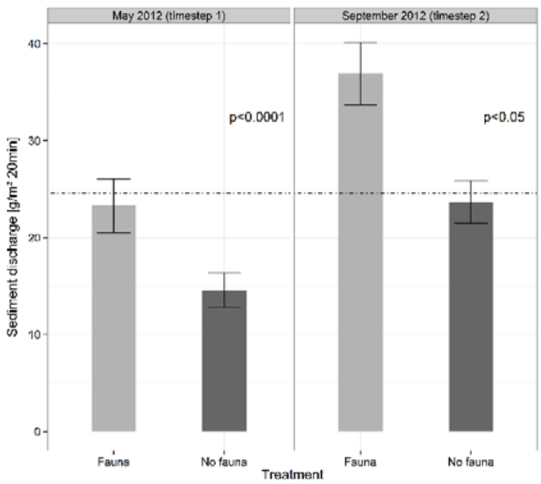


Figure 3: Effect of mesofauna on sediment discharge at two timesteps (May and September 2012) (from Seitz et al. 2015)

CONCLUSIONS

Influences of tree species richness, identity and species-specific functional traits on interrill erosion and TKE and the role of leaf litter cover and soil mesofauna were studied in a subtropical Chinese forest. Tree or leaf species richness did not affect sediment discharge, runoff and TKE in this subtropical forest ecosystem. Nevertheless, neighbourhood diversity effects on TKE were present. The absence of a species richness effect on soil erosion is likely attributable to the early successional stage of the forest experiment. Nevertheless, mixed tree stands showed a more balanced and homogenous vegetation development than monocultures. Thus, tree plantations with multiple species seem to ensure a more advanced soil cover than plantations with only one species.

Moreover, results showed that tree and leaf species identity influences initial soil erosion processes. Therefore, the appropriate choice of tree species plays a major role for erosion control. This effect becomes already considerable in an early successional stage and thus can be of importance during the establishment of tree plantations. Furthermore, species-specific functional traits affected soil erosion rates. High crown cover and leaf area index reduced soil erosion, whereas it was slightly increased by increasing tree height. At the same time, low LAI, low tree height, simple pinnate leaves, dentate leaf margins, a high number of branches and a low crown base height effectively minimize TKE. At last, results showed that the presence of soil mesofauna in the leaf litter layer increased initial soil erosion. Effects of this fauna group on sediment discharge have to be considered in soil erosion experiments.

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4.5.O

USING ORIBATID MITES TO ASSESS SOIL QUALITY AFTER THE ADDITION OF PRUNINGS IN SUBTROPICAL ORCHARDS

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INTRODUCTION

Biological diversity is one of the keys for the maintenance of ecosystems that have been damaged by anthropogenic activities. Some researches have focused on the role of mites (Acari) in biomonitoring and their importance as soil bioindicators in agroecosystems (Iturrondobeitia et al., 2004). Mite communities are extremely sensitive to all types of soil disturbance, especially oribatid mites. Oribatid mites are found suitable as robust bioindicators due to their low fecundity and dispersion ability. In fact, oribatid mites have been used as bioindicactors in many studies, some of them related to their tolerance to metal contamination (Zaitsev & van Straalen, 2001) or their ability to assess the effects of reforestation (Moraza, 2011). However, there are no studies related to their response to the addition of prunings in agroecosystems.

Changes in the dominance structure of mites communities can be used for measuring anthropogenic disturbances in soil ecosystems (Badejo et al., 1995). Since agricultural ecosystems are proved to absorb a large amount of CO_2 from the atmosphere, we examined the utility of oribatid mites for monitoring the progress of pruning decomposition and its potential for enhancing carbon storage in soils. Recycling of agricultural and urban wastes as soil mulchings is increasingly becoming a sustainable practice Microhabitats created by the application of mulchings may accelerate colonization and thus increase the resilience of the ecosystem.

The aim of this study was to investigate the effects of the application of different prunings on the abundance, diversity and community structure of free-living oribatid mites in soils with subtropical orchards in Almuñécar (Granada, S Spain). The potential of these mulchings to be considered as an agricultural practice for increasing the biodiversity of mites and for influencing their species composition and population dynamics in agroecosystems is discussed in this study.

MATERIAL AND METHODS

Three soils with subtropical orchards were sampled in Almuñécar (Granada, S Spain): mango (*Mangifera indica L.*), avocado (*Persea americana Mill.*) and cherimoya (*Annona cherimola Mill.*). The predominant climate is Subtropical Mediterranean and the soils are Eutric Anthrosols (IUSS Working Group WRB, 2015). Two treatments were applied: prunings from orchards (P), placed on the soil surface underneath their correspondent trees, and prunings from gardens (G) under the three orchard trees. Garden prunings come from green areas surrounding the town centre, thus promoting the recycling of urban debris. Control soils (C) without prunings were also sampled underneath all orchards. These experienced were followed for three years and the samples were collected in February 2016. The experimental design is shown in Figure 1.

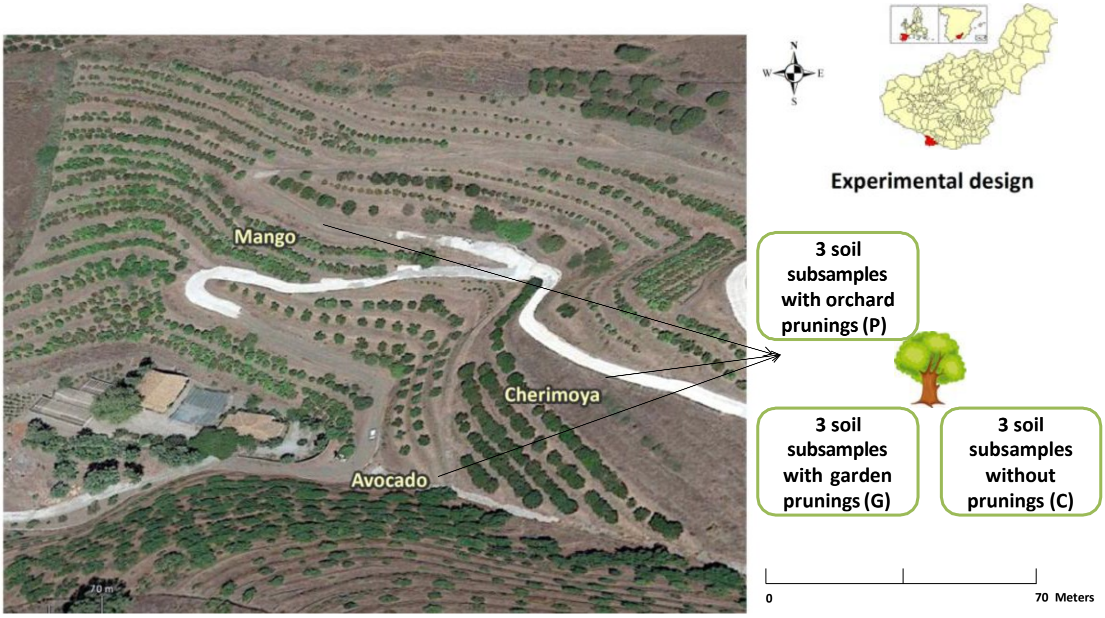

Figure 1. Study site and experimental design.

In every sampling plot, 3 soil subsamples were taken in a 50 cm^2 cylinder, 4 cm in depth. The mites were extracted in Tulgreen funnels (based on the Berlese funnel), preserved in 70% ethanol and determined to genus level, including also the juvenile's stages. The nomenclature of the mite groups followed Lindquist et al. (2009), whilst that of the Oribatida followed Weigmann (2006) and Pérez-Íñigo (1972). Oribatid mite communities were characterized by the number of individuals from family and the Shannon (H') diversity index.

RESULTS

Our results show that Oribatida were more abundant than Astigmata or Mesostigmata in all the control soils except for mango. Oribatida were numerically higher under all prunings than those in their respective controls (Fig. 2). Although no significant differences were found between the different prunings (orchards and gardens), all prunings significantly increased Oribatida populations in relation to the control soils. Oribatida are *k*-selected mites known for their long life span and late colonization (Reference), thus their presence may be attributed to a higher soil organic matter content. The presence of Astigmata and Mesostigmata, *r*-selected mites, indicates that soil fauna is still in an early succession state (Hasegawa et al., 2013). The decrease in abundance of Astigmata and Mesostigmata in the soils under orchards and garden prunings compared to their controls and the increase in Oribatida suggest that the addition of prunings may be increasing the soil organic matter content.

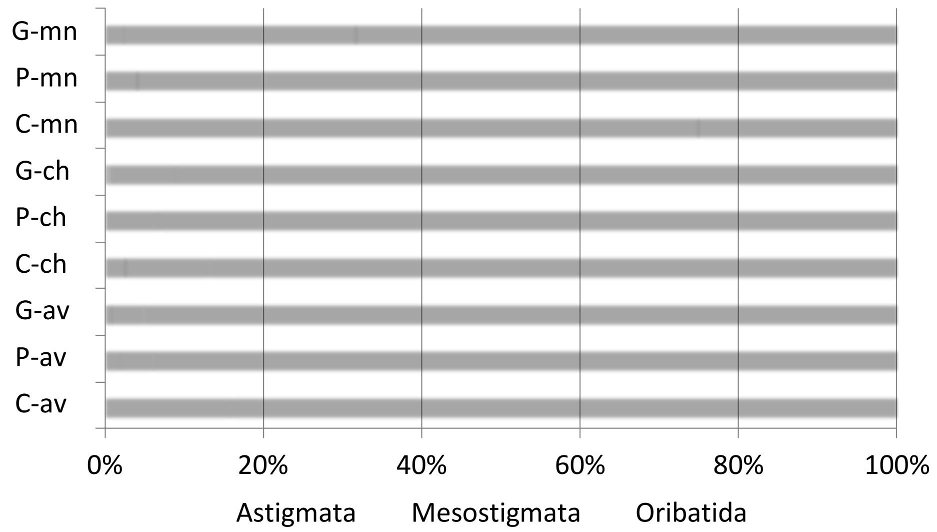

Figure 2. Density (Abundance in m^2) of different mite groups (adults individuals) in all orchards for the different treatments (*G, garden; P, pruning; C, control; mn, mango; ch, cherimoya; av, avocado*).

As shown in Table 1, the complexity of the Oribatid mite community was influenced by the application of the different prunings. For cherimoya and mango, no significant differences for density were recorded between garden and orchard prunings, but there were significant differences between prunings and controls. For avocado, there were significant differences between prunings (avocado and garden) and the control, and also between avocado and garden prunings, probably due to the original higher organic matter content (data not shown) in avocado soils compared to mango and cherimoya soils. On the other hand, the Shannon-Wienner index was generally low in every soil, probably due to the subtropical/dry Mediterranean climatic conditions (Magurran, 2004). Significant differences in this index were found for cherimoya prunings and mango control. However, no significant differences in this index were found in avocado, despite obtaining clear higher values with avocado prunings. These differences in density and diversity can be explained by the different chemical composition of prunings (between garden and orchard), that probably influenced the abundance of different Oribatida families.

Table 1. Adults density (individual·m^2), number of families (*S*) and Shannon index (*Hs*) of Oribatida adults in experimental plots. Significant differences are shown within each orchard at $p \leq 0{,}5$, between plots without and with mulching.

Oribatida	C-av	P-av	G-ag	C-ch	P-ch	G-ch	C-mn	P-mn	G-mn
Density	1180	4940*	1320	2280*	10804	8400	20*	1140	700
S	10	11	7	6	15	12	1	7	9
Hs	0,66	0,69	0,60	0,41	0,89*	0,62	0*	0,78	0,82

CONCLUSIONS

1. All prunings significantly increased Oribatida populations within the surface soil layer compared to the control soils, especially in mango, suggesting that the addition of prunings may be increasing the soil organic matter content.
2. The addition of prunings increased the diversity of the oribatid mite community, and also for other functional groups of mites (Astigmata and Prostigmata). Thus, organic debris should be applied or preserved as refuges for dispersion of soil fauna and for improving soil quality in terms of fertility and resilience. Further studies at genus level in Oribatida should be conducted for achieve accurate results.
3. Recycling and application of organic debris in agroecosystems resulted in an environmental-friendly management practice for improving soil quality and productivity.

ACKNOWLEDGEMENTS

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KEYWORDS

Oribatid mites, soil biodiversity, mulching, agroecosystem, subtropical orchards.

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4.6.O

ASSESSMENT OF SOIL EROSION RATES IN TWO AGRICULTURAL REGIONS OF EUROPEAN RUSSIA FOR THE LAST 50 YEARS IN THE VARIOUS SCALES

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Soil erosion totals can be dominated by a few extreme events, thus monitoring, as well as erosion model simulation need to be long enough to capture these erosive events. The land use and climate changes are also influenced on the average erosion rates. Quantitative assessment of erosion rate trends for the last 5 decades was undertaken for the Veduga River and the Mesha river basins, located in two regions of the Russian Plain (Voronezhskaya oblast' and Tatarstan Republic respectively). The both river basins are located in the forest-steppe zone with very high proportion of croplands (Fig.1, Table 1). Detail assessment of erosion rates was done for two time intervals (1963-1986 and 1986-2015) in the middle scale for the studied river basins. In addition, the soil redistribution for period 1963-1986 and 1986-2015 were studied in one small catchment within each river basin for the verification of model calculations.

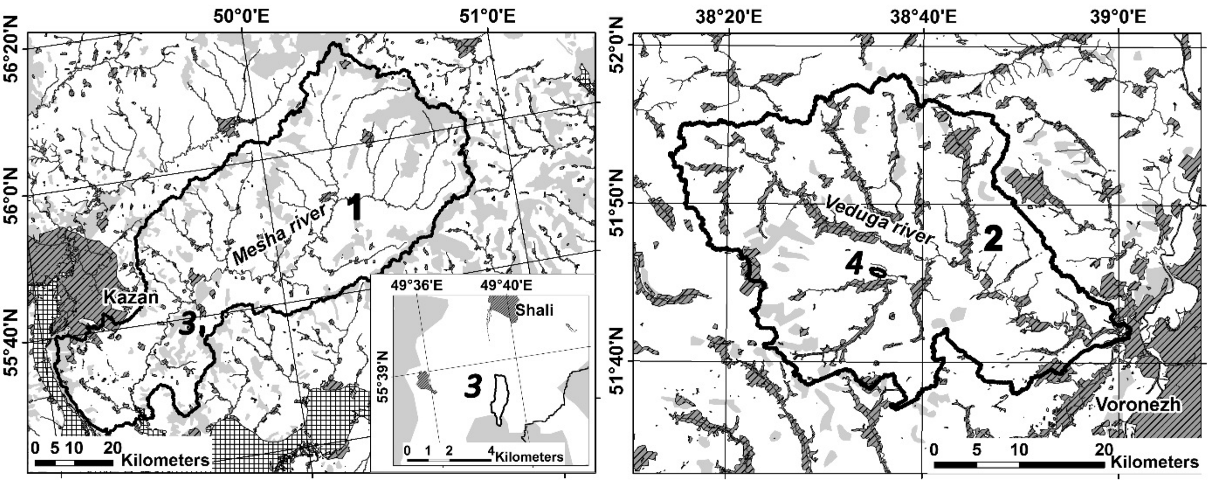

Figure 1. Land use maps for the period 2015-2016 for the Mesha river basin (left) and the Veduga river basin (right) and the location of small key catchments within the studied basin. Legend:

- boundary of basin or catchment; **1** - Mesha river basin; **2** - Veduga river basin;

3,4 - key catchments location.

- croplands; - woodland and grassland; - reservoirs; - settlements

Modified versions of USLE and State Hydrological Institute (SHI) models were applied for calculation of soil losses for warm period of year and snow-melt respectively. Land use changes were identified by visual interpretation of multi-seasonal images Landsat 5 and Landsat 8 which was done in 1985-1986 and 2013-2015. Meteorological characteristics, including precipitation, soil temperature during cold season, total water storage before the snow-melt was collected for the period 1960-2015. Crop management coefficients was evaluated based on statistical information about area under the different crops on the regional scale for period 1960-2015. It was assumed that soil erodibility and the relief characteristics don't change between both time intervals. Similar approach was used for calculation of the soil losses using erosion models at the key catchments, located within the river basins. The bomb-derived and Chernobyl-derived ^{137}Cs were used for the determination of sedimentation rates in the valley bottoms in the key catchments. The detail geomorphological mapping was undertaken for the both key catchment allowing to identified the different types of slope and slope catchments for the cultivated parts and to distinguish the main pathways of sediment transport from the slope to the valley bottom, area of the valley bottom and some other elements of relief which are influenced on the connectivity between cultivated slope and valley bottom. Previously determined values of sediment delivery ratio coefficient were applied for evaluation of the proportion of eroded from the cropland sediment delivered to the valley bottom (Panin et al., 2001). In the result, it was possible to evaluate preliminary sediment budgets for the key catchments. The more detail description of methods and approaches used for sediment budget calculation can be found elsewhere (Golosov, 1998; Golosov and Belyaev, 2009 etc.).

Table 1. Land use structure in the Mesha river and the Veduga river basins for two time intervals and changes of the land use structure

Land use type	Area, hectare	% of total area	Area, hectare	% of total area
Mesha river basin				
	1984-1988 period		2013-2016 period	
Woodland	77203.4	17.7	83383.4	19.1
Grassland	75263.5	17.3	86413.2	19.8
Cropland	269116.5	61.8	241886.1	55.5
Waterbodies	1958.5	0.5	2193.1	0.5
Settlements	12209.8	2.8	21875.9	5
Total	435751.7	100.0	435751.7	100.00
Veduga river basin				
	1984-1988 period		2015-2016 period	
Woodland	6801.9	5.7	10893.5	9.1
Grassland	18889.5	15.8	19644.5	16.5
Cropland	82877.5	69.4	75395.1	63.2
Waterbodies	300.9	0.3	351.9	0.3
Settlements	10477.1	8.8	13061.8	10.9
Total	119346.9	100	119346.8	100

It was found that cropland area decrease in the both studied river basins on about 9-10% during past 30 years due to mainly cultivated land abandonment (Table 1). Crop management coefficients (CMC) reduced at the Mesha River basin due to growth of proportion of perennial grasses in crop rotation. CMC don't change at the Veduga River basin. Growing of air temperature during last decades lead to serious reduction of coefficient of runoff from the cultivated slope in the period of snow-melt (Petelko et al., 2007) in the both regions. In addition, frequency of extreme rain-fall decrease at the

Mesha River basin and slightly increase at the Veduga River basin. The potential annual erosion rates were calculated for the both river basins for 1980 and 2015, assuming that all cultivated lands are used as fallow, and the map of soil losses were constructed (Fig.2). It is allowing to compare the soil losses without possible temporal-spatial variation of CMC. It was established that potential annual erosion rates at the Mesha River and in the Veduga River basin were 23 t ha^{-1} and 11 t ha^{-1} respectively in 1980. Given relationship between soil losses for two basins in a good agreement with assessment of mean annual soil losses for Tatarstan Republic and Voronezhskaya oblast' on the middle 1980^{th} (Litvin,2006). The trend of reduction of potential soil erosion rates was found for the both river basins to 2015, if it is compare with 1980, due to serious reduction of soil losses during snow-melt.

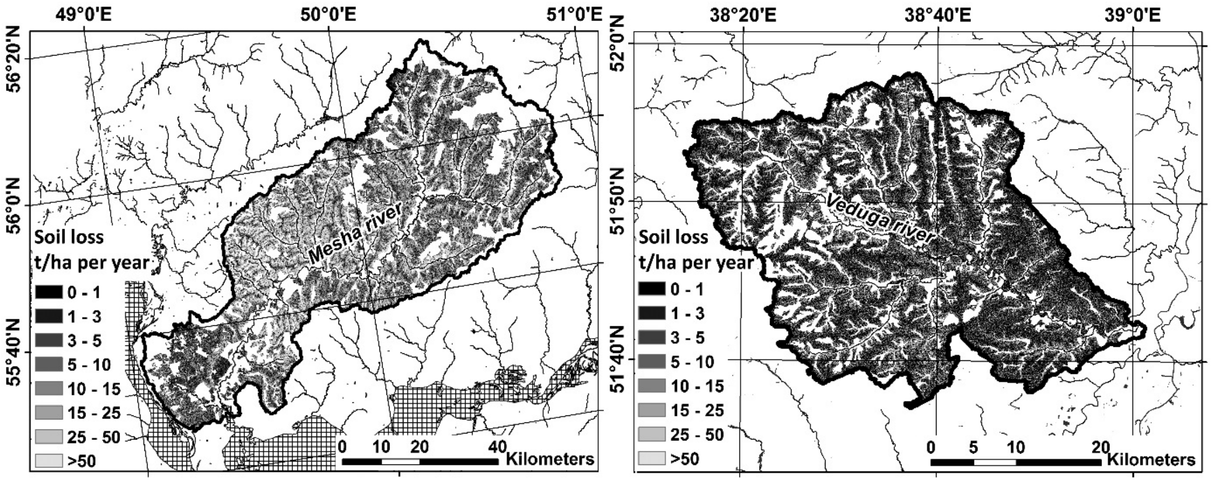

Figure 2. Maps of potential soil losses from cultivated fields of the Mesha River basin (left) and the Veduga River basin (right), constructed based on modified version USLE and SHI empirical models.

Results of the detailed assessments of sediment redistribution rates for periods 1963-1986 and 1986-2015 at the key catchments were used for verification of the results of soil erosion rate calculations within the studied river basins. Both key catchments are characterized by a high proportion of cultivated land, which, according to the available evidence (topographic maps, aerial photographs, satellite imagery and data archives), has not changed appreciably since the 1950s. Both bomb- and Chernobyl-derived ^{137}Cs were used as chronological markers for documenting changes in soil erosion rates on croplands during the last 50-55 years in key catchments. The sediment accumulation rates in the dry valley bottoms reflect the intensity of soil erosion on the slopes of the dry valley catchments (Golosov, 1998). Eight ^{137}Cs depth profiles located in representative cross-sections along the dry valley bottoms have been used to document sediment accumulation rates at key catchments. It was established that the mean annual sedimentation rates had reduced in 4-5.4 times in the both key catchments (Table 2). In case of Mesha River basin key catchment is located in area with slightly higher erosion rates if it is compare with mean values for the entire river basin (see Fig.1). The calculated mean annual erosion rates had reduced from 8.4 to 7.1 t ha-1 (the Mesha key catchment) and from 5.5 to 5.0 t ha^{-1} (Veduga key catchment) in 1963-1986 and 1986-2015 respectively. It is very likely that soil losses were underestimated for 1963-1986 due to incomplete estimation of erosion losses during snow melt. The ephemeral gully erosion usually had observed in the hollows due to water concentration, and this type of erosion isn't calculated by the model SHI. Soil losses from cultivated hollow catchments is almost one order magnitude higher if it is compare with soil losses from slope without hollows according field monitoring data obtained as a result of 15 years of observations during snow-melt in the southern part of the forest zone of the European Russia (Litvin, 2002).

Table 2. Estimates of mean annual sedimentation rates in dry valley bottoms at the key catchments for different periods based on interpretation of ^{137}Cs depth profiles.

Dry valley location	Landscape zone	Co-ordinates	Mean annual sedimentation rates, cm yr$^{-1,}$ for indicated time interval		Reduction of sedimentation rate
			1963-1986	1986 – 2015	
Veduga River basin	Forest-steppe	51.760539N 38.602139E	2.0 ± 0.2	0.5 ± 0.05	x 4
Mesha River basin	North of forest-steppe	55.645670N 49.646451E	1.4 ± 0.15	0.26 ± 0.05	x 5.4

CONCLUSIONS

The decreasing trend of erosion rates during last 30 years was established for the two typical small river basins of the forest-steppe zones of the Russian Plain due to mostly serious reduction of erosion rates during snow-melting. The erosion rate trend established on the model calculations in a good agreement with a quantitative assessment of sediment redistributions in the key catchments, located within the studied river basin. However, the high reductions of sedimentation rates in the dry valley bottoms of the key catchments during 1986-2015 if it is compare with 1963-1986 allows to suggest, that it is very likely that SHI model underestimated the annual erosion rates during snow-melt for period 1963-1986.

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4.8.O

ASSESSING ENVIRONMENTAL SENSITIVITY TO DESERTIFICATION IN AREAS AROUND IDKU LAKE (EGYPT) UTILIZING AN ADJUSTED MEDALUS MODEL

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ABSTRACT

Desertification is the consequence of multiple processes in arid and semi-arid environments, where water is the main factor limiting ecosystem development. Several factors may cause desertification, including climate change and human activities (Parvari *et al*., 2011; Torres *et al*., 2015). In the United Nation Organization Development and Environment Conference in Rio de Janeiro (1992), desertification was defined as: "*Land degradation in arid, semi-arid and dry sub-humid areas affected by human activities and climate changes"* (Khosravi *et al.,* 2013). Several methods have been developed to assess desertification processes, including direct observations and measurements, parametric equations, mathematical models and remote sensing. Compared to direct field observations, remote sensing data are cost-effective, time-efficient, and valuable in mapping land degradation risks (Gao and Liu; 2008; Li *et al*., 2007; Mansour *et al.,* 2011).

Several methods have been developed to assess desertification processes, including direct observations and measurements, parametric equations, mathematical models and remote sensing. The 'Mediterranean Desertification and Land Use' (MEDALUS) approach (Kosmas *et al.,* 1999) is simple, robust, widely applicable and adaptable (Kosmas *et al.,* 2003).

The study area is located to the west of the Nile Delta in UTM Zone 36 (31º02'30"-31º29'00"N; 30º01'20"-30º32'05'E) and covers 1092 km^2. The lands around Idku Lake are considered as problematic and reclamation costs are high (MALR, 1994).

The objectives of this paper are:
1) To develop a geographic soil database for the area that can be used for the development and management models required by decision-makers.
2) To assess and map areas sensitive to land degradation processes in the Idku Lake area, using the desertification method of the ESAs and spatial analyst tool in a GIS.
3) To use these results as a basis to propose possible actions to combat desertification in the most affected areas.

Digital image processing was executed for a Landsat ETM+ satellite image (path 177, row 38) acquired during 2013. The landform map was generated from the enhanced Landsat ETM+ images and the SRTM using ENVI 5.1 software (Dobos *et al.,* 2000, 2002). The physiographic units were defined and classified into groups and then the map legend was established adopting the procedures of Zinck and Valenzuela (1990).

Two sample areas were investigated in detail, typically covering ~10% of the investigated area. A detailed morphological description of 14 soil profiles was performed, adopting the procedures of FAO (2006). A total of 56 representative disturbed soil samples were collected, which were then air-dried and the fine-earth (<2.0 mm) fractions were used for chemical analyses, adopting the procedures of USDA (2004). The soils were classified to the Sub-Great Group level, based on USDA (2010).

The MEDALUS Model was developed to calculate the Environmentally Sensitive Area (ESA) Index, which is used to determine desertification patterns and processes. Sensitivity to desertification is defined by an index (SDI: Sensitivity to Desertification Index) obtained from the geometric mean of three indices of soil quality, climate and vegetation. The value of each index is divided into several classes, the thresholds of which have been determined empirically from extensive field work.

The major landscapes within the study area are coastal, lacustrine and flood-plains.The marine plain covers 87.1 km^2 of the northern part of the Nile Delta, and is represented by the landforms of sand-sheets (58.9 km^2), sand-dunes (14.8 km^2) and hummocks (13.4 km^2). Lacustrine plains dominate the middle parts of the area; covering 159.5 km^2. The floodplain is the main landscape unit and dominates the southern parts of the study area, covering 803.5 km^2. The different landforms within this landscape are decantation basins, high river terraces, low river terraces, overflow basin and river levees, with areas of 342.8, 281.0, 113.0, 46.2 and 20.5 km^2, respectively. The main Sub-Great Groups on the flood-plain are Typic Torrifluvents, Vertic Torrifluvents and Typic Natriargids and represent 42.2, 27.1 and 4.3% of the total area, respectively. On the lacustrine plain, Vertic Torrifluvents and Aquic Torripssaments are the main Sub-Great Groups and cover 6.2 and 1.3%, respectively, of the study area. The marine plain has Typic Torripssaments, covering 5.4% of the area.

The soil quality of the study area is classified as high (value <1.2), moderately high (value 1.2-1.4), moderately low (value 1.4-1.5) and low (value >1.5). These qualities represent 29.75, 43.33, 25.41 and 1.51% of the total area, respectively**.** The low soil quality class occur in the southern part of the Idku Lake area, resulting from high water-table levels (<50 cm from the surface) and high salt contents in soil (≤16.43 dS/m) and irrigation water (≤3.96 dS/m).

The Climate Quality Index (CQI) is assessed based on the amount of rainfall, aridity and slope aspect parameters. The results indicated that 56.14 and 43.86% of the area were assigned as low (value >1.81) and moderate (value 1.81-1.15) quality, respectively. The lowest CQI occurred in the south-east of the study area, which will induce low crop productivity (Thornes 1995; Bahreini and Pahlavanravi, 2013).

The Vegetation Quality Index (VQI) is given by the geometric mean of the indexes for erosion protection, drought resistance and vegetation cover. The low vegetation quality class occurred in the southern part of Idku Lake, representing 4.92% of the total area. Low vegetation quality is mainly due to the low density of plant cover, resulting from high water- table levels and high salt contents in soil and irrigation water.

The adjusted MEDALUS Model was used to define the desertification characteristics of various types of ESA. The Model applies a geometric mean of the three quality indexes, in order to provide a sensitivity diagnosis. The distribution of areas sensitive to desertification shows that 4.9, 51.2 and 12.7% of the study area were classified as very sensitive, highly sensitive and moderately highly sensitive, respectively. The moderately sensitive zone covers 31.2% of the study area. Causal factors include degraded soil properties, poor vegetation cover and land mismanagement.

Most of the study area is classified as highly to moderately sensitive to desertification. Highly sensitive areas are in the southern shores of Idku Lake. We suggest that in order to mitigate desertification, particularly on the southern shores of Idku Lake, soil properties should be improved. This will involve decreasing soil salinity to acceptable limits, which can be achieved by leaching, establishment of suitable drainage networks for lowering high water-tables and improving water quality. Thus, decision-makers should pay more attention to the areas which are most sensitive to desertification.

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4.9.O

ASSESSING THE WATER EROSION'S RISK ON THE MADEIRA'S ISLAND

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INTRODUCTION

A storm of 20 February 2010 in Madeira's island, with intense water erosion, triggering huge debris flows, was responsible for devastation and the losses of 50 lives. The Madeira Island, was an area of 742.0 km^2 with the maximum altitude of 1858 m (average 689.7 m), is a mountainous island with an average slope of about 65%, surpassed in 40% of the area. The annual precipitation ranges between 420 and 3 125 mm (average 1380 mm).

The objective is the assessment of the rainfall erosion's risk on the island; to obtain rainfall erosion rates and provide erosion maps, the RUSLE methodology was adopted. The main complexity was obtaining the annual erosivity (R) for the entire island, as, besides the Funchal-Observatório climatic station, there are few stations with rainfall records of 10 minute discrimination, most with durations of less than 10 years. The scarcity of data, lead to establish relations for assessing the erosivity; the analyses showed suitable relations with the Modified Fournier Index (MFI), and MFI and R. The practise allowed the calculation of the annual R for the Island, and mapping of erosivity parameter; other RUSLE parameters where obtained following normal recommended procedures.

From the preliminary results, one can indicate that the average Madeira's erosion is ≈ 28 t.ha^{-1}.yr^{-1}. The extent of "in situ" erosion's threat is obtained by dividing the soil surface horizons' volume by the amount of the annual erosion; the result is expressed as the number of years required for the total exhaustion of soil cover; in average, the estimated duration for soil resource depletion exceeds 21 000 years. For **high and very high** erosion's threat (exhaustion in < 100 yr) the areas are very significant, matching a surface of about 263 km^2 (35.6 %), for **very high threat** (depletion in < 10 yr) the area is about 40 km^2 (5.4 %); high slope areas relate to high (or very high) erosion risks.

METHODOLOGY

A brief description of the methodology employed in this study is presented. A **USLE/RUSLE** simplified methodology was adopted (Wischmeier and Smith, 1978; Renard et al., 1997) The estimation and the surface distribution of the incidence levels of rainfall erosion followed the of Wischmeier methodology - Universal Soil Loss Equation USLE with adjustments included in RUSLE – Revised Universal Soil Loss Equation with conditions of the available data. Each parameter of the USLE/RUSLE procedure is rendered by a factor:

$$E = R \times K \times LS \times C \times P$$

Where: E – estimated value of Soil Loss ($t.ha^{-1}.yr^{-1}$); R – Rainfall Erosivity Factor ($MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$); K - Erodibility Factor ($t.h.MJ^{-1}.mm^{-1}$); LS - Morphology Factor (-); C - Crop Cover Factor (-); P - Agricultural Practice Factor (-). The characteristics for each parameter are briefly stated.

RAINFALL EROSIVITY - The EI_{30} (Wischmeier – R), require high discrimination rainfall records. The erosivity - R_i - for a set of events (m), with duration (i), in $MJ.h^{-1}.mm.ha^{-1}$, is set by the equation:

$$R_i = \sum_{j=1}^{m} E \times I_{30j}$$

Reference periods usually are in years; the value of index EI_{30}, for each rain event (j), is given by the product of kinetic energy of rainfall by the maximum rainfall intensity with 30 minutes duration (I_{30maxj}). The values are expressed per unit area and time, in $MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$.

The evaluation rainfall erosivity (R) is complex; because besides the Funchal station, there are only a few stations with precipitation records with 10 minute discrimination, but with less than 10 years. With scarce data one had to rely on correlations to estimate the distribution of erosivity; the analyses showed appropriate correlations for Modified Fournier Index (MFI) and R. This allowed mapping of RUSLE's R. Other RUSLE parameters where obtained following standard procedures.

RAINFALL EROSIVITY PARAMETER – R - The assessment of erosion results from the calculation of the Wischmeier parameters. Major importance was assumed in estimating the rainfall erosivity and in establishing of their spatial distribution. Regression equations were used, on a regional and annual basis as well, as other relationships between R and the Modified Fournier Index (MFI), to outcome the data scarcity and to preserve the necessary accuracy of the erosivity estimation.

The expression presented next, results from the adjustment of the mentioned relationships, based on available data for a longer period at Funchal station, meteorological stations in Southern Portugal and data for the island of Sicily. The factor – R - and their distribution on the island, resulted from 26 climatic stations corresponding, generally, to the period of hydrologic years of 1949/50 to 1979/80.

$$R_2' = 1.365 \times MFI^{1.408}$$

Where: R_2' - average annual erosivity ($MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$) and MFI - Modified Index Fournier (mm). For the coastline the values were adjusted, by interpolation from the Madeira's rainfall map published by Plano Regional da Água da Madeira (Madeira's Regional Water Master Plan).

EROSION LEVELS **-** Universal Soil loss Equation (USLE/RUSLE) - The procedures prescribed in the Agricultural Handbook, Nr. 537 (Wischmeier and Smith, 1978) including adaptations from the Agricultural Handbook, Nr. 703 (Renard et al., 1997) to evaluate erosion. Besides the aspects mentioned for erosivity, some relevant aspects, regarding the asses of soil erodibility, morphology factor, crop and agricultural practice factors are briefly referred.
For **soil erodibility – K** – which is relatively complex and requires knowledge of the local conditions, the information was obtained from the report of Madeira's Island Soils Map; in case there was not sufficient analytical data values resulting from the experience were adopted; to obtain the **morphology factor – LS** – the estimates of slope length were based on the drainage density; the parameter L was assessed using ArcMap software tools and the slope factor – S - was produced directly through the same software tools, from a digital terrain model (DTM).
The calculation of the **crop cover factor – C** – was based on data of land use classes of Madeira's land use map (Carta de Ocupação do Solo da Região Autónoma da Madeira – COSRAM). In this study

the **agricultural practice factor – P** – was considered equal to 1.0 to the entire territory; in places with terraces the value 0.1 was assigned.

SOIL EROSION THREAT (in situ) - Adopting as reference a recent study for Sicily, the threat of soil erosion was estimated as the time (number of years) in which the total exhaustion of soil cover resource should occur. Same threat classes have been adopted: a) **low** - more than 500 years to occur total erosion; b) **moderate** - time to exhaustion varies between 100 and 500 years; c) **high** - resulting total erosion between 10 and 100 years; d) **very high** - when, at most, in 10 years the total soil exhaustion occur. The soil apparent specific weight adopted is 1.7 $t.m^{-3}$. From the distribution of rainfall erosion and soil thickness it was possible to calculate the time for the total exhaustion of soil.

ANALYSES OF RESULTS - A Table with values of Average Annual Precipitation **P** (mm), Modified Fournier Index **MFI** (mm) and Rainfall Erosivity **R** ($MJmmh^{-1}ha^{-1}yr^{-1}$) for climatic stations in Madeira with more years is presented. The distributed Modified Fournier Index (MFI) values obtained for the Island based on 26 stations and a methodology for assessing distribution, range from 81.0 to 465.0 mm; the rainfall erosivity – R - in annual basis, was then obtained. These values present an average value of 2 964, ranging from 660 to 8 515 $MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$.

Table - Average Annual Precipitation P (mm), Modified Fournier Index MFI (mm) and Rainfall Erosivity R ($MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$) for the climatic stations.

Climatic Station	P (mm)	MFI (mm)	R ($MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$)
Areeiro (antigo)	2955.2	495.7	8511.3
Bica da Cana - IGA	2984.8	464.6	7769.3
Camacha	1539.0	268.7	3594.5
Canhas	811.0	147.0	1537.5
Cascalho	1863.8	262.2	3471.9
Curral das Freiras (Igreja)	1957.6	392.8	6133.1
Encumeada de S. Vicente	2723.2	470.6	7910.2
Funchal - Observatório - IM	628.4	128.9	1277.0
Lugar de Baixo – JG	611.6	121.6	1177.0
Montado do Pereiro	2357.9	409.6	6507.3
Ponta Delgada	1160.1	178.9	2027.3
Porto Moniz (Feira do Gado)	1340.5	207.4	2495.6
Queimadas	2263.8	344.4	5096.1
Ribeiro Frio	2404.3	406.5	6436.4
Sanatório	873.1	172.1	1918.9
Santana	1432.0	226.7	2828.8
Santo António (Trapiche)	991.0	199.5	2363.3
Santo da Serra - IGA	1806.3	296.9	4135.8

Following the developed methodology, for Funchal, with P ≈ 630 mm and MFI ≈ 130 mm, the estimated value of annual average erosivity is R ≈ 1280 $MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$. For Funchal's river basins - P ≈ 1900 mm and erosivity is R ≈ 3800 $MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$. For the storm event of February 20, 2010 - the rainfall was P = 147 mm, with maximum 30-minute intensity I_{30} ≈ 62.8 mm h^{-1} and an erosivity R ≈ 2 145 $MJ.h^{-1}.mm.ha^{-1}.yr^{-1}$. This value is about 1.7 fold the average rainfall erosivity and the obtained return period for the event is in the order of 80 years.

OTHER WISCHMEIER PARAMETERS – For the soil erosivity factor, K, the values range between 0.0001 and 0.0192, the average factor is close to 0.0080 ($t.h.MJ^{-1}.mm^{-1}$); the slope-length factor, L, presents values between 1.28 and 4.75, the calculated values for the slope factor, S vary from 0.03 and 16.12 and the product of these the - morphology factor, LS, range to a maximum of about 76.4 with average of 22.7; for the crop cover, C, the values, depending on various classes of soil use and land occupation, vary from 0.001 and 0.35, with an average of around 0.05. Since the area of terraces is significantly small the agricultural practice factor, P, as the average value of 0.98.

APPLYING USLE/RUSLE - The mean estimated value of erosion is ≈ 28 $t.ha^{-1}.yr^{-1}$. The surface distributed values vary from zero up to the maximum of 920 $t.ha^{-1}.yr^{-1}$; the value of 100 t ha^{-1} yr^{1} is exceeded in more than 4.9% of the area. It is observed that central zone of Madeira's Island, as well more steep slope areas located along the coast, present high rainfall erosion levels.

SOIL EROSION THREAT (in situ) - Dividing the volume of soil surface horizons by annual erosion rates erosion's threat result; the average soil depletion value in average exceeds 21 000 years. For maximum threat of erosion - **very high and high** - the area is approximately 263 km^2 (35.6%) for - **very high** - the area is about 40 km^2 (5.4%). In steep slopes the erosion threat is very high or high; most areas classified as **very high** threat, are the areas mostly affected by storm of February 20, 2010. The biggest "off site" impacts occurred in densely urbanized areas, as downtown of Funchal.

CONCLUSIONS

- From rainfall erosivity studies one can extract the major conclusions:

- Very good relations were detected between Wischmeier rainfall erosivity R and precipitation P (for Wischmeier events), as well as between the rainfall in Wischmeier events and rainfall days with more than 10 mm (erosive rainfall); some good or very good relationships between the monthly rainfall erosivity and monthly precipitation or- monthly erosive rainfall were obtained.

- Monthly, the relationships for the monthly rainfall erosivity and monthly Fournier Modified Index (MFIm) are much better than the monthly relations with precipitation. Annually, the relationships for annual rainfall erosivity and MFI, or annual rainfall are not as good, being advisable that the annual rainfall erosivity estimates to be obtained by adding estimated monthly values. Is recognized the advantage of obtaining regional relations of the type $R = a.P^b$, or $R = c.MFI^d$, on monthly or annual basis. The used methodology showed robustness and justified the work.

To assess the erosion threat, accounted as soil exhaustion time, major conclusions follow:
Rainfall erosivity of the February 20, 2010 storm at Funchal's climatic station showed impressing values, with a value around **R = 2 145** MJ.mm $h^{-1}.ha^{-1}$, about 1.7 times the average annual erosivity. The estimated return period for the storm is in the order of 80 years. A more accurate evaluation of the return period associated with rainfall erosivity must still be obtained.

As shown, the mean estimated value for rainfall erosion in Madeira is about 28 $t·ha^{-1}·yr^{-1}$ with a maximum value of about 920 $t·ha^{-1}·yr^{-1}$. The highest values occur in the central area of the island and in steep slopes along the coast and in the valleys of the watercourses. In urbanized areas, as the case of Funchal and other boroughs, the values of erosion are relatively small.

For maximum erosion's threat - **very high and high** - the area is approximately 263 km^2 (35.6%); for - **very high** - the area is about 40 km^2 (5.4%). In steep slopes the erosion threat is very high or high. Most areas of **very high** threat are the areas major affected by storm of February 20, 2010. The

biggest "off site" impacts occurred in densely urbanized areas, as downtown of Funchal. The methodology shown very suitable for assessing the threats of erosion but there is a need for improvement. Records for extreme events are still being analysed.

KEYWORDS: land degradation; erosion risk, precipitation, rainfall erosivity, revised universal soil loss equation.

4.10.O

COMPARISON OF EROSION MODELLING BASED ON HIGH-RESOLUTION RADAR RAIN DATA WITH AERIAL PHOTO EROSION CLASSIFICATION

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INTRODUCTION

The Universal Soil Loss Equation (USLE) is the most frequently applied erosion prediction model. The equation of the USLE is originally based on complex algorithm developed from a large dataset of ~10,000 plot years of standardized field plots, and is used for the prediction of long-term annual soil loss (Wischmeier and Smith, 1978; Renard *et al*., 1997; Foster, 2005). Due to large differences between plot and strongly varying real field conditions (Boix-Fayos *et al*., 2006; Auerswald *et al*., 2009), the USLE was already validated by different approaches (e.g. Cohen *et al.,* 2005; Konz *et al*., 2009; Pradhan *et al*., 2012;…). But hardly any of the existing validation methods could quantify sheet and rill erosion exclusively and at the same temporal and spatial scale as the USLE. In particular, the validation should also include the errors that result from data and procedures which are needed in the typical application case. Therefore, the approach of this study was to validate soil loss predicted with methods and data sources from the official erosion modelling of Bavaria (Germany) by a comparison to erosion classifications from aerial photos which were taken shortly after erosive rain events. Soil loss was predicted with high-resolution radar rain data for many fields with different crops (e.g. grassland, spring and winter small grain, maize, potato, soy bean, beet, hops,…) and different cultivation methods (conventional and mulch tillage).

MATERIAL AND METHODS

About 2500 aerial photos were used for soil loss classifications of 8100 fields located in Bavaria (Germany). Visual soil loss classes 0, 1, 2, and 3 where assigned to the individual fields when 0%, <10%, 10-30% and >30% of the field area were covered by signs of erosion, respectively. To reduce subjectivity of the classifications, four classification rounds were run out in total. In the end, 6 140 fields were visually classified once, 1 399 fields twice, 517 fields three times and 45 fields were classified four times. The classifications were compared and analysed for the validity of this approach by regressions. For the comparison with predicted soil loss, the median soil loss class of the individual fields was used, resulting also in intermediate classes (0.5, 1.5, 2.5). The USLE predicts the long-term soil loss by the product of six factors quantifying the effects of rain (R), soil (K), slope length (L), slope steepness (S), land use (C) and permanent erosion control measures like terracing and contouring (P). As series of erosion events were examined, the factor C was replaced by the soil loss ratio (c_{SLR}) which is depending on crop and cultivation method. The factor c_{SLR} is defined as the relative soil loss compared to a bare fallow under otherwise same conditions. The values for c_{SLR} were taken from literature (e.g. Schwertmann *et al*., 1990) considering the respective crop and cultivation method registered for the individual fields in an official inventory. The R factor of the single events was calculated from high-resolution (1 x 1 km², 5 min), rain-gauge adjusted radar rain data (RADOLAN; Fischer *et al*. 2016). Data for the factors *K*, *L*, *S* and *P* were mainly taken from the official high-resolution (5 x 5 m²) erosion cadastre of Bavaria implemented in ArcGIS and maintained at the Bavarian State Research Centre for Agriculture (Bayerische Landesanstalt für Landwirtschaft, 2010; Kagerer and Auerswald 1997). Soil loss was predicted for all events between the last soil tillage and the taking of the aerial photo and the sum of soil loss was compared to the classified degree of erosion damage visible on the aerial photos for individual fields.

RESULTS

The visual soil loss classifications correlated very highly significantly ($r^2 = 0.74 – 0.78$, $p < 0.001$) and regression lines were near to the line of unity with slopes between 0.9 and 1.1 and intercepts between 0 and 0.2. Differences of soil loss classes between the classifications rounds correlated with field conditions like field size (Spearman's rho = 0.1, $p < 0.0001$) or maximum slope length (rho= 0.1, $p < 0.0001$).

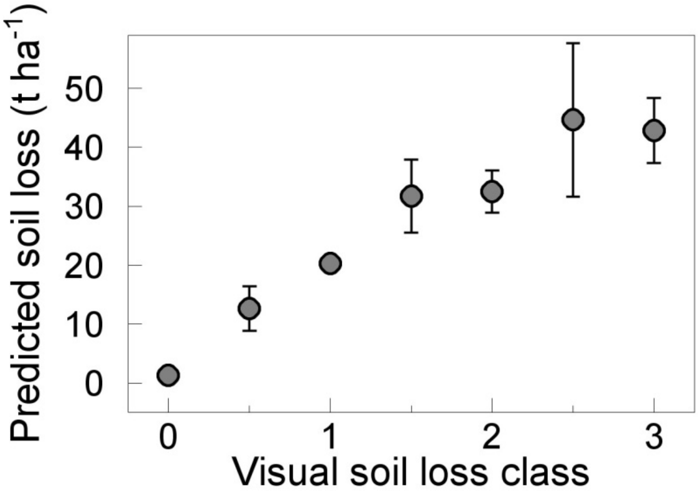

Figure 1 : Mean predicted soil loss (t ha^{-1}) per visual soil loss class with its 95% confidence interval (not always visible).

Predicted soil loss ranged from 0 t ha^{-1} to 400 t ha^{-1} for individual fields. The prediction and visual classification of soil loss correlated very highly significantly (rho = 0.68, $p < 0.001$). Mean predicted soil loss increased from 1 t ha^{-1} in visual soil loss class 0 to 19 t ha^{-1} in class 1, 31 t ha^{-1} in class 2, and 43 t ha^{-1} in class 3 (Figure 1). The 95% confidence intervals were narrow, while scatter of predicted soil loss was large in all classes (standard deviation between 7 t ha^{-1} and 46 t ha^{-1}). The intermediate classes showed large scatter due to their low number of fields within these classes (1% of all fields respectively).

Also, all individual USLE factors, except factor *K*, correlated very highly significantly with predicted and with visually classified soil loss in a similar ranking. The factor c_{SLR} explained far more of the variation of the visual soil loss classification of arable land (rho = 0.71, $p<0.001$) than the factor *R* (rho = 0.11, $p<0.001$) although higher influence of *R* would be expected. This was caused by the lower spatial resolution of the data for *R* than for c_{SLR}. Within one rain radar pixel of 1km², *R* has to be assumed to be constant while c_{SLR} varies strongly depending on different crops growing on the 20 fields, theoretically located in this pixel. The site specific factors *L*, *S* and also *P*, correlated similarly to visually classified soil loss (rho = 0.14 – 0.18, p<0.001). The high influence of these factors was caused by the fact that *L* and *P* were influenced by maximum erosive slope length which was again also highly correlated to visual soil loss classifications (rho = 0.22, p < 0.001). The lower correlation and lack of correlation of factor *K* to predicted and visually classified soil loss was caused by the low variation of *K* (coefficient of variation 0.25).

Furthermore, mean predicted soil loss and mean visual soil loss class of different crops were ranked in a similar order (Table 1). Hops, maize, beet and potato had the largest visually classified and predicted soil loss while small grains and grassland had lowest loss of all crops. Fields with mulch tillage had lower predicted and visually classified soil loss than fields with conventional tillage.

Table 1 Mean predicted soil loss and mean visual soil loss class, and number of fields *n* for an excerpt of crop groups. Hops, maize and beet & potato are separated by cultivation method with conventional (conv.) and mulch tillage. The group grain legume contains pea and broad bean.

Crop group	Mean predicted soil loss (t ha^{-1})	Mean visual soil loss class	*n*
Grassland	0.3	<0.1	3463
Winter wheat	0.4	<0.1	1007
Spring barley	0.2	0.1	242
Oat	0.3	0.1	143
Grain legume	1.4	0.3	56
Canola	7.7	0.4	275
Beet & potato (mulch)	7.2	0.5	34
Beet & potato (conv.)	15.8	1.1	36
Maize (mulch)	18.9	1.1	526
Maize (conv.)	28.3	1.3	1509
Soy bean	21.0	1.4	10
Hops (mulch)	26.5	1.5	18
Hops (conv.)	42.2	1.9	32

DISCUSSION AND CONCLUSIONS

For the average of many fields, the visual classification of soil loss worked sufficiently well, and there was no reason to assume a general invalidity of the USLE and the official parameterization procedures. For individual fields, differences between prediction and visual classification could be rather large. Event predictions mainly suffered from errors in the assumed crop stage period and tillage practices, which cannot consider inter-annual and farm-specific variation. No other data source like e.g. remote sensing data was used to include this variation, as the approach was to predict soil loss following the official procedures of the Bavarian agricultural administration. The resolution of radar rain data seemed insufficient to predict short-term erosion on individual fields given the strong spatial gradients within individual rains (Fiener and Auerswald, 2009; Fischer *et al.*, 2016). The quality of the input data clearly controlled the prediction quality. Many cases where USLE predictions and observations deviate may likely be caused by parameterization weaknesses but not by a failure of the model itself.

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4.11.O

MULTI-SCALE COMPARISON OF SEDIMENT DISCHARGE IN AGATSUMA WATERSHED, JAPAN: OBSERVATION DATA VERSUS GEOWEPP DATA

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INTRODUCTION

Soil erosion can have a significant impact on agricultural activities and watershed management, affecting the sustainability of the land, the aquatic habitat, river aggradation, and reservoir sedimentation, which can lead to decreased water storage. Soil-erosion and sediment-yield prediction models have been developed and are widely used to predict and measure the effects of land cover changes and human activities in a watershed.

However, variability in watershed properties often prevents accurate prediction of runoff and sediment yields. Negi (2001) pointed out that different hydrological processes and factors are dominant at different watershed scales. The rainfall–runoff process defines the hydrograph at the hillslope and sub-catchment scales, the channel network system controls it at the mesoscale basin scale (up to 100 km^2), and spatial variability in precipitation becomes significant in large basins. Burns et al. (2010) attempted a multi-scale calibration to improve sediment yield prediction, using observation data from plots, hillslopes, and small watersheds for a physically based hydrologic model and a process-based soil erosion model and found that the sediment yield was apparently affected by the scale. Nevertheless, De Boer and Crosby (1996) reported that clear correlations between sediment yield and basin scale cannot be determined because the sediment yield may be controlled by the sediment supply in small parts of the basin area.

The geo-spatial interface of the Water Erosion Prediction Project (GeoWEPP) is a process-based model for the prediction and assessment of water and sediment yields in a watershed. As each process involving water and sediment dynamics is described physically, this model is capable of identifying critical areas at any location within a watershed. This study aimed to compare GeoWEPP outputs with observation data at various scales to examine the factors that can be used to characterize the relationship between sediment discharge and watershed scale.

MATERIALS AND METHODS

The study was conducted in Agatsuma watershed (Fig. 1), Gunma Prefecture, Japan. which has an area of 711 km^2 and elevations of 482–2,568 m. The watershed includes two volcanic areas, around Mt. Asama (2,568 m) and Mt. Kusatsu-Shirane (2,160 m). The soil type within the watershed is strongly influenced by volcanic activity, which results in high soil porosity. The land cover at the study site is forest (80%), agricultural (6%), and other (14%). Plantation is the dominant forest, and most of the agricultural area is used to grow cabbage. The climatic conditions were represented by the monthly rainfall and average temperatures at the Kusatsu and Tashiro stations, where annual average precipitation amounts to 1,672 mm and 1,476 mm, respectively. The wet season from April to October is defined by rainfall amounts exceeding 100 mm, and the highest rainfall occurs in summer (June to September).

This study used previous observation data collected by Namba et al. (2007), which consist of 11 years' daily discharge data obtained at Murakami Point and 617 suspended-sediment and flow-rate samples obtained at 96 subwatersheds in Agatsuma watershed. The daily discharge data were routine monitoring data observed at Murakami Point, located approximately 26 km downstream of Agatsuma watershed.

Using the lysimeter method, the observation plot data were obtained from other research conducted by the Forestry Agency (2007). Each observation plot had an area of 5 m^2, a width of 1 m, and a length of 5 m. Plots were observed in four types of forest and one cabbage field. The four types of forest plot were broadleaf forest, fir, larch, and cypress. The numbers of observations of the forest and cabbage plots were 184 and 72, respectively.

The GeoWEPP discharge and sediment yield comparison was conducted using a developed hydrological model and suspended-sediment rating curves. The hydrological model was developed using a rainfall-runoff model based on the tank model and the Modified Priestley–Taylor equation of Sawano et al. (2015) (Amaru et al., 2016). The suspended sediment rating curves were developed from suspended sediment observation data and flow-rate data according to the classified forest percentages. The forest coverage of 96 subwatersheds was classified into four classes: 0–25, 25–50, 50–75, and 75–100%. The forest percentages used in this study were forest class 2 (F2), forest class 3 (F3), and forest class 4 (F4). F2 represents the low-forest percentage and F4 is the high-forest percentage.

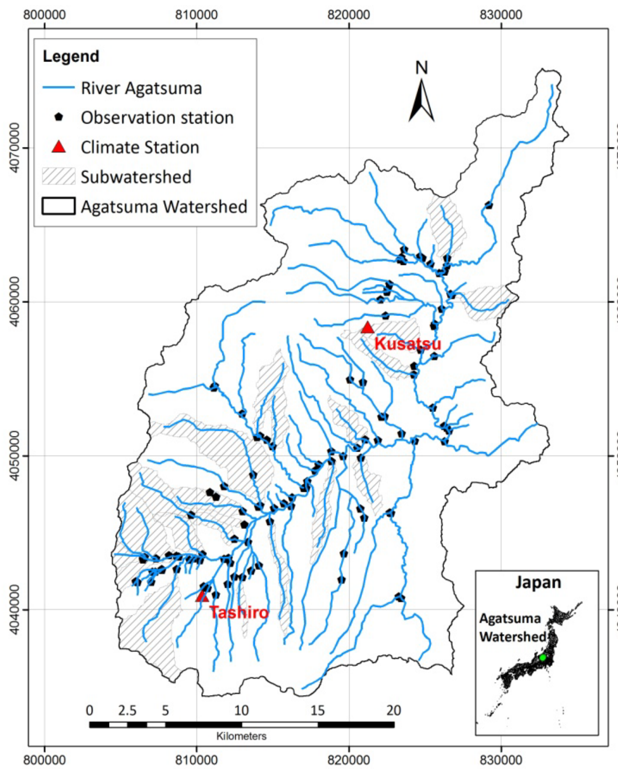

Figure 1. The Agatsuma watershed

A sensitivity analysis was conducted to identify the parameters that most affected discharge, soil loss on hillslopes, and sediment yield by examining soil interill erodibilty, rill erodibilty, saturated hydraulic conductivity, soil saturation level, and critical shear stress. In the channel section, the parameters used for the sensitivity analysis were main channel critical shear stress, channel erodibility, and depth to the non-erodible layer.

As many as 20 subwatersheds, with forest coverage ranging from 29% to 97%, were chosen to represent forest percentage and used to conduct the GeoWEPP simulation. The model inputs in this study were digital elevation model (DEM) data with 10-m resolution obtained from the Geospatial Information Authority of Japan, a land cover map with 10-m resolution from the Japan Aerospace Exploration Agency (based on ALOS AVNIR 2 version 201602), and a soil map from Agrimesh. The watershed method in the

GeoWEPP model was used to run the simulation. The watershed method consists of hillslope soil loss prediction and watershed sediment discharge at the outlet.

A comparison between observation data and GeoWEPP simulation values was conducted to ensure that the GeoWEPP model could represent the sediment discharge in the forested watershed. The GeoWEPP hillslope soil loss was compared roughly with plot observation soil loss for only the high- and low-forest percentages. The GeoWEPP sediment yield at the watershed scale was compared with suspended sediment rating curves developed based on forest percentage. The relationship between sediment yield and watershed area was assessed by the specific sediment yield to calculate the sediment yield per unit area (ton hectare^{-1} year^{-1}). The specific sediment yield was used to evaluate the relationship between the hillslope and watershed scales within the area of the watershed, to reveal any differences an d examine the influencing factors.

RESULT

The GeoWEPP hillslope soil loss seemed to be overestimated for the forest watershed. Plot observation data were compared with the GeoWEPP hillslope soil loss to detect the range of soil loss in several types of coverage. In the hillslope section, GeoWEPP predicted the soil loss for F2, F3, and F4 to be 5.15, 4.21, and 0.61 ton/ha/year, respectively. Plot observations in the four types of forest and the cabbage crop showed that the forest plots had an average soil loss of 0.0258 ton/ha/month, whereas the cabbage plot had an average soil loss of 0.471 ton/ha/month. If these values are compared with plot observation data for 1 year (5.652 ton/ha/year for cabbage and 0.309 ton/ha/year for forest), the GeoWEPP prediction of soil loss in the forest area (high-forest percentage) was overestimated, while that in the cabbage area (low-forest percentage) was underestimated.

The comparison results indicate a good-fit simulation result using GeoWEPP for discharge and sediment yield. The GeoWEPP discharge and sediment yield evaluations were conducted before the simulation was started. These evaluations were conducted to ensure that GeoWEPP could represent a mountainous and forested watershed in Japan. The linear correlation coefficients between GeoWEPP discharge and the hydrological model were 0.97, 0.95, and 0.93 for F2, F3 and F4, respectively. The sediment yield comparison between GeoWEPP and the suspended sediment rating curves produced values of 0.88, 0.91, and 0.59 for F2, F3 and F4, respectively. However, GeoWEPP still showed overestimation compared with the sediment yield calculated by the suspended sediment rating curves.

According to the sensitivity analysis of the GeoWEPP model, the main parameter influencing the discharge results was the restricted layer, which facilitates subsurface lateral flow and return flow to the river. In the channel section, the depth of the non-erodible layer was the main factor affecting the amount of channel soil loss.

The main source of sediment in the forest watershed was from soil loss in the channel section, as shown in Figure 2. The high-forest percentage (F4) showed less hillslope soil loss compared to the low forest percentage (F2), but it also showed a channel soil loss rate quite similar to that of F2. As a result, the sediment discharge at the outlet of the subwatershed was not very different from that of F2.

Figure 2 shows the sediment yields along the area of the subwatershed. The hillslope section shows that F2 had high soil loss because it was dominated by cabbage fields and the soil surface was sometimes tilled and bare. However, in F4, annual hillslope soil loss ranged from low to high soil loss. Soil loss in the hillslope section resulting from increased area shows that F2 had a flat trend, whereas the trend for F4 seems to have been slightly increasing.

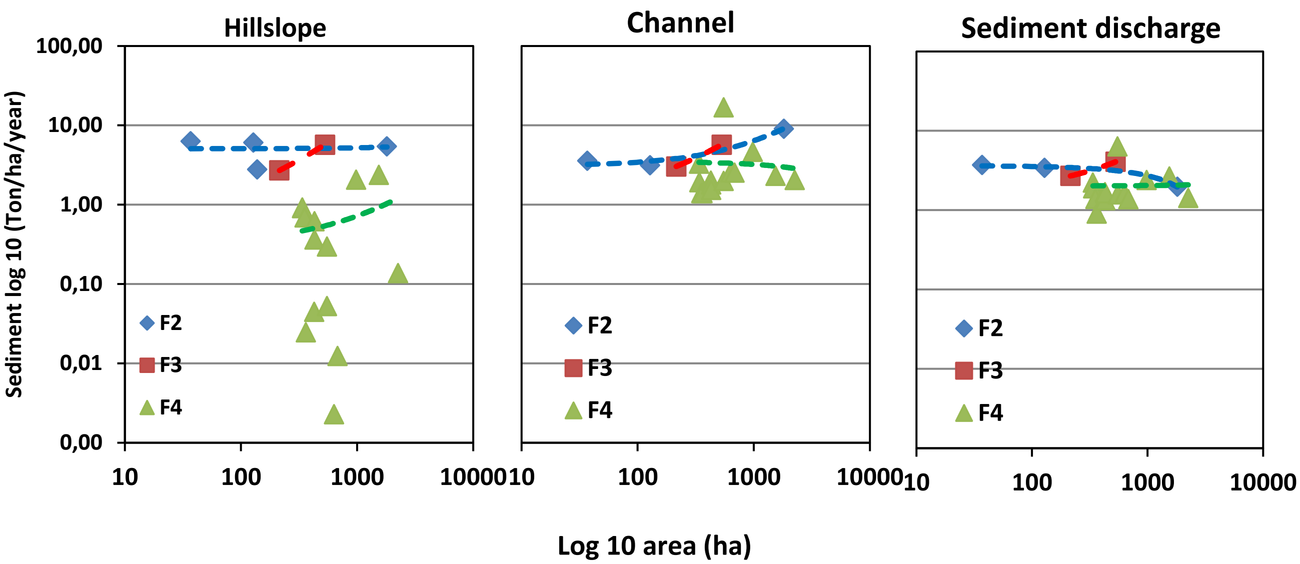

Figure 2. Relationship between Area and sediment in hill slope, channel and subwatershed outlet

Simulation of the channel section indicated that F4 and F2 had similar soil loss values, even though F4 had lower values than F2. The soil loss in the channel section increased slightly for F2 and decreased slightly for F4. The sediment discharge at the subwatershed outlet resulting from soil loss in the hillslope and channel sections is shown as “sediment discharges” in Figure 2. The results show that the sediment yield of F2 decreased slightly, while that of F4 appeared to have a flat trend with increasing area. Overall, the GeoWEPP model could not show the scale effect in the simulation, possibly because GeoWEPP has a limitation as regards modeling gully and stream-bank erosion processes. Additionally, the channel soil loss from sediment storage or supply requires further investigation.

After soil is eroded in hillslope and channel sections, the sediment has various probabilities of being transported down the slope or being deposited. The deposition process in the channel section appears to decrease the amount of sediment discharge found at the outlet of the watershed.

CONCLUSION

A hydrological model and sediment rating curves were developed to compare with GeoWEPP discharge results. The comparison showed that GeoWEPP can represent the hydrological processes in a mountainous forest watershed and represent the sediment yield. GeoWEPP was then used to predict soil losses and sediment yields in the hillslope, channel, and outlet portions of the subwatershed.

The WEPP soil loss predictions at the plot scale were overestimated in the forest plots but underestimated in the cabbage plot. Even though the GeoWEPP sediment yield results at the watershed scale showed a good fit with the observed suspended sediment, they still showed a tendency for overestimation.

The GeoWEPP hillslope soil loss for the high-forest-percentage region increased slightly with area increase. However, the GeoWEPP sediment discharge results considering the area of watershed did not show the scale effect.

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4.12.O

LAND DEGRADATION AS A DRIVER OF WINE ECONOMIC STRUCTURE

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INTRODUCTION

Land degradation is a worldwide issue which affects cropping patterns, agriculture structure and economics (Wu and Brorsen, 1995, Blaikie and Brookfield, 2015). In Mediterranean countries, in particular, land degradation has been frequently associated with specific land use and management types (Hill et al., 2008). Also viticulture has been included within the possible drivers of land degradation, especially because of soil losses due to intense erosional processes (e.g. Cerdan et al., 2010; Martínez-Casanovas et al., 2009; Prosdocimi et al., 2016), excessive earthworks (Costantini et al., 2015; Bazzoffi et al., 2006), nutrient depletion (Ramos and Martínez-Casanovas, 2006), compaction (Lagacherie et al., 2006), salinization and sodization (Crescimanno et al., 2007). The relationships between land degradation and viticulture, however, have been mainly investigated under a biophysical point of view, while the relationships with wine economics have been largely neglected. The effect of environmental characteristics on wine structure and economics has been verified at a national scale by Costantini et coll. (Costantini et al., 2016). In this study, we specifically focus on the possible influence of land degradation on the overall system of production and organization of the viticultural farms and wine industry.

MATERIALS AND METHODS

The investigation was carried out at the national scale in Italy, the largest wine production country of the world. We examined Italian viticultural lands singled out at the 1:500,000 scale, called viticultural “Macroareas”, which included Denomination of Origin areas (DO). We selected 32 out of the 92 Macroareas of the “Atlas of the territories of Italian wine” (Pollini et al., 2013), randomly distributed across the country. The 32 Macroareas resulted well representative of the environmental features of the Italian vineyard. The viticultural and oenological characterization of the Macroareas was based on the official data reported in the Italian wine production declarations (AGEA, Agenzia per le Erogazioni in Agricoltura, 2008). A set of variables was chosen to discover the relationships between the system of production and organization of the viticultural farms and wine industry, and land degradation. Four indices were chosen to represent different aspects of land degradation. They provide for a synthetic information on climatic conditions, vegetation, and soil quality, which are the main biophysical drivers of land degradation (Salvati et al., 2016). They are: i) the Index of sensitivity to desertification (ESAI), which identifies areas with increasing sensitivity to degradation by summing up the contributions of soil, climate, and vegetation qualities, and land management sustainability (Basso et al., 2000) ii) the soil quality index (SQI), which specifically focuses on soil fragility, taking into account texture, depth, available water capacity, and slope. The weights given to these soil properties relate to the resistance to water erosion (Basso et al., 2000); iii) the land vulnerability index (LVI), which describes the land vulnerability to dryness and desertification. It takes into account the changes occurred in climate, land use, coverage of vegetation, soil properties and population amount, during the last two decades (Salvati, 2009); iv) the soil aridity index (SAI), which represents

the number of days when the soil is dry in the moisture control section (Costantini and L'Abate, 2009).
A set of 19 oenological and viticultural variables was considered to characterize the production and organization system of the viticultural farms and wine industry. They regarded the general quantitative features of the wine producers, the typology of winery (independent, aggregated in cooperatives, afferent to industry), the incidence of independent farms producing wine, the quality orientation of the wine producers (DO, table wine), the acreage investment into vineyards. A multivariate and non parametric statistical approach was used to identify, among the 19 variables , the most correlated with the environmental degradation indices. A Spearman test was adopted with a significant p-level ≤0.5. Macroareas were clustered on the basis of the environmental degradation indices through a K-means analysis. A discriminant test was used to highlight the most powerful variables in separating the groups.

RESULTS AND DISCUSSION

The multivariate statistical analysis highlighted that the ESAI land degradation and LVI indices were inversely correlated with DO wine production. LVI and SQI were positively correlated with table wine production. Territories with high SQI values showed a large number of small wineries and farms conferring grape to the industry. Low values of ESAI, SQI, and SAI were all associated to a large diffusion of the vineyard land use in the grape suitable areas (i.e., agricultural area at altitude less than 800 a.s.l.) (Table 1)

Table1: Correlation between structural, oenological and viticultural variables, and land degradation indices (Spearman test: *p level<0.05). Nw: total number of wine producers; CWm: mean capability of the wineries; DO%: proportion of wine of denomination of origin; Tab%: table wine on the total wine production; Ev%: proportion of existing vineyards on grape suitable area; Gsa%: proportion of grape suitable area on total surface.

		Land degradation indices			
		LVI	ESAI	SQI	SAI
structural variables	Nw	-0,016	0,147	0,497*	-0,007
	CWm	0,169	-0,144	-0,486*	0,044
oenological variables	DO%	-0,476*	-0,458*	0,137	-0,305
	Tab%	0,485*	0,243	0,030*	0,279
viticultural variables	Ev%	-0,299	-0,578*	-0,455*	-0,352*
	Gsa%	0,498*	0,455*	0,124	0,335

Four vulnerability clusters (Fig. 1) were significant discriminates by the studied indices, although LVI had a lower discriminant power (p value= 0.056). LVI and ESAI showed a constant decrease from Macroareas with high sensitivity to land degradation to Macroareas of very low sensitivity, while pedoclimatic (SAI) and soil features (SQI) discriminated Macroareas of the high from the moderate class, and low from very low class.
DOC and Table wine production had a inverse trend from higher to lower land vulnerability, with better quality production matching low and very low degradation clusters. Few big wineries characterized both the very low and high sensitivity lands, but the wineries differed for typology,

either cooperative or industrial, respectively. Macroareas clustered within the intermediate classes of sensitivity were dominated by the presence of little, independent wineries.

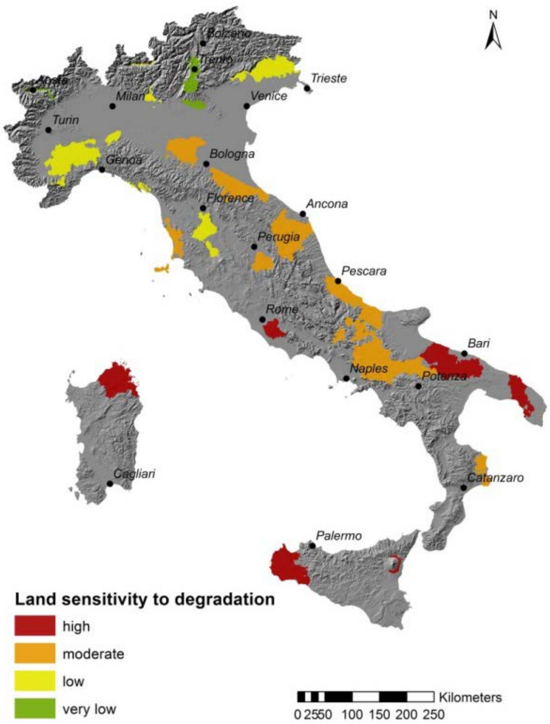

Figure 1 –Macroareas clusters with different land sensitivity class to degradation.

CONCLUSIONS

It is well known that land degradation and risk of desertification in Italy are relatively more widespread in the southern regions and toward the costs of the Mediterranean sea, while the action of the biophysical factors enhancing the sensitivity to land degradation and desertification, starting from climate, is relatively milder in northern and central Italy (Salvati and Bajocco, 2011; Costantini and Lorenzetti, 2013). Viticultural territories which developed in lands with high sensitivity to land degradation show a low intensity of viticulture and high a tendency to concentrate the production in a lower amount of large wineries, marketing table wine. On the other hand, viticultural territories which are relatively less sensitive to land degradation are those where viticulture occupies a larger part of the agricultural surface and mainly produces DO wines. This trend is particularly evident in the least sensitive to degradation territories. It is noticeable to observe that in these viticultural areas, similarly to the first class, there is a dominance of few wineries producing large quantity of wine but, in this case, in an associated cooperative form.

In conclusion, form this study it is evident that, at the broad scale, there is a linkage between land degradation and wine economic structure. This outcome would suggest a change in the strategy of agricultural policies, which usually separate agro-environmental measures from those addressed to structural interventions.

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4.15.O

SOIL EROSION, SOIL QUALITY AND CROP YIELD IN THE CHINESE MOLLISOL REGION

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INTRODUCTION

With the growth of economic and population in the world, agricultural sustainability has been considered as crucial for meeting food demand and economic development in developing countries. However, the increase rate at which corn, soybean and rice yields has slowed since 1995 (FAOSTAT, 2007). The contiguous areas of northeast China with black soil, chernozem and meadow soil are called the north-eastern black soil region, including three provinces (Heilongjiang, Jilin, and Liaoning) and the eastern part of the Inner Mongolian autonomous region, which is considered important for Chinese crop production (Xu et al., 2010), which provides one fourth to one third food supply in China. However, since large-scale cultivation of the region began in the 1950s, severe soil erosion has occurred and the thickness of Mollisol soils has decreased from 60–70 cm in the 1950s to 20–30 cm at present (Fan et al., 2005). In some places the loess parent material of Mollisol soil has been exposed to the surface, which reduces soil productivity (Wang et al., 2009). Consequently, the above changes further threaten the food security in China (Liu et al., 2010). Therefore, it is important to quantify how soil erosion affect soil quality and crop yield. The specific aims of this study were to analyze the impacts of soil erosion/deposition on soil characteristics and to select retainable soil quality assessment indicators, to discuss the corresponding relations of corn yield to soil quality and soil erosion/deposition rate, to fit equations among crop yield, soil quality, and soil erosion.

METHODS

The study was conducted in the Binzhouhe catchment, located in Bin County in the north of Heilongjiang Province in China. The catchment has a temperate continental monsoon climate, which is hot and rainy in summer and cold and arid in winter. The mean annual precipitation is 548.5 mm, and 80% of which is received from June to September. The mean monthly temperature is 3.9ºC. The dominant soil association in this study catchment is classified as Mollisol based on USDA Taxonomy. As regards soil use, the field devotes to agriculture with corn [Zea mays L.] being the dominant crop for several decades and irrigation is not used in the study area. According to field investigation, 168 soil samples in the collected based on a 200*200 m grid of the research watershed. Meanwhile, four years of corn yields and 15 soil indicators, including physical, chemical and biological were measured. Moreover, ^{137}Cs tracing technique was used to estimate soil erosion/deposition rate.

RESULTS

1) The soil erosion and deposition rates on the watershed ranged from -7122 to 5471 t km^{-2} yr^{-1}. Soil erosion was dominated in the upstream area, erosion and deposition was coexisted in the midstream region, and soil deposition was dominated in the downstream area. At hillslope scale, severe soil erosion occurred at the midslope, and deposition occurred at the footslope with 20-30 cm depth.

2) Soil erosion significantly influenced soil characteristics, especially affected soil organic matter, total nitrogen, available phosphorus, urease, alkaline phosphatase, microbial biomass nitrogen. According to correlation analysis and principal component analysis, the thickness of mollic soil layer, mean weight diameter of soil aggregate, soil organic matter, soil total nitrogen, pH, soil sucrase and soil microbial biomass nitrogen were chose as the minimum data set indices for soil quality evaluation. The mean soil quality index (SQI) at the catchment and the hillslope scales were 0.453 and 0.471, respectively. And the spatial distribution of soil quality index at both scales of watershed and sloping were reverse with soil erosion rate.

3) The spatial distribution characteristic of crop yield at the watershed was as follows: downstream > midstream > upstream. At hillslope scale, the minimum corn yield occurred at the midslope, which was the corresponding to soil quality and while the reverse with the soil erosion rate.

4) Corn yield declined as mollic thickness decreased. When the mollic was completely lost, the corn yield reduced by 24.2%. Particularly, within 20 cm mollic thickness, reductions in the corn yield ranged from 46.2 to120.8 kg ha^{-1} when 1 cm of mollic thickness was lost. In the extreme rain years, deposition depth at the slope foot greatly influenced corn yield, which could reduce 11.0%-31.7% of corn production, depending the deposition thickness.

5) Corn yield had a highly significant positive correlation with soil quality index and negative correlation with soil erosion rate. The equations among crop yield, soil quality, and soil erosion rate were established and validated.

CONCLUSIONS

This study analyzed the impacts of soil erosion and deposition on soil quality and crop yield. The results showed whether the spatial distribution of soil quality index or and crop yield was reverse with soil erosion rate. Corn yield declined as mollic thickness decreased. When the mollic thickness was less than 20 cm, corn yield was reduced by 8.2%-24.2% with decreasing mollic thickness. Particularly, within 20 cm mollic thickness, reductions in the corn yield ranged from 46.2 to120.8 kg ha-1 when 1 cm of mollic thickness was lost. In the extreme rain years, deposition depth at the slope foot greatly influenced corn yield, which could reduce 11.0%-31.7% of corn production. Corn yield had a highly significant positive correlation with soil quality and negative correlation with soil erosion rate. The equations among crop yield, soil quality, and soil erosion rate were established and validated. It indicated that preventing soil erosion in the Chinese Mollisol region is important for feeding Chinese people.

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4.16.O

EFFECT OF TILLAGE EROSION ON SOIL DISPLACEMENT AND SOIL PRODUCTIVITY (CASE STUDY: NORTH OF IRAN)

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INTRODUCTION

In most developing countries, often steep lands are being cultivated which can lead to sever tillage/water erosion, the loss of productive top soil, reduction of water retention capacity, increasing bulk density, and soil structural degradation (Ferreras et al., 2000). Tillage erosion is the greatest when carried out up-down the slope (Boardman and Poesen, 2006). Many studies of soil redistribution have been conducted and have documented the roll of tillage erosion in soil redistribution (Lindstrom et al., 1992; Govers et al., 1994; Lobb et al., 1995; Van Oost et al., 2006). The magnitude of soil redistribution due to tillage may often exceed that of water erosion (e.g., Govers et al., 1996). In terms of shoulder slopes, it has been estimated that tillage erosion accounts for at least 70% of the total soil erosion (Lobb et al., 1995). Based on ^{137}Cs tracer studies, tillage erosion and deposition rate estimates frequently exceed 10 Mg ha^{-1} yr^{-1} (Govers et al., 1996; Van Oost et al., 2006). In terms of long-term agricultural practice, development of a variety of terraces by dissecting long slopes into several short segments, is effective at decreasing soil erosion by water, but may cause more intensive soil redistribution by tillage and the formation of tillage banks (Van Oost et al., 2006; Zhang et al., 2014). A few studies have documented the severity of tillage translocation and erosion in Iran, therefore, the objectives of this study were to; (i) quantify rates of tillage translocation and deposition, and (ii) examine the impact of tillage erosion on the soil productivity on farmers' fields on a steeped slope of the southern Guilan, Iran.

MATERIALS and METHODS

The study area was a sloping land of rainfed agriculture in north of Iran (located about 36° 53′ N, 49° 32′ E). The studied area was about 11 ha, and it was under conventional up-down moldboard ploughing for 40 years. Seven fields were selected for surveying and five for soil and crop sampling (Fig. 1). To predict downward and lateral soil translocation, surveying was performed by Theodolite in two directions to produce digital elevation map. The soil translocated in both side of each boarder was calculated by:$V= ((\Delta h/2)\times(\Delta x/2)\times W)/W$

where, V is soil translocated per unit width (m^3 m^{-1}), Δh is elevation difference between adjacent fields (m), Δx the length of field (m), and W the field width (m). It was assumed that net translocation is zero in the middle point of each field, and therefore the soil loss has occurred on the uppermost portion of the field and soil accumulation has occurred on the lower portions.

To study the effect of tillage erosion on soil quality and productivity, soil and crop sampling was performed at 18 points (Fig. 1). The soil samples were analyzed for their physicochemical properties and the yield and yield components of wheat were determined.

RESULTS and DISCUSSIONS

SOIL REDISTRIBUTION

The elevation difference between the adjacent fields ranged from 1.04 to 3.20 m in slope direction, and ranged from 1.0 to 1.3 m in lateral direction. The maximums were observed between A and C fields, and

between F and G fields, respectively. Paperdick and Miller (1977) noted that repeated downslope moldboard ploughing above and below permanent field borders in the Palouse created soil banks 3-4 m high. Tillage erosion is a function of landscape erodibility (Boardman and Poesen, 2006), including slope direction, steepness and curvature, and as well field size and shape (Van Oost et al, 2006). The higher slope steepness at fields A, B and D resulted in higher soil redistribution and higher tillage banks.
The results of soil loss and accumulation calculations are presented in Tables 1 and 2 for both directions. The highest soil loss and accumulation were observed at Field C in both directions. The minimums were for Field D. Filed C and D have the biggest and smallest areas, respectively. There was an agreement between soil translocation rate in slope direction and slope steepness. But the lateral translocation was lower in steeper slopes (fields G and F) in compare to gentle slopes (fields A and B). The results in general agreements with those of Nyssen et al. (2000) and Zhang et al. (2009). Govers et al. (1994) noted that 4000 ton soil is redistributed during each tillage practice by moldboard plough.

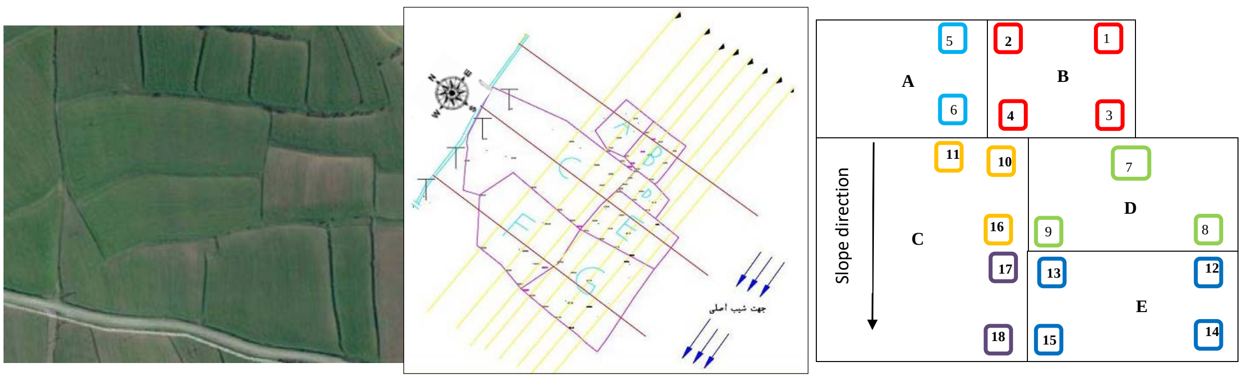

Figure 1. Study area and surveying and sampling design

Table 1. Soil translocation along the slope above and below the field boundaries

Profile No.	Field	Soil accumulation ($m^3\ m^{-1}$)	Soil loss ($m^3\ m^{-1}$)	Soil accumulation ($m^3\ m^{-2}$)	Soil loss ($m^3\ m^{-2}$)	Soil accumulation (kg ha^{-1})	Soil loss (kg ha^{-1})
1	E	34	-	0.50	-	486200	-
	G	-	80	-	0.50	-	1144000
2	E	46	-	0.67	-	657800	
	G	-	96	-	0.67	-	1372800
3	D	25	-	0.67	-	359260	
	E	51	49	0.70	0.67	729300	700700
	G	-	98	-	0.70	-	1401400
4	B	47	-	0.62	-	667400	-
	D	22	25	0.55	0.62	312400	355000
	E	28	43	0.36	0.55	400400	614900
	G	-	48	-	0.36	-	693550
5	B	51	-	0.70	-	728460	-
	D	22	27	0.56	0.70	319500	390500
	E	30	49	o.36	0.56	430430	700700
	G	-	47	-	0.36	-	672100
6	B	55	-	0.80	-	785260	-
	C	65	102	0.50	0.80	923000	1448400
	F	-	64	-	0.50	-	915200
7	A	56	-	0.76	-	802300	-

	C	32	93	0.26	0.76	455820	1324860
	F	-	31	-	0.26	-	450450
8	C	29	-	0.26	-	416060	-
	F	-	31	-	0.26	-	444730

Table 2. Lateral soil translocation before and after the field boundaries

Profile No.	Field	Soil accumulation (m^3 m^{-1})	Soil loss (m^3 m^{-1})	Soil accumulation (m^3 m^{-2})	Soil loss (m^3 m^{-2})	Soil accumulation (kg ha^{-1})	Soil loss (kg ha^{-1})
1	B	15.3	-	0.25	-	217260	-
	A	-	13.4	-	0.25	-	190280
2	E	32.4	-	0.27	-	463320	
	C	-	63	-	0.27	-	894600
3	G	45	-	0.32	-	463500	
	F	-	40.4	-	0.32	-	577720

Soil fertility properties including organic carbon content, available potassium and phosphorous, and total nitrogen of the sampling points (Fig. 1) are given in table 3. Table 3 also presents yield components of wheat measured at 1 square meter. Both soil properties and crop yield show very high spatial variability, which are in part correlated to each other and as well to field position in terms of soil loss or accumulation. Soil organic carbon, available potassium and phosphorous were lower than the critical values needed for appropriate plant growth in most sampling points.

Cluster analysis grouped the sampling point into 3 groups based on the soil properties. The group with the highest fertility includes points 4, 12, 17 and 18. The second group includes points 2, 5, 6, 10, 11 and 14. The rest was grouped as lowest fertility. In the same way, the sampling points were grouped based on wheat yield. The points 2, 12, 14 and 16 were the first group with high wheat yield. The second group includes points 1, 4, 7, 9, 15, 17 and 18. The combined effects of landscape and field positions (Nyssen et al., 2000; Poesen et al., 2000; Van Oost et al., 2006) have caused these complications. The points of higher fertility were not necessarily those of higher productivity. Cultivation management of the farmers seems to be another important factor affecting the situation. Each field is managed separately and in most cases differently. The most important differences are plant density, using manure, fertilizer rate, planting time, time and rate of pesticides/herbicides etc.

Table 3. The soil properties and yield component of wheat

Point	Organic carbon (%)	Potassium (mg kg^{-1})	Phosphorous (mg kg^{-1})	Nitrogen (%)	Spike No. (m^{-2})	Grain per spike	Grain Weight (g)	Grain yield (kg ha^{-1})
1	0.50	174	6.2	0.045	224	7.47	36.54	924
2	0.75	154	14.1	0.048	377	1.51	39.87	1791
3	0.62	91	8.2	0.051	255	2.14	43.54	357
4	1.13	116	12.4	0.095	365	5.28	46.82	897
5	0.82	150	10.5	0.068	272	3.66	31.20	1061
6	0.76	224	10.3	0.064	344	1.43	33.68	261
7	0.57	86	6.1	0.047	760	2.08	48.70	1021
8	0.64	172	10.1	0.054	258	1.19	51.98	445
9	0.66	89	10.1	0.055	596	0.97	56.85	794

10	0.89	101	10.5	0.075	108	6.48	39.84	131
11	0.76	94	14.5	0.064	210	9.33	38.54	299
12	1.10	269	16.4	0.092	310	9.78	47.80	1856
13	0.68	157	8.2	0.057	135	4.00	58.90	652
14	0.89	126	8.5	0.074	326	7.02	60.02	1841
15	0.76	106	6.1	0.064	157	22.31	58.90	1174
16	0.65	100	8.1	0.054	234	10.37	61.36	2022
17	1.40	136	14.5	0.116	164	15.72	55.89	1043
18	1.36	133	14.2	0.113	203	14.15	57.50	1439

CONCLUSION

Long term soil redistribution by tillage was predicted by surveying method on a sloping land of rainfed agriculture in north of Iran. The field borders had resulted in formation of tillage banks of 1.04 to 3.20 m high in slope direction, and of 1.0 to 1.3 m high in lateral direction during 40 years. Soil loss and accumulation were estimated to range 8.7-36 and 7.5-23 ton ha^{-1} $year^{-1}$, respectively. Tillage translocation was affected by slope gradient and field size, indicating that slope gradient and field size are the dominant factors influencing tillage translocation. While the downward translocation was higher in higher slopes, the lateral translocation was higher in gentler slopes. Soil and crop samples were collected from different parts of the fields and landscape, to study the effect of tillage erosion on soil quality and productivity. Cluster analysis of the sampling points based on soil properties and wheat yield, both grouped the points into three groups. But there was not necessarily agreement between these two grouping. The complex effects of landscape and field positions, in combination with different cultivation practices performed separately by each farmer were recognized as the affecting factors.

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4.1.P

EVALUATION OF VEGETATION INDICES (NDVI AND EVI) FROM MODIS FOR MONITORING AREAS SUSCEPTIBLE TO DESERTIFICATION IN BRAZIL

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INTRODUCTION

Drylands are areas with aridity index (AI) less than 0.65 and cover 41% of the planet's surface and are inhabited by 35% of the global population. These areas have a population of more than 2 billion inhabitants, with 90% of that population living in developing countries. Some of these regions are under food insecurity and have a poor quality of life, being prone to accelerated desertification (D'odorico et al., 2013). Around the world dryland areas include 15% of Latin America, 66% of Africa, 40% of Asia and 24% of Europe. There is a significantly greater proportion of drylands in developing countries (72%). Nevertheless, the United States, Australia and several countries in Southern Europe also contain significant dryland areas (Niemeijer et al., 2005).

In Latin America more than 5,160,000 km^2 are affected by desertification. As a result of this process, 24 billion tons per year of arable lands are lost which affects significantly farming and economic development (Fao, 1998). In Brazil the areas susceptible to desertification correspond to semi-arid area, dry sub-humid area and their surrounding areas. These regions occupy about 1,340,000 km^2 and affect approximately 30 million people. At about, 14% of these areas, located in the northeast and southeast, are in a strong desertification process (Santana, 2007).

The Moderate Resolution Imaging Spectroradiometer (MODIS) sensor is the main instrument aboard the TERRA and AQUA satellite. It was designed to provide global information of the Earth's surface, ocean and atmosphere, covering Earth every 2 days. It has 36 spectral bands, a polar orbit of 705 km, spectral range of 0.4 to 14.4 µm and a spatial resolution of 250, 500 and 1000 meters.

Bands from 1 to 7 are used to monitoring land surface, bands from 8 to 16 the oceans and the bands 17 and 19 the atmosphere. Finally, the bands from 20 to 36 are thermal bands of the spectrum (3660 nm at 14385 nm). The MODIS sensor provides products useful for studies in climatology, hydrology, oceanography, atmospheric sciences, biology, agronomy, among others (Justice et al., 2002; Rudorff et al., 2007). The MOD13 product provides two vegetation index NDVI (Normalized Difference Vegetation Index) and EVI (Enhanced Vegetation Index), with global coverage images, temporal resolution of 8, 16 and 30 days and spatial resolution of 250 m, 500 m, 1 km and 25 km.

The objective of this study was to evaluate the correlation between MODIS sensor data (NDVI and EVI) and rainfall data in order to monitoring the susceptibility to desertification of large areas.

METHOD

This research was carried out from 2000 to 2014 with rainfall data from a rain gage network with 120 gages located in the study area and its surroundings. The NDVI and EVI vegetation index data were obtained from MODIS images (MOD13Q1 product) downloaded from the Land Processes Distributed

Active Archive Center from NASA and The United States Geological Survey. The images used had temporal resolution of 16 days and spatial resolution of 250 m.

The correlation between rainfall data and NDVI and EVI data in January (rainy season) and August (dry season) was performed. For NDVI and EVI data in January was used accumulated rainfall from the previous five months (September, October, November, December and January). For NDVI and EVI data in August were used the previous five months (April, May, June, July and August).

RESULTS

It was observed that vegetation indexes and rainfall have a direct proportion which was already expected since NDVI and EVI indicate the presence of vegetation. Bigger the rain is, bigger the NDVI and EVI are (Figure 1 and 2).

Correlations between NDVI and EVI data were strongly dependent upon rainfall data. The coefficients of determination R^2 and Pearson R were really close to each other, although they were closer to NDVI data than to EVI (0.91 and 0.85 respectively). Therefore, it is possible to infer that NDVI is more influenced by rainfall than EVI is. This fact may be understood by the dependence of EVI on other factors, such as soil conditions, water availability in the soil, climatic interferences or luminosity, requiring more specific studies to detect these relationships.

Figura 1: Seasonal analysis of NDVI and Rain data registered in Barra do Cuieté meteorological stations.

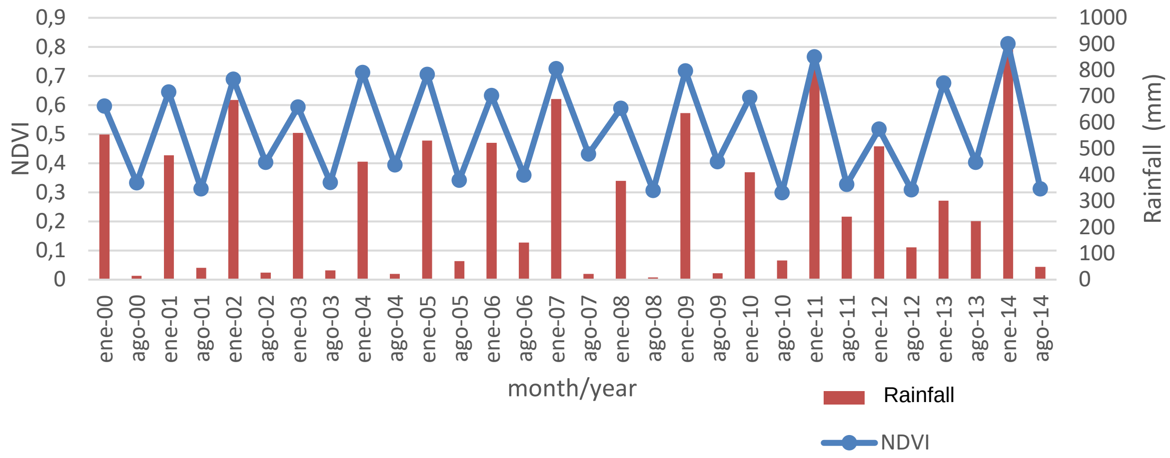
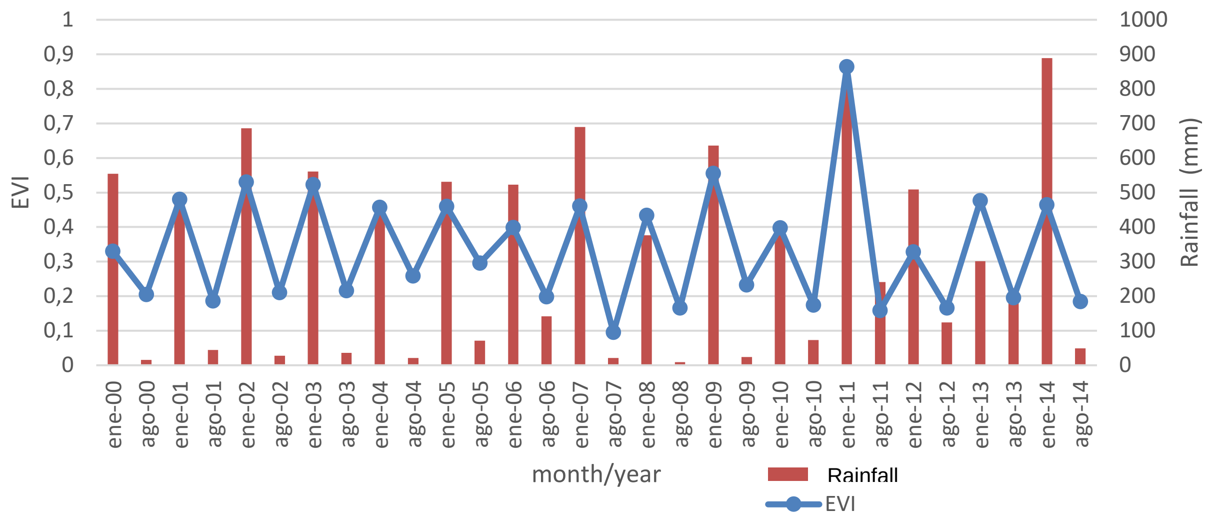


Figura 2: Seasonal analysis of EVI and Rain data registered in Barra do Cuieté meteorological stations.



CONCLUSIONS

The use of NDVI and EVI data from MODIS can be a good tool for monitoring vegetation in large scale in Brazil. The variability of NDVI and EVI is associated with a dense or less dense vegetation which has a direct relationship with rainfall, because of this, it is useful for monitoring susceptible areas to desertification.

The data acquired from the MODIS sensor are free of charge, have a high temporal resolution and cover the whole earth. Thus, it can be used in areas and projects with few financial resources.

ACKNOWLEDGEMENTS

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4.3.P

ANALYSIS OF THE ERODIBILITY INDEX OF THE SOILS OF THE BARRAGE BASIN SANTA BÁRBARA, PELOTAS RS

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INTRODUCTION

Sharp soil erosion is one of the main environmental problems caused by agriculture. In tropical regions, water erosion is relevant because it contributes to soil degradation and causes the loss of its most fertile layer by carrying particles through the slopes to watercourses, resulting in siltation (Macedo et al., 2010).

Studies on erosion are intended to measure the influence of several factors determinant to this process, allowing the estimation of soil losses and promoting practices that reduce such losses to a minimum. However, the determination of soil losses by erosion through direct methods is costly and time consuming. These are the main causes of the growing interest of researchers on erosion prediction methods (Foster et al., 1985). Among the methods used to predict such losses under certain management conditions and assess the effectiveness of conservation practices, there is the Modified Universal Soil Loss Equation (MUSLE), which is a tool that can be used for the analysis of environmental degradation, especially aiming the estimation of soil loss, predicting the amount of sediments in small and medium-sized river basins in a given outlet.

MUSLE is a variation of the USLE model. The main difference between the two models is that USLE estimates erosion for a determined period (annual), while MUSLE calculates the amount of sediments generated by isolated rainfall events (Alcântara and Souza, 2010). MUSLE is composed of the following factors: Y - amount of sediments in a given outlet of a basin after a rainfall event (Mg); Q - total runoff volume (m^3); qp - peak flow of the resulting hydrogram (m3 s^{-1}); K - soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}); C - crop use and management (dimensionless); LS - topography (slope and declivity length) (dimensionless); P - conservation practices (dimensionless) (Williams, 1975).

In MUSLE, erodibility is the factor responsible for soil susceptibility to erosion. It is conditioned mainly by mineralogical, chemical, morphological, physical and biological attributes of the soil. Erodibility is the effect of the processes that regulate infiltration of water into the soil, breakdown by raindrop impacts, and transport resistance by surface flow, which are responsible for the behavior of the soil in relation to erosion. The value of erodibility is variable because of the wide variety of soils with different attributes, making it risky to estimate a value based solely on soil classification (Martins et al., 2011).

Based on the above, this study aimed to indirectly determine erodibility (K factor) for the main representative soil classes of the Santa Barbara Dam Basin in the city of Pelotas (Rio Grande do Sul state, RS).

MATERIAL AND METHODS

The study area was the Santa Barbara Dam Basin located between the geodetic coordinates 31°37'52" - 31°47'16" S and 52°20'20" - 52°27'20" W at the southwestern portion of the city of Pelotas. According to Cunha and Silveira (1996), the soils in the Santa Bárbara dam area are classified as Red-Yellow Argisol and Gray-Brown Argisol (PVd1), Planosol and Gray-Brown Argisol (PLe1), Planosol and Gleysol (PLe3), Planosol and Gleysol (PLe4), and Gleysol, Organosol and Neosol (HGe1).

Soils classified as PVd1 present as source material anatectic granites and homogeneous magmatites, with an undulating topography, 3-15% of slope and deep or very deep drainage. It has a medium texture (sandy loam; in deeper parts, sandy-clay-loam) and a weak structure (small to medium sub-angular and angular blocks). PLe1 soils have clayey gravel, arkosic sand and silt as sources, with a gently rolling topography (0-3%) and a standard drainage. They have a medium texture (sandy loam) and a weak structure (massive). PLe3 soils have a slightly clayey and sandy gravel as source material, with a medium relief formed by ancient depressions of drains clogged with silty sediments considered deep, with an imperfect drainage. They have a medium texture (sandy loam, sometimes sandy) and a poor structure (massive). PLe4 soils, whose source materials are arkosic clays and silts, presenting a plain macro-relief but a sharp medium relief, have an effective drainage due to the fossil depressions of the High Plains. They have a relatively sandy texture and great amounts of gravel. Finally, HGe1 soils have as source materials alluvial clay sediments and recent peats. They are flat and have a mesorelief and a micro-relief poorly differentiated, very deep and poorly drained. They have good structures (disaggregated sub-angular blocks) and a good consistency (firm).

The area of São Gonçalo Channel, as well as the municipality of Pelotas, is characterized by a humid subtropical climate, with cold winters and hot summers. Its average annual temperature is 17.6°C and the average annual rainfall is 1,249 mm (Simon et al., 2003). The average monthly rainfalls show that it is not possible to clearly define the months of the year corresponding to the dry and rainy seasons, ranging between and 153.36 mm for November and September, respectively (Teixeira et al., 2013).

Soil erodibility was calculated from the equation proposed by Wischmeier and Smith (1978), considering the results of geotechnical parameters (Cunha and Silveira, 1996), according to the to the following formulation:

$$K = \frac{0.137}{100}\left[2.1\times10^{-4}.(12\text{-}OM).((Sil+Fs).(100\text{-}Cla))^{1.14} + 3.25.(S_1 - 2) + 2.5(P_1 - 3)\right] \quad (1)$$

where:
K - soil erodibility (Mg h MJ-1 mm-1);
OM - organic matter (%);
Sil+Fs - silt + fine sand content (%);
Cla - clay content (%);
S1 - parameter describing soil structure (Tab. 1);
P1 - parameter describing soil permeability (Tab. 1).

The values for organic matter, silt, sand and clay contents were obtained according to Cunha and Silveira (1996). The equation to calculate K, according to Wischmeier and Smith (1978), has no limitations in its application.

Table 1. Classification of the parameter describing the soil structure (S1)

Classification	Structure
1	Very fine granular
2	Fine granular
3	Average or large granular
4	Block or massive

Source: Wischmeier and Smith (1978)

Table 2. Classification of soil permeability (P1)

Texture	Permeability (cm s^{-1})	Classification
Silty clay, clay	$< 2.8 \times 10^{-5}$	6 – very low
Silty clay, sandy clay	2.8×10^{-5} a 5.6×10^{-5}	5 – low
Sandy clay	5.6×10^{-5} a 1.4×10^{-4}	4 – low to moderate
Silt	1.4×10^{-4} a 5.6×10^{-4}	3 – moderate
Clayey sand	5.6×10^{-4} a 1.7×10^{-3}	2 – high
Sand	$> 1.7 \times 10^{-3}$	1 – very high

Source: Hann et al. (1994)

RESULTS AND DISCUSSION

Soil Erodibility (K) values calculated for basin and soil class are shown in Table 3. The values were 0.0145 Mg h MJ^{-1} mm^{-1} for HGe1, 0.086Mg h MJ^{-1} mm^{-1} for PLe1, 0.047 Mg h MJ^{-1} mm^{-1} for PLe3, 0.0397 Mg h MJ^{-1} mm^{-1} for PLe4 and 0.260 Mg h MJ^{-1} mm^{-1} for PVd1. Fernandes (2011) obtained values between 0.20 and 0.30 Mg h MJ^{-1} mm^{-1}, which were classified as medium erodibility, and values above 0.30 Mg h MJ^{-1} mm^{-1}, classified as high erodibility.

Thus, it is possible to observe that the PVd1 soil has a medium susceptibility. The other soils showed a low susceptibility. This may be because Red-Yellow Argisol and Grey-Brown Argisol (PVd1) are prone to erosion. Miqueloni and Bueno (2011), estimating the erodibility of a soil in a basin headwaters area of the Tijuco Stream, Monte Alto/SP, where the predominant soils were classified as Red-Yellow Argisol classes, concluded that the average erodibility estimated for this type of soil was considered high, correlated with soil particle size and varying in specific areas according to the relief.

Table 3. Values for Soil Erodibility (K) of Santa Bárbara Dam Basin in relation to soil classes.

Soil	**K (Mg h MJ^{-1} mm^{-1}) (Basin)**
HGe1	0.0145
PLe1	0.0860
PLe3	0.0470
PLe4	0.0397
PVd1	0.2600

Souza and Lima (2013), upon assessing the laminar erosion potential of the Ribeira Sozinha Basin (GO), classified the degree of soil erodibility in that basin as High, Medium, Low and Null according to its textures. The classifications were High: Cambisols and Litholic Neosols, sandy/medium texture; Medium: Red Argisol, Red-Yellow Argisol with a medium/clayey texture and Red Latosol with a sandy/medium texture; Low: Red Latosol with a medium and clayey texture and Petric Plinthosols resistant to erosion; and Null: highly clayey Gleysols. This was also observed for the soils of the Santa Barbara Basin for PLe3 and PLe4 soils, which have Gleysol in their composition, i.e., lower erodibility values, the PLe1 soil, composed of Argisol, has a higher erodibility value than those soils, they can be classified (PLe3 and Ple4) as null and low erodibility according to Souza and Lima (2013), respectively.

CONCLUSIONS

It can be concluded that, through the values found, it is possible to classify PLe3 and PLe4 soils as null erodibility, PLe1 as low erodibility and PVd1 as medium erodibility. Through the results obtained for the calculation of erodibility, it was found that the same calculation can be used to predict loss by soil erosion.

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4.4.P

SOIL WATER MODELLING IN A DRYLAND AGRICULTURAL SYSTEM.

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ABSTRACT

In Spain, dryland agriculture covers 84% of the area devoted to grain cereals. Slurry applications are used widely as fertilizers in order to reduce costs. Slurries have a high nitrogen content and low C:N ratio. Consequently, there is a risk of groundwater contamination by nitrates, through the leaching process in the soil profile. Soil water content (SWC) can be simulated with mathematical models that allow us to predict drainage and leaching losses. They also help to set up decisions regarding agricultural practices and the reduction of environmental impacts. The aim of this work was to model the SWC and its dynamics in a winter cereal crop rotation (barley, wheat) in a dryland Mediterranean agricultural system. One layer, from 0–90 cm, was used in the modelling process. Daily temperature and precipitation were collected from an experimental plot in Oliola municipality (NE Spain). Model simulation was applied for a three year period (2012–2015) and was validated with field data obtained from the 2012/13 cereal cropping season. A small quantity of drainage water was obtained, equivalent to 4% of mean annual precipitation (MAP). Nitrate leaching only occurred on those days in which annual precipitation was higher than the MAP (443mm). The one layer model was useful for SWC quantification in dryland agricultural systems. Nevertheless, a more detailed approach involving different soil layers is recommended to accurately represent SWC dynamics and to quantify nitrate leaching.

INTRODUCTION

Intensive livestock farming has environmental impacts on landscape, air quality, climate change and subsuperficial water quality. Agro-hydrologic modelling, because of its physical base, is a key element of environmental impact prediction. In dryland agricultural systems from Mediterranean areas, one of the strategies to mitigate such environmental impacts is to reuse animal slurries as fertilizers. Slurry application is done on winter cereals at sowing and at tillering development stage. Slurries are composed of more than 90% of a liquid fraction (Yagüe *et al.,* 2012). In consequence, monitoring their movement throughout the soil profile together with water displacement is important to reduce environmental risks in groundwater, such as nitrogen leaching.

Different reference models on soil water monitoring exist (Groot, 1987; Eckersten and Jansson, 1991; Porter, 1993). Some models use dozens of parameters to describe SWC dynamics in soil, which include many physiological processes. Simpler compartmental models can help to monitor soil water dynamics as drainage or SWC. They treat the soil as different layers and make the calculations using soil properties, crop characteristics and weather information.

The aim of this work was to model the drained water and its dynamics in the soil in a winter cereal crop rotation in a dryland Mediterranean agricultural system as a first step to model N leaching.

MATERIALS AND METHODS

Study area.

A long term experimental field located in Oliola, Lleida, NE Spain. Coordinates are 41º 52" 34'N, 0º 19" 17' E with altitude of 440 m a.s.l. was studied. It was set up in a 3-year rotation of barley (*Hordeum vulgare*) and one of wheat (*Triticum aestivum*), with 18 strategies of nitrogen fertilization, including combinations of mineral nitrogen fertilizers, pig slurry and control (no nitrogen applied). The region has a semiarid Mediterranean climate with low annual precipitation (443mm), a mean annual temperature of 12.6°C with high temperatures in summer. The soil is deep (>1 m), it has a silty loam texture, the organic matter content is below 2%. The soil is non-saline and calcareous with pH of 8.2 (1:2.5 soil: distilled water). It is classified as a Typic Xerofluvent.

Data acquisition.

Daily precipitation, air temperature and evapotranspiration (ETo) were recorded from an automatized meteorological station next to the field. For modelling, data from three year cropping seasons (2012-2015) were collected. Core samples to measure the real values of water content were collected in the 2012/13 growing season.

Compartmental model.

The model was developed in MatLab. The maximum ETo, stored soil water and drainage were calculated. The program took the information from an Excel file. It required initial edaphic conditions, daily precipitation, reference crop ETo, the percentage of the soil surface covered and the crop coefficients. Crop data introduced in the model were adapted from a work under similar conditions (Villar, 1989). One layer of 90 cm deep (0-90cm from the top soil surface) was used. As it is a dryland environment, precipitation is the unique water source. Calculations were done daily for the cropping seasons.

The results appeared in an output Excel file. The program plots the drainage versus time and the water layer versus time. As a first approach, results were evaluated with graphical representations from observed and simulated data.

RESULTS

Climatic conditions for the evaluated years are shown in Fig. 1 and Fig. 2. ETo is higher than the precipitation for the main part of the year. It can be four times the rainfall figure in the hottest month of the year. The variability of precipitation among the studied years was important and directly affected the drainage results. The 2012/13 cropping season was the wettest year with 30% more rainwater received than the 2014/15 season. The latter one was the hottest year.

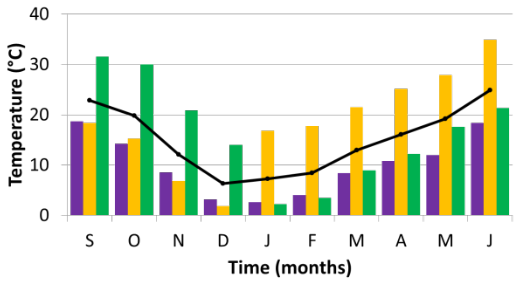

Figure 1. Mean monthly temperature (°C) (black line) and precipitation (vertical bars), for each cropping season evaluated: purple, 2012/13; yellow, 2013/14; green, 2014/15.

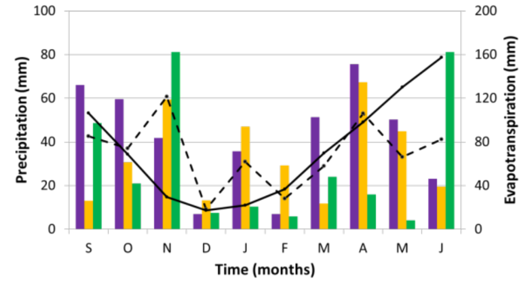

Figure 2. Mean monthly precipitation (bars and dotted line), evapotranspiration (continuous line) and total precipitation (vertical bars) for each cropping season: purple, 2012/13; yellow, 2013/14; green, 2014/15.

Soil water content

Soil water dynamics of the soil reservoir in the Oliola study area are shown in Fig. 3. The water reservoir graph can be divided in two big groups based on time of year. From September to February, all years had the same tendency. However, from March to June, the water reservoir was different for each year studied.

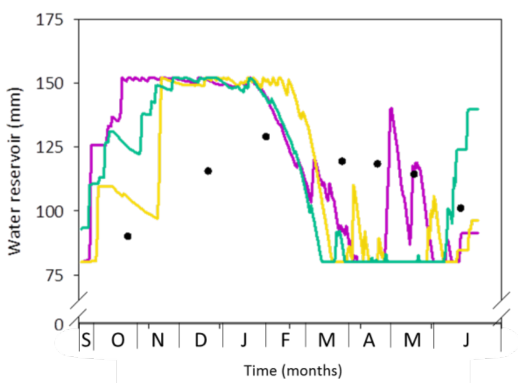

Figure 3. Water reservoir (mm) in a soil layer (0-90cm) for three cropping years. Color lines mean; purple, 2012/13; yellow, 2013/14; green, 2014/15; black points, field data (2012/13).

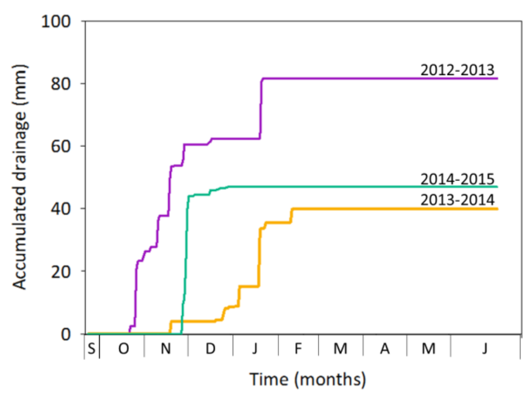

Figure 4. Monthly accumulated drainage (mm) for three years of the experimental model period.

Accumulated values for water drainage are shown in Fig. 4. Results were associated with rain events. The highest drainage was found in the months where precipitation was higher than ETo (November and January). Because of the low precipitation during the cropping year 2014/15, drainage only occurred during November and December.

The maximum quantity of soil water was achieved during November, December and January. Water content decreased the fastest from February to March, probably because of the growing period of the barley crop. Plant growth was encouraged by fertilization with slurries and the increase in temperature. From March to June, there was no drainage in spite of the rain events (Fig. 2) and the changes observed in the water reservoir (Fig. 3). Precipitation was the unique source of water in the dryland areas, thus the water stored in the soil profile was used for the crop necessities.

A common fertilization practice in Oliola fields is the application of slurries before sowing, at the end of September, and as a top-dressing at the beginning of February. According to these results, the application of slurries as top-dressing could enhance nitrogen leaching from the soil surface in rainy periods.

CONCLUSIONS

In rainy years, water losses calculated with the compartmental model were up to 4% of the annual precipitation.

Compartmental modelling was a practical method to simulate water movement in the soil. Nevertheless, a more detailed approach is suggested to improve the results and contribute to decreasing the environmental hazards of N losses.

ACKNOWLEDGMENTS

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4.6.P

RELATIONSHIPS BETWEEN SLOPE EROSION PROCESSES AND AGGREGATE STABILITY OF ULTISOLS FROM SUBTROPICAL CHINA DURING RAINSTORMS

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1 INTRODUCTION

Process-based erosion models applied to agricultural soils usually rely on the rill-interrill concept, which requires different relationships and algorithms for each component. Soil erodibility, K_i for interrill erosion and K_r for rill erosion, is regarded as a key parameter for evaluating a soil's susceptibility to erosion and is essential for predicting soil loss and evaluating the environmental effects thereof. Both K_i and K_r can be determined either by data from experimental plots or from soil properties, including soil texture, cohesion strength, soil shear strength, and aggregate stability. Among the aforementioned properties, aggregate stability is a key soil structural trait that describes the resistance of aggregates to the disintegrating action of water (Valmis et al., 2005). Previous studies have established formulas to describe the relationship between interrill erosion and topsoil aggregate stability, which is reflected through different indicators such as percolation stability (*PS*) (Mbagwu and Auerswald, 1999) and aggregate instability (β) (Valmis et al.,2005). The stability index, A_s, is probably a better aggregate indicator for slaking by fast wetting and mechanical breakdown due to raindrop impact effects (Yan et al., 2008). However, interrill and rill erosion are continual and simultaneous processes on slopes with rill present. For the purposes of accurately describing the erosion process and calculating the relative contributions of interrill and rill erosion, it is necessary to determine the relationships between K_i, K_r and A_s simultaneously during rainstorms.

Conventional erosion monitoring techniques such as field plots and stereo-photo-surveys, cannot readily distinguish between interrill- and rill-eroded sediments. Numerous methods have been applied in recent decades for distinguishing interrill- and rill- eroded sediments in experimental plots using tracers, e.g., beryllium-7, glass particles, artificial radionuclides (^{134}Cs & ^{60}Co). REEs are found at low background concentrations in soils, chemically stable, environmentally safe and undergo low plant uptake. REEs therefore make ideal soil tracers because they also strongly adsorb to soil particles without interfering in their movement and can be analysed readily and accurately (Liu et al., 2016a,b). Since interrill erosion generally occurs from the upper 10 mm of soil, REEs applied at different depths have the potential to distinguish interrill and rill erosion during rainstorms K_i and K_r can then be estimated simultaneously.

Against this background information, the purposes of this study were (i) to partition interrill and rill erosion for Ultisols using REEs as tracers during simulated rainstorms; (ii) to develop new equations for predicting both interrill and rill erosion rates that incorporate relevant soil aggregate stability indices to replace the erodibility factors; and (iii) to validate the newly developed equations.

2 MATERIALS AND METHODS

2.1 Experimental design

Soils developed over Quaternary red clay or shale, classified as Ultisols, were collected from Xianning

County (29°39′-30°02′ N and 114°06′-114°43′ E) and Yidu City (30°05′-30°35′ and 110°05′-111°35′E). Samples collected from the uppermost 30-cm layer comprise 58.3% clay, 32.5% silt and 9.2% sand for the soil developed over Quaternary red clay (clay soil) and 21.5% clay, 38.3% silt and 40.2% sand for the soil developed over shale (loam soil). The bulk density and organic matter content for soil developed over Quaternary red clay or shale were 1.16 g cm^{-3} and 1.5% or 1.25 g cm^{-3} and 1.7%, respectively. All soil samples used in the experiment were air-dried and screened through a 2-mm sieve. Six REE oxides in powder form (Yb_2O_3, Tb_4O_7, Sm_2O_3, CeO_2, La_2O_3, Dy_2O_3) were chosen for this study based on their price, quantity to be applied, and susceptibility to detection.

Each REE oxide was initially mixed thoroughly with 1 kg of air-dried soil and then mixed with additional air-dried soil approximately five times until it reached the target application concentration. Soil samples, each containing a corresponding REE, were prepared for packing into metal plots with dimensions measuring 2.25 m in length, 0.5 m in width, and 0.2 m in depth. The plot was subdivided into three equal parts of length 75 cm each. The plots were set at 10°, 20° and 30°. Packing was carried out layer by layer to achieve the desired uniform mean bulk density (1.16 g cm^{-3} for the soil developed over Quaternary red clay and 1.25 g cm^{-3} for the soil developed over shale). Erosion within the depths of 0.0-1.0 cm and 1.0-20.0 cm was deemed to be interrill and rill erosion, respectively. Thus, six areas with different REEs were existing. The bottoms of the plots were perforated and covered with a 20 cm layer of sand to facilitate even drainage of percolating soil water. Runoff was funnelled to a collection vessel placed at the lower end of the plot. After packing, the surface soil was watered to saturation, covered with a rain shelter and maintained for three days without any disturbance to enhance the adsorption of the REEs to soil particles.

Three rainfall simulations, with intensities of 60, 90 and 120 mm h^{-1}, were conducted for 30 min following the initiation of runoff. These intensities were based on the natural maximum rainfall intensity occurring for 30-min duration with return periods of 2, 5 and 20 years near Xianning County and Yidu City. In total, there were 18 treatments, comprising three slope gradients (10°, 20° and 30°), three rainfall intensities (60, 90 and 120 mm h^{-1}), and two soil types (soil developed over Quaternary red clay or shale). Each treatment was performed in triplicate; therefore, a total of 54 rainfall simulations were carried out in this study. Runoff and sediments were collected in a series of plastic containers at 3 min intervals over the 30-minute period. The volume of water in each container was measured, and the sediment was air dried and weighed.

2.2 Laboratory analysis

A modified standard methodology for extracting metals from environmental samples was used to extract the REE from the various soil-REE mixtures and sediment samples in a sequence that combined ten steps. Inductively Coupled Plasma Mass Spectrometry (X Series 2 ICP-MS, Thermo Fisher Scientific, US) analysis of the extracts containing REEs was carried out at China Three Gorges University. A stock internal standard solution containing Rh and Re (10 $\mu g \cdot L^{-1}$) was added in the analysis process for analytical quality control. Three separate measurements were taken for each extract and the mean value of the measurements was calculated. The LB-method was used to measure aggregate stability for the investigation of the following breakdown mechanisms: fast wetting (FW); slow wetting (SW); and mechanical breakdown by stirring pre-wetted aggregates (WS).

2.3 Calculation

Aggregate stability for each sample was expressed in terms of the mean weight diameter for the different size classes calculated using Equation 1:

$$MWD = \sum_{i=1}^{n} \overline{x_i} w_i \qquad (1)$$

where w_i is the weight fraction of aggregates in the size class *i* with an average diameter x_i.
The relative slaking index (*RSI*) and the relative mechanical breakdown index (*RMI*) were used to determine the resistance to slaking and the mechanical breakdown of the soils, respectively. A_s is the stability index in this research.

$$RSI = \frac{MWD_{sw} - MWD_{fw}}{MWD_{sw}} \tag{2}$$

$$RMI = \frac{MWD_{sw} - MWD_{ws}}{MWD_{sw}} \tag{3}$$

$$A_S = RSI \times RMI \tag{4}$$

where MWD_{fw}, MWD_{ws}, and MWD_{sw} are the mean weight diameter obtained by the FW, WS, and SW treatments, respectively.
As mentioned above, the stability index (A_s) may have the potential to express K_i in interrill erosion equation and K_r in rill erosion equation in the WEPP model synchronously. So we make an assumption that D_i and D_r can be calculated by the following equations:

$$D_i = aA_s S_f I^2 \tag{5}$$

$$D_r = bA_s(\tau - \tau_c) \tag{6}$$

where *a* and *b* are coefficients.

3 RESULTS

3.1 The stability index

The aggregate stability values of the soils developed over Quaternary red clay or shale calculated by Equations 1-4 are shown in Table 1. The values of MWD_{fw}, MWD_{ws} and MWD_{sw} for the soil developed over shale were lower than those for the soil developed over Quaternary red clay, indicating that the aggregate in the soil developed over Quaternary red clay was more stable than that in the soil developed over shale. In both types of soil, the values of aggregate stability could be ordered as $MWD_{fw} < MWD_{ws} < MWD_{sw}$ for the three treatments, suggesting that slaking (FW) is the most effective mechanism for aggregate breakdown, followed by mechanical breakdown (WS); chemical dispersion (SW) is the weakest breakdown mechanism. The *RSI* and *RMI* of the soil developed over shale were higher than those of the soil developed over Quaternary red clay (Table 1). Higher *RSI* and *RMI* of aggregates imply greater susceptibility to slaking and mechanical breakdown, respectively. The A_s of the soil developed over shale was greater than that of the soil developed over Quaternary red clay, indicating that the former had a greater potential for detachment during both interrill and rill erosion (Yan et al., 2008).

Table 1 Aggregate stabilities measured by the LB-method on the soils developed over Quaternary red clay or shale

Soil	MWD_{fw} (mm)	MWD_{ws} (mm)	MWD_{sw} (mm)	RSI	RMI	A_s
The soil developed over Quaternary red clay	0.72 ± 0.06	2.24 ± 0.09	2.74 ± 0.04	0.737	0.182	0.135
The soil developed over shale	0.62 ± 0.04	1.64 ± 0.13	2.63 ± 0.10	0.764	0.376	0.288

3.2 Establishment of predictive equations

The aggregate stability index A_s reflects soil resistance to both raindrop impact during interrill erosion and runoff shear stress impact during rill erosion. Based on data from thirty events selected at random, Fig. 1 presents the results of a linear regression analysis with a zero intercept for the measured and estimated interrill and rill erosion rates calculated using Equations (5) and (6) with coefficient values set to *a* and *b*=1, respectively. The relationships indicate that coefficient values *a* and *b* should be 2.59 and 6.12, respectively (Fig. 4). Therefore, the interrill and rill erosion rates for the soils developed over Quaternary red clay and shale can be individually estimated by using the following equations:

$$D_i = 2.59 A_s S_f I^2 \quad (n=30,\ R^2=0.97,\ p<0.01) \qquad (7)$$

$$D_r = 6.12 A_s(\tau - \tau_c) \quad (n=30,\ R^2=0.81,\ p<0.01) \qquad (8)$$

3.3 Validation of the equations

The remaining interrill (n=24) and rill erosion data (n=24) were used for model validation. Fig. 2 illustrates moderate to good relationships between measured and predicted (using Equations 14 and 15) values of interrill and rill erosion rates (R^2=0.95 and 0.76, respectively). The R^2 value for the validation of Equation 15 is not as high as that for Equation 14. Nevertheless, the results show that Equations 14 and 15 are effective for predicting interrill and rill erosion rates simultaneously during rainstorms.

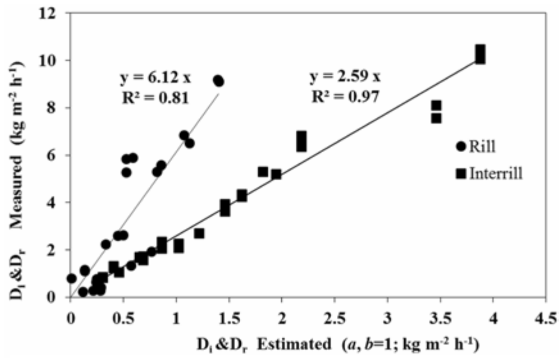

Fig. 1 Linear regressions with zero intercept between measured and estimated values of interrill and rill erosion rate.

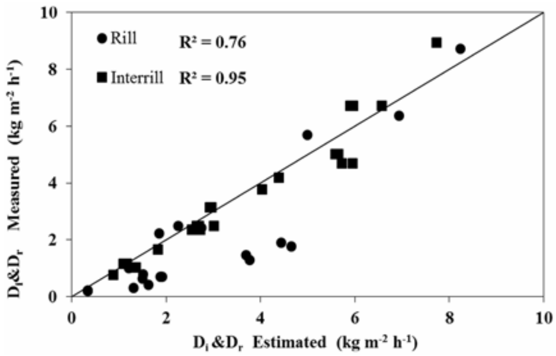

Fig. 2 Relationship between measured and estimated rate of interrill and rill erosion for validating.

4 CONCLUSIONS

In conclusion, the equations incorporating the stability index, As, reliably reflect the soil interrill and rill erosion rates. Equations using an aggregate stability index, As, were constructed to replace the erodibility factors, Ki and Kr for interrill and rill erosion, respectively, in the WEPP model; the equations were subsequently verified. Our results show that it is possible to estimate interrill and rill erodibility simultaneously on the basis of aggregate stability measurements using the REE tracing method. The equations developed herein have great potential to be applied to a wider range of soil types. As our results are based on experiments on only two soils, future studies should investigate a wider range of soil types with various aggregation characteristics under more varied rainfall conditions in order to support the results obtained herein.

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4.7.P

IMPACT OF TILLAGE PRACTICES ON EARLY AMMONIA LOSSES

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1. INTRODUCTION

The major source of ammonia (NH_3) losses to the atmosphere (85%) is linked to agricultural management. The excessive application of N fertilizers and surface application of manure N leads to approximately 60% of NH_3 being volatilized (Cameron et al., 2013).

The amount of NH_3 volatilised is a variable percentage of the applied NH_3, depending on several factors such as the fertilizer type, the manure characteristics, the application technique, the soil type and humidity, the weather conditions, the crop cover and the tillage systems (Sommer et al., 2003; Grandy et al., 2006; Huang et al., 2013). No-tillage (NT) systems are being promoted as the best tillage practice as they improve soil structure, increase soil moisture, reduce erosion, increase soil organic carbon sequestration and subsequently increase yields and economic benefits to the producer (Grandy et al., 2006; Huang et al., 2013). However, some reports have shown that NT systems increase NH_3 losses compared to minimum tillage (Mkhabela et al., 2008; Zhang et al., 2011). Large NH_3 losses reduce the N available for plant uptake, contribute to environmental degradation (i.e., acidification and eutrophication) of natural ecosystems and atmospheric air pollution (FAO, 2001).

The dry matter content (DM) content of the slurry affects the emissions in several ways. The slurry DM contributes to the sealing of soil pores, reducing infiltration and sometimes causing ponding at the soil surface (Donovan and Logan 1983). Thus slurry dry matter influences the amount of NH_3 losses (Bosch-Serra et al., 2014).

For Mediterranean areas, information on ammonia emissions is scarce. Spanish soils may be prone to NH_3 atmospheric losses due to high soil pH, hot climatic conditions and nature of soils (calcareous soils), but there is a lack of information on field strategies for mitigation of losses.

The objective of this study was to compare NH_3 volatilization losses under no-till and minimum tillage (MT) when pig slurry is applied at sowing in a calcareous soil.

2. METHODS

2.1. Site description

The experiment was carried out at Oliola (NE Spain), where the climate is dry Mediterranean: annual average temperature of 13.4°C and an annual average rainfall is 610 mm. The soil is well drained, non-saline, with a silty loam texture in the surface layer and it is classified as a Typic Xerofluvent (Soil Survey Staff, 1999).

2.2. Experimental design

A broad fertilization experiment was established in 2002 in Oliola (NE Spain). From 2009 onwards, the plots were divided by tillage practice into either minimum tillage (MT) or no-till (or direct sowing).

This study was conducted during two winter cereal growing seasons (2011-2012 and 2012-2013). Barley (*Hordeum vulgare* L.) was sown in 2011 and 2012 (indicated on the treatment name as -11 and -12) under rainfed conditions. In both years, the sowing rate was 190 kg ha^{-1}, and the distance between rows was 0.12 m.

The treatments were arranged according to a randomized block design with three replicates, although only two replicates were used for this study. The applied fertilizer at sowing was pig slurry (25 t ha^{-1}) from fattening pigs (FS). The treatments were (Table 1): a control (C, with no N, no organic matter applied) and FS at a rate that equalled 25 t ha^{-1} in 2011 and 22 t ha^{-1} in 2012

Table 1. Fertilizer N applied at sowing as slurry from fattening pigs (FS) under minimum tillage (MT) or no-till (NT) in 2011 and in 2012

Year	Treatment	Fertilizer applied at sowing		
		Total NH_4^+-N (kg ha^{-1})	Total N (kg ha^{-1})	Pig slurry applied (t ha^{-1})
2011	C-MT-11	0	0	0
	C-NT-11	0	0	0
	FS-MT-11	111	185	25
	FS-NT-11	111	185	25
2012	C-MT-12	0	0	0
	C-NT-12	0	0	0
	FS-MT-12	80	130	22
	FS-NT-12	80	130	22

The composition of the applied pig slurry is stated in Table 2.

Table 2. Main properties of the applied fattening pig slurry

	Fattening pig slurry	
Year	2011	2012
pH	8.70	8.70
Dry matter (%)	11.87	10.37
Organic matter (%, d.m.)	66.40	60.59
Total N (%, d.m.)	6.27	5.71
NH_4^+-N (%, d.m.)	3.75	3.50

d.m.: dry matter

2.3. Gas sampling and analysis

The NH_3 volatilization was measured using a system composed of a dynamic chamber method and a photoacoustic analyser. Undisturbed soil samples before slurry application were taken with PVC tubes (15 cm height * 7cm Ø). They were taken from field plots from each combination tillage/fertilization. Two field blocks were sampled. Samples were transported to the laboratory within one hour, and they were placed inside a glass jar (1.5 L). The jar was equipped with a system that allowed us to directly measure the air stream NH_3 concentration with a photoacoustic analyser (Innova 1412®). Slurry was applied upon arrival at the laboratory, over the tube surface, according to the established treatments. From it, the air stream NH_3 concentration was measured continuously for a further 12h. This time of measurement was taken because farmers must incorporate the slurry within the first 24h after application but they usually incorporate it on the day of application. Thus, 12h is the maximum time that slurries remain on the surface.

3. RESULTS

Preliminary results (Figure 1) show that in both years, the total cumulative NH_3 volatilization, or as a percentage of total ammonium nitrogen (TAN) applied, were significantly ($p<0.05$) affected by tillage practices.

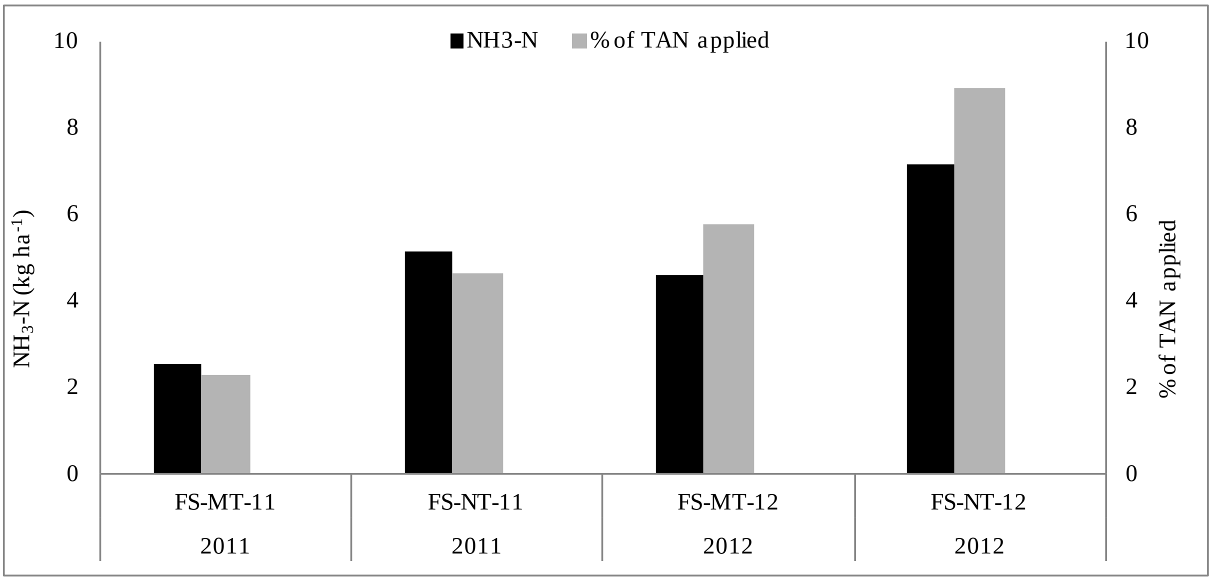

Figure 1. Total ammonia volatilization in 2011 and 2012 and as a percentage of the total ammonium nitrogen applied. Measurements were done at sowing, during 12h after pig slurry (FS) application in minimum tillage (MT) and no-till (NT) plots.

The highest losses (up to 9% of TAN) were achieved in the NT treatment, which is approximately two times higher than the value corresponding to the conventional tillage treatment.

4. DISCUSSION

The differences between NT and MT would be greater if measurements were taken over for a longer period of time, after slurry incorporation in soil. The reduction of slurry infiltration into the NT soil was related to the stubble left on the surface and to higher soil compaction in NT conditions.

5. CONCLUSIONS

Tillage practices had a significant effect on NH_3 volatilization. Ammonia volatilization was higher from NT than from MT treatments. The present study suggests that fattening pig slurry application at agronomic doses in combination with minimum tillage to promptly incorporate the slurry can be an effective strategy to reduce NH_3 losses to the atmosphere.

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4.8.P

ACHIEVEMENTS OF SOIL AND WATER CONSERVATION BASED ON NATIONAL SURVEY ON SOIL EROSION IN CHINA

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ABSTRACT

Four times survey on soil erosion have been conducted in the past 30 years. Although it was recognized that China is one of the countries suffered from very severe soil erosion over the world, it should also be noticed that a series of achievements have been obtained after long time hard treatments. Data from these surveys not only revealed the erosion situation in details such as distribution, area, intensity, but also provided the crucial information to develop measures to reduce soil loss. The *Law of Soil and Water Conservation of People's Republic of China* was promulgated in 1991 and revised in 2010. It together with the relative regulations and technical standards have played very important roles in controlling erosion caused by human activities. National key projects of soil and water conservation were put into effect across the country, numerous small watersheds located in different regions were well treated and protected, these both contributed to the ecological improvement greatly. In the background of "Soil and water conservation is one of the basic state policies in China", in the following years, greater achievements must be obtained under the support of more investments from governments and companies, more advanced technologies from research institutes, and deeper awareness from all ordinary people living in China.

KEYWORDS*:* Soil and Water Conservation, Erosion Survey, China

INTRODUCTION

The National Land and Water Resources Census plays a very important role in China. It provides reliable information on the national soil erosion status and helps government take effective and efficient treatments to control such erosion. Compared with America, which began the National Erosion Reconnaissance Survey in 1934 (Nusser and Goebel, 1997; Goebel, 1998). China develops national scale census was quite late, the mid-1980s Landsat Multi-Spectral Scanning satellite images were adopted to as the main information source in the first remote sensing survey of soil erosion. Then the second survey began in 1999, using the mid-1990s Landsat Thematic Mapping (TM) image as the main information source. Shortly, the third survey was developed in 2000-2001. As a result, these surveys clarified the key controlling areas in the main rivers of China successfully, and further contributed a lot to the enforcement of "national ecological construction planning" and "national ecological protection planning". With the development of economy in China, many constructions were started to build, this has caused serious problems related to soil erosion. It is crucial to develop a new census for making clear the causes and processes of erosion. Under this circumstance, the fourth national census was conducted. Generally, after 3-years census, the condition of soil loss caused by water and wind was ascertained, the status of soil erosion control measures was obtained, this is very helpful and useful for building a beautiful and environmental-friendly country.

METHODS

The Chinese Soil Loss Equation (CSLE, Liu et al. 2002) developed from the Universal Soil Loss Equation (USLE), was introduced to this census. Compared to the USLE, the biological, engineering and tillage factors were used in the CSLE to replace the cover and management factors. In addition, a steeper slope steepness factor equation was inclusive in the equation based on the measured data in the loess plateau (Liu et al, 1994).

RESULTS

(1) Area and Intensity of Water and Wind Erosion

The area and intensity of erosion caused by water and wind are shown in Table 1. Compared to 3,556,000 km^2, obtained in the second national remote sensing survey in 2002, after about 10 years treatment, the total soil erosion area is reduced by 606,400 km^2.

Table 1. Area and intensity of soil erosion caused by Water and Wind

Area based on the Degree of Erosion (10^4 km^2)	Soil Erosion Type		Total area of Erosion
	Water Erosion	Wind Erosion	
Slight	66.7	71.6	138.4
Moderate	35.1	21.7	56.9
High	16.9	21.8	38.7
Severe	7.6	22.0	29.7
Extreme	2.9	28.4	31.3

(2) Quantity, area, and distribution of gullies in the Losses Plateau and the black soil region of Northeast China

The characteristics, including numbers, area and length for the gullies in the Loess Plateau of Northwest China and in the black soil region of Northeast China are shown in Tables 2 and 3.

Table 2. Characteristics of the gullies in the Loess Plateau of Northwest China

Region	Number of Gullies	Area (hm^2)	Length (km)
Hilly and gully region	556,425	15671,937	470,978.8
Gully region of loess plateau	110,294	3049,520	92,299.4

Total	666,719	18,721,456	563,278.2

Table 3. Characteristics of the gullies in the Black soil region of Northeast China

Type	Number of Gullies	Area (hm^2)	Length (km)
Developing gullies	262,177	303,606	168,382.4
Stable gullies	33,486	61,236	27,130.3
Total	295,663	364,842	195,512.6

(3) Area, quantity, and distribution of soil erosion control measures

The basic information on soil erosion control measures are shown in Table 4.

Table 4. Area of soil erosion control measures

Type	Engineering measures	Biological measures	Other measures	Total
Area (km^2)	200,300	778,500	12,800	991,600

CONCLUSION

The census results demonstrate that the areas suffering erosion were decreased either caused by water or by wind. However, attention should be paid that there are still some regions suffering from severe erosion, including the upper and middle reaches of the Yangzi River - even the vegetation coverage increased more than 30% during the past decade, the upper and middle reaches of the Yellow River, the soil and stone mountainous area in Southwest China, and the black earth area in Northeast China. In addition, erosion in some mining areas are still quite severe, this man-made erosion type has attracted more attentions both from the governments and also from the enterprises.

Generally, those national plans enforced after the first 3 surveys and a sires of regulations including the revised *Law of Soil and Water Conservation of People's Republic of China* have played important role to alleviate erosion in a great extent. But there is still a need to continue implementing such engineering or biological measures, especially in these regions suffering from serious erosion mentioned above.

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4.10.P

PHYSICAL QUALITY OF A YELLOW ARGISSOL USING THE "S" PARAMETER

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INTRODUCTION

The quality of a soil can be considered in three aspects: the physical, the chemical and the biological, being the physical quality important to evaluate the degree of degradation of the soil and, once evaluated, it is possible to select the appropriate practices for the recovery of the soil. Themselves. A soil is considered to be physically degraded when it presents low water infiltration capacity, high surface runoff, low aeration, reduced root system, difficult mechanization, ie, negative changes in soil structure. According to Richart et al. (2005), the degradation of the soil structure can compromise the development of plants and, consequently, agricultural production. However, due to the complex dynamics of the soil, its quality can not be directly measured, and can be estimated from man-made indicators. The changes in the status of soil quality have been evaluated through indicators, comparing them with desirable values at different time intervals (Chaer, 2001; Melo Filho et al., 2007).

Ramos et al. (2013) with the objective of evaluating the physical and hydraulic properties of a dystroferric Red Latosol under three management systems: pasture, coffee cultivation and native forest, found that in the system under bushes the lowest value of S was obtained, 0.0404, followed by coffee cultivation and pasture, with S values corresponding to 0.0453 and 0.0753, respectively. However, based on the values of S, they concluded that the adopted management did not alter the physical quality of the soil. According to Beutler et al. (2008), the parameter S evaluates the porosity that corresponds to the interparticle pores, cracks, biopores and macrostructural pores resulting from the management, the parameter still identifies the degradation of the physical quality, considering, for example, animal trampling, the different systems of use of the soil (soil under forest, soil under coffee plantations).

Andrade and Stone (2009), with the main objective of verifying the adequacy of the S index in the physical quality diagnosis of cerrado soils, selected 2364 samples from the Embrapa Rice and Feijoo and Embrapa Cerrados Soil Laboratories databases. Soils under cerrado of the regions CO, N and NE of Brazil. The authors correlated the S index values obtained from these samples with soil density, macroporosity and total porosity and found a negative and highly significant correlation between soil density and S index for the various textural classes considered. Total porosity and macroporosity, on the other hand, presented positive and also highly significant correlations with the S index. According to the authors, these results are in agreement with the one observed by Stone et al. (2005), for a dystrophic Red Latosol under cerrado vegetation.

Despite the importance of the incorporation of organic matter in the improvement of soil structure, the effects of organic fertilization on the physical properties of the same are still little known, from the use and management systems adopted. Therefore, the objective of this work was to evaluate the possible effect of different combinations of soil preparation and treatment (liming and organic and/or mineral fertilization) on the value of the parameter S, obtained from the water retention curve in the soil, in a Yellow Distrófico Argissolo in the municipality of Capão do Leão/RS/Brazil.

MATERIAL AND METHODS

The experiment was conducted in the municipality of Capão do Leão, Rio Grande do Sul, Brazil, whose geographic coordinates are 31º 45 'south latitude and 52º 27' west longitude, with an average altitude of 28 m. The soil where the experiment was deployed was classified by Cunha and Silveira (1996), as a typical Distrófico Yellow Argissolo, with sandy loam texture. The experiment was carried out in a randomized complete block design, with four replications, with 2.0 m wide and 4.0 m long, totaling 8 m^2, the treatments used being presented in Table 1. The vermicompost used was produced at from bovine manure and that, after the vermicompost process, presented the following characteristics: 0.63% of nitrogen; 0.15% phosphorus; 0.49% potassium; 0.18% calcium and 0.12% magnesium.

Table 1. Description of the treatments used in the experiment

Treatment	Description
T_1	Control
T_2	Vermicompost + Limestone
T_3	NPK+ Limestone
T_4	Limestone
T_5	Vermicompost
T_6	Vermicompost + Phosphorus + Limestone

For the determination of water retention curves in the soil, undisturbed samples were collected using volumetric rings of approximately 50 cm^3, in triplicate, in the 0 - 0.20 m depth layer. The analyzes were performed in the Physics Laboratory of the Department of Soils of FAEM/UFPel, applying the voltages of 0; 0.1; 6; 10; 33; 100; 500 and 1500 kPa. For the tensions up to 10 kPa the tensile table method was used and for the higher tensions, the Richards porous plate extractor (Klute, 1986). The data on the relationship between potentiometric and volumetric moisture were adjusted using the Soil Water Retention Curve- SWRC software developed by Dourado Neto et al. (2001), through the function proposed by van Genuchten (1980):

$$\theta = [\theta_r + (\theta_s - \theta_r)/[(1+ \alpha.\psi)^n]^m] \quad (1)$$

Being θ = water content (kg kg^{-1}); θr = residual water content (kg kg^{-1}), defined as the water content where $d\theta/d\Psi = 0$; θs = saturation water content (kg kg^{-1}); α (m^{-1}); n and m ($m = 1 - 1/n$) are the empirical parameters of the model.

$$S = -n(\theta_s - \theta_r)\left[1 + \frac{1}{m}\right]^{-(1+m)} \quad (2)$$

The soil density was obtained using the volumetric ring method, contained in the Manual of Soil Analysis Methods (Embrapa, 1997), in the same rings used to make the retention curve. Macro and microporosity were determined after adjusting the moisture versus potential data to the van Genuchten equation (1980), considering the matric potential of 6 kPa (Embrapa, 1997).

RESULTS AND DISCUSSION

The results showed that there was no statistical difference for the variables soil density, macroporosity, microporosity and total porosity among the different treatments, demonstrating that the use of organic fertilization and chemical fertilization, isolated or associated, in the amounts recommended by the Soil Fertility Commission For the RS and SC (1995), present the same effect when compared to the treatment without fertilization (Control). The average values obtained were 1530 kg m^{-3}, 0.077 $m^3 m^{-3}$, 0.282 $m^3 m^{-3}$ and 0.359 $m^3 m^{-3}$ for soil, macro and microporosity, and total porosity attributes.

Table 2 shows the parameters of the adjustment equation of the model proposed by van Genuchten (1980), as well as the coefficient of determination obtained, considering the different treatments used.

The values of the S parameter for the different no-tillage treatments are shown in Figure 1. The S-parameter values were higher in the treatments T4 (Limestone), T5 (Vermicompost) and T6 (Vermicompost + P + limestone), (0.037, 0.037 and 0.036, respectively), which, according to Dexter (2004), indicated favorable conditions for root growth. However, for treatments T1 (Control), T2 (Vermicompost + limestone) and T3 (NPK + limestone), the values found were 0.021, 0.024 and 0.026, respectively, indicating that there is some physical restriction to root growth. The higher values of S indicate a better pore configuration in the soil and, therefore, less physical restriction for plant root growth, either by aeration, mechanical restriction or water retention characteristics. Perhaps the low values found for the parameter are due to the effect of time in soil structuring after the incorporation of the fertilization, that is, only a sample collection may have been insufficient to detect improvements in soil structure.

Table 2. Adjustment parameters of the van Genuchten model (1980) for the different treatments used

	Parameters of soil water retention curve					
Treatment	α (m^{-1})	m	n	θr ($m^3 m^{-3}$)	θs ($m^3 m^{-3}$)	r^2
T_1	0.234	0.3064	0.6068	0.108	0.339	0.995
T_2	0.839	0.0887	1.6640	0.125	0.361	0.996
T_3	0.537	0.1060	1.5660	0.136	0.340	0.993
T_4	0.977	0.0395	4.2487	0.101	0.360	0.996
T_5	0.423	0.1934	1.3839	0.143	0.376	0.999
T_6	0.894	0.0304	5.1521	0.114	0.376	0.995

T_1: Control; T_2: Vermicompost + limestone; T_3 NPK + limestone; T_4: Limestone; T_5: Vermicompost; T_6: Vermicompost + P + limestone

Da Silva (2004), working with Red Yellow Latosol under two management conditions, forest and orange orchard, found S values of 0.1071 for the forest and 0.0265 for the orchard. According to the author, this shows that the soil in the forest presents a structural quality superior to the soil of the orchard, which presents an area trafficked by agricultural machines, with greater soil density due to compaction and, consequently, reveals a degraded physical structure.

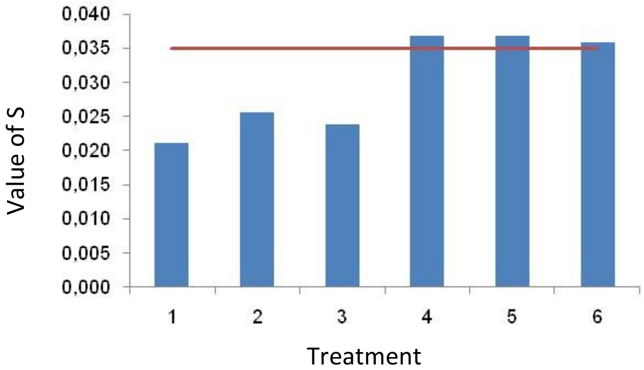

Figure 1. Behavior of the parameter S according to the treatments used, as well as the limiting value, suggested by Dexter (2004). T1: Control; T2: Vermicompost + limestone; T3: NPK + limestone; T4: Limestone; T5: Vermicompost; T6: Vermicompost + P + limestone.

CONCLUSIONS

The parameter S was higher (> 0.035) for the treatments in which limestone and vermicompost were applied alone and when there was an association between organic fertilization, phosphorus and limestone. For the 0 - 0.20 m layer there was no statistical difference between the treatments in the attributes soil density, macroporosity, microporosity and total porosity.

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4.11.P

WATER EROSION OF ULTISOLS UNDER CONVENTIONAL TILLAGE IN THE CERRADO-PANTANAL BRAZILIAN ECOTONE

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INTRODUCTION

Soil erosion has affected the agricultural productivity over time, and, therefore, it threatens food security (Smith, 2013). Even with the advance in soil conservation systems such as no-tillage system, conventional tillage practice continues to prevail, and it is inevitable in cases such as implantation and restoration of sugarcane and pasture fields (Youlton et al., 2016).

The climate condition is a key factor of understanding erosion process in some regions. In Mato Grosso do Sul State, located in Central-West Brazil, heavy rainfall have occurred during the summer resulting in an increase in soil loss ratio (Simonneaux et al., 2015) that is worsened due to planting or harvesting delays (Bussmann et al., 2016). In this study, we evaluate rainfall characteristics on the erosion of Ultisols under conventional tillage in the Cerrado-Pantanal Ecotone.

MATERIALS AND METHODS

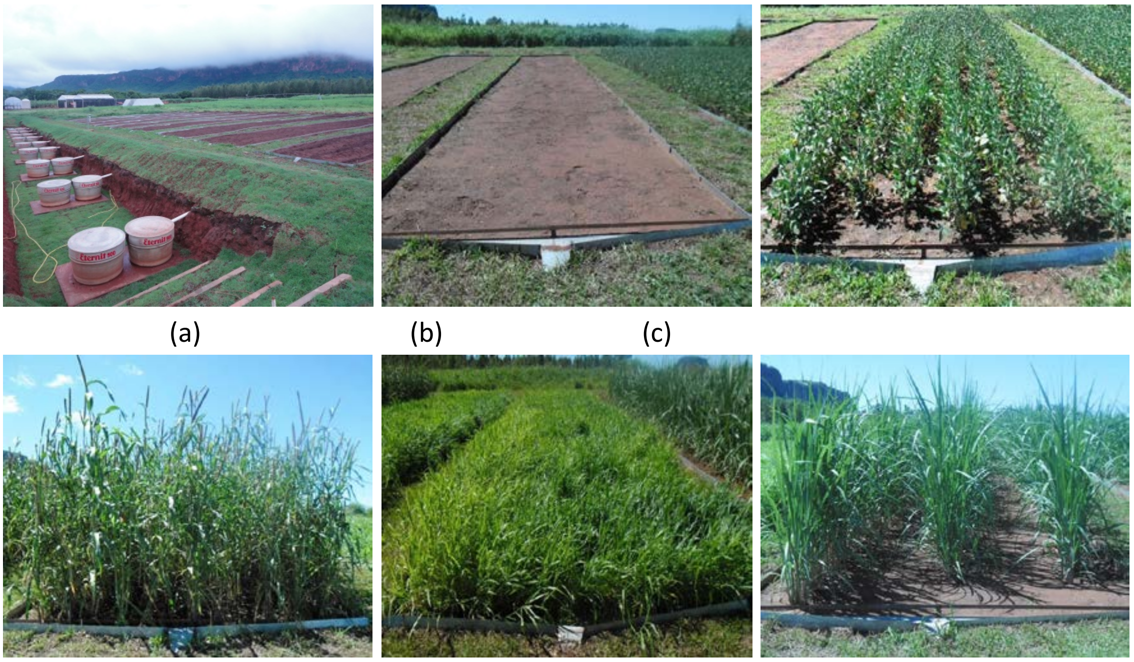
The experiment area is located at the State University of Mato Grosso do Sul – UEMS, Aquidauana City, MS (latitude 20°28′ S, longitude 55°40′ W, average elevation 191 m). This region is in the Cerrado-Pantanal Ecotone. The average slope is 0.03m m^{-1} and the soil is classified as Ultisols.

For each segment of uniform intensity, the kinetic energy of the rainfall was calculated by the equation (Wischmeier and Smith, 1978), and EI_{30} is the erosivity index of individual erosive rainfall events (MJ mm ha^{-1} h^{-1}), Ec is the total kinetic energy (MJ ha^{-1}), and I_{30} is the maximum 30-minute intensity (mm h^{-1})

The rainfall patterns were determined for each erosive rainfall event, classifying them as advanced, intermediate, and delayed time-sequence patterns.

The experiment plot area is 22.15 m long downslope by 3.50 m wide (77.53 m^2 per unit area basis). At the bottom of each plot was installed a collecting system (channel) using a PVC pipe connected to the first 500 L water tank, which is connected to another identical water tank by a 1/9” Geib-type divisor to allow the excess runoff to flow.

All treatments were cultivated under conventional tillage system with plowing followed by harrowing of two operations downslope by using leveling disk harrow and with fertilization and sowing or planting crops. The treatments, with two replications, were: (1) bare soil (BS); (2) soybean (SO); (3): millet (MI); (4): pasture (PA) e (5) sugarcane (SU) (Figure 1 a, c, d, e, and f, respectively). It was studied for 140 days.



(a) (b) (c)

(d) (e) (f)

Figure 1. Erosion plots in the treatments: plots at a moment before crops planting (a); bare soil (b); soybean (c); millet (d); pasture (e) e sugarcane (f). Photos taken in September, 2012 (a) and in March, 2013 (other photos).

For each treatment, the Soil Loss Ratio (SLR) was obtained by the ratio of soil loss in each treatment to losses observed in the bare soil. To evaluate soil and water loss in different stages of crop development, 140 days was divided into five equal periods.

The experiment was carried out following a completely randomized design using analysis of variance and subsequent application of the Turkey test ($p<0.05$) for comparing the control effect on water and soil loss mean values.

RESULTS AND DISCUSSION

The two heaviest rainfall events (33% of the total precipitated) caused 49% of the total soil loss and 37% of the total water loss in the bare soil treatment. However, in the other treatments with soil coverage, it was observed a different behavior between each other and the bare soil in the percentage of soil loss throughout the development of the crops.

Table 1. Precipitation, erosivity (EI_{30}), rainfall patterns, and soil loss by water erosion of Ultisols under different cultivation systems, in Aquidauana (MS)

Date	Rainfall mm	EI_{30} *	Treatment								
			BS	---- ²SO ----		---- MI ----		---- PA ----		---- SU ----	
			PS	PS	SLR ⁴	PS	SLR	PS	SLR	PS	SLR
12/13/2012	25	410 (AD)	350.6	342.8	0.978	357.2	1.019	371.2	1.059	371.1	1.058
12/17/2012	49	215 (IN)	681.9	634.6	0.931	787.5	1.155	805.0	1.180	751.9	1.102
12/18/2012	13	37 (IN)	15.1	7.8	0.518	4.4	0.289	1.8	0.122	7.2	0.474
12/30/2012	16	46 (IN)	19.1	19.0	0.997	12.5	0.655	12.8	0.672	6.5	0.341
01/07/2013	15	92 (AD)	407.3	138.4	0.340	97.5	0.239	67.1	0.165	120.2	0.295
02/04/2013	21	825 (IN)	2101.1	844.8	0.402	560.5	0.267	0.6	0.000	801.7	0.382

02/06/2013	18	135 (IN)	1000.8	209.7	0.209	0.3	0.000	0.1	0.000	258.0	0.258
02/08/2013	25	270 (AD)	1823.6	401.5	0.220	3.2	0.002	0.6	0.000	625.6	0.343
02/15/2013	15	164 (IN)	1110.1	258.0	0.232	5.4	0.005	2.7	0.002	297.4	0.268
02/16/2013	13	75 (DE)	1312.7	419.2	0.319	1. 9	0.001	0.9	0.001	393.8	0.300
03/13/2013	12	80 (AD)	1366.7	191.7	0.140	11.1	0.008	10.8	0.008	349.5	0.256
03/20/2013	39	441 (IN)	4506.7	573.3	0.127	9.5	0.002	0.1	0.000	1234.0	0.274
03/24/2013	14	45 (IN)	3305.1	297.1	0.090	2.1	0.001	0.7	0.000	819.4	0.248
03/25/2013	13	76 (AD)	960.5	20.2	0.021	0.6	0.001	0.1	0.000	14.3	0.015
04/02/2013	104	3430 (IN)	16281.1	2231.2	0.335	46.1	0.037	5.1	0.093	1214.4	0.164
04/04/2013	77	453 (DE)	3161.1	473.5	0.150	52.3	0.016	0.1	0.000	462.9	0.146
04/05/2013	29	169 (DE)	2397.1	25.3	0.011	1.1	0.000	0.8	0.000	54.4	0.023
04/06/2013	120	2482	22286.2	2014.9	0.090	174.1	0.008	54.1	0.002	1104.8	0.050
04/07/2013	22	78 (AD)	922.9	129.8	0.141	0.1	0.000	0.3	0.000	19.2	0.021
04/09/2013	12	77 (AD)	1542.2	115.4	0.075	0.2	0.000	1.8	0.001	13.4	0.009
04/10/2013	41	884 (IN)	9330.2	1485.7	0.159	56.0	0.006	2.6	0.000	459.4	0.049
04/13/2013	43	213 (IN)	3870.3	377.4	0.097	6.2	0.002	0.5	0.000	106.7	0.028
Total	672	10697	78752.5	11211.2	-	2189.8	-	1339.8	-	9487.0	-
Mean	29	-	3424.0	487.4	0.29	95.21	0.16	58.2	0.14	431.22	0.27
(%) Rel [5]	-	-	100 A [6]	14.2 B	-	2.8 C	-	1.7 C	-	12.0 B	-

* MJ mm ha^{-1} h^{-1}; [1] SB: bare soil; [2] SO: soybean, MI: millet, PA: pasture, SU: sugarcane; [3]AD: advanced rainfall pattern, IN: intermediate rainfall pattern, DE: delayed rainfall pattern; [4] SLR: Soil Loss Ratio (Mg ha^{-1}) (Mg ha^{-1})$^{-1}$; [5] (%) Rel: soil loss in relation to the treatments T1 after crops planting; [6]Identical capital letters for the mean soil loss values and small letters for the mean values are not different by Tukey Test 0.05.

The intermediate time-sequence pattern (IN) accounted for 50% of the total rainfall, the advanced time-sequence pattern (AD) accounted for 36%, and the delayed time-sequence pattern (DE) accounted for 14% of the total rainfall after crop sowing or planting. This rainfall patterns contributed to soil loss of, respectively, 40, 51 e 9%, relative to the total soil loss (Table 1).

The EI_{30} index values tended to behave as the rainfall patterns, with higher EI_{30} associated to the intermediate pattern (6435 MJ mm ha^{-1} h^{-1}) and the lower to the delayed pattern (697 MJ mm ha^{-1} h^{-1}).

Since rainfalls classified as intermediate pattern occurred more frequently, they led to EI_{30} index 9.2 times higher than the EI_{30} from the rainfall amount concentrated in the final third period for the delayed pattern. Thus, soil loss was 5.9 times higher than those rainfalls with a delayed pattern.

During the study period, EI_{30} index was 10697 MJ mm ha^{-1} h^{-1} (Table 2); also, the erosivity index found meets the Brazilian range, which is from 1672 to 22452 MJ mm ha^{-1} h^{-1} year^{-1} (Oliveira et al., 2013).

Larger soil loss was observed in BS treatment, exceeding seven to 59 times those observed in the other treatments, due to plant cover absence and soil overturning. The raindrop impact on the soil surface contributed to the formation of soil seals and, therefore, to erosion, supporting results found by (Kinnel, 2005).

Table 2. Precipitation, water loss (Lw) by water erosion of Ultisols under different cultivation systems, in Aquidauana (MS).

	Rainfall	---- BS ----	---- SO ----	---- MI ----	---- PA ----	---- SU ----
	mm	wl (mm)				
Total	672.34	256.47	233.96	112.37	31.00	235.11
(%) Rel	-	100 A	91 A	44 B	12 C	92 A

BS: bare soil; SO: soybean, MI: millet, PA: pasture, SU: sugarcane; wl: water loss (mm); (%) Rel: losses in relation to T1 treatment after crops planting, considering as 100; Identical capital letters for mean values are not significantly different by Tukey Test 0.05.

In between, there were different runoff values among treatments. Therefore, the infiltration rate values in treatments BS, SO, MI, PA, and SU accounted, respectively, for 62, 65, 83, 95, and 65% of precipitation. Overall, soils with a fine-textured subsoil horizon show limitation to water infiltration into the soil due to the high soil texture ratio between subsoil horizons A and B.

In BS treatment, accumulated soil loss was 59 times higher than PA treatment while accumulated water loss (Figure 2b) was 8 times higher. It was noticed less variation in water loss results than in soil loss results, considering different growth stages. We also observed in (Eduardo et al., 2013) that this higher variation mainly occurs at the final stages of growing cycles.

At the first stage, results from water and soil loss were similar to all treatments. However, higher soil loss was observed in BS treatment after the first stage while higher water loss data was accounted for BS, SO and SU treatments. Divergent results between soil and water loss among treatments were noticed at the final stages of growing cycle due to different biomass yield between cultivations, which differently covered the soil, and to higher rainfall intensities that occurred during the growth stages.

As SO, MI, PA e SU treatments had 91, 44, 12, and 92% of the total water loss observed in BS treatment, respectively (Figure 3b), PA was more efficient at controlling water loss. In general, increasing water loss was observed at the fourth growth stage because of higher rainfall amounts and intensities during this stage that led to an increase in soil loss in BS, SU, and SO treatments.

CONCLUSION

In the study period, we note that the predominant pattern of rainfall is the intermediate with a total rainfall erosivity of 10.697 MJ mm ha^{-1} h^{-1}. Surface runoff and soil loss are greater in the early stages of crop development. We find values of soil loss of 78.750, 11.200, 2.190, 1.340 and 9.490 kg for the bare soil, soybean, millet, pasture, and sugarcane, respectively. The soil loss ratio for soybean, millet, pasture and sugarcane is 0.29; 0.16; 0.14, and 0.27, respectively. Our results indicate that the presence of plant residues on the soil surface is more effective in reducing soil loss than the runoff.

ACKNOWLEDGMENTS:

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Session V: Methodological advances in the evaluation of soil and water degradation processes

5.2.O

PREDICTION TECHNOLOGIES FOR ASSESSMENT OF CLIMATE CHANGE IMPACTS

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Temperatures, precipitation, and weather patterns are changing, in response to increasing carbon dioxide in the atmosphere. With these relatively rapid changes, existing soil erosion prediction technologies that rely upon climate stationarity are potentially becoming less reliable. This is especially true of empirical technologies such as the Universal Soil Loss Equation which utilize maps of constant rainfall/runoff erosivity factors derived from historic climate data. The objectives of this presentation are to: 1.) Explore potential issues with existing empirical and process-based technologies for erosion prediction, 2.) Present new tools and procedures that can be used to assess impacts of climate change on future soil erosion, and 3.) Provide examples of how soil loss may change in the future at specific locations. Assessments of the impacts of future climate change are typically conducted by selection of a Global Circulation Model (GCM), obtaining predictions from the GCM for a particular (large) grid cell over the Earth's surface, and then down-scaling the information to a much finer local set for prediction of future temperatures and precipitation. For the application described here, we utilize the MarkSim web tool that can automatically access and downscale future climate projections from single or ensemble GCMs using the IPCC 5th Assessment Report. Output from MarkSim is used together with a custom software tool to automatically create future climate inputs and parameter files for use in CLIGEN (CLImate GENerator), which then creates climate input files for the process-based WEPP (Water Erosion Prediction Project) soil erosion model. Additionally, an updated WEPP version contains the ability to simulate the effects of changing atmospheric carbon dioxide levels and plant growth and cover production. In the evaluations to this point, future climate impacts are quite location specific. These new tools should provide options for conservation planners to assess possible impacts of changing climate, and effectiveness of existing land management practices.

Issues Related to Soil Erosion Prediction and Climate Change

A number of issues exist with existing empirical and process-based technologies for erosion prediction, related to their climates inputs. Empirical models such as the Universal Soil Loss Equation (USLE, Wischmeier and Smith, 1978) or its derivatives utilize weather information to obtain the rainfall/runoff erosivity "R" factor that is the summation of the products of the raindrop energy (E) and the maximum 30-minute rainfall intensity (I_{30}) for each storm event measured along with erosion plot soil losses. Additionally, R factor values can be computed using observed rainfall data at weather stations. These empirical models assume stationarity, which is that basically the driving rainfall erosivity factor remains constant into the future. Process-based soil erosion modeling technologies (e.g. WEPP, Flanagan et al., 2012) also depend upon climate inputs, usually from daily climate information created by a stochastic weather generator. Again, the parameterization for the weather generator may use observed climate data for the previous 20-50 years, and stationarity is often assumed for erosion model applications. Some of the questions that arise are:

1. How rapidly is climate changing, that updates to model climate inputs would be necessary?
2. How are changes in the temperatures and precipitation occurring at a location affecting all of the USLE/RUSLE factors? Certainly R-factors will be changing, but what about the C factor – changes in cover driven by warmer temps and greater CO_2 concentrations?, P factor – particularly effects of vegetative strips – changes in their growth, K factor – changes in soil properties due to changing temperatures, more loss of OM, more inputs of surface biomass?
3. Crop varieties are changing almost every year, resulting in plants that can produce more biomass and yield, different residue amounts, etc. Can these changes in plant characteristics be reflected in model inputs, and how can this be (easily) accomplished?
4. Models do not consider all processes that may be important. For example, the current WEPP model does not use the atmospheric CO_2 levels as an input that might affect plant growth and biomass production. Increases in biomass production and cover might somewhat offset potential increases in soil detachment from more frequent and/or more intense rainstorms. How important is it to include these types of inputs and processes?

Tools at the USDA-ARS NSERL

WEPP/SWAT Future Climate Input File Generator

Some recent efforts at the USDA-ARS National Soil Erosion Research Laboratory (NSERL) are targeted toward developing tools that can assist with determining effects of projected climate changes on runoff and soil loss predictions. Assessments of the impacts of future climate change are typically conducted by selection of a GCM, obtaining predictions from the GCM for a particular (large) grid cell over the Earth's surface, and then downscaling the information to a much finer local set for prediction of future temperatures and precipitation. One of the tools developed by the NSERL is a Microsoft Excel workbook (Figure 1) used in conjunction with the MarkSim DSSAT Weather File Generator (Jones and Thornton, 2013) web tool (Figure 2) to automatically access and downscale future climate projections from single

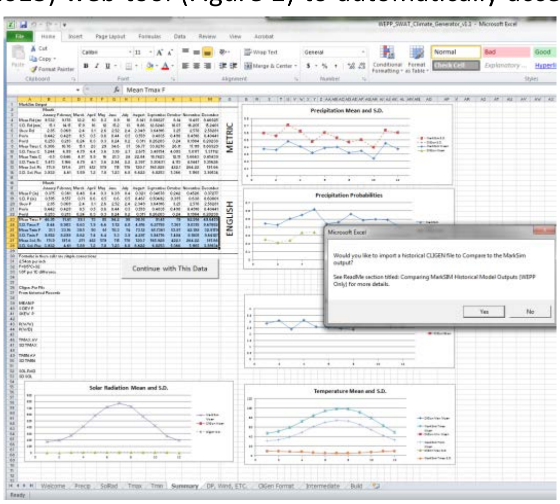

Figure 1. Future Climate Input File Generator

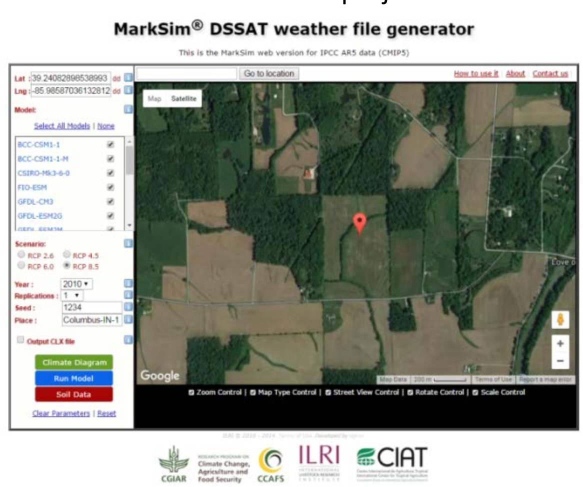

Figure 2. MarkSim web application

or ensemble GCMs using the IPCC 5th Assessment Report. Output from MarkSim (daily temperatures, precipitation, solar radiation) is used together with the custom software tool to automatically create future climate inputs and parameter files for use in CLIGEN, which then creates climate input files for the process-based WEPP (Water Erosion Prediction Project) soil erosion model. The WEPP/SWAT Future Climate Input File Generator software (Figure 1) is described in Trotochaud et al. (2016). It relies upon downloads of annual strings of projected daily weather parameters (typically 50 years) produced by the online MarkSim software (Jones and Thornton, 2013). The workbook then processes the data to create a new CLIGEN .par file (some of the parameters from an existing station .par file are also still used).

WEPP-CO2 Version

To simulate the effects of increased temperature and atmospheric carbon dioxide (CO_2) changes on crop yield, runoff, and soil loss, the Penman-Monteith method (Monteith, 1965; Allen et al., 1989) was incorporated into the WEPP (v2012.8) model for potential evapotranspiration calculation. The effects of the CO_2 level on canopy resistance and biomass energy conversion were computed following the procedure described by Stockle et al. (1992). The modified WEPP model requires a 'wepp-co2.txt' input file containing parameters needed to simulate the impact of increased atmospheric CO_2 level. To demonstrate the functionality of this modified WEPP version, model simulations were performed for a continuous corn (*Zea mays*) field under a spring-chisel plow tillage system, and a continuous soybean (*Glycine max*) field under a no-till cropping system on a uniform 100 m long, 3% slope gradient hillslope and a Miami silt loam soil. A 30-year base climate file was stochastically generated for West Lafayette, Indiana, USA using the CLIGEN weather generator (v5.3). To evaluate the effects of future rising temperatures and CO_2 levels, or a combination of both on simulated crop yields, runoff, and soil loss, four scenarios were simulated: 1) base climate and the current CO_2 level of 380 ppm (control); 2) increase in minimum and maximum daily temperatures in increments of 0.8°C; 3) increase in CO_2 level in increments of 100 ppm; and 4) increase in minimum and maximum daily temperatures in increments of 0.8°C and increase in CO_2 level in increments of 100 ppm. Changes in crop yield, runoff, and soil loss were computed as a ratio of future values to base values.

Increases in temperature resulted in crop yield reductions for both corn and soybeans (Figure 3). Runoff and soil loss exhibited small changes for corn and increasing trends for soybeans with increasing temperatures. Increases in CO_2 levels caused greater increases in crop yields for soybeans compared to corn (Figure 4).

Runoff for corn and soybeans showed slight increasing or decreasing trends, respectively, with increasing CO_2 level. Soil loss for soybeans was greater compared to that for corn. The combined effect of increasing temperatures and CO_2 levels on crop yield for soybeans was much greater compared to corn (Figure 5). Soil loss from the soybean field showed greater reductions than from the corn field.

CONCLUSIONS

Most available information indicates that climate is changing as a result of increasing global temperatures due to increasing greenhouse gases in the atmosphere. Soil erosion is a threat to soil and water resources and crop productivity. Being able to model the effects of climate change on soil erosion will allow for the development of land management strategies to minimize the impact of changing precipitation and temperatures. Software developed at the USDA-ARS NSERL provides some of these tools (Available via url: http://www.ars.usda.gov/Research/docs.htm?docid=10621).

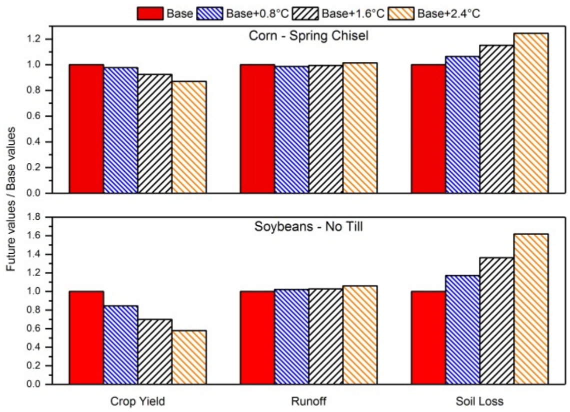

Figure 3. Effect of increase in daily maximum and minimum temperature on simulated crop yield, runoff, and soil loss for corn under spring-chisel and soybeans under no-till cropping systems in West Lafayette, Indiana, US (CO_2 at 380 ppm).

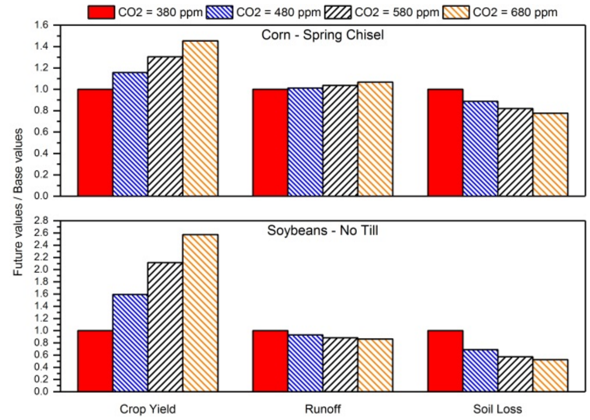

Figure 4. Effect of increase in CO_2 level on simulated crop yield, runoff, and soil loss for corn under spring-chisel and soybeans under no-till cropping systems in West Lafayette, Indiana, US.

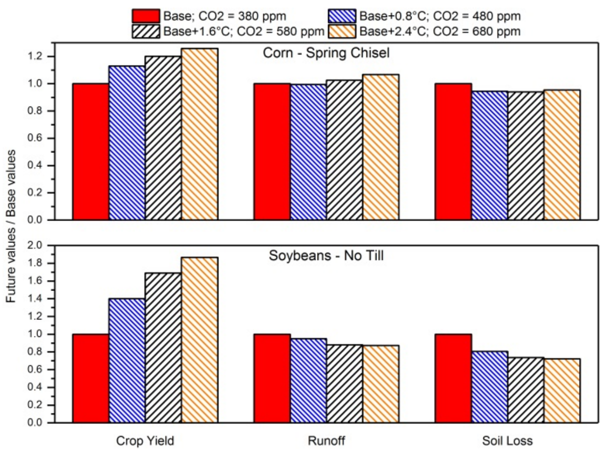

Figure 5. Combined effect of increase in daily maximum and minimum temperatures and increase in CO_2 level on simulated crop yield, runoff, and soil loss for corn under spring-chisel and soybeans under no-till cropping systems in West Lafayette, Indiana, US.

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5.4.O

AN AGROCLIMATIC RISK ANALYSIS APPROACH TO THE DEVELOPMENT OF RESILIENT AGRICULTURAL PRODUCTION SYSTEMS

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ABSTRACT
The objective of this study was to analyze the agroclimatic risk on several agricultural production systems as a basis for the development of resilient systems and the generation of agroclimatic adaptation and prevention models. The methodological approach is based on the estimation by temporal and spatial analysis of risk, which includes: (1) the identification of climatic hazards, through the analysis of anomalies or limiting extreme climatic events; estimation of the vulnerability of the productive system, defined by its exposure, sensitivity of the species to water stress and the adaptive capacity of the system against agroclimatic risk; and (3) obtaining the agroclimatic risk, which allows the identification of zones of agroclimatic adaptation where the production systems present a greater or lower risk of losses as a consequence of the identified hazards. This study shows the progress made so far in the analysis of agroclimatic risk for four production systems: melon (*Cucumis melo*), Key lime (*Citrus aurantifolia*), Andean raspberry (*Rubus glaucus*) and avocado (*Persea americana*) in the department of Cauca and Valle del Cauca (Colombia – South America). This risk analysis under deficit or excess precipitation scenarios allowed, through a participatory exercise with producers and technical assistants, to identify technological options based on soil and water management that contribute to increase the adaptation capacity of the production system as a first step in the development of resilient systems.

INTRODUCTION
Climatic variability associated with ENSO (La Niña and El Niño) has caused economic and environmental losses affecting enormously the agricultural sector, particularly in Colombia (Corpoica, 2013). Extreme weather events are also expected to become more frequent and with higher intensity. Additionally, water stress (deficit or excess) and the geomorphological susceptibility of the land are considered main factors affecting agricultural systems. As consequence of Climate Change, decrease of plant growth rate, changes in plant development, and changes in incidence and severity of diseases, pests and weeds are expected to affect crops (IPCC, 2012; Corpoica, 2013).

Corpoica and The Adaptation Fund carried out a project titled “Risk Reduction and Adaptation to Climate Change - Adaptation and Agroclimatic Prevention Models (MAPA)”, addressing future climate change scenarios in Colombian agriculture. The objectives of this study were to i) analyze the agroclimatic risk of four agricultural systems: melon (*Cucumis melo* L.), key lime (*Citrus aurantifolia* S.), andean raspberry (*Rubus glaucus* B.) and avocado (*Persea americana* var. Hass) in the departments of Cauca and Valle del Cauca; and to ii) validate technological options that could improve the agricultural systems, allowing the system to tolerate the impacts of extreme climatic events and keeping their socio-economic and environmental viability.

MATERIAL AND METHODS

We carried out the analysis in the municipalities of Roldanillo, Ginebra and Andalucia in the department of Valle del Cauca, and in the municipality of El Tambo in the department of Cauca. Melon, key lime, andean raspberry and avocado were evaluated. Crops were established between 980 to 1695 MASL. The rainfall ranges between 900 - 2010 mm/year, and the reference evapotranspiration (ETo) ranges between 1400-1600 mm/year. Minimum and maximum mean temperature were 14°C and 32°C respectively, with a relative humidity values between 65% to 90% (Corpoica, 2015a; Corpoica, 2015b).

The risk assessment used in this approach is the result of linking the threat, the vulnerability and risk elements of the agricultural system, to determining the social, economic and environmental consequences of a weather event (IPCC, 2012; McCarthy et al., 2001; Turner et al., 2003). The methodological approach we used was based on the estimation by temporal and spatial analysis of the risk, in the following stages:

1. Identification of climate threats: we performed a spatial and temporal characterization of climatic variables, behavior patterns and anomalies when events such as “El Niño” and “La Niña” and extreme events occurred. Data from 32 years (1980 to 2011) with emphasis on precipitation were used. Data were recorded monthly. The ARCGIS ®10.1 software was used to generate the spatial models of the analyzed variables and to (extrapolate) the data we used REGNIE and IDW software.
2. Vulnerability analysis of the system: we identified the susceptibility of the territory where crops were located (exposure) and the sensitivity of the crops to each of the climatic threat. We used the FAO land evaluation methodology (1976) to compare the crop requirements with the environmental supply. Additionally, we used the Palmer index to determine the probability of occurrence of water normality, water excess or deficit on the soil (Martínez et. al., 2015). We performed the analysis using soil studies and mapping at 1:100.000 scale (IGAC and CVC, 2004; IGAC, 2009).
3. Estimation of agroclimatic risk: we estimated overlapping the layers generated by the climatic threats and the vulnerability analysis for each crop. We identify zones of agroclimatic adaptation where the production systems present a greater or lesser risk of losses as consequence of the identified threats. To set the representative scenario of increased risk in every water condition, the most extreme and persistent condition in the area was selected. Scales for the evaluation of agroclimatic aptitude were established using a grading matrix under restrictive moisture conditions for the crops (Martínez et al., 2015). Finally, we generated descriptive cartography at a scale of 1: 100,000.

RESULTS AND DISCUSSION

The analysis of the climatic risk allowed the delimitation of zones of water stress for the crops evaluated, finding that the water deficit is the main factor climate threat for key lime and avocado crops in most of the municipalities of Andalucía (Valle del Cauca) and El Tambo (Cauca) respectively, while, the water excess is the main climatic threat identified for melon and andean raspberry crops in the municipalities of Roldanillo and Ginebra (Valle del Cauca) (Fig 1).

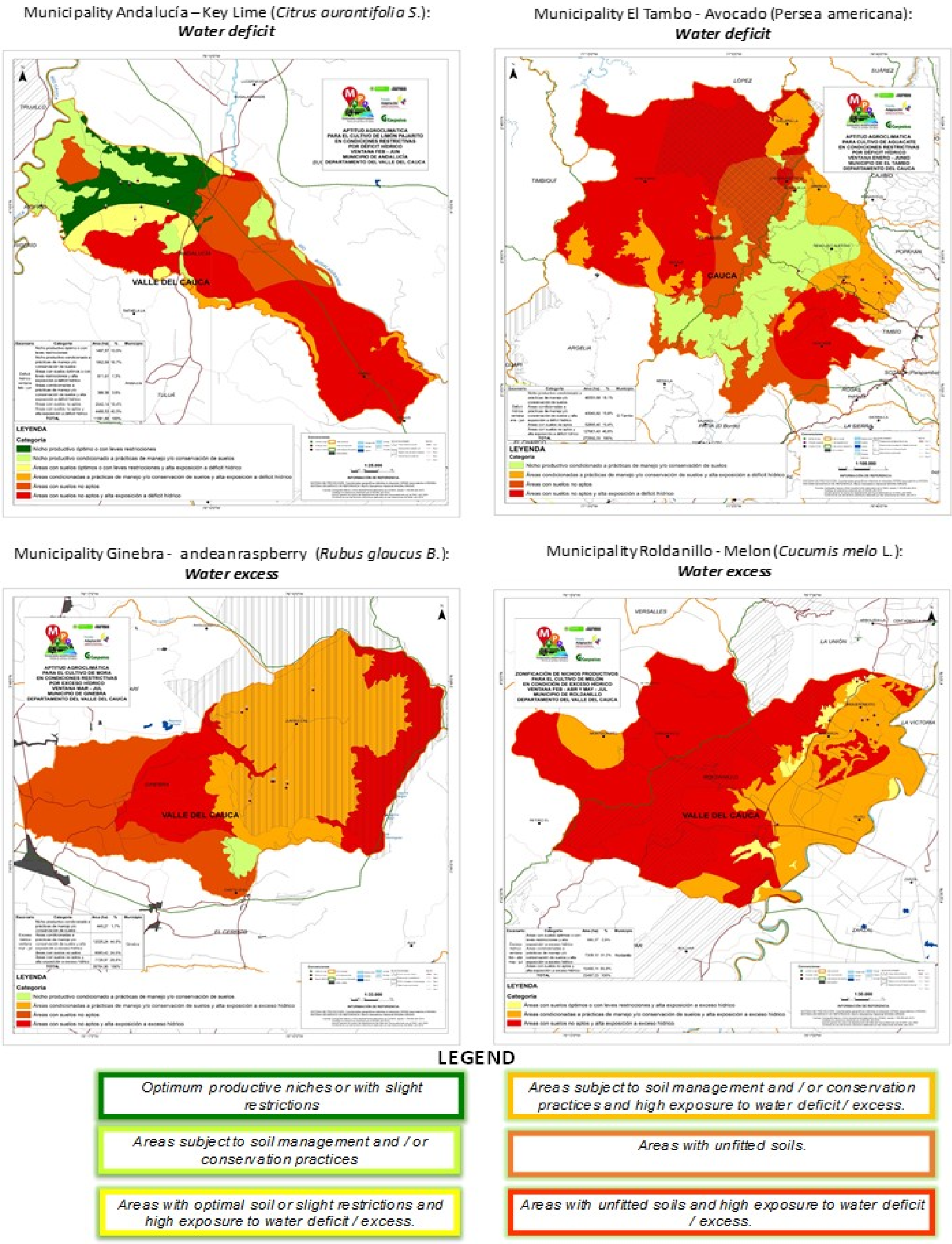

Figure 1. Identification of agroclimatic aptitude areas for prioritized crops. Modified of CORPOICA (2015a); CORPOICA (2015b).

A high probability was found (> 60%) for both climatic scenarios and distributed temporally according to analysis of each crop: Deficit: key lime (April - June and October - December) and avocado (throughout the year); Excess: andean raspberry (March - July) and melon (February - April and May - July). ENSO events influence the precipitation behavior in the evaluated municipalities, where the periods of excess precipitation are associated with La Niña, registering an increase between 22 - 25%, while the decrease of precipitation is associated with El Niño events, with decreases between 18 - 21% (table 1).

Table 1. Influence of ENSO events, and territorial susceptibility in four municipalities of Cauca and Valle del Cauca.

Municipality	Precipitation mm/year	Precipitation anomaly		Reference evapotranspiration (mm/year)	Frequency of water stress conditions (PDSI)	
		El Niño	La Niña		Excess	Deficit
El Tambo	2500 - 5000	-40% a -20%	0% a 60%	1200-1600	High	Middle
Andalucía	1300 - 2250	-40% a -20%	20% a 60%	1200-1600	Low	High
Ginebra	1300 - 2000	-40% a 0%	20% a 40%	1000-1400	Middle	Middle
Roldanillo	1100 - 1750	-40% a -20%	20% a 60%	1200-1400	Low	Low

Source: CORPOICA (2015a); CORPOICA (2015b).

The territorial susceptibility to deficit or excess water conditions was categorized as high to very high in the analysis municipalities (>50% of the total area), influenced by topographic conditions (slope > 30%) and soil characteristics (texture and effective depth) that directly affect the water balance and the availability of water for the crops (fig 1).

These factors affect directly the water balance and the availability of water for crops, as in the case of key lime and avocado, and indirectly in the predisposition for the appearance of phytosanitary problems such as those identified for melon and andean raspberry crops. In this way, the agroclimatic risk analyzed allowed, through a participatory exercise with producers and technical assistants, to identify technological options focused on the management of soil and water resources, which contribute to the efficient use of irrigation and fertilization, and to planning of practices for crop diseases control, during the periods of greatest exposure to the climatic hazards identified at each site.

CONCLUSIONS

The analysis of the agroclimatic risk for the evaluated crops allows the approach to the development of resilient production systems, as being a planning tool, based on the identification of limiting factors of the territory and the vulnerability of the crop against climate variability, for the timely implementation of local and regional adaptation measures such as crop management practices and the management of natural resources, among others.

ACKNOWLEDGMENTS
The authors express their thanks to the Adaptation Fund for project financing MAPA, the team agroclimatología and technological options - Corpoica responsible for processing project information and other people who made possible the development of work.

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5.6.O

RISK TO DESERTIFICATION IN TROPICAL AREAS. APPLICATION OF THE MULTIFACTORIAL MODEL OF RISK ANALYSIS TO DESERTIFICATION (MRAD)

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ABSTRACT

The aim of the study was the validation of a multifactorial methodological model to perform a comprehensive risk analysis for desertification (MRAD) in tropical areas. The risk assessment used by this investigation is the result of relating the hazard, the vulnerability and the elements under risk (subject). The model includes: (1) the estimation of the hazard desertification by using climatic, geomorphometric and land use/land cover indicators; (2) the generation of the edaphic vulnerability index to desertification (exposition, sensibility and adaptability), reached through the estimation of the inherent soil quality; and (3) the generation of the risk index to desertification. An evaluation of the Amaime river basin, applying the approach showed that it is possible to differentiate areas with higher or lower risk, supported by the analysis through spatial modeling of available data surfaces, that weight the influence through a regional level of biophysical and socioeconomically factors on the process of soil desertification and degradation. Furthermore, the output of the model has higher accuracy and resolution; a reason to propose its application to similar regions in tropical areas. It was identified that the Amaime river basin has a medium risk to desertification. It is explained by: high to medium hazard given by areas with water deficit in the flat part of the basin and by relief with slopes greater than 25%, with transient crop cover predominating, permanent crops and grasses, with high intensive land use, which are distributed from the western sector to the northeastern part of the basin. The vulnerability of the basin classified between medium to high is determined by the parental material, the fertility of its soils and by the low capacity of resistance to degradation, and by limitations such as the low content of organic matter in the flat zone and by the shallow effective depth feature of mountainous reliefs.

Keys: hazard; vulnerability; climate change; spatial analysis; land degradation.

INTRODUCTION

Desertification is explained by the combination of multiple social and biophysical factors, including agricultural activities, increased levels of aridity, increased infrastructure, and extractive activities (UNCCD, 1994; Geist and Lambin, 2004; UNCCD, 2013; Salvati and Bajocco, 2011). The objective of this work was to validate a multifactorial methodological model to perform a comprehensive risk analysis for desertification (MRAD) in tropical areas, based on the spatial analysis for the comprehensive evaluation of desertification at the regional and local levels. This includes: (1) the estimation of the hazard to desertification, through the use of climatic, geomorphometric and land cover indicators; (2) generation of the soil vulnerability index to desertification, by estimating inherent soil quality; and (3) obtaining the risk index for desertification.

MATERIAL AND METHODS

The MRAD calibration was carried out in dry areas in the western slope of the central Andes of Colombia, corresponding to the Amaime River Basin (Valle del Cauca - Colombia), located on, between 900 and 4100 MASL, and geographical coordinates 3º 29 'and the 3º 47' of latitude North and 75º 55 'and the 76º 12' of longitude West. The annual rainfall values in the region vary between

900 and 2200 mm. The most of this precipitation falls during two wet seasons throughout the year, the first one between March and May and the another one between September and November while the month with the lowest rainfall values is July. The proposed methodological model to perform a comprehensive risk analysis for desertification (MRAD) assumes the Risk (R) is the result of relating the hazard (H), the vulnerability (V) and the elements at risk (subject or system) (Cardona, 2003; De-Lange, 2010), which is represented by the function (Eq. 1):

$$R_{ij} = \left[(\mathrm{NHI}_{ij} * \ \mathrm{AnHI}_{ij})\right] * \left[(\mathrm{ExI}_{ij} + \mathrm{SI}_{ij}) - (\mathrm{RDI}_{ij})\right] \quad \text{(Eq. 1)}$$

Where:

Rij	=	Risk index to desertification.
NHIij	=	Natural Hazard Indicators.
AnHIij	=	Anthropic Hazard Indicators.
ExIij	=	Exposure Indicators.
SIij	=	Sensitivity Indicators.
RDIij	=	Resistance Capacity to degradation Indicators.

Data processing was done using three steps, (1) the estimation of the threat to desertification, using climatic, geomorphometric and land cover indicators (Eq. 2, 3, 4); (2) the generation of the vulnerability index soil, achieved by estimating the inherent soil quality (exposure, sensitivity and adaptability) (Eq. 5,6 y 7), and (3) obtaining the risk index to desertification (Fig. 1).

(Eq. 2)

$$NHI\ geomorphometric\ ij = (\ Slope\ *\ Slope\ Lenght\ Factor\ *\ Curvature\ *\ Topographic\ humidity\ I.)^{(1/4)} \quad \text{(Eq. 3)}$$

$$AnHI\ ij = (\ Vegetation\ quality\ *\ Intensive\ Land\ Use\ )^{(1/2)} \quad \text{(Eq. 4)}$$

$$ExI\ ij = (Parent\ material\ *\ Texture\ *\ pH\ *\ Cation\ Exchange\ Capacity\ *\ Rock\ fragments)^{(1/5)} \quad \text{(Eq. 5)}$$

$$SI\ ij = (\ Erosion\ *\ Electrical\ conductivity\ *\ Exchangeable\ Sodium\ Porcentage\ )^{(1/3)} \quad \text{(Eq. 6)}$$

$$RDI\ ij = (\ Efective\ depth\ *\ Structure\ *\ Drainage\ *\ Carbon\ )^{(1/4)} \quad \text{(Eq. 7)}$$

The product of the hazard and the vulnerability was performed by analyzing and processing spatial information of cartographic soil units. These units have associated attributes of one or more selected soils as modals, and containing numeric and alphanumeric information, which extracted data variables, involved in MRAD. These variables were assigned dimensionless values to pre-established ranges and categories, regarding the degree of participation in the causes or processes of degradation (Fig. 1). Basic secondary and primary information of the zone were used as daily records of climatic variables, satellite images and soil studies at a scale of 1: 100.000 and 1: 50.000. The risk index for desertification is achieved with the product of the information on the threat and vulnerability indexes of each Regions of Risk Analysis to Desertification (RRAD), based on the information processing through spatial analysis and mapping to define areas with an equal level of risk (Fig. 1.) Through of the differentiation of risk scales, the identification of nodes with risk to desertification are classified in three categories: Node 1: Low risk (<1.5), Node 2: Moderate risk (1,5 - 2.5) and Node 3: High risk (> 2.5).

RESULTS AND DISCUSSION

The spatial analysis outputs of the model allowed to establish that 98.2% of the total area of the basin presents important degree of risk to desertification, of which 67.744 ha (64.9%) are located in the alluvial plain, 34683.2 ha (33%) in the mountain position and 34.683 ha (33%) in piedmont (Fig. 2).

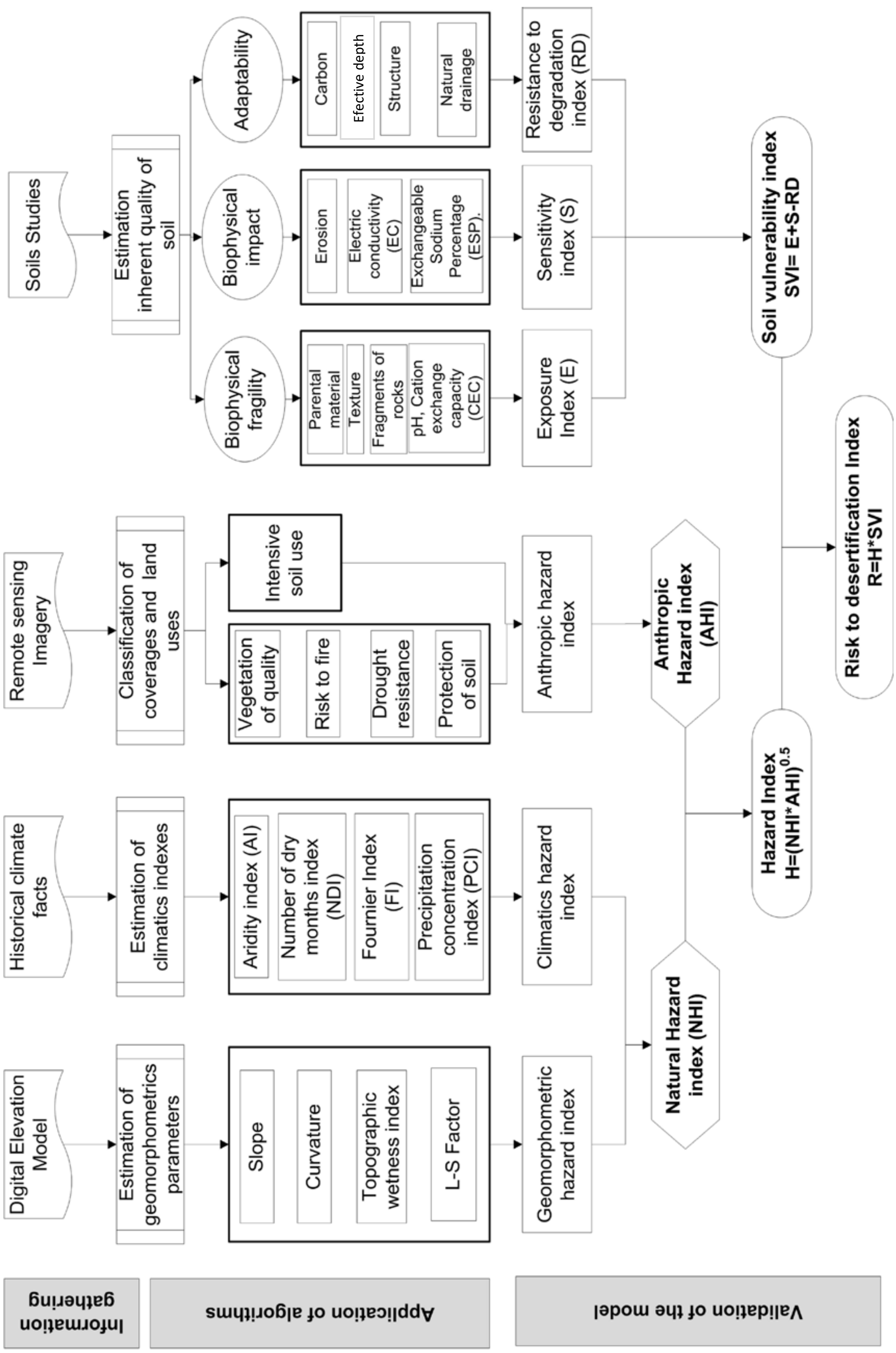

Figure 1. General structure of the methodological Model risk analysis to desertification (MRAD).

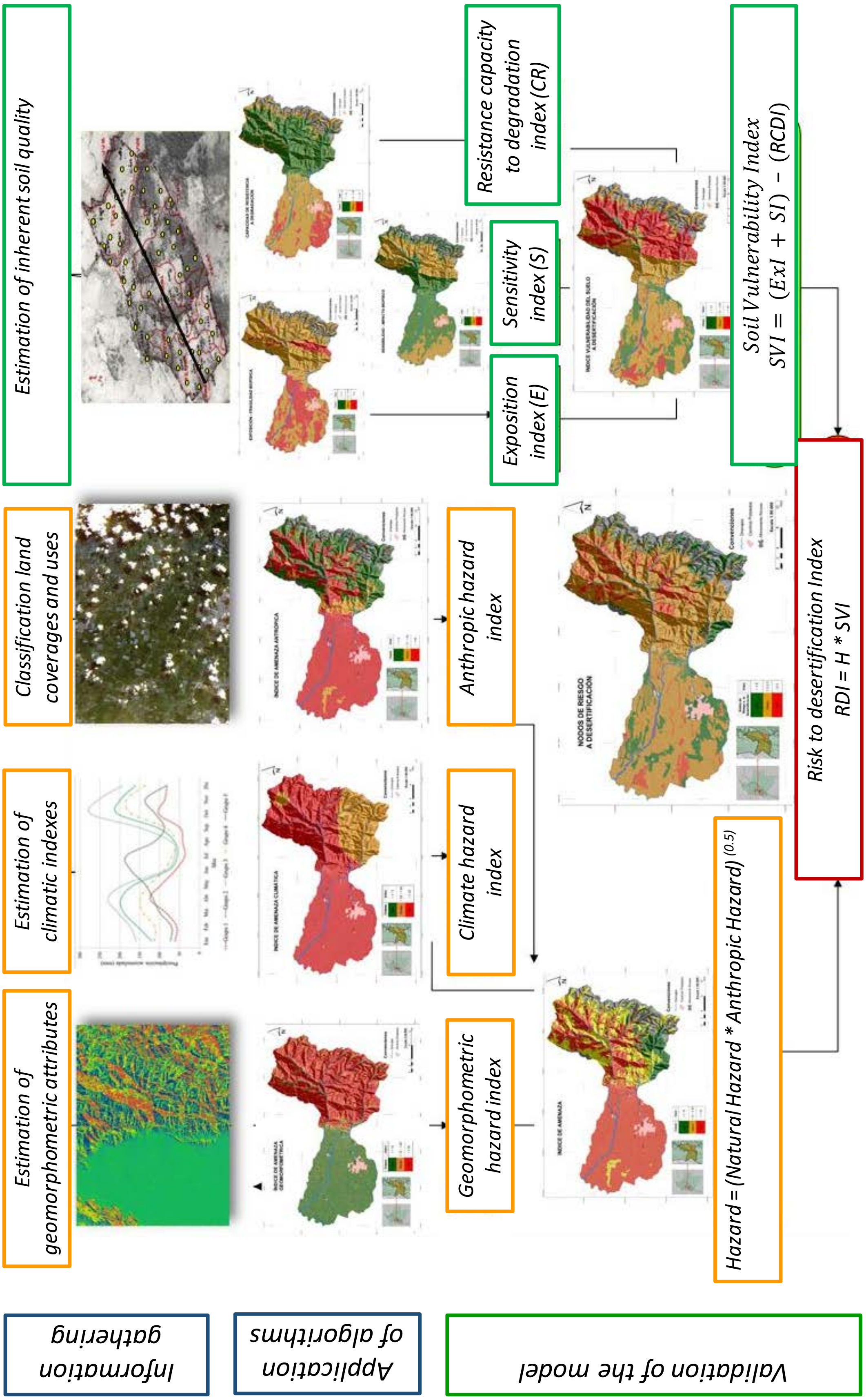

Figure 2. Calibration of the risk analysis for desertification (MRAD). Amaime River Basin (Valle del Cauca - Colombia).

The results of the evaluation of the hazard and vulnerability indices allowed to identify that the moderate risk is the dominant condition in the Amaime river basin. It is because a high to medium hazard, given by the conditions in the areas with water deficit in the flat part of the basin and by relief with slopes greater than 25%, made up mainly of transient crops, permanent crops and grasses with an intensive use of soils, which are distributed from the western sector to the northeastern part of the basin. The high risk to desertification covers areas of the territory with high vulnerability in areas where erosion and salinity processes occur, and with high hazard represented mainly by steep slopes in mountainous, areas on which pastures have been established and which are in expansion, and by the water deficit determined by the climate in flat area.

The assessment of soil vulnerability to desertification in the RRADs found in the Amaime River Basin allowed the study area to be classified in the category of zones with medium vulnerability, reaching an average of 1.27 for the estimated index (IV) (Fig.2). With the analysis of the RARD soil quality indicators it was possible to establish that 28.8% of its territory presents areas with high vulnerability to desertification, affecting approximately 29683 hectares, mainly located in mountain areas; While 43.9% and 15.7% of the total basin area are classified as areas with medium and low vulnerability to desertification.

CONCLUSIONS

The application and calibration of the MRAD model to the Amaime River Basin (Colombia), allow us to affirm that, through this methodological approach it is possible to differentiate areas with greater or lesser risk to desertification by spatial modeling of available data surfaces. The advantage of the model is the way to weight the influence of biophysical and socioeconomic factors, on the processes of desertification and land degradation with greater precision and resolution.

The methodological sequence of the MRAD and the spatial data processing allow the identification of potential degradation, and loss of land quality processes by obtaining intermediate and final products (hazard, vulnerability and risk indices) of interest for Modeling, diagnosis and decision making at regional and local level.

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5.7.O

NEW METHODOLOGY TO CALCULATE THE EROSIVITY OF A STORM WITH A 10 YEAR RECURRENCE (R_{10}) FOR APPLICATION OF RUSLE IN SPAIN

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SUMMARY

The present report is focused on a study on the quantitative characterization of a very important parameter of the rain which is important because it is used in the models of water erosion estimation and, therefore, of the loss of soil that our fields withstand. This parameter is the erosivity factor from RUSLE (Revised Universal Soil Loss Equation) in each of its versions. The R factor is an index of the erosivity at a location, it is obtained with the product of total storm kinetic energy (KE) times the maximum 30-min intensity (I_{30}).

RUSLE1 and 2 uses runoff to compute the effect of ponding on erosivity, the effect of contouring, critical slope length when contouring fails and sediment transport capacity and deposition associated with strips of dense vegetation, concave slopes and low grade terraces and diversions channels. RUSLE2 uses a storm with a 10 year recurrence interval in its runoff computations.

R_{10} ($KE_{24h,10yr}$*I_{10yr}) value is used to compute runoff. This factor represents the erosivity of a storm with 10 year return period. Therefore, it is the maximum storm erosivity that occurs in any year that has the probability of occurring once every 10 years which is the erosivity of an infrequent moderately erosive rain.

This report presents a new methodology to calculate R_{10} factor in Spain. This new methodology includes modern methods to obtain maximum 30-min intensity (I_{30}), such as regionalization, and more adequate functions, such as the SQRT-ET max function of distribution. And in order to obtain KE, the relationships existing between the kinetic energy and the daily rainfall with the data of rainfall and kinetic energy registered in pluviometers are presented.

The results have been processed with GIS and AJAX techniques. This processing has allowed the design of an interactive informatics application, in such a way that it is possible the easy and fast knowledge of R_{10} in somewhere of peninsular Spain.

INTRODUCTION

Excess rainfall rate or runoff is a key variable used by RUSLE1 and RUSLE2 to compute erosion reduction by support practices and reduction of R factor for ponding.

R_{10} value is used to compute runoff, it is the product of Kinetic energy ($KE_{24h,10yr}$) for a 10 year-24 hour precipitation ($P_{24h,10yr}$) times maximum 30-min intensity (I_{30}) for this precipitation. This precipitation is the storm amount that occurs in a 24 hour period that has the probability of occurring once every 10 years (a 10-year return period).

RUSLE1 and RUSLE2 use flow rate values for runoff to compute sediment transport capacity, contouring effectiveness and contouring failure and besides to compute the adjustment factor for flat slope of R factor (Foster, 2005).

The practices are more or less effective depending on storm severity, and the 10 year 24h precipitation is an index of storm severity that varies by location. A more erosive storm than an average annual storm is used because support practices effectiveness and its loss depend on large storm severity (Foster et al, 1997).

RUSLE 1 obtain the value for this storm from 10 year R event (R_{10}), and RUSLE2, one way to obtain this value is to enter values for the 10 year R event like that used in RUSLE1 (Foster, 2004).

METHODOLOGY

This research proposes a new methodology in order to calculate R_{10} factor ($KE_{24h,10yr}*I_{30}$) for $P_{24h,10yr}$ in Spain.

Obtaining $P_{24h,10yr}$

$P_{24h,10yr}$, the 24 hours rainfall associated to a 10 year return period is obtained in the study "Máximas lluvias diarias en la España peninsular" (1999) ; the nomograph was edited by the Ministry of Public Works in Spain. It is based on a regional analysis of this data, obtained in the 1545 rainfall gauge network system available in Spain. The regional method adopted was the Index Method. Regional frequency analysis increases data using series from other sites that are judged to have similar frequency distributions to the site of interest, forming a homogeneous region. 'Regional approach pretends to solve lack of temporal data with space data' (Hosking and Wallis, 1997). Second part of the study implied choosing an adequate frequency distribution, and the SQRT-max was justified to be the best. Finally in this study the authors developed an application to obtain this variable in the Spanish peninsular area.

Obtaining $KE_{24h,10yr}$

This paper presents the relationships existing between the kinetic energy and the daily rainfall with the data of rainfall registered by pluviometers (Roldán, 2006). The KE is estimated with potential equations from daily rain.

In order to obtain these relationships, the rainfall Kinetic energy in 24 hours is obtained from the sum of kinetic energies of precipitations calculated of registered rainfall in a day. The KE is obtained with a disdrometer, instrument of measurement that provides information on the rainfall and allows the characterization of this rainfall (Joss and Waldvogel, 1967).

It is recommended the combined use of the two potential equations.

The equations that provided the best fits and correlation coefficients were the following:

$$KE_{24h} = 17882\,(1 - 3.5^{(-1.8392*10^{-6}*P_{24h}{}^{2} - 8.322*10^{-4}*P_{24h})}) \text{ .... } (\textit{Equation 1}) \qquad R^2 = 0.997 \text{ for}$$

P_{24h}>140mm, unusual P_{24h}

KE_{24h} is the kinetic energy in J/m2 caused by the daily precipitation P_{24h} in mm.

$$KE_{24h} = 17882\,(1 - 3.5^{(-3.279*10^{-6}*P_{24h}{}^{2} - 6.2967*10^{-4}*P_{24h})}) \text{ ....} (\textit{Equation 2}) \qquad R^2 = 0.994 \text{ for}$$

$1<P_{24h}\leq 140$mm, usual P_{24h}

KE_{24h} is the kinetic energy in J/m^2 caused by the daily precipitation P_{24h} in mm.

With these expressions $KE_{24h,10yr}$ for the $P_{24h,10yr}$ is obtained

Obtaining I_{30}

I_{30}, maximum intensity for 30 minutes is obtained using MAXIN, (http://www.ecogesfor.org/recursos.html) application based on regional approach of annual maximum rainfall intensity. Hosking and Wallis (1997) techniques were used to form a region; where a region included dimensionless "short durations" series, and authors established 1 hour as the threshold to be incorporated in a "region"- short or large duration series- . Regionalization technique, established in this way, was a solution to lack of spatial annual maximum intensity information in Spain, and besides it improved robustness in estimates.

In addition, SQRT-max was used for the adjustment, as it had been used for 24 hours maximum rainfall studies in Spain with good results.

Finally, in order to extend results where no recording rain gauge was available, the following expression was used

$$I(d,T) = \frac{P_{24,\mathrm{T}}}{24} * g(d)h_c(T)$$

$$g(d) = \frac{\overline{\mathrm{I}}_{\mathrm{d}}}{\overline{\mathrm{I}}_D} = \left(\frac{\overline{\mathrm{I}}_1}{\overline{\mathrm{I}}_D}\right)^{\frac{24^{\mathrm{a}} - d^{\mathrm{a}}}{24^{\mathrm{a}} - 1}} = K^{\frac{24^{\mathrm{a}} - d^{\mathrm{a}}}{24^{\mathrm{a}} - 1}}$$

hC-1(T)= -0,0004*(Ln(T))2+0,0092*Ln(T)+1,0044

hC-2(T)= -0,007*(Ln(T))2+0,1066*Ln(T)+0,9086

Where,
d, duration (hours)
K,a, parameters depending on the place of study
And two expressions for h(T), depending on the place of study.

RESULTS

Each parameter involved in the estimation of R_{10} factor was represented as raster geospatial data. AJAX techniques were used to transform these geographic static information into an interactive web application. This application support an interactive map (Figure 1) where by typing geographic coordinates or researching a location in Spain, the R_{10} value can be obtained.
This application is available in http://www.ecogesfor.org/recursos.html

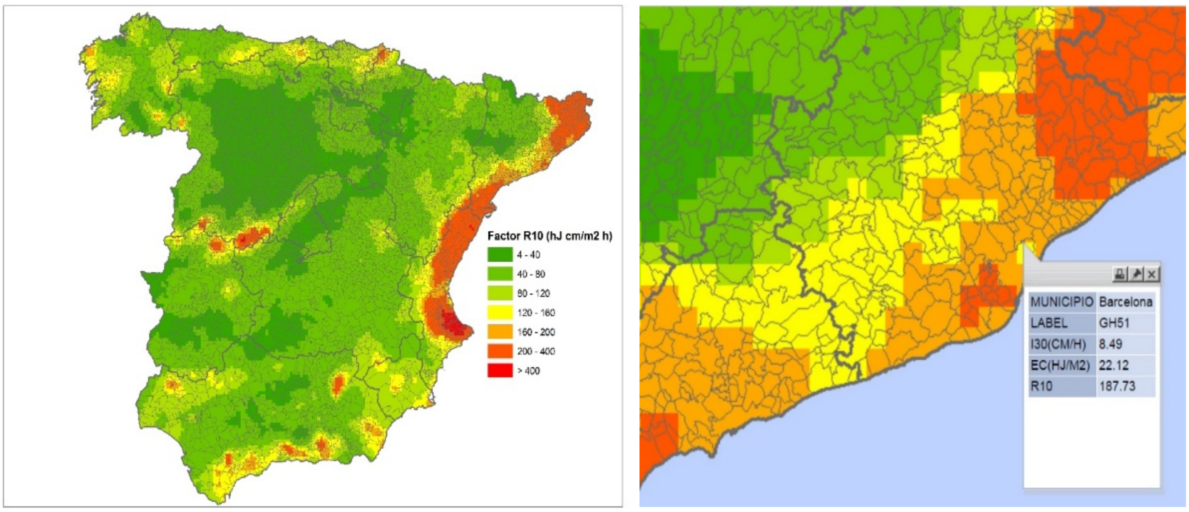

Figure 1. Raster Map of the R_{10} Factor for Spain

CONCLUSIONS

The methodology has the advantage of obtaining with facility the data $P_{24h,10yr}$ and $KE_{24h,10yr}$ and I_{30} for this precipitation.
In addition, methodology has permitted to design an interactive informatics application and its use presents others advantages like simplicity and rapidity in the estimate of the R_{10} factor ($KE_{24h,10yr} * I_{30}$) and therefore, its use in the application of RUSLE (v.1 and v.2) to estimate soil loss and sediment emission.
The R_{10} factor values should be proposed as constitutive standards for the whole country. This change will result in more efficient conservation practices and will lead in future to better protection of soil from erosion.

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5.8.O

THE EROSION PROCESSES OF GULLY BEDS AND ITS INFLUENCES ON HEADCUT RETREAT BASED ON AN IN-SITU SCOURING EXPERIMENT

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1. INTRODUCTION

Headcut retreat is one of the important processes of gully erosion, which leads to intensive sediment yield and threaten the soil quality of a region (Oostwoud Wijdenes et al., 2000; Vandekerckhove et al., 2001; 2003). The waterfall at the headcuts dissipated flow kinetic energy to gully beds and formed plunge pools right under the headcuts (Stein et al., 1993). The development of plunge pools caused rill headcut migration to upstream drainage area (Bennett et al., 1999), and the shape factors were applied to predict the linear headcut retreat rates of rills (Alonso et al., 2002). Flores-Cervantes et al (2006) extended the model to hillslope gully heads in the model channel-hillslope integrated landscape development (CHILD). Campo-Bescos et al (2013) calibrated the model in northeast Spain and found the differences between field observation and model simulation were only 0.05 m^3 $year^{-1}$. While the plunge pools were not typical in some gullies in fields, and how to predict the headcut erosion rates without shape factors of plunge pools needs further study. The aim of this study is to detect the relationships between headcut retreat rate and flow hydraulics and erosion processes of gully beds, and find out the key influencing factors on gully head eroison based on an in-situ scouring experiment on 5 simulated gully heads in Dry-hot valley in southwest China.

2. MATERIAL AND METHODS

Five new gully head plots (named as gully A, B, C, D and E) were constructed on former gully heads in July 2012. The five gully heads had similar initial topographies. The gully beds were 20 m long and 2 m wide. The headcut height was 0.5 m; the initial slope of the five gully beds ranged between 18.2-19.1%. The soil dry bulk density ranged from 1.65-1.73 g cm^{-3}, and the soil textures were mainly sandy clay loams (USDA) with sand content over 50% by mass. The flow scouring tests on each gully bed lasted 1 h, including 7 tests with a flow discharge of 83.3 l min^{-1} and 4 tests with a flow discharge of 166.7 l min^{-1}. For each gully head, four cross-sections (located at end of drainage area, 1m, 2m and 4m from bottom of headuts at gully beds) were used to monitor the mean flow velocity (v, by dyeing method), mean flow depth (h) and mean flow width (w) (measured by tape) every 10 minutes during the flow scouring tests. Other parameters including Reynolds Number (Re), Froude Number (Fr), shear stress (τ, N m^{-2}), Darcy–Weisbach friction factor (f), and Manning coefficient (n) were calculated also.

3. RESULTS AND DISCUSSION

3.1. LANDFORM CHANGES OF GULLY HEADS

All 5 gully heads were clearly erode by runoff, and the headcut retreat processes were accompanied with the incision of gully beds close to headcut (0-4 m). The total incision depths of gully beds were ranged from 0.08 to 0.26 m, and the total linear headcut retreats were ranged from 0.24 to 1.85 m. The plunge pools were not clearly developed during the tests (Fig 1). The linear retreat distances generally decreased with the decreasing incision depth of gully beds.

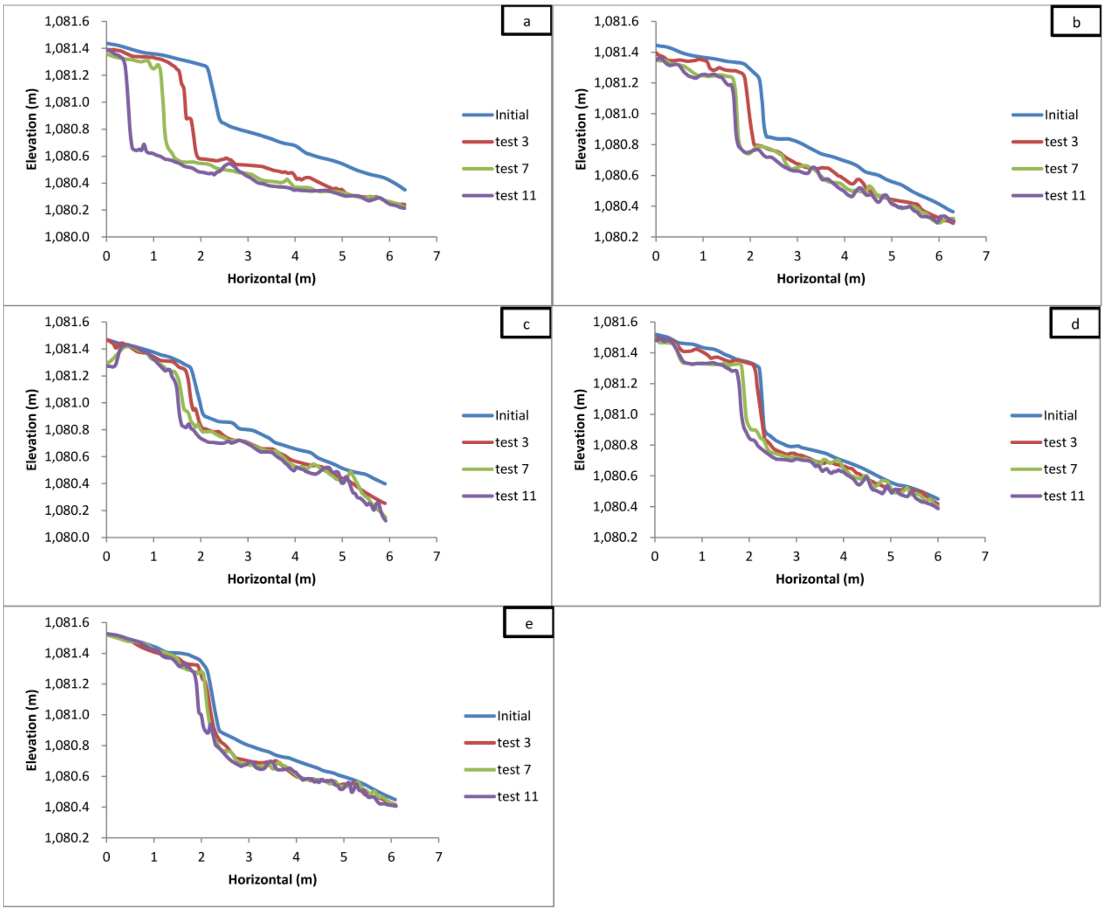

Fig 1 The Longitudinal profile of the five gully heads during the tests (a. plot A, b. plot B, c. plot C, d. plot D, e. plot E)

3.2 THE RELATIONSHIPS BETWEEN HYDRAULICS AND EROSION IN GULLY HEADS

The average *Re* values of 5 gully beds were above 5000, which indicated the turbulence flow conditions on gully beds. The average *Fr* values were mainly ranged from 0.8 to 1.2, which indicated the near-critical flow in gully beds. The *f*, *n* and τ weak correlated with the erosion volume of gully beds ($R^2 < 0.25$, $P < 0.01$, $n = 55$), and no relationships were found between hydraulic parameters and erosion volumes of headcut retreat.

3.3 THE RELATIONSHIPS BETWEEN GULLY BED INCISION AND HEADCUT RETREAT

Though plunge pools were not clearly developed during the tests, the erosion volumes of headcut retreat during each test showed a significant linear relation with the volume changes of gully beds (R^2 = 0.80, P < 0.01, n = 55). This indicated the erosion rates of gully beds close to headcut (0-4 m) have potential to predict the retreat rates instead of shape factors of plunge pools.

The hydraulics and soil conditions did not show clearly differences between each plot in drainage area, while the clay content and organic matters showed clearly difference between each gully bed, which affected the erodibitlites of soils of gully beds. The total erosion volume of gully bed A was 0.92 (m^3), while the value was only 0.09 m^3 in gully bed E. The soil clay content and organic matter in gully bed E was 46.9% (by mass) and 11.21 g kg^{-1}, which was about 2.7 and 2.2 times higher than those in gully bed A, which indicated the different erosion volumes of each gully head in this study were mainly attributed to variation of soil erodibilities of gully beds.

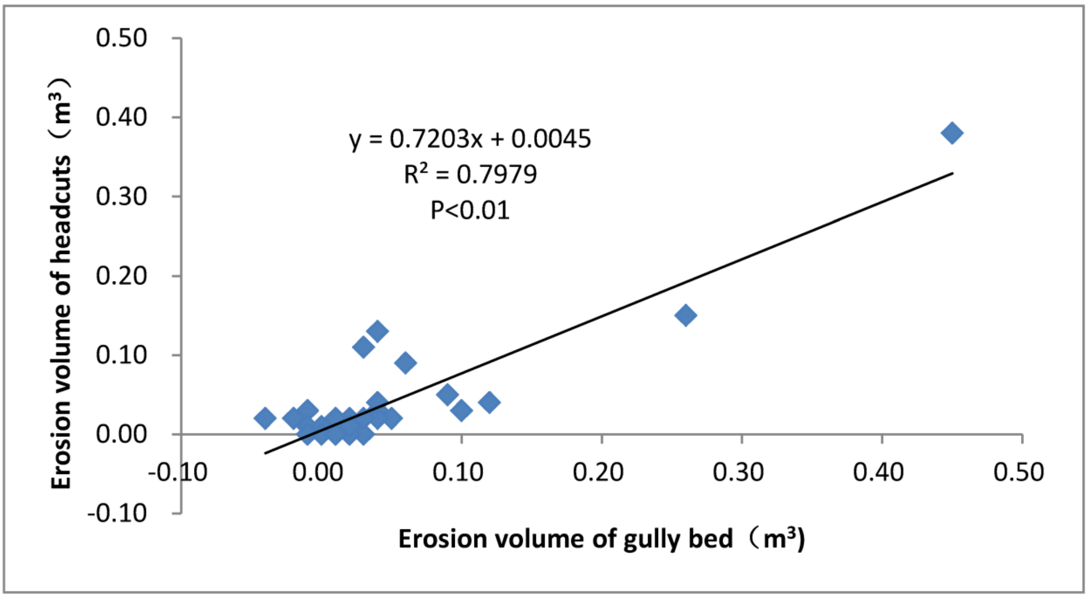

Fig 2 Relationships between erosion volumes of gully headcut and beds

4. CONCLUSION

The headcut retreat rates were strongly correlated with the gully bed erosion volumes, which indicated the erosion volumes of gully beds close to headcut (0-4 m) could be used to predict headcut retreat when plunge pools were not developed significantly. The differences of headcut retreat rates were mainly attributed to the variation of soil erodibilities in gully beds.

5. ACKNOWLEDGEMENT

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5.9.O

AGRICULTURAL, RUNOFF, EROSION AND SALINITY (ARES) DATABASE TO BETTER EVALUATE RANGELAND STATE AND SUSTAINABILITY

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ABSTRACT

Rangelands comprise approximately 40% of the earth's surface and are the largest land cover type in the world. Degradation from mismanagement, desertification, and drought impact more than 50% of rangelands across the globe. The USDA Agricultural Research Service (ARS) has been evaluating means of reversing degradations and increasing rangelands sustainability for decades. The ARS has recently develop a relational database, the **A**gricultural **R**unoff **E**rosion, and **S**alinity – ARES, to store information from rainfall/runoff/soil experiments across the United States. The ARES database contains more than 100 rangeland plant communities and over 2,000 experiments designed to quantify the interactions between vegetation, soils, and slope on runoff and soil erosion processes. This database is being used to understand impacts of land management practices on hydrologic processes and rangeland sustainability. The ARES database is the foundation for developing a quantitative means of estimating the risk of sustainability of the site. The database can be used to identify ecological sites in areas that have limited information and then use the information contained within the database to estimate sustainability. If this site is at risk the database and tools such as the Rangeland Hydrology and Erosion Model can be used to evaluate alternative management practices and the likelihood of reversing the degradation.

INTRODUCTION

Rangelands comprise over 40% of the total landmass in the United States, with nearly 80% in the western states. The estimated annual costs of damage caused by soil erosion and excessive sediment in surface waters within the U.S. is approximately $6 billion to $16 billion annually for water users (Osterkamp et al., 1989, and Lal, 1994). Over 55% of sediment and salts entering the Colorado River are derived from accelerated soil erosion from federally owned rangelands with damages estimated to be $385 million per year to water users (Kenney et al., 2009). Historically, information on the types, patterns, causes, spatial location, severity, and extent of land degradation through soil erosion at global and national scales have not been available in sufficient detail for developing cost-effective conservation policies. ARS and its partners (NRCS, BLM, USFS and BOR) have implemented large scale experiments to evaluate rainfall, runoff, soil loss, and water quality on rangelands for the last forty years across the west which began with a rotating boom rainfall simulator (Swanson 1965; Figure 1), and for the last ten years using a new fixed boom rainfall simulator (Paige et al. 2003) with standardized sampling protocols

(Simanton et al. 1991). These datasets have not been archived and are vulnerable to being lost due to the majority of scientists involved having retired.


Figure 1. Swanson rotating boom simulator (left) and Walnut Gulch rainfall simulator (right) used to collect data for the Agricultural, Runoff, Erosion and Salinity (ARES) Database.

MATERIALS & METHODS

ARS has developed a relational database to store ongoing research data; as well as, recovered historical datasets, to make them available to the national and international scientific community. We have identified 23 sites and over 1,500 plots and runs that can be added to an existing WEPP-IRWET rainfall/runoff database developed to support the Rangeland Hydrology and Erosion Model (RHEM) along with other tools developed by ARS. Originally, the RHEM model was developed

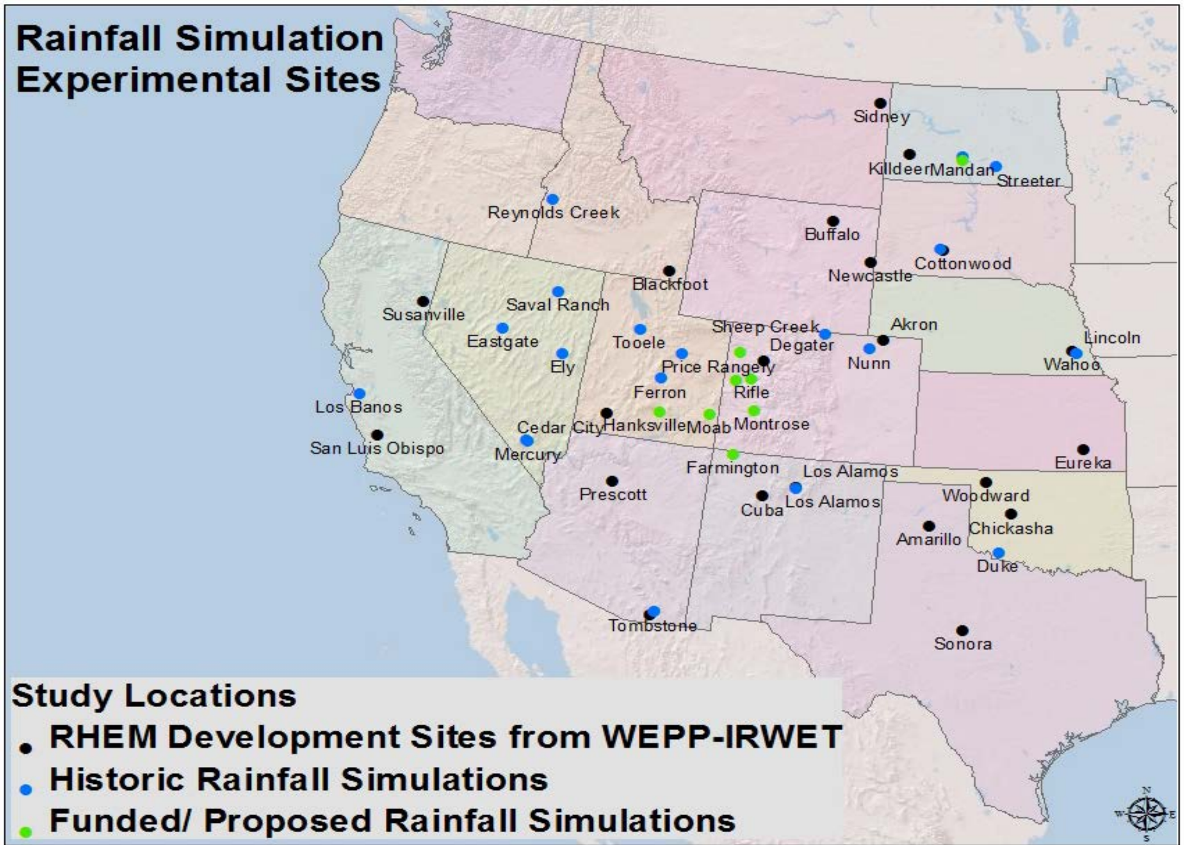

Figure 2. Map of historical, current and proposed future research that will populate the Agricultural, Runoff, Erosion and Salinity (ARES) database.

from 204 plots at 49 rangeland sites (Figure 2) (Nearing et al., 2011). The data recovered from this effort will expand the existing data available fivefold for use in developing runoff, soil loss and water quality models for use in natural resource and rangeland management. The data can be used to determine sustainability of rangelands and the hydrologic impact of changing from a State within an Ecological Site to another State (Williams et al., 2016). We have collaborated with scientists and retrieved data in various formats including: field datasheets, floppies in obsolete formats (i.e. Quattro Pro spreadsheets), Access database, Excel spreadsheets, SAS database, publications, etc. We have identified an additional 15 sites and approximately 250 plot/runs and are seeking collaborating arrangements with these USDA scientists to recover this data. In addition, we have identified numerous experiments and sites that the Department of Energy evaluated at the Pacific Northwest National Laboratory in Washington on sagebrush steppe rangelands we will be adding soon.

RESULTS & DISCUSSION

Scientists, and a team of IT Specialists from National Agricultural Library (NAL) are developing the ARES database using an SQL Server with sites cross-referenced to NRCS soil series and Ecological Site databases. The database will be hosted on a permanent NAL server for public access which will classify and display plant community types using Omernik Level IV ecoregions, MLRA, watershed boundaries, and more. Photos and site descriptions of each location will be available for users to make comparisons with their own sites. Datasets will be cross-referenced and hot-linked to publications retrievable through the National Agricultural Library. That data has been used to develop a training tutorial for the RHEM model and factsheets to assist in developing hydrologic sections of Ecological Sites. The database contains the dominant rangeland vegetation types for the western United

States (Table 1). Soils in the database cover all 12 texture types: Sand, Loamy sand, Sandy loam, Loam, Silt loam, Silt, Sandy clay loam, Clay loam, Silty clay loam, sandy clay, Silty clay, and Clay. Slopes range from 3% to 38%.

Table 1. Vegetation type and dominant plant(s) at the site contained with the Agricultural Runoff, Erosion, and Salinity database.

Vegetation Type	**Common Name**	**Scientific Name**
Desert Shrubland	Creosote brush	Larrea tridenta
Sagebrush Steppe	Sagebrush	Artemisia spp.
Salt Desert Shrub	saltbush	Atriplex spp.
Shortgrass	Buffalo grass	Buchole dactylodies
Mixed Grass	Gramma grass	Bouteloua spp.
Tall Grass	Indiangrass/Big Bluestem	Sorghastrum nutans and Androgogon gerardii
Annual Grassland	Cheatgrass	Broums tectorum
Desert grassland	Tobosagrass	Pleurapjis mutica
California Annual Grassland	Wild oats	Avena spp.
Invaded mixed grass	Kentucky bluegrass	Poa pratensis
Pinyon-Juniper Woodland	Pinyon-Juniper	Pinus edulis and Juniperous osteosperma
Clubmoss invaded mixed grass	Clubmoss	Huperzia spp.

CONCLUSIONS & IMPLICATIONS

The ARES Database (100+ plant communities and 2,000+ plots/runs) will be utilized to validate and expand the utility of Rangeland Hydrology and Erosion Model (RHEM) for plant communities not currently addressed (i.e., meadows, blackbrush, etc.). Data outputs applied to RHEM will assist in developing standardized hydrologic sections for USDA Natural Resources Conservation Service rangeland Ecological Site Descriptions used to describe optimum vegetation cover for reducing soil erosion and improving water quality. The database will also be employed to develop new equations to estimate total dissolved solids in runoff water, and be provided as an additional resource for natural resource management programs. Overall, the ARES database will standardize methods of archiving past, present, and future experiments while offering a high degree of automated output that will be available to run process-based models such as RHEM.

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5.10.O

NEW TOOLS TO ESTIMATE RUNOFF, SOIL EROSION, AND SUSTAINABILITY OF RANGELAND PLANT COMMUNITIES

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ABSTRACT

Rangelands are the largest land cover type in the world. Degradation from mismanagement, desertification, and drought impact more than 50% of rangelands across the globe. The USDA Agricultural Research Service has been evaluating sustainability of rangeland for over 40-years by conducted rangeland rainfall simulation experiments (>100 plant communities) across the U.S.A. The data from these experiments has been assembled into a relational database (**A**gricultural **R**unoff **E**rosion, and **S**alinity - ARES). This database is being used to understand impacts of land management practices on hydrologic processes and rangeland sustainability. Results from analyzing the data in central Colorado, U.S.A. indicate that long-term heavy grazing does increase runoff rates and volumes. The database was also used to evaluate salt transport and soil erosion processes in central Utah, U.S.A. with the Rangeland Hydrology and Erosion Model (RHEM). RHEM did an excellent job in predicting runoff at the Utah, U.S.A. sites (R^2 0.90) over all rainfall intensities applied. The model did a good job of estimating sediment yield (R^2 0.58) for saline and sodic sites. There was a very strong correlation (R^2 0.84) between observed sediment yield and observed total dissolved solids. The ARES database and REHEM decision support tool can be used to quantify sustainability of rangelands in relation to land management practices impacts on rangeland hydrologic and soil erosion processes and water quality.

INTRODUCTION

Rangelands are the largest land cover type in the world. Degradation from mismanagement, desertification, and drought impact more than 50% of rangelands in Asia and 70% of African and Middle Eastern rangelands. Approximately 18% of non-Federal U.S. rangelands would benefit from conservation to reduce accelerated soil loss (Weltz et al. 2014). In the U.S. estimated annual costs of damage caused by soil erosion and excessive sediment is approximately $16 billion annually to water users (Osterkamp et al., 1989). However, financial impacts from degraded rangelands are likely significantly higher when other ecosystem services are included. Historically, information on the types, patterns, causes, spatial location, severity, and extent of land degradation through soil erosion at global or national scales have not been available in sufficient detail for developing specific policies for targeting conservation in a cost-effective approach. USDA Agricultural Research Service has implemented large scale experiments to evaluate rainfall/runoff/soil loss/water quality on rangelands for the last forty years using a rotating boom rainfall simulator (Swanson 1965) or a fixed boom Walnut Gulch rainfall simulator (Paige et al. 2003) with standardized sampling protocols to estimate rangeland sustainability. These data have not been archived and are vulnerable to being lost as the majority of scientists who conducted these experiments have retired. Cost to reproduce this data would exceed $30 million and take ten years to resample if possible. The **A**gricultural **R**unoff **E**rosion, and **S**alinity (ARES) was developed to support the need to secure and make accessible this historical data (Nesbit et al., 2016). This database can be used to develop recommendations on methods to reduce upland soil erosion and increase sustainability of rangelands when the information is used to parameterize the Rangeland Hydrology and Erosion Model – RHEM (Nearing et al., 2011 and Williams et al., 2016). Specifically, there is a need to improve the understanding of sources and transport mechanisms of dissolved solids in rangelands and develop methods to estimate soil loss processes from different management practices on rangelands across the globe to ensure sustainable food supplies from agricultural enterprises.

MATERIALS & METHODS

The ARES database is publicly available through the USDA National Agricultural Library and contains information from 100 sites and over 2,000 plots/runs. The database contains information on soils, plant community, foliar and ground cover, rainfall intensity, runoff, soil loss, and water quality (on selected sites). Data from 1 site in Colorado, U.S.A. and 2 sites in Utah, U.S.A. were utilized to demonstrate database utility for understand rangeland hydrologic processes. A Swanson rotating boom rainfall simulator was used for measuring runoff within an existing variable grazing intensity study at Central Plains Experimental Range (CPER), Nunn, Colorado, U.S.A. Three consecutive years of rangeland rainfall simulation measurements were conducted at CPER to better understand dynamics of annual vegetation conditions and interactions of grazing and plant productivity on runoff and infiltration. CPER had three treatments of light, moderate and heavy grazing. Rainfall simulation was applied on 3 x 10 m plots at controlled intensities of 60 and 110 mm/hr. Rainfall simulations were replicated 4 times for each grazing intensity for a total of 12 plots. Three soil moisture conditions were assessed during each simulation: dry soil at 60 mm/hr; 30 minutes later a wet soil run of 60 mm/hr; followed by a very wet run of 110 mm/hr. Rainfall intensities were ~100 and ~1000 year average recurrence intervals. Simulations were conducted within 3-ha enclosures which excluded cattle from each plot for the duration of this study to evaluate recover processes from past grazing intensities. Rainfall simulation was used to quantify the hydrologic, erosion, and salt mobilization and transport processes on 2 sites in central Utah, U.S.A. A Walnut Gulch rainfall simulator was used on 2 x 6 m plots where a single rainfall intensity was applied to each plot as either a 2yr (44 mm/hr), 10yr (80 mm/hr), 25yr (104 mm/hr), or 50yr (136 mm/hr) rainfall return rate on dry soil for approximately 45 min. At each site 3 replications for each rainfall intensities was sampled for a total of 12 plots. For both studies runoff and sediment samples were collected every 2 min. in 1 l bottles, dried and weighed. Canopy and ground cover were measured with line-point intercept. Results from Utah study were used to validate RHEM. RHEM was designed to require minimal input that is readily available for most rangeland ecological sites: soil texture; slope; plant life-form; canopy and ground cover; and precipitation. RHEM estimates runoff, soil erosion, and sediment delivery rates and volumes at the spatial scale of the hillslope and the temporal scale of a single rainfall event.

RESULTS & DISCUSSION

At CPER the differing effects of grazing treatments on runoff were most evident during the very wet (110 mm/hr) rainfall rate (Figure 1). The time difference between simulator runs ('92 vs'94) were most evident on the heavy grazing treatment with very little change over time on the light and moderate grazing treatments. Runoff decreased ~50% from 1992 to 1994 during the dry run heavy grazing treatment. A governing principle of land management is that changes in plant cover and species result in changes in watershed condition, hydrologic response, and sustainability. Grazing management practices influence runoff and soil erosion on rangelands because they affect plant distribution and density, biological diversity, canopy and ground cover, and soil properties.

The second study was conducted in the Price River Basin in central Utah, U.S.A. on 2 different ecological sites using the Walnut Gulch Rainfall simulator. Soils were derived from Mancos shale and are naturally saline, sodic and highly erosive. The RHEM model did an excellent job in predicting runoff at the 2 Utah sites (R^2 0.90) over all rainfall intensities applied (Figure 2). RHEM predicted sediment yield (R^2 0.58) reasonably well with no significant bias in the predicted sediment yield. For saline and sodic sites, such as these, the soils are highly dispersive and the RHEM model slightly under predicted sediment yield. New parameterization equations designed specifically for saline and sodic soils should improve sediment yield predictions. There was a very strong correlation (R^2 0.84) between observed sediment yield and observed total dissolved solids (Figure 3) in the runoff at these 2 sites in Utah, U.S.A.

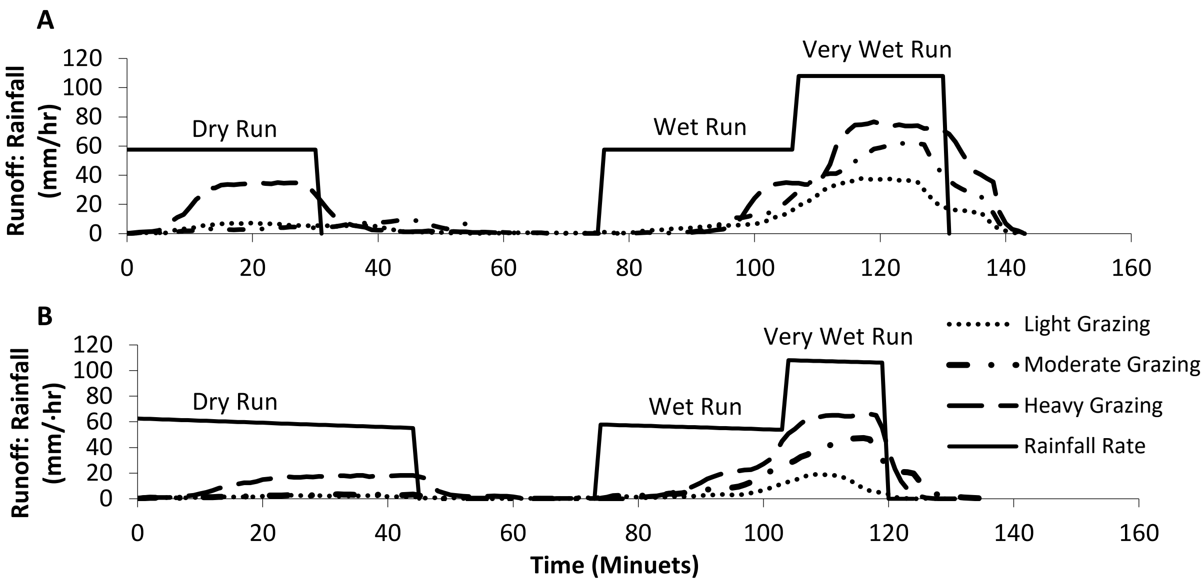

Figure 1. Mean runoff rate (mm/hr) by grazing intensity at Central Plains Experimental Range, Nunn, Colorado, U.S.A. in 1992 (A) and 1994 (B).

CONCLUSIONS

Vegetation is the primary factor controllable by human activity that influences the spatial and temporal variability of surface runoff and soil erosion on rangelands. RHEM can be used to address the best methods to obtain sustainable ecosystem services and food security for livestock based food production systems. RHEM is the easiest and most accurate soil erosion tool for use on rangelands. RHEM's simple, available inputs and web-enabled interface make it the choice for international rangeland management as countries combat desertification, drought, and land degradation through soil erosion processes on rangelands. Further work will involve additional rainfall simulations on saline and sodic rangelands to determine if these results are consistent and when or if soil erodibility parameters need to be altered. Updates to RHEM will include options to predict total dissolved solids in runoff on saline soils; and impacts of prescribed burns, wildfires and invasive plant species on runoff and sediment yield. Hydrologic response (runoff rate and sediment yield) at the hillslope scale can be used quantify and predict salt mobility and transport as a function of rainfall intensity to estimate total dissolved solids in surface runoff on rangelands. Runoff and sediment yield can be used to assess the sustainability of rangelands when site specific results are compared to baseline conditions of runoff and sediment yield for undisturbed sites. This allows land managers to quantify the benefits of conservation, prioritize areas where conservation would be beneficial, and cost-effectively increase the sustainability of rangelands.

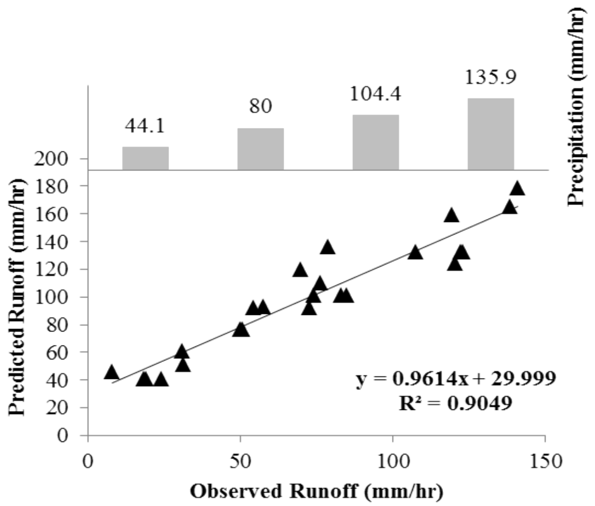

Figure 2. Observed vs. RHEM predicted runoff mm/hr for saline soils in Utah, U.S.A.

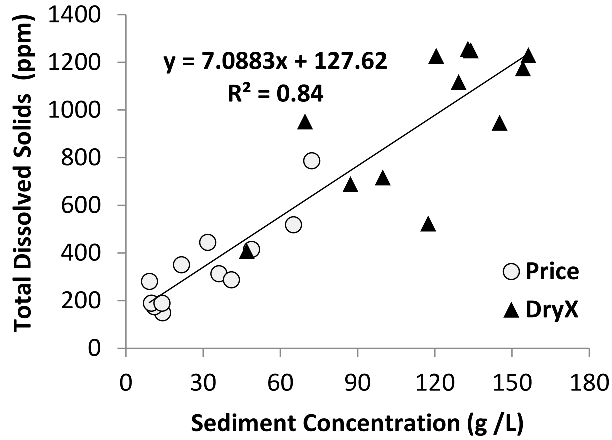

Figure 3. Sediment concentration vs. total dissolved solids (ppm) for saline soils Utah, U.S.A.

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5.11.O

ADVANCES IN MODELING SOIL EROSION AFTER DISTURBANCE ON RANGELANDS

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INTRODUCTION

Disturbance on rangeland such as fire or woody plant encroachment can alter the site ecological characteristics and hydrological behavior which, in turn, can lead to increased soil loss. Many rangeland conservation practices are aimed at reducing soil loss associated with disturbances. Research has been undertaken to develop process based models, such as the Rangeland Hydrology and Erosion Model (RHEM; Nearing et al., 2011; Al-Hamdan et al., 2015), that predict soil erosion rate after disturbance on rangelands. In these models soil detachment is predicted as a combination of multiple erosion processes, rain splash and thin sheet flow (splash and sheet) detachment and concentrated flow detachment. In RHEM, Splash and sheet detachment is calculated by (Wei et al., 2009):

$$D_{SS} = K_{SS}(I^{1.052}\, q^{0.592}) \qquad [1]$$

where D_{SS} is the splash and sheet detachment rate (kg s^{-1} m^{-2}), Kss is the splash and sheet erodibility, I is the rainfall intensity (m s^{-1}), and q is the discharge per unit width (m^2 s^{-1}). Concentrated flow detachment is a function of concentrated flow detachment capacity (Dc, kg s^{-1} m^{-2}) which is calculated by (Al-Hamdan et al., 2012):

$$Dc = K_{\omega}(\omega) \qquad [2]$$

where K_{ω} is concentrated flow erodibility (s^2 m^{-2}) and ω is the stream power (kg s^{-3}). Knowing soil erodibility associated with each process is a key factor for achieving satisfied performance of such models. In this paper, we present advancement in developing equations to predict erodibility on disturbed rangeland where two different parametrization approaches can be used, which accommodate a wide range of vegetation, ground cover, soil texture, and disturbance conditions.

METHODS

The approaches for erodibility estimation on degraded sites were conceptualized based on observations and results of experimental studies on rangelands. In the first approach, where concentrated flow detachment is negligible, only the splash and sheet erodibility (Kss) parameter is needed to predict erosion due to disturbance. In this approach (Kss approach), it is assumed that sediment detachment is dominated by rain splash and thin sheet flow (splash-sheet), while the major role of concentrated flow paths is transporting the splash-sheet detached sediments. The approach uses empirical equations that were developed by applying piecewise regression analysis to predict the differences of splash and sheet erodibility before and after disturbance using two continuous relationships and across a wide range of soil textures as a function of vegetation cover and surface slope (Al-Hamdan et al., 2017):

Bunch Grass:

$$\mathrm{Log}_{10} Kss = \begin{cases} 4.154 - 2.547 * G - 0.7822 * F + 2.5535 * S & \text{if } G \leq 0.475 \\ 3.1726975 - 0.4811 * G - 0.7822 * F + 2.5535 * S & \text{if } G > 0.475 \end{cases} \quad [3]$$

Sod Grass:

$$\mathrm{Log}_{10} Kss = \begin{cases} 4.2169 - 2.547 * G - 0.7822 * F + 2.5535 * S & \text{if } G \leq 0.475 \\ 3.2355975 - 0.4811 * G - 0.7822 * F + 2.5535 * S & \text{if } G > 0.475 \end{cases} \quad [4]$$

Shrub:

$$\mathrm{Log}_{10} Kss = \begin{cases} 4.2587 - 2.547 * G - 0.7822 * F + 2.5535 * S & \text{if } G \leq 0.475 \\ 3.2773975 - 0.4811 * G - 0.7822 * F + 2.5535 * S & \text{if } G > 0.475 \end{cases} \quad [5]$$

Forbs:

$$\mathrm{Log}_{10} Kss = \begin{cases} 4.1106 - 2.547 * G - 0.7822 * F + 2.5535 * S & \text{if } G \leq 0.475 \\ 3.1292975 - 0.4811 * G - 0.7822 * F + 2.5535 * S & \text{if } G > 0.475 \end{cases} \quad [6]$$

In this approach, concentrated flow erodibility (Kw) is set as the insignificant value that is typical for undisturbed rangeland (i.e., 7.747×10^{-6} s^2 m^{-2}).

In the second approach (dual erodibility approach), concentrated flow erodibility is also calculated to predict soil erosion due to disturbance. This approach is used in the case of abrupt disturbance that exposes loose sediments on a steep slope. The approach uses empirical equations that were developed by applying regression analysis to predict the differences of concentrated flow erodibility before and after disturbance as a function of vegetation cover and soil texture. It also uses empirical function to predict temporal erodibility variation within a runoff event as a function of cumulative flow, (Al-Hamdan et al., 2012):

$$K_\omega = K_{\omega(max)adj} e^{-5.53 q_c} \quad [7]$$

where q_c is the cumulative unit flow discharge (m^2 s^{-1}) and $K_{\omega(max)adj}$ is the maximum stream power erodibility (s^2 m^{-2}) which can be calculated by:

$$log_{10}\left(K_{\omega(\max)adj}\right) = -3.64 - 1.97(litter + basal + crypto) - 1.85rock - 4.99clay + 6.0silt \quad [8]$$

where *litter, basal, and crypto* are the fraction of area covered by litter, basal, and cryptogam to total area (m^2 m^{-2}), *rock* is the fraction of area covered by rock to the total area (m^2 m^{-2}), and *clay* and *silt* are in fraction.

To combine the two approaches into one, the following equation can be used to calculate detachment from concentrated flow:

$$D_c = P\left[\left(K_{\omega(max)} e^{-5.53 q_c}\right)\right](\omega) + (1-P) * K_\omega(\omega) \quad [9]$$

where P is the probability of overland flow to become a significantly high eroding concentrated flow. Equation 9 is like the first parameterization approach when P equals zero, and Kω is set as 7.747×10^{-6} s^2 m^{-2}. The probability of overland flow to concentrate can be calculated with the following equation (Al-Hamdan et al. 2013):

$$P = \frac{\exp(-6.397 + 8.335S + 3.252bare + 3440q)}{1 + \exp(-6.397 + 8.335S + 3.252bare + 3440q)} \quad [10]$$

where S is slope gradient (expressed as a fraction), *bare* is the fraction of bare ground, and q is the flow discharge per unit width ($m^2\ s^{-1}$).
Applicability of the two erodibility parameterization approaches was evaluated using the test of percent bias (PBIAS; Gupta et al., 1999). To test the performance of the two erodibility approaches, infiltration parameters in RHEM were optimized on total volume of runoff for each plot. The data used for evaluating the erodibility parameter estimation approaches were obtained from rainfall simulation databases maintained by the USDA-Agricultural Research Service. The data span undisturbed and disturbed conditions such as wild fire, prescribed fire, and/or tree encroachment.

RESULTS

In general, model simulations overestimated low sediment yield values and underestimated high sediment yield values. The bias at the two ends of the erosion rates is typical for all erosion models because of the limitations in representing the random components in measured data within replicates (Nearing, 1998). However, the simulations were still able to match more than 50% of the measured sediment yield at highly disturbed sites. The overall performance of the Kss approach when using weighted averaging between Equations 3 through 6 based on the percentage of life form presented in each plot was satisfactory with a PBIAS of 54.9 (Figure 1). Applying the dual erodibility approach improved the overall performance of the model with lower absolute value of PBIAS (Figure 2). The dual parameterization approach enhancement is more evident in the burned sites.

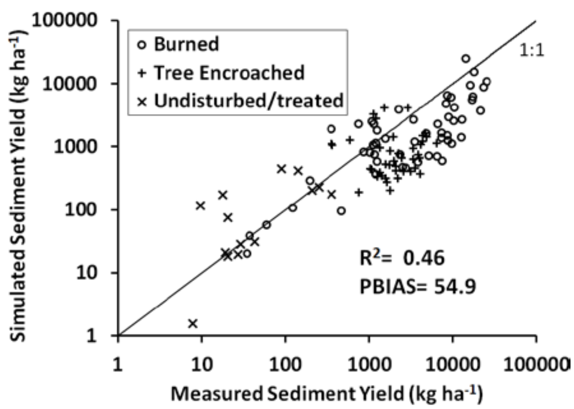

Figure (1) Measured sediment yield vs. sediment yield estimated by RHEM when using weighted averaging between Equations 3 through 6 based on the percentage of life form to estimate *Kss* while assuming no significant eroding concentrated flow.

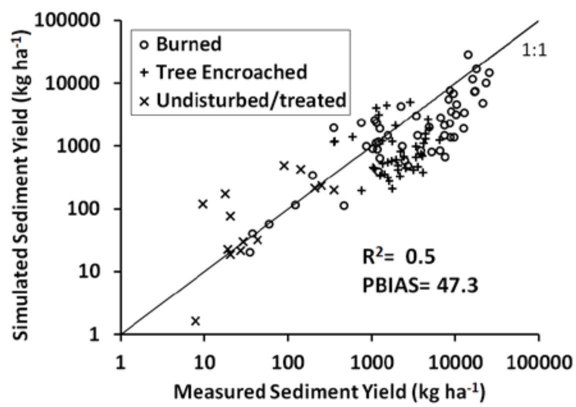

Figure (2) Measured sediment yield vs. sediment yield estimated by RHEM when using weighted averaging between Equations 3 through 6 based on the percentage of life form to estimate *Kss* while concentrated flow detachment rate calculated using Equation 9.

CONCLUSIONS

Results show that one erodibility parameter approach (Kss) is effective in most cases for predicting erosion, which minimize the error that can be generated from the parameterization process. The approach uses empirical equations that were developed by applying piecewise regression analysis. The breakpoint generated by the piecewise regression identifies a threshold at which there is a substantial change in the rate of erodibility increase with respect to bare soil area, and therefore provides an objective means for detecting changes between natural and disturbance phases. The dual erodibility approach is needed in the cases of abrupt disturbance that exposes loose sediments on a steep slope (e.g., immediately post-fire). The approach uses empirical equations to predict the differences of concentrated flow erodibility before and after disturbance and to predict temporal erodibility variation within a runoff event as a function of cumulative flow. The equations for estimating erodibility in the two approaches use readily available data for estimating erodibility values. The two parameterization approaches expand the applicability of RHEM to a greater scope of landscapes, soil textures, and disturbance conditions.

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5.12.O

IMPROVED UNDERSTANDING OF HYDROLOGY AND EROSION PROCESSES AND ENHANCED APPLICATION OF THE RANGELAND HYDROLOGY AND EROSION MODEL (RHEM) FOR DISTURBED RANGELANDS

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INTRODUCTION

Land managers across arid to semiarid regions around the world are challenged with addressing broad-scale ecohydrologic changes caused by plant community transitions and altered disturbance regimes. For example, the sagebrush (*Artemisi*a spp.) ecosystem occupying nearly 40% of the Great Basin Region (~380,000 km^2), USA, is considered one of the most imperiled ecosystems in the United States due to woody plant encroachment and introduced annual weeds. Encroaching conifers on these rangelands outcompete shrub and herbaceous vegetation and facilitate development of extensive patches of bare ground (Williams et al., 2014a). The increase in bare area promotes connectivity of runoff and erosion processes and results in elevated runoff rates and soil loss (Williams et al., 2014a, 2016a, 2016b).
Invasion of sagebrush rangelands by the annual grass cheatgrass (*Bromus tectorum* L.) has increased the size and frequency (by more than 10-fold) of wildfires on these landscapes, making these rangelands more vulnerable to high rates of runoff and soil loss (Pierson et al., 2011; Williams et al., 2014b). Other regions around the world are undergoing similar changes in vegetation, disturbance regimes, and hydrology and erosion rates (Williams et al., 2014b).

Land managers need enhanced process-based knowledge and predictive tools to evaluate and target conservation and restoration practices that address these broad-scale ecological changes. This study highlights findings from decades of field hydrology and erosion experiments and recent enhancements to the Rangeland Hydrology and Erosion Model (RHEM; Nearing et al., 2011). The field experiments, conducted by USDA scientists, provide an enhanced process-based knowledge to address broad-scale changes in vegetation, disturbance regimes, and runoff and erosion processes on arid to semiarid rangelands. The experimental data further provide a basis for enhancing and evaluating the RHEM tool for predicting rangeland hydrologic and erosion responses to disturbance and management practices.

METHODS

A suite of rainfall simulation and overland flow experiments were conducted on multiple rangeland sites across a range of undisturbed and disturbed vegetation and soil conditions (Table 1, Pierson et al., 2010;

2013, Al-Hamdan et al., 2015, Williams et al., 2016a, Al-Hamdan et al., 2017). The field experiments were conducted to improve understanding of infiltration, runoff, and erosion processes across a diversity of vegetation, soils and topography and to provide a dataset for developing and testing hydrologic and erosion parameter estimation equations for the RHEM tool (Al-Hamdan et al., 2015, 2017). Rainfall simulations were run at the small (0.5 m^2) and large (13 to 32.5 m^2) plot scales. Rainfall intensities ranged from near 60 mm h^{-1} to more than 100 mm h^{-1} and the duration of rainfall ranged from 45 min to 60 min. Small plot simulations were used to quantify runoff and erosion from rainsplash and sheet flow processes and large plot simulations were used to quantify runoff and erosion from rainsplash, sheet flow, and concentrated flow processes. Overland flow experiments (8 m^2) using computer controlled flow regulators were employed to isolate and quantify runoff and erosion processes by concentrated overland flow (Pierson et al., 2010). Canopy and ground cover and soil properties were characterized for all experimental plots using methods described by Pierson et al. (2010). Plot data were analyzed using regression approaches to develop parameter estimation equations for predicting the probability of runoff to concentrate, various overland flow hydraulic variables, and soil erodibility (Al-Hamdan et al., 2015). The parameter estimation equations were applied in the RHEM tool to enhance its predictive capability across diverse rangeland conditions (Al-Hamdan et al., 2015, 2017; Williams et al., 2016b).

Table 1. Numbers of small (0.5 m^2), rill (8 m^2) and large plots (13-32.5 m^2) and the number of simulation runs.

	Field Plots	Simulations
Small Plots	994	1590
Large Plots	194	340
Rills	710	2552
Total	1898	4482

RESULTS

Our field studies across diverse rangeland sites indicate that the amount of vegetation, bare ground, the connectivity of processes, and sediment availability are primary controls for site hydrologic vulnerability to runoff generation and sediment delivery (Figure 1, Pierson et al., 2011, Williams et al., 2014b). Data across numerous sites suggest runoff generation, runoff velocity, and erosion increase at an increasing rate where bare ground exceeds 50-60% (Pierson

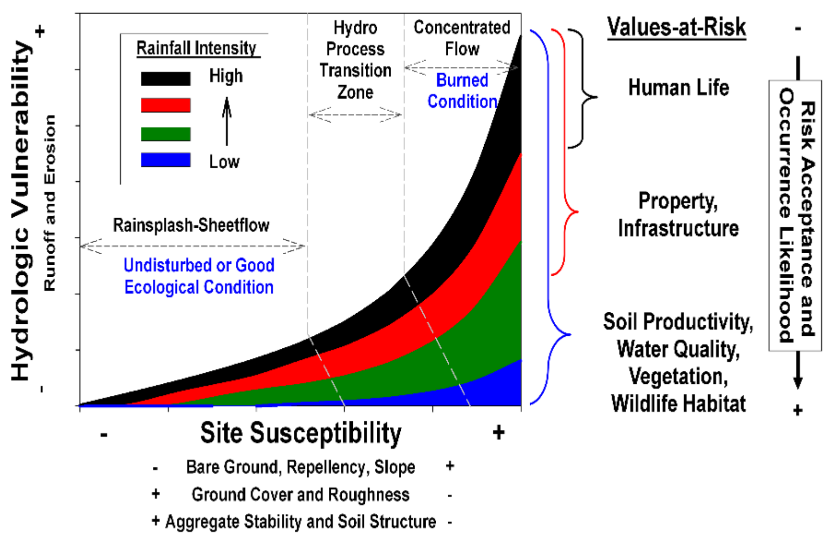

Figure 1. Model of hydrologic vulnerability (y-axis), as function of site conditions (site susceptibility, x-axis), and the potential values-at-risk impacted by varying magnitude of runoff and erosion responses.

et al., 2010, 2013, Williams et al., 2014b). For undisturbed rangelands, bare ground is restricted to isolated patches. Runoff and sediment leaving these patches travels a limited distance downslope before being captured in or behind ground cover (Williams et al., 2016a, 2016b). Runoff generation and sediment detachment and delivery processes are poorly connected under these conditions. Following disturbance, bare ground and runoff and erosion processes become well connected and runoff travels downslope at a higher velocity and with greater sediment detachment and transport energy (Pierson et al., 2011, Williams et al., 2014a, 2014b, 2016b). This connectivity of processes initiates as bare ground approaches 50% (Pierson et al., 2010, 2013). Erosion rates are highest where processes are well connected and ample sediment is available (e.g., immediately after fire). The magnitude of the overall erosion response is governed by process connectivity, sediment availability, and the amount of water input or runoff discharge and may be amplified with increasing slope steepness (Al-Hamdan et al., 2015, Williams et al., 2014b).

Regression analyses of the above noted field data have supported our field observations of the drivers of runoff and erosion processes on rangeland ecosystems. Our analysis of data pooled across our extensive dataset clearly demonstrate that ground cover and soil texture are primary predictors of soil erodibility on rangelands (Al-Hamdan et al., 2017). Further, our analyses demonstrate that the velocity and hydraulics of overland flow and the likelihood for flow to become concentrated are strongly regulated by the percentage of bare ground, hillslope angle, and the amount of runoff available (Al-Hamdan et al., 2015).

RANGELAND HYDROLGY AND EROSION MODEL

Collectively, the field data and regression analyses have resulted in new representative equations and parameter estimation procedures for runoff hydraulics, flow concentration, and soil erodibility in the RHEM tool specific for rangeland applications (Al-Hamdan et al., 2015, 2017). These enhancements have made RHEM an excellent tool for predicting landscape ecohydrologic and erosion responses to disturbances (Al-Hamdan et al., 2015), such as plant community transitions and wildfire, and for targeting and predicting potential hydrologic and erosion responses to management across diverse rangeland ecosystems (Williams et al., 2016b). RHEM predicts runoff and associated soil loss using a risk assessment approach based on erosion event return intervals (Figure 2).

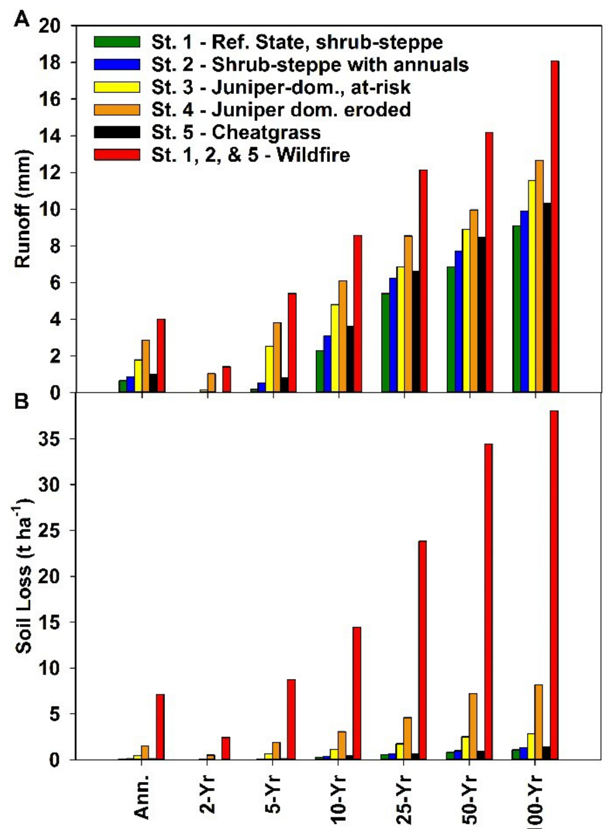

Figure 2. RHEM predicted runoff and soil loss for various ecological states on Great Basin, USA, rangelands and for conditions after wildfire.

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5.13.O

PROCESS-BASED MODELING OF UPLAND EROSION AND SALT LOAD IN THE UPPER COLORADO RIVER BASIN

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INTRODUCTION

Hillslope runoff and soil erosion processes are indicators of sustainability in rangeland ecosystem due to their control on resource mobility. Hillslope processes are dominant contributors to sediment delivery on semi-arid rangeland watersheds (Nichols, et al., 2013). The influence of vegetation on hillslope runoff and sediment production forms the basis of current rangeland hydrology and erosion modeling (Nearing, et al., 2011). Soil erodibility parameters are functions of many intrinsic (e.g. soil texture, clay mineralogy, etc.) and extrinsic (e.g., management, soil amendments, subsurface hydrology, etc.) factors. In particular, soil salinity and sodicity have been shown to significantly affect soil structural stability (Abusharar, et al., 1987, Goldberg, et al., 1988, Ruiz-Vera and Wu, 2006) with implication in susceptibility to erosion. While the fundamental understanding of relationship between salinity / sodicity and aggregate stability has been well established in laboratory settings, this information has seldom been incorporated into hydrologic and erosion models. The Rangeland Hydrology and Erosion Model (RHEM) for example has been developed from an extensive experimental data collected on four dominant rangeland community types (i.e. shrub-dominated, sod grass, annual grasses and forbs and bunch grasses). Soils at most of the original experimental sites used in the development of RHEM were undisturbed and with no appreciable influence of soil salinity and sodicity on soil erodibility and infiltrability. This is reflected in hydraulic parameterization equations used in RHEM showing no dependence on soil chemistry (Al-Hamdan, et al., 2015, Nearing, et al., 2011, USDA-ARS, 2014). Nevertheless,Simanton, et al. (1991) found erosion rates of saline soils developed in the Mancos Shale of the Colorado Plateau to be outliers compared to other non-saline soils. Higher erosion rates were attributed to a greater susceptibility to rill development at this site. In this paper, we develop parameterization equations for RHEM to predict soil erosion and salt load on saline rangelands of the Upper Colorado River Basin (UCRB).

METHODS

Rainfall simulation experiments were conducted at six saline sites in the Upper Colorado River Basin (UCRB). Soils at these sites varied in classification and properties but were all derived from parent material of marine geology (Mancos Shales or Eagle Valley Evaporites). Published maximum Sodium Adsorption Ratio (SAR) ranged from 3 to 13 (USDA-NRCS Web Soil Survey, 2017). Vegetation at these sites was shrub-dominated with varying degrees of grass cover.

On each experimental site, a series of rainfall simulations were conducted on 6 m x 2 m erosion plots to quantify sediment and salt transport processes during rainfall-driven erosion processes. Rainfall was applied with a Walnut Gulch Rainfall Simulator (Paige, et al., 2004). During each simulated event, soil loss, runoff and solute transport were measured under four rainfall intensities corresponding to return periods of 2 (44.1 mm/hr), 10 (80 mm/hr), 25 (104.4 mm/hr) and 50 (135.9 mm/hr) years. Intensities were calculated based on the 5-minute depth return frequencies published in the National Oceanic and Atmospheric Administration (NOAA) atlas 14 (Bonnin, et al., 2006). Each rainfall intensity at both sites was replicated three times leading to a total of twelve plots per site. Before and after each rainfall event, soil surface microtopography digitized from a series of convergent photographs processed through a Structure from Motion (SfM) software.

The RHEM model was calibrated and used to predict runoff and soil loss on these saline sites. RHEM uses parameter estimation equations to relate soil biophysical conditions to hydraulic and hydrologic parameters. The effective hydraulic conductivity Ke and the sheet and splash erodibility Kss are examples of such parameters linked to soil biophysical conditions through the following relationships:

$$Ke = a\exp\left(b\left(basal + litter\right)\right)$$

$$Kss = 10^{\{c + d \cdot GroundCover + f \cdot FoliarCover + g \cdot Slope\}}$$

where coefficients *a* and *b* vary as a function of soil texture and vegetation community type (i.e. shrub, sod grass, bunch grass and forbs and annual grass) while coefficients *c-g* are functions of vegetation community type and ground cover and basal, litter, GroundCover and FoliarCover are expressed in areal fractions. The concentrated flow erodibility Kω is used to predict the contribution of small rills and channels to total soil erosion. This parameter is typically set to negligible value on undisturbed non-saline sites due to the lower occurrence of concentrated flow erosion in undisturbed rangeland conditions. On disturbed rangelands however, concentrated flow erosion processes are active and contribute to high sediment delivery (Al-Hamdan, et al., 2012). In such disturbed conditions, (Al-Hamdan, et al., 2012) has proposed the following estimation equation for Kω:

$$K\omega = 10^{-4.14 - 1.28 \cdot litter - 0.98 \cdot rock - 15.16 \cdot clay + 7.09 \cdot silt}$$

In addition to these three primary parameters, calibration was also performed on parameter α used in the Smith-Parlange infiltration equation. Furthermore, all rainfall simulation experiments were conducted on soils initially dry therefore with very low saturation ratios. These low saturation ratios were experimentally calculated from soil moisture samples and bulk density measurements. Nevertheless, it was anticipated that variability these initial saturation ratios would have insignificant effects on observed runoff and erosion and therefore a unique saturation ratio was calibrated for the entire dataset to fix the effect of this parameter on predicted runoff and erosion values.

The calibration scheme used in this study assumes that the historically determined relationships between hydrologic and hydraulic parameters and soil biophysical properties on non-saline soils also apply to saline conditions but at a different magnitude. Thus Ke, Kss and Kω on saline sites can be determined by applying multipliers ce, cs and cω to these parameters. Data from three sites (36 plots) were used to calibrate the model to saline conditions and model performance validated against data from all six sites. Scalars ce, cs and cω as well as the other calibration variables were determined with a Monte Carlo scheme implemented in SPOTPY, an optimization tool developed for the Python programing language (Houska, et al., 2015). Model performance metrics used in this paper are the coefficient of determination (R^2) and Nash-Sutcliffe Efficiency (NSE) between observed and predicted cumulative runoff and erosion.

RESULTS AND DISCUSSIONS

The calibration on the 36 plots resulted in R^2 of 0.83 for runoff and 0.46 for sediments while the NSE values were 0.77 and 0.45 respectively. Calibrated values were Saturation ratio = 1.7%, ce = 2.27, cs = 0.69 and cω = 2.48. Figures 1 shows the results of the RHEM calibration and evaluation on the 72 saline plots. Runoff prediction using the traditional non-saline parameter estimation equations yielded R^2 of 0.75 and NSE of 0.61 and these values were improved to 0.82 and 0.81 with the saline equations (Fig. 1a). Soil loss prediction had a lower performance compared to runoff (Fig. 1b). The use of the saline parameter estimation equations resulted in no effect on R^2 values which were unchanged at 0.64 but a degradation of NSE was noted falling from 0.61 with the non-saline equations to 0.47 with the saline equations.

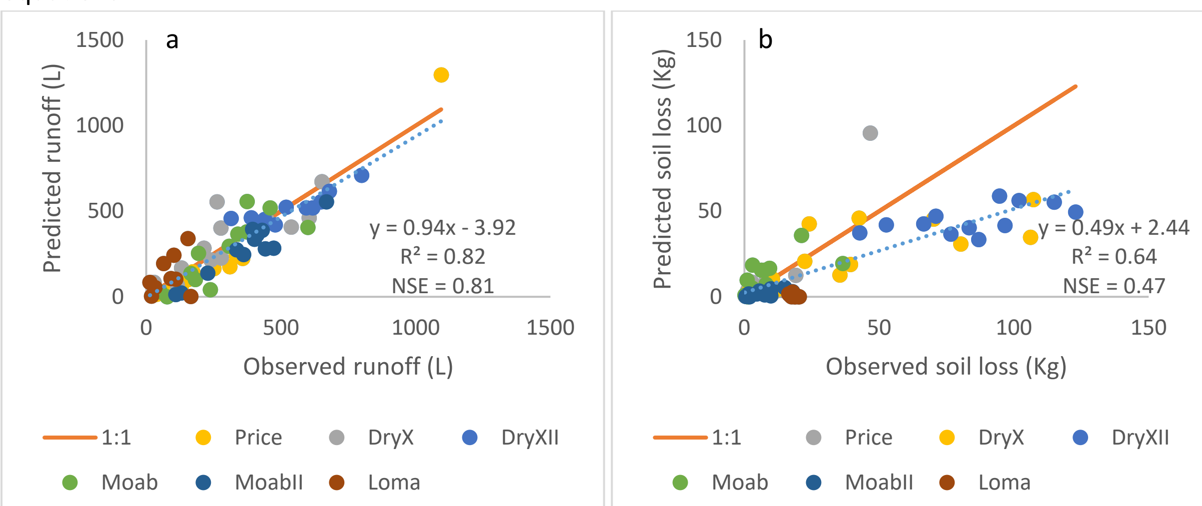

Figure 1 Runoff (a) and soil loss (b) predictions using the parameter-estimation equations developed for saline soils. Dots are colored based on the experimental site they represent.

Two sites (DryX and DryXII) seem to be responsible for a significant portion of the lag in performance on soil loss prediction. Soils at these two sites were mapped as Chipeta-Badland complex and were visibly more erodible and more incised by rills compared to other saline sites. This result suggests that additional soil properties information need to be incorporated in soil erosion prediction on these sites. It is likely that dissolved solids chemistry might explain the erosion behavior of soils at the DryX sites. In such case soil intrinsic properties such as Sodium Adsorption Ratio (SAR), gypsum content, electrical conductivity, etc. could be valuable additional variables to improve soil loss prediction. In addition, a more specific calibration of RHEM for concentrated flow erosion with the help of erosion estimates measured with Structure for Motion (SfM) might help improve soil loss prediction.

CONCLUSIONS

The RHEM model has been successfully calibrated to address soil erosion and salt load on saline rangelands of the UCRB. Performance on runoff were appreciably improved with the use of the parameter estimation equations developed for saline soils while soil loss estimates were predictable with a fair R^2 but could be improved by incorporating additional soil property information and augmenting RHEM concentrated flow prediction with microtopographycally-observed erosion/deposition patterns.

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5.15.O

METHODOLOGY FOR EDAPHOCLIMATIC ASSESSMENT IN THE PROTECTED DESIGNATION OF ORIGIN "WINES OF MADRID"

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INTRODUCTION

Edaphoclimatic assessment is a powerful tool to prevent soil degradation and erosion processes, which occur extensively in agricultural areas, such as central Spain. Soils with poor general aptitude for most agricultural uses, as a consequence of such processes, have traditionally shown a particular high aptitude for the vineyard. The territory of the Protected Designation of Origin (PDO) "Wines of Madrid" includes 11,500 ha currently under vineyard cultivation, partially characterized by soils which may be considered as degraded, generally as a consequence of erosive processes. The selection and cartography, in a semi-detailed scale, of the most suitable areas for rainfed vineyards, have been the main objective of the edaphoclimatic assessment project developed in this territory by IMIDRA.

The vine requires cold winter, spring rains, and sun with moderate heat during the growth and ripening of the fruits in the summer (Cara, 2009), being sensitive to spring frosts. Blij (1983) considers that the geographical area of the vine is delimited between the annual isotherms of 10ºC and 20ºC.

The presence of vineyards focuses on the southern third of Madrid Region. This territory shows a Continental Mediterranean climate with average temperatures ranging from 5-6 ^{o}C (January) to 24-25^{o}C (July) and average annual rainfall from 400 to 600 mm. *Xeric* soil moisture regime (SSS, 2014) is the only one defined in that context. Substrates show a wide variety from the lithological and geomorphological viewpoint, with gently undulating reliefs and altitudes ranging from 500 to 800 m. Soil types define two distinct areas: West, dominated by *Haploxeralfs*, *Xerorthents*, and *Dystroxerepts* groups, and East, with a predominance of *Calcixerepts* and *Haploxerepts* groups (SSS, 2014). A total of 21 pilot areas have been established in the PDO territory for a semi-detailed study at 1:25,000 scale.

MATERIALS AND METHODS

The methodology used in this project was developed in the framework of the Land Suitability Classification system as established by FAO (1976). The structure of Land Suitability Classification has been applied in the present work on the basis of diverse territorial units defined and delineated by soil and climate criteria, with the support of a Geographic Information System. Land Characteristics and their expression as Land Qualities have been applied as the base of defining limitations in relation to vineyard requirements (FAO, 1976). Thus, a key component of the developed methodology is the selection and rating of edaphoclimatic parameters that significantly affect vineyard development as well as must production and quality.

Morphological and physico-chemical data from 469 soil profiles have been used, as well as climatic data from 24 meteorological stations in the southern third of the Madrid Region. Soil profile description (Schoeneberger *et al*., 2012) constitutes the basis of soil sampling and subsequent physicochemical characterization. Five severity levels, from "very favorable" to "very unfavorable"

(FAO, 1976) applied to each land characteristic from soil profiles and meteorological stations, are the core for the subsequent assignment of Suitability Classes to the Land Evaluation Units. Soil profiles and meteorological stations characterize certain territories that are considered as Land Mapping Units (LMUs). The delineation of such territories has been carried out according to physiographic criteria such as altitude, lithology, landforms and contrasting vegetation or land uses patterns. The definition of the vegetative period of the vineyard is crucial. Most of the land characteristics established as limitations are referred to this period. For the purposes of the study carried out, the following periods of interest for the vineyard have been established: vegetative period, from April 1st to October 7th (including vintage period from September 1st to October 7th) and dormancy period, from October 8th to March 31st. Limitations and intervals have been established in the context of the PDO from the background revision and the experience of IMIDRA researchers, as summarized in Table 1.

Table 1. Land characteristics and severity levels for vineyard evaluation.

Limitations	Very unfavourable	Unfavourable	Slightly favourable	Favourable	Very favourable
Climatic parameters (for meteorological stations)					
Average absolute minumum (ºC) annual/vegetative period	<-15 / -	<-12 / <-1.5	-	- / -1.5<T<-0.5	- / >-0.5
Average minimum during vintage	-	≤8	-	>8	-
Annual average (ºC)	<9		-	9≤ T <11	11≤ T <18
Average absolute maximum (ºC)	-	≥40	-	<40	-
Average annual rainfall (mm)	-	<350	-	>350	
Average wind speed (may and june) (km.h^{-1})	>70 in ≥20% of years	15-70 in ≥20% of years	-	<15 in ≥20% of years	-
Air humidity conditions	-	High frequency of ≥2 consecutive days with ≥10 mm rainfall and T≥12º**C**	-	Low frequency of such combination	-
Sunshine hours per year	<1500; or <1200	<1500 and >1200 during vegetative period	-	>1500 and >1200 during vegetative period	
Soil morphological and physico-chemical parameters (for root zone in soil profiles)					
Internal drainage (classes) *(1)*	very poorly drained	somewhat poorly drained	excessively or moderately well drained	well or somewhat excessively drained	
effective depth (cm)	<35	35-50	50-80	>80	
USDA textural class (<2mm fraction)	-		silty loam; silty clay; silty; sandy, clayey	other classes	Loamy sand; sandy loam
AWC compensated *(2)* (mm /150 cm)	-	Very low (<64)	low (65-127)	moderate (127-190)	high (>190)
Structure and consistency	-	Dominant rock structure and very hard, or compacted or very compacted	Dominant rock and hard, soft or loose. Weak structure and moderately compacted	Moderately structured and weakly compacted	Strongly structured or compound granular. Not compacted
Coarse fragments (>2 mm; % volume)	-	abundant (>40)	many (15-40)	frequent (5-15)	few (<5)
ECe (dS.m^{-1})	>16	8-16	4-8	2-4	<2
pH (H_2O, 1:2.5)	<4.5; >9.0	4.5-5.5; 8.5-9.0	8.0>pH>8.5	7.0>pH>8.0	5.5>pH>7.0
Base saturation (%)	<15	15-35	35-60	60-75	>75
CEC (cmol$_c$.kg^{-1})	<5	5-10	11-20	21-25	>25
Active lime (%)	>20	-	-	10-20	<10
Organic matter (%)			<1.0; >5.0	1.0-2.0	2.0-5.0

Note: cells with "-" indicate not relevant conditions in the PDO. *(1)* Schoeneberger et al., 2012; *(2)* "c" is referred to the compensating effect exerted by the precipitation during the vegetative period (Pveg) on the AWC. It´s applied when Pveg>200 mm.

The relative frequency of the most limiting values determines the assignment of general limitations to each of the meteorological stations and soil profiles (as basic evaluation units), from "very severe" to "no-significant" limitations (Table 2).

Table 2. Land Suitability Classes for Land Mapping Units

Soil limitations (for a specific Land Mapping Unit -LMU-)	**Climatic limitations (for a specific LMU)**	**Suitability Classes for LMUs**
LMUs with soil profiles with only slight limitations	slight	S1
LMUs dominated by soil profiles with slight limitations and presence of soil profiles with moderate limitations. Absence of severe limitations		S1/S2
	moderate	S2/S1
	severe	S2/S3
	very severe	N2
LMUs dominated by soil profiles with moderate limitations. Presence of others soil profiles with severe limitations. Absence of soil profiles with very severe limitations	slight	S2
	moderate	S2/S3; S3/S2
	severe	S3
	very severe	N2
LMUs dominated by soil profiles with severe limitations.	slight	S3
	moderate	S3/N1
	severe	N1; N2
	very severe	N2
LMUs dominated by soil profiles with very severe limitations	slight	N1; N2
	moderate	
	severe	N2
	very severe	

Thus, a single land characteristic classified as "very unfavorable" leads to qualify the basic evaluation unit as with "very severe limitations", or three or more "unfavourable" characteristics leads to define "severe limitations", among other combinations. Since edaphic profiles constitute the basis for the definition of cartographic units, the integration of edaphic and climatic parameters allows assigning a Suitability Category (FAO, 1976) to each of the Land Mapping Units. Four suitability categories have assigned for each LMU, namely: "Orders" (suitable -S- or not -N-), "Classes" (degree of suitability: S1 to S3; N1 or N2, including specific combinations of them), "Subclasses" (main limiting factors that determine suitability; eg: w, climate; m, low AWC) and "Units" (homogeneity of LMU in terms of kind of cartographic unit: "consociation" or "complex" according to Van Wambeke and Forbes (1986).

RESULTS

The climatic limitations observed in the scope of this study refer specifically to temperatures during vegetative period (spring frosts) and have been applied in the Eastern zone of the PDO, especially in the great valleys (Tagus, Jarama and Tajuña rivers). No relevant limitations by frost are considered in Western areas of the Region, as well as a higher annual rainfal than that recorded in the Eastern zone, although extremely high temperatures (≥40ºC) must be taken into account in these areas. Regarding soil types, a significantly high vineyard frequency has been currently found in *Xerorthents* and *Calcixerepts* Groups. Conversely, have been observed a significantly low frequency of vineyards in *Haploxeralfs*. However, *Haploxeralfs* group show the highest frequencies of profiles with "slight" or "moderate" limitations for vineyard cultivation, while *Xerorthents* present the highest frequency of soil profiles with "severe" and "very severe" degrees of limitation (Figure 1).

As for the evaluation of Land Mapping Units (LMUs), "Moderately suitable" (S2) and "Marginally suitable" (S3) constitute the most frequent Suitability Classes obtained in the PDO territory. LMUs dominated by *Haploxeralfs* (widespread in the PDO) mainly show slight and moderate limitations derived from structure and consistency, fine textures, and low CEC as well as numerous areas with moderate limitations by low temperatures. LMUs dominated by *Calcixerepts* (Eastern areas) are mainly characterized by moderated limitations both climatic and edaphic (structure, texture and low CEC). In this case, such limitations are mainly related to the presence of massively carbonated subsurface horizons. On the other hand, LMUs with *Xerorthents* as dominant group reflect moderate and severe limitations associated to low AWC and CEC, both generally as a consequence of coarse textures with low organic matter content. By contrast, such LMUs generally show slight climatic limitations.

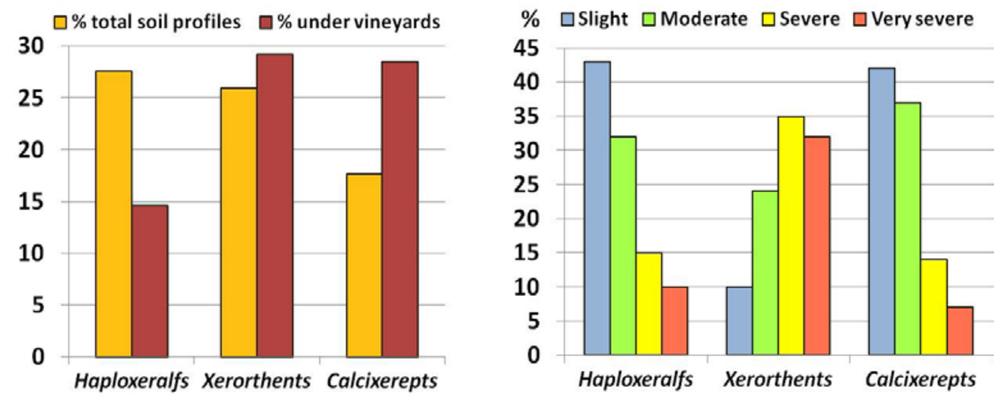

Figure 1. *Left*: percentage of each Soil Group regarding total and vineyard soil profiles. *Right*: percentage of limitation degrees according to Soil Groups

CONCLUSIONS

This methodology has allowed to select the most suitable areas for vineyard in the PDO "Wines of Madrid", and is susceptible of application in any other current or potential areas of vineyard, being convenient some appropriate adaptations fo the established land characteristics and severity levels, according to the use of different vine varieties in a given area. The obtained results suggest that the vineyard is currently underrepresented in extensive areas classified as "suitable" or "moderately suitable" for vineyards, and is, likewise, a crop widely established in soils that can be considered degraded as a result of erosion. Although the erosive processes are significant in the whole of PDO, the effects are particularly noticeable in areas dominated by *Calcixerepts* and *Xerorthents*, so that the C horizons commonly appear very close to the surface. The characteristics of these horizons, however, do not imply, in general, severe or very severe limitations that compromise the aptitude of these surfaces for the vineyard. As more widespread edaphic limitation, the soils of the PDO present a low storage capacity for cations (low CEC), suggesting the addition of organic fertilizers such as a general amendment which is, in any case, appropriate for other defined edaphic limitations.

ACKNOWLEDGEMENTS

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5.16.O

GATHERING ESSENTIAL DATA AS A PRELIMINARY STEP FOR WATER ACCOUNTING IN AN IRRIGATED BASIN

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1. INTRODUCTION

Water accounting in irrigated land requires determining the water budget components at a field and basin scales. These data will be the basis for calculating water use efficiency and water use impact and the effect of water government and management options as well as climate change scenarios on the water use indicators. Remote sensing data is a potential candidate for closing the water budget of vast areas. Our study aims at integrating remote sensing data, field measurements and land surface models for calculating the water balance components of the Algerri-Balaguer basin (Ebro Basin, Spain) in a way determining the water consumption of the basin fields. Considering outlets of the two drainage channels in the basin , our approach requires determination of the areas drained by the two channels with verification of the land use. The drainage area is commonly determined by using digital elevation models (DEM) (Ariza-Villaverde et al., 2015). However, the DEM resolution is of main importance due to its influence on the determination accuracy (Kevin J. McMaster, 2002). In particular, Junming Chen (2010) specified low resolution DEM; cell size of range between 200 m and 1000 m; as poor candidate for the automatic extraction of channel networks due to producing area with little or no relief. In this work, two digital terrain models (DTM) of 5 m and 25 m resolution were compared to a 200 m resolution DTM in order to accurately determine the drainage basin of Algerri-Balaguer. Furthermore, field survey along with the data base of the Noguera was used for identifying the land use and land cover of the basin.

2. MATERIAL AND METHODS

2.1. Study region

Figure 1 shows the Algerri-Balaquer basin which is located at the north east of Spain. The basin is bounded by Noguera Ribargozana is at the west of the region while Farfanya, and Segre rivers are at the east and the channel is at the north which separates the hilly area of Noguera from the basin. The basin has above sea level elevation range between 180 m at the south and 420 m at the north and is cultivated by cereal crops, fruit trees, olive trees, and vegetables (as onions) and grass bush based mainly on irrigation. Two main drains serves the basin with outlets at (41 48 4 N, 0.0 38 53 E) and (41 45` 37`` N, 0.0 40` 13`` E). The outlets are named, hereinafter, as AB1 and AB2 respectively. .

2.2. DTM

Three digital terrain models (DTM) of 5 m, 25 m and 200 m were provided by the Spanish national geographic institute in order to determine the areas drained by the two channels. The DTMs are built based on geodetic reference system ETRS89, in the Canary Archipelago REGCAN95, which is compatible with ETRS89 with Universal Transfer Mercator projection (UTM) and can be found at https://www.ign.es/ign/layoutIn/modeloDigitalTerreno.do?locale=en.

2.3. Noguera database

Noguera is a well defined database for the land and land use provided by the Government of Catalonia, the Ministry of Agriculture, Livestock, Fisheries and Food to be used for this study. Given that information on the study basin is included in Noguera, the database is used as a reference for this work since it incorporates satellite images and aerial ortho-imagery. The incorporated imagery meets the official minimum scale mapping of 1:5000 and provides a continuous view of the land around the Catalan territory.

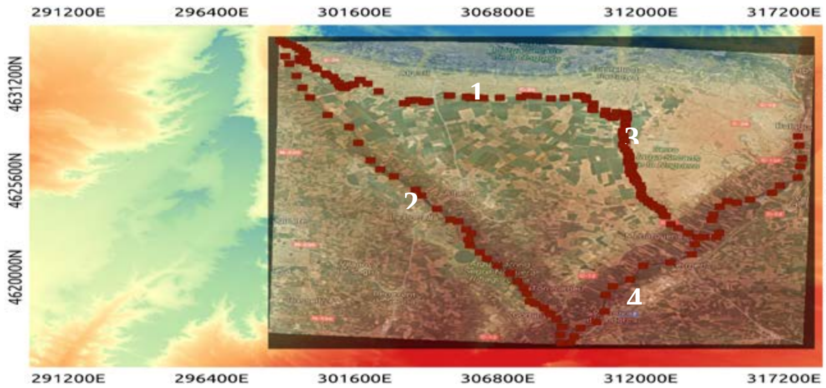

Figure 1: a Google map overlaid on DTM of 25 m resolution to show the study basin bounded by 1) man-made channel, 2) Noguera Ribagozana, 3) Farfanya and 4) Segre rivers which are in red dots

2.4. Feature extraction

The boundaries of the study basin are digitized using a Google map image in order to extract the basin area from three DTMs (figure 1). The stream networks and boundaries of the sub-basins served by the two ditches are automatically extracted by tools of Grass GIS 7 within QGIS 2.18. The tools apply the AT least-cost search algorithm which minimizes the impact of DTM data error (Ehlschlaeger 1989) in a way gives more accuracy in comparison to similar tools used by other software (Kinner et al., 2005). Thus, the depressions are filled and obstacles are removed in order to determine the flow accumulation and flow directions. Specifically, the tools enables producing accumulation map giving the amount of overland flow of each pixel, drainage raster map showing the drainage directions, output basin map contains the boundaries of watershed basins, and other essential maps such as the one representing the slope steepness.

2.5 Features extraction evaluation

The boundaries and area of the sub-basins delineated from the three DTMs are compared to information extracted from the data base of Noguera region. The criterion used for measuring how accurately the delineated sub-basins match the reference ones is the longitudinal root mean square error (LRMSE) (Anderson et al., 2014). The LRMSE is defined as the root mean square error computed between a number of paired sets of points located along both the delineated and reference sub-basins. The boundaries of the reference sub-basins were divided into (m) equal length segments and (n) evenly spaced points. The distance (d) from each segment of the reference boundaries to the delineated one was measured in order to calculate LRMSE addressed by the following equation.

LRMSE = sqrt{sum(d2)/n}

3. RESULTS

3.1. Features accuracy

Figure 2 shows the extracted area of the DTM together with the calculated slope of each pixel, the sub basins, the main streams and the drainage areas of the two outlets.

A general tendency of mismatching the northern boundary of the basin drained by the AB1 ditch is observed from the three resolutions of the DTMs. The 200 m resolution resulted in the less accurate delineation where LRMSE was more than 35% and the 25 m resolution gave better result (LRMSE = 27%). The best result shown in figure 2F was obtained from the 5 m resolution (< 20%). In contrast, the calculated LRSME of other boundaries of the basin are less than 25 % for all DTMs. The elevation variability might explain this result where the slop is very steep at the northern boundary of the basin in comparison to other boundaries. Indeed, the 200 m resolution DTM is smoothing the elevation in a way makes the delineated boundary deviates from the reference one. The delineation accuracy improves as higher resolution DTM is used where more details of the boundary are captured. For the sub-basin drained by the AB2 ditch, similar results were obtained. The calculated LRMSE are 31%, 22% and 13% for the 200 m, 25 m and 5 m resolutions respectively.

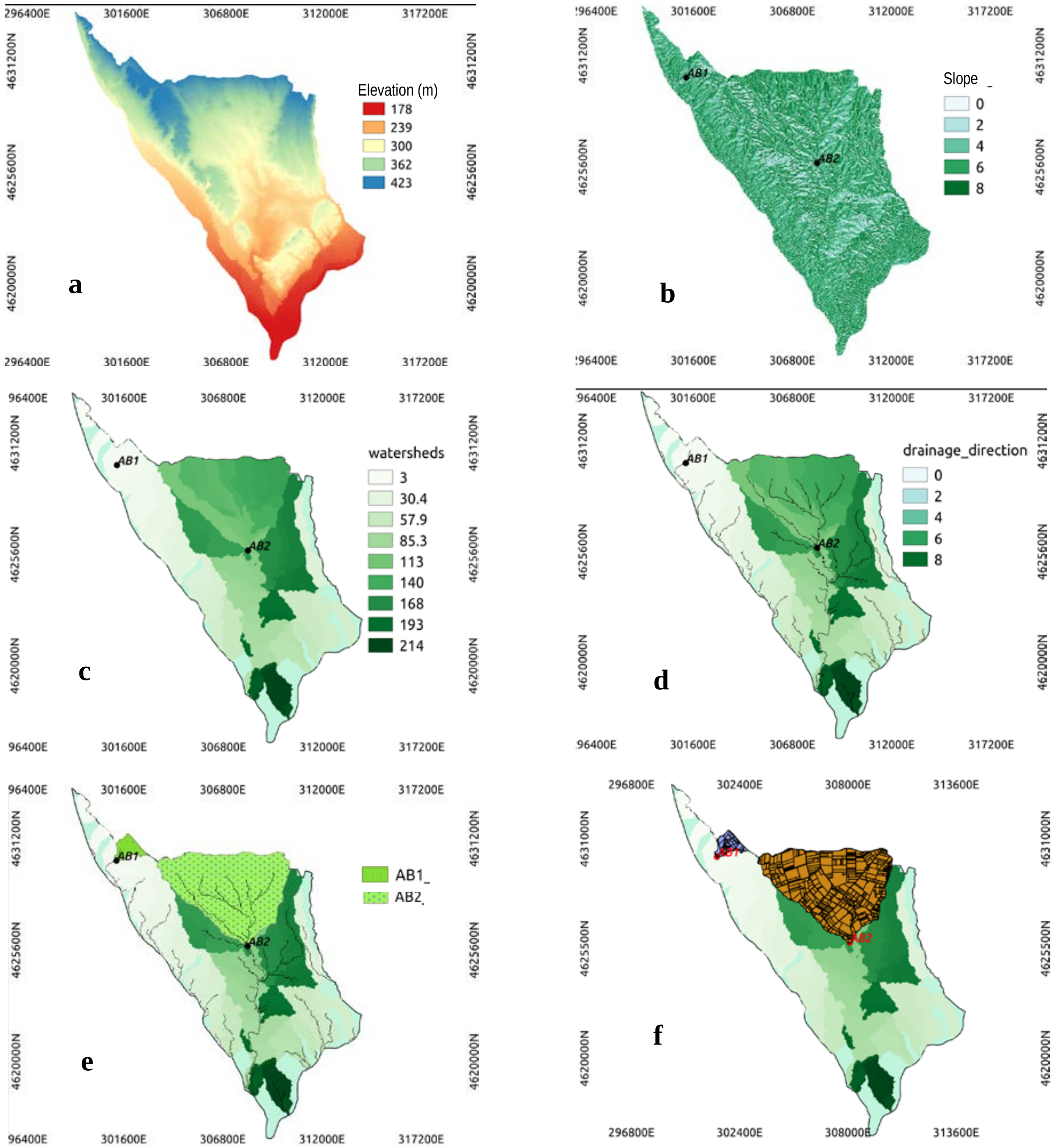

Figure 2 shows a) the extracted area of the DTM together with b) the calculated slope of each pixel, c) the sub basins, d) the main streams and e) the drainage areas of the two outlets matching the Noguera data base (f)

3.2. Drainage area

The areas of the delineated sub-basins drained by the two ditches are calculated by multiplying the pixels of each sub-basin by the resolution of the DTM. It was found that the most accurate DTM is the one of 5 m where the calculated area of the AB1 sub-basin is 198.21 ha while the area of the sub-basin AB2 is 2386.1 recording a relative error of 0.05% and 0.07% respectively. The associated relative error to using the 25 m DTM was 0.1% and 0.2% for the sub-basins AB1 and AB2 respectively. The 200 m resolution DTM resulted in the highest relative error of 0.3% and 0.5% for AB1 and AB2 sub-basins respectively.

3.3. Land use/Land cover

A field survey is conducted by checking the land use and land cover of randomly selected 32 fields. The collected data is compared to the extracted data set of the two basins from Noguera. The comparison showed that the 57 fields are matching the data set of Noguera where 10 fields were covered by grass, 7 fields of cereal crops, 5 fields of fruits and 10 fields of olives trees. In general, the Noguera database shows that main land covers of the two sub-basins are cereal crops (TA), fruits (FY), water surface (AG), grass bush (PR), roads (CA), not used areas (IM), olive trees (OV), building (ED), pasture (PS), and vegetable garden (TH). Figure 3 summarizes the area of each land cover of the two sub-basins.

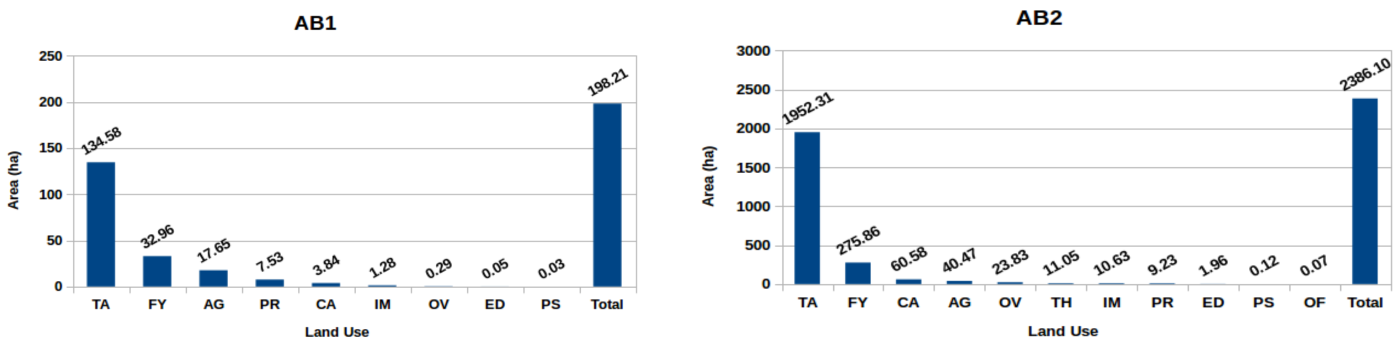

Figure3: areas of the land covers served by the drainage channels AB1 and AB2

4. CONCLUSIONS

Three DTMs of different resolutions (5, 25 and 200 m) were used for automatically delineating the drainage areas served by two ditches in the Algerri-Balaguer basin, Spain. The drainage areas were delineated using the tools of Grass GIS 7 within QGIS 2.18. The tools apply the AT least-cost search algorithm which minimizes the impact of DTM data error. It was found that the higher the resolution the more accuracy in delineating the sub-basins boundaries especially where steep slopes exist. In particular, the longitudinal root mean square error calculated for the three DTMs recorded lowest value for the 5 m resolution. As a result, it was found that the resolution of the DTM affects the calculation of the drainage area. Similarly, the 5 m resolution DTM gave the most accurate area in comparison to the reference data set followed by the 25 m resolution and, then, the 200 m resolution DTMs.

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5.17.O

APPLICATION OF SPACE SERIES ANALYSIS TO COMPARE THE EFFECT OF TILLAGE DIRECTION ON SOIL PROPERTIES IN ADJACENT FIELDS

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INTRODUCTION

Land use/management changes are certainly the most important factors that influence the protection of natural ecosystems (Vitousek et al., 1997). Land use changes from grassland to agriculture reduce soil organic matter content (Dawson and Smith, 2007) and adversely affect soil physicochemical properties (Tejada and Gonzalez, 2008). Tillage type can have an impact on the soil aggregate stability and quantity of soil organic matter (Xin et al., 2015). Incorrect tillage and cultivation can lead to soil erosion, reduced water retention capacity and increased bulk density and soil structural degradation (Ferreras et al., 2000). The comparison has been made between the adjacent fields in many studies (e.g. Kirkby et al., 2000; Celik, 2005; Blanco-Moure et al., 2016; Mloza-Banda et al., 2016) on the effects of land use changes, management, and tillage type and operation technique on erosion rate, soil properties, soil carbon content, etc. The problem with such comparative studies is the assumption of initial uniformity of the soil characteristics in adjacent areas, which is in opposite of soil spatial variability. Our aim was to test the hypothesis that spatial variability of soil properties is ignored in comparative studies of management/land use change, and to present a method based on time series analysis for comparison of soil properties between adjacent fields of different management/land use considering this variability.

MATERIALS AND METHODS

The study area was a sloping land of rainfed agriculture in Iran (located about 36° 22′ N, 49° 35′ E). The mean annual precipitation and potential evaporation of the region are 325 mm and 1200 mm, respectively. The study area is a south face hillslope having average gradient of 10 percent, consists of two parts (Fig. 1). The eastern part is a Research Station in which the contour tillage (COT) is performed for more than 30 years and grazing is also prevented. The western part belonged to farmers in which the cultivation is up-down the slope by moldboard plow (UDT) and the fields are usually grazed after harvest. To conduct the study, we select the middle position of the hillslope. Soil sampling was performed in July 2013 at 85 points of 5 m intervals along a straight line at the middle slope position (Fig. 1). The number of soil samples taken from UDT and it's adjacent COT were 45 and 40 samples, respectively. The samples were analyzed for common physicochemical properties.

The effect of tillage type on soil properties was evaluated by selecting ten sampling points from each field, and the Mann-Whithney U test was applied to compare mean values of the soil properties determined in the UDT and COT fields. This analysis is determined the gross difference between the two tillage/cultivation types which includes natural spatial variability. The net effect of up-down slope tillage on soil properties was determined in comparison with contour tillage in the same field at three steps. The method is based on the assumption that the UDT field would have had what soil properties if it was under the COT management practices. **Step one.** Space series diagrams were plotted for 40 soil samples of COT field. **Step two.** The data normality was checked for all data sets. The autocorrelation function (ACF) and partial autocorrelation function (PACF) were computed for space series of data to estimate the order of the models (Parlange et al., 1992). Autoregressive (AR), moving average (MA) and autoregressive-moving average (ARMA) models were used in this study. **Step three.** The best model obtained for each soil properties based on 40 soil samples of COT was used to predict the value of the property in ten adjacent points (separated by 5 m interval) in UDT field. Finally the difference between the predicted and the measured data of the same ten points was considered as the net effect of UDT in comparison with COT. Also a Mann-Whithney U test was run for the data of UDT (for 10 measured points and 10 predicted points).

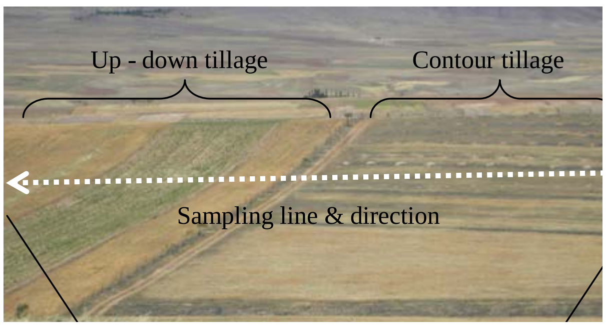

Figure 1. Study area and sampling design

RESULTS AND DISCUSSION

Mean comparison of the data by Mann-Whithney U test showed a significant difference between the two tillage types (UDT and COT) in terms of geometric mean diameter of soil aggregates, calcium carbonate equivalent, saturated soil water content and pH at $P < 0.05$, and the percentage of sand, silt and clay, mean weight diameter of soil aggregates, bulk density and organic carbon at $P<0.01$. The test shows no significant difference in EC between two tillage types. The geometric and mean weight diameters of soil aggregates, soil bulk density, saturated soil water content, sand percentage, calcium carbonate equivalent and organic carbon content were higher in the contour tillage than in the up-down tillage field. But the results were reversed in terms of soil pH and clay and silt percentages. Accordingly, based on the analysis by classical statistics (Mann-Whithney U test), the difference in tillage type has resulted in changing of most measured soil properties with at least 95 percent confidence level ($P < 0.05$). This is in agreement with the literature in general, but this analysis like most of the previous ones is unable to realize the soil spatial variability and separate it from the change induced by management (including tillage type, cultivation, land use etc.). Marzvan et al. (2015) reported that spatial variability and dependency of soil properties are quite different in adjacent fields of different tillage management probably due to the effect of tillage on the random component of soil spatial variability.

Our results showed that ARMA and high-order MA models had better capabilities for modeling the space series of soil properties. ARMA and MA were the best models for 6 and 4 soil properties, respectively. MA model of high order (5 to 9) obtained for sand, clay, soil pH and electrical conductivity of soil solution probably indicates the dominancy of random component of spatial variability in these cases affecting 5 to 9 previously located points in COT field. The selected model of each soil property (based on 40 soil samples of COT) was used to predict the values of the property in ten adjacent points (separated by 5 m interval) in UDT (Fig. 2; selected properties).

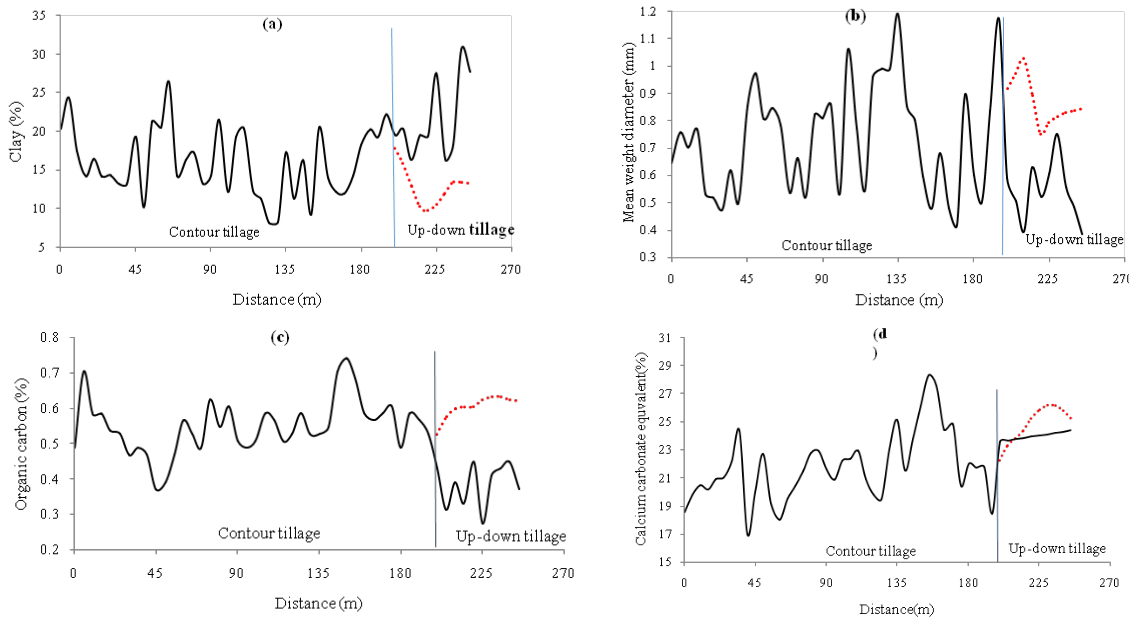

Figure 2. Space series of measured (solid line) and predicted (dash line) data for clay (a), mean weight diameter (b), organic carbon (c), calcium carbonate equivalent (d).

In contrast with the results of Mann-Whithney U test run on the original data in which all soil properties (except EC) were significantly different between COT and UDT, comparison between the predicted and measured data in UDT showed that there is no significant difference ($P<0.05$) between these two data sets for sand percentage, bulk density, electrical conductivity and pH. Also, the significant level increased from 0.05 to 0.01 for GMD and Saturated soil water content. Figure 5 presents space series of measured (40 points in the COT field and 10 points in the UDT field) and predicted (10 points in the UDT field) data for those soil properties show significant difference by Mann-Whithney U test between the predicted and measured values and for sand and calcium carbonate equivalent an example of those do not show significant difference. The predicted values are higher than the measured values for weight mean diameter, geometric mean diameter, organic matter and moisture saturation percentage. This means that the soil of the field with up-down tillage would have higher weight mean diameter, geometric mean diameter, organic matter and moisture saturation percentage if it was under the cultivation and tillage management of the adjacent field with contour tillage. In other words, contour tillage especially decreases soil erosion in compare to up-down tillage results in increasing soil quality (Truman et al., 2011). Higher measured clay content in UDT in comparison to predicted one could also be an indicator of surface layer removal by tillage-water erosion. The results of Asadi et al. (2017) measuring soil properties in 24 points with three replications in these fields show that clay percentage is higher 4.5 percent at 15-30 cm depth than at 0-15 cm depth in average. There were not significant differences between the two depths for sand in overall and for calcium carbonate equivalent at mid slope position (Asadi et al., 2017). Figure 5 also shows the results for calcium carbonate equivalent in which there were no significant differences between predictions and measurements. Therefore, in the current study, they were not affected by tillage.

CONCLUSION

The assessment of tillage type effect on soil properties by space series analysis illustrate how we can realize and differentiate the natural soil spatial variability from management induced changes when comparing the adjacent fields. The comparison is usually made between the adjacent fields to study the effects of land use /management changes on soil characteristics, is based on the assumption of initial uniformity of the soil in adjacent areas. We hypothesized that this assumption is contrary to the spatial variability of soil characteristics. The effect of up-down tillage on soil properties was compared with contour tillage by predicting soil properties in the same field. The generated soil data of UDT field were predicted by the ARMA and high-order MA models developed by space series analysis of soil data of CT field. The results of statistical comparison between the predicted and measured data in UDT field were different from the results of statistical comparison between COT and UDT. The results of space series analysis of this study reject the assumption of initial uniformity of the adjacent fields of previous studies.

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5.1.P

PERFORMANCE EVALUATION OF DECANTO-DIGESTOR IN A DOMESTIC WASTEWATER TREATMENT PLANT OF A RURAL SETTLEMENT

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ABSTRACT
The utilization of wastewater after proper treatment is indicated as an alternative source of water and fertilizer for agriculture, but because of its limitations on chemical, physical and biological properties, these effluents must be monitored and evaluated with a view to environmental safety, the security of quality of life and generation of rural socioeconomic development. The aim of this study was to evaluate a low cost modular system for the collection and treatment of domestic sewage in areas of rural settlements in order to reuse it in production of ornamental plants, grasses and seedlings for reforestation. In view of the results, it was concluded that there was significant removal of most of the studied parameters.

1. BACKGROUND AND INTRODUCTION

The scarcity of water resources in the world, sometimes by absolute lack of water and other times by contamination from natural sources, is a cause for concern in the scientific community and has fostered discussion and research in the field of wastewater reuse.

According to the IBGE (Brazilian Geographical and Statistical Institute), atlas sanitation through 2011, despite significant regional differences persist in the scope of municipal sewage services, water supply, stormwater management and solid waste between 2000 and 2008 there was a breakthrough the number of municipalities covered by sanitation in all regions of Brazil.

Wastewaters are rich in chemicals and they pose potential biological risk to human health. Due to these features, the National Environmental Council - CONAMA and the Environmental Sanitation Technology Company - CETESB set limits for acceptable agronomic and environmental water reuse in agriculture. The concentration of heavy metals, total content of salts and biological parameters are seen as essential in assessing the quality of wastewater for irrigation.

The decant-digestor, followed by anaerobic filters, constitutes a system that can be very advantageous for wastewater treatment. It associates, in series, a reactor resistant to variations in the affluent with a reactor that is also efficient on the dissolved portion of the sewage. It occasional operation and requires no specialized operator, having immediate departure with proper functioning from the beginning absorbs shocks and toxic overload with rapid recovery and does not lose efficiency over the long term (ANDRADE NETO et al, 2000). According to Lo Monaco et al., (2009), the volume of wastewater applied in agriculture should be based on the recommended dose of nutrients for crops, and not on water need. Since if the nutrient content reaches high values, it can cause soil, surface water and groundwater pollution.

The aim of this study was to evaluate a low cost modular system for the collection and treatment of domestic sewage in areas of rural settlements in order to reuse it in the production of ornamental plants, grasses and seedlings for reforestation.

2. MATERIALS AND METHODS

2.1 Case study sites

The municipality of Apodi, geographical coordinates 5°39'55'' South latitude and 37°48'13''West longitude, is located in the West Zone of Rio Grande do Norte (Chapada do Apodi), 375 km from the capital Natal, RN, Brazil. It has an area of 1556.1 km^2, corresponding to 2.75% of the surface of the state. Its climate is dry, hot and wholesome, with predominance of a semi-arid climate. It has a maximum temperature of 37° C and a mean annual temperature of at least 21°C.

The area consists of a sewage collection network and Wastewater Treatment Plant (WWTP). This WWTP was constructed with local material resources, and involves an anaerobic digester-type decanter, followed by anaerobic filters downstream, forming a primary treatment complex. Subseq uently, we built a wetland area of 20 m width by 20 m length, or 400 m^2.The sewage collection network directs the sewage generated in the settlement to the Wastewater Treatment Plant (WWTP).

2.2 Sampling and analytical methods

The experimental system was monitored by collecting samples of raw sewage (EB) between 10:00 a.m. and 12:00 noon. The raw sewage was collected during preliminary treatment, while the effluents were collected in boxes outside of the reactors. The samples were analyzed according to CETESB (2007) and CONAMA (2008) criteria. Analyses were performed using the methodology of Standard Methods for the Examinations of Water and Wastewater of AWWA, 20th Edition, 1998.

3. RESULTS AND DISCUSSION

3.1 Water quality – heavy metals

In Table 01 are shown the results of the heavy metal parameters analyzed over the period from June to October 2011. Based on the results, a gradual increase in the efficiency of the system is perceived in most of the parameters evaluated.

The zinc (Zn) values ranged from 0.94 mg L^{-1} to 0,055 mg L^{-1}, showing efficiency of reduction between the inflow and outflow values of 94.15% (Table 1). These values are within the limits of 5 mgL^{-1} established by CONAMA Resolution 430/2011 and of 2 mgL^{-1} established by CETESB through decree 8.468/76; however, they are outside the limit of 0.01 mg L^{-1} determined by the FAO. Corroborating this study, Costa (2012), evaluating the use of wastewater of domestic origin on the sunflower crop through a wastewater treatment plant on a rural settlement, found a zinc reduction of 57.89%, with its influent and effluent values fitting within the standard established by CONAMA 430/2011.

The decanto-digestor provided average removal of 42.86%, 51.52% and 23.53% of copper (Cu), iron (Fe) and manganese (Mn), respectively. These values are above the reference value established by the FAO; however, they are within the limits determined by CONAMA resolution 430/2011 and by decree no. 8.468/76 established by CETESB. Similarly, Silva (2012), analyzing wastewater coming from cashew nut processing, observed a variation in iron (Fe) concentration of 0.43 mgL^{-1} at inflow and 0.16 mgL^{-1} at outflow from the wastewater treatment plant, generating an iron reduction efficiency of 62.79%. According to Ayers e Westcot (2001), copper (Cu) concentrations greater than 0.2 mg L^{-1} may cause toxicity in plants grown in nutrient solutions. In addition, zinc concentrations greater than 2.0 mg L^{-1} also cause toxicity in plants; however, in soil with a pH greater than 6.0 and fine texture, this effect is minimized.

The chrome (Cr) values found both in the influent and effluent were 0.05 mg L^{-1}, without removal through the treatment system (Table 1). According to the reference values of the FAO, this value is outside the established limits, in contrast with CONAMA resolution 430/2011 and decree no. 8.468/76 established by CETESB, in which this value is within the desirable range. It may be observed in Table 1 that there was no reduction in the concentration for cadmium (Cd) and lead (Pb), both remaining below 0.02 mg L^{-1} and 0.03 mg L^{-1}, respectively. These values are within the limit established by the FAO and CONAMA 430/2011; however, they are above the limit of 0.01 mg L^{-1} determined by CETESB decree no. 8.468/76.

Through the efficiency of removal of the decanto-digestor system, a variation in nickel (Ni) levels of 0.163 mg L^{-1} was observed at inflow of the system and 0.044 mg L^{-1} at outflow, thus generating a reduction of 73% (Table 1). The values found are well above the reference values established by the FAO; however, they are at acceptable levels according to CONAMA resolution 430/2011 and decree no. 8.468/76 established by CETESB.

The mercury (Hg) concentration did not vary after treatment of the influent, remaining below 0.002 mgL^{-1} Hg (Table 1). According to CONAMA resolution 430/2011, the value found is below 0.01 mgL^{-1} Hg, the limit for release of effluents in bodies of water.

Table 1. Total mean composition (mg L^{-1}) of heavy metals in wastewater from the settlement of Milagres, RN, Brazil.

Organic parameters	Reference values	INFLOW	OUT-FLOW	CONAMA 430/2011	CETESB DC.8.468/76
Zinc	0.01 ± 0.01	0.94	0.055	5.0	2.0
Copper	0.038±0.037	0.07	0.04	1.00	0.2
Iron	0.251±0.062	0.99	0.48	15.0	5.0
Manganese	0.113±0.065	0.17	0.13	1.00	0.20
Chrome	0.004±0.001	< 0.05	< 0.05	0.05	0.10
Cadmium	0.002±0.002	< 0.02	< 0.02	0.20	0.01
Lead	0.025±0.014	< 0.03	< 0.03	0.50	0.50
Nickel	0.008±0.007	0.163	0.044	2.00	0.20
Hg		< 0.002	< 0.002	0.01	

Source: FAO (Food and Agriculture Organization); CONAMA Resolution 430/11 and CETESB State Decree 8.468/76.

CONCLUSIONS

The decanto-digestor provided average removal of 42.86%, 51.52% and 23.53% of copper (Cu), iron (Fe) and manganese (Mn), respectively. The mercury (Hg) concentration did not vary after treatment of the influent, remaining below 0.002 mgL^{-1} Hg. The zinc (Zn) values ranged from 0.94 mg L^{-1} to 0,055 mg L^{-1}, showing efficiency of reduction between the inflow and outflow. The chrome (Cr) values found both in the influent and effluent were 0.05 mg L^{-1}, without removal through the treatment system. In view of the results, it was concluded that there was significant removal of most of the studied parameters.

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5.2.P

MEASURED DATA of SOIL PROBES ANALYZED FROM 4 DIFFERENT ARABLE LAND LOCATIONS IN HUNGARY

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INTRODUCTION

Hungary has excellent agricultural potentials. We produced agricultural products on 5.4 million ha in 2016 (http1). The growing population demand more and better quality foodstuff, whereupon increasing amount of yield have to be produced on decreasing agricultural lands. The global climate change, limited water resources, increasing extreme hydrological events (drought, extreme precipitation, floods etc.) and temporal variability of rainfall put a big emphasis of effective water use (Szépszó, 2007; Jolánkai, 2009; Bartholy et al. 2011). We do the best to utilize the potential capacity of our soils and try to reduce the effects of both extremities (drought and floods) (Várallyay, 2009). It not only needs experimental knowledge but efficient, science-based measurement network as decision support system.

The soil temperature and moisture content limit the time and methods of agro-technical operations. Nowadays the tillage of wet or dry soil counts as erratic operation in agriculture which reduces soil resistance and buffer ability. In addition, soil temperature also influences the date of sowing which has effects on germination. There are several methods for measuring soil temperature, soil moisture and groundwater conductivity. Classical methods like analyzing in laboratory take a lot of time. Together and separately, soil temperature and soil water content are widely studied in science as well also in Hungary. The subject of many investigations are in connection with litter cover, canopy and mulch and its effect on temperature and soil water content of the upper layer of soil (Krakomperger, 2011; Bidló et al., 2014; Kim et al., 2016; Király et al., 2016). In our research we use self-developed soil probes measuring soil temperature in the field.

MATERIALS AND METHODS

The present study was carried out on 4 arable lands in Hungary. The common point of these was the crop (maize) grown on them. The sampling plots possess different soil parameters, different locations and exposure, and different cultivation techniques. The aim of this research is defining dynamics in soil vertically, across the plot or finding differences between sites. Soil temperature was measured by newly developed soil probes. We used 2 probes, the short probes *(Fig. 1.a.)* measure in 8 and 20 cm depths, and the long probes *(Fig. 1.b.)* measure in 8, 20, 40, 60 and 80 cm depths. The probes were placed the sampling plots (2 long and 3 short probes/sampling plot) after sowing and gathered before harvest. Measurements were made every 15 minutes. Preliminary investigations demonstrate the utility of sensors, because sensors provide fresh and accurate data, too. The measuring range of soil temperature is -40 – +60 °C with ±0.1 accuracy. The Sampling plot 1. *(Fig. 2.)* covering approximately 25 ha and located in North-East Hungary. The east side of plot is bordered by grasslands which is due to the necessity of a ditch in the middle of grasslands. Water-effected soils are dominant in the region. The plot has a slope of south-east direction with a total of 2-3 m height difference. Characteristically meadow soil was determined with texture of loam-loamy clay. The sampling plot is rich in organic matter and has favorable soil structure.

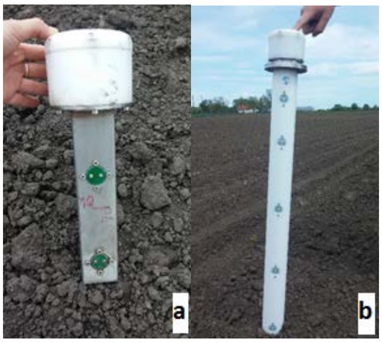

Figure 1. The used short probe (a) and long probe (b) for measurements (Photo: Dobó, 2016)

The calcium-carbonate content is increasing with depth. Except of the top, cultivated layer, the soil is compacted.

Figure 2. Sampling plots of research in Hungary (Google Earth)

The Sampling plot 2. *(Fig. 2.)* covering 13 ha is located in North-West Hungary. The slope of the plot is very little which means 30 cm. Meadow soil was determined with the texture of loam-loamy clay. The Sampling plot 3. *(Fig. 2.)* covering 45 ha is located Middle-Trans-Danubia. There is a lake nearby. The lake has effect on microclimate and also on soil parameters. The Sampling plot 3. declines 100 cm during approximately 800 m distance, between south-east and north-west corners. Brown forest soil is representative with mostly sandy-loam texture. The soil is carbonate-free and it has really low humus content. The Sampling plot 4. *(Fig. 2.)* covering 18 ha is located in Southern-Hungary. This region of Hungary is one of the most suitable areas for agricultural production. In the past there were rivers and water courses running across the area which is visible on *Figure 2.* The height difference is about 70 cm between the south-east corner and the middle of the plot. According to water effect and pressure, the higher parts' common soil type is chernozem, while alluvial soils were found at the deeper parts. The prevailing texture is loam and sandy-loam. The sampling soil is carbonate-free and has high humus content. The probes collected big amount of measured data and sent it to the server via GSM connection. If probe doesn't find network for various reasons, we have incomplete data tables. To simplify statistical analysis needed filtering empty cells, which leads to missing data in every 15 minutes. Despite of filtering we still have huge amount of data to analyze and find similarities or differences. To complete statistical analyses IBM SPSS Statistics 22 was used.

RESULTS

In order to simplify the presentation of soil temperature *(Fig. 3.)* we chose 3 days of summer randomly. Despite of small interval, soil temperature shows daily changes perfectly. July is the hottest month in Hungary when air temperature can reach 40 °C on sunny spots. This high air temperature is also reflected in the soil which is most intense at 8 cm depth. The daily fluctuation decreases with the depth which means that at 40 or even 80 cm depth the fluctuation is normally under 10 °C *(Tab. 1.)*.

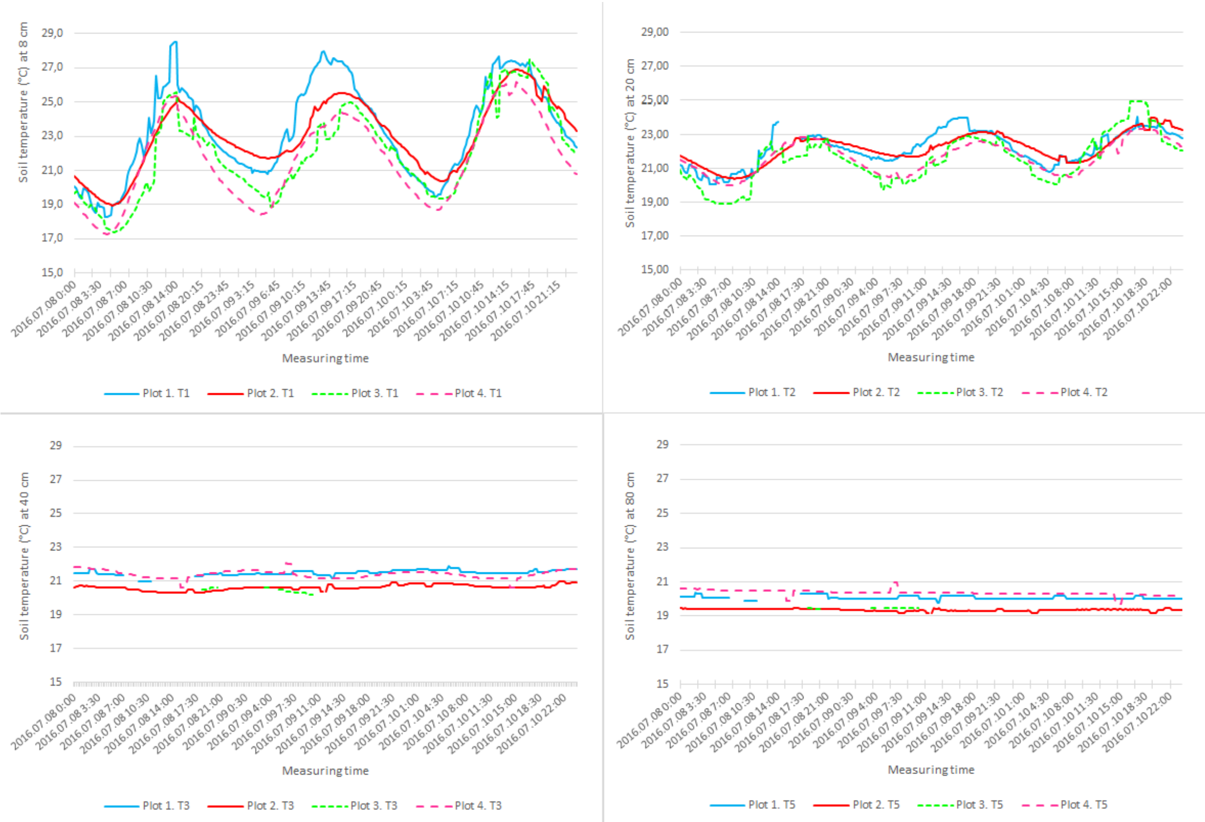

Figure 3. Soil temperature of sampling plots in different depths during a 3 day period

The amount of fluctuation is also represented in *Tab. 1.,* where statistics are based on all measured data. It is clear that the average fluctuation is above 20 °C at 8 cm depth while it's about 10 °C at 80 cm depth. It is conspicuous that Sampling plot 4. keeps the heat well in the upper layer because its range is the lowest and min. temperature is the highest in all of the 4 examined plots while the others go cold during the night.

(°C)	8 cm (T1)				20 cm (T2)				40 cm (T3)				60 cm (T4)				80 cm (T5)			
	Plot 1	Plot 2	Plot 3	Plot 4	Plot 1	Plot 2	Plot 3	Plot 4	Plot 1	Plot 2	Plot 3	Plot 4	Plot 1	Plot 2	Plot 3	Plot 4	Plot 1	Plot 2	Plot 3	Plot 4
Min	5,9	9	7,4	11,8	7,4	8,7	9,8	13,6	10,5	17,3	11,3	14,9	11,3	16,6	11,8	14,8	11,3	15,8	12	14,6
Max.	31,5	31,5	32,4	32,2	27,1	26,3	27,6	27,2	23,9	23,1	23,4	24	21,9	22,4	21,6	22,3	20,8	22,2	20,4	21,1
Range	25,7	22,5	25	20,4	19,7	17,7	17,9	13,6	13,4	2,2	12,1	9,1	10,7	5,8	9,8	7,6	9,6	6,4	8,4	6,5

Table 1. Minimum, maximum and range of soil temperature (°C) of sampling plots

During the statistical analyzes means of soil temperature data (measured at 8, 20, 40, 60 and 80 cm depths) were compared first. As data have no normal distribution, Kruskall-Wallis test was used. Distribution of means are not the same across the sampling plots ($p<0,05$). It means that there are difference between sampling plots which is clear on *Fig. 3*. Sampling plot 1. has the highest max. temperature at 8 cm depth, while sampling plot 2. has the highest min. temperatures in represented 3 days. To find out which plots have similar characteristics, cluster analyzes were used. The cluster analyzes results showed connection between Sampling plot 1. and plot 3 while plot 2. and plot 4. were in similar in all depths.

CONCLUSION

To get around imperfect data IT background needs to be proved.

One of our aim was to find dynamics in soil temperature. Probes placed in different depths showed differences. It was clear on line charts but also proved by statistically. While average fluctuation of soil temperature is around 10 °C, until then there is only 1-2 °C daily alternation in randomly chosen 3 days' soil temperature at already 40 cm depth.

According to cluster analyzes Sampling plot 1. and 3., and Sampling plot 2. and 4. have similar characteristics based on soil temperature. The slope and exposure could be the common point of 2-2 plots. While Sampling plot 1. and 3 have steeper exposure, until then Sampling plot 2. and 4. are declivous. According to results represented in this paper soil temperature is highly affected by exposure and slope of table.

To make similarities/differences more explicit, more sampling plots and use other statistical tests are needed to determine these relations. In future we will select not only one but two or three sampling plots in a region firstly and after that compare them.

ACKNOWLEDGEMENT

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http1: https://www.ksh.hu/docs/hun/xstadat/xstadat_hosszu/h_omf001a.html

5.4.P

STONINESS: A SOIL PARAMETER DETERMINED BY DRON

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INTRODUCTION

Coarse fragments constitute a large part of the soil volume. They derived from the weathering process of the original rock or they are being transported and deposited by erosive agents. This fact has direct implications on the physical and chemical properties of soil resource, such as temperature, moisture, infiltration, or erosion capacity among others (Poeson et al., 1994).

In irrigated agriculture, stoniness is usually removed from the soil surface to facilitate tillage. However, in the case of rainfed or conservation agriculture the stoniness can be left to maintain moisture in the field, and therefore increase the available water for plants. In addition, stoniness prevents erosion by the direct impact of water droplets and reduces the natural sealing of soil. By this reason, stoniness is an important factor in degraded areas due to soil erosion, where soil losses are important. From an agricultural point of view, the efficiency of water and the calculation of available water in the soil for the plants has been a clear link between agronomy and leading technologies such as remote sensing or sensor development technologies (Malone et al., 2013; Dimitrov et al., 2014). However, in most of the researches, the areas selected to develop remote sensing studios show the same characteristics: fields with bare soil in order to avoid the effect of stoniness on the soil parameter studied. Therefore, the study of the influence of coarse elements has been avoided in those experiments where some soil variables are being studied. The area discrimination with high and low percentages of surface stones, their influence on the outcomes and the validation of these results in the field, are still waiting for a technical solution that will validate or counteract this situation. Nowadays, with the development of unmanned aerial vehicles, known as UAVs or drones, the studies allow to work with a much more detailed scale and with much greater precision and speed. From the large-scale work with satellites and remote sensing, currently, we have images taken at lower heights, with a very good resolution (centimetres). Studies on plant physiopathology, soil erosion, determination of soil variables or geomorphology are some examples of the application of drones to the field of agronomy and research (Miřijovskýa, 2014)

According to these ideas, the present research seeks to combine diverse sciences such as geostatistics, aeronautics and soil science, in order to identify the surface stoniness through the use of drones. This action will allow the researchers to discriminate areas with coarse elements from soil with the purpose of a better and more efficient management of the soil, because the soil variables analysed by images can be identified in soil and not in a mixture of soil and stones.

MATERIAL AND METHODS

The study was carried out in plots of the Universitat Politècnica de València (39.4839 N and 0.3405 W). Ten small plots of 70 cm x 70 cm were used, being repeated identically 3 times (Figure 1). In each set of 10 plots, different percentages of surface stoniness were established: Control (0% or bare soil), 25% and 50% of stoniness on the soil surface. This percentage was also combined with two colour ranges of the coarse elements: whitish or light colour [W], reddish colour [R] and mixture of both types [M]). At the same time, two vegetation plots [Veg H and B] with 2 levels of soil coverage were used (50% and 100% of soil surface). However, the influence of these plots have not been analysed in this article. The influence of soil moisture was identified as Low (L), Medium (M) and High (H), according the water content in soil. Low was related to dry soil, whereas medim and high were related to wet and soaked soil respectively.

The drone was a BEBOB-Parrot model with four rotors, a camera with Fisheye objectives 180 ° 1 / 2.2" and a sensor of 14 megapixels. There were two flights per session, at the same height and at the same time. To avoid possible differences due to the incidence of light, after the flights were made, frames were extracted from the videos obtained (Figure 3) and the calculation of the stoniness was carried out after the classification of the image by a classification method. In this method, the user must configure the number of classes for the classification, the criterion to stop the iterative process, or the number of iterations of the procedure in the algorithm. K media in the OpenCV library, which is used in many remote sensing and digital image processing programs (Bradski and Kaehler, 2008; Laganière, 2011). In the case of the OpenCV version for the Python programming language, the user can use the *kmeans* command to perform unsupervised classifications. Supervised and unsupervised classifications were carried out to obtain the outcomes. In addition, all the frames were located with UTM coordinates in order to identify the plots studied.

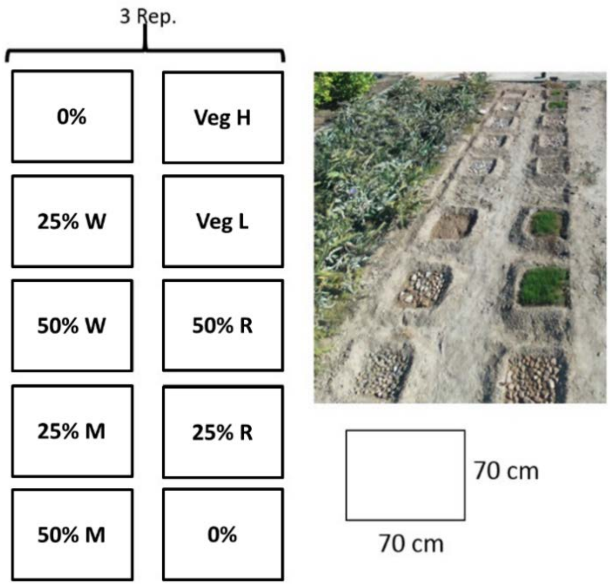

Figure 1 –Plot sketch. Coarse elements: B: Whitish stones; A: Reddish stones; M: Mix of both colours.

The *k-media* method analyses a digital image in a multidimensional space determined by the number of spectral layers in the image. In our case, the space was three-dimensional because we used the trichromatic camera available in the drone. In the multidimensional space a series of points that are called centroids are defined. Each point is a pixel, and therefore at the end, the image is materialized in a set of points equal to the product of the number of rows by the number of columns of the image. In order to perform classification, the file containing the image must be converted to a multidimensional matrix of dimensions (m, n, 3), where m is the number of rows, n is the number of columns and 3 is the number of bands in the image. Once these steps were performed, the stop criterion in the classification was specified in the process. This stop was defined when convergence was supposed to be achieved. In the same regard, it was defined the number of iterations to guarantee convergence (between 10 and 20, obtained experimentally). Tolerance was also defined by comparing the displacement of the centroids of the classes after each iteration. The threshold was set to a digital level unit. Finally, we specified the number of classes to be classified by the method. There were three classes in total: i) stones, ii) soil and iii) background.

The complete images contain a large area of land in addition to the area of interest for the calculation (Figure 2a). Therefore, it was necessary to define the area of interest of the image that would be processed before each calculation. This task is the responsibility of the user since the comparison between the theoretical values of stoniness and the values obtained by the image analysis. It depends logically on the coherence between the processed sub-image and the experimental plots prepared for our study. The area of interest was defined by a mask (Figure 2b).

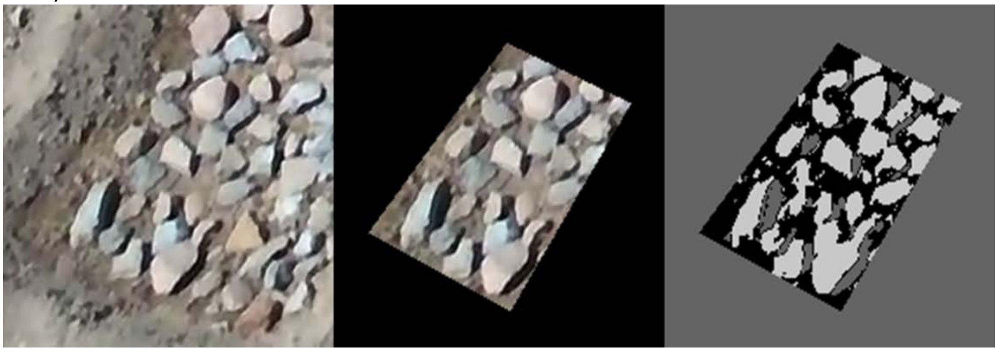

Figure 2 – a) Plot of study with 50% of stoniness, b) mask on the plot, c) classification outcome

Once the parameters were established, the unsupervised classification process was executed and the algorithm automatically assigned to each pixel of the original image a value among the 3 possible classes, representing it by a different gray color or tone for each class (Figure 2c). The strict calculation of stoniness was obtained as a percentage of surface area covered by stones with respect to the total area of the area of interest.

RESULTS AND DISCUSSION

The results of the study are shown in Table 1. The stoniness measurement and soil moisture at the time of making the images appear in the table. As a general view, outcomes show low error values and the moisture is the key on the goodness of the outcomes.

Table1 – Stoniness outcomes. Moisture: High (H); Medium (M) and Low (L).

Plot/image	Stoniness(%)		Moisture
	Real	Calculated	
1-2/35	25	48	L
1-3/35	50	65	L
1-8/35	25	18	L
1-5/36	50	48.4	H
2-3/38	50	47.2	H
2-5/67	25	26.8	H
2-8/17	50	49.5	H
2-7/67	50	43.1	H
2-4/68	25	27.5	H
2-9/68	25	21.4	H

3-2/83	50	47.4	H
3-9/84	25	20	M
3-3/84	50	46.4	M
3-4/89	50	44.5	H

Soil moisture is an important variable, which is reflected in the results. Three qualitative levels of soil moisture were indicated: high (H), medium (M) and low (L), according to the previous irrigation management of each set of plots. Although initially this variable was not part of the study, it has been observed that soil moisture is essential for good results. This is because when the soil moisture was greater, the differentiation between soil and stones was easier, both for visual effects and classification.

This circumstance is clearly observed in the results of low moisture plots (L) where the differences between the theoretical value of the stoniness and its calculated value were clearly unacceptable (15 units of percentage on average value). In contrast, the differences between theoretical values and calculated values were generally below 5 percentage units in the rest of the plots. Exactly 4.3 units for plots with a medium moisture content and 3.08 units for plots with a high water content. Colour of stones did not show any influence on the goodness of the calculations.

CONCLUSIONS

The unsupervised classification system is an automatic method applicable to drone technology, noting that the best results were found when the plots had a high content of water. The moisture allows differentiating more clearly the three different classes. In that case, the error was less than 5 percentage units. It is therefore a good method to quickly evaluate the bare soil surface roughness and therefore to discriminate the study area independently for the analysis of the soil variable. The fact that the result is much more reliable in wet areas than in drylands is an interesting indicator for planning flights destined to the calculation of stoniness. In this sense, the program has also not found differences between the colour of the coarse elements.

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5.5.P

GLOBAL CHANGE EFFECTS IN LOW CASAMANCE (SENEGAL): A METHODOLOGICAL APPROACH

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1. INTRODUCTION

The last decades of the XXth century are characterized by an acceleration of environmental changes at a planetary scale with effects on socioecological systems (BOADA & SAURÍ, 2002). These changes have both biophysical and socioeconomic origins, and the connection between them constitutes the global change (GC). This paper discusses a methodological approach to study the main effects of GC in the Low Casamance (Senegal) (Figure 1), an estuarine area with subtropical climate, where traditional rice cultivation (in flooded rice fields on reclaimed mangroves with hydraulic works of maintenance), fishing and hunting are the most important traditional socioeconomic activities. Under these conditions, in the area there are mainly poorly developed soils, ferralitic soils, hydromorphic soils and halomorphic soils with non-degraded structure (IRD, 2014).

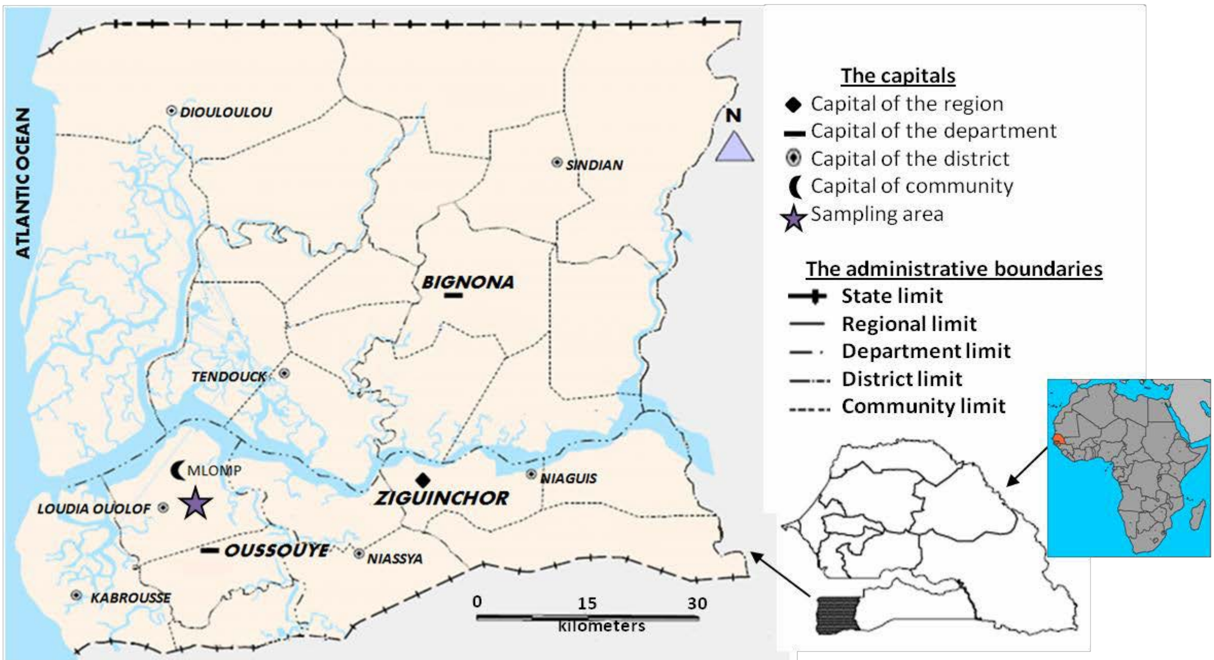

Figure 1: Localization of Ziguinchor region (Low Casamance). Source: Own elaboration from Bodian and Ndiaye (2010) and es.pinterest.com (2017).

In the study area, the local scale expression of GC is caused by a set of biophysical factors, mainly the variability of rainfall, changes in its seasonal distribution, and an increase in temperatures; and socioeconomic factors related with environmental policies, and demographic and economic changes.

1.1 *Biophysical Factors*

Regarding climate, a study of rainfall during the last century shows (Figure 2) how the average annual rainfall before 1970 was over 1.500 mm, while between 1970 and 1990 decreased to 1.100 mm. In the 90s, there was an increase in rainfall but it still remains about 10 % smaller than it was before the 1970s drought. On the other hand, the high annual variability of rainfall is still a dominant characteristic. In the period 1950 – 2014, the average annual temperature in Senegal has a clear tendency to increase, being 0.5ºC higher after the 80s than the annual average over the entire period (Sagna et al., 2015).

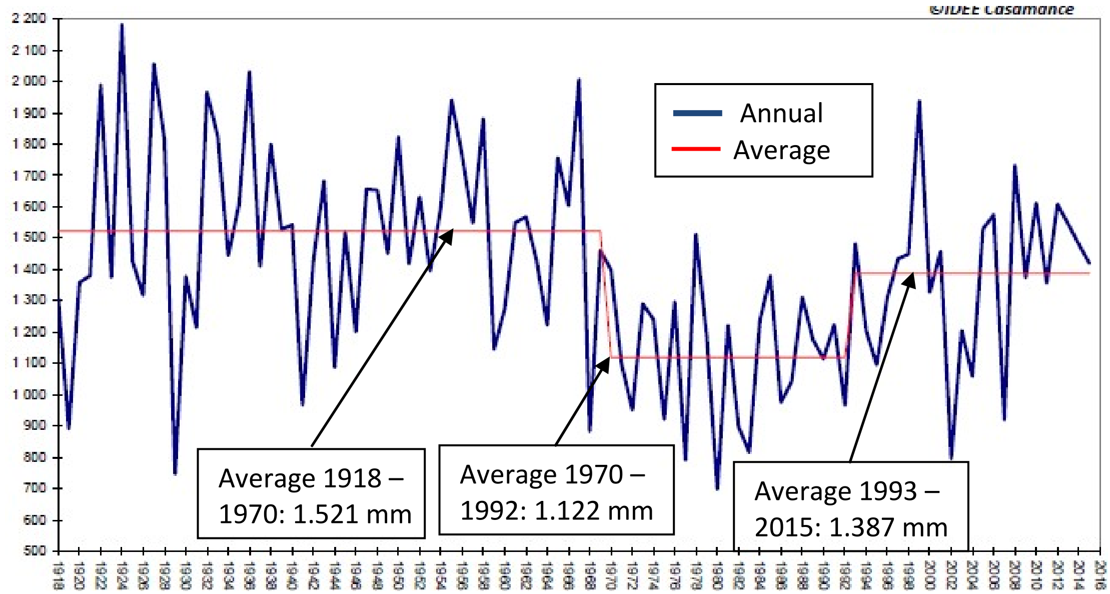

Figure 2: Annual rainfall in Ziguinchor. Source: Own elaboration from IDEE Casamance (2016)

1.2 Socioeconomic Factors

The socioeconomic factors that have led to changes in the area have several origins. On the one hand, energy is a key factor: in 2013, biomass contributed 47% of the supply of energy in Senegal, and 45% of fuel products (Thiam, 2015); energy transition strengthens an urban – rural duality, where the former consumes more charcoal and butane, and the latter more biomass. In 1974, the State of Senegal started the campaign of “butanization” through which the butane gas was subsidized to relieve the pressure on forests areas. The rural areas of Ziguinchor are characterized by an intense use of wood (95%) and charcoal (78%) but less butane gas (31%) (Bodian & Ndiaye, 2010). Moreover, globalization induces an increasing role of the monetary economy, forcing the population to abandon the traditional activities that provide few monetary benefits; rural population migration has decreased the active population on the traditional agricultural sector with effects on the territory, such as the increase in the difficulty of maintaining the earth dikes that protect the rice fields from the saltwater. Institutional policies also have an impact on the territory: institutional projects to build anti-salt dams that finally ended up increasing soil salinity in the area (DIÉDHIOU, 2001) are examples.

1.3 *Consequences Of Gc*

All these factors have resulted on changes in the land use and cover, as well as a severe soil salinization (due to the 1970’s drought, when the estuary became hyper saline (Savenije & Pagès, 1992)). Rainfall variability and the salinization of rice fields have had strong impacts in this area that depends on rain fed agriculture, favouring an abandonment of agricultural areas with direct effects on the food security and vulnerability of the population.

The **objectives** of this study are: i) To identify the processes of GC occurred in the last decades; ii) To quantify land use and cover changes; iii) To evaluate the effects of these changes on soils fertility.

2. METHODOLOGY

In order to develop this study, a socioecologic methodology is proposed, considering: i) Documentary and cartographic research to determine the biophysics and socioeconomic factors that have caused the GC in the region; ii) interviews and questionnaires with local population to understand how they are perceiving the GC; iii) diachronic analysis of land use and cover to quantify the changes; iv) Soil analysis focused on salinity problems to quantify the effects of GC on this process. See Figure 3.

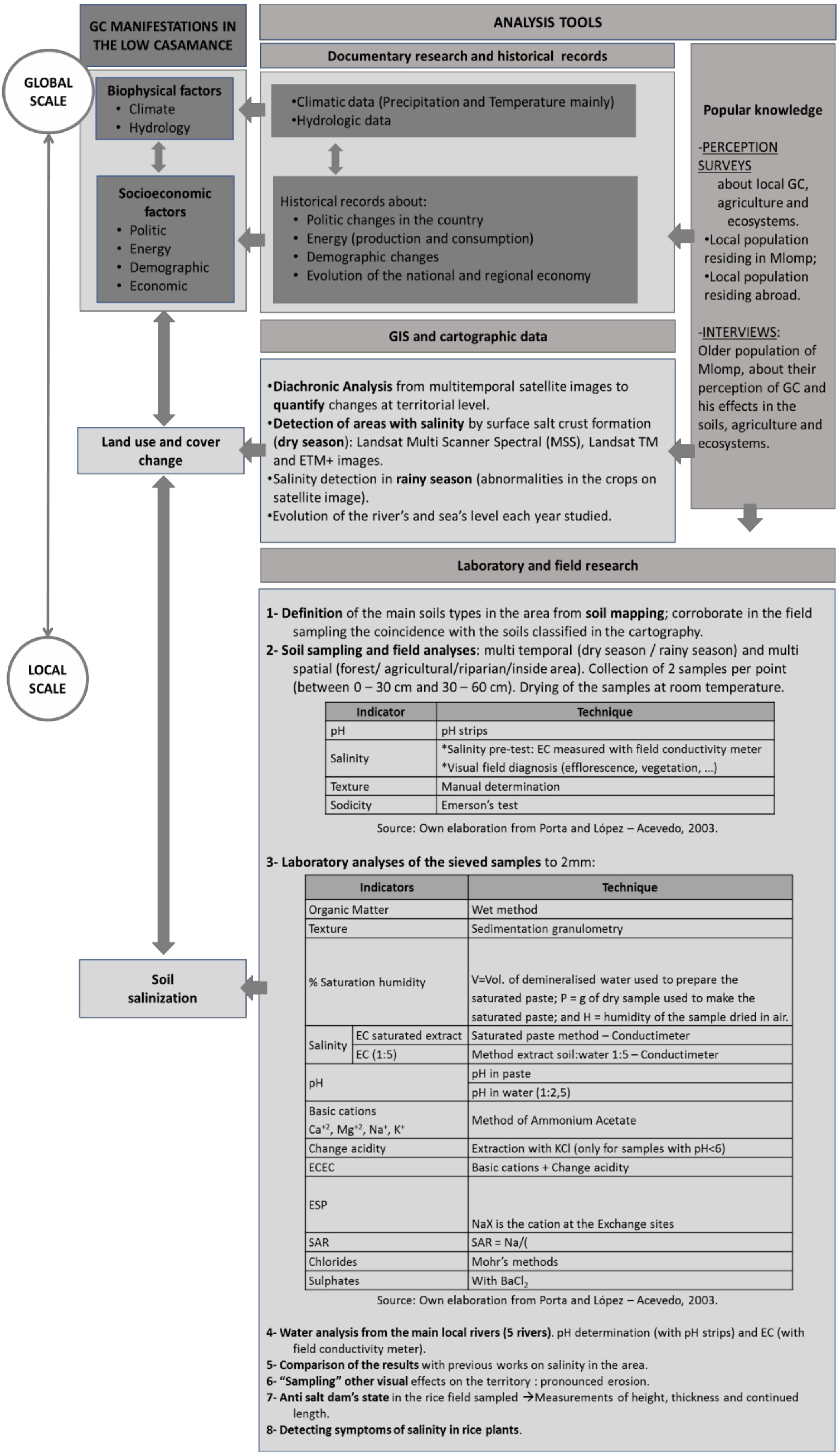

Figure 3: Methodological framework for the analysis of land degradation in the Low Casamance

3. EXPECTED RESULTS AND CONCLUSIONS

By studying diachronic changes in the land uses and covers, we may determine whether the GC effects have produced a land cover transformation (complete replacement of a landscape by another, as the case of abandoned rice fields, or the mangrove affected by salinity) or modification (changes in components of the land cover but without change in the general plant formation, such as the death of certain tree species sensitive to salinity within the same forest type).

Moreover, applying this methodology for determining the level of soil salinization provide an overview of the current salt content of this land and its suitability for growing rice. A comparison of the results with data on salinity obtained by other studies conducted in the area in the past will allow an assessment of the evolution of salinity in the region.

Conclusions: the holistic approach to the study of global change is very important to integrate all the factors that influence the land configuration and the changes that occur therein, and accounting for both biophysical and socioeconomic factors will allow a deep study of all the processes that take place.

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Session VI: Reclamation of degraded soils and waters. Use of amendments

6.1.O

EFFECT OF INORGANIC AND ORGANIC AMENDMENTS ON WATER RETENTION AND YIELD OF WHEAT IN A SANDY SOIL

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INTRODUCTION

The amount of water held in soil at any given time is influenced primarily by soil texture. Factors such as soil type, clay content, organic matter (OM) content and pore size determine soil water retention (ref). Sandy soils have low CEC, poor fertility, low water retention and high potential to allow ground water contamination **(**Dixon, 1991; Reuter, 1994; Hillel, 1998). Sandy soils are the dominant soil type is many parts of the world, so the challenge to meet the need of increasing human population while mitigating effect of climate change call will require better and more sustainable management of sandy soils.

Many researchers have suggested that OM and clay have capacity to improve suitability of sandy soils (Reuter, 1994; Arthur et al., 2011; Wang et al., 2014); however, the beneficial effect of co-application have not been fully explored. Kramer (1983) reported that the benefits of amending sandy soils with OM alone do not persist for longer time in the absence of enough soil clay. Therefore, the aim of the experiment reported here is to investigate the potential of combined application of clay and peat to improve water retention of a sandy soil. A test crop of spring wheat was used to assess the response.

METHOD

Soil preparation

The trial was conducted at the University of Warwick, UK. The soil was sandy loam belonging to a Wick series developed from Triassic rocks (Whitfield, 1973). The soil was collected from plough layer, air-dried and shredded to remove pebbles. The soil contains 65% sand, 18% clay and 17% silt, pH was 6.1 and organic matter was 2.5%.

The inorganic material used as amendments were calcium bentonite (B) and kaolin (K). Their pH was 9.5 and 5.1, respectively. The organic material used was a medium grade, pure sphagnum peat (P) with particle size ranges from 0-14 mm and the pH was 4.2. The treatments consist of three P rates (0, 20 and 30% V/V), two K rate (0 and 5% w/w) and three B rates (0, 2.5%, and 5% w/w).

Field trials were undertaken using 60 microplots (concrete sleeves 60cm in diameter, sunk 100 cm into the ground), each having between rows and inter row spacing of 1m. The bottom 20 cm was filled with gravel layer to ensure proper water drainage. Three hundred kg air-dried, homogenized soil was treated with equivalent soil amendment, and filled into the lysimeter. A free area of about 10 cm was left on the surface of lysimeters to prevent soil loss through splashing during rainfall.

Sampling and Analysis

Soil moisture content (MC) was monitored using direct soil sampling and soil moisture probes for 12 months. All lysimeters were sampled monthly for gravimetric soil moisture content. Soils were sampled randomly using 1cm diameter soil auger. Subsamples are weighed and oven dried at 105^{0}C

overnight. Soil MC was determined as the difference between the weight of wet and oven dried soil, and reported as percentage of oven dried soil. Data from logger was used as back up.
Wheat was sown in April 2016 and grown to maturity. Phosphate and potassium fertilizer were applied into the soil at the rate of 70kg and 60 kg /ha, respectively, before sowing. Nitrogen was applied as surface dressing using ammonium nitrate at the rate of 160 kg N /ha, split across two applications. At GS51, 10 plants were randomly selected per plot. The roots were cut off from the crown, and the shoot was weighed. Data was presented as percentage increase over that of control. Wheat ear was harvested in August 2016. All the results are means of five replicates. Data were analysed using ANOVA and means separated using Tukey HSD at $p \leq 0.05$. Correlation analysis was performed using Excel.

RESULTS

Soil moisture content

Amendment increased water retention of the test soil. P application and rate increased soil MC; and 30%P was significantly higher than 20%P and soil only (Table 1). Clay K reduced soil MC compared to the control but the difference was not significant. Water retention of 2.5%B also was not significantly different from soil only. Among the clay alone treatments, only 5%B was significant. Weight for weight, water retention of clay B was significantly higher than K. Effect of clay rate was observed in clay B in that higher clay rate retained more water. Interaction between clay and P was significant ($p \leq 0.05$) in respect to soil moisture.

Table 1: Average soil moisture content and wheat fresh biomass yield in sandy soil amended with clay and peat

Treatment	Average ear weight (g/m^2)	Average soil moisture content for 12 months (%)
Soil only	243.6c	12.19 (0.174)
20%P	314.6ab	13.69 (0.223)
30%P	346.8a	15.11 (0.142)
5%K	282.0bc	12.27 (0.169)
5%K+20%P	318.2ab	14.54 (0.511)
5%K+30%P	281.0bc	15.23 (0.265)
2.5%B	265.2bc	12.73 (0.116)
2.5%B+20%P	304.0abc	15.60 (0.460)
2.5%B+30%P	279.8bc	16.54 (0.162)
5%B	258.4bc	13.88 (0.197)
5%B+20%P	283.0bc	15.97 (0.164)
5%B+30%P	312.2ab	17.79 (0.337)

Figures with similar letter in the same column are not significantly different ($p \leq 0.05$; n = 5).
Standard Error (S.E.) in bracket.

Combined application of clay and P increased soil moisture content compared to clay alone.
For each clay type/rate, the increase was higher at 30% than 20%. Among the treatment, 5%B+30%P was significantly higher than the rest except 2.5%B+30%P (Table 1). Synergy was observed for combined application of clay and P in all the clay regardless of their type or rate.
Observation showed changes in soil MC retention pattern over season. Highest MC was recorded in April while July had the least. Clay B in the presence or absent of P, had higher water retention both during April and July period. Where soils were amended with just peat, 30%P had the highest water retention throughout (Figure 1). In 5%K amended soil, 30%P had highest water retention in wet period, however, as the soil became drier in July, water retention of 20%P treated soils became

higher than 30%P (Figure 1). This trend suggests that soil amended with K+20%P would hold more water in drier season than those receiving higher volume of P, possibly due to increased porosity at higher volume. Similar observation was seen in P only treatment in July where the differences in soil MC between the two P rates diminished greatly during the driest month (Figure 1). In clay B amended soil, the data predicts higher retention in 5%B amended soil compared to 2.5%B+P in both July and April, suggesting an increase availability of soil water as clay rate increases.

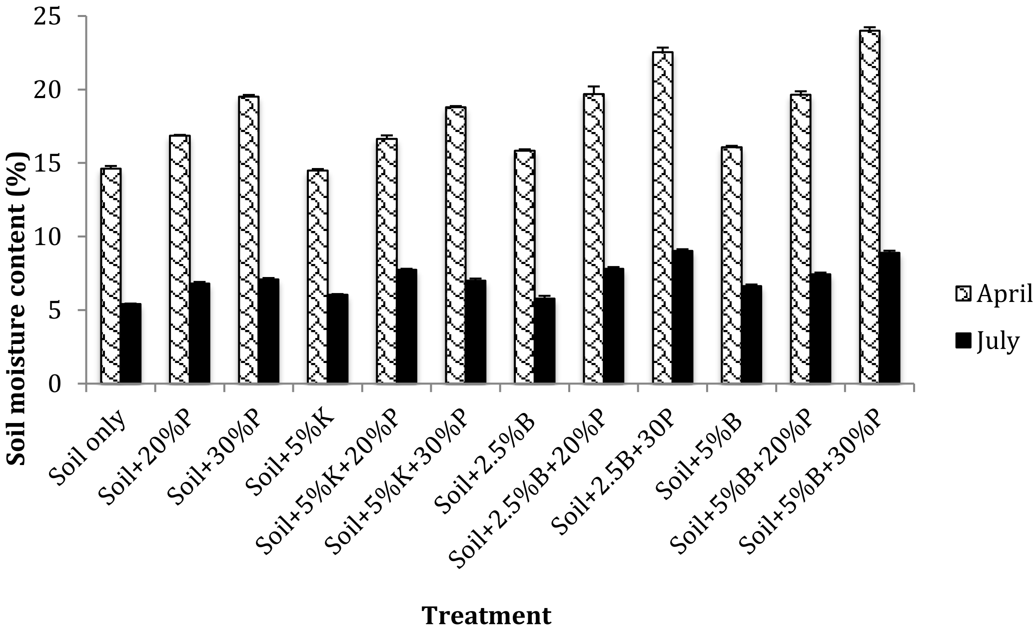

Figure 1: Seasonal variability in soil moisture content. Highest water retention was recorded in April and lowest values in July 2016.

Wheat fresh shoot weight and ear weight

All amendments increased ear weight over that of soil only, but only 20%P, 30%P, 5%K+20%P and 5%B+30%K were significantly higher than control (Table 1). Among the treatments, 30%P had highest ear weight. At the same application rate, soil treated with P only had higher ear weight than clay-peat treated soil except 5%K+20%P, suggesting yield suppression when clay and P was co-applied. Similar result was observed for fresh shoot weight (Figure 2). This is possibly due to factors such as additional nutrient release from P decomposition and reduced mineralization rate in clay amended soils. The latter could support sequestration of carbon and increase microbial activity in the long run.

A comparison between amended and unamended soils showed that the former resulted in greater plant biomass. The increase ranges from 21 percent in soil amended with 5%K only to 91 percent in 5%K+30%P (Figure 2), suggesting improved vegetative growth in amended soils. However, in 2.5%B amended soils, the trend predicted reduction in fresh shoot weight as P rate increased unlike at 5% clay rate.

The relationship between soil water retention and wheat fresh shoot weight was examined using correlation. There was positive correlation between soil water retention and biomass yield (r^2 = 0.70, n = 60), suggesting higher yield as soil water increases.

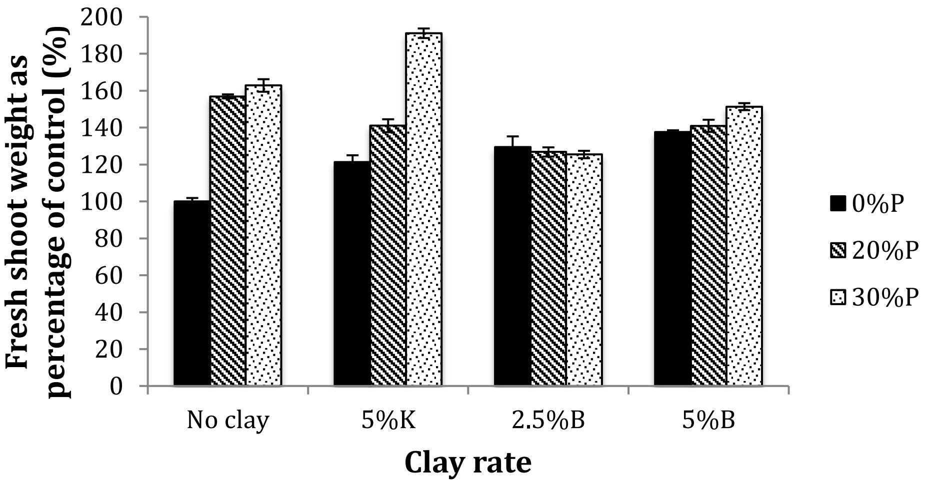

Figure 2: Fresh shoot weight in different clay amended soil in response to P rate (n = 5)

CONCLUSIONS

Amending sandy soil with clay and OM increased water retention and wheat yield compared to a control. Bentonite retained more water than kaolin, whether applied alone or with OM, but that did not necessarily increase yield. During wet period, response of both clay to moisture retention as OM increases were similar, however, during dry months, soil amended with kaolin and 20%P retained more water than at 30%P. Co-application of clay and P improved water retention over clay alone and P, but supressed wheat yield. Soil amended with kaolin appeared to support higher wheat yield than bentonite except for total ear weight when applied with 30%P. In all, application of P on its own and combined application of clay and P at the rate $\geq 20\%$P increased performances of the test sandy soil. The results demonstrated that under field condition, amending sandy soil with clay and OM could improve its moisture retention and wheat yield.

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6.3.O

EFFECTIVENESS OF COMBINED APPLICATIONS OF CLAY AND ORGANIC MATERIALS ON THE HYDROLOGY OF A SANDY LOAM USING RAINFALL SIMULATION.

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1. INTRODUCTION

The relatively large particle sizes of sandy soils (0.06 – 2.0 mm) create macro- and mesopores which generate high rates of infiltration following rainfall or irrigation (Massoud, 1975). On agricultural lands, high infiltration rates can cause loss of water and nutrients from the soil profile, which may adversely affect root uptake and subsequent crop growth. To mitigate these stresses on crops, irrigation and / or fertiliser applications may be need to be increased in both frequency and magnitude to produce a viable crop. However, these practices can lead to the leaching of excessive nutrients into groundwaters, leading to environmental pollution (Reuter, 1994) and significant water treatment costs.

One alternative practice is to alter soil properties to increase soil water holding capacity and reduce infiltration rates. This project seeks to measure the effect of additions of clay and organic matter, both separately and in combination, on the hydrological properties of a sandy soil using laboratory-based rainfall simulation techniques. Rainfall simulators are valuable research tools as they reduce the variability and uncertainty of natural rainfall. Vadas et al. (2007) reported similarity between rainfall studies conducted in the laboratory and the field to justify the use of this technique in predicting real life conditions. Although many previous studies have focused on the potential of clay or organic matter amendments to improve soil retention of water and /or nutrients, few have considered the consequences of this practice with regard to potential runoff generation and the susceptibility of soil to erosion (erodibility). Therefore the objective of the current study is to investigate the effect of clay and organic matter amendments on infiltration and runoff from a sandy soil for different storm durations (15 minutes and 30 minutes).

2. MATERIALS AND METHOD

2.1. Soil collection and preparation

Surface horizon soil was collected (0-20cm deep) from the University of Warwick, Wellesbourne experimental field, Warwickshire, United Kingdom. The soil was air dried, homogenised and sieved through a 10mm screen. The soil was identified as a typical brown earth, belonging to the Wick series (Whitfield, 1973). The soil contains 65% sand, 18% clay and 17% silt. Its pH was 6.1 and organic matter content was 2.5%.

2.2. Soil amendments

The soil was amended with two types of clay, and an organic material. The clays were kaolin (K) and bentonite (B) representing 1:1 and 2:1 clay minerals respectively. The organic material used was a medium grade, pure sphagnum peat (P), with a pH of 4.2. The treatment combinations consisted of three P rates (0, 20% and 30% v/v) and three clay rates (0, 2.5% and 5% w/w). Each treatment had four replicates.

2.3. Soil packing

Soil erosion trays (20cm x 11cm x 5cm) were used. Each tray was lined with a perforated metal mesh, covered with a layer of fine cloth to prevent soil from washing out, whilst providing free drainage. Each tray was fitted with a funnel at its downslope edge, which collected surface runoff that then discharged into a plastic container via a 30mm diameter pipe. Any infiltrate was collected via a 10mm diameter pipe at the bottom of the tray. All treatments were packed to the same volume. For the clays (K and B), treatments with the same application rate were packed to the same density. All samples were saturated and allowed to drain to field capacity. The trays were placed on a sloping table (15%) and placed under the rainfall simulator.

2.4. Rainfall simulation

The gravity-fed rainfall tower at the Soil Management Facility, Cranfield University, UK was used. Raindrops were generated by ponding a constant head of water above an array of hypodermic needles. Raindrop size was randomised by letting the drops fall through a metal mesh located 1 m below the hypodermic needles. Total drop fall height was c. 8.8 m, ensuring over 95% of drops reach terminal velocity. Following calibration, target areas with similar rainfall intensity were identified. The rainfall intensity was approximately 65 mm/hr. The soil trays were placed in the marked position on the rainfall table. Two storm durations were used: 15 and 30 minutes. After each storm event, infiltration and runoff volume were measured.

3. RESULTS

Data were analysed using ANOVA and means separated using HSD at $p \leq 0.05$. Correlation analysis was performed using Excel.

3.1. Infiltration

The hypothesis is that the increase of clay and organic matter in the sandy soil would reduce the infiltration rate by reducing pore sizes. The results support this. In soils treated with P only, infiltration reduced as P rate increased (Fig 1), both after 15 minutes and 30 minutes of rainfall application. However, only 30% P at 15 minutes was significantly different from the unamended soil, suggesting that the reduced infiltration in soil treated with 30%P up to 15 minutes might be due to water retention capacity of the amendment at this higher rate. A similar reduction in infiltration was not seen for the 20% application rate. After 15 minutes of rainfall, the 30%P treatment reached its maximum water holding capacity so that more water was able to infiltrate. This suggests an increase in infiltration occurs over time as the capacity of the organic material to hold water reduces.

Both K and B clays significantly reduced infiltration volume compared to the unamended soil, with the exception of 2.5%K after 15 minutes. The reduction in infiltration volume increased with clay rate, as expected. The difference in infiltration volume between the two K rates (2.5% and 5%) was significant ($p \leq 0.05$, standard error (SE) = 20), while the difference between the two B rates was not. Comparing the two clay types, at 2.5%, B reduced infiltration more than K, and vice versa at 5%. The reduced infiltration in the clay amended soils is likely associated with the ability of the clays to reduce soil pores sizes compared with the unamended sandy soil.

Combined applications of clay and P reduced infiltration volume compared to the unamended soil. The mean differences were significant (except 5%K+20%P at 30 minutes) at 15 and 30 minutes for both clays. At 2.5% clay, infiltration reduced as P rate increased, while at 5% clay, infiltration increased with P rate (Fig 1), possibly due to increased porosity. When applied with clay, infiltration reduced in P amended soil compared to P only.

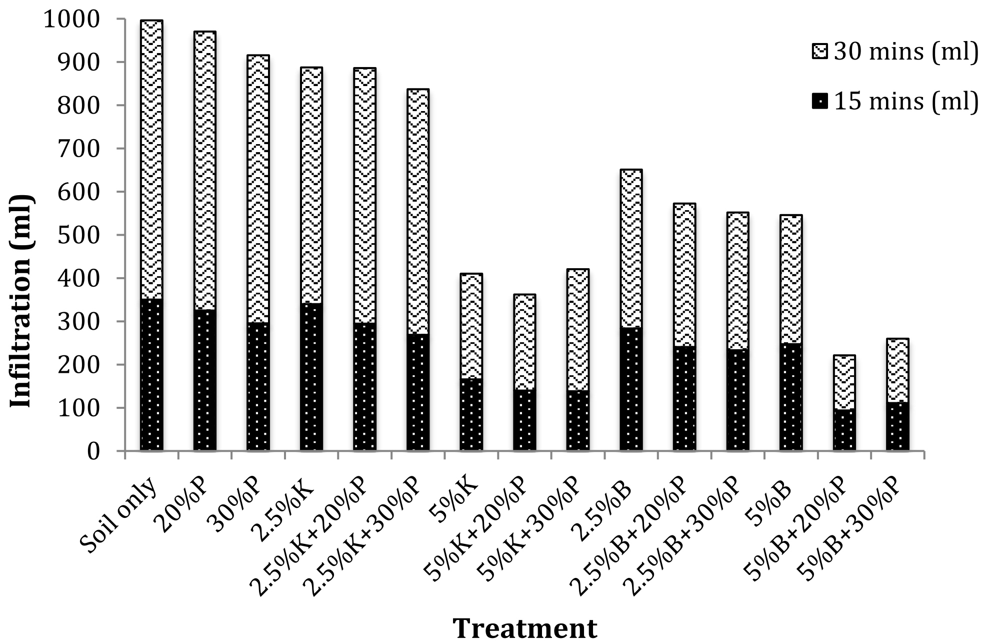

Figure 1. Infiltration volume as affected by clay type and organic matter amendments

3.2. Runoff

Figure 2 shows the response of runoff to clay (type and application rate) and P additions. The unamended soil had the lowest runoff (5.3 ml). Where P was added by itself (i.e. no clay added), runoff was similar to that of the unamended soil for the first 15 minutes. However, as rainfall duration increased to 30 minutes, addition of P reduced runoff by up to 71%, compared to the unamended soil, showing the potential of P in reducing runoff in sandy soils.

For the clay amended soils at low application rates of 2.5%, the response with K was similar to that of the unamended soil as P increased, except for 30%P after 30 minutes. For 2.5%B, runoff increased with P rate over the two rainfall durations. At the 5% clay rate, within the first 15 minutes, the response of the two clays was similar, having higher runoff at 20%P, but reduced as P rate increased to 30%. The results after 30 minutes followed similar patterns (Figs 2 and 3.). When combined with P applications, there was more runoff from the soil amended with B than with K. The increased runoff in soil amended with clay is associated with a number of factors such as reduced infiltration arising from a reduction in soil pore sizes. When wet, clay particles absorb water and expand, which will further reduce soil pore size; this could explain the increase in runoff as rainfall duration increases and the soils get wetter.

4. CONCLUSION

Adding clay and P to a sandy soil reduced water infiltration compared to an unamended control. Weight for weight, B has lower infiltration at 2.5%, while K has lower infiltration at 5%. Combined application of clay and P further reduced infiltration. Runoff volume was correlated with rate of clay application. Over time, runoff reduced as P rate increased.

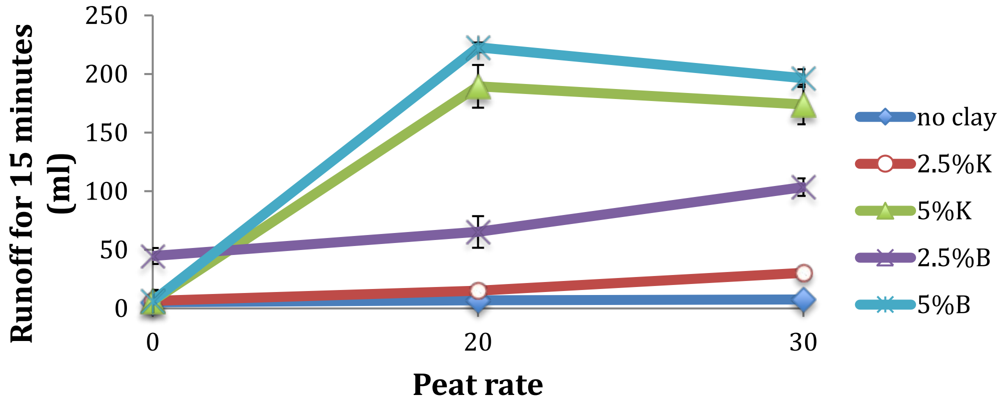

Figure 2. Runoff after 15 minutes as a function of clay application rate and type, and peat application rate

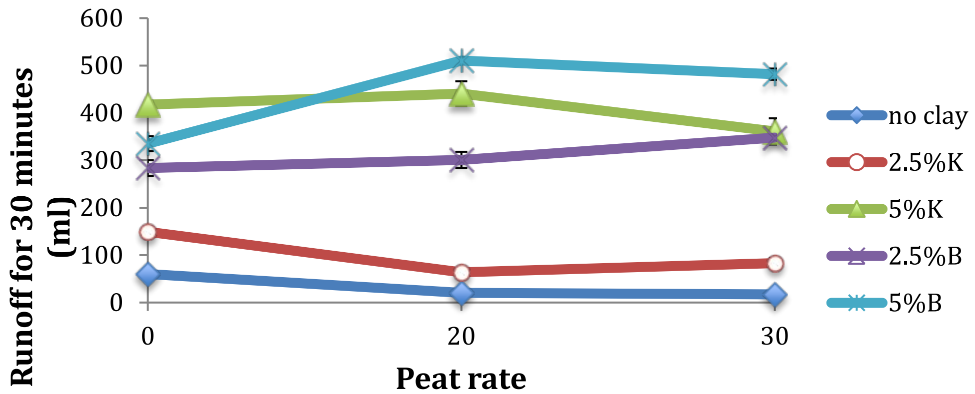

Figure 3 Runoff after 30 minutes as a function of clay application rate and type, and peat application rate

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6.5.O

TOWARDS SUSTAINABILITY IN ARID LANDS

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ABSTRACT
Facing desertification is a pregnant priority for humankind.

Since the early 30's its spreading mechanism has been unveiled by Pr. Henri Erhart[1].
When considering the Bio-Rhexistasy cycle, it is noticeable that a strong unbalance prevails between a slow colonisation rate of soil creating vegetation, and abrupt reversing conditions of soil erosion when the vegetation hold is lost.
As illustrated in Figure 1, Africa is perceived as the most vulnerable ground for desert expansion.

When seeking for remediation in reforesting and re-cultivating arid lands, the crucial question is how to restore the water retention capacity of surface formations, essential for plants to survive. In most desert landscapes very altered soil conditions prevail; in fact former elaborated soils have generally been washed away by scarce torrential rains, leaving lateritic cuirass or barren rock grounds.

The basics of the PLAN.T.E project is to remediate the consequent lack of water retention capacity by creating local systems of small reservoirs able to collect a significant part of the impluvium and retain shallow depth water out of reach of intense evaporation, but still available for plants.

PLAN.T.E is an imperative that can be read as "**PLAN**t **T**rees with **E**xplosives".

The method is known as "cratering" and allows to produce from appropriate blasting in 4-6 m deep drilled holes a series of cones of crushed rock with a high fracture porosity. When applied to impervious rocks like claystone, marl or shale, individual reservoirs with up to 10 m^3 capacity are made available. Success in planting trees of selected resilient stocks in these locations is then to be achieved through appropriate fertilisation with organic compost and adapted watering till self-sufficiency, when roots gain access to the eventual water reserve .

The method is thought to provide to most of the fragile ecosystems, that currently vanish in arid regions due to the double penalty of adverse climatic drift and increasing human pressure, the necessary impulse for a medium term recovery and long term prosperity.

Besides the technical presentation of the method, our contribution mostly focuses on the framing conditions of the project PLAN.T.E, through specific studies and partnerships that back the first projected developments and illustrate our collaborative strategy to rally expertise and funding.

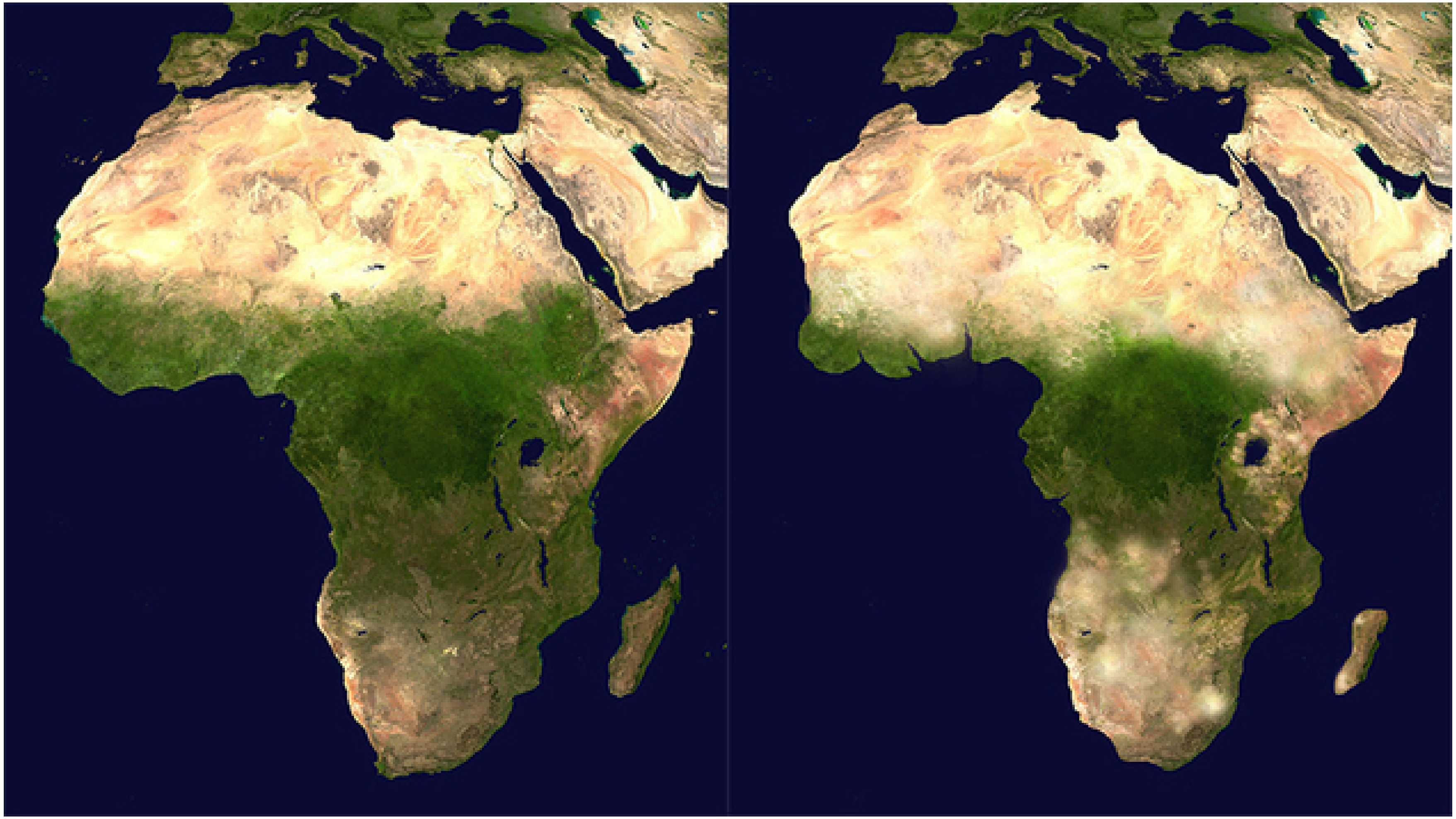


Figure 1 : Africa within 1 000 years (or less ?)   An illustration of probable desertification progress

Source : Data from UN, GRID-Arendal, PreventionWeb, UN Convention to Combat Desertification

THE PLAN.T.E PROJECT
***
A STEP TOWARDS A BETTER RESILIENCE OF SAHELIAN POPULATIONS FACING THE DESERTIFICATION PROCESS

Pascal Bernasconi (Links Consultants), Emilio Neto & Patrick Pierron (GéO-CSP SAS)

INTRODUCTION

The PLAN.T.E project is currently supported by GéO-CSP, an independent consultancy in Applied Geology with significant experience in the mining sector as well as in cement and construction material industries.

GéO-CSP provides customized services to match its institutional and private client's needs in project framing and development, through a very proactive network of small to medium sized engineering and consulting partners, focused on sustainable solutions and industrial integration. The activity includes site rehabilitation and waste management in order to meet the highest criteria of environmental compliance in a realistic approach of project integration and sustainability.

So, the PLAN.T.E project appears as a natural extension of our field experience based on the realization of mining and environmental audits carried out in the Sahelian region. These actions were accompanied by a reflexion on the process of desertification.

The PLAN.T.E project presented here is the result of our commitment to contribute to the emergence of efficient processes and techniques in line with a logic of sustainable development.

CONTEXT AND OBJECTIVES

The "post-Rio" Convention to Combat Desertification (CCD) refers to desertification as "land degradation in arid, semi-arid and dry sub-humid areas as a result of various factors, including climatic variations and human activities ". This serious phenomenon affects nearly 40% of the land area and affects one-third of humankind; inaction against this plague would cause 10 million hectares of agricultural land to vanish every year, equivalent to 1/5th of the area of Spain.

The objective of the PLAN.TE project is to impulse a new dynamic of land rehabilitation in the regions affected by desertification through the reforestation of pilot zones, now firstly located in southern Tunisia (Tataouine) and Morocco (Merzouga), using methods inspired by the mining industry.

Concretely, it means creating locally a system of small reservoirs capable of collecting a large part of the impluvium and of retaining water at depth, protected from evaporation. This step is essential for restoring a significant soil cover ratio through planting suitable ligneous species and then facilitate the regeneration of ecosystems favouring agricultural development and reinforcing the resilience of local populations to environmental changes.

Finally, the specificity of the project lies in the use of explosives, as in the case of mining, which requires the implementation of special safety and security measures.

THE PLAN.T.E PROJECT: from concept to operations
An innovative planting method

The most practical and effective way to reach the aforementioned objectives is to use the so-called cratering technique, as illustrated in the following diagram:

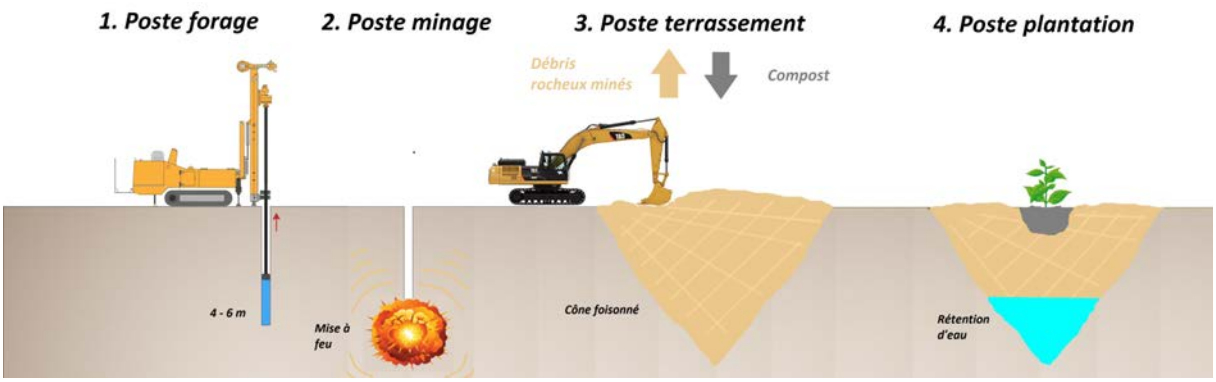


Figure 2 : Illustrative diagram of the "cratering" method.

1- Use of a hammer drill for the realisation of short drillings from 4 to 6 m deep
2- Use of ANFO explosives (ammonium nitrate-Fuel). Under ideal conditions (94.7% NH4 NO3 and 5.3% diesel) the reaction essentially releases water (H2O), Nitrogen (N2) and minor Carbon Dioxide (CO2):
This excludes any danger of soil contamination.
3 – Minor excavation works to host compost and tree to be planted. Excavated rock can be used as dykes for run off collection. A composting station for organic waste will provide the fertilizer needed to start planting.
4 - The final planting operation of adapted trees will be accompanied by planting of non-invasive and non-toxic protective plants. Planting will require limited watering until the first significant rains. So coupling with a temporary irrigation system is required.

The application of the PLAN.TE method is relatively easy, however it requires a precise approach in terms of choice of target areas, mobilization of specific teams and equipment, safety and security in the use of explosives, also acceptance and participation of the communities concerned. The financing of such works, followed by the necessary monitoring and maintenance of the plantations, is also a sensitive issue, in particular because of the scarce economic resources of the potential beneficiaries.

A threefold strategy

1 / Carry out two pilot projects in two target areas of which we already have a good knowledge, particularly through local contacts favourable to a partnership. These are the aforementioned regions of Merzouga-Hassilabied and Tataouine-Douiret.

2 / Create, in synergy with these pilots actions, a cross-cultural component, further presented as "DESERT FESTIVAL" which will potentiate and support our approach, and also promotes our action by expanding awareness through media coverage and also mobilise local support.

3 / Form local teams to spread over targets more difficult to access, especially in Algeria, then expand operations in the countries of the Sahel belt, Senegal first, then Burkina Faso, Mali, Mauritania, Niger and Chad if possible.

Our scope for project development

In accordance with the common concept of industrial greenfield project development the following steps are observed :

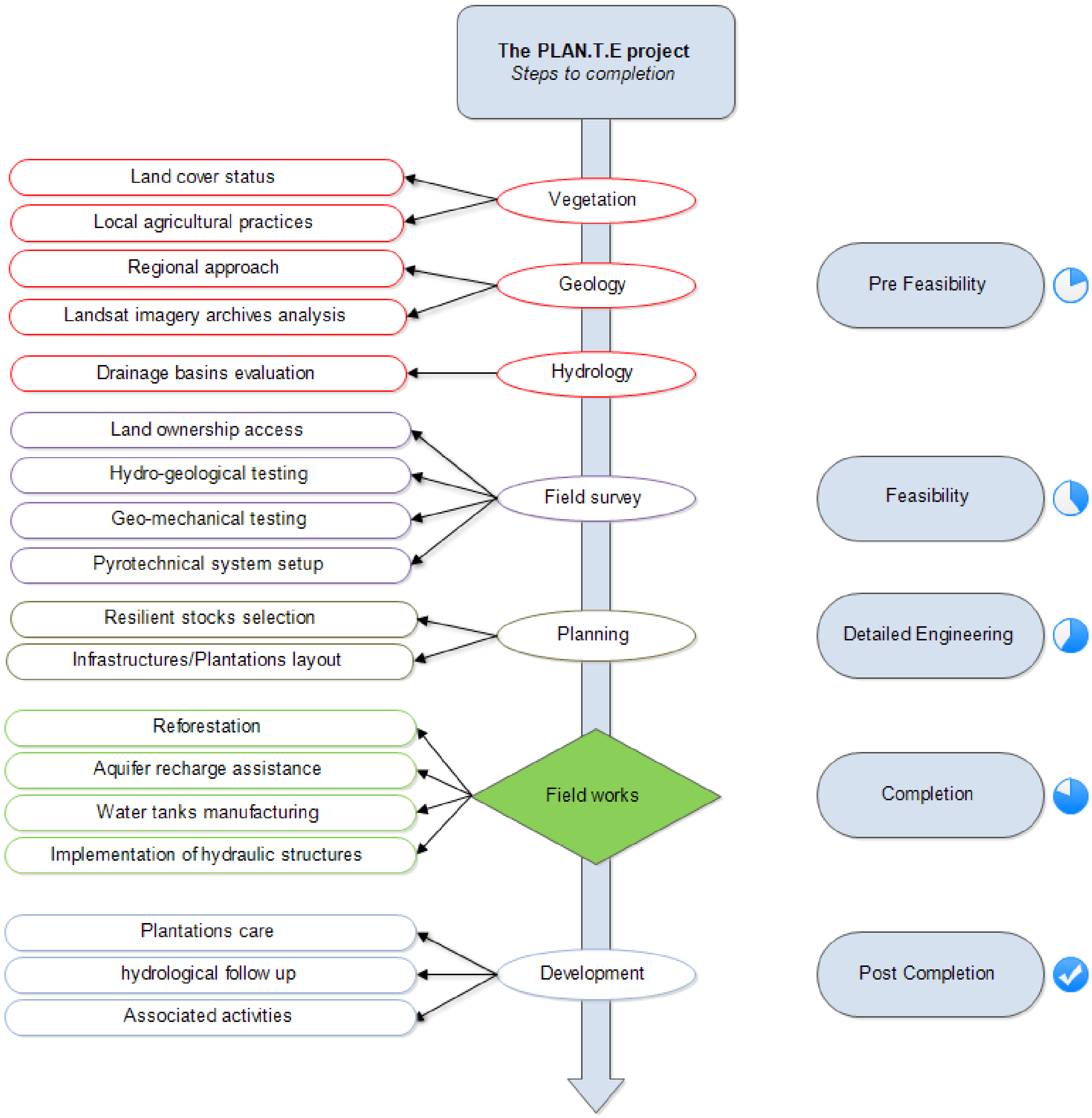


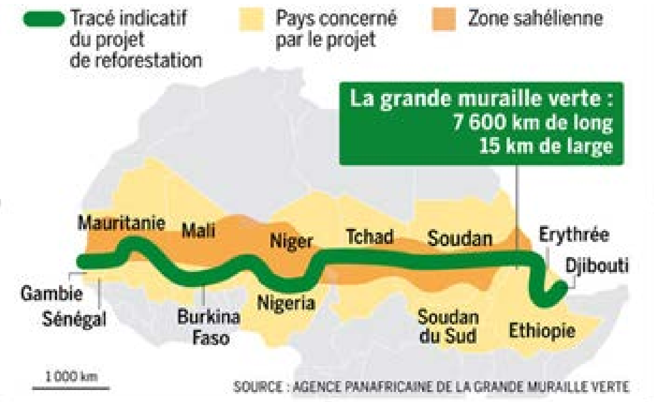
Figure 3 : Logigram of the PLAN.T.E Project development

The initial implementation of the PLAN.T.E method is now being planned in Tunisia, where the MARHP (Ministry of Agriculture, Hydraulic Resources and Fisheries) and IRA (The Arid Regions Institute) are incorporating it to their current projects with three main targets :
- the assistance to surface aquifers recharge, through blasting enhanced permeability of flooding racks,
- the creation of ponds and cisterns in impervious rock sites, in gypsum as an instance,
- the simple reforestation in rocky areas with combined simple water capture systems.

For each of these targets, a sheet describing the project draft, the means to be implemented, and the corresponding preliminary budget calculation will be presented at CONSOWA.

Our perspectives and communication plan

The PLAN.T.E Project is conceived as collaborative and therefore open to all scientific and practical contribution in form of either expertise or grants. It is developed using the collaborative project management platform **Atikteam** sponsored by Demotera [2].
This approach ensures rapid access to the bulk of information and enables better tracking of project progress by partners and stakeholders; it is a guarantee of a transparent and effective collaboration. Thus any new entrant can very quickly access the main part of the project and put himself in situation to be able to collaborate effectively in his field.



Usage makes Atikteam the instrument of a remarkable increase in collective effectiveness .

Operation wise, the pacing to be considered for the PLAN.T.E Project is dictated by the drilling capacities. Our target is 60 000 m/year for a standard drilling equipment, meaning 200 holes per week at 6.0m depth, or potentially 100 hectares per year, planted along a 10 m x 10 m grid.

The PLAN.T.E project may naturally be part of the Green Wall Project that should ultimately cover a 15 km wide strip over 7 600 km of east-west extent in Sahel, meaning more than 10 million hectares.

Figure 4 : The Great Green Wall Project

Assuming that the PLAN.T.E method is applied to 10% of this surface, which would have the effect of an initiator of certain beneficial character, it would be necessary to mobilize 1000 workshop units to achieve this goal in 10 years. This would create about 50 000 direct jobs and would require some 3Mt of explosives. These figures are important, but remain low compared to the estimated cost of the impacts of desertification (over $ 40 billion / year in the World).

To eventually plant 100 million trees, the yearly PLAN.T.E budget would be less than $ 1 billion.

Our communication plan, detailing the “DESERT FESTIVAL” issues will also briefly be presented at CONSOWA.

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2 http://www.atikteam.com/en/demotera

6.6.O

SOIL INOCULATION WITH CYANOBACTERIA AS A PROMISING TECHNIQUE TO COMBAT DEGRADATION PROCESSES IN DRYLANDS

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ABSTRACT

Biocrusts (communities of cyanobacteria, algae, lichens and mosses in association with soil particles) are considered a key component of dryland soils due to their important role increasing soil fertility, and water retention and their protective effect against water and wind erosion. All these functions joined to their ability to survive in dry environments where water is the most limiting factor for plant establishment, convert biocrusts on ideal candidates as restoration agents to promote soil recovery in degraded drylands. Concretely, soil inoculation with cyanobacteria is one of the most promising biotechnological strategies due to the possibility of growing these organisms under laboratory conditions and their capacity to survive under harsh environmental conditions. The objective of this study was to analyse the effect of three native N-fixing cyanobacteria strains, inoculated individually and as a mixture, on the development of a new biocrust and on soil properties. To do this, the selected strains were inoculated on soils collected from three semiarid areas in the province of Almeria. Preliminary results show that cyanobacteria-inoculated soils show higher CO_2 uptake, organic carbon content and spectral absorptions by photosynthetic pigments than control soils. However, this effect greatly depended on the inoculated strain. Our results point to the feasibility of using individual cyanobacteria or a consortium of cyanobacteria species as a novel and viable restoration approach to combat degradation problems in drylands.

INTRODUCTION

Increasing world's population, land use changes and impacts of climate change are important threatens to the conservation and stability of ecosystems in the next decades. Drylands are especially vulnerable to induced land degradation processes, which lead to losses of soil fertility, modification of carbon and nutrient cycling and a decrease in biodiversity and vegetation productivity. Combating land degradation in drylands is a challenge due to the intrinsic peculiarities of these areas, characterized by limited water availability, low vegetation cover and high erosion rates. In these regions, biocrusts (communities of cyanobacteria, algae, lichens and mosses in association with soil particles) are considered a key soil component that have multiple functions regulating hydrological, geomorphological, and biogeochemical processes. They are able to survive in extreme arid environments and are activated even after erratic rainfalls or dew events, promoting C and N fixation. As a result of their metabolic activity, biocrusts contribute to soil stabilization, soil fertility and water retention, thus improving the surrounding environment for other soil biota and annual and vascular plants. Due to all these beneficial effects, a number of studies have claimed the use of biocrusts for restoration in water-limited environments and some of them have proved their success in soil recovery (Antoninka et al., 2016). Cyanobacteria are envisaged as ideal candidates for this purpose due to the possibility of growing them under laboratory conditions and their remarkable adaptation to combinations of stressful environmental conditions such as extremes of temperature, UV, irradiance, drought, salinity and short wetting cycles. However, knowledge about the most suitable species and inoculation strategies is necessary previous to their sucessful application in the field. The main goal of this study was to analyse the effect of three native N-fixing cyanobacteria, inoculated individually and in combination, on key soil properties related to water availability and fertility on different soil types collected from semiarid areas in the province of Almeria (SE Spain).

MATERIAL AND METHODS

Samples of cyanobacteria crusts were collected from three different sites in the province of Almeria (SE, Spain), where biocrusts are one of the main surface components: a limestone quarry located in the Gádor range, with soils having a silty texture, poor structure and very low organic matter content; the Tabernas badlands area with soils having also a silty loam texture and poor structure and low organic matter content; and Las Amoladeras (Cabo de Gata Natural Park), a flat grassland with sandy loam soils and relatively higher organic carbon content. Soil native cyanobacteria belonging to three representative N-fixing genera (*Nostoc*, *Scytonema* and *Tolypothrix*) were isolated from such soils and cultured in $BG11_0$ medium. Each strain was inoculated (6 g m^{-2}), individually and in combination (each in the same proportion), on Petri dishes with 80 g of each soil type. Samples were maintained in the laboratory under a constant temperature of 28ºC and a light intensity of 60 µmol photons $m^{-2} s^{-1}$, during three months. During the experiment, two irrigation treatments were applied simulating a dry and a wet hydrological year in the study sites. After 3 months, we measured surface reflectance of the samples with a ASD spectroradiometer and based on this, we determined surface albedo and spectral absorptions by photosynthesic pigments as proxies of biocrust development. Net CO_2 flux was measured with an infrared gas analyzer coupled to a transparent methacrylate chamber with a volume of 668 cm^3 (Ladrón de Guevara et al., 2015). After these measurements were conducted, the surface biocrust was collected and organic carbon (OC) content was determined using the Walkley and Black method modified by Mingorance et al. (2007). Significant differences between inoculation treatments were analysed using one way ANOVA and the Fisher post hoc test.

RESULTS AND DISCUSSION

Results show significant differences in spectral properties, net CO_2 flux and OC content among the different treatments. The inoculated samples showed lower reflectance (Figure 1a) and deeper spectral absorptions by carotenoids (around 500 nm) and chlorophyll *a* (680 nm) (Figure 1b) compared to the control samples. Decreased albedo and increased absorption peaks due to higher pigment content are usually used as indicators of biocrust development (Chamizo et al., 2012). Samples inoculated with *Nostoc* and the mixture of the three strains showed the lowest reflectance and the deepest absorption peaks, thus indicating a greater biocrust development compared to the other cyanobacteria treatments.

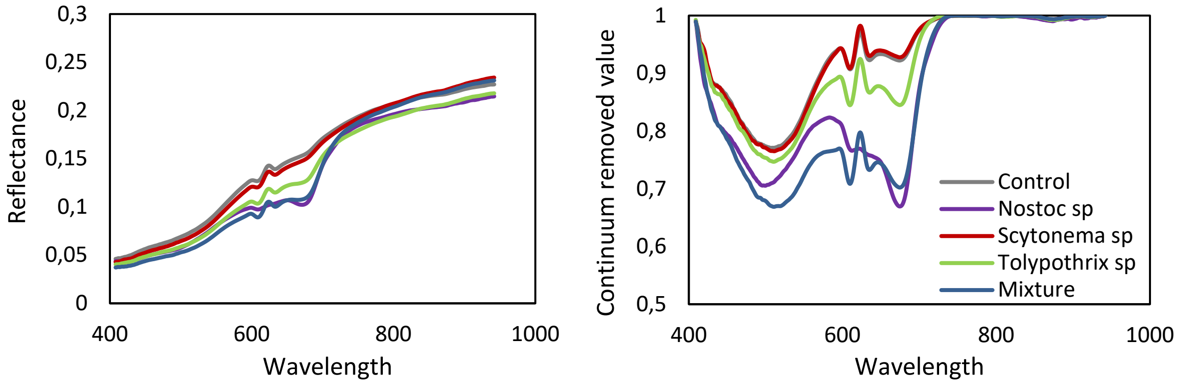

Figure 1. Mean reflectance (a) and continuum removed spectra (value of absorption peaks) (b) in the control and the different inoculation treatments in the sandy loam soils from Las Amoladeras (wet irrigation treatment).

Due to the development of an artificial biocrust, the inoculated samples showed positive values of net CO_2 flux, indicating a net CO_2 uptake, whereas control soils showed CO_2 fluxes closed to zero (Figure 2). No significant differences (p>0.05) were found between the dry and wet irrigation treatments.

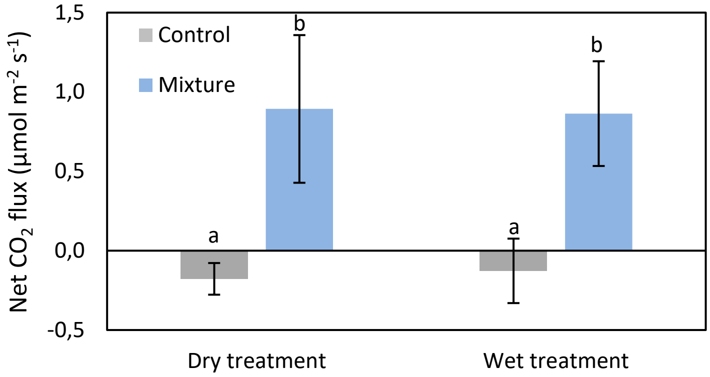

Figure 2. Net CO_2 flux (mean ± sd) measured in the control and the mixture-inoculated samples in the silty loam soils from El Cautivo. Different letters indicate significant differences (p<0.05) between the control and inoculated samples within each irrigation treatment.

The higher CO_2 fixation in the inoculated samples resulted in higher OC content in these samples compared to the non-inoculated ones. From the different treatments, samples inoculated with *Nostoc* individually and the combination of the three cyanobacteria strains exhibited the highest OC content (Figure 3). This pattern was observed in the three types of soils tested. Thus, an important property affected by cyanobateria inoculation is the soil organic carbon content, which is significantly increased by the photosynthetic activity of the induced biocrust and the synthesis of exopolysaccharides compounds, which greatly contribute to increasing soil fertility (Xu et al., 2013). However, no significant differences were observed in OC content (p>0.05) bewteen the dry and wet irrigation treatments, suggesting that water availability was not a key driver affecting cyanobacteria development.

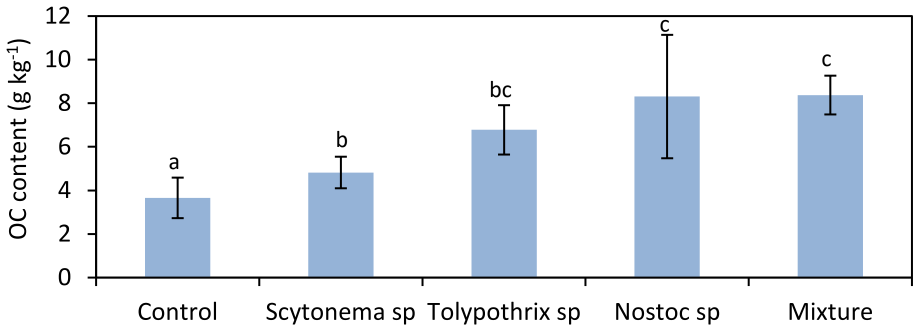

Figure 3. OC content (mean ± sd) in the control and the different inoculation treatments in the soils from El Cautivo (wet irrigation treatment). Different letters indicate significant differences (*p*<0.05) between treatments.

We can conclude that, according to the variables analysed, soil inoculation with *Nostoc* and the mixture of the three tested cyanobacteria strains appear to be the most effective inoculation strategies to promote biocrust development and improve soil properties.

CONCLUSIONS

Our results point to the feasibility of using individual cyanobacteria strains or a consortium of them to promote the formation and development of artificial biocrust communities and improve key soil properties related to soil fertility. This improvement in soil quality by cyanobacteria inoculation can be crucial to facilite further plant establishment. However, previous to their broad application, research is necessary to study the effectiveness of these strains under field conditions.

ACKNOWLEDGEMENTS

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6.7.O

EVALUATION OF SOIL WITH THREE ORGANIC RESIDUES ON EROSION USING RAINFALL SIMULATOR

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INTRODUCTION

Soil erosion has been damaged agricultural lands around the world and became a serious problem to developing countries. In Brazil, soil erosion is an environmental, social and economic problem because affects agricultural productivity, the work in countryside areas, water contamination and river siltation (Silva at al., 2013; Castro and Queiroz Neto, 2009; Beskow et al., 2009).

To solve or to reduce the soil losses it is necessary to protect the soil against erosion implementing techniques such as: (i) soil bioengineering; (ii) soil conservation practices; (iv) structural works; and (v) increase of soil organic matter content.

Considering the importance of sustainable practice in the environment, some researchers have used animal and agro-industrial residues such as sugar cane bagasse, cattle manure, coffee pulp and husk, cassava bagasse and sewage sludge because they provides: (i) an alternative substrates; (ii) helps solving contamination problem that their uncontrolled disposal may cause; and (iii) a good source for nutrient and organic matter that can be used to reclamation of degraded soils (Pandey et al., 2000; Eneji et al., 2008; Ferrer at al., 2011; Bhattarai at al., 2011; Moreno-Ramon at al., 2014).

The aims of this work were to determine the variations in runoff, infiltration and soil loss, in a clay soil mixed with three different kind of organic matter (composted cattle manure, sewage sludge and coffee husk) very abundant in the state where this study was carried out.

METHOD

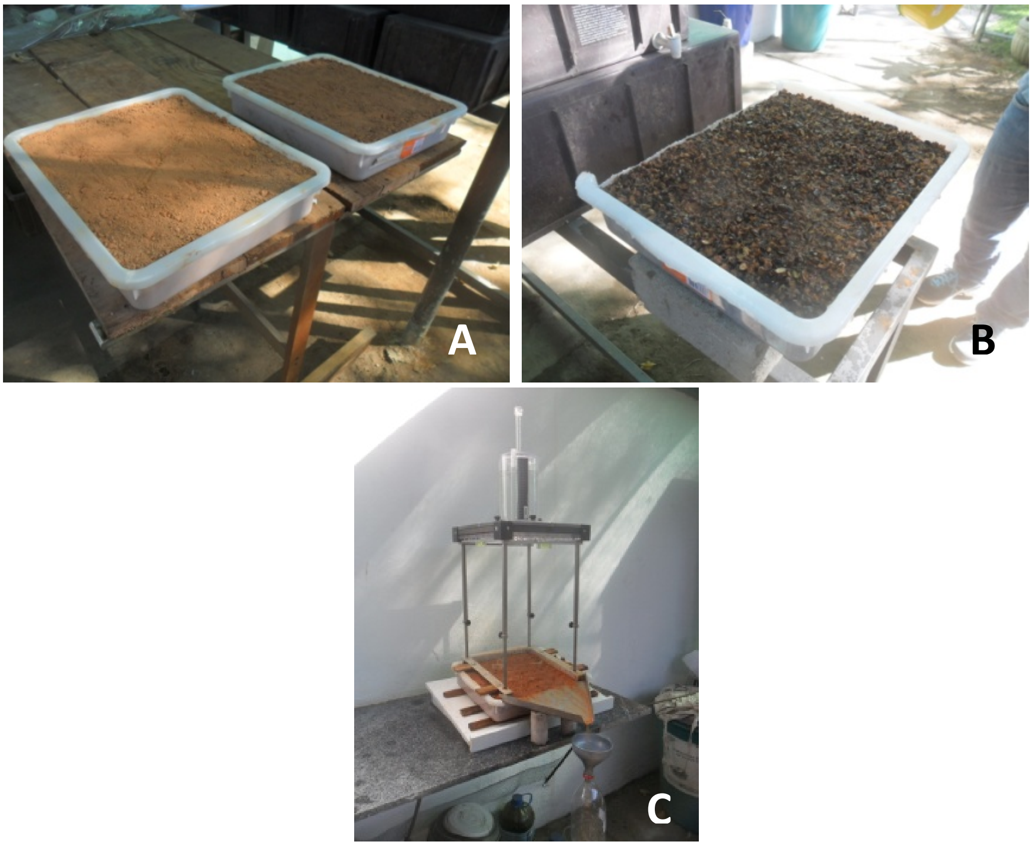
The experiment was realized in plastic containers (Figure 1) using a rainfall simulator at soil laboratory at IFES, Brazil. The residues were used in two different doses under two slope angles and the results will improve the soil management and the disposal of residues.

The soil used in this study was a loamy soil. It was collected from the agricultural area in Cariacica, state of Espírito Santo, Brazil. After being collected, it was air-dried and then the soil clods were broken using hands and a rubber hammer; then stored for being used into containers.

An experimental factorial design with three independent variables was designed. The factors were (i) treatment: pure soil (PS), composted cattle manure (CCM), sewage sludge (SS), coffee husk (CH); (ii) residues dosage (10% and 30% of total solids); (iii) slope angle: 10^{o} and 20^{o}. The combination of these factors resulted in 48 experimental simulations, which were replicated three times (4 soil treatments x 2 amount of residues x 2 slope angles x 3 replicates = 48). The residues had a content of 10% moisture. The measured dependent variables were runoff (mm), soil loss (gm^{-2}) and infiltration (mm).

The rainfall simulator used in this study is manufactured and sold by Eijkelkamp Agrisearch Equipment. It is a small rainfall simulator to assess infiltration and soil erosion. Some technical specifications of the rainfall simulator are: 49 capillary tubes with a calibrated cylindrical reservoir with capacity of 2300 mL of water, the stainless steel ground frame is connected to a gutter installed on the downstream site of the plot for the collection of the runoff and sediment; the surface area of test plot is 0.0625 m^2; the kinetic energy of rain is 4 Jm^{-2} mm^{-1}; intensity of rain simulation is 6 mm min^{-1} and duration of rain simulation is 3 minutes. Calibration tests, some practical use and its performance are detailed in Iserloh et al. (2013) and Eijkelkamp (2016).

Figure 1: Container with soil and organic residues user (A and B); Rainfall simulator (C).



RESULTS

It was observed that the runoff generated for all soil treatment was reduced when compared to pure soil. However, in the coffee husk treatment, with dosage of 30%, the runoff was zero (Figure 2) because all rainfall was infiltrated into the soil. This result was very favorable to the coffee husk treatment.

The concentration of total sediments ($g.L^{-1}$) in the runoff increased for all treatments when compared to the pure soil. The composted cattle manure treatment had the highest sediment concentration. However, in the coffee husk treatment with 30% dosage, the concentration was null due to no runoff (Figure 3).

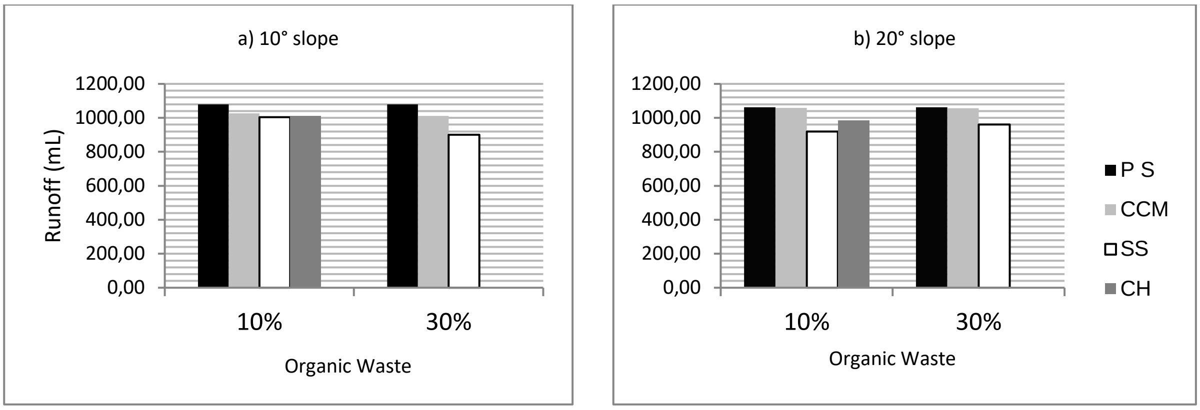


Figure 2: Simulated runoff.

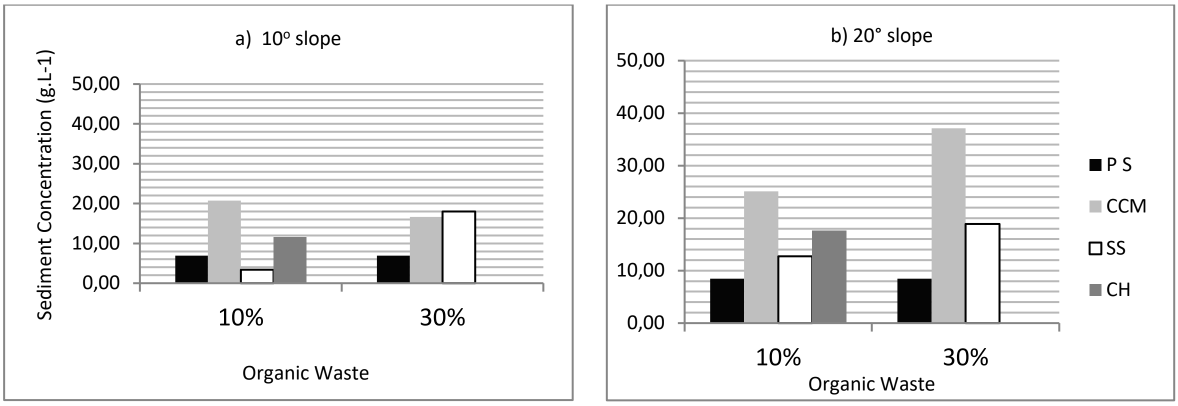


Figure 3: Sediment concentration on runoff.

CONCLUSIONS

It was observed in this research that the use of organic residues such as coffee husks, sewage sludge and composted cattle manure have improved soil infiltration. The coffee husk was the residue that presented the best result because when applied at 30% dosage allowed a total infiltration of the rainfall simulated by rainfall simulator.

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6.8.O

APPLICATION OF COMBINED ORGANIC/MINERAL FERTILIZER AMENDMENTS TO SUSTAINABLY ENHANCE SOIL FERTILITY IN TRADITIONAL OASIS PRODUCTION SYSTEMS

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1. INTRODUCTION

Arid and semi-arid biomes cover 41% of the terrestrial land surface, they are home 31 % of the of world population (Schimel, 2010), and because of global change, demography accentuation and land degradation issues its area is increasing. Naturally, dryland soils are low in organic carbon (<1%) due to the low productivity of the agroecosystems they support (Feng and Fu , 2013; Reynolds et al., 2007; Seager et al., 2007). Nevertheless, the size of the areas concerned means that the organic carbon stock in arid and semi-arid regions is far from being negligible with close to 750 Gt of carbon (Bernoux, 2011). Depending on the classification criteria, dry regions represent less than 30% of the total organic carbon stocks in the soil.

The Middle Eastern and North African (MENA) region spread from Morocco in the west to Iran in the east, including the majority of the Middle Eastern and Maghreb countries, it is currently characterized by high population growth, and in significant parts also by high temperature and low rainfall conditions. This dryland climatic conditions accelerated soil erosion, nutrient imbalances and soil organic matter (SOM) depletion, which individually and jointly are major problems threatening the economic livelihood and food security of the population in the region (Dregne , 2002).

In south Tunisia over half of the rural population is directly dependent on locally grown crops and date palm production. Reduction in soil fertility depletion has been highlighted as the single most important constraint to food security in South Tunisia (Mlih et al., 2015). Maintaining organic carbon (OC) and total nitrogen (TN) status; hence also soil fertility, is therefore critical in the food security of this climate vulnerable region. Date palm production is the only agriculture source of employment and income in southern Tunisia, it plays social and economic important role with 30% of world production and 70% of total Tunisian date production (frutrop, 2001)

The Nefzaoua region (22,900 km^2) of South Tunisia does cover 48% of the total area of all Tunisian oasis with nearly 15,600 ha of palm trees juxtaposed in landscape of old and new ownership (Ministry of Environment, 2012). The Nefzaoua region contains over 4 million palm trees of which ca. 1.6 million are Deglet Nour (the most valuable date variety). The date palm production in this region has over the last decades date decreased from 190.6 Gg 2011-2012 (RCDA , 2012) to 182.5 Gg in 2014-2015 (Ministry of agriculture , 2015).

The lack of precipitation water leads to a reduced input of shoot and root material, both due to the limitations on growth of crops and the natural vegetation, and additional crop offtake, which in turn leads to low SOC.

In addition, in these arid and hyper-arid regions OM turnover and decomposition rates are rapid as a result of high soil temperatures, which also limit date palm and forage production in these Tunisian oases systems. Furthermore, sub-optimal land management does further enhance soil degradation, and the unfavorable location of Nefzaoua oasis in and around Chott Jerid and Gharsa salt plains (Kadri et al., 2002; Ben Aissa et al., 2004; Marlet et al., 2007), may further exacerbates soil degradation problems

Therefore, new agricultural fertilizer and management practices are required to help reduce the rapid mineralization of OM, enhance SOC stock, available nitrogen (N) and ultimately the need for costly continuous additions of farm manure (FM) by farmers to maintain fertility in these oasis soils. We propose that a combined organic amendment and mineral amendment (bentonite clay), both have in single amendments to soils shown proven benefits, may provide additional benefits when applied together. The aim of the present study was to examine the viability of new proposed amendment materials, consisting of such mixtures of organic (farm manure or compost) and mineral (bentonite clay) materials. The potential of these new materials was compared with the FM amendment only based strategies traditional employed by the farmers in the region.

The present study compared the effectiveness of the five organic/mineral fertilizer amendments compared to an unamended control on soil fertility in oasis production systems, as evaluated through rate and persistence of the increase in soil organic C and total N achieved following the application of these \amendments.

2. MATERIALS AND METHODS

2.1. Site characteristics and trial design

Soil samples were collected from a traditional oasis in Fantassa (33.8° N; 8.7° E), in Kebili governorate in southern Tunisia. Climate data indicated the site had an annual mean precipitation (MAP) of 40 mm and a mean annual temperature of 21.4°C. Temperature ranges from mean maximum temperature of 46.6 °C, mean minimum temperature of 1.9 °C. The annual potential evapotranspiration exceeds 2000 mm per year. The wind current of sirocco blows strongly (120 D yr^{-1}) in the summer carrying in high quantities of sands and dust.

The experimental design was a randomized complete block with six treatments and tree replicates; the plot size was 3 m long and 3 m wide. The six treatments were: (i) sandy soil + farm manure (SF), (ii) sandy soil + compost (SC), (iii) sandy soil + bentonite clay (SB), (iv) sandy soil + bentonite + farm manure (SBF), (v) sandy soil + bentonite + compost (SBC) and (vi) untreated control (U) The treatment amendments were applied on 17 June 2014. Different treatments are mixed in top layer of 0-20cm and then tilled the soils manually with the traditional tools of the farmer up to a depth of 30 cm.Farm manure (FM) was applied at a rate equivalent to 30 ton ha^{-1}. The compost (C) was applied at 10 ton ha^{-1} and Bentonite Clay was added of 12% of the total soil surface. The amounts applied for each amendment were the same both for single or multiple amendments (photo 1).

The applied farm manure (from goats and sheep) is the same as traditionally used by farmers. The compost was derived from the waste of date palm in the Kebili oasis and was produced by the Institute of Arid Land also in Kebili. The bentonite clay was from a geological source in El Hamma in Gabe's region. Physical and chemical characteristic of different amendments is found in Table 1.

2.2. Soil sample collection and preparation

Soil samples from three depths (0–20, 20-40 and 40-60 cm depth) were collected at four different times: (i) directly after amendment application on the 17th June 2014 (t= 0), (ii) the 18th September 2014 (t=90 days), (iii) the 20th December 2014 (t= 180 days) (iv), and finally on the 20th March 2015(t=270 days). All samples were placed in plastic bags, labelled and taken to the laboratory. Soil from each replicate was air dried, sieved and stored for chemical analysis.

2.3. Chemical and physical analysis

Soil chemical analysis, i.e. organic carbon (Walkley–Black), nitrate nitrogen (Kjeldahl) was conducted by Institute of Arid Land laboratories. The soil organic matter in soil samples was oxidized with $K_2Cr_2O_7$ in concentrate sulphuric acid for 30 min followed by titration of the excess of $K_2Cr_2O_7$ with ferrous-ammonium sulphate. Nitrogen in the digest was determined by Kjeldahl distillation and titration method (Bremner and Mulvaney, 1982) The C: N ratio was calculated by dividing the OC concentration in the soil with that of TN for the same depth (Blanco-Canqui and Lal, 2008) Soil bulk density (BD) was determined using the core method with three replications per plot. The soil depth intervals for the bulk density measurements were the same as those mentioned above for the soil samples for the SOC and TN analyses. The stainless steel cylinder was 5cm in diameter and 5 cm in height. Soil BD was computed from the weight of oven-dried soil at 105 °C and using (Okalebo and al, 1993):

BD =W1-W2/V (*equation 1*)

Where, BD indicates the soil bulk density (gcm^{-1}), W1 is the weight of soil after drying in an oven and plus the weight of the stainless steel cylinder (g), W2 is the weight of the stainless steel cylinder (g), and V is the volume (cm^{-3}) of the stainless steel cylinder.

2.4. SOC and TN stock calculations

The SOC and TN stocks were computed by the equivalent soil mass method using

An equation was used to characterize SOC stock in Mg ha^{-1}

C Stock = BD × SOC (g kg^{-1}) × H (*equation 2*)

Where, C Stock represents SOC Stock (Mg ha^{-1}), BD is soil bulk density (Mg m^{-3}), SOC is SOC concentration (g kg-1) and H is soil sampling depth (m)

C is the carbon content estimated by elemental analyzer and calculated as follows:

SOC concentration (g.kg^{-1}) =4*((V-v)/V) *10 (*equation 3*)

Where, V Control titration volume, v is titration volume of the samples.

An equation was used to characterize TN stock in Mg ha^{-1}

TN Stock (Mg ha^{-1}) = BD (Mg m^{-3}) ×TN concentration (g kg^{-1}) ×H (m) (*equation 4*)

Where, BD is soil bulk density, TN is the total nitrogen content estimated by elemental analyzer and calculated as follows:

N ($g.kg^{-1}$) = 0, 14 × V_{HCl}×10 (*equation 5*)

Where V_{HCl} is the titration volume after distillation.

2.5. Statistical Analyses

SAS software 9.0 was used for the meta-analysis described above to examine the significance difference test ($p<0.05$) of the response of the OC and TN stock and C: N ratio to different amendments, soil depth, and experimental duration. Sigma Plot 11.0 (Systat Software, 2008) was used to plot the graphic.

3. RESULTS

3.1. Effect of depth trend on OC and TN Concentration and Stock

The OC and TN concentration still gradually decreased significantly ($p<0.05$) with the depth in soil, regardless of the type of amendment applied treatment. The OC and TN concentration were significantly decrease (P‹ 0.05) in SF and SC amendment with depth from 24.0±0.4 $g.kg^{-1}$at 0-20cm depth to1.9±0.07 $g.kg^{-1}$ at 40-60cm for OC concentration and from 1.6±0.1 $g.kg^{-1}$ at 0-20cm depth to 0.5±0.2 $g.kg^{-1}$ at 40-60cm.

When we combined mineral and organic amendment in soil OC and TN concentration decrease significantly (P‹ 0.05), but this decrease is less than the decrease in soil amended with mineral or organic amendment alone. OC decrease significantly from 25.6± 0.4 $g.kg^{-1}$ in soil amended with SBC at 0-20cm depth to 2.5±0.1$g.kg^{-1}$ at 40-60cm (table 1).

Table 1. Organic carbon (OC), total nitrogen (TN) concentration and C: N ratio's for 0-60 cm depth in the oasis soil under various amendment treatments at the start of the experiment.

	Depth (cm)	U	SB	SC	SM	SBM	SBC
OC	0-20	0.2±0.01	1.6±0.3	24.0±0.4	16.1±0.1	15.7±0.1	25.6±0.4
Concentration	20-40	0.6±0.4^{a}	5.2±0.3^{a}	4.7±0.1^{a}	5.2±0.3^{a}	3.5±0.1^{a}	6.7±0.2^{a}
g.kg^{-1}	40-60	0.7±0.3^{a}	1.0±0.2^{a}	1.9±0.1^{a}	1.7±0.04^{a}	3.0±0.2^{a}	2.5±0.1^{a}
TN	0-20	0.9±0.1	0.7±0.4	1.6±0.1	2.0±0.3	2.1±0.6	1.2±0.2
concentration	20-40	0.6±0.2^{b}	0.5±0.4^{b}	1.3±0.3^{a}	1.8±0.8^{a}	1.1±0.3^{ab}	0.6±0.2^{b}
g.kg^{-1}	40-60	0.4±0.2^{a}	0.4±0.2^{b}	0.5±0.2^{b}	1.2±0.8^{a}	1.5±0.3^{a}	1.4±0.1^{a}
C:N ratio	0-20	0.2±0.01^{a}	2.5±0.4^{a}	14.9±0.4^{a}	8.1±0.1^{a}	7.5±0.2^{a}	23.8±0.4^{b}
	20-40	1.2±0.2^{a}	1.8±0.2^{a}	3.7±0.1^{a}	2.9±0.1^{a}	3.2±0.04^{a}	22.6±0.2^{a}
	40-60	2.7±0.1^{a}	2.2±0.02^{a}	3.5±0.2^{a}	1.4±0.1^{a}	1.8±0.1^{a}	1.7±0.1^{a}

OC stock decrease significantly ($p<0.05$) in soil amended with organic amendment SC treatment from 6.9±0.4 kg.m-^{2} in the surface layer (0-20cm) to 2.6±0.1 kg.m-^{2} in 20-40cm of depth. In deeper layer 40-60cm this decrease don't have significant effect ($p=0.17$). However, when we combined organic with mineral amendment, OC stock decrease significantly ($p<0.05$) but more slowly than the decrease of OC stock in organic amendment only. OC stock decrease from 7.4± 0.4 kg.m-^{2} in surface layer (0-20cm) to 4.1±0.2 kg.m^{-2} in 20-40cm of depth, in deeper layer (40-60cm) this decrease don't have significant effect ($p=0.17$).TN stock increase significantly ($p<0.05$) with depth from 0.6±0.3 kg.m^{-2} in SM treatment in surface layer to 1.0±0.7 kg.m^{-2} in 20-40cm of depth, to attend maximum with significant increase in 40-60cm of depth with 1.1±0.2 kg.m^{-2} , when we added mineral with organic amendment, TN stock increase but more slowly than with organic amendment, it increase from 0.3 ±0.1 kg.m^{-2} in surface layer (0-20cm) to 0.4±0.2 kg.m^{-2} in 20-40cm of depth, to achieve 1.3±0.4 kg.m^{-2} in deeper layer (40-60cm) (Table 2).

Table 2. Organic carbon (OC), total nitrogen (TN) Stock for 0-60 cm depth in the oasis soil under various amendment treatments at the start of the experiment

	Depth (cm)	U	SB	SM	SC	SBM	SBC
OC Stock $Kg.m^{-2}$	0-20	0.05±0.01	0.9±0.3	5.04±0.1	6.9±0.4	4.7±0.1	7.4±0.4
	20-40	0.3±0.1	0.5±0.4	3.1±0.3	2.6±0.1	3.1±0.1	4.1±0.2
	40-60	0.7±0.3	0.9±0.1	1.6±0.1	1.7±0.1	2.8±0.3	2.3±0.2
TN Stock $Kg.m^{-2}$	0-20	0.2±0.1	0.2±0.1	0.6±0.3	0.4±0.1	0.6±0.5	0.3±0.1
	20-40	0.3±0.2	0.3±0.1	1.0±0.7	0.7±0.3	0.7±0.2	0.4±0.2
	40-60	0.3±0.1	0.4±0.1	1.1±0.2	0.4±0.2	1.5±0.3	1.3±0.4

4. CONCLUSION

Combined organic/mineral amendment enhance the duration of the fertility of soil, OC and TN decrease significantly ($p<0.05$) with the amendment duration but more slowly than organic amendment only. So, combined organic amendment with mineral amendment should be considered as a smart option in oasis system to sustainably increase OC and TN Stock therefore increasing soil fertility to reduce organic matter in the sever climatic condition of the region.

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6.2.P

RESTORING ABANDONED AGRO-SILVO-PASTORAL LANDSCAPES USING THE *COCOON* ECOTECHNOLOGY

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INTRODUCTION

Agro-silvo-pastoral systems are a type of mixed land use characteristic of the Mediterranean region. There is a triple and complementary use of the land, adapted to areas of low soil potentialities and to the Mediterranean climate: open evergreen forests, grazing and rain fed crops. Due to their mixed characteristics and to the extensive form of exploitation, these systems constitute varied landscapes of high biological diversity (Pinto, 1993). Moreover, these systems can help meet the needs of a growing population and protect the environment: they can be exploited with both crop and animal-production systems and may enhance the supply of regulating ecosystem services, such as nutrient retention, carbon storage, pollination and pest control, as well as cultural services, such as recreation and landscape aesthetics (EC, 2016). They have been managed through decades furnishing the necessary outputs for the rural population of the area, but nowadays they are in regression in the European Mediterranean basin (Torralba et al. 2016). Due social, economic and ecologic constraints, the restoration of these systems is a difficult task despite a new rural demand exists.

The Green Link project (LIFE15 CCA/ES/000125) aims to demonstrate the environmental and economic benefits of an innovative tree growing method that has the potential of replacing traditional planting techniques with the “Cocoon”, a low-cost and 100% biodegradable device that improves water supply to seedlings.

METHODS

Near to 4.000 Cocoons will be installed between autumn 2016 and spring 2017 in a burned area in El Bruc (Catalonia, NE Spain) in order to help the restoration of the old agro-silvo-pastoral landscape that exist in this zone at the beginning of the 20th century (figure 2). This area is mainly private, with small properties (<100 ha). It is located at 500 m.a.s.l., at the foothills of the Montserrat Mountains. The mean annual precipitation is about 600 mm, with a market dry period during summer. Soils are mostly *Xerorthent tipic*, *Xerorthent litic* and *Xerepts*. Geomorphology is dominated by >15% slopes, despite old terraces and flat croplands exist. Except some croplands, the vast majority of the area is abandoned.

Plantations will be monitored over 4 years, in order to evaluate the Cocoon effect on growth and survival rates of the seedlings, and the general effect on the restoration of biodiversity. Moreover, the effect on soil quality will be evaluated through physicochemical and biological analysis. Regarding socio-economic aspects, surveys will be carried out in order to know the perception of different stakeholders about ecosystem services and the restoration of the old agro-silvo-pastoral landscape.

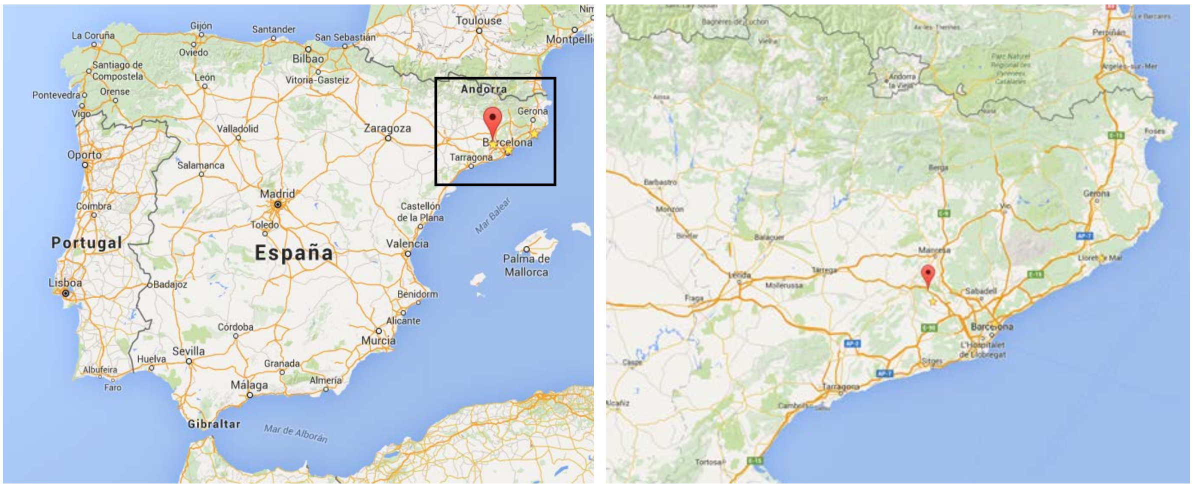

Figure 1. Location map of El Bruc area.

The species to plant are selected with the engagement of landowners, farmers and local and regional authorities, and in coordination with another LIFE project (LIFE Montserrat, LIFE13/BIO/ES000094). Near 4.000 seedlings will be planted in 20 ha in different densities (table 1). Selected species include forest native species (*Q. ilex, Q. faginea*), forest species not native and adapted to driest climates (*Ceratonia siliqua, Argania spinosa*), agronomic native species (*Figus carica*, *Juglans regia, Vitis vinifera, Prunus dulcis,* vera olive tree) and crop species not native and adapted to driest climates (cornicabra and arbequin olive trees). Despite this distinction between cultivated and forest species, some forest species with high added value will be planted in croplands (oak tree for truffle production, argan tree) and some cultivated ones will be planted in burned forests, in order to restore the ancestral agricultural use.

Table 1. Species selected to plant in El Bruc, area occupied per specie, prevision of seedlings planted per area and median planting density.

Specie	Plantation area (ha)	Number of seedlings to plant	Planting density (seedlings/ha)
Olea europaea var. *europaea* (arbequin)	8	770	100
Olea europaea var. *europaea* (vera)	0,3	100	300
Olea europaea var. *europaea* (cornicabra)	1,8	980	550
Quercus ilex subsp. *ilex* (truffle)	1,4	330	240
Quercus ilex subsp. *ilex*	0,3	60	200
Quercus ilex subsp *ballota*	2	400	200
Quercus faginea	2,5	500	200
Ceratonia siliqua	0,5	100	200
Figus carica	2,1	300	150
Prunus dulcis	0,2	60	300
Vitis vinifera	0,8	240	300
Juglans regia	0,3	60	200
Argania spinosa	0,2	25	125

TOTAL	**20,4**	**3925**	**200**

Figure 2. General view of the hills to be restored in El Bruc area. Old terraces, soft slopes and croplands are the places where Cocoon technology is more suitable.

RESULTS

In November 2016 plantations started using a drill installed in a tractor: 667 olive trees (var. arbequina), 142 holm oaks (truffle inoculation) and 137 fig trees were planted. Moreover, some of the plots were fenced (electric and hunting fence), and chemical (chilly feed) and physical (canon) repellents were installed in order to prevent damages from wild boards. During plantation some lessons were learned in relation to Cocoon installation and design (figure 3). These problems, mainly those related to installation tasks, will be corrected in order to not repeat them in the spring plantation.

Figure 3. Symptoms of deficient installation or device limitations or failures: Cocoon outstanding from soil surface, incorrect fulfil of holes (space between Cocoon and soil walls), protector fall and lip collapse.

Figure 4. Hunting fence, electric fence and canon installed in order to prevent wild boards damages.

Planting technique, using a drill installed in a tractor seemed to be very convenient due its polyvalence, allowing to use the tractor for drilling, watering (with the installation of a barrel) and nailing sticks (for fencing).

In the medium term, the expected results of the project are to increase environmental quality and green economy:

- Increasing soil fertility (increasing organic matter and water retention capacity, reduction of soil erosion)
- Increasing biodiversity (woody plants establishment and encroachment, new habitat creation, fauna attraction and establishment)
- Reducing forest fire risk (open landscapes creation, low biomass charge)
- Creation of green jobs linked to the agro-silvo-pastoral landscape: maintenance and cultivation of the plantations, harvesting (olives, figs, truffle), elaboration (olive, cheese, marmalades, meat), commercialization, ecotourism.

ACKNOWLEDGMENTS

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6.3.P

THE GREEN LINK PROJECT: RESTORING DESERTIFIED AREAS WITH AN INNOVATIVE TREE GROWING METHOD ACROSS MEDITERRANEAN BASIN TO INCREASE RESILIENCE

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INTRODUCTION

Desertification is defined as "land degradation in arid, semiarid and sub-humid areas resulting from various factors, including climatic variations and human activities" (UNCCD, 2013). In this sense, soil degradation is one of the most threatening consequences of climate change. Approximately 45% of soils in Europe are in a vulnerable state and 15% even being considered extremely vulnerable. Some southern parts of the EU, including Spain, Greece, Portugal, Italy and France (Corsica) are significantly affected (EEA, 2008). Soil degradation problems not only have environmental implications: more and more agricultural land are degrading and their cultivation becoming unprofitable, adding serious economic problems to rural areas already weakened by depopulation. This project aims to compensate impacts of climate change and contribute to increase resilience of Mediterranean ecosystems, specifically in Spain, Greece and Italy.

There is a huge number of studies on the potential impacts of climate change on water resources which involve many different approaches. A common element is the reduction of water availability for forest ecosystems and for irrigation purposes across all regions (EEA, 2012). Reforestation and afforestation efforts in the Mediterranean region cannot be called cost-efficient currently, since the percentages of growth failure and seedling mortality rates are extremely high, mainly when broad-leaved resprouting species (e.g. *Quercus* species) where planted (Vallejo et al 2012).

METHODS

The Green Link is a collaborative LIFE project (LIFE15 CCA/ES/000125) that aims to demonstrate the environmental and economic benefits of an innovative tree growing method that has the potential to restore desertified areas across the Mediterranean basin. This consists of replacing traditional planting techniques with the "Cocoon". The Cocoon ecotechnology consists of a water reservoir, combined with mycorrhizal fungi inoculum, and a tree shelter (see figure 1). The Cocoon is put into a small hole of 60 cm wide and 30 cm deep made into the soil. In the middle of the hole, a seed or a sapling is planted first, and then the Cocoon device is put around it. The water reservoir is made of recycled paper pulp sealed with a biodegradable coating to ensure water tightness during the first year. It is only water filled once at the moment of planting. Water is slowly transported to the tree using wicks. As the reservoir degrades and empties over time, the remaining shallow pits will serve as a micro-catchment to collect surface runoff during rain events. Additionally, the degraded reservoir becomes an organic amendment ameliorating the soil. Mycorrhizal fungi are added to the soil surrounding the plant roots, increasing the absorbing surface of roots 100 to 1000 times, thanks to a very efficient symbiosis between roots and fungi by very fine mycelia. This improves soil moisture availability as well to the soil nutrients. Mycorrhizal fungi also release enzymes into the soil that dissolve hard-to-capture nutrients, such as organic nitrogen, phosphorus and iron, commonly fixed on to the soil complex. These fungi are present in 90% of natural forests and woodlands and form a characteristic symbiotic association with the roots. A cylindrical shelter is placed around the tree to protect the seedling against the sun, desiccating winds and smaller animals feeding on the young plant. The Cocoon practically eliminates evaporation of water from the adjacent soil and prevents the growth of weeds near the saplings, which would otherwise compete for water, nutrients and light. Hence, all available resources are directed towards tree establishment.

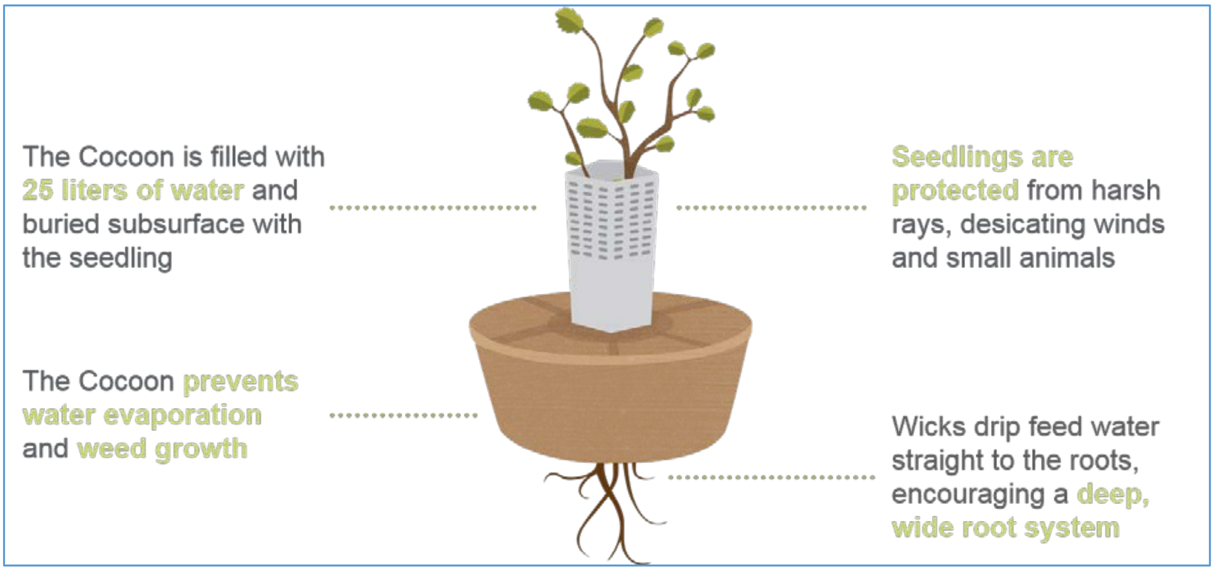

Figure 1. Scheme of the Cocoon device.

In order to prove the viability of the Cocoon technology and demonstrate its potential, the project foresees planting a variety of woody species on different soil types located in areas on a climate gradient from semi-dry to extremely dry climates across the Mediterranean border. As a whole, 7 pilot areas located in Italy, Greece and Spain, covering 60 ha planted with 24.000 seedlings of 20 plant species, will be evaluated (see table 1).

Table 1. Pilot areas description.

Village / area	Tifaracás (Canary Islands, Spain) Population: 2,028 Municipal unit: 103,30 km^2 Included in the Biosphere reserve	Municipality of Almería (Natural Park de María Vélez, Spain) Population 191.443 Municipal unit: 296,21 km^2	Municipality of Jijona* (25 km north of Alicante, Spain) Population: 7.575 Municipal unit: 163,76 km^2	Municipality of El Bruc (10 km East of Igualada and 15 km West of Terrassa, Spain) Population: 254.217 Municipal unit: 47,21 km^2	Municipality of San Marco Argentano, province of Cosenza, in the Calabria region, Italy Population: 7.500 Municipal unit: 78 km^2	Ptolemaida (part of the municipality of Eordaia, Greece) Population: 32.142 Municipal unit: 217,901 km^2
Climate/Soil	Very dry, eroded soil, with steep deforested hills	Extremely dry, desertic in some areas	Very dry conditions with low levels of soil moisture due to the lack of rain (especially in summer)	Dry conditions, with soil vulnerable to erosion due to recent wildfires	Very eroded soil, frequent droughts and floods have led to land degradation	Very heterogeneous soils: clay, sandy and loam (black, brown, grey); partly rocky, intersected by lignite
Altitude	300-600 m	1200 m	453m	489m	200m	648m
Former use	Forest	Agriculture (almond orchards)	Agriculture (cereals)	Forest	Agriculture	Opencast mining
Current use	Part of a natural park and testing of new agricultural species	Part of a natural park and testing of new agricultural species	Abandoned farmland	In August 2015, 1.277 Ha. burned down in a forest fire	Farmland turned into disuse and the land has been abandoned	Lignite mining site owned and operated by PPC (Public Power Corporation)
Challenge	Enhance connectivity between the green patches in the park and provide safe haven for local and migrating birds	Enhance connectivity between green patches	Converting eroded soil into fertile soil, as well as to offer an economic alternative for land-owner and farmers.	Reforest part of damaged area using adaptive and economically interesting plant species	Converting eroded soils into fertile soils & Support plants in the first growth stages	Converting eroded soils into fertile soils & Support plants in the first growth stages

*The information about Tous, which is the second pilot area in Valencia (Spain), is not included on the table.

The potential benefits of Cocoon will be validated through monitoring of various objective indicators:

- Survival rate: monitoring of planted individuals and estimation of their physiological state. Tree height and stem base diameter will be measured in a selection of individuals.
- Biodiversity: flora inventories and key species abundance will be carried out in each zone before plantations and at the end of the project. Fauna recruitment and entrance (use) will be estimated through monitoring qualitative indicators: tracks, excrements, lairs, nests and direct observations.
- Vegetation structure and growth rate: vegetation structure will be measured in transects, identifying main species height and width, and vegetation cover.

- Biomass carbon stock: above ground carbon stock will be estimated with modelling tools according to vegetation structure data.
- Edaphic parameters: soil quality indicators will be analyzed at the beginning and at the end of the project: total organic carbon, particulate organic matter, soil pH, electrical conductivity, soil nitrate, microbial activity, microbial biomass, potentially mineralizable N, soil enzymes, soil respiration.
- Root growth and soil carbon stock
- Mycorrhizal infestation will be assessed, not only by microscopy observation, but also with innovative molecular techniques (DNA sequencing) to identify the type of mycorrhiza.

The socioeconomic perception of the actions by the local economy and population in all areas will be monitored during the project's lifetime. This action involves two rounds of interviews, at the beginning and at the end of the project.

EXPECTED RESULTS

The Green Link project expects to:

1. Demonstrate that the Cocoon technology allows planting trees in dry climates and on poor soils in response to combat desertification phenomena. It is expected to achieve over 80% survival rate after planting, for all the species selected.
2. Offer a competitive market solution to plant trees, without the use of irrigation with the Cocoon, by demonstrating that there will be significant savings of up to 30% for planters (taking into account lower maintenance and dead trees repositioning costs) vs traditional methods in these areas or alternatives.
3. Improve soil quality since water scarcity will be compensated; further green cover, microorganisms and mycorrhiza will enhance the association among roots and soil, planting along height lines will help prevent erosion while improving water retention in the area.
4. Enhance ecosystem services provision by increasing biodiversity and positive growth of soil carbon stocks over time.
5. Quantify and value the variety of ecosystem services provided by trial areas to promote potential for regional socio-economic development.
6. Increase awareness and dissemination of adaptation strategies on forest management and among other stakeholders (particularly on EU relevant legislation and objectives).

ACKNOWLEDGMENTS

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6.5.P

BIOCHAR AMENDMENT DO NOT INCREASE SOIL WATER RETENTION IN A SANDY-LOAM MEDITERRANEAN SOIL

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INTRODUCTION

Biochar is a highly stable carbon-rich material obtained from pyrolysed biomass, and used as a soil amendment. It is also a very resistant material to decomposition when is applied to soil. Previous works reported biochar benefits, mainly in acid soils, improving soil fertility, nutrient retention microbial activity and contributing to organic carbon sequestration. Furthermore, biochar can improve soil aggregation, pore size distribution and bulk density (Downie et al., 2009) but it depends on the biomass feedstock, the production temperature during the pyrolysis process and the native soil properties. As biochar enhance porosity and reduce bulk density and evapotranspiration, several studies have reported greater aeration and water holding capacity or changes in hydraulic properties in treated soils (Major et al., 2010; Ouyang et al., 2013), but others indicates the opposite or non-effect (Laird et al., 2010). Therefore, evidences for hydrological effects remains unclear and the mechanisms behind the contradictory results are not yet well elucidated (Jeffery et al., 2015). In this sense, Novak et al. (2009) observed different effects on soil water retention capacity after the addition of several types of biochars to a loamy sand soil, being the characteristics of raw material and pyrolysis conditions (such as temperature and residence time) determinant in the resulting biochar's properties, for example its hydrophobicity.

The effects of biochar on soil water retention and availability can be explained as biochar promotes macroaggregate formation in soils, being this formation more intense in coarse texture soils (Ouyang et al., 2013). As a result, biochar amendment could increase saturated hydraulic conductivities and change the shape of the soil water retention curves. In a similar way, Abel et al. (2013) found that several biochars increased total pore volume as well as water content at the permanent wilting point on a sandy soil, poor in organic matter, but no appreciable effects on soil hydrophobicity were detected. Recently, Obia et al. (2016) showed an increase in soil aggregate stability, porosity and available water capacity at moderate doses (1-2%) of two biochars in a tropical soil. In addition, Ma et al. (2016) reported an improvement of soil physical properties in a three-year consecutive combined study after biochar (7.8 Mg ha^{-1}) and inorganic fertilizer application in a Mollisol. In this experiment, significant relationships between soil organic carbon and available water, confirm the close connection between improvement of soil structure and soil water.

Regarding water holding capacity in a biochar amended soil, some experiments found positive results, but at high rates as 60 t ha^{-1}, (Ulyett et al., 2014), being questionable as undesirable ecotoxic effects could appear (Marks et al., 2014). Other authors, who used a more realistic doses, found that biochar may not improve water retention when was applied only one-time (Brantley et al., 2014). Therefore, the aim of this experiment was to demonstrate the influence of two different biochars (6.5 Mg ha^{-1}) in soil water holding capacity and bulk density in an agricultural soil.

EXPERIMENTAL AND METHODS

In this study, a neutral pH, stony rich, sandy-loam *Fluventic Haploxerept* soil, cultivated as a vineyard, has been amended with two different types of biochars. One was obtained from pine wood splinters as a residue of a biomass gasification process, and was characterized as a hydrophobic material (P biochar); the other one was obtained from corn cobs by slow pyrolysis and was hydrophilic (Z biochar). Table 1 shows data of the main chemical composition and properties. Both were applied at a dose equivalent to 6.5 Mg ha^{-1}, uniformly distributed on soil surface, and then incorporated into the soil by tilling two times to a plough depth of 15 cm.

Table 1. Main chemical characteristics of biochar from pine wood (P), corn cobs residues (Z) and the Ap horizon of agricultural soil (S). Total carbon (TC), O/C and H/C molar ratios, electrical conductivity (EC), ash, pH, weight loss-on-ignition (LOI), Soil Organic Mater (SOM).

Parameter	Soil (S)	Pine biochar (P)	Corn cob biochar (Z)
pH (water)	7.26	11.5±0.04	10.3±0.04
EC (dS m^{-1} 25°C)	0.06	0.69±0.02	2.54±0.5
LOI $_{550°C}$ (g kg^{-1})	41.7	892.1±0.3	897.9±0.2
Ash (g kg^{-1})	-	91.9	91.1
TC (g kg^{-1})	10.7	793.4	785.8
H/C	-	0.19	0.29
O/C	-	0.11	0.11
$CaCO_3$ (g kg^{-1})	9.5	33.3	22.5
SOM (g kg^{-1})	16.47		
Coarse sand (%)	34.4		
Fine sand (%)	24.0		
Coarse silt (%)	10.9		
Fine silt (%)	14.9		
Clay (%)	15.2		

Samples from control soil (S) and biochar-treated soil (S+P and S+Z) were collected two months after biochar application. Bulk and skeletal soil density, particle size and water holding capacity at different times were measured.

RESULTS AND DISCUSSION

Biochar physical properties

Table 2 shows the differences about particle size distribution of both biochars. Corn cobs biochar has higher proportion of coarse particles, mainly at the 2-0.5 mm size range. Conversely, pine biochar is richer in fine particles which could influence physical properties such as specific surface area. As expected, both biochars have low bulk density values, especially pine biochar according to its finer particle size.

Table 2. Particle size distribution (%) and bulk density (kg dm^{-3}) of a pine (P) and corn cobs (Z) biochar used in this experiment.

Biochar raw material	>5 mm	5-2 mm	2-1 mm	1-0.5 mm	0.5-0.2 mm	<0.2 mm	Bulk density (kg dm^{-3})	Solid density (kg dm^{-3})*	Porosity (m^3 m^{-3})
Pine (P)	0.1	1.7	6.2	7.36	26.6	58.0	0.188	1.8	0.89
Corn (Z)	0	2.8	40.1	24.6	27.3	5.2	0.339	1.5	0.77

*According to Brewer et al. (2014)

Biochars also differs greatly in water affinity (Table 3). While pine biochar is highly hydrophobic, as water drops remains over the surface of particles during several hours, corn biochar is hydrophilic. This fact is also reflected by water holding capacity at different times (Table 3).

Table 3. Water drop penetration time (WDPT) and water holding capacity (WHC) of a pine (P) and corn cob (Z) biochar used in this experiment.

Biochar raw material	WDPT (sec)	Hygroscopic water (%)	Maximum WHC (%)	WHC 24h (%)	WHC 96h (%)
Pine (P)	>10 000	5.5	56.2	53.3	49.7
Corn (Z)	<3	8.2	194.3	187.6	186.9

Effect of biochar on soil physical properties and water retention

The addition of a moderate dose of biochar (6.5 Mg ha^{-1}) do not have a significant effect on soil bulk density and therefore on whole soil porosity (data not shown), due to the high proportion of gravel (64%) in this soil.

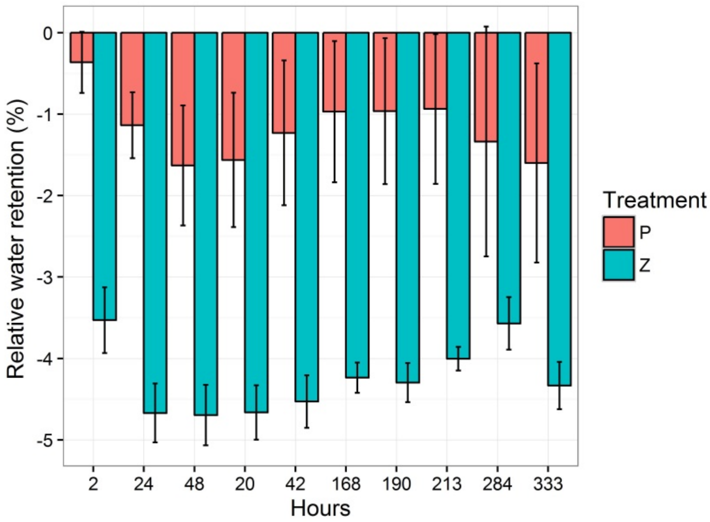

Figure 1. Differential water holding capacity (%) of a soil amended with pine biochar (P) or corn cobs biochar (Z) compared to the control soil along a slow air drying process (333h).

Soil amended with both biochars showed lower water retention capacity than control soil (Figure 1). Corn cob biochar had a more reductive effect (-4.25%) than pine biochar (-1.17%), comparing to control soil. These effects persisted along two weeks in a slow air drying process. Water holding capacity diminution could not be explained by hydrophobicity due to corn cob biochar was highly hydrophilic. Moreover, all biochar-amended soils were not water-repellent. Therefore, other biochar properties could explain the water filling in amended soil. Biochar particle size, larger in corn cob ones, could influence wettability as inner pores of biochar particles could not be easily filled with water. These results could contribute to clarify some current assumptions about biochar effect on soil water retention.

CONCLUSIONS

Pine biochar is a highly hydrophobic material richer in fine particle than the hydrophilic corn-cob biochar. Soils amended with both biochars showed lower water retention capacity than control soil, being the biochar particle size, larger in corn cob ones, one of the reason of the wettability.

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6.9.P

CAN HUMAN WASTE SAVE LAND FROM BEING WASTED?

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INTRODUCTION

Soil erosion is a long known, yet still persistent threat to food security (Pimentel & Burgess 2013). Current soil erosion rates on cultivated land generally exceed soil production rates by > 30 times and as a result almost one third of this land became degraded within the last 40 years.

The amount of soil organic carbon (SOC) in the soil is considered to be ”one of the key factors that control the stability of aggregates” (Blanco-Canqui and Lal, 2008, chapter 2).The higher stability of the aggregates leads to a higher resistance against the impact of forces from raindrops, while at the same time it increases the porosity and permeability of the soil and in turn leads additionally to less runoff generation (Gliessman 2000, Blanco-Canqui & Lal 2008).
Conventional farming methods like tillage reduce SOC and degraded land is usually low in SOC and thus more susceptible to soil erosion (Lal 2006). However, there is a cheap and always available source of organic matter, which is usually discarded and thereby causes many problems including 1.8 million child deaths/ year: human waste (Corcoran 2010).
Conventional toilets flush excreta away. But this away means only out of sight: 90% of all sewage is discharged into water bodies without any treatment (Corcoran 2010). Missing or inappropriate sanitation causes many health problems and a number of deaths per year which outnumber deaths caused by all forms of violence together - including wars (Corcoran 2010). Even in regions where expensive sewers and treatment plants are installed, most of the nutrients are lost:

- The cleaned water still contains high amounts of valuable nutrients which are washed to the sea and thereby lost for agriculture. On the other hand these nutrients also cause problems in the water bodies, where they cause eutrophication and dead zones (Corcoran 2010).
- 40% of the sewage sludge in Europe is applied as biosolids in agriculture, the rest goes into landfill or incineration (European Commision, 2002). This is often due to being highly contaminated with heavy metals from industrial waste waters (Venglovsky et al., 2006).

However, there is a solution, which is cheap and solves not only the sanitation issue without wasting large amounts of water, but at the same time also keeps the valuable nutrients on land and provides a step towards closing the cycle and the perspective of a circular economy. Dry toilets and especially Urine-Diverting-Dry-Toilets (UDDTs) provide a sustainable solution for sanitation and also provide a free and highly available source of organic residues with high nutrient content.
The use of human excreta as an agricultural fertilizer is therefore starting to adhere more and more interest, especially with regard to the looming phosphorus crisis (e.g. Krause et al. 2015, Mihelcic et al. 2011, Mnkeni & Austin 2009). However, other uses of the residues should also be considered, since the use in agriculture can be opposed due to the potential risk of pathogens but also due to fear and misconceptions.

The objective of this study is thus to evaluate the use of human waste from urine-diverting dry toilets for non-agricultural purposes by investigating (1) their effect on the erodibility of soil and (2) the potential usage as a soil amendment for the restoration of degraded land.

MATERIALS & METHODS

Human faeces are being collected from different UDDTs in Ecuador. Faeces can contain many pathogens such as bacteria, viruses, protozoa and helminth eggs. To ensure a safe use and also improve the perception of the material, the faeces are being sanitized prior to application, using a solar sanitizer as shown in Figure 1. The sanitized waste will be applied to degraded land on the farm of the University “El Romeral” –coordinates -2.7657251,-78.7250273 in the province of Azuay, Ecuador. The experimental design consists of plots of 1m² size on a highly degraded sloped area (Figure 2), where two treatments and one control with 7 replicates of each are applied.

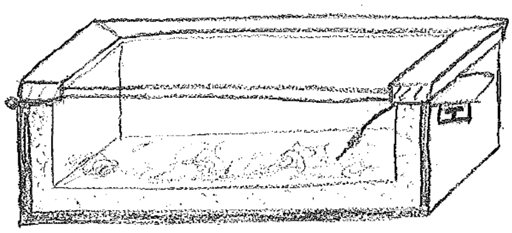

Figure 1: Sketch of a solar sanitizer: top: double glass; sides and bottom: insulated walls, painted black

Figure 2: Experimental area in El Romeral, Ecuador

In order to quantify the potential for restoration of degraded land, key parameters regarding soil quality and established plant biomass will be assessed. Furthermore the effect on the erodibility of the soil will be analysed after different time intervals (2 weeks, 6 months and 1 year after application). Therefore the runoff will be captured, measured and analysed for sediment and nutrient content.

EXPECTED RESULTS

Since positive effects of animal manure application on erodibility have been demonstrated (Grande et al. 2005), we expect comparable positive effects from the application of human waste, This would be observed as increased biomass production, improved soil quality parameters, increased infiltration and reduced sediments in the run-off.

CONCLUSION

The dissemination of sanitation concepts like the UDDT especially in areas of poor sanitation infrastructure have a large potential to improve health and living standards in developing countries. The safe use of human waste collected from such ecological sanitation systems as a soil amendment for degraded areas could be a cheap restoration method. It could facilitate the distribution and acceptance of the use of this type of sanitation as well as the utilization of the residues as a fertilizer also on agricultural land.

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6.10.P

RECLAMATION OF MULTI-METAL(LOID) POLLUTED SOILS USING NANOSCALE ZERO VALENT IRON

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INTRODUCTION

Soil contamination is a worldwide problem. In fact, it is estimated that there are about 2.5 million of potentially contaminated sites in the EU, of which about 14% (340,000 sites) are expected to be contaminated and in need of remediation measures (van Liedekerke et al., 2014). The main contaminant categories are mineral oils and metal(loi)s. Anthropogenic contamination of soils by metal(loid)s occurs from different sources, such as mining, atmospheric deposition and the application of sludge, mineral fertilizers and pesticides. Although legislation is based on total concentration of metal(loid)s in soil, it is a poor indicator of their toxicity effects. The available fraction of metal(loid)s in soils is the most ecologically relevant because it is readily mobile and bioavailable for plants and other soil organism (Adriano, 2001) and depends on the soil properties and environmental factors (Adriano, 2001). The application of immobilization technologies which reduce the metal(loid) availability in soil can be an alternative to traditional techniques, including excavation and landfilling, which are unfeasible on a large scale because they are environmentally disruptive and cost prohibitive. In the last few years, the use of nanoscale zero-valent iron (nZVI) for site remediation has been investigated for its potential to degrade organic pollutants and immobilize metal(loid)s (Jegadeesan et al., 2005; O'Carrol et al., 2013; Stefaniuk et al., 2016; Gil-Díaz et al., 2017a, 2017b). Most of the nanoremediation studies have been performed with water samples and its utility in soil has recently attracted attention (Vítková et al., 2016; Gil-Díaz et al., 2017a, 2017b). Furthermore, most of the applications of nZVI to soils have been conducted with single metal(loid) contamination, and little data are available regarding the effectiveness of nZVI for the simultaneous immobilization of several metal(loid)s in soil. The objective of the present work was to evaluate the effectiveness of nZVI to immobilize metal(loid)s including As, Cd, Cr, Pb and Zn, present in two different soils, a calcareous and an acidic one.

METHODS

Soil samples were collected from the surface layer of two agricultural areas from the Madrid region. Physico-chemical properties were determined according to the Spanish official methodology (MAPA 2004) (Table 1). The air-dried soil samples were spiked with a solution including all of the metal(loid)s to concentrations of 100 mg/kg for As and Cd and 200 mg/kg for Cr, Pb and Zn. The compounds used were As_2O_5, $CdSO_4$, $K_2Cr_2O_7$, $Pb(NO_3)_2$ and $ZnSO4{\cdot}7H_2O$. Spiked soil samples were incubated for 30 days at 25ºC and 50% humidity. Then, the soils were air-dried and sieved (<2 mm). A subsample (2.5 g) was weight in a 50-mL centrifuge tube and mixed with a given mass of the nZVI suspension diluted 1:10 with Milli-Q water to obtain the treatments 0%, 1%, 5% and 10% (w/w). The iron nanoparticles used were NANOFER 25S, supplied by NANO IRON s.r.o. Three independent tubes were used per treatment. The tubes were shaken for 72 h at 100 rpm with a Reax 2 shaker. Then, the mixtures were air dried and the availability of the metal(loid)s was analyzed by applying the sequential extraction procedure proposed by Tessier et al. (1979). The Tessier extraction separates the metals concentration into five fractions, in decreasing order of availability: exchangeable (EX), bound to carbonate (CB), Fe and Mn oxides (OX), bound to organic matter (OM), and residual (RS). Cadmium, Cr, Pb and Zn were quantified with a flame atomic absorption spectrometer and As was measured using a graphite furnace atomic absorption spectrometer.

Table 1. Physico-chemical properties of the soils included in the study.

Soil	pH	EC	$CaCO_3$	N	OM	P	Ca	Mg	Na	K	Cr	Pb	Zn	Sand	Silt	Clay
		dS/m		%						mg/kg					%	
Calcareous	8.23	0.27	8	0.11	1.1	26	2866	374	34	217	17	13	44	29	38	33
Acidic	5.50	0.07	0.15	0.03	0.6	27	500	72	11	97	10	9	20	43	42	15

RESULTS

The use of nZVI significantly reduced the metal(loid) availability in soils although the effectiveness of nZVI to immobilize metal(loid)s in soil strongly depended on the metal characteristics, soil properties and dose of nZVI (Table 2 and 3). As other experiments have concluded, the presence of other metal(loid)s can affect the immobilization effectiveness due to the competitive phenomenon between metal(loid) ions (Vítková et al., 2016; Gil-Díaz et al., 2017a).

Arsenic: The application of nZVI in the calcareous soil induced a reduction of As in more available fractions (EX+ CB) of 37, 86 and 93% at the dose of 1, 5 and 10%, respectively, and in the acidic soil available As decreased 50, 77, 89% at increasing nZVI dose. The As immobilization can be produce by by inner-sphere complexation with the iron oxides present in the shell of ZVI nanoparticles (Jegadeesan et al., 2005; Gil-Díaz et al., 2017b).

Cadmium: Cd showed high availability in the studied soils and the treatment with nZVI at 10% reduced its availability moderately, from 90 to 52% in the calcareous soil and from 96 to 78% in the acidic one. The low effectiveness of nZVI to immobilize Cd can be explained by its redox potential: Cd(II) has a standard oxidation-reduction potential (-0.40 V) that is very close to that of Fe(II) (-0.41 V); therefore, the removal mechanism of Cd(II) is purely sorption or complex formation (Li and Zhang, 2007).

Chromium: Cr availability was lower in acidic soil sample than in calcareous one, probably due to the lower soil pH value. Similar to As, Cr(VI) in soil is found as oxyanion and the presence of elements with opposite charge can increase its immobilization due to the increase of positive charge on the soil particles. After the application of nZVI at 5%, available Cr showed a decrease of 90 and 77% in the calcareous and acidic soil, respectively. The reaction mechanism can be the reduction of Cr(VI) to Cr(III) which precipitates as iron-hydroxides and/or oxyhydroxides. The Cr(VI) reduction is thermodynamically favorable because the standard potential of Cr(VI) is 1.36 V, which is more positive than the Fe (-0.41 V).

Lead: EX-Pb was negligible in the calcareous soil whereas was next to 31% in the acidic one. Better immobilization results at lower nZVI doses were observed in calcareous soil, which is probably due to the higher soil pH values at which is more favorable the sorption of metal cations. Moreover, under alkaline conditions, Pb can precipitate as phosphate. After the treatment with nZVI at the highest dose (10%), Pb in more available fractions was next to 5 and 7% in calcareous and acidic soil samples, respectively. Lead has a standard potential of -0.14 V, that is slightly more electropositive than iron, thus, Pb can react with nZVI by mechanisms, chemical reduction and sorption on the nZVI surface (Li and Zhang, 2007).

Zinc: Zn, as Cd and Pb, it is a cationic metal and it was more immobilized in untreated calcareous soil than in acidic one (Vítková et al. 2016; Gil-Díaz et al., 2017). In the calcareous soil the available Zn was mainly bound to CB fractions and the application of nZVI at 10% reduced the CB-Zn from 36% to 26%. In the acidic soil, at the same dose of nZVI, EX-Zn decreased more than 95% whereas CB-Zn significantly increased from 2% to 19%. Previous experiments have concluded that the main mechanism of Zn immobilization with nZVI is sorption or complex formation; reduction is not thermodynamically favored because Zn(II) has a standard potential of -0.76 V, which is more negative than that of Fe(II) (-0.41 V) (Li and Zhang, 2007).

Table 2. Concentration of metal(loid)s in EX and CB fraction at the different treatments in the calcareous soil.

Tessier fraction	nZVI dose	As (%)	Cd (%)	Cr (%)	Pb (%)	Zn (%)
EX	**0%**	9.70	70.0	50.6	0.54	1.39
	1%	5.10	64.3	23.5	0.44	0.74
	5%	0.70	31.5	2.24	0.00	0.18
	10%	0.17	21.9	1.51	0.00	0.00
CB	**0%**	20.9	20.5	4.17	31.7	35.8
	1%	14.1	25.0	3.80	32.9	32.9
	5%	3.67	23.5	3.20	9.91	28.5
	10%	2.03	30.5	2.95	4.99	25.6

Table 3. Concentration of metal(loid)s in EX and CB fraction at the different treatments in the acidic soil.

Tessier fraction	nZVI dose	As (%)	Cd (%)	Cr (%)	Pb (%)	Zn (%)
EX	**0%**	0.94	92.2	22.0	31.3	58.9
	1%	0.15	88.7	0.77	22.3	39.8
	5%	0.07	80.0	0.27	1.69	5.24
	10%	0.22	59.3	0.20	1.64	0.87
CB	**0%**	10.9	3.78	9.03	38.7	2.16
	1%	5.76	6.00	9.53	26.6	3.71
	5%	2.61	11.3	6.87	10.7	24.6
	10%	1.04	19.1	5.43	5.17	18.8

CONCLUSIONS

The effectiveness of nZVI to immobilize metal(loid)s in soils strongly depended on the metal(loid) characteristics, soil properties and dose of nZVI. The use of nZVI significantly reduced the metal(loid) concentrations in more available fractions (EX and CB), and the poorest results were obtained for Cd, especially in the acidic soil. Anionic metal(loid)s (As and Cr) were more easily retained in acidic soil, whereas cationic metal(loid)s (Cd, Pb and Zn), were immobilized more in calcareous soil. In general, a similar efficiency was found at 5 and 10% of nZVI for As and Cr whereas the efficiency was dose-dependent for the rest of metal(loid)s. The concentrations of As, Cd, Cr, Pb and Zn in more available fractions after the application of nZVI at 10% were 2, 52, 4, 5 and 26%, for the calcareous soil, and 1, 78, 6, 7 and 20% for the acidic one, respectively. Thus, the use of nZVI for Cd immobilization in polluted soils is not an adequate strategy whereas it can be a useful reclamation technique for calcareous or acidic soils polluted with As, Cr, Pb and Zn. Further experiments are necessary to evaluate the stability of the immobilization at long-term.

ACKNOWLEDGEMENTS

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6.11.P

IMPACT OF TWO ORGANIC AMENDMENTS ON HERBICIDES BEHAVIOR IN SOIL AND ON SOIL PROPERTIES: A COMPARATIVE FIELD STUDY

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ABSTRACT

The objective of this work was to study the effect of the application of two different organic amendments (composted spent mushroom substrate and green compost) on the soil properties and on the mobility of the herbicides chlorotoluron and flufenacet under field conditions. The results showed the ability of these organic residues i) to maintain/improve the soil´s physical and hydric properties; and ii) to preserve the groundwater from the contamination by herbicides.

1. INTRODUCTION

The application of organic residues rich in nutrients and organic matter (OM) to soil as organic amendments is a conservative agricultural practice often used to increase crop yields as well as to preserve the soil from degradation. Among the numerous residues potentially used as organic amendments are those from agricultural and industrial activities such as composted spent mushroom substrate (SMS) and green compost (GC). However, the OM of these residues can modify the dynamics and environmental fate of herbicides applied to the amended soils. Despite the potential environmental impact of the application of organic amendments along with pesticides in agriculture, no many field studies have assessed the effect of organic amendments on pesticides fate (Herrero-Hernández et al., 2011). Thus, the objective of this work was to study the effect of the addition of two different organic amendments, SMS and GC on the soil properties and on the mobility of the herbicides chlorotoluron and flufenacet under field conditions.

2. METHODS

2.1. *Amendments, soil, herbicides and tracer ion*

The organic amendments used were composted spent mushroom substrate (SMS) from mushroom cultivation, and green compost (GC) from pruning of plants from gardens and parks of Salamanca. The organic residues were supplied by Sustrates of La Rioja S.L. (La Rioja, Spain) and Viveros el Arca (Salamanca, Spain), respectively. The OM contents of both residues were 59.4% (SMS) and 46% (GC).

A sandy soil (14.9% clay, 4.7% silt and 80.4% sand) was selected to perform the field experiment. A 100-cm-depth soil profile relative to unamended and amended plots was physicochemical and hydraulically characterized. Soil water content at field capacity (θ_{FC}, ¸33 kPa) for each soil horizon were estimated from their soil texture, bulk density and soil organic matter content using the pedotransfer function of Rawls et al. (1982). The soil characteristics corresponding to the top (30 cm) at the beginning of the experiment are shown in Table 1.

Table 1. Characteristics of unamended and SMS- and GC-amended soils (0-30cm) at time 0 days.

Treatment	Depth (cm)	Bulk density (g cm^{-3})	pH	OC (%)	N (%)	C/N	θ_{FC} (cm^3 cm^{-3})
S	0-10	1.27	6.34	0.77	0.053	14.5	0.203
	10-30	1.40	6.62	0.91	0.073	12.5	0.195
S+SMS	0-10	1.02	7.11	2.53	0.237	10.7	0.290
	10-30	1.21	7.15	1.45	0.070	20.7	0.231
S+GC	0-10	1.10	6.99	1.63	0.136	12.0	0.249
	10-30	1.28	6.70	0.86	0.073	11.8	0.202

The herbicides studied were chlorotoluron and flufenacet, which are used to control the weeds in pre- and post-emergence in winter cereals. Both herbicides were used under the commercial formulations Erturon (chlorotoluron 50% p/v, Cheminova Agro S.A., Madrid) and Herold (Flufenacet 40% p/v, Bayer CropScience S.L., Valencia). The analytical standards of both herbicides (99.5% purity) were supplied by Sigma Aldrich Química S.A. (Madrid). Both chemicals have a moderate solubility in water, 74 mg L^{-1} for chlorotoluron, and 56 mg L^{-1} for flufenacet. However, chlorotoluron is characterized by a lower hydrophobicity (log Kow= 2.5) than flufenacet (log Kow= 3.2) (PPDB, 2017).

The dispersive characteristics of herbicides through the soil profile were assessed with the conservative tracer transport using bromide as tracer ion (KBr).

2.2. *Field experiment, extraction and analysis of herbicides and tracer ion*

A field study corresponding to the treatments unamended soil (S), soil amended with spent mushroom substrate (S+SMS), and soil amended with green compost (S+GC) was conducted by quadruplicate in experimental plots of 9 m × 9 m (12 plots). The study has been conducted in the experimental farm Muñovela of IRNASA-CSIC (Salamanca, Spain). The GC and SMS were applied at rate of 85 t ha^{-1} and 140 t ha^{-1} (dry weight basis), respectively, and incorporated to the 0-20 cm soil layer. Bulk density from 0 to 30 cm-depth was determined for each plot. Nine plots were instrumented with polyvinyl chloride tubes of 120 cm (length) × 5.2 cm (inner diameter) in order to measure the soil water content (SWC) in the soil profile periodically and to assess its temporal evolution each 20 cm from 20 to 100 cm-depth using an electric probe. Then, chlorotoluron and flufenacet, and bromide (tracer ion) were applied at 5, 15 and 53 kg a.i./ha, respectively, in 9 plots after winter wheat sowing in December 2016. One more plot per treatment was not treated with chemicals to be used as control plots for determination of soil biochemical parameters. Weather conditions were recorded throughout the experiment in a meteorological station placed at site. Temperatures varied from -11.6°C to 18.4°C (4.3°C mean temperature) during the eighty first days of the experiment and the cumulative precipitation corresponding to this period was 69.8 mm.

Soil profiles (0-100 cm) were sampled at 1, 17, 33, 60 and 80 days after herbicides and bromide application, then divided into 10 segments of 10 cm. The soil contained in each segment was sieved by 2 mm and moisture content of the bulk sample was determined. Triplicate of moist soil samples (6 g) were taken from each soil segment, sonicated for 1h and shaken for 24 h with 12 mL of acetonitrile or deionized water, centrifuged and filtered (< 45 µm) to determine the herbicides and the bromide ion, respectively. The herbicides were concentrated by evaporation of 8 mL of herbicide extracts until dryness under a nitrogen stream using an EVA-EC2-L evaporator (VLM GmbH, Bielefeld, Germany) and then, the residue was redissolved in 0.75 mL of acetonitrile. In the soil extracts, herbicides were quantified by high-performance liquid chromatography /diode array detector/mass spectrometry (HPLC-DAD-MS) (Waters Assoc., Milford, USA). Detection by HPLC-DAD was at 243 nm (chlorotoluron) and 232 nm (flufenacet). Herbicides determined by HPLC-MS were quantified by monitoring the positive molecular ions [m/z] 213.04 (chlorotoluron) and 364.03 (flufenacet). The retention times of chlorotoluron and flufenacet were 6.1 and 7.9 min, respectively. The bromide concentrations in the soil extracts were determined using a Metrohm ion chromatograph (Metrohm Ltd., Switzerland).

3. RESULTS

The application of both SMS and GC amendments to the soil modified the soil properties as well as the bromide ion and herbicides mobility. On the one hand, the amended soils showed a higher OC content than unamended soil at the top 30 cm (1.6-3.3-fold in S+SMS and 0.9-2.1-fold in S+GC) (Table 1). Soil amendments decreased the bulk density by factors of 1.2 for S+SMS, and 1.1 to 1.2 for S+GC relative to the unamended soil. In addition, an increase in the water holding capacity linked to the addition of the organic amendments was observed. The soil water holding capacity enhanced 1.2-1.4 times and 1.0-1.2 times after the application of the SMS and GC, respectively.

The distribution of bromide, chlorotoluron and flufenacet through the soil profile in the unamended, SMS- and GC-amended plots at 1 and 80 days after application is shown in Figure 1. Concentration profiles showed that mobility increased in the order flufenacet < chlorotoluron < bromide.

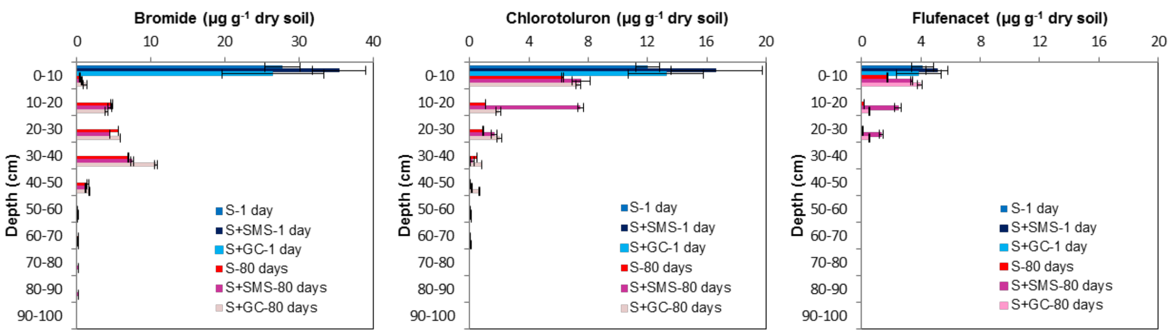

Figure 1. Distribution profile of bromide, chlorotoluron and flufenacet in unamended, SMS- and GC-amended plots at 1 and 80 days after application. Error bars represent the standard error of the mean value (n=3).

At 80 days after application, and following 69.8 mm of cumulative rainfall, amounts of bromide lower than 0.7% of the applied dose were detected at 60-70 cm-depth in S and S+GC treatments, and at 80-90 cm-depth in S+SMS plots. Independently of the treatment, the highest amounts of bromide were measured at 30-40 cm-depth. Chlorotoluron, the less hydrophobic herbicide, reached 50-60 cm-depth in S+SMS, and 60-70 cm-depth in S and S+GC (<0.6% of the applied dose), while the movement of flufenacet was limited to 20-30 cm-depth in all treatments. The highest amounts of both herbicides were determined at 0-10 cm-depth in both unamended and amended plots. However, the percentage of herbicides recovered at this depth was generally higher for amended soils than for unamended soils, showing S+GC the highest percentages. These percentages ranged from 45% (S+SMS) to 55% (S+GC) for chlorotoluron and from 42% (S) to 73% (S+GC) for flufenacet relative to the applied doses. These results agree with the higher OC content of amended soils, which could enhance the herbicides adsorption by soil and consequently decreased their mobility (Herrero-Hernández et al., 2011, 2015). Additional adsorption and dissipation studies of these herbicides along with microbiological studies are being accomplished at this moment in order to interrelate all these processes with the environmental fate of the herbicides.

4. CONCLUSIONS

The results obtained show that, the application of organic amendments contributes to improve the soil properties, and at the same time decreases herbicides mobility. Therefore, SMS and GC could help to reduce groundwater contamination by herbicides.

5. ACKNOWLEDGEMENTS

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6.12.P

FINE LEAF LITTER PRODUCTION AND NUTRIENT CONTRIBUTION OF VEGETABLE SPECIES IN MULTI STRATUM SILVOPASTORILIAN ARRANGEMENTS

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INTRODUCTION

The degradation of soils and grasslands is one of the most serious limitations of the livestock production system in the low tropics of the Colombian Caribbean Region (Mejía et al., 2008; Cajas et al., 2010). This leads to low availability and poor nutritional quality of native or introduced grasses, which in turn generates a low biological and economic efficiency of this production system, particularly during the prolonged dry season (Cajas, 2002; Cuadrado et al. ., 2005). In addition, this region is highly vulnerable to extreme weather events (floods, droughts, landslides, etc.), with consequent social, economic and environmental impacts (Boshell, 2008). Therefore, the primary productivity of these systems is severely restricted, as well as the recycling of nutrients due to the scarce production of litter in the pasture (Vitousek et al., 1995). Thus, silvopastoral systems are presented as one of the alternatives to overcome the described limitations of the traditional livestock production system in the Caribbean region in particular and in the tropical region in general (Cajas et al., 2008, Cajas et al. Al., 2010). The objective was to determine the annual fine leaf litter production and annual potential and actual nutrient input of plant species within a 13 - year - old multispecies silvopastoral arrangements compared to a grassland pasture in the Middle Valley of the Sinú River, Córdoba.

METHODS

This research was developed at the CORPOICA Turipaná Research Center, located in Cereté, Córdoba, Colombia (8 ° 51 'N, 75 ° 49' W, Altitude 18 m). The annual average rainfall is 1380 mm, with an annual average temperature of 28 °C, a potential annual evapotranspiration of 1240 mm and relative humidity of 81%. The study area corresponds to the tropical dry forest ecological life zone (Holdrige, 1967). Experimental plots of multiple strata silvopastoral systems established in 1998 (Cajas and Sinclair, 2001) were used. The experimental design was randomized complete blocks, treatments consisted of three arrangements of silvopastoral systems (A1, A2 and A3) of different structural complexity and diversity and a traditional prairie (A0), used as control. Each arrangement had three replications in plots of 2 ha each.

The treatments were**:**

Arrangement A0 (control pasture): This consisted of the combination of the angleton grasses -ANG- (*Dichanthium aristatum*) and mombasa -MOM- (*Panicum maximum*).

Arrangement A1: angleton pastures (*Dichanthium aristatum*) and Mombasa (Panicum maximum) associated with guasimo trees –GUA- (*Guazuma ulmifolia*), caña fistula –CÑF- (*Cassia grandis*) and Campano –CAM- (*Albizia saman*). The trees are at a distance of 16 m x 16 m.

Arrangement A2: Same as Arrangement A1 but includes forage shrubs forage (Leucaena leucocephala) and totumo (Crescentia cujete), planted at a distance of 4 m x 4 m.

Agreement A3: Same as Agreement A2 but also includes timber species ceiba tolúa -CEI- (Pachira quinata) and Caoba -CAO- (Swietenia macrophylla).

In order to measure the rate of fine leaf litter fall of the species within the silvopastoral arrangements, we used circular traps with an individual area of 0.45 m2. Leaf litter was manually taken from the area delimited by circles of 0.45 m2. The collected material was collected in the traps of each tree, classified by species and fractions, and its dry mass was then determined by the area unit (DM) (Jones and Case, 1990). The nutrient input in the leaf litter (LL) of each species and arrangement was determined from the DM and the concentration of each feed in the LL (kg ha-1 year-1). An analysis of variance was performed and when effects of the treatments were detected, the media were separated through the Tukey test. In both cases a level of significance (α) ≤ 0.05 was used.

RESULTS

Annual LL production was higher in silvopastoral arrangements (P <0.0001) than in the grassland. The grassland showed an LL production of 770 kg ha-1 year-1, whereas the arrangements including trees, shrubs and timber fluctuated between 2877 and 3148 kg ha-1 year-1, with no significant differences between them. Overall, this indicates that there are about four times more LL inputs in the silvopastoral systems annually than in the prairie. On the other hand, the amount of LL (dry basis - DM) depended on the plant species included in the evaluated silvopastoral arrangements (Figure 1). In the case of the pastures, the mean values of leaf litter production for MOM were 551 kg ha-1 year-1 and for ANG 218 kg ha-1 year-1, which were not different from each other (CV = 19.1%)

In all the observed periods, P was the nutrient that presented the lowest concentration in the LL, which varied between 0.05 and 0.25%. It should be noted that ANG, MOM, GUA and CEI recorded the highest concentrations (0.10 - 0.25%), while the lowest P content was presented in CÑF, CAM and CAO. In contrast, N was the nutrient with the highest concentration in this leaf litter fraction, ranging from 0.50 to 3.39%.

ANG and MOM grasses recorded the lowest N contents during the observed period (ANG: 0.79-1.29%, MOM: 0.51-1.04%), while CAM, CÑF and GUA stood out with the highest concentrations of N (CAM: 1.54 -3.29%, CnF: 1.38-2.41%, WF: 1.44-2.58%). Additionally, C represented about 50% of the dry mass of the pasture LL. Among the arboreal species the CAM excelled by its contributions of C; its leaf litter content fluctuated between 56.6 and 59.2%.

The potential return of nutrients through LL was statistically different between the evaluated arrangements. In general, the amount of nutrients contributed through leaf litter followed a pattern similar to that found in LL production (Figure 2). That is, in the silvopastoral systems the nutrients returned by leaf litter were significantly higher than those found in the grassland.

In the grassland, potential nutrient return through LL (kg ha-1year-1) followed the decreasing pattern: C 390> N 6.8> Ca 5.5> K 3.0> Mg 2.5> P 1.1. In contrast, the potential return of nutrients (kg ha-1 year-1) by leaf litter in silvopastoral arrangements averaged: C 1674, N 74, P 2.9, K 12.6, Mg 6.9 and Ca 40.7 (in arrangement A3) And 55.1 (in A1 and A2).

Figure 1. Annual leaf litter (LL) yield by species in multiple strata silvopastoral arrangements (A1, A2 and A3) and in a tree-less grassland (A0). Different lowercase letters on the columns indicate significant differences in the dry mass of LL between arrangements. Tukey test (P≤0.05).

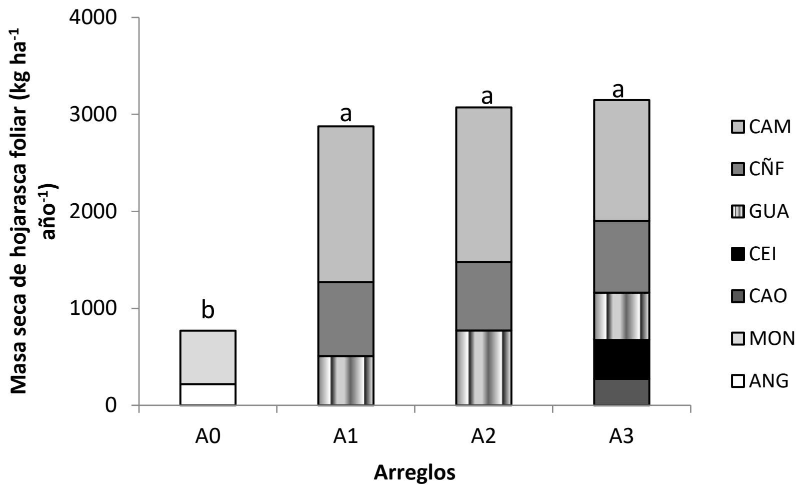
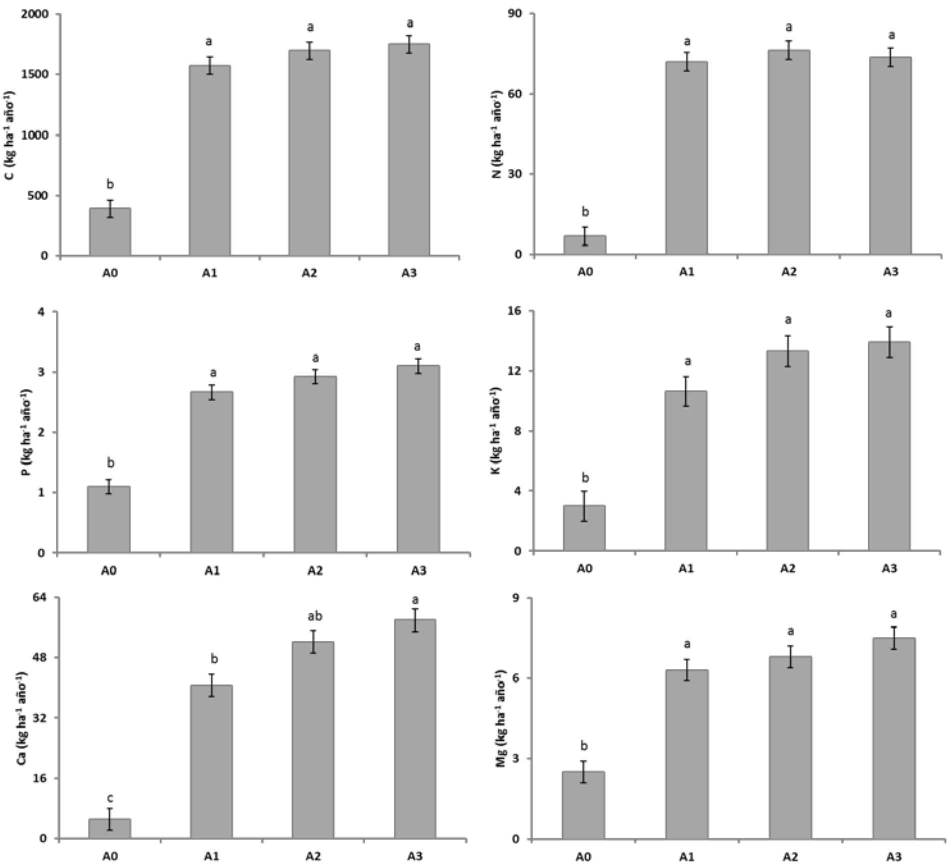


Figure 2. Potential annual return of nutrients through leaf litter in multiple strata silvopastoral arrangements in the middle valley of the Sinú River. *The vertical bars in each column represent the standard error. Different letters above the columns denote significant differences (P <0.05), and letters equal non-significant differences between the arrangements. A1, A2 and A3: Silvopastoral arrangements and A0: grassland without trees*



CONCLUSION

Silvopastoral arrangements generate more LL and nutrients than the grasses of a tree-less grassland. In addition, the litter supply of the species is independent of the silvopastoral arrangement in which it is found. In our knowledge the findings reported in this study are among the earliest carried out with these species in silvopastoral systems of multiple strata under the environmental conditions of the Sinú river valley, Colombian Caribbean.

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6.13.P

DAIRY SLURRY USED AS FERTILIZER MODIFIES SOIL POROSITY AND SHAPE OF PORES

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INTRODUCTION

Organic amendments (OM) are related to soil quality (*Thangarajan et al.*, 2013). Physical fertility is affected as OM promotes important changes in the physical soil properties such as porosity. Pore space governs critical aspects of almost everything that occurs in the soil: the movement of water, air, and other fluids; the transport and the reaction of chemicals; and the residence of roots and other biota (*Nimmo*, 2005). The description of porosity and its quantification requires working with undisturbed samples. Micromorphology constitutes a useful tool for such studies. In micromorphology procedures (*Stoops*, 2003), undisturbed soil samples are impregnated with resin and thin soil sections (30 µm thick) are obtained. As a consequence, materials become translucent to light. Such thin sections can reveal small changes in the soil before they can be detected by physical or chemical analysis, making this method a valuable and powerful tool to monitor experimental studies and understand incipient processes in soil (*Stoops*,2010).

In thin sections a wide range of soil pore sizes and shapes can be observed, either between or within aggregates (*Dalal and Bridge*, 1996). Besides, their changes over time can be evaluated by image analysis.

Microporosity, mesoporosity and macroporosity are terms referring to the size and functionality of pores; however, the boundaries between these terms are diffuse. Microporosity comprises pores with diameters of less than 30 µm and is related to soil holding capacity, while mesoporosity (diameters between 30 and 75 µm) and macroporosity (diameters >75 µm) are linked to soil water drainage (*Brewer*, 1964). Large pores (>30 µm) include biopores, cracks, and pores between aggregates. Pore size and pore quantity affect soil's quality and its concentration of organic carbon. Conversely, soil organic carbon and soil texture also influence porosity (*Thomsen et al.*, 1999). Biopores are important as they improve the diffusion of fluids (gas and liquid water diffusion), which affects decomposition. Besides, small pores protect soil organic carbon against microbial decomposition by limiting both microbial access and gas diffusivity (*Thomsen et al.*, 2003).

Our research aims to evaluate the effect of two dairy slurry application rates on macroporosity and on the pores' shape.

METHODS

Description of the study area

This study was conducted at field level in Tallada d'Empordà (Girona, Spain), where the climate is dry Mediterranean: annual average temperature is 15.8º C and annual average precipitation is 602 mm. The soil is very deep (>1.2 m), well drained, non-saline, calcareous, and it is classified as a Oxiaquic Xerofluvent (*Soil Survey Staff*, 2014).

Experimental design

A double-annual forage cropping system (ryegrass-maize) was maintained during six years (2006- 2012). From 2013 onwards an annual grain rotation: maize-oilseed rape, was introduced. Maize (*Zea mays*), as a summer crop, was irrigated in both rotations. In the 2015-2016 cropping season oilseed rape (*Brassica napus* L.), as a winter crop, was established.

Treatments were a control (without N application) and two dairy slurry rates equivalent to 170 and 250 kg total-N ha^{-1} yr^{-1}. Organic fertilizer was applied at sowing. At tillering, a mineral fertilizer as a complement was applied (Table 1). Treatments were distributed in three randomized blocks.

Table 1: Fertilizer treatments.

Treatment	Winter crop		Summer crop	
	kg total- N/ha using dairy slurry at sowing	**kg total- N/ha using mineral fertilizer at tillering**	**kg total- N/ha using dairy slurry at sowing**	**kg total- N/ha using mineral fertilizer at tillering**
T-1	0	0	0	0
T-5	70	80	100	200
T-2	100	50	150	150

Soil thin sections

At oil seed rape physiological maturity (2nd June 2016), one prism (0.06*0.09*0.19 m^3) of undisturbed soil was collected from the surface (0-0.06 m deep) for each plot. As the field trial includes three blocks, nine undisturbed soil rectangular prisms were obtained. From each prism, a horizontal thin section (0.05 m high by 0.13 m wide) was obtained using the procedure described by *Stoops* (2003).

Porosity parameters

For each thin section, two areas were photographed (3.15 x 4.20 cm) with an Olympus® C-7070 Wide Zoom camera. For each area, three images were obtained: i) an image using ordinary light and parallel polarisers; ii) another image using ordinary light and crossed polarisers; iii) and another one using incident UV light. The latter image was binarized using ImageJ® software developed by *Rasband* (2012).

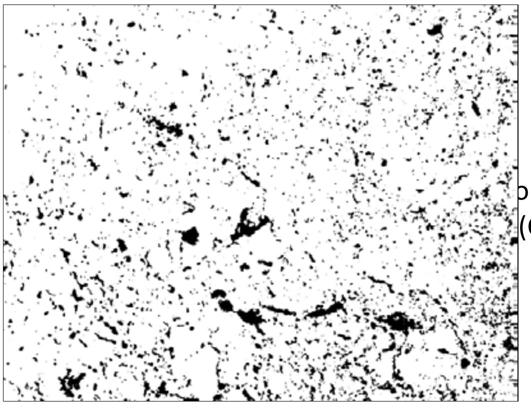
The pore size distribution were determined thanks to an algorithm of mathematical morphology using Quantim4 library (*Vogel*, 2008) for six ranges of apparent pore diameter (APD): over 30 µm, 30- 60 µm, 60-100 µm, 100-200 µm, 200-400 µm and over 400 µm.

Four shape descriptors defined in *Ferreira and Rasband* (2012) (Circularity, Axis ratio, Roundness and Solidity) were determined, thanks to ImageJ® software, in five ranges of apparent diameter: 30-60 µm, 60-100 µm, 100-200 µm, 200-400 µm and over 400 µm.

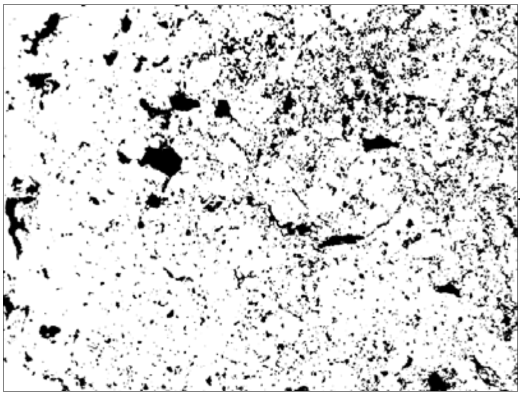
RESULTS

Preliminary results show that porosity increased (pores bigger than 30 µm) as the amount of organic matter applied increased (Figure 1a, b, c).

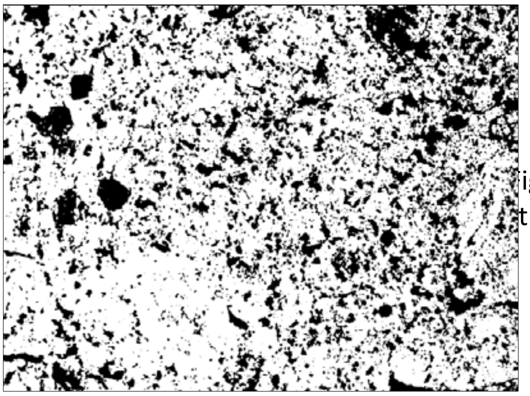
The organic matter applied also enhanced aggregation. The presence of pores with an apparent diameter higher than 400 µm was reduced in the control (Figure 1a).



ore space of a standard image of the treatment T-1 (Control).



igure 1b: Pore space of a standard image of the treatment T-5 (170 kg N/ha using dairy slurry).



igure 1c: Pore space of a standard image of the treatment T-2 (250 kg N/ha using dairy slurry).

As the amount of organic matter increased, preliminary results show that circularity decreases in pores higher than 400 µm.

CONCLUSIONS

Dairy slurry application increases soil porosity. In Mediterranean agricultural systems, the use of slurry from livestock, as fertilizer at agronomic rates (170-250 kg N /ha) is a good practice in order to maintain sustainability.

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F. Domingo-Olivé (IRTA-Mas Badia) was responsible for field experimental design, field maintenance and sampling, A.D. Bosch-Serra (UdL) established the aim of this experiment and reviewed it as scientific coordinator, A. Valdez-Ibañez (UdL) obtained soil images and classified them and N. Mateo- Marín (CITA) obtained the numeric database from images, performed the statistical analysis and wrote the extended abstract for discussion within authors.

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6.15.P

PHOSPHORUS SOIL CONTENT AS AN INDIRECT INDICATOR OF MANURE MANAGEMENT

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INTRODUCTION

Phosphorus (P) is one of the most important nutrients to ensure food production. P is essential on field crops to achieve the optimal yield. However, an inadequate use of P fertilizers can produce undesirable effects on the environment, especially to water quality, contributing to eutrophication of freshwater systems.

P can affect both surface and underground water. P flowing to surface water is primarily a function of soil P availability and the processes of runoff and erosion (van Es et al., 2004). Also underground water may be affected, mainly due to P leaching, which may occur under certain circumstances, for example, high historical P applications (Schoumans and Groenendijk, 2000) or high mineralization of organic matter in organic soils (Duxbury and Peverly, 1978).

Manure application is the main P input on agricultural land in Catalonia, where 47% of the total P applied comes from livestock (Figure 1). This balance (Figure 1) shows that P applied doubles crop uptake and results in a surplus of 19.287 tonnes of this nutrient (MAPAMA, 2015; data from 2013) which is mostly accumulated in the upper horizons of farmlands. As a consequence, there is a high risk of P pollution of fresh water from fields through runoff and erosion. This situation might be worse in such Mediterranean climate with dried periods followed by torrential rainfall.

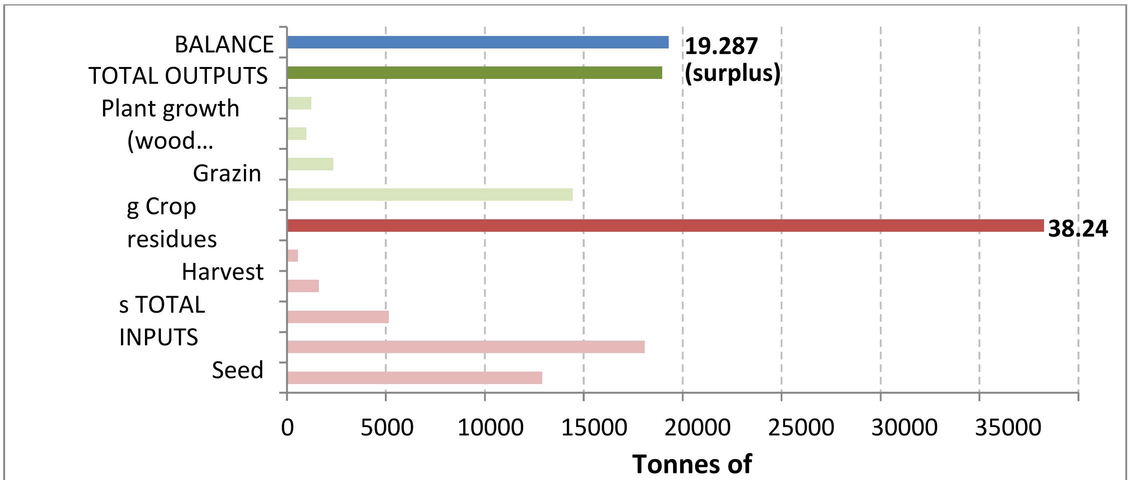

Figure 1. Phosphorus balance in Catalonia, year 2013 (MAPAMA, 2015)

Villar et al. (2015) detected higher values of P in the drainage water in comparison with the contents of the irrigation water and identified a connection with the application of manure and mineral phosphate fertilizers.

The LIFE project *Futur Agrari* works to help farmers making decisions about fertilization management in order to maximize the agronomic and economic benefits and minimize the potential environmental impact.

METHODS

The area

Irrigation was implemented in the Algerri-Balaguer scheme about 20 years ago to complement the limited amount of rainfall, which is about 400 mm per year. Irrigation transformed the area and made possible to grow intensive crops like maize and rotations of barley plus maize with high nutrient requirements. Livestock also increased in the area and manure is usually applied before crop's sowing.

The project

The LIFE+ project *Futur Agrari* deals with manure management in Catalan areas with a high number of intensive pig farms. During a 4-year period (2014-2017), several maize plots have been surveyed, soil sampled and advised in order to implement good agricultural practices to improve soil and water conservation in the Algerri-Balaguer irrigation scheme.

Samples and analysis

Close to 900 ha (Figure 2), which belong to 20 farmers, have received fertilizer advice each year by technicians on the project. Soil samples from 0 to 30 cm were taken in 2014 to evaluate different soil fertility parameters at the beginning of the project. Composite samples included a minimum of 8 sub-samples for every 3-4 hectares. Available P, using the Olsen method, and K (ammonium acetate method) was determined in 102 maize plots.

Figure 2. Sampled plots (in blue) of the studied area. Algerri-Balaguer (Catalonia).

RESULTS

High levels of P were observed in the upper layers (0-30 cm) of agricultural soil, with a high potential to be lost through soil erosion. The average value of all the project samples was 43 mg*kg^{-1} (Figure3), which is considered a very high level of available P according to most of the existing interpretation tables for the Olsen method (Cottenie, 1980; Mallarino and Sawyer, 2013; Horneck et al., 2015) considering as high level those above 25 mg*kg^{-1}. Close to 60 % of the top soil samples had an available P content higher than that limit and occasionally it exceeded 100 mg*kg^{-1} for some plots. As part of the LIFE project, soil will be resampled during 2017 and results used as an indicator to evaluate the effect of the implemented actions in the area during these 4 years.

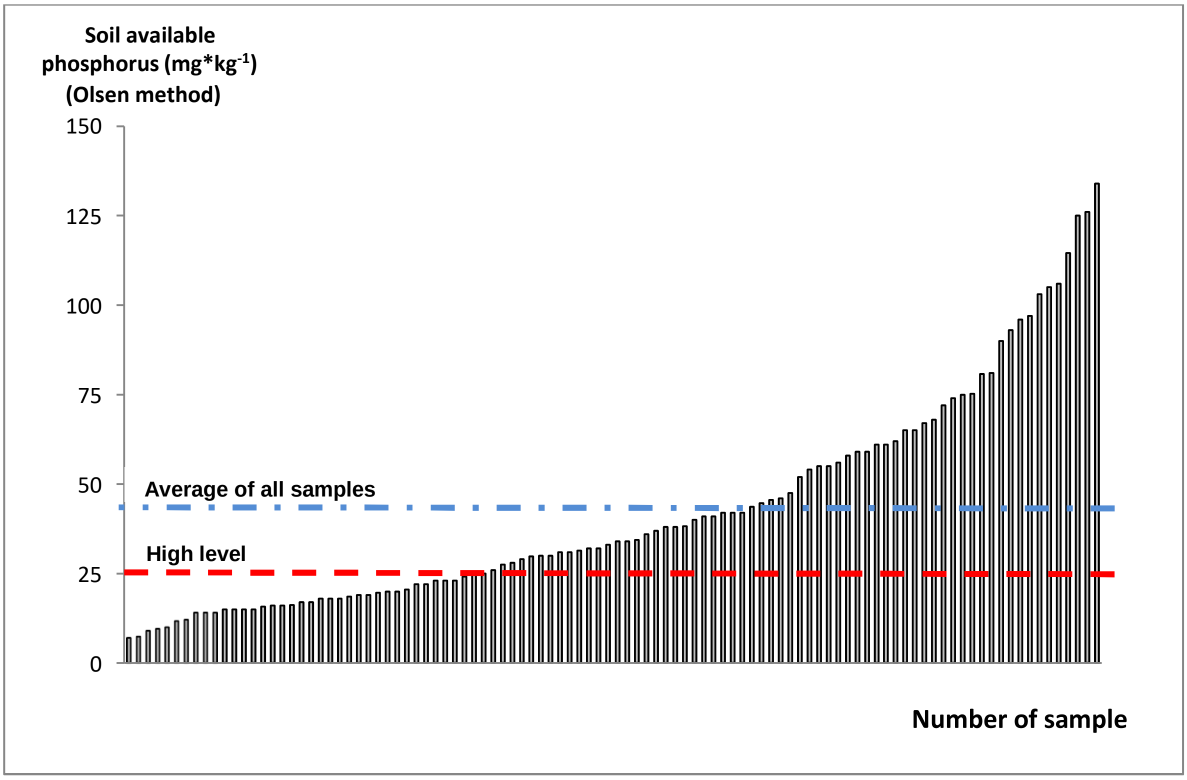

Figure 3. Soil available phosphorus (mg*kg^{-1}) content of samples from the studied area, 2014. Red dashed line corresponds to high level and blue dashed line corresponds to the average of the samples.

Since the amount of P required by crops is lower than the amount received from manure application, P accumulates in the upper layers of the soil. Therefore, nutrient imbalances, especially for P, are frequently linked to mixed farming based on manure applications, whereas no surplus of such element is found in soils that receive mineral fertilizers (Figure 4).

The inefficient use of manure is partially caused by the disproportion of nutrients in manure, the large variability according to the source, the difficulty in estimating nutrient availability, the high density of farms in some areas and the low precision to calculate and apply the appropriate rate for each crop and field situation.

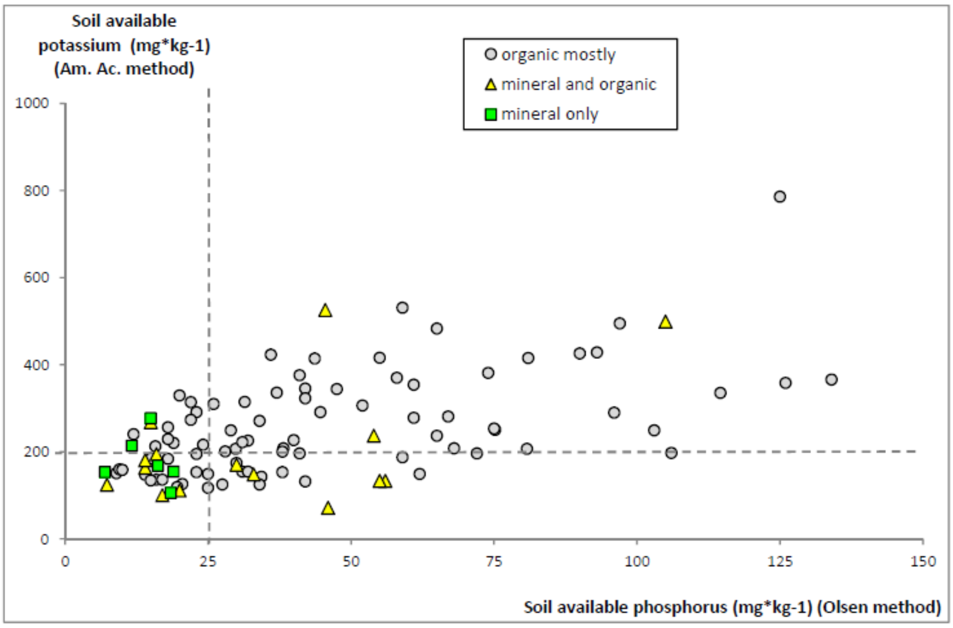

Figure 4. Soil available phosphorus and potassium content (mg*kg-1) according to 3 fertilization managements: only mineral (green square), mineral and organic (yellow triangle) and mostly organic (grey circle). Grey horizontal dashed line corresponds to threshold for high level of available phosphorus and grey vertical dashed line corresponds to threshold for high level of available potassium.

ACKNOWLEDGMENT

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6.16.P

MODEL FOR DEGRADED LAND RESTORATION ON REGIONS UNDER HYDRIC STRESS, IN RIO DE JANEIRO STATE, BRAZIL

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INTRODUCTION

One of Brazil's aims is to recuperate 15,000,000 ha of degraded pastures, between 2010 and 2020, although there are around 30,000,000 ha with some degree of degradation. The term *recuperate* is used indiscriminately and does not make distinction between the lands which will return to the productive system and those which will be reforested. In Rio de Janeiro State, some municipalities use large fractions of their territories for cattle ranching, usually not respecting springs and other Permanent Preservation Areas which are protected by the Brazilian Forest Code (Law nº 12651, of 25th May 2012) and have to be reforested according to respective legal requirements.

The recovery programs for degraded areas aim to improve biological productivity and reduce the additional pressure on forest remnants (Lovejoy, 1985); and interrupt the loss of sediments and restore the perenniality of rivers (Korte and Kearl, 1993). Some landscapes in the state of Rio de Janeiro resemble badlands due to the high degree of land degradation, with the need to initiate a process of recovery of degraded areas. But what is the environmental, economic, and social return of implementing a large-scale RDA program? Korte and Kearl (1993) wondered what would be the effects on the production and quality of water for the basin as a whole, if the restoration were done on a large scale, that is, small watersheds inserted into a larger river basin? Would restoration of watersheds across the basin be cost-effective? The authors conclude that due to the high environmental return, large-scale restoration of watersheds should be a public policy.

Batchelor et al. (2015) evaluated the effects of cattle disposal in riparian systems after 23 years of cattle removal. The authors' results indicate that channels widths and erosion banks decreased by 64% and 73% of sites, respectively. The exposed soil area (bare) was reduced by 90% and there was a 63% reduction in the number of exposed river channels. For the authors the removal of livestock can result in dramatic changes in riparian vegetation, even in semi-arid landscapes and without the introduction of degraded area recovery programs or other active restoration efforts.

Therefore, the aim of this research is to present a proposal for defining which areas have to be restored in environments under hydric stress, using the example from *Santo Antonio de Pádua* Municipality (Figure 1), which has 82.9%, or 505.61 Km^2 of its territory covered with pasture.

METHODS

In order to facilitate the execution of the objective of this research, it was necessary to create a database that would allow the delimitation of priority areas for restoration in detail scale. The information provided by the Brazilian Institute of Geography and Statistics (IBGE) at a scale of 1: 50.000 does not meet the objectives of the work, as several first-order channels in the range of 1.50.000, are not mapped, precluding the quantification of total areas to be restored. To do so, it was necessary to use the Digital Elevation Model (DEM) at a scale of 1: 25,000 and orthophotos referring to Santo Antônio, Digital Elevation Model that are also available on the IBGE website. The Digital Elevation Model served as a basis for mapping the drainage at 1: 25.0000 scale. In addition, the DEM was also used for the processing of contour lines, generated with equidistance of 10 meters.

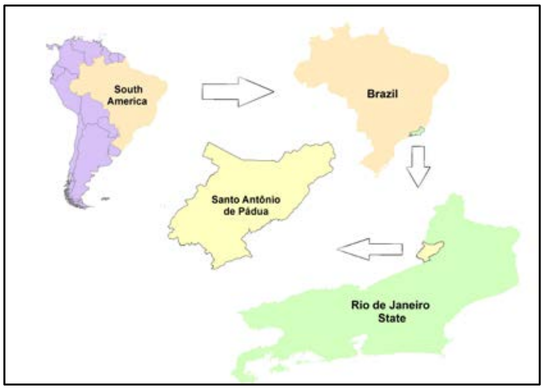

Figure 1. Location of *Santo Antônio de Pádua* Municipality (Rio de Janeiro State, Brazil)

The process of drainage network extraction was done in Esri's ArcGis® 10.1 software using the Spatial Analyst_Hidrology extension and was based on Souza and Almeida (2014). At first, we used the tool "Fill" that creates a raster from the filling of possible gaps existing in the DEM. From this new raster, the direction of the water flow in the drainage network was determined by the "Flow Direction" function. This tool creates a new raster from the determination of the flow direction based on the line of greater slope of the terrain. Next, the accumulated flow was determined; this parameter indicates the degree of confluence of the flow and can be associated to the factors of ramp length and horizontal curvature. For this purpose, the "Flow Accumulation" function was used. The "Stream Order_Strahler" and "Stream to Feature" tools, to determine the orders of the generated channels and to convert the raster file to a vector file, respectively.

In general, the automatic extraction of the drainage presented satisfactory results, however, in the flat areas, some channels were identified as errors. The corrections of the drainage system were made manually in ArcGis® 10.1 software with visual control on the scale of 1: 25,000. To assist the corrections were used contour lines and orthophotos (IBGE) of the municipality.

After this step the segmentation of the first-order channels was done, with the delimitation of a buffer of 15 meters on each side of the channel and 50 meters around the springs. The data were tabulated in Microsoft Excel ® worksheet to quantify the total area to be restored.

RESULTS

The restoration of degraded areas is a necessity in several regions of Rio de Janeiro. However, the climatic characteristics of each region impose different models to increase the efficiency of the programs of recovery of degraded areas. The municipality of Santo Antônio de Pádua has a high hydric deficit between February and October, but even in months where higher precipitation volumes are expected, high hydric deficits may occur. The average annual rainfall ranging between 1001-1210 mm.y-1. The month of August usually presents the highest water deficit with 40 mm (Brandão et al., 2016).

The decision of starting the process of forest restoration at springs, recharge areas and first-order channels is due to the higher availability of water within the soil, making it easy to carry out degraded areas restoration projects, minimizing the seedling losses and reducing the cost of restoration. Restoration actions outside these areas increase the losses of seedlings that can reach 50%, making recovery of degraded areas very expensive.

The figure 2 shows the map with the drainage at a scale of 1:25,000. The total amount of first-order canals is 1,751; with a total length of 806.70 km. A buffer zone of 15 meters has been established along each bench of the canals, comprising 25.42 km^2 (2,542 ha), which corresponds to 4.2% of the municipal territory.

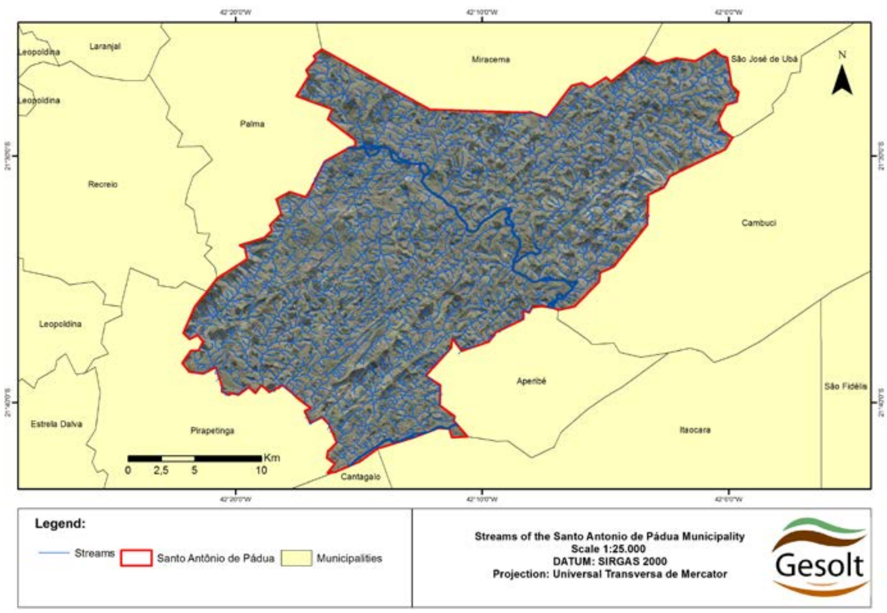

Figure 2. Streams of the Santo Antonio de Pádua Municipality.

The comparison between the drainage network at 1: 25,000 and 1: 50,000 scales shows that the total of channels at a 1: 25,000 scale is 1,431.33 km, with drainage density of 2.37 km.km-2, while at a scale of 1: 50,000 is 1,269.70 km and drainage density is 2.10 km.km-2. The efficiency of the model can be evaluated by restoring an experimental area of 0.66 ha (Figure 3). The restoration began in August 2012 with the installation of a fence in the area to be restored to keep out cattle ranching and planting six species native to the Atlantic Forest and after four years the area is completely covered by woody vegetation and restored as a recharge and spring area.

Figure 3. Spring reforested in Aug. 2012 (left) and the result 4 years later (Nov 2016 - right).

It is believed that with this model, the value to restore 1 ha is less than US $ 1,000. It is even possible to reduce this value by adopting cheaper but more time-consuming measures, such as artificial perches (Tomazzi et al., 2010) and assisted natural regeneration with enrichment planting of desired species (Shono et al., 2007). Considering the amount of US $ 3,000 spent per hectare, the total financial investment for the springs, recharge areas and first-order channels would be US $ 7,625,934.00.

CONCLUSION

The research aimed to present a forest restoration model in environments under water stress. The choice to start reforestation at the springs was reliable, because in four years the restored area was shown through visual monitoring that was already stabilized. Furthermore, native vegetation cover at spring areas remarkably contributes to groundwater recharge in first order catchments (Braumann et al., 2012) It is noteworthy that even under severe conditions of water stress, the higher humidity in the springs and valley bottoms of the first-order channels contributed to the reduction of costs and improved the efficiency of the restoration.

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6.17.P

POTENTIAL OF ROCK DUST AND SEWAGE SLUDGE TO IMPROVE SOIL FERTILITY IN DEGRADED BRAZILIAN PASTURES

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INTRODUCTION

Atlantic Rainforest is one of the most threatened biomes in Brazil. Due to proximity to the most populated and economically developed areas, much Atlantic Rainforest has been cleared for urban development, mineral exploitation, farms and cattle ranges. When Brazil was discovered, in 1500, this biome totalled 1,315,460 km² (15% of Brazil). Nowadays, only 8.5% remains of the original forest, considering remnants covering >100 ha, and 12.5%, considering remnants >3 ha (Fundação SOS Mata Atlântica, 2015). Deforestation has mainly occurred for the production of Pau-Brasil (*Paubrasilia enchinata*), sugar cane, coffee and for gold extraction. More recently, the fragmentation of Atlantic Rainforest has been promoted by the expansion of rangelands for cattle.

This research has been conducted in *Santo Antonio de Pádua* Municipality, situated in north-western Rio de Janeiro State, Brazil (Figure 1). Some 13% of the Municipality is forest, including fragments of all sizes. Some 83% of Municipal territory is used for pasture, especially on slopes, river banks, uplands and around springs. Although within the Atlantic Rainforest biome, the Municipality has serious water shortages (Brandão et al., 2016), which poses problems for farming activities.

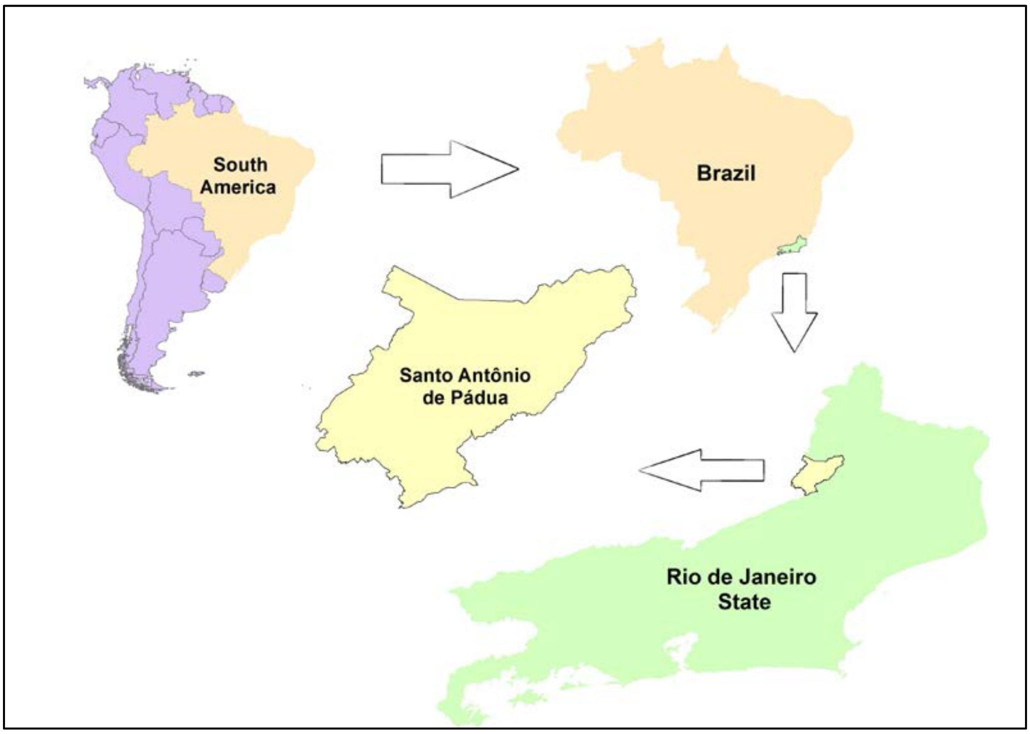

Figure 1. Location of *Santo Antônio de Pádua* Municipality (Rio de Janeiro State, Brazil).

The Municipality has various industries, including ornamental rock quarries. This has caused serious environmental degradation, because most waste was dumped into local rivers. Rock dust utilized in this experiment consists of reused quarry residues. When mining companies became legally obliged to collect and store the rock dust, some vegetables began to grow on sites where the rock dust had been stored. Vegetal growth would only have been possible with the presence of nutrients or due the plant resistance.

There are several programmes in Brazil to rehabilitate degraded lands. Amongst the multiple costs involved in land rehabilitation, there is the cost of plant seedlings and fertilizers. In Brazil, this cost can exceed US$ 3,000 per hectare. Such costs make it difficult for small stakeholders to implement environmental protection projects. In order to reduce costs, NGOs request donations for the costs of seedling, but famers must still pay for fertilizers and labour. Therefore, this research work assesses the application of rock dust and sludge as sources of nutrients to improve degraded pasture soils.

Rock dust applications are becoming relatively common in environmental improvement projects in Brazil (Amparo, 2013; Rezende et al., 2013; Jorge and Souza, 2015). Applications can improve soil nutrient budgets and can be used by stakeholders who practise agroecology and organic agriculture (Haraldsen and Peersen, 2003; Silva et al., 2005, Jones et al., 2009; Souza et al., 2013). Another benefit is the gradual release of nutrients, thus minimizing potential water pollution.

METHODS

Before material collection, several visits were made to the companies, and in all of them plant growth could be observed in the rock dust storage area. Rock dust is more efficient when applied together with organic compost (Silva et al., 2012). Initially, we planned to use sludge, due to its high organic content, but the Municipality does not treat its sewage. Therefore, we used sludge from the local water treatment station. Experiments were conducted using different combinations of Soil (S), Rock Dust (RD), and Water Treatment Station Sludge (WTSS).

Soil samples were taken from the B horizons of Ultisols, which are the dominant soil types within the Municipality. This choice was due to this horizon being exposed after severe sheet erosion. The utilized rock dust was collected from a local company, which only uses Mylonite gneiss (local name: *pedra paduana cinza*). This rock dust has a similar texture to silt (97% <0.075 mm, 87% <0.044 mm). The sludge was collected during the maintenance of the treatment station. It was then dried on a conveyor and powdered.

Soil samples, rock dust and water treatment station sludge, and mixtures of these components underwent laboratory analysis. The mixtures used were: (1) S/RD, 50% each component; (2) S/RD/WTSS, 12.5, 25 and 50%, respectively, of RD and WTSS; (3) S/WTSS, 12.5, 25 and 50%, respectively, of WTSS; and (4) RD/WTSS, 12.5, 25 and 50%, respectively, of WTSS.

RESULTS

The analytical results indicate the low soil fertility and the high potential of the rock dust and WTSS (Table 1). The soil had relatively low pH (5.8), low phosphorus (P) content (2.9 $mg.dm^{-3}$) and low base saturation (V) (21.9%). These results generally concur with results from weathered Ultisol B horizons. In comparison, rock dust had high pH (7.3), high P and potassium (K) contents of 362.4 $mg.dm^{-3}$ and 524.0 $mg.dm^{-3}$, respectively (Figure 2). Base saturation is rather high (86.4%). WTSS was rather acidic (pH 5.3). It was believed that P content would be higher, but the concentration was rather low (18.6 $mg.dm^{-3}$). K content was 136 $mg.dm^{-3}$ and base saturation was 38.3%.

On all samples, in which rock dust was added, independent of its concentration, pH was close to neutral. The S/RD sample of 50% each increased base saturation to 74.3%. Although when the mixture involved only S/WTSS, there was no increase in base saturation, being 23.8%, with 12.5% WTSS. The aluminum (Al) saturation was between 31.7-40.9% for the S/WTSS.

The different combinations of RD and WTSS exhibited the best chemical indices for soil fertility, although the high pH value (7.7) might diminish P availability. Higher values for P and K occurred at the concentration of 25% for WTSS, with 594.6 $mg.dm^{-3}$ and 524.0 $mg.dm^{-3}$, respectively. Base saturation reached 86.9%. With 12.5% WTSS, P content was slightly higher than with RD only (382.9 $mg.dm^{-3}$).

The addition of RD and WTSS improved all chemical indices within the soil. Base saturation increased to 81.6, 83.0 and 79.9%, with 12.5, 25 and 50% RD/WTSS, respectively.

It was postulated that WTSS addition could increase soil organic matter content. WTSS had an organic content of 146.00 $g.kg^{-1}$. However, its concentration in the different combinations utilized, was ≤5 $g.kg^{-1}$, when 50% WTSS was added.

Table 1. Soil chemical analyses of the soil rock dust and sludge mixtures

Sample	pH (H_2O)	P	K	Na	Ca	Mg	Al	H + Al	SB	(t)	(T)	V	m	OM
		$mg.dm^{-3}$			$cmol.dm^{-3}$				$cmol.dm^{-3}$			%		$g.kg^{-1}$
S	5.8	2.9	36	-	0.2	0.3	0.4	2.1	0.6	1.0	2.7	21.9	40.8	3.0
RD	7.3	362.4	524	-	2.5	0.5	0.0	0.7	4.3	4.3	5.0	86.4	0.0	3.0
WTSS	5.3	18.6	136	-	3.1	0.4	1.3	6.3	3.9	5.2	10.2	38.3	25.0	146.0
S/RD	6.9	183.4	362	-	2.0	0.5	0.0	1.2	3.4	3.4	4.6	74.3	0.0	3.0
S/WTSS (12.5%)	5.3	3.8	34	-	0.2	0.3	0.4	1.9	0.6	1.0	2.4	23.8	40.8	4.0
S/WTSS (25%)	5.2	3.5	36	-	0.2	0.3	0.4	2.1	0.6	1.0	2.7	21.8	40.9	2.0
S/WTSS (50%)	5.1	5.5	36	-	0.2	0.3	0.3	2.1	0.6	0.9	2.7	23.7	31.7	5.0
RD/WTSS (12.5%)	7.7	382.9	524	-	2.6	0.4	0.0	0.7	4.4	4.4	5.1	86.6	0.0	6.0
RD/WTSS (25%)	7.7	594.6	524	-	2.7	0.5	0.0	0.7	4.5	4.5	5.2	86.9	0.0	4.0
RD/WTSS (50%)	7.7	404.5	512	-	2.6	0.5	0.0	0.6	4.4	4.4	5.1	87.9	0.0	5.0
S/RD/WTSS (12.5%)	7.0	190.1	368	-	2.7	0.6	0.0	1.0	4.2	4.2	5.2	81.6	0.0	3.0
S/RD/WTSS (25%)	7.0	288.6	418	-	2.6	0.4	0.0	0.9	4.2	4.2	5.0	83.0	0.0	3.0
S/RD/WTSS (50%)	7.2	11.5	356	-	2.4	0.5	0.0	1.0	3.8	3.8	4.7	79.9	0.0	5.0

Captions. P – Na – K – Fe – Zn – Mn – Cu - Mehlich extractor 1; Ca – Mg – Al – Extractor: KCl - 1 mol/L; H + Al – Extractor: SMP; SB = Sum of Bases; CTC (t) - Cation Exchange Capacity; CEC (T) - Cation Exchange Capacity pH 7.0; V = Base saturation; m = Aluminum saturation; Organic Matter (OM) – Oxidation: $Na_2Cr_2O_7$ 4N + H_2SO_4 10N.

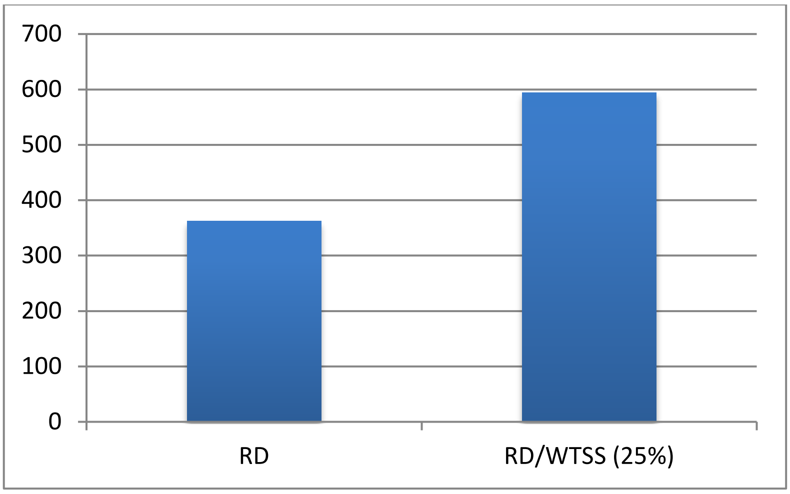

Figure 2. Phosphorus content ($mg.dm^{-3}$) in rock dust and in the 25% rock dust/WTSS combination.

CONCLUSIONS

The results showed that RD and WTSS additions can increase soil fertility. The local availability and the low cost of these residues may provide an economic case for their utilization in land improvement projects. It is possible that other organic residues, especially those with higher organic contents, might be combined with rock dust and increase soil fertility more.

ACKNOWLEDGEMENTS

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6.18.P

PHOTODEGRADATION EEFECTS ON DECOMPOSITION OF ORGANIC AMENDMENTS. APLICATION IN DRYLAND RESTORATION

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INTRODUCTION

Drylands cover more than 40% of the global land surface and are home to a third of the global population (MEA, 2005). These areas are highly vulnerable to land degradation; thus developing cost-effective large-scale strategies to restore these landscapes becomes critical to preserve their biodiversity, functionality and resilience to climate change (Kildisheva et al., 2016). Belowground processes and combined aboveground-belowground interactions are critical to ensure the functionality of soils and ecosystems (Muñoz-Rojas et al., 2016a). Nevertheless, knowledge of soil processes and functions in arid ecosystems is currently limited, and often, not effectively addressed in restoration programs. The use of organic amendments and mixtures of soil substrates has been proposed as an effective strategy to increase the levels of organic matter and water holding capacity of degraded soils. These approaches have proved to increase plant recruitment and recover soil fertility and quality, thereby enhancing biogeochemical nutrient cycles (Bateman et al., 2016; Luna et al., 2016; Muñoz-Rojas et al., 2016b).

In drylands, water availability plays a fundamental part in controlling biotic ecosystems processes (Oyonarte et al., 2016). However, recent evidence suggests that other decisive drivers take part in these processes. Unexplained high rates of litter decomposition, nitrogen mineralization, soil enzymatic activity and carbon turnover have been observed in drylands. These observations have been partly explained by photodegradation, a process that consists of the breakdown of organic matter via solar radiation (UV) and that can increase decomposition rates and lead to changes in the balance of carbon and nutrients between plants, soil and atmosphere (Austin et al., 2016). Photodegradation can have a critical impact on the decomposition rates of organic amendments used in dryland restoration applications. However, the role of this important abiotic process on restoring water-limited environments is yet to be harnessed.

In this research, we conducted a multi-site field experiment to test the effects of photodegradation processes on decomposition of organic amendments with potential use in ecosystem restoration of drylands.

METHODS

The study was carried out during 12 months in two biodiverse semi-arid areas that are affected by mining activities: the Pilbara bioregion in North Western Australia (22°17’ S, 118°27’ E) and the Cabo de Gata Nijar Natural Park, South East Spain (36°43' N, 2°11’ W). In the Pilbara (NW Australia), mean annual rainfall ranges between 250 and 400mm, mostly concentrated in the summer months (December to March). This rainfall originates from sporadic summer convection thunderstorms and tropical cyclones. Mean annual temperatures range between 19.4 and 33.2°C with average maximums over 40◦C in the summer period. In Cabo de Gata-Nijar (SE Spain), mean annual precipitation is 220 mm and mean annual temperature is around 18.1 °C, with a minimum of 14.6 °C and a maximum of 21.7 °C. Temperature and precipitation distribution is characterised by strong interannual variation and random precipitation patterns.

At both sites, four treatments were applied in replicated plots (1x1 m, n=4) (Figure 1) that included: a control (C) without application of a soil amendment; organic amendment covering the soil surface (AS); organic amendment incorporated into the soil (AI); and a 50:50 combination of both, with the amendment covering the soil surface and incorporated into the soil (AS-AI). Different organic materials (native-plant mulch versus compost) and soil substrates (sandy versus calcareous) were used at each site (NW Australia and SE Spain, respectively). To monitor UV radiation and soil conditions for the duration of the trial, a radiometer or UV sensor (UVM, 250–400 nm, Apogee Instruments Inc, Logan, Utah, USA) (Figure 2) and a HOBO micro station data logger with three connected soil temperature and moisture probes, were installed at both locations.

Soil microbial activity, soil CO_2 efflux, and nitrogen and organic matter fractions were measured repeatedly during the experiment. To determine soil microbial activity, we used the 1-day CO_2 test (Muñoz-Rojas et al. 2016a). Soil CO_2 efflux was measured with a soil CO_2 flux chamber attached to a LI-COR 6400 (LI-COR Inc. Lincoln, NEB, USA) using PVC soil collars; and nitrogen and carbon fractions were measured following the methods established by Oyonarte et al. (2007). Differences in soil characteristics across treatments were tested using analysis of variance (ANOVA), and comparisons between means were analysed with the Tukey’s HSD (honest significant difference) test (P <0.05). Statistical analyses were performed with R statistical software version 3.3.3 (R Core Team 2013).

RESULTS AND DISCUSSION

After 12 months, levels of microbial activity were significantly (P <0.05) higher in those plots amended in the surface in both study sites (Figure 3). Largest soluble fractions of C and N and soil CO_2 efflux were also found on the surface-amended plots (results not shown). These results reflect a fast C decomposing process directly related to UV radiation (light exposure), evidencing the facilitation of microbial litter decomposition by photodegradation (photopriming). These UV-related processes can be highly relevant at global scales as they can contribute to forcing biogechemical cycles; however, responses may vary depending on the type of soil and substrate. Overall, the results of this study are expected to shed light on the critical role of photodegradation on decomposition of organic amendments used in dryland restoration.

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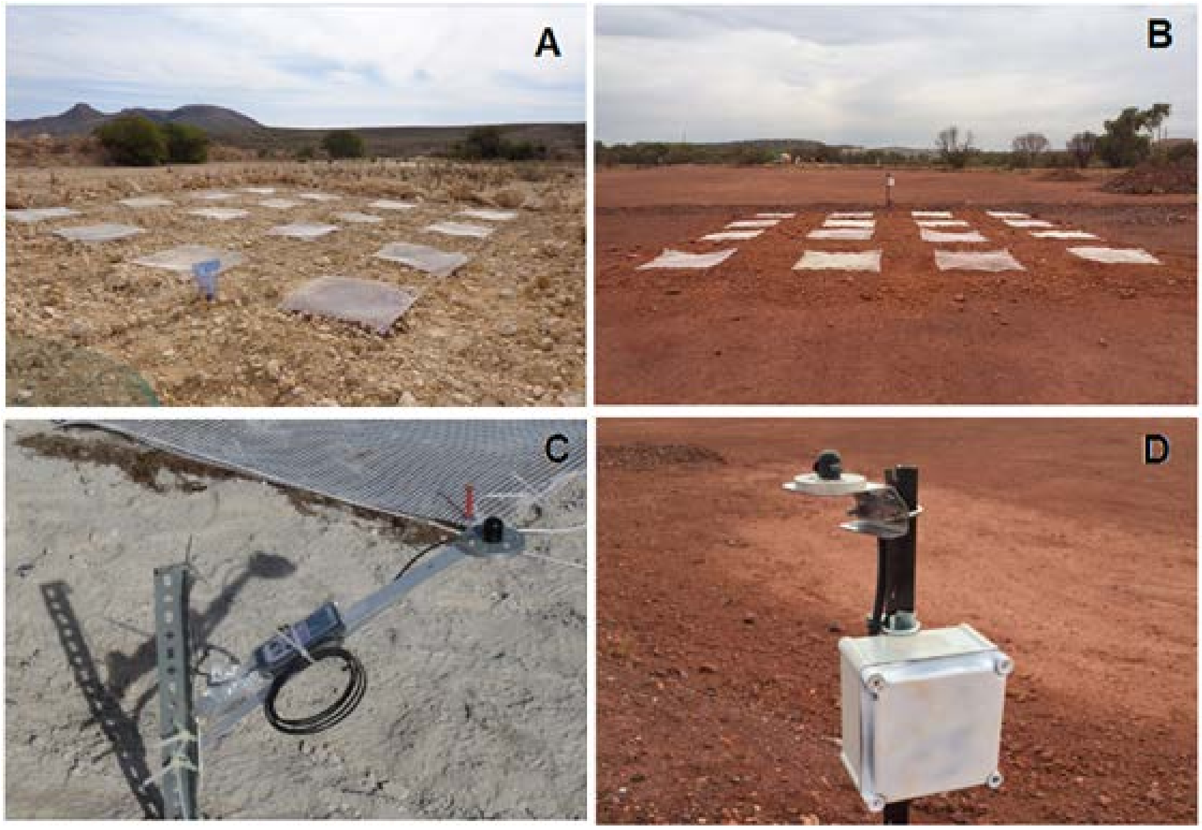

Figure 1. Covered plots (A and B) and UV sensors (C and D) installed in both study sites, Cabo de Gata-Nijar (SE Spain) (left) and the Pilbara (NW Australia) (right).

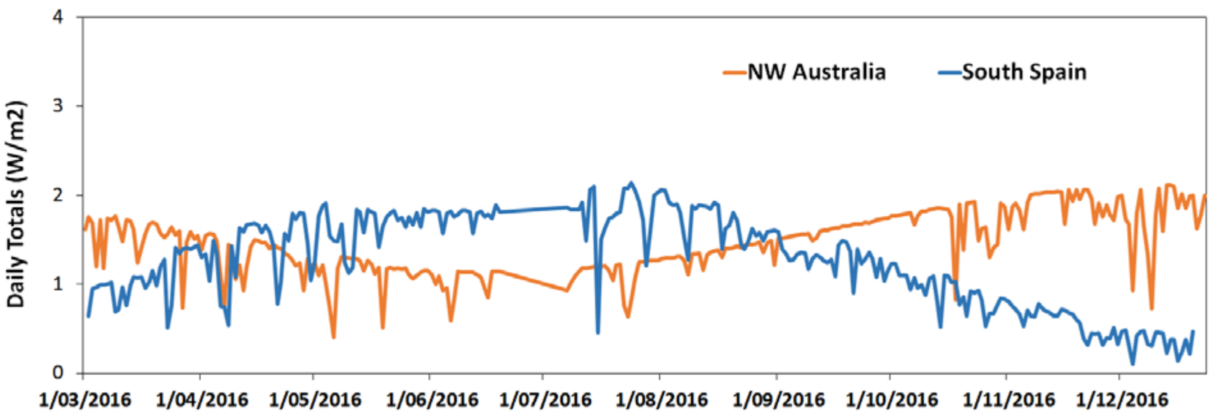

Figure 2. Variation of UV radiation during the field experiment in Cabo de Gata (SE Spain) and Pilbara (NW Australia).

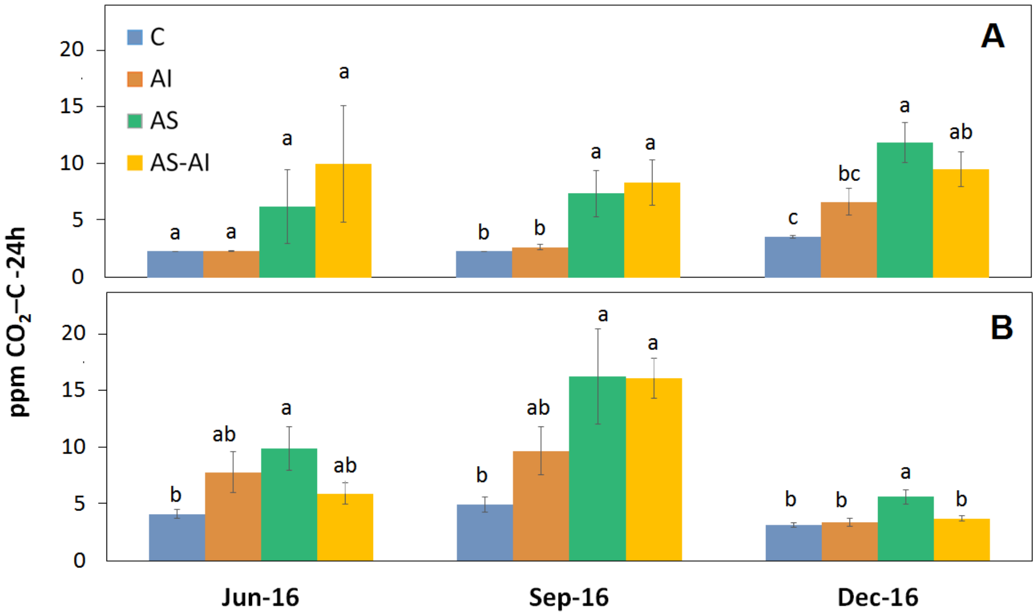

Figure 3. Soil microbial activity in experimental plots in Cabo de Gata (SE Spain) (A) and Pilbara (NW Australia) (A). C: control, AI: organic amendment incorporated into the soil; AS: organic amendment covering the soil surface and (AS-AI): combination of both techniques, both covering the surface and incorporated into the soil.

6.19.P

EFFECT OF PIG SLURRY AND TILLAGE ON SOIL QUALITY PARAMETERS

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INTRODUCTION

Mediterranean agricultural systems are characterized by low soil organic carbon (SOC) content and, as a consequence, they are susceptible to erosion and disaggregation. These degradation processes are particularly important in dryland systems. Furthermore, low soil organic matter (SOM) constrains water-holding capacity (Rasool et al., 2008) and water infiltration rate (Grande et al., 2005).

Gobin et al. (2011) underlined the importance of the continuous C supply as a food source for microorganisms and also in the building process for stable C in soil that contributes to soil aggregate formation. Fertilization and tillage practices affect SOM turnover.

Actually, Spain is the leading European pig producer. Slurries from fattening and sow pigs have different composition, mainly in organic matter content. However, in both slurries, ammoniacal N (easily available for crops) is the predominant form, which accounts for 65-70% of the total N (Yagüe et al., 2012a). In Spain, fertilization based on pig slurry (*Sus scrofa domesticus*) is a feasible option for farmers, as it is easily available, minimizes cost inputs and allows nutrient recycling.

The addition of pig slurry also induces a reactivation of soil microbial growth and activity (Hernández et al., 2007) associated with the decomposition of the labile fraction of its OM content (Bernal et al., 1991; Rochette et al., 2000).

Pig slurries have mainly been studied as N sources and to a lesser extent on their impacts on total SOC and soil quality parameters, probably because their low OC content. Continuous applications can affect SOC and aggregate stability (Yagüe et al., 2012b; Domingo-Olivé et al., 2016). Furthermore, they are relevant to soil carbon dynamics and also to enzymatic activities.

In semiarid areas, conservation tillage has also been recognized as a suitable system to increase SOC (Harvorson, 2002; Alvaro-Fuentes et al., 2008).

The experiment was established in a dryland winter cereal system. The aim of this research was to evaluate the effect of slurries of different origins (fattening pigs or sows) applied during 9 cropping seasons and combined with minimum tillage and no tillage in the last 4 seasons, on soil quality. The studied parameters were: total SOC, soluble C in water (Cw), C microbial biomass (CMB) and enzymatic activities: urease and acid phosphatase.

MATERIALS AND METHODS

The study was conducted in an experimental field located in Oliola, Catalonia, Spain. The region has a semiarid Mediterranean climate with high summer temperatures and low annual precipitation.

The experiment was established in 2002. The soil is deep (>1m) and calcareous, with carbonate content forming 30% in the surface layer (0-30 cm). It is non saline, electrical conductivity (1:5; soil:distilled water) is lower than 0.2 dS m^{-1}. The superficial layer has silty loam texture (131 g kg^{-1} sand, 609 g kg^{-1} silt; and 260 g kg^{-1} clay), pH (1:2.5; soil:distilled water) is 8.2, and organic matter content low (< 2%). The soil is classified as Typic Xerofluvent (Soil Survey Staff, 2014).

Soil was sampled (0-0.10 m depth), on the 18th May 2012, after 2.5 months from the last fertilization using slurries (at cereal tillering stage). Overall, this sampling was done after 9 yrs of continuous fertilization with slurries (historical fertilization application is detailed in Table 1) and after 4 yrs of different tillage implementation. Fertilization treatments included applications at cereal tillering of pig slurry of two origins: fattening (FS) and sow (SS) slurry at similar rates of 54 and 56 Mg ha^{-1}, the OC applied equalled 2.23 and 0.83 Mg OC ha^{-1} respectively

Each fertilization treatment was combined (for the last 4 years before sampling was done) with two tillage systems: minimum tillage (MT) and no-tillage (NT).

In this system, straw was collected and packed after harvest (July) and removed. Stubble was buried at the end of summer (September) through tillage based on disc harrowing (15 cm) in minimum tillage plots, and was left in no-tillage plots.

The evaluated soil quality parameters were: soil organic carbon (SOC) by AFNOR (2007), soil C microbial biomass determination (CMB) by fumigation (Vance et al., 1987), Soluble carbon in water (Cw; 1:2.5; soil:water; Yakovchenco et al., 1998, urease (Kandelen and Gerber, 1988 modified by Kandelen et al., 1999) and acid phosphatase (Tabatabai and Bremner, 1969).

Table 1. Averages of the total nitrogen (organic plus mineral) applied (TN)†, ammonium N (NH_4^+-N) and organic carbon (OC) applied annually to the experimental plots and also in the sampling year. Fertilizers from different slurry sources were applied at cereal tillering

Fertilization treatment	Annual slurry fertilizer application		Averages from October 2002 (sowing) to June 2011 (harvest)			Fertilization on 6th March 2012 (tillering)		
	Sowing	Tillering	TN†	NH_4^+-N	OC	TN	NH_4^+-N	OC
			--------------- kg ha^{-1} yr^{-1} ---------------			----------- kg ha^{-1}------------		
FS¶	0	54FS	389	259	1648	371	248	2229
SS††	0	56SS	92	59	385	135	93	828

†The difference between the total nitrogen applied and the nitrogen in NH_4^+-N form equals to the organic N applied.

¶FS: slurry from fattening pigs. Average rate (3 blocks): 54 ± 2 Mg ha^{-1} yr^{-1} (± standard deviation, SD).

††SS: slurry from sows. Average rate (3 blocks): 56 ± 4 Mg ha^{-1} yr^{-1}(± SD).

The experimental design was a split-plot with three blocks (replications). The statistical analyses were performed using SAS V8 (SAS Institute, 1999-2001) statistical software. When the analysis of variance (ANOVA), was considered significant ($p<0.05$), Duncan's Multiple Range Test (DMRT) was performed to compare the means at the 0.05 probability level.

RESULTS

The origin of the slurry, although applied at the similar rate of 55 Mg ha^{-1} yr^{-1} had influence on the CMB which was higher following FS than SS. No significant differences were found in SOC and Cw parameters, although FS achieved higher values than SS (Table 2).

The CMB was significantly different associated to the origins of the slurry. Nevertheless, enzymatic activities as urease and acid phosphatase were not influenced by these changes (Table 3).

Tillage had no effect on the evaluated soil quality parameters, probably because tillage was done before sowing.

Table 2. Average values of total soil carbon (SOC) and fractions: soluble carbon in water (Cw) and soil carbon microbial biomass (CMB)

	SOC	Cw	CMB
	----- mg C g^{-1} soil -----		----- mg C kg^{-1} soil -----
Fertilization†			
FS	19.1	18.2	254 A
SS	16.1	12.8	228 B
Significance¶	NS	NS	**(0.0097)
Tillage ††			
MT	18.3	17.2	248
NT	17.5	12.8	237
Significance	NS	NS	NS

†FS: slurry from fattening pigs; SS: slurry from sows.
††MT: minimum tillage; NT: no tillage.
¶Significance: NS not significant p>0.05; ** 0.01<p<0.001.

Table 3. Average values of urease and acid phosphate activities

	Urease	Acid phosphatase
	µmol NH_4-N g^{-1} h^{-1}	µmol p-nitrophenol g^{-1} h^{-1}
Fertilization†		
FS	3.39	1.01
SS	3.36	1.05
Significance	NS	NS
Tillage ††		
MT	3.48	1.07
NT	3.27	1.00
Significance¶	NS	NS

†FS: slurry from fattening pigs; SS: slurry from sow.
††MT: minimum tillage; NT: no tillage.
¶Significance: NS not significant p>0.05.

CONCLUSIONS

Tillage practices did not affect the evaluated soil parameters. Differences in the amount of OC applied were not reflected in SOC or Cw increments, but resulted in increased CMB in the FS treatment (254 mg C kg^{-1}) *vs.* the SS one (228 mg C kg^{-1} soil).

The urease activity (3.39 and 3.36 µmol NH_4^+-N g^{-1} h^{-1} for FS and SS respectively) and acid phosphatase activity (1.01 and 1.05 µmol p-nitrophenol g^{-1} h^{-1} for FS and SS respectively) were not affected by the established treatments. The absence of negative effects on soil quality, even at the highest FS rate, partially explains why farmers are not worried about the high amount of N applied (371 kg N ha^{-1} in FS *vs.* 137 kg N ha^{-1} in SS).

ACKNOWLEDGEMENTS

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6.20.P

RECOVERY OF EUROPEAN MARGINAL SOILS: THE INTENSE PROJECT

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INTRODUCTION

Soil resources in many parts of Europe are being lost due to inappropriate land management practices, industrial activities or/and land use change which is of great concern. Global food security, use of renewable raw materials and production of energy from biomass are some of the "Great Challenges" for the 21st century. Within the framework of the project "Intensify production, transform biomass to energy and novel goods and protect soils in Europe (INTENSE)", a team of eight research institutes from seven European countries are contributing to reconvert marginal lands into sustainable agriculture production soils. INTENSE will explore ways applying different tools at field scale. The Spanish partner is represented by the Centro de Investigaciones Energéticas, Medioambientales y Tecnológicas (CIEMAT) research centre and its objective is to improve the soil production of two marginal lands by using biochar of local vegetable wastes and pellets from compost of spent mushroom substrate during three years in order to assist in solving two problems that rural areas are facing at present: low agricultural productivity and converting an abundant residue into a useful resource.

METHODS

Study area

The experimental work is being carrying out in two different test sites 35 km away from each other. One of the sites is located in *Buendía (Cuenca, Castilla-La Mancha, Spain), which* has a stony soil with a nearly level topography and contains marle and dolomite. The second site is located in *Casasana (Guadalajara, Castilla-La Mancha, Spain), which* has a gently sloping slope and contains marl and gypsum. Both of them are subjected to dry summer conditions, and the soils have a basic pH and high calcium carbonate content. These sites differ from the rest of the scenarios of the European partners with acid soils and in wet climate.

Soil amendment and cultivation

Barley and sunflower are cultivated in both sites. The amendments are pellets from compost of spent mushroom substrate, bio-rest from biogas and straw; biochar from woody remains obtained from the surroundings; and ammonium sulphate nitrate. The experimental design for both sites consists of

39 plots of 4 m^2. Furthermore, there are 6 plots of 1 m^2 as control of the amendment effect in the soil without cultivation. Each crop and amendment combination is triplicated and has been randomly located (Figure 1 and 2).

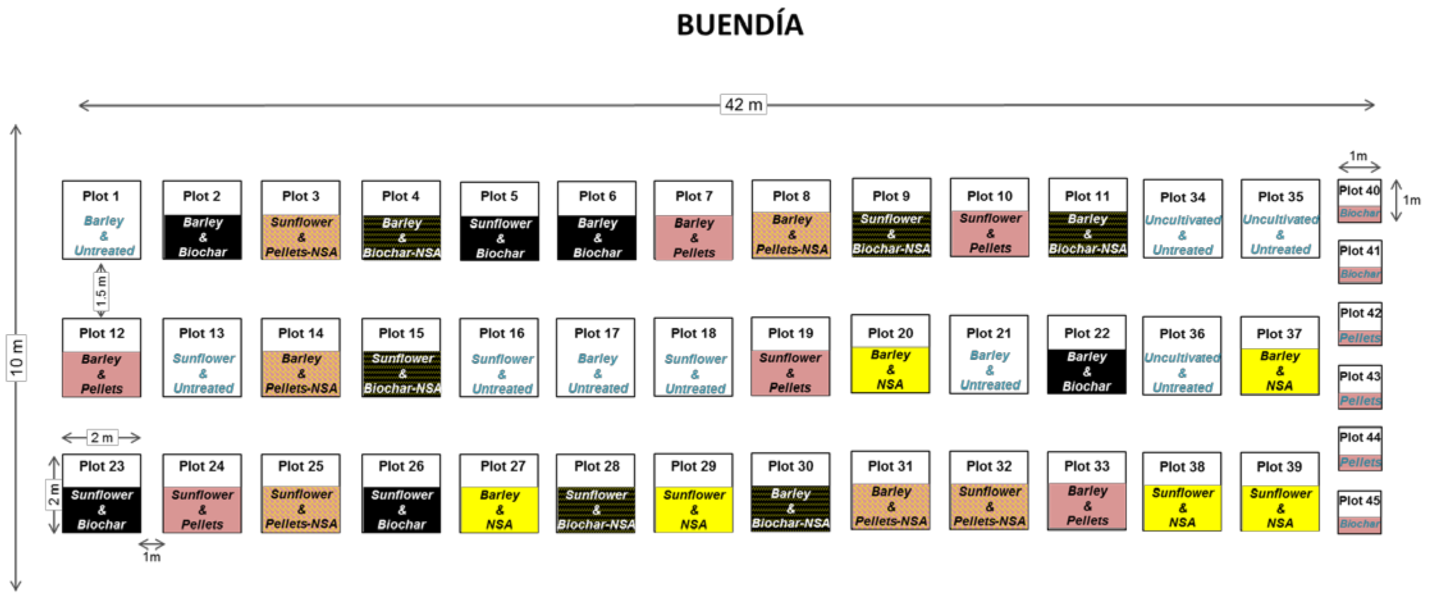

Figure 1. Experimental design of test site from Buendía

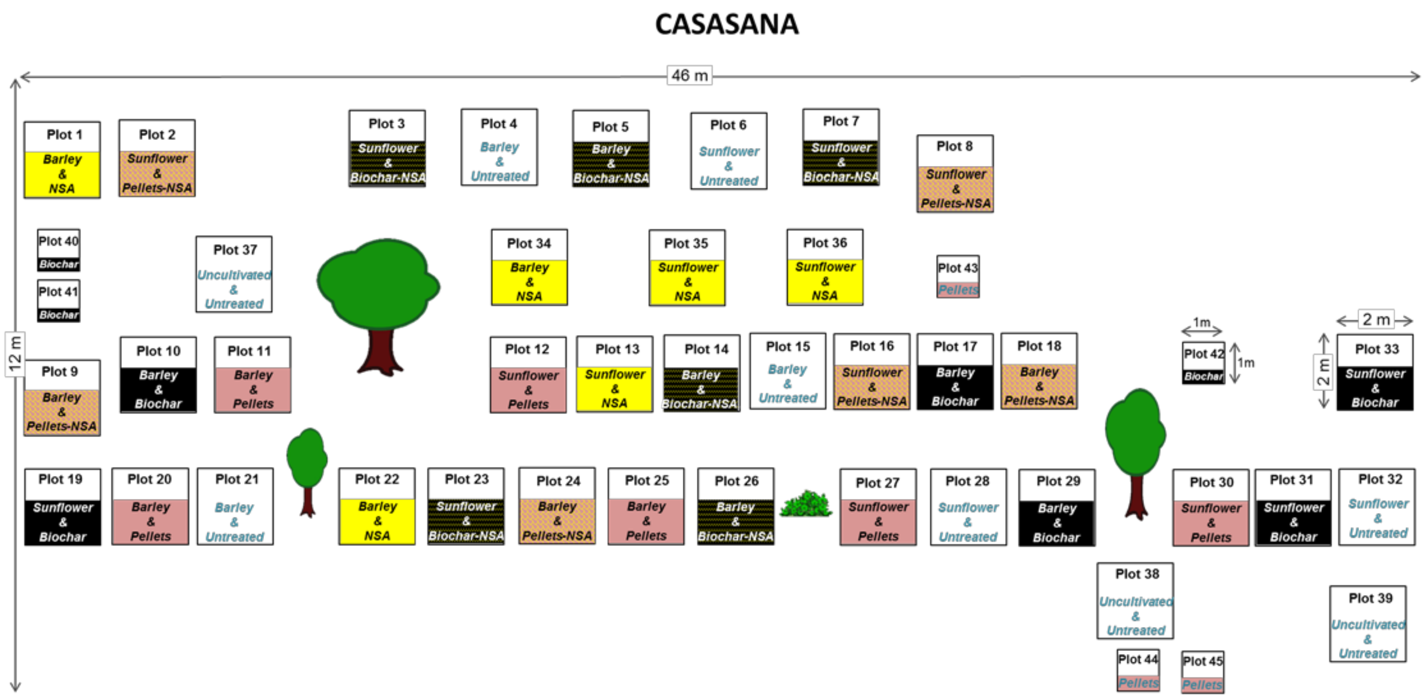

Figure 2. Experimental design of test site from Casasana

Soil and crop characterization

In the case of soils, two samplings are to be carried out every year: before sowing and after harvesting. Analyse include bulk density, pH (H_2O, 1:2.5), electrical conductivity (EC) (H_2O, 1:5), organic matter (OM) (Walkley-Black method), exchangeable cations (Ca, Mg, K, Na) (after extraction 1 mol CH_3COONH_4), color determination according to Munsell Soil Charts (Munsell Color 2009) and, water holding capacity and field capacity (Richards and Weaver, 1944).
In the case of the plant samples, samples are to be collected at the maturity, and include measuring the length of the aerial parts and the roots, the fresh and dry weight of the aerial parts and roots, the content of N, the yield, the number of grains/seeds per ear/flower and the yield index.

RESULTS

The interpretation of physico-chemical characterization (Andrades and Martínez, 2014; Garrido Velero, 1994) of the soils from the test sites (Table 1) show that both soils are basic and these pH values are adequate to support barley and sunflower life. Regarding the OM, the Buendía value is more than twice as high as compared to the Casasana value. However, both test sites have a low organic matter content. According to the EC value, Casasana soil is stronglysaline while Buendía soil is nonsaline. Regarding the exchangeable cations, both soils have elevated contents of calcium which agree with the lithology of the study area. Magnesium content in Buendía soil is medium, and is related with the presence of dolomite, whereas in Casasana the content is low; the potassium content in both soils is medium; and sodium content in Buendía and Casasana soils is very low and low, respectively. Finally, regarding field capacity, it is medium in the case of Buendía soil and high in the case of Casasana soil.

Table 1. Physico-chemical characterization of initial soils from Buendía and Casasana (n = 33, mean ± SD)

	BUENDÍA SOIL	**CASASANA SOIL**
pH(H_2O)	8.4 ± 0.1	7.8 ± 0.1
$EC_{1:5}$ (mS cm^{-1})	0.130 ± 0.014	2.135 ± 0.050
OM (%)	1.19 ± 0.14	0.47 ± 0.14
Ca (cmol kg^{-1})	Saturated	Saturated
Mg (cmol kg^{-1})	1.36 ± 0.15	0.41 ± 0.07
K (cmol kg^{-1})	0.52 ± 0.05	0.34 ± 0.07
Na (cmol kg^{-1})	0.04 ± 0.01	0.13 ± 0.02
Bulk density	1.20 ± 0.09	1.01 ± 0.07
Field capacity (%)	23.7 ± 1.6	38.0 ± 3.7
Water holding capacity (%)	13.3 ± 1.0	15.8 ± 2.3
Color	10YR 5/4	2.5Y 7/2

The results of Table 1 show that the plots within each test site are homogeneous. However both test sites show certain differences between them. The Buendía soil has better conditions for cultivation than the Casasana soil, although both of them have problems related to nutrient immobilization.

CONCLUSIONS

In summary, the Buendía soil is more fertile than the Casasana soil but according to the lithology of the study area, both test sites have high amounts of Ca, a high percentage of calcium carbonate ($CaCO_3$) and alkaline pH and these features immobilize certain nutrients such as P or iron. The application of the different amendments aims to improve the fertility and help to release these nutrients. At the end of the three years of the INTENSE project, data will be obtained data to evaluate which are the best combinations of crop and amendment.

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6.21.P

THE ROLE OF GYPSUM SPOIL IN THE REMEDIATION OF SOILS CONTAMINATED BY HEAVY METALS.

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INTRODUCTION

Nineteen years after one of the worst European ecological disasters occurred in Aznalcóllar (Seville, Southern Spain), when a pyrite mine pond breached and released highly polluted tailings and acidic water, there are still numerous remnant spots of polluted and barren soil (Martín et al., 2015) where vegetation cannot even germinate (García-Carmona et al., 2017). Another mining issue comes when, during the extraction of raw gypsum, large quantities of mining waste material, called gypsum spoil, are produced and accumulated in the exploited area. Being the Iberian Peninsula the largest area of gypsum outcrops in Europe, and Spain one of the world's largest producers, big quantities of this gypsum spoil urge to be sustainably managed in the country. Due to the physical and chemical properties of gypsum, this mineral has an important use as a fertilizer (Chen et al. 2011) and as amendment of certain degraded soils (Frazen et al., 2006; Zdruli et al., 2010; Chen et al., 2011). As gypsum spoil is composed of a 50-60% of gypsum mixed with fine material such as silt and clay, this waste material could also meet the requirements to improve soil properties and plant growth, as well as to immobilize heavy metals in soils, which would therefore prevent them from accessing the food chain or reaching aquifers, and so to promote soil and vegetation recovery. Consequently, an ex situ remediation experiment has been performed in order to test if gypsum spoil can help to restore soils highly polluted with heavy metals along with their associated vegetation.

MATERIALS AND METHODS

First, we collected soil samples containing high concentrations of heavy metals such as Pb, As, Cu and Zn, from contaminated spots in the Guadiamar Green Corridor in Aznalcóllar (Seville, Southern Spain). Then we mixed them with gypsum spoil in four different proportions: 0% (T1), 10% (T2), 20% (T3) and 50% (T4). Afterwards, we sowed seeds of two unrelated species (*Medicago sativa* L. and *Cynodon dactylon* (L.) Pers.) separately in pots (6 cm x 5.6 cm x 8 cm) filled with the four different substrate mixtures. Thirty-two replicates per treatment and per species were prepared, which makes a total of 256 pots randomly placed within a greenhouse (Figure 1). For 12 weeks, we watered pots daily and monitored them three times per week, recording plant emergence and survival. After 12 weeks, we collected seedlings and washed them with distilled water. Subsequently, we separated shoots from roots and dried the samples in an oven (70º C for 48 h). We weighed the samples in a precision scale (0.0001 g), after stabilization at room temperature. We also collected, dried and sieved five soil samples per treatment and species. Finally, we analyzed the content of heavy metals in plant and soil samples by means of ICP mass spectrometry (PE SCIEX ELAN-5000 spectrometer).

Statistical analyses of the collected data were performed applying generalized linear models (GLMs) using the R "stats" package with R version 3.3.2 (R Core Team, 2016).

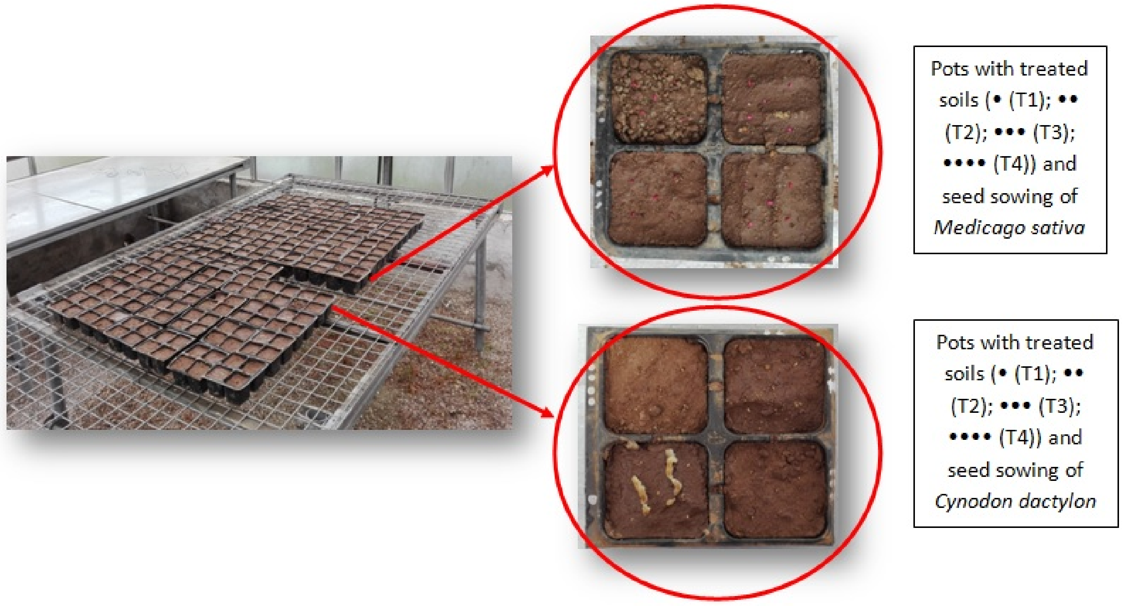

Figure 1. Experimental design and seed sowing in treated soils (*Medicago sativa* –top-right-; *Cynodon dactylon* –bottom-right-)

RESULTS

Our results show that the no addition of gypsum spoil (T1) totally inhibited the emergence of the seeds sowed, whereas the addition of the amendment in any proportion (T2, T3, T4) promoted, not only seed emergence (81% ±4.24 for T2, 88% ±0 for T3, and 99% ± 2.12 for T4), but also growth for both species. On one hand, *Medicago sativa* showed the highest survival rate for T4 treatment (97% versus 30% and 64% for T2 and T3 treatments, respectively) and high mortality rates for T2 and T3 treatments (70% and 36%, respectively). On the other hand, *Cynodon dactylon* registered high survival rates for all gypsum treatments, being the highest the one on T4: 72% (T2), 84% (T3) and 94% (T4). In terms of biomass, the most effective treatment was T4 for both species, as showed in Figure 2. However, comparing the weight of the seedlings grown per treatment for each species, *Medicago sativa* seedlings weighed on average 91% (T1), 93% (T2) and 88% (T3) less than these of *Cynodon dactylon*. With regards to the effects of gypsum spoil on soil properties, the application of the amendment may have enhanced water infiltration and aeration. As well, the acidic pH (3.8 ± 0.01) of this polluted soil was partially corrected by T2 and T3 treatments (5.6 ± 0.24 and 6.4 ± 0.01, respectively) and completely neutralized by T4 treatment (6.8 ± 0.14). Moreover, the relation between gypsum spoil addition and heavy metal immobilization is direct, and therefore, after watering daily for 12 weeks, most heavy metals leached out in control pots (where no gypsum spoil had been applied), whereas the addition of the amendment partially immobilized heavy metals (Table 1); consequently, heavy metals bioavailability was reduced, especially in the pots with T4 treatment, where the most vigorous and least polluted seedlings grew.

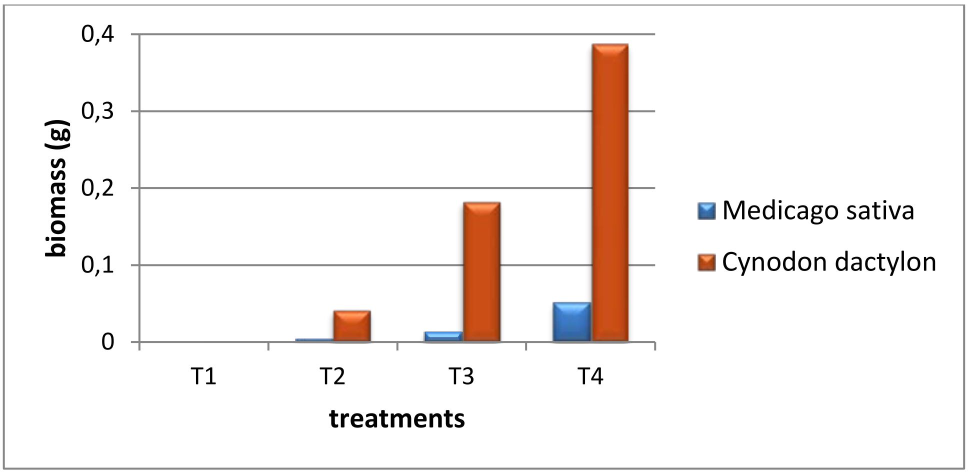

Figure 2. *Medicago sativa* and *Cynodon dactylon* seedling weight per treatments.

Cynodon dactylon				
	Cu	Zn	As	Pb
T1	71 ±1,14 d	86 ±1,90 c	69 ±2,30 d	78 ±1,21 d
T2	67 ±1,70 c	66 ±2,68 b	65 ±2,37 c	76 ±1,43 c
T3	64 ±1,78 b	61 ±0,79 b	63 ±1,74 b	74 ±0,90 b
T4	50 ±4,55 a	46 ±5,15 a	50 ±5,71 a	62 ±4,69 a
Medicago sativa				
	Cu	Zn	As	Pb
T1	54 ±6.51 c	74 ±6,56 c	52 ±6,99 c	73 ±3,11 d
T2	49 ±11,67 bc	48 ±9,29 b	52 ±8,82 c	74 ±4,17 c
T3	46 ±10,25 b	42 ±5,25 b	49 ±10,50 b	72 ±3,74 b
T4	40 ±24,56 a	32 ±14,29 a	40 ±19,46 a	66 ±5,47 a

Table 1. Percentage (mean ±SE) of heavy metals per treatments leached out from soil after 12 weeks of daily watering.

CONCLUSIONS

Our findings suggest that gypsum spoil could be used as an effective amendment to recover soils polluted with heavy metals. The main evidences supporting it are:

- It has the capacity to take acidic pHs towards neutrality, and to react with heavy metals reducing its mobility, and so their toxicity.
- It can also improve the physical properties of soils, promoting aggregation of soil particles, which facilitates water infiltration and aeration.
- All these chemical and physical improvements, promoted by the addition of gypsum spoil, allow that seeds can once again emerge and grow in these soils, since heavy metals will be less available for them, as opposed to essential nutrients, oxygen and water.

Thus, if gypsum spoil can be used to amend these kind of polluted soils, two urgent environmental issues would be simultaneously addressed: (i) the sustainable management of mining waste material; and (ii) the effective remediation of degraded soils polluted with heavy metals.

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KEYWORDS

Guadiamar Green Corridor; Polluted tailings; Polluted soils; Gypsum spoil; Amendment; Remediation; *Medicago sativa*; *Cynodon dactylon*

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Session VII: Sedimentation processes. Causes and effects

7.2.O

RELATIONSHIP BETWEEN HYDRO-PEDOLOGICAL AND SEDIMENTATION, FOLLOWING THE RE-VEGETATION OF THE BADLANDS OF THE 'TALAKHAYA' WATERSHED IN THE MICRONESIAN ISLAND OF ROTA".

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ABSTRACT

The Talakhaya watershed in Rota is identified as a Coral Reef Management Priority site for CNMI (Commonwealth of Northern Mariana Islands). Since 2006, CNMI resource management agencies collaborated with USDA Natural Resources Conservation Service (NRCS) on reclaiming the eroded areas (badlands of the Talakhaya watershed in Rota (CNMI)). In 2010, federal and jurisdictional partners including NOAA came together to develop a Conservation Action Plan (CAP) for the Talakhaya watershed. The CAP highlights the need for re-vegetation of eroding areas as well as addressing intentional burnings with education and fire prevention campaigns, among other objectives related to preservation and protection of badland in Talakhaya watershed in Rota.

The existing re-vegetation efforts occurring in the watershed however, did not have a component of quantifying the effects that their project made on reduced sedimentation. The Talakhaya Watershed Soil Loss Assessment project reported here aimed to quantify the reduction in sediment by determining the hydro-pedology of the watershed and by monitoring the level of sediment that is travelling through the streams and deposited in the nearby shorelines. These objectives are achieved by measuring the hydrological parameters following the installation and the use of the equipment such as; stream flow meters, barometric level loggers, turbidity meters, and rain gauges. The water flow as well as the turbidity level of streams leading to the ocean from the Talakhaya watershed was monitored and sedimentation level was assessed accordingly. The stream monitoring was compared in areas where no mitigation techniques were applied with the areas where the 'Vetiver' grass was planted as a major means of controlling erosion and reducing sedimentation load into the streams leading to the ocean shorelines. The comparison was also made with other areas where tree plantation was being practiced as means of badland preservation and for reducing sedimentation load into the ocean and protecting the coral reef near the shorelines.

The analysis of the soil and water sampling from the areas of the watershed planted with Vetiver grass was compared with areas of watershed without any Vetiver plantation in order to evaluate the effect of the Vetiver grass on sedimentation. The environmental impact of the re-vegetation on the coral reef areas affected by the stream water originating from the Talakhaya watershed was also evaluated.

The results of hydro-pedological monitoring for evaluating the effectiveness of the aforementioned mitigation techniques on sedimentation from the Talakhaya watershed are being reported in this presentation.

KEYWORDS:*Watershed Management, Hydropedology, Vetiver grass system, Sedimentation, Degraded lands, Commonwealth of Northern Mariana Islands.*

INTRODUCTION

The Talakhaya watershed in Rota (Fig. 1) is identified as a Coral Reef Management Priority site for the Commonwealth of the Northern Marianas Islands (CNMI). Since 2006, CNMI resource management agencies including Division of Forestry, Division of Environmental Quality, Department of Lands and Natural Resources, and the Luta (Rota) Soil and Water Conservation District have collaborated on restoring the Talakhaya watershed badlands (severely degraded lands) in Rota. These agencies have worked with USDA Natural Resources Conservation Service in identifying Best Management Practices (BMPs) and restoration projects. Beginning in 2007, NOAA Coral Reef Initiative (CRI) funds were awarded to CNMI to begin re-vegetation efforts of the badlands areas in Talakhaya in Rota. These efforts endured mixed success, with successful re-vegetation followed by human-induced as well as wild fire burning of the area. This project therefore aimed to quantify the reduction in sediment (if any) as a result of re-vegetation efforts currently occurring in the Talakhaya watershed. As it is reported here the results of the on-going re-vegetation efforts, especially by modifying the planting techniques for Vetiver grass (*Chrysopogon zizanioides*) which were being planted prior to this investigation, have shown although narrowly a positive impact on the sedimentation in the streams and rivers surrounding the Talakhaya Watershed.

The island in Rota is the southernmost island in the Commonwealth of the Northern Marianas Islands (CNMI) and the second southernmost island of the Marianas Island Archipelago. The island is located roughly 60 km north of the island of Guam

The island of Rota consists of well-developed limestone terraces with six different levels and Sabana being the uppermost terrace (Sugawara, 1934).

The Talakhaya Watershed has been described by Keel, Mylorie, and Jenson (2005) as “a large, relatively steep exposure of weathered volcaniclastic material” and contains the island’s only surface streams. The only areas on the island that contain exposed volcanic rock are on Sabana and the Talakhaya Watershed (USDA SCS, 1994). The streams of the Talakhaya Watershed are fed by the Water Caves in Sabana, several of which also act as the source of the island’s potable water. Studies of the Matan Hanom Spring discharge have shown large variation in the amount of discharge during wet seaon at “5.4 million gallons per day (mgd), dry season discharge at 0.5 mgd, and an average daily flow of 1.8 mgd (USDA SCS, 1994).”

Although there has been some limited unpublished data collected for rainfall on Rota and the Talakhaya Watershed area by USGS, attempts to obtain this data and its sources proved unfruitful as it is unavailable online or by both the Hawaii and Guam offices. Other rainfall information for the island has been collected by UOG-WERI (Water and Environmental Research Institute of the University of Guam) and by the National Weather Service from the Rota

International Airport, although both data sources are collected outside of the project area (USDA NRCS, 2008).

GIS data is largely limited as well in the area. Rainfall, soils, land cover, and elevation data is also constrained by limited information. Furthermore, access to majority of the watershed is limited by lack of proper road infrastructure and land accessibility as several areas are privately owned. On the other hand, several studies in the Talakhaya Watershed area have focussed on the water caves, which supply the island's drinking water (Matan Hanom and As Onan Caves) and also serve as the source for the area's streams (Keel, Mylorie, & Jenson, 2007 and USDA SCS, 1994). The baseline soil descriptions for the study site were obtained from the soils of Rota and the Northern Marianas Islands by Young (1986) and were utilized in this study as a reference for the expected soil types in the Talakhaya Watershed.

METHODOLOGY

Site visits to the Talakhaya Watershed area along with representatives from Rota Department of Lands and Natural Resources (DLNR) Forestry section representatives provided insight into the planning of the approach and equipment to determine soil loss from the watershed that would be most feasible in terms of cost, safety, manpower, environmental impact, and ensuring accuracy of data to meet project goals.

Upon determining the measurement parameters the types of devices that were used to best collect data included: 16 OnSet Hoboware level loggers, 4 Hoboware rain gauges, 1 Hach portable flow meter, and 1 Horiba multi-parameter meter (temperature, pH, dissolved oxygen, turbidity, electrical conductivity, and total dissolved solids measurements). Data collected from such instruments were used to determine the hydrology of the watershed and the level of sediment travelling through the streams and deposited in the nearby Bays (Fig. 1).

Following the installation and the use of the equipment including a water quality meter, flow meter, barometric level loggers, and rain gauges, the stream flow and water quality measurement as well as collection of data from the fixed level loggers of each stream leading to the ocean from the Talakhaya watershed were measured on monthly basis. In addition to stream monitoring, the rain gauge data was also collected from the fixed loggers during these monthly site visits. This monitoring was compared with the areas with the Vetiver grass plantation in order to assess the effect of vegetation type on reducing the sedimentation load into the ocean and the coral reef near the shorelines. It should be pointed out that prior to the aforementioned investigation the Vetiver grass had been planted rather in sporadic bunches which was not effective in reducing sedimentation. A modification strategy for planting the Vetiver grass as a technology and in the form of hedges was introduced and preformed thereafter.

The analysis of the soil and water sampling from the areas of watershed re-vegetated areas with new planting techniques using Vetiver grass (Fig. 2, 3, and 4) is also being compared with areas of watershed without any Vetiver plantation in order to evaluate the effectiveness of the environmental impact of the Vetiver plantation on the watershed as well as the coral reef area fed by the stream water from the Talakhaya watershed area.

Hydrologic data was gathered in four of the major stream outlets within the Talakhaya Watershed area (Fig. 1 and 5). This included two re-vegetated sites (including the Vetiver grass technology), one un-vegetated site, and a naturally vegetated site, which was used as a control location. The hydrologic data was collected to develop a correlation with the amount of rainfall, stream water level, stream flow, and turbidity which was used to represent in-stream turbidity levels. This correlation was used to assist in improving the understanding of the watershed's dynamic behaviour through understanding the interaction between rainfall rates with stream output and sedimentation. An understanding of this correlation can aid in predicting future watershed behaviour based upon projected future activities within the watershed. The hydrologic data that was collected include rainfall level, stream level, discharge, and water quality data (i.e. turbidity, pH, total dissolved solids (TDS)).

Rainfall was measured using four tipping bucket HoboWare® data logging rain gauges. The rain gauges were placed at four sites (Fig. 5) around the Talakhaya watershed at different altitudes and proximities to the coastline.

Stream water quality was measured monthly using a multi-parameter water quality meter, which measures among others: acidity (pH), dissolved oxygen (DO), turbidity, total dissolved solids (TDS), and conductivity.

The primary water quality measure observed in this study was turbidity. Stream turbidity is a measure of the cloudiness of the water in terms of Nephelometric Turbidity Units (NTU), which indicates the amount of sediment carried in the stream. The turbidimeter is a device that measures the transmission of light reflected by particles through a solution. Turbidity measurements are useful in this study as an indicator of sediment loading in the affected streams.

Stream level was measured using fourteen level loggers with up to three loggers placed in each river to measure water pressure and one logger placed above the water surface and in the proximity of each river logger to measure the atmospheric pressure (Figure 1, 5). The water level was measured regularly at 1-hour intervals using the level loggers in the streams within the watershed. The recorded pressure of the in-stream loggers was compared against level loggers on land, which measured atmospheric pressure. The atmospheric pressure was subtracted from the in-stream pressure to accurately calculate the water level of the streams based on pressure and temperature of the water level on the logger using Pascal's principle [Eq.1]:

$$\Delta P = \rho g(\Delta h) \qquad [1]$$

Flow was measured bi-weekly to monthly in the four Talakhaya stream sites using an electronic flow meter. Flow measurements were taken along transects running perpendicular to the flow direction from edge to edge of each stream under the study.

Soil Sampling and Analysis:

Soil composite samples were taken and tested in the University of Guam Soil Labs in Guam in order to identify the various soil types represented in the Talakhaya Watershed. The samples were taken at areas within relatively close proximity to rain gauge sites and revegetation areas including both badland and vegetated areas within proximity of such sites given the representation of the various rain gauge locations within the watershed. Composite samples were ground and sifted through a two-millimetre standard sieve. An additional sample (one sampling event) was taken off the coastline to illustrate the sediment being deposited from the nearby streams.

A total of 7 samples were then obtained and analyzed for; pH, texture, nutrients analysis, and organic matter content determination. All soil testing methodology were derived from the Methods of Soil Analysis: 'Chemical and Microbiological Properties' monograph (Page, Miller, & Keeney, 1982). This methodology has been adapted for use on Guam soils by the University of Guam Soil Research and Testing Laboratories.

Soil pH Analysis

Soil acidity, or pH, was analyzed using an electronic pH meter which analyzed 10 gram sieved samples suspended in 10 mL of distilled water in order to achieve a 1:1 soil to water ratio (Page et al, 1982). The soil mixture was measured using a calibrated Oakton glass electrode – calomel electrode pH meter.

Soil Organic Matter Analysis

Soil organic matter is the organic, or carbon-based, fraction of the soil, which is inclusive of plant, animal, and microbial residue, which can be fresh, or at any other stage of decomposition as well as high-carbon compounds such as coal, charcoal, or graphite. The soil's organic matter can act as a reservoir for plant nutrients, increase the water-holding capacity and aggregation of the soil, lower the soil's bulk density, and increase the cation exchange capacity of the soil. Organic matter in the soil also supports microbial activity in the soil as a required energy source for microorganisms.

Analysis of the soil organic matter utilized the Walkley-Black Method, also known as the dichromate oxidation method, which reacts with readily available organic carbon through oxidization (Page et al, 1982). The reaction is incomplete in reacting with all organic carbon in solution and therefore a correction factor of 1.3 is introduced to the calculation of total organic carbon based on the titration reaction of the organic carbon in soil with ferrous ammonium sulphate.

Soil Texture Analysis

Several of the physical and chemical properties of soils are largely affected by the soil texture, which is the proportion of sand, silt, and clay particles in the soil body. The soil texture of the composite samples taken for the Talakhaya Watershed were determined based on the rate at which soil particles settle into solution is determined primarily by particle size. This is the most commonly used method of determining soil texture at the UOG soil labs in Guam. The method uses the Bouyoucos hydrometer method of mechanical analysis, which is founded upon Stoke's Law.

Soluble Phosphorous in Sodium Bicarbonate

The soluble phosphorous was extracted from the soil and measured using a Bausch and Lomb Spectronic 21 colorimeter was set to 888nm. The methodology was developed by Page et al (1982) and adopted for use on Guam soils by the University of Guam Cooperative Extension Service (UOG CES).

Nutrient Analysis of Potassium, Calcium and Magnesium

Two reagents were made for the nutrient analysis and analyzed using a Perkin-Elmer Atomic Absorption Spectrophotometer Model 305B using the standard conditions for the equipment used with the magnesium sample diluted with the Lanthanum oxide solution.

Results and Discussion:

Hydrology

Stream and Rainfall Data:

The four streams under monitoring have been categorized based on the upstream vegetation type to include; one un-vegetated stream site within the project area that requires re-vegetation yet (coded TK1), two already re-vegetated areas within the project (coded TK2 and TK3), and a naturally well-vegetated site just outside of the project scope (coded TK4) to serve as our ideal model area.

The project design in this manner attempts to compensate for the lack of background data prior to the start of the re-vegetation efforts being undertaken. Although not ideal, the comparison of streams allows for a better understanding of the areas of the watershed under study than simply monitoring after the re-vegetation efforts with no other means of comparison to form a known baseline. As such, however there are various differences within each stream aside from vegetation including but not limited to land use, watershed sub-basin size, stream length, and geology.

Our un-vegetated stream site (TK1) is perhaps the most hydrological dynamic of the sites (Fig. 6). Although oral history from local community members has indicated that this stream was normally perennial, the stream has become more intermittent. This has made data collection within the streams far more difficult especially during the dry season and through the early wet season. When reaching enough rainfall volume to fill the stream large quantities of sediment are also carried with the rainfall to be deposited within the stream and out into the ocean. Figure 6, 7, 8 and 9 illustrates time series graphs including rainfall, stream level, and turbidity measurements for streams TK1, TK2, TK3, and TK4 respectively. These figures illustrate the reactions of the various streams to rainfall events, especially with peak stream levels in response to large rainfall events.

Due to an error with the Hobo Data Shuttle, there was a data gap between September 13, 2014 and October 27, 2014 for most of the level logger data. Stream TK1 was also affected by Typhoon Vongfong on October 5, 2014 when a fallen tree blocked one of the in-stream level loggers and created a small dam, blocking additional debris, sediment, and water from flowing further downstream.

The re-vegetated stream sites (TK2 and TK3) act to illustrate the effects of the Talakhaya

Revegetation Project on the lower reaches of the watershed. Although shown to be overall less dynamic than TK1, these streams still display high turbidity levels during high rainfall events (Fig. 7 and 8). Given the need for the re-vegetated specially the Vetiver grass and other vegetation to become established before becoming fully able to protect sediment from reaching the streams and thereby the ocean, this lack of drastic difference in the unvegetated and revegetated areas is expected. Once given a few years for the plants to establish themselves and provide their extensive barrier capacities it is expected that the turbidity levels of the re-vegetated sites will then decline further.

Although the stream sites chosen were indicated by local residents as having been historically perennial streams the observations over the duration of the project have shown all streams (TK1, TK2, TK3, and TK4) to behave more intermittently. It may also be noted, “perennial streams are found only on the volcanic soils of Talakhaya,” but “not all of the streams run perennially” (USDA SCS, 1994).

The natural vegetation stream (TK4) represents a stream without large badlands issues requiring the re-vegetation efforts that are required of the other stream basins in the study. As such, this stream represents an ideal vegetative cover that the re-vegetation sites should reflect or surpass in sedimentation and stream turbidity (Fig. 9).

According to Title 65 of the CNMI Administrative Code, “Turbidity at any point, as measured by nephelometric turbidity units (NTU), shall not exceed 0.5 NTU over ambient conditions except when due to natural conditions” for Class 1 waters which include all fresh waters in Rota such as the Talakhaya streams.

A comparison of the daily stream turbidity measurement (Fig. 10) illustrates the high variability of stream turbidity levels. Figure 7 also illustrates the frequency for the peak turbidity measurement to be represented by the non-vegetated stream, TK1.

Rainfall vs. Turbidity:

The rainfall interval providing the most effect on stream turbidity appears at a 6 –hour interval in comparison against four selected intervals prior to turbidity measurement including 3, 6, 12, and 24- hours of rainfall data prior to stream turbidity readings (Fig 11, 12, 13, 14). This information however is limited by sampling frequency and the random determination of turbidity data collection relative to storm events.

Because of a lack of fixed loggers or regular intervals to measure in-stream turbidity, the information provided by the graphs (Fig 11, 12, 13, 14) includes data collected during site visits with a water quality meter to measure turbidity and other water quality data. As such, this comparison between turbidity and rainfall prior to such measurements is not representative of peak turbidity levels, which would be more indicative of the effect of rainfall on turbidity including reaction time and overall change in turbidity during storm events. Given the available information however, the peak reaction of turbidity is within 6 hours of rainfall events for TK1 (R^2=0.58584), TK2 (R^2=0.09473), and TK4 (R^2=0.16298) (Fig. 9). The peak reaction for TK3 (R^2=0.32768) was shown to have occurred within 12 hours of a rainfall event (Fig. 13).

Rainfall v Discharge:

Similarly, peak stream discharge and their rainfall reaction times cannot be fully determined because of lack of permanent in-stream loggers, sampling frequency, and also the safety of monitoring individuals (research assistants). Some of the streams, especially the non-vegetated TK1, are known to have dynamic reactions to changes in rainfall and therefore in-stream discharge measurements during such events would pose significant safety hazards. Based upon the same 3, 6, 12, and 24-hour interval comparisons between streams discharge measurement (Fig 15, 16, 17, 18) and prior rainfall intervals the 12-hour rainfall interval (Fig. 17) appeared to have the largest influence upon stream discharge with the effect peaking off at the 24-hour interval (Fig. 18) comparison. The stream discharge however was also seen represented in earlier intervals including both the 3-hour (Fig. 15) and 6-hour (Fig. 16) rainfall events indicating a varying influence of rainfall on stream discharge over the 24 hour period of rainfall measured. For TK1 (R^2=0.59635) and TK3 (R^2=0.21014), the peak reaction occurred within 3 hours of a rainfall event. For streams TK2 (R^2=0.55069) and TK4 (R^2=0.34024), the peak reaction correlation occurred within 24 hours of a rainfall event. These R^2 values might be improved with additional data collection.

The time-reaction for stream discharge to rainfall events further illustrates the difficulty in comparing the measured streams against one another given the large variability in stream behaviors. Of the streams measured, TK2 (one of the re-vegetated stream sites) was the most dynamic in reaction to rainfall events with much larger increases in stream flow in reaction to varying amounts of rainfall than any of the other streams (i.e. TK1, TK3, and TK4), which all had similar discharge reactions to varying rainfall events. This difference may be attributed to the varying sizes of the stream basins and such comparisons of size must thereby also be made.

Stage Discharge Curve:

A preliminary stage discharge curve (Fig. 19) was developed for the four streams measured in the study. The stage discharge curve was developed from the stream flow measurements conducted and the stream level measured by the installed level loggers.

An accurate stage discharge curve should utilize several years' worth (e.g. 25 or 50 years of stream level and discharge data) of water level and stream flow data. The development of an accurate stage discharge curve for the primary rivers of the watershed is essential to future management of the watershed because the stage discharge curve removes the need for the weekly flow measurements of the watershed by providing a measurement of flow level in the rivers.

The stage discharge curves developed for this study utilized only one-year of data collected (Fig. 19). Therefore, this does not provide a fully accurate estimate of the flow and water level relationship of the streams. However, the stage discharge curve developed can serve as the basis for future hydrologic studies within the Talakhaya Watershed. It is recommended that flow and level recordings of the streams continue to be measured in order to obtain a more accurate estimate of the watershed behavior for future studies.

Soils

A total of seven composite samples were collected (Table 2) for testing at various areas within the Talakhaya Watershed including badland (Samples 1 and 3), savanna (Samples 2, 4, 6, and 7), and shoreline (Beach) (Sample 5). The mapped sample areas include Akina Badland Complex, 30 to 60 percent slopes and Chinen Clay loam, 15 to 30 percent slopes, while the beach area is identified as Takpachao Variant-Shioya Complex, 1 to 10 percent slopes (Young, 1986). Akina soils are identified as "Loamy, mixed, isohyperthermic, shallow Udorthenthic Haplustolls", Chinen soils are identified as "Clayey, oxidic, isohyperthermic Lithic Arguistolls", and Takpachao variant soils are "Loamy-skeletal, carbonatic, isohyperthermic Lithic Haplustolls (Young, 1986)."

All samples tested (Table 2) with the exception of the sample 5 (the beach sample) contained low pH levels and were thereby more acidic. Organic matter was low in all areas except for sample 6 and 7, which were sites located near several *Acacia* trees which were planted as a part of the revegetation project. The leaf litter produced by these trees can explain the high organic content seen in these areas (soil samples).

Soil texture measurements (Table 2) showed high sand content in samples 1, 3, and 5, which are expected as samples 1 and 3 were taken from badland areas where most of the clay content would have already eroded away and sample 5 was located along the beach. In contrast, the clay content was high in samples 6 and 7, which were taken in similar savanna environments in close proximity to one another.

Nutrient content measured (Table 2) included calcium (Ca), Magnesium (Mg), and Phosphorus (P). Potassium (K) data was unavailable (and continues to be) due to machine calibration issues. However, these soils tend to be generally low in potassium content due to their inherent low fertility status (Young, 1986). Calcium and Magnesium levels in all samples taken were shown to be very high, where 2000 ppm Ca and 200 ppm Mg are at the higher limits of the spectrum. Especially, high Magnesium levels may therefore pose toxicity risks to plants and limit non-tolerant vegetation from growing in these areas. Phosphorus levels in contrast were very low in all areas measured, where 10 ppm P is the lower limit of the spectrum. These soils are therefore lacking (P, K) in available nutrients for large vegetation to become established.

Recommendations:

From the up-to-date data, it appears that re-vegetation could possibly have a positive impact on reducing sedimentation. However, new growth, especially the Vetiver grass technology, must have more time to establish itself especially following the modified planting techniques. In addition, more data is required as re-vegetation is still ongoing and becoming established and may have more distinct effect on reduced sedimentation. Therefore, continued hydrologic and soil and stream water monitoring of the area would be necessary to obtain more data for producing more defined lines in the graphs obtained from the data. This would establish a stronger understanding of the effects of the re-vegetation efforts with regards to sedimentation and stream hydrology. Other monitoring procedures such as stream water sampling from the upper sections of the rivers (where there is continues water flow) as well as frequent and established pattern of shoreline samplings are also needed to compliment the obtained data from data loggers and turbidity probe. Shoreline sediment sampling at varying distances from the mouths of the various streams in the Talakhaya watershed is strongly recommended as it would provide information certainly beneficial to the understanding the effect of the sedimentation as it reaches the ocean community.

Furthermore, there is a continued need for increased community awareness in order for them to appreciate the effects of conservation and the preservation of natural resources in this island. The efforts of this project should develop a sense of community stewardship for protecting the watershed from further degradation possibly caused by human induced burning as well as other degrading factors in order to protect the coral reef in the ocean surrounding the island.

ACKNOWLEDGEMENT

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APPENDIX I - Tables

Table 2: Soil Sample Analysis for Talakhaya Watershed

#	pH	% O.M.	E.C. µS/cm	% Sand	% Silt	% Clay	Soil Texture	Ca (ppm)	Mg (ppm)	P (ppm)
1	5.59	0.16	78.7	60.92	20.52	18.56	Sandy Loam	11949	941	1.62
2	4.98	0.16	134.8	34.92	29.80	35.28	Sandy Clay Loam to Clay Loam	12013	972	1.23
3	5.45	0.16	60.6	50.92	23.80	25.28	Sandy Clay Loam	17133	824	1.36
4	4.50	1.48	55.3	30.92	33.80	35.28	Clay to Clay Loam	12765	890	0.12
5	8.92	0.16	632.0	78.92	7.44	13.64	Sandy Loam to Loamy Sand	8153	963	0.09
6	4.47	3.45	101.8	10.92	23.80	65.28	Clay	5016	860	1.49
7	4.49	5.42	129.0	19.28	19.44	61.28	Clay	3725	736	1.89

APPENDIX II - Images

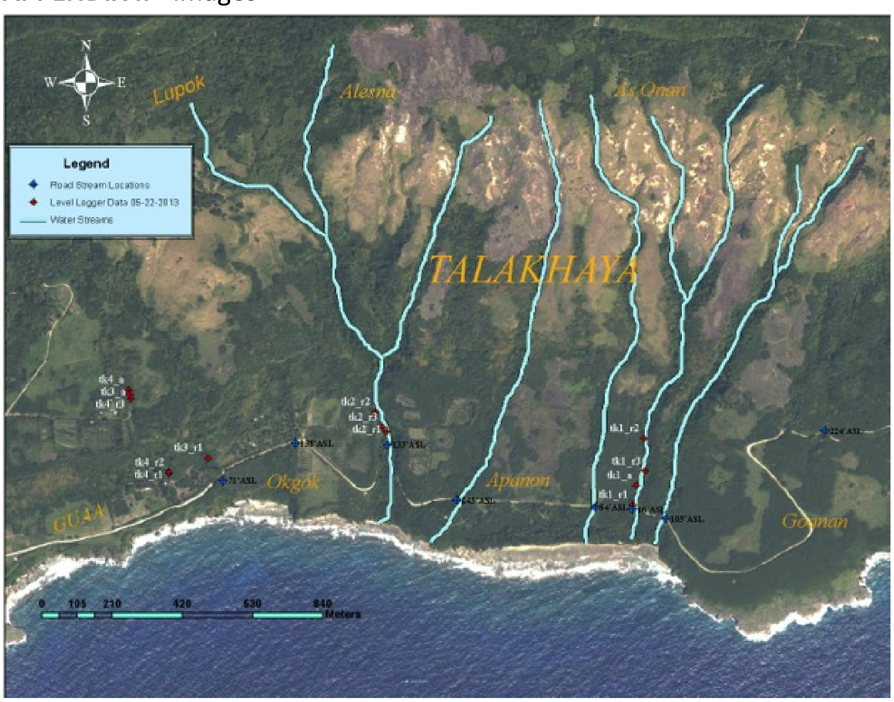


Figure 1: Talakhaya watershed working Map (courtesy of the Rota BECQ office) shows the 'major' streams and the location of the level loggers that are used for monitoring the effect of re-vegetation of the watershed.

Figure 2: Vetiver grass (Chrysopogon zizanoides) planted in hedgerows in the Talakhaya Watershed.

Figure 3: Vetiver grass hedgerows in Talakhaya Watershed planted about three months prior.

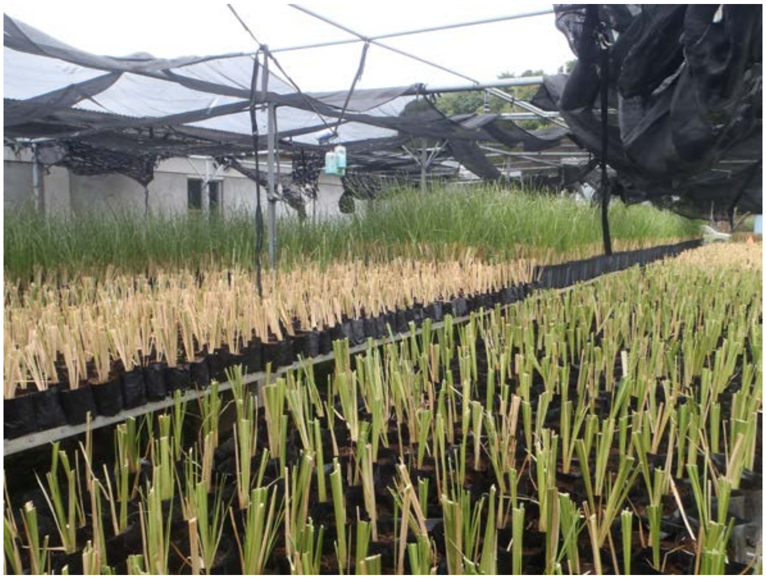


Figure 4: Vetiver grass mass production preparation in the nursery.

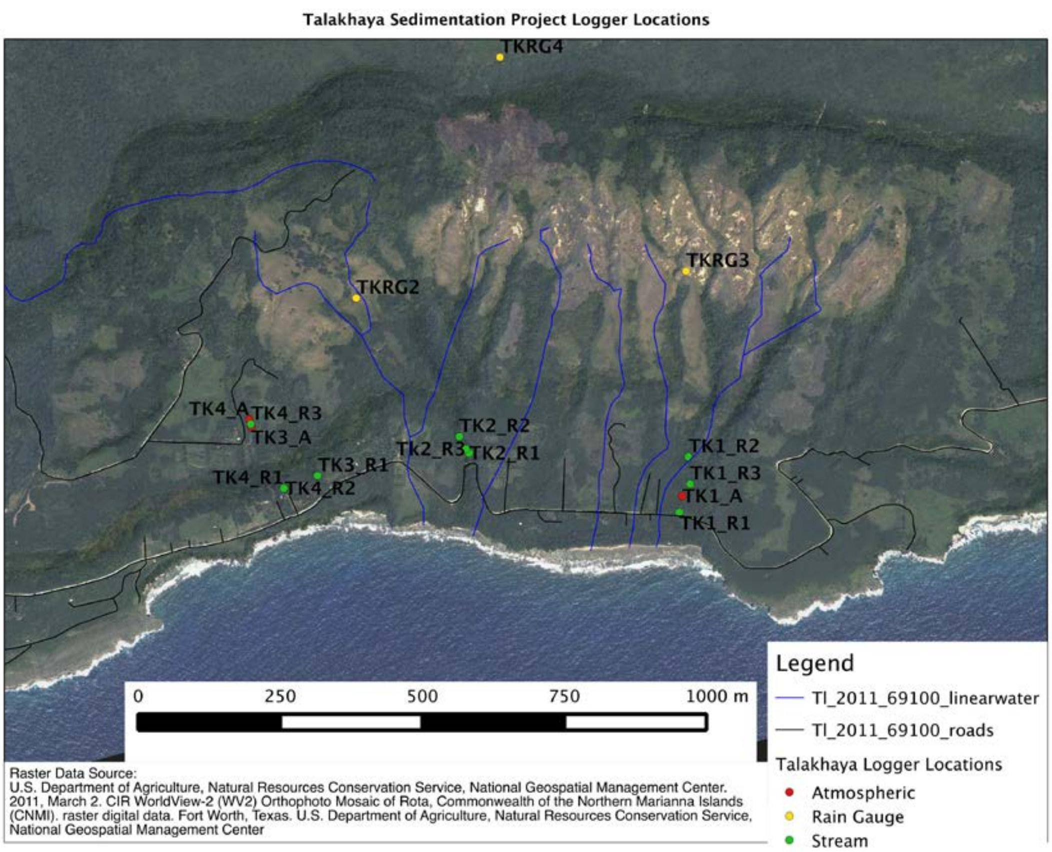


Figure 5: Talakhaya Sedimentation Project Logger Locations with code names color labelled by logger type (USDA NRCS, National Geospatial Management Centre, 2011).

APPENDIX III – Graphs and charts

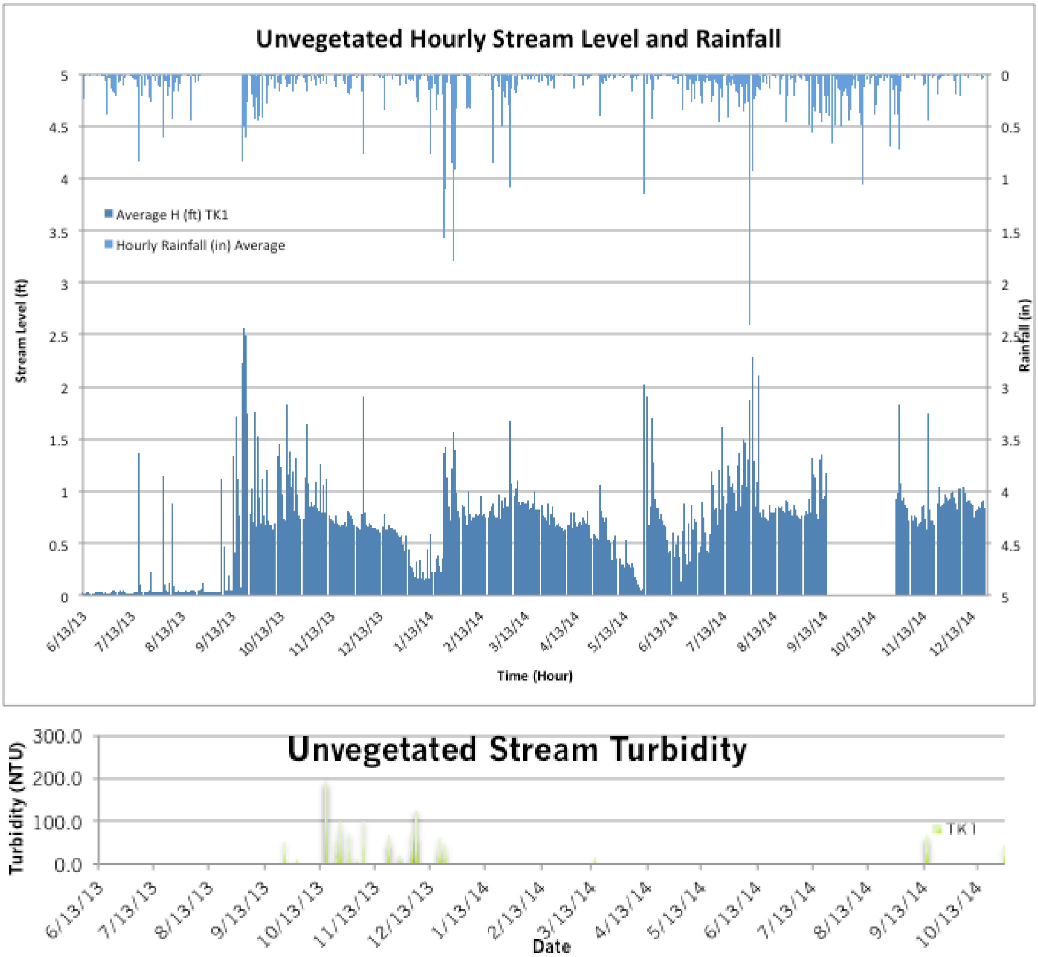


Figure 6: Unvegetated Hourly Rainfall, Stream Level, and Turbidity for TK1

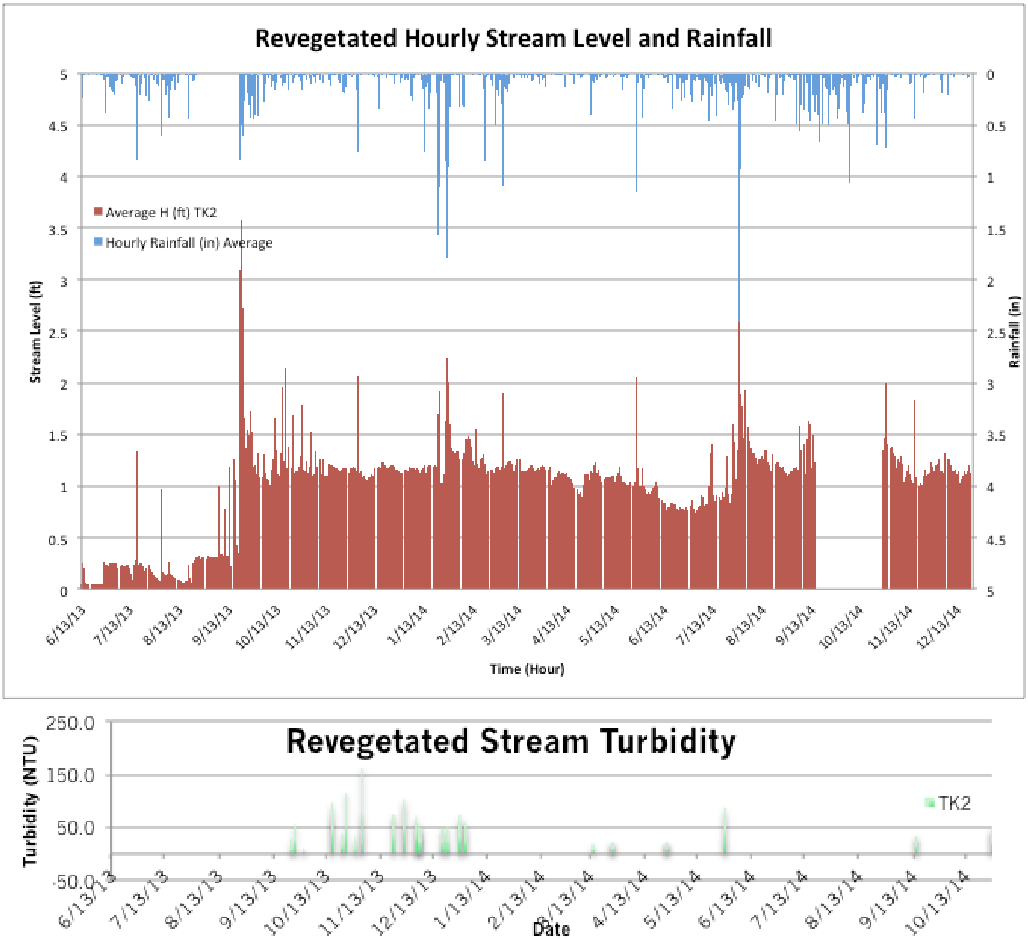
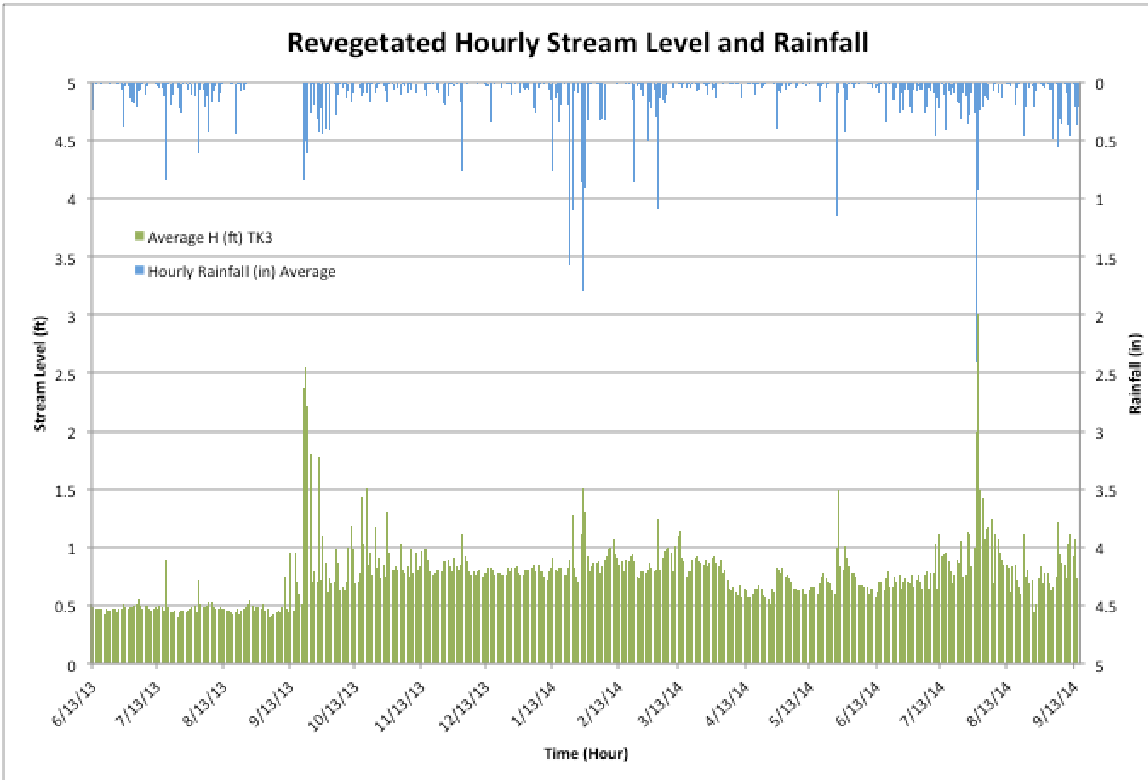


Figure 7: Revegetated Hourly Rainfall, Stream Level, and Turbidity for TK2



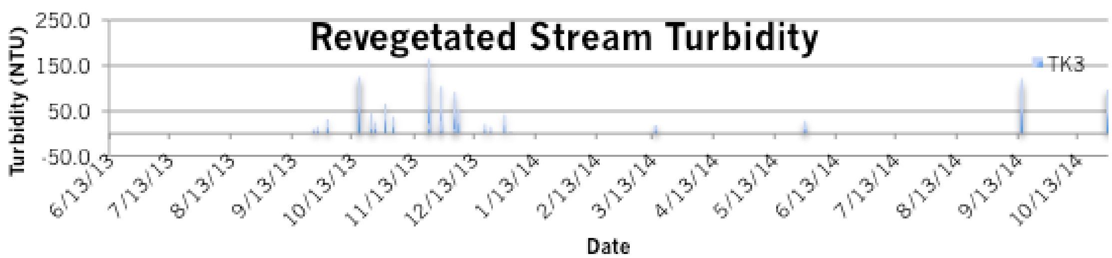


Figure 8: Revegetated Hourly Rainfall, Stream Level, and Turbidity for TK3

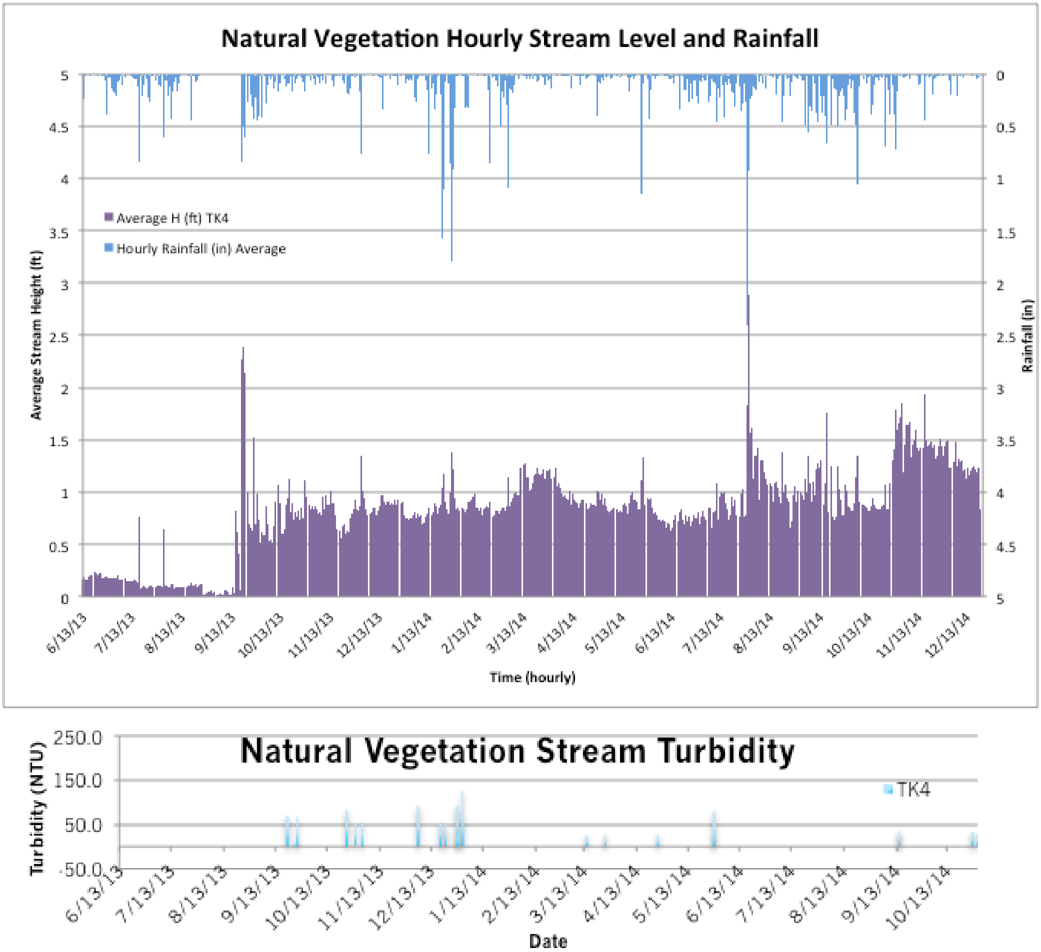


Figure 9: Non-vegetated Control Hourly Rainfall, Stream Level, and Turbidity for TK4

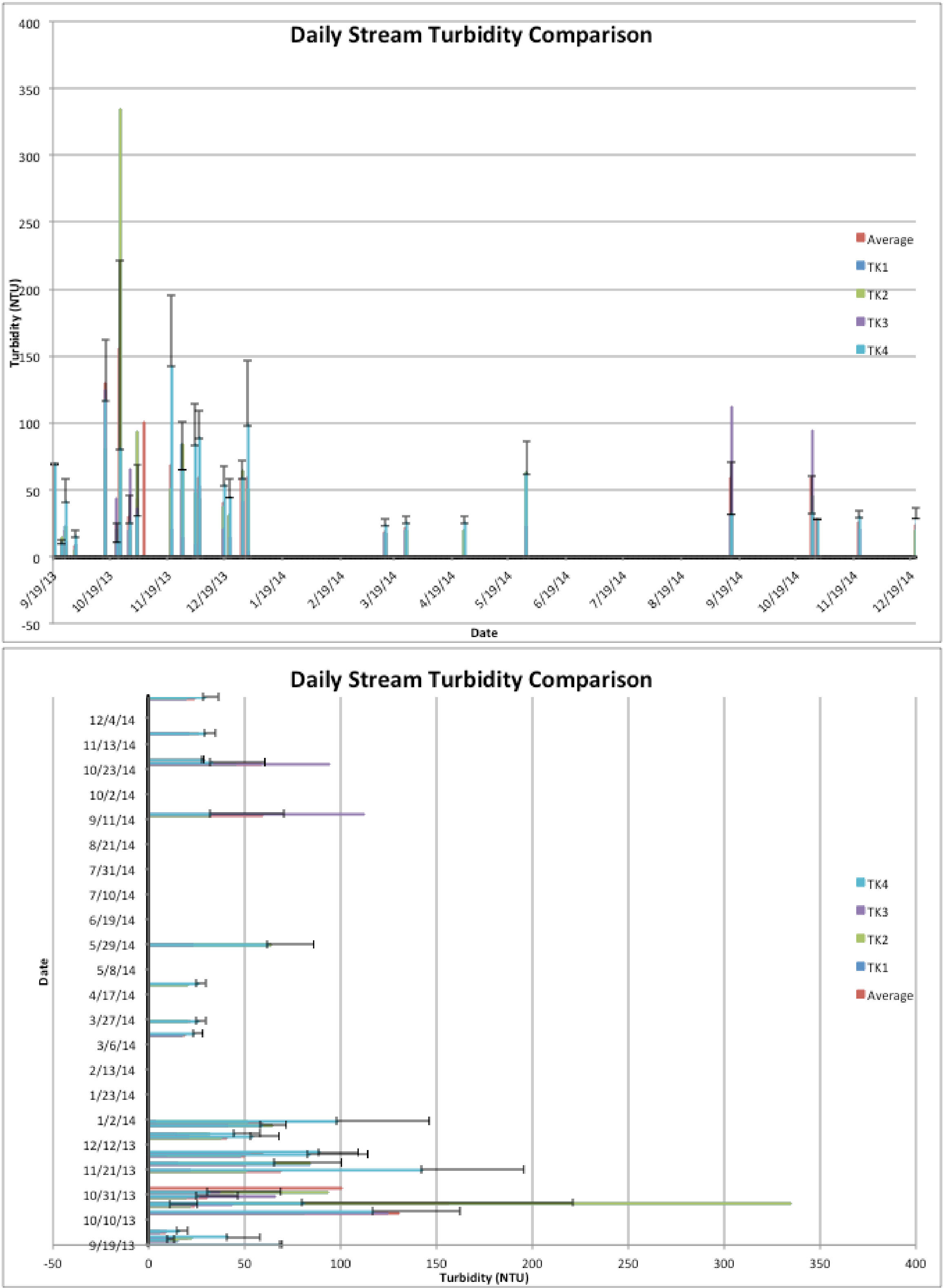


Figure 10: Daily Stream Turbidity Comparison.

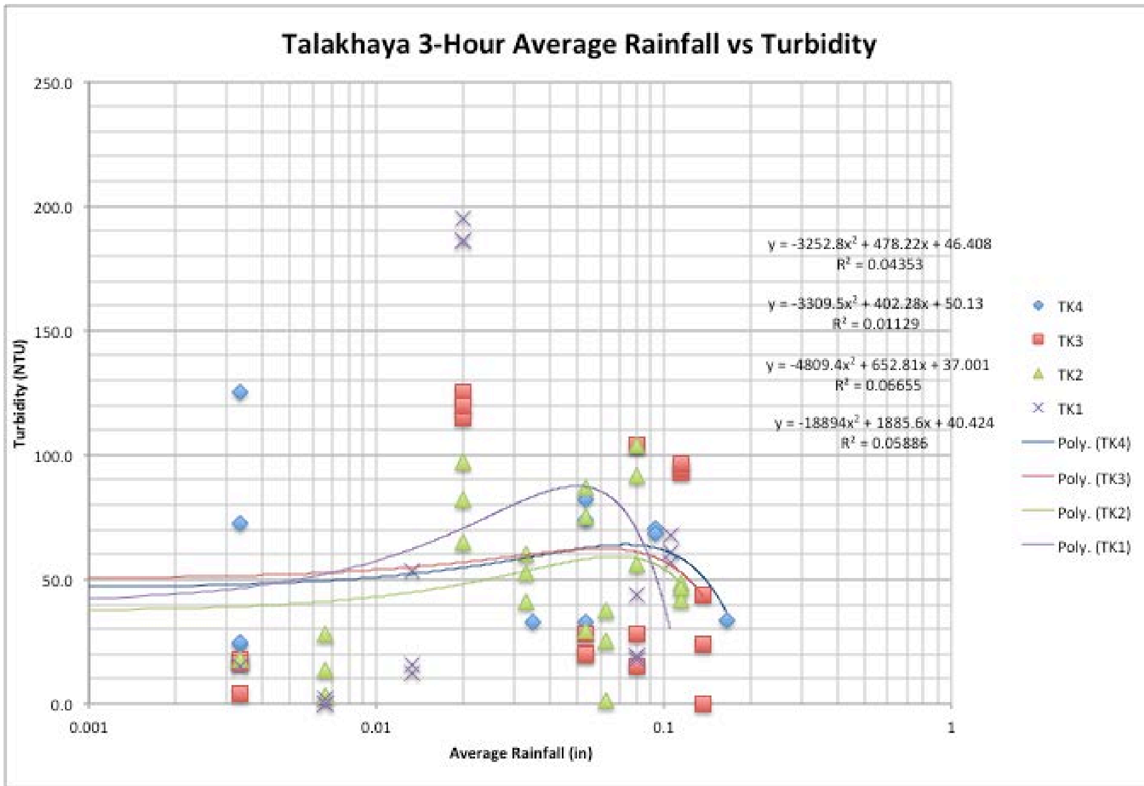


Figure 11: Talakhaya 3-Hour Average Rainfall vs. Turbidity

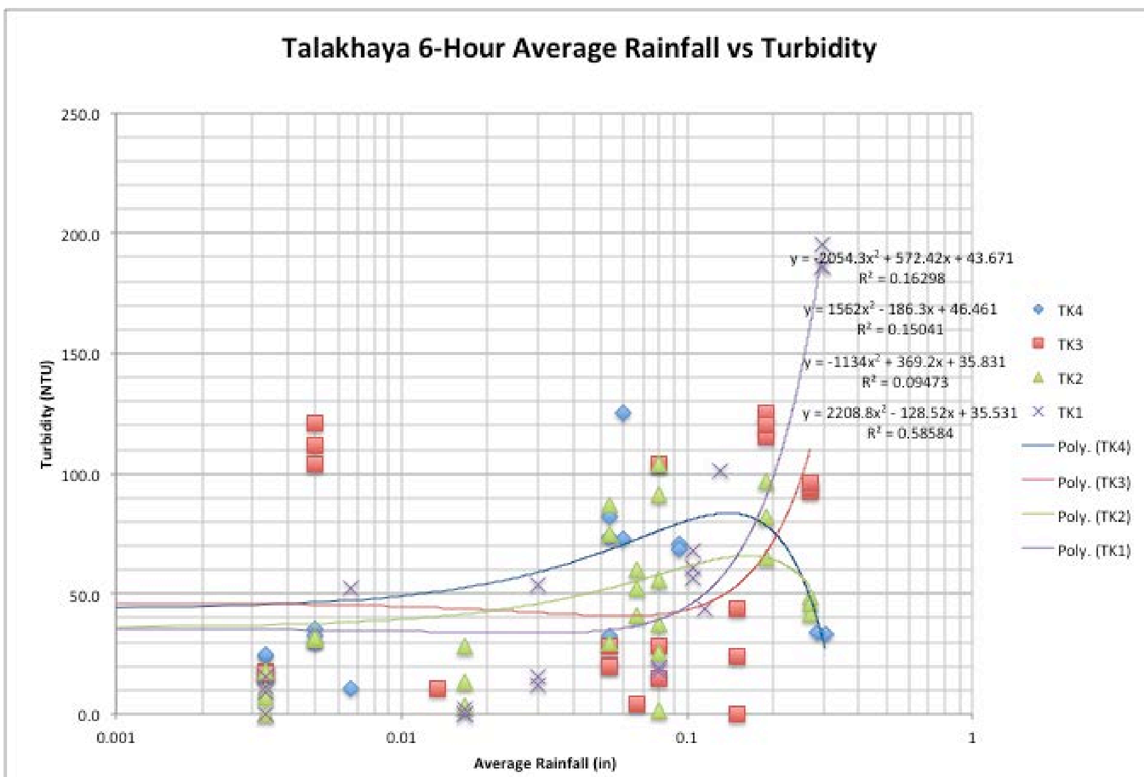


Figure 12: Talakhaya 6-Hour Average Rainfall vs. Turbidity

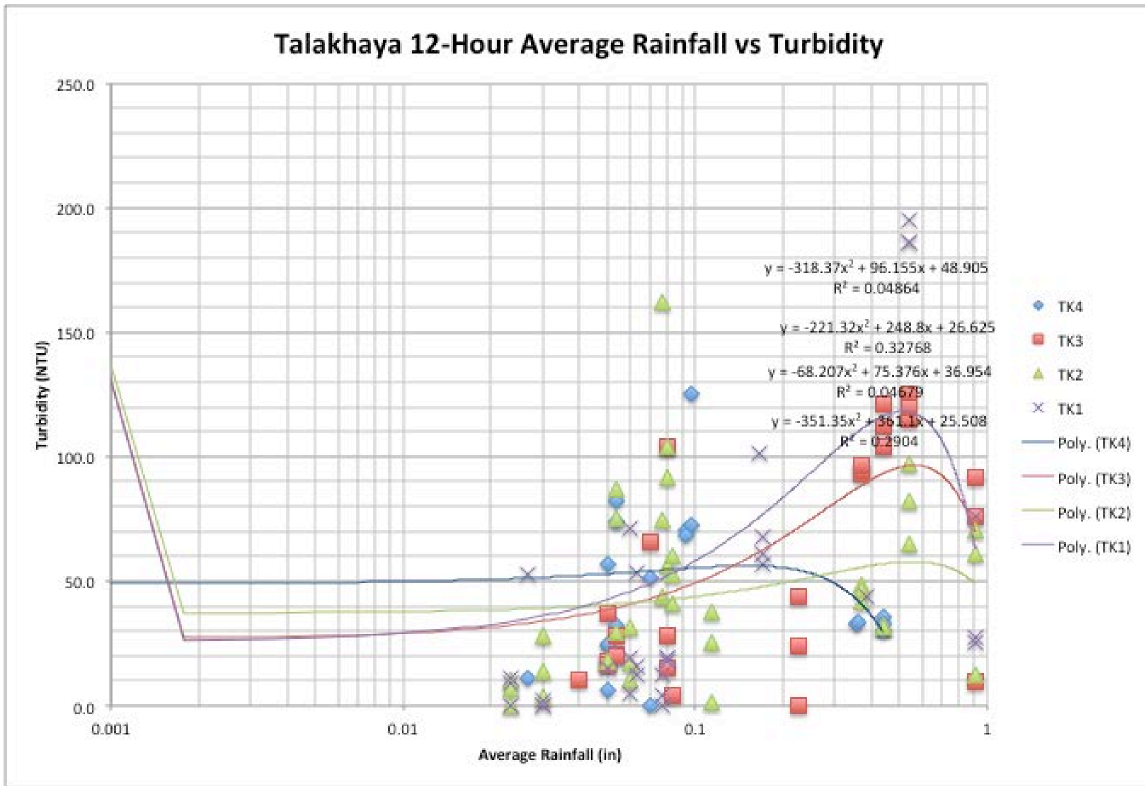


Figure 13: Talakhaya 12-Hour Average Rainfall vs. Turbidity

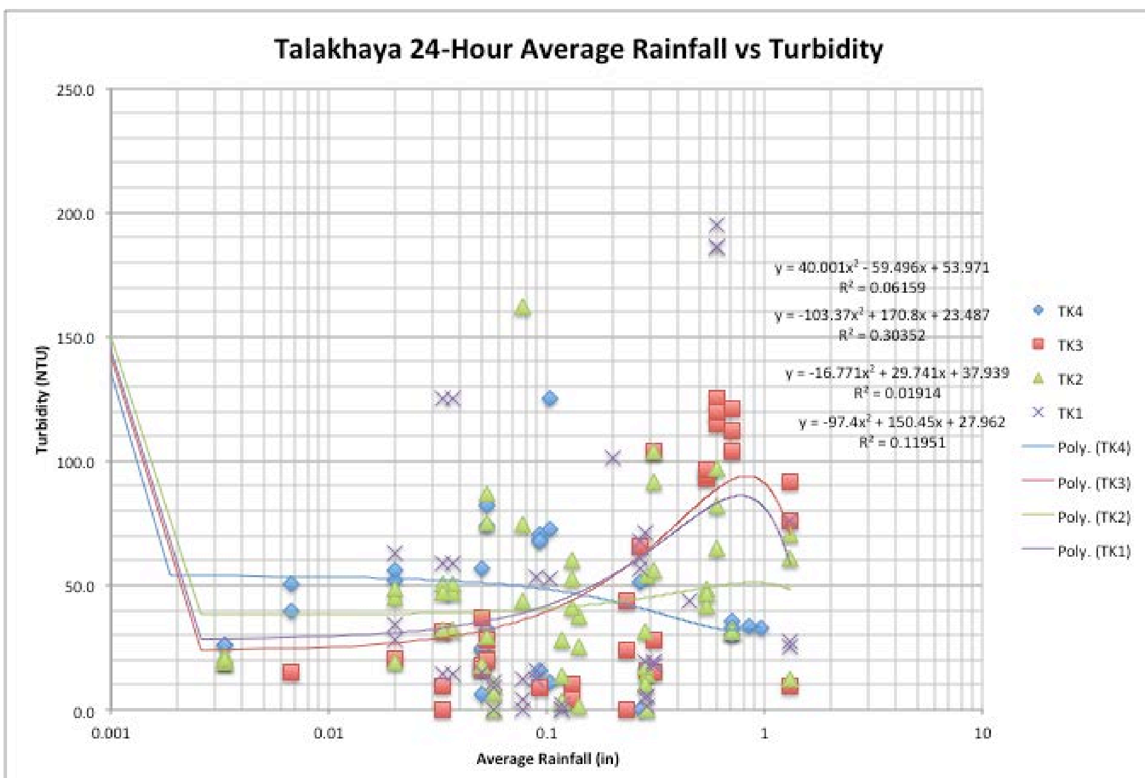


Figure 14: Talakhaya 24-Hour Average Rainfall vs. Turbidity

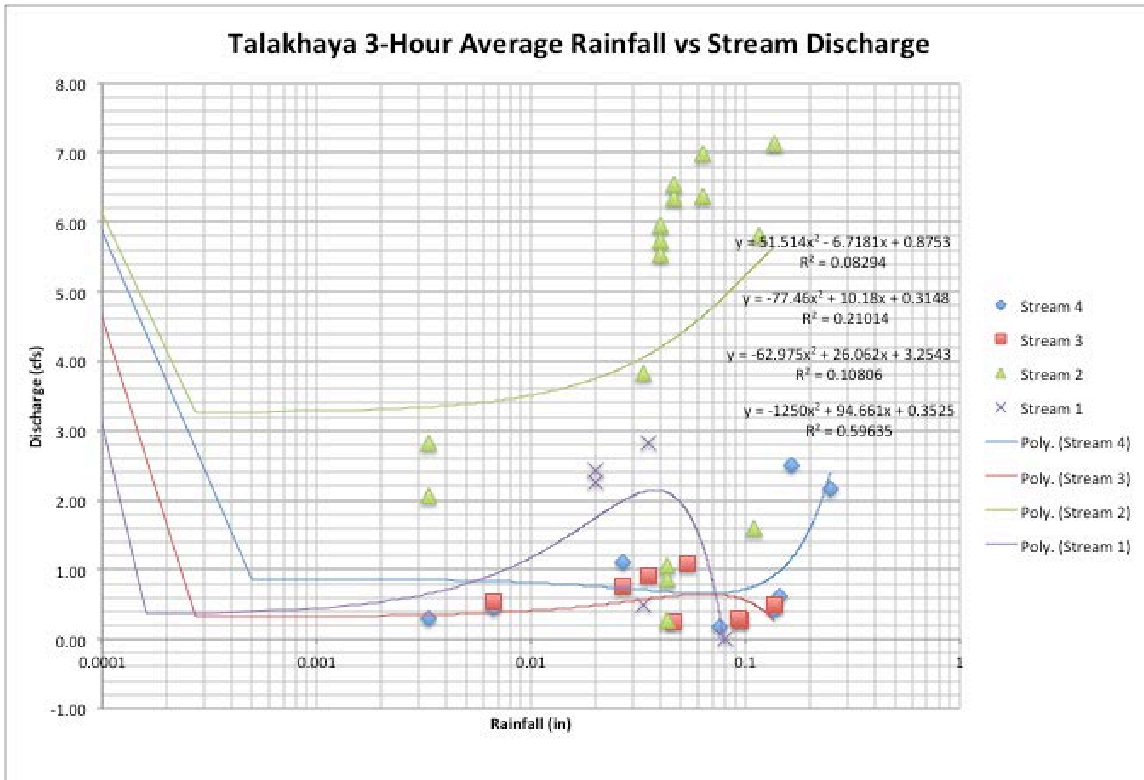


Figure 15: Talakhaya 3-Hour Average Rainfall vs. Stream Discharge

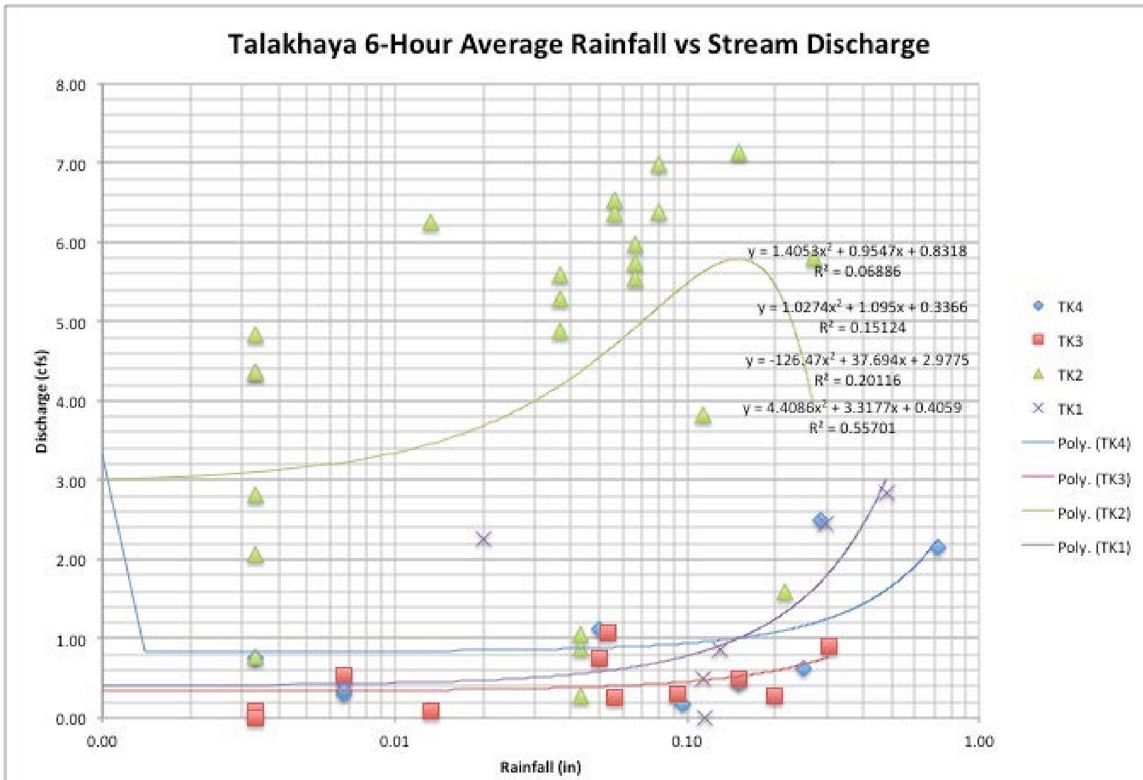


Figure 16: Talakhaya 6-Hour Average Rainfall vs. Stream Discharge

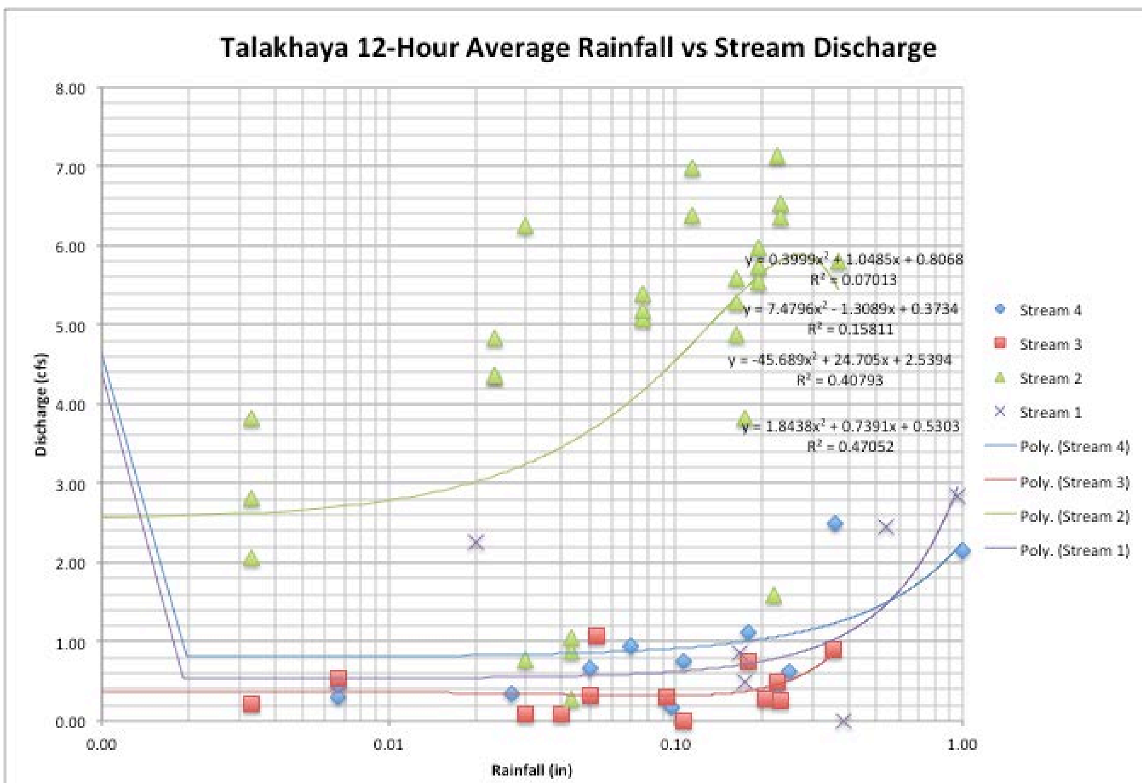


Figure 17: Talakhaya 12-Hour Average Rainfall vs. Stream Discharge

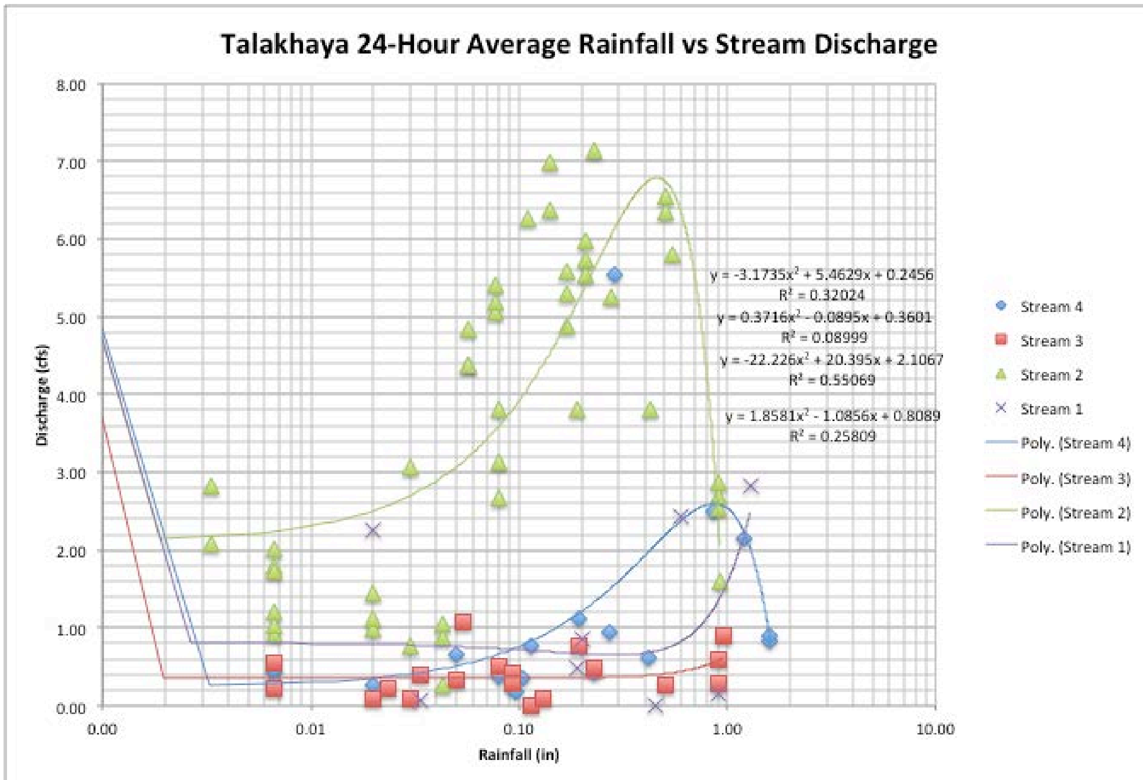


Figure 18: Talakhaya 24-Hour Average Rainfall vs. Stream Discharge

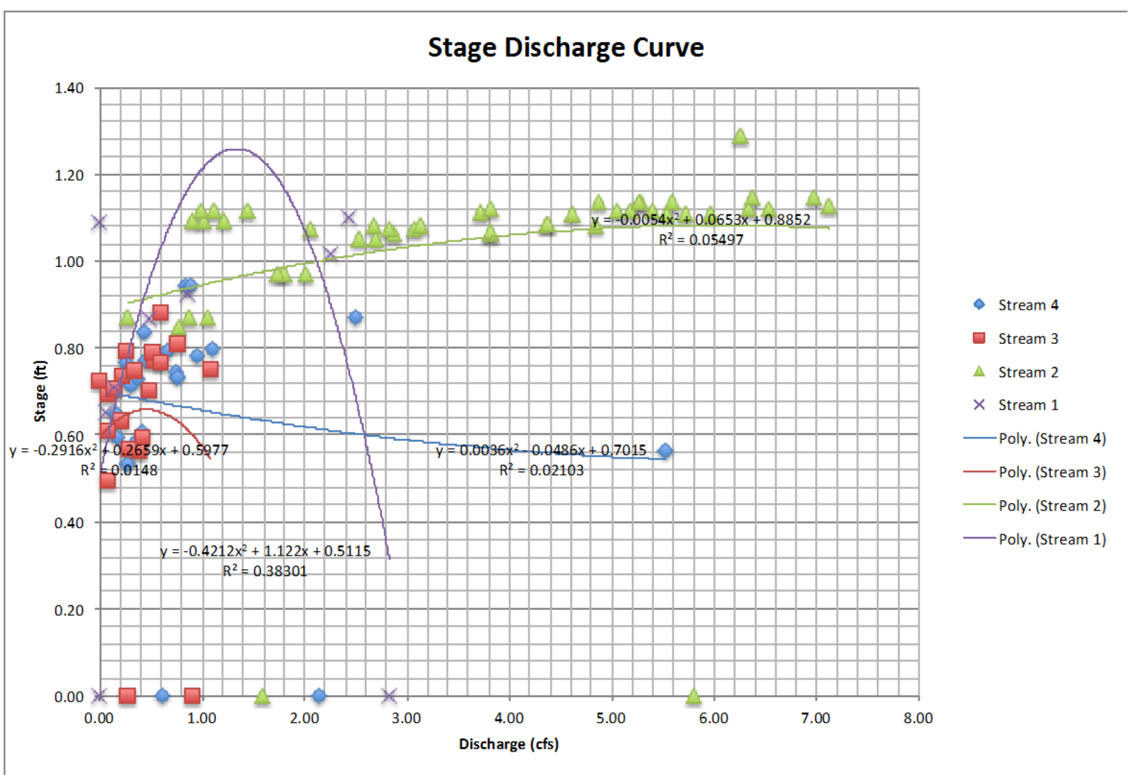


Figure 19: Stage Discharge Curve for Talakhaya Gauged Streams

7.3.O

CHANGES IN RUNOFF ENERGY, SURFACE LANDFORM AND SEDIMENT YIELDING DURING THE BANK GULLY EROSION PROCESS IN THE YUANMOU DRY-HOT VALLEY REGION, SOUTHWEST CHINA

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1. INTRODUCTION

Gully erosion has been recognized as an important source of sediment and a major land degradation process in a range of different environments (Valentin et al., 2005). Soil loss produced by gully erosion represents a minimum of 10% and up to 94% of the total sediment yield (Poesen et al., 2003), decreasing the agricultural land productivity (Pimentel et al., 1995) and damaging downstream water supplies (Verstraeten et al., 2003). Headcut is a step change in the bed surface topography where intense, localized erosion occurs (Bennett et al., 2000), whose height is often viewed as an important parameter to describe the activity degree of gully heads (Li et al., 2014). Many studies have focused on the gully headcut development and migration in concentrated flow, providing important insight into the processes and mechanisms of gully headcut erosion (Bennett and Alonso, 2005; Wells et al., 2009). However, most of them were conducted with laboratory flume experiments whose soil condition is much different from the natural ones. The research is still needed to study the relationship between runoff energy loss, surface landform changes and sediment yielding during the bank gully erosion processes under field conditions with larger scale, thus to offer more precise knowledge on gully head developments.

2. METHODS

A bank gully was selected as the experimental site and its 5-m-long upstream catchment area were divided into five field flume experimental platforms (mean width of 1.43 m). The mean horizontal length of the upstream catchment area was 4.86 m, while the average horizontal length of the bed of the bank gully was 1.93 m, with a mean slope gradient of 9.0° (Figure 1). Five gully headcuts were constructed based on the in situ active head with different heights of 25, 50, 75, 100 and 125 cm (Figure 2).

A series of simulated scouring tests was carried out to simulate the concentrated flows in the bank gully. The flow rate was set as 120 $L \cdot min^{-1}$, and the simulated scouring tests lasted 140 min. During these experiments, the hydraulic and sediment yielding parameters were observed at regular time intervals.

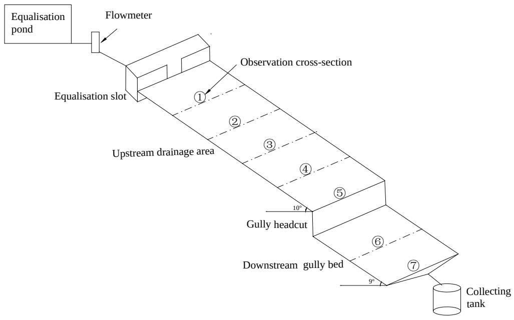

Figure 1. Sketch of the simulated experimental platfor

Figure 2. Field experimental platforms with different headcut heights

Image-based three-dimensional (3D) reconstruction was used to collect the topographical data prior to scouring and after each single scouring test to help generate high-resolution digital elevation models (DEM). 3D models were automatically generated by Agisoft PhotoScan 1.1.6 professional software by importing the photos and control points. The erosional characteristics and morphological parameters of the gullies were obtained from pre- and post-scouring DEMs using the 3D analyst and spatial analyst tools of ArcGIS 10.1.

3. RESULTS

3.1. Changes in runoff energy

The potential energy decreased gradually with the flow down, however, there were no significant differences in the potential energy at the upstream areas for the five gullies, but significant differences were observed in the gully beds, which can be attributed to the effects of the steep headcut (Fig.3). The kinetic energy also showed a significant decreasing trend over the cross-sections, which was contrary to some previous studies.

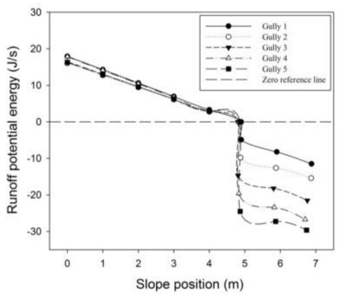

Figure 3. Spatial characteristics of runoff potential energy for five plots

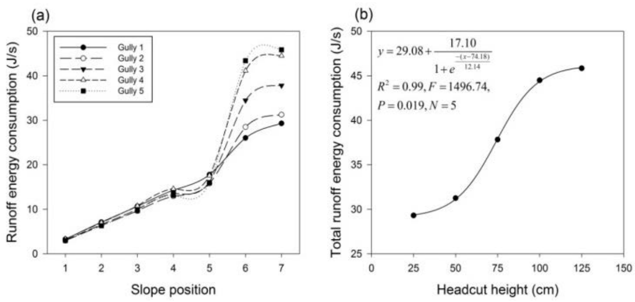

Figure 4. Change of runoff energy consumption over the slope position (a) and over the headcut height (b)

The total runoff energy consumption increased significantly as the headcut height increased (Fig.4). Both the decreases of potential and kinetic energy indicated that more runoff energy consumption occurred in the bank gully erosion process compared to the energy transformation.

3.2. Gully erosion processes

3.2.1. Landform change characteristics

Figure 5 shows the longitudinal elevation profiles of each bank gully pre- and post-scouring along the flow direction. Significant vertical incisions were observed in the upstream areas and gully beds, and both the vertical incision and headcut retreat occurred in the gully heads. The downstream gully beds have been subjected to much deeper incisions than the upstream areas, which can be attributed to the gentle slope gradient and the loose underlying surface soil.

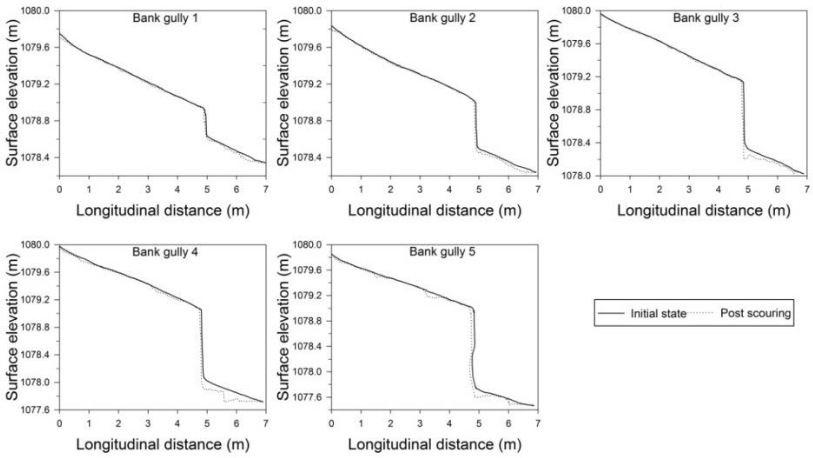

Figure 5. Longitudinal elevation profiles of each bank gully pre- and post-scouring

In addition, significant vertical incisions were observed because of the plunge pool development under the gully headcut, and the mean incision depth values were much higher than those in the upstream areas and gully beds. The incision depth of the plunge pool showed a significant increasing trend as the headcut height increased under the experimental conditions.

3.2.2. Soil erosion variations of gully heads

The result showed a significant linear increasing trend between the soil loss volume and headcut height (Figure 6(a)). Furthermore, Figure 6(b) shows the mean soil loss volume per unit area variation in the head position for five gullies. As the experiments progressed, the mean soil loss volume gradually increased, with a significantly logarithmic trend. Following an initial period of rapid increases, the mean erosion volume values tended to be stable state conditions gradually.

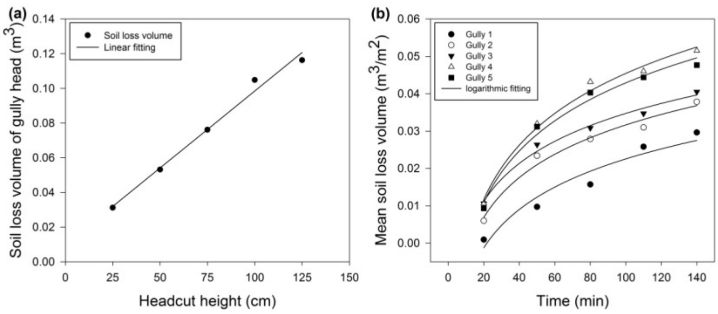

Figure 6. (a) Soil loss volume variation over headcut height and (b) change of mean soil loss volume per unit area over time for the tested gully heads

3.3. Characteristics of sediment yielding and delivering

The runoff sediment concentration showed a gradually declining trend over time and tended to steady-state conditions at the end. At the initial period of each scouring test, the runoff sediment concentration showed several sudden increase points, which can be explained by the fact that newly loose topsoil was generated due to the alternating dry and wet conditions. The result indicated that more sediment has been transported in the gully head and downstream gully bed, and the result also verified that the headcut migration would be associated with significant increases in the sediment yield (Wells et al., 2009). In addition, greater sediment increases have been determined with the increase of the headcut height, which indicated the potential effects of the headcut height on the sediment yielding during the bank gully erosion.

4. CONCLUSION

The results showed that (1) both the potential and kinetic energy showed decreasing trends from the spatial perspective, with much more energy being consumed in the gully headcuts, and the total runoff energy consumption has increased significantly as the headcut height increased; (2) landform changes exhibited some variations in different parts, with more landform changes in the gully heads due to the vertical incision and headcut retreat; while higher gully headcuts experienced much more landform change and soil loss; and (3) headcut migration made greater contributions to the sediment yield than the upstream areas, and the average runoff sediment concentration showed an exponential growth trend as the headcut height increased. All the results demonstrated the significant impacts of the headcut height on the characteristics of runoff energy, landform change and sediment yielding during the gully headcut erosion process in the Dry-hot valley region.

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7.1.P

QUANTITATIVE ASSESSMENT OF SEDIMENTATION IN DIFFERENT SECTIONS OF THE NIIDA RIVER FLOODPLAIN AFTER EXTREME EVENT

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Extreme flood events are very typical of small mountain river basins in different parts of tropical and sub-tropical landscape zones. They usually result in both intensive erosion and sediment deposition on the river valley bottom. Also, sedimentation rates considerably influence alluvial soil formation and contamination, depending on the sources of sediment. Detailed study of different sections of floodplain was undertaken in the Niida River basin (Fukushima Prefecture) after an extreme flood event which occurred in the middle of September 2015, when 385-456 mm of precipitation fell during 6 days with maximum 200 mm/day in the end of typhoon. The objectives of the study are: to evaluate the erosion/deposition processes and its spatial changes in different sections of the river floodplain, with particular attention to the low reaches of the Niida River, and to assess changes in alluvial soil contamination in comparison with the level of contamination before the extreme flood.

The Niida River basin has the total area of 248 km^2 and it is located on the eastern part of the Fukushima prefecture. Level of initial radionuclide deposition after the FDNPP accident varies considerably along the Niida River with maximum contamination in its southwestern part (the Hiso River basin), with decrease in both northern and eastern directions. All previously used agricultural lands of the headwater uplands both in depressions and on hillslopes have been abandoned since spring 2011 immediately after the accident. Intensive decontamination work (complete removal of the contaminated upper topsoil layer for the long-term storage) is underway now starting from the northwestern part of the basin. Coastal lowlands were not seriously affected by the radionuclide fallout. Agricultural lands in this part of the basin are still in use. The Niida basin is characterized by wet monsoon climate with highly variable total annual precipitation (1300-1500 mm). Less than 5-7% of precipitation falls as snow. The alluvial sandy and loamy sandy soils is typical of the river floodplain. They are characterized by high moisture content and stratification with interlayers of light loam. More detailed description of the Niida river basin can be found elsewhere (Konoplev et al., 2016).

Field and GIS methods were used, including direct field measurements of the depth of fresh sediment and its area with soil descriptions for the typical floodplain sections, measurement of dose rates; interpretation of space images of the river valley for a few time intervals (before and after flood event) with the following evaluation of spatial transformation of the river bottom, including river channel and different floodplain levels.

Results of quantitative assessment of sedimentation rates were used for understanding of the effect of extreme flood on alluvial soils of the different reaches. The sedimentation rates in the upper part of the Niida River basin within intermountain depressions depended on the type of the channel. They are in the range 10-15 cm (one order of magnitude higher than the mean annual) for the artificially canalized channels with concrete embankments and levees, which are the dominant type for main rivers draining the basin headwaters upland depressions. However, sedimentation rates are only 1-3 cm for the uppermost section of the artificially canalized channels. Some valley reaches within uplands depression have wider channels, but unconfined by levees or embankments (though there can be levees on one or both sides, but remote from the channel along the higher floodplain or low terrace level). Here in some section within the floodplain there was extremely high deposition of sand and pebble material with an average depth of 30-40 cm and a total weight of 400-500 tons, which, in fact, amounts to one tenth of the total measured flow volume of suspended sediments. The most detailed study was undertaken for the lower reach of the Niida River located within the coastal lowlands. It was possible to divide the lower reach into three sections according to decreasing of the channel gradients towards to the river mouth. Evaluation of areas which were undergone by bank erosion, in-channel deposition and active overbank sedimentation was undertaken for each section and presented for the different types of channel, determined according to the river channel classification (Alabyan and Chalov, 1998) (Table 1).

Table 1. Mean areas of the lower reach of the Niida river valley bottom affected by bank erosion, in-channel deposition and intensive overbank sedimentation

Type of channel	Bank erosion, m^2	In channel deposition, m^2	Floodplain sedimentation, m^2
	Section 1 *		
Meandering	16732	7178	32970
Braided	13228	608	21615
Straight	14983	1196	7894
	Section 2		
Meandering	36431	13144	79499
Braided	10691	2290	19523
	Section 3		
Meandering	14562	19713	71510
Braided	7473	1675	18776
Straight	3232	369	644

* *Section of the river valley bottom of the lower reach were separated according to the decrease of the river channel gradient towards the mouth*

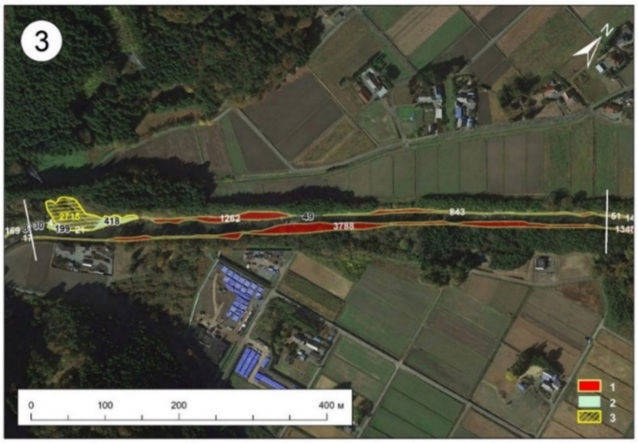

Figure 1. The erosion (1), deposition (2) and sedimentation (3) zones appeared after extreme flood at the section 1 (semi-mountain sub-reach) within the straight type of the Niida River channel.

It was found that bank erosion is particularly high at the section 1, because of the high energy of the flow after the exit of the Niida river from the mountains to the coastal plain in particular in the river reaches with straight type of channel (Table 1, Fig.1). Generally, erosion prevails the in-channel deposition for the all sections of the lower reach of the Niida river except meandering channel type at the section 3 (Table 1). However, the meandering type of channel is characterized by the relative equilibrium relationship of erosion and in-channel deposition even at the section 1 with the most active valley bottom transformation including overbank sedimentation (Fig.2; Table 1).

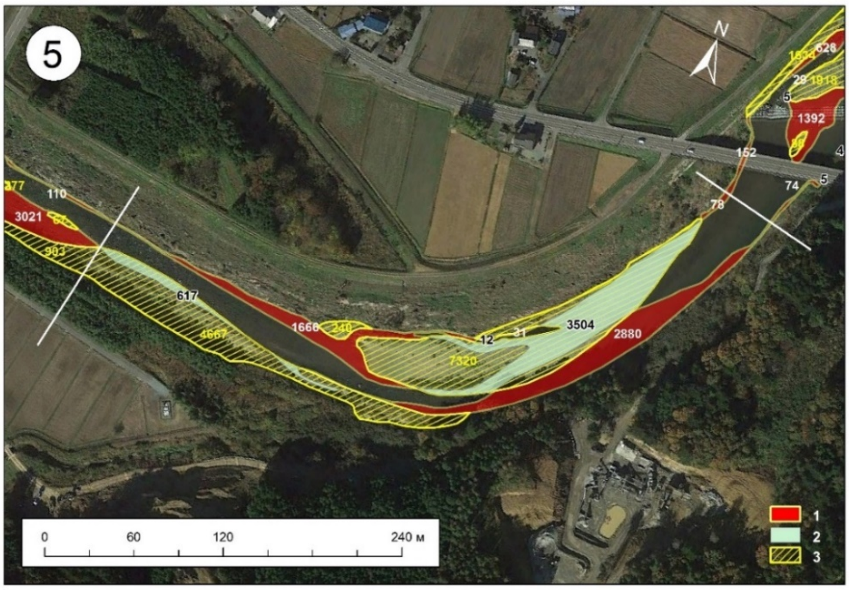

Figure 2. The erosion (1), deposition (2) and sedimentation (3) zones appeared after extreme flood at the section 1 (semi-mountain sub-reach) within the meandering type of the Niida River channel.

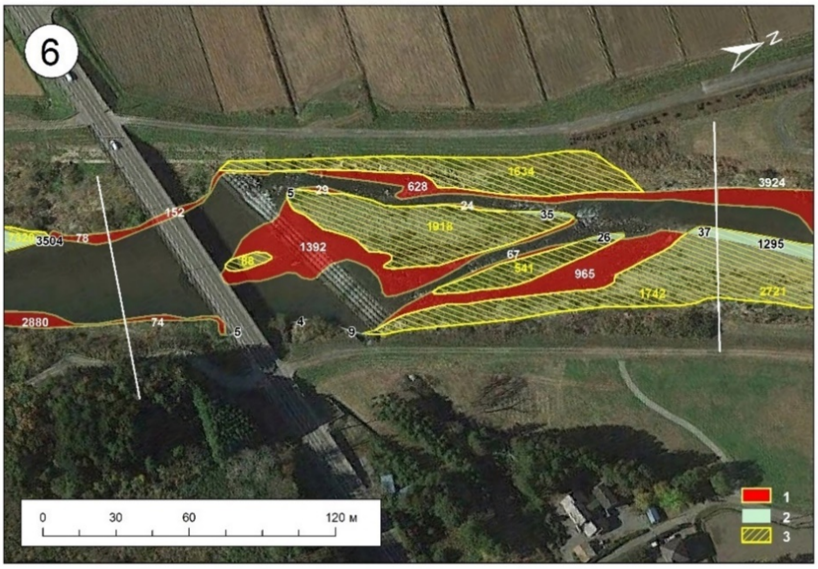

Figure 3. The erosion (1), deposition (2) and sedimentation (3) zones appeared after extreme flood at the section 1 (semi-mountain sub-reach) within the braiding type of the Niida River channel.

The braided type of the river channel is characterized by the different modification of the river valley bottom depending on the river channel gradient. Active erosion was identified for the braided channel at the section 1 (Fig.3), while intensive in-channel deposition was observed at the section 3 (Fig.4). It is necessary to stress that overbank sedimentation at the lower reach of the Niida River leads to the reduction of the dose rate, even though the dose rate was low initially because of sand-gravel grain size of the sediment. Dose rates were reduced by 3-5 times for all the floodplain sections in both upper and lower reaches of the Niida River with high sedimentation rate because the top soil layers with high radionuclide contamination were buried under fresh sediments produced mostly due to bank erosion and mass movements.

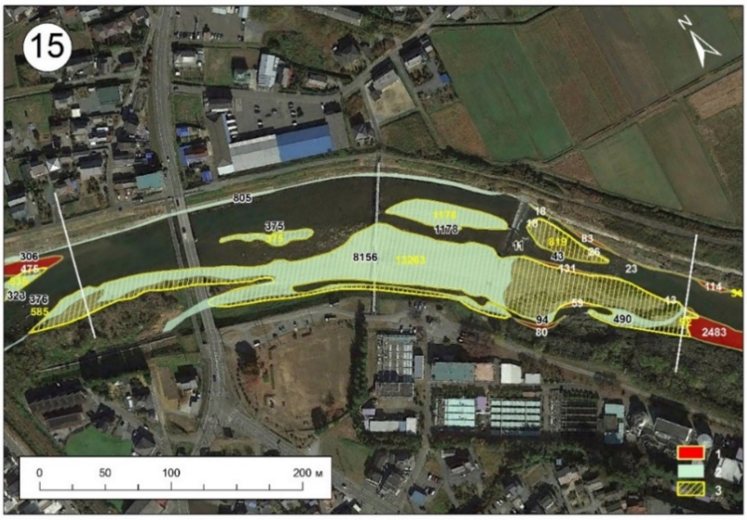

Figure 4. The erosion (1), deposition (2) and sedimentation (3) zones formed after extreme flood at the section 3 (plain sub-reach) within the braided type of the Niida River channel.

CONCLUSIONS

The detailed assessment of geomorphological and radioecological consequences of the extreme flood in the Niida River basin allows to conclude that different exogenic processes (landslides, scree, bank erosion etc.) were the main sediment sources for the river sediment discharge. Maximum channel deformations were observed within the lower reach of the Niida River because of extremely high energy of the stream with maximum water and sediment discharges. However, sedimentation rates on the floodplain of the upper reaches of the Niida River were also one order of magnitude higher than the mean annual rates. Finally, extreme event contributed to the serious reduction of the dose rate of the river bottom except for some high floodplain levels where sedimentation rates were negligible.

ACKNOWLEDGEMENTS

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7.2.P

MODELING MONTHLY STREAM FLOW AND SEDIMENT DISCHARGE WITH SWAT IN A HEADWATER CATCHMENT, SOUTHEASTERN BRAZIL

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INTRODUCTION

The São Paulo Metropolitan Region is the most populous region of Latin America, with approximately 22 million inhabitants, of which 47% are supplied with water from the Cantareira System. Because of the high water demand from this supply system, the Water Conservation Project was created with the objective of improving the environmental suitability of rural properties. The Posses watershed, a small tributary river that drains to Cantareira, was selected as a priority area to receive the reforestation and the soil and water conservationpractices in order to reduce soil losses and optimize the provision of water downstream.

Several studies are being done in this basin in order to evaluate the impact of conservation practices on water regime and sediment yield. Water erosion is the main form of soil degradation in Brazil, responsible for pollution and silting of rivers and dams. Given the complexity of hydrosedimentological processes, modeling is an important tool that can help to understand the water and sedimentological behavior in river basins.

The Soil and Water Assessment Tool (SWAT) is a time-continuous, semi-distributed and semi-physically based hydrological model that operates on a daily time step (Arnold et al., 1998). It is developed for assessing the impact of management and climate change on water supplies, sediment production and agricultural chemical yields in large basins. The model has been widely studied and applied to a number of regions in the world with satisfactory results, even for Brazilian conditions.However, there are still few studies that evaluate the modeling of soil losses and sediment production with SWAT, especially in small headwater watershedsin Brazil.

In this context, the objective of this work was to evaluate the modeling of the average monthly water flow and the sediment yield with the SWAT hydrological model in the Possess watershed.

METHODOLOGY

The Posses watershed is located between the coordinates 22,83° and 22,90° of latitude south and, 46,22° and 46,26° of longitude west. The basin has 12km^2 drainage area with altitudes between 968 and 1,420 m. The average annual temperature is 18 °C, and the average annual rainfall is 1,652 mm.

The area is located in mountainous region with Atlantic forest biome of the southeastern Brazil. The predominant soils classes are: Red-YellowArgisols (41% of the area), Cambisols (39%), FluvicNeosols (11%) and LitholicNeosols (9%). The predominant soil use is extensive pasture without conservation practices (Figure 1).

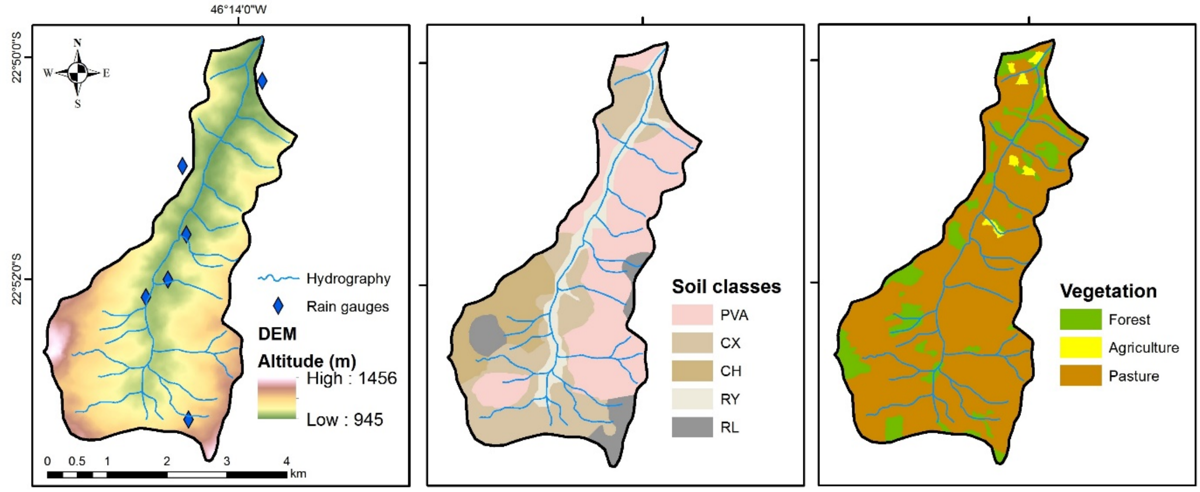

Figure 1.Posses watershed Digital Elevation Model (DEM), soil classes and vegetation. PVA – Red-yellow argisol, CX – HaplicCambisol, CH – HumicCambisol, RY – FluvicNeosol and RL – LitholicNeosol.

The hydrologic simulation was developed with the 2012 version of ArcSWAT. The relief was characterized using a digital elevation model (DEM) with spatial resolution of 15m. The daily climatic data was obtained from a nearby climatic station and the rainfall data fromgauges located within the Posses watershed. Data was gathered from 2009 to 2014.Water discharge measurements from 2009 to 2011 were used for model calibration, whereas the remaining data was used for validation of water flow hindcasts.

For the evaluation of sediment production, a regression was performed between turbidity data and suspended solids for water samples from the Posses Stream. With the generated equation the sediment concentration was estimated for bi-monthly turbidity data of the water. Then, a regression was generated between the sediment concentration and the observedwater flow, in order to generate a curve of suspended solids discharge.

The SUFI-2 algorithm from SWAT-CUP software was used for the calibration, validation and uncertainty analysis. This algorithm allows the stochastic evaluation of the simulation through the p-factor and r-factor statistics, based on the 95% prediction uncertainty(95PPU). The 95PPU is calculated at the levels of 2.5% and 97.5% of the cumulative distribution of the results generated by the propagation of the parameter uncertaintiesusingLatin hypercube sampling. The p-factor represents the fraction of observed data inserted in the 95PPU range and the r-factor is the ratio of the mean width of this range to the standard deviation of the observed data. Values of p-factor> 0.7 and r-factor <1.5 are recommended(Abbaspour et al., 2015). The Nash-Sutclife coefficient (NSE) was used to evaluate the modeling results.

RESULTS AND DISCUSSION

The observed and simulated hydrograms for the calibration and validation periods as well as the 95PPU range are shown in Figure 2.

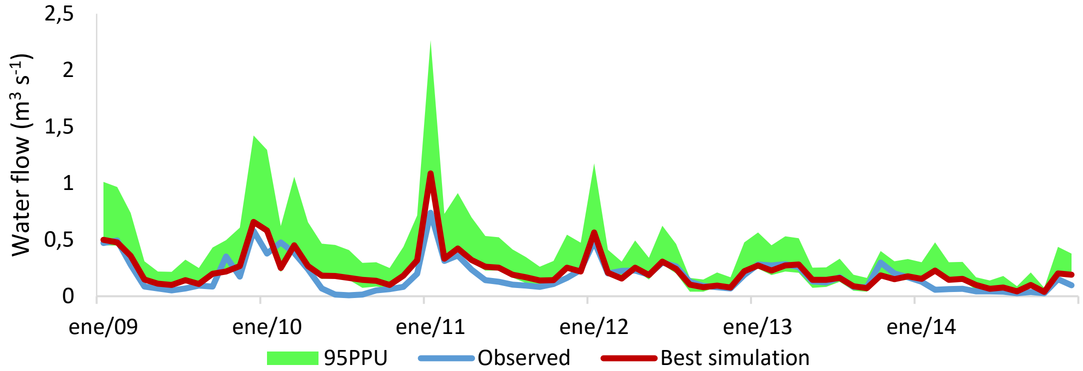

Figure2. hydrograms for calibration and validation periods.

Agood match between the simulated and observed average monthly water flow can be observed,specially at the calibration period. For the validation period it was observed a general overestimation at the minimum and maximum water flow. The validation period presents a greater variation between the observed values of maximum and minimum flow, which may explain the lower values of NSE, which can still be considered acceptable (Table 1).

Table1. Results of the indexes used to evaluate the modeling performance calibration and validation for the monthly mean water flow and sediment yield (SED).

	Water flow		Sediment yield	
Index	Calibration	Validation	Calibration	Validation
p-factor	0.64	0.31	0.94	0.89
r-factor	0.54	0.58	1.84	2.17
NSE	0.79	0.51	0.56	0.45

The p-factor for the streamflow 95PPU was below the recomended (Abbaspour et al., 2015).However, the r-factor is also small, that is, the 95PPU range is very narrow, indicating that the range of variation of the calibrated parameters could be higher, which would increase the p-factor.

Good results have been obtained with SWAT in large Brazilian river basins. Pontes et al. (2016)reportedNSE values of 0.85 and 0.88 for the calibration and validation of the average monthly flow in the Jaguarí River Basin, of which the Posses Stream is tributary. For small basins, the model also presents satisfactory results in streamflow simulation, and unsatisfactory for sediment yield (Bonuma et al., 2014).

NSE values for sediment yield modeling were lower than the ones for water flow, in both calibration and validation stages. However,these valuesare considered satisfactory for the modeling of a phenomenon as complex as erosion.

Regarding the uncertainty analysis,the 95PPU for the sediment yield envelops 89% of the discharge ofsolids variation for the validation period and 94% for calibration. However, the 95PPU range is too wide (Figure 3), which indicates a high degree of uncertainty in the input parameters related to the erosive process.

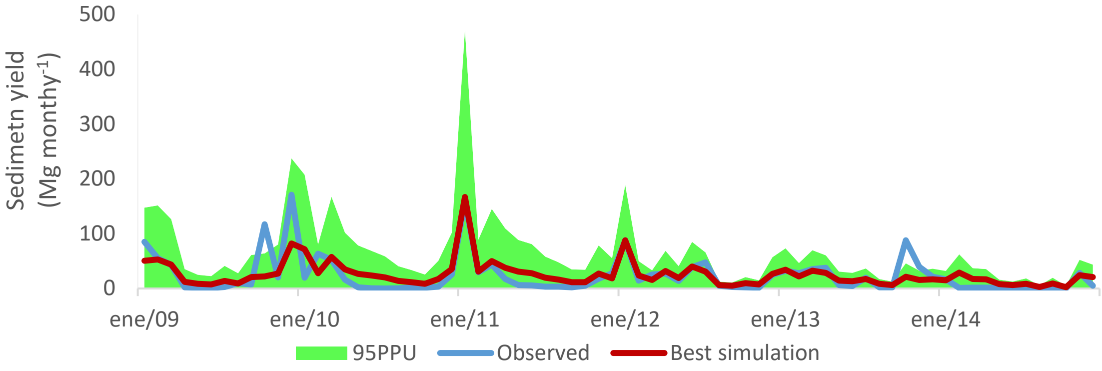

Figure3. Discharge of suspended solids for calibration and validation periods

During the modeling process, it was observed that the parameters related to soil loss in the landscape did not show any sensitivity in the SWAT, being calibrated only the parameters referring to the channel routing. Thus, the model cannot be used to assess the impacts of changes in landuse or the adoption of conservation practices on sediment yield. On the other hand, the model can be used to estimate the siltation in downstream reservoirs.

CONCLUSION

Satisfactory results were obtained for the calibration and validation of the average monthly streamflow and the sediment yield in the basin.
The SWAT was not sensible to landscape soil losses parameter and does not allow to predict the impact of landuse changeon sediment yield.

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7.3.P

MODELING SOIL LOSSES AND SEDIMENT YIELD IN THE UPPER GRANDE RIVER BASIN, BRAZIL

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INTRODUCTION

The Upper Grande River Basin is mostly covered by degraded pastures used for dairy cattle raising. Shallow and little permeable Cambisols (Inceptisols) are the predominant soil class in the area, which, combined with an intensive and highly concentrated summer rainfall, characterize an erosion-prone scenario. The Grande River is one of the main tributaries of the Paraná River, and an important source of hydroelectric power in Brazil. Therefore, off-site erosion can be a great issue due to reservoir sedimentation.

In spite of the problems regarding on-site and off-site erosion in the Upper Grande River Basin, very few government owned river gauging stations measure sediment transport in this area. Hence, modeling erosion and sediment delivery is a useful tool in order to estimate sediment yield.

The Sediment Delivery Distributed model (SEDD) (Ferro and Porto, 2000) combines RUSLE (Renard et al., 1997) gross erosion predictions with a semi-empirical and spatially distributed calculation of sediment delivery ratio (SDR). Hence, the model is able to estimate sediment yield from river basins. Moreover, sediment sources and erosion-prone areas can be identified.

The aim of this work was to apply SEDD in the Upper Grande River Basin in order to model soil losses and sediment yield. It also sought to identify the main sediment sources in the study area.

MATERIALS AND METHODS

Study area

The Upper Grande River Basin covers an area of 15,705 km². The Köppen climate type is Cwb: Humid subtropical with dry winter and temperate summer. The average annual precipitation is about 1500 mm. Haplic Cambisols and Red Yellow Latosols are the predominant soil classes, spreading through 44% and 31% of the study area, respectively. Haplic Cambisols are shallow, not much permeable and often graveled, usually associated to hilly, mountainous slopes; Red Yellow Latosols are severely weathered, deep, very permeable soils, usually found in the flatter, gentle slopes along the landscape. Rangeland is the primary land use, frequently degraded by overgrazing and water erosion. Elevations range from about 800 m, near the basin outlet, to 2600 m, at the Mantiqueira mountain ridges. The basin can be divided in three main sub-watersheds: the Mortes River sub-basin, to the north; the Grande River sub-basin, in the central and southern regions; and the Capivari River sub-basin, to the west (Figure 1).

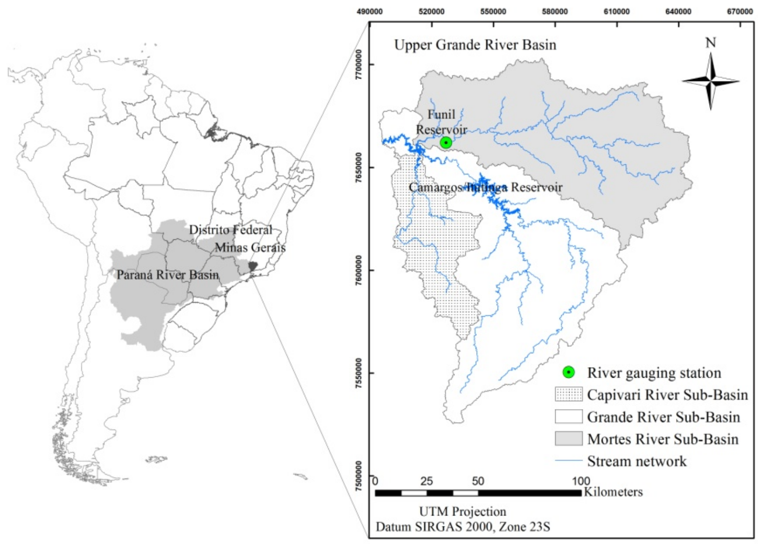

Figure 1. Location of the Upper Grande River Basin.

RUSLE

RUSLE estimates average annual soil losses by a direct equation in which five empirical factors are used to describe the processes affecting erosion (Renard et al., 1997):

$$A = R\ K\ LS\ C\ P$$

where: A is soil loss per unit area (t ha^{-1} yr^{-1}); R is the rainfall and runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}); K is soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}); LS is the topographic factor, representing slope length and steepness (dimensionless); C is cover management factor (dimensionless), and P is support practice factor (dimensionless).

In this study we composed grid layers of each RUSLE factor using GIS software. The R factor was derived from monthly and rainfall maps. A soil map of the study area was used to assign K factor values to specific soil classes. The LS factor was calculated using a 30m resolution DEM. C factor values were assigned to uniform land uses. A constant value of 1.0 was appointed for the P factor, assuming the absence of support practices.

SEDD

In the SEDD model, the area specific sediment yield (SSY) can be expressed as:

$$SSY_i = SDR_i * A_i$$

where: SSY_i is the specific sediment yield for a grid cell *i*; SDR_i is the soil delivery ratio for a grid cell *i* and A_i is the annual soil loss computed by RUSLE for a grid cell *i*.

The sediment delivery ratio (SDR) is defined as:

$$SDR_i = exp(-\beta t_i) = \exp\left(-\beta \frac{l_i}{v_i}\right) = \exp(-\beta \frac{l_i}{a_i * s_i^{0.5}})$$

where: SDR_i is the soil delivery ratio of a grid cell *i*; β is a catchment specific parameter (h^{-1}); t_i is the overland flow travel time (h) from a grid cell *i* to the nearest stream channel along the flow path; l_i is the flow length from cell *i* to the nearest stream channel (m); v_i is the flow velocity for cell *i* (m s^{-1}); a_i is a surface roughness coefficient for cell *i* (m s^{-1}) and *Si* is the slope for cell *i* (m m^{-1}).

Flow length and flow velocity were calculated based on DEM processing and a land use map. However, parameter β is catchment specific and is usually calibrated using measured sediment yield. In this study, β was calibrated using sediment data from a river gauging station located at the outlet of a sub-watershed (Figure 1). The best-fit value of β (which yielded the lowest error in relation to the observed SSY) was then applied to the SEDD equations in the whole Upper Grande River Basin. Although the assumption that β is constant throughout the basin may be questionable (Porto and Walling, 2015), a series of sediment yield measurements within sub-catchments, which are not available at present, would be necessary to establish independent *β* values.

RESULTS AND DISCUSSION

RUSLE predictions of average annual soil losses for the Upper Grande River Basin were of 22.35 t ha^{-1} yr^{-1} (Figure 2). Rangelands, the main land use in the study area, presented average soil losses of 16.63 t ha^{-1} yr^{-1}. Many pastures found in the basin are degraded, and therefore, may experience greater erosion than well-managed rangelands. Also, during the beginning of the rainy season, pastures are usually sparsely vegetated as a result of overgrazing and the lack of rainfall during the winter. Therefore, the single C factor value appointed to such land use might not represent the spatial and temporal variability of the parameter, which associates uncertainty to the model predictions.

The average soil losses for eucalypt and agriculture were of 65.93 and 57.29 t ha^{-1} yr^{-1}, respectively, in spite of the greater C factor assigned to croplands. In the Upper Grande River Basin, agricultural areas are mainly located where less erodible soils occur; that means 59% of the croplands were found on Latosols, whereas eucalypt was mainly associated with Haplic Cambisols. Also, agriculture tends to be established on smooth landscapes, more suited to mechanization. Mean values of the LS factor for croplands were 17% lower than those of eucalypt forests.

The calibration of the β coefficient for the SEDD equation indicated that the model was sensitive to the parameter. Average modeled SSY_i for the Mortes River sub-basin varied 58% as β ranged from 1.0 to 4.0 h^{-1}. By setting the β parameter to 3.0 h^{-1}, SEDD predictions yielded a mean SSY_i value of 1.58 t ha^{-1} yr^{-1}, which resulted in an error of 0.01 t ha^{-1} yr^{-1}, or 0.6%. It is important to highlight that the β parameter may be a source of great uncertainty in the SEDD model. The parameter highly increases the user's degree of freedom, and the model is able to accommodate a wide range of results during calibration of β. Uncertainty and sensitivity analysis should be employed to verify the model's prediction capacity. However, the lack of proper validation data in the study area hampered such investigation

The mean value of SDR_i for the study area was of 0.07. The spatial distribution of SDR_i indicates that the sediment sources closer to the stream channels have higher probability of delivering eroded particles to the water courses (Figure 2a). However predicable this might look, flow velocity, which depends on slope gradient and surface roughness, is used in the SEDD model as a proxy for overland flow transport capacity. Therefore, wherever flow velocity is high, eroded particles have a greater possibility of being transported to the stream network as opposed to being deposited along hillslopes. The average SSY_i in the Upper Grande River was of 1.93 t ha^{-1} yr^{-1} (Figure 2b). The distance to streams did not influence SSY_i values as much as it influenced SDR_i. The zoom-in data frames in Figures 7c and 7d demonstrate how SSY_i varies within a close distance to the stream channel.

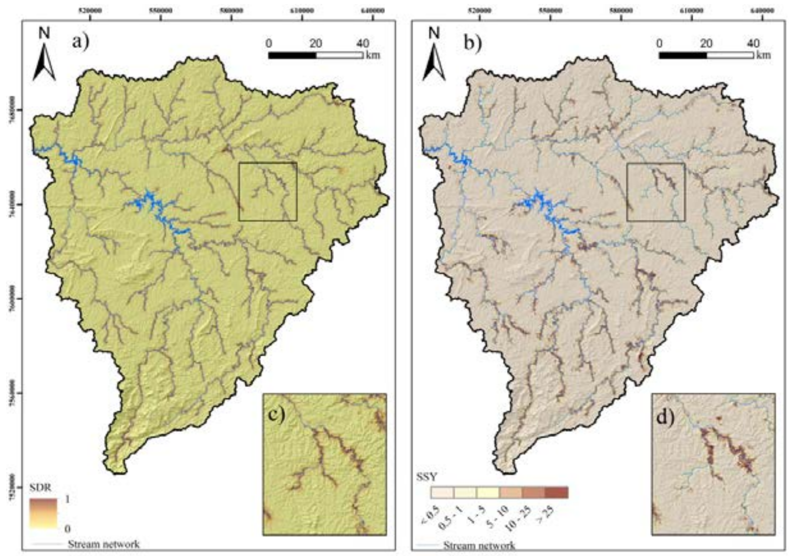

Figure 2 (a) (b) Sediment delivery rate and (c) (d) specific sediment yield (t ha^{-1} yr^{-1}) in the Upper Grande River Basin.

Although croplands comprise only 5.5% of the study area, such land use generated 25% of the total SY in the basin. Eucalypt forests also showed expressive SSY_i values, which were only lower than the ones for bare soils and agriculture. Rangeland and forests which, combined, occupy 85% of the Upper Grande River Basin account for only 46% of the total SY. Therefore, according to the model predictions, sediment production in the study area is highly influenced by intensive land use.

CONCLUSIONS

In the Upper Grande River Basin, bare soils, eucalypt forests and agriculture presented the highest soil losses among the identified land cover classes, according to the RUSLE predictions.

Bare soils, agriculture and eucalypt presented the highest area-specific sediment yield values. Such land uses generate a great amount of sediment within relatively small areas. Hence, in order to reduce the off-site erosion impacts in the basin, soil management support practices on croplands and eucalypt forests should be widely encouraged.

These results provided by this study are an initial estimation of the erosion and sediment delivery dynamics in the Upper Grande River Basin. Field data must be gathered in order to verify the quantity and the sources of sediments that reach the water courses. A model may accurately predict the sediment yield from a river basin without correctly identifying the spatial source, especially when calibration from observed data is employed.

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7.2.P

BANK EROSION AGGRAVATED BY MEANDER CUT-OFF AT THE ALPINE MEADOW IN THE SOURCE REGION OF YELLOW RIVER

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ABSTRACT

Meandering river is a most common river pattern on the earth. Most of the rivers developed at the alpine meadow in the source region of Yellow River are meandering rivers. Meander evolution and cut-off has been the major reason for soil erosion of grassland on the alpine meadow. A large neck cutoff was investigated and measured on the Baihe River, a large tributary of the upper Yellow River. Two erosion stages were detected in the meander evolution: the one was continuous bank erosion caused by secondary flow, and the other was the slump block failure caused by cutoff. Large amount of bank soil eroded by cutoff was washed away from the neck of meander. In addition, bank erosion propagated along both upstream and downstream from the cutoff and caused more slump blocks collapsed at the nearby and neighbor segments. Enormous slump blocks were measured to illustrate how bank failure propagated.

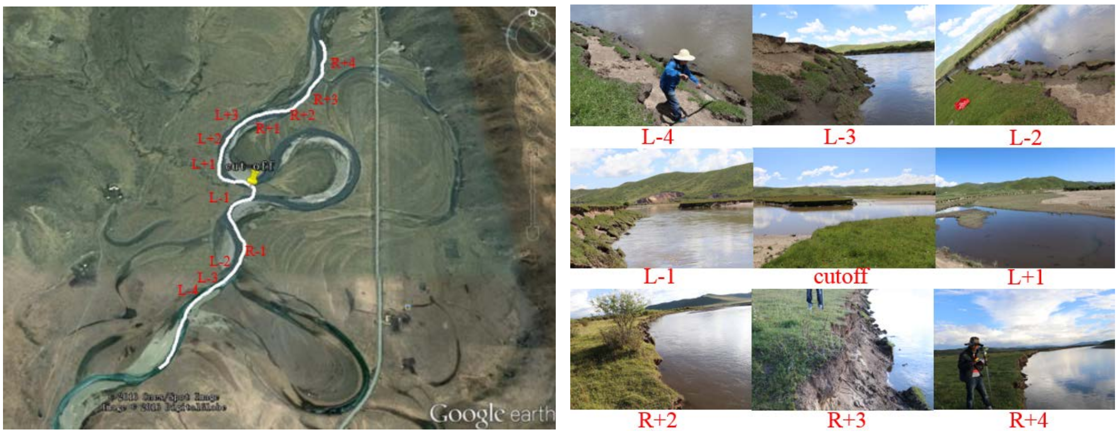
Keywords: meandering river, alpine meadow, meander cutoff, bank erosion, slump block

1. INTRODUCTION

Meandering river is a most common river pattern on the earth. Most of the rivers developed at the alpine meadow in the source region of Yellow River are meandering rivers. Rather than common slope erosion or gully erosion, meander evolution and cut-off has been the major reason for soil erosion of grassland on the alpine meadow. Meander cutoff usually caused a higher (1-5 magnitude) sediment erosion rate than lateral migration of individual bend (Zinger, 2011). Morphological analysis indicated that neck cutoff of meander occurred when the neck-river width ratio was larger than 1.5 or sinuosity was larger than 2.2 (Howard, 1992). New channel broadened and abundant sediment was released to form point bar and sandbar after neck cutoff occurred. Simulation of long-term meandering evolution and dynamic morphology analysis has been widely conducted since recent years ago (Camporeale, 2008; Asahi, 2013). However, extreme pulse of sediment erosion associated with meander cutoff and short-term bank erosion at the nearby river segments of the cutoff has not yet been studied.

2.DATA ANALYSIS AND RESULTS

An investigation was carried out at a large neck cutoff on the Baihe River, a large tributary of the upper Yellow River in July 2016, about two years after the investigated cutoff occurrence. Twelve river segments were selected for measurement (Fig. 1): 454 slump blocks were detected and measured in the selected segments (Table 1).



(Note: L, R mean left and right bank; number indicated segments sequence ordered by increasing distance from the cutoff; -, + mean upstream and downstream; labels were connected with Table 1

Fig. 1 Neck cut-off of meanders and slump blocks caused by meander cutoff of the Baihe River, tributary of the upper Yellow River (map from google, 2013; photos taken by the authors in 2016).

The location, size, area of each block was indicated in Table 1. The collapse year of each block was estimated through age recognition of plants developed on the block, or bend of erect stem of annual herb on new slump blocks. Bank erosion rate was indicated by the speed of bank retreat rate/migration rate.

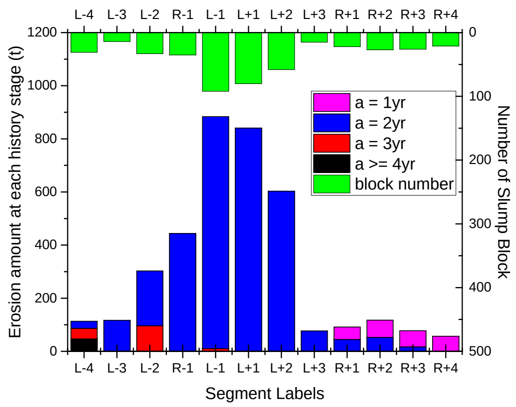
Table 1. Statistics of slump blocks at the twelve selected bank segments

Label	**Distance (m)**	**Length (m)**	**Block Number**	**Total area (m^2)**	**Area (m^2) for different erosion period (yr)**				**Bank Erosion (t)**	**Migration rate (m/yr)**
					a>=4	**a=3**	**a=2**	**a=1**		
L-4	-1602.73	78.00	31	25.21	10.41	8.85	5.95	0	113.45	0.08
L-3	-1194.47	15.00	14	21.3	0	0	20.25	0	122.69	1.42
L-2	-1059.19	46.00	33	49.64	0	15.29	32.74	0	312.73	0.54
R-1	-931.50	108.00	35	78.265	0	0	78.265	0	443.76	0.72
L-1	-77.43	113.95	92	175.34	0	2	173.34	0	883.71	1.54
L+1	324.58	175.55	80	166.42	0	0	166.75	0	838.76	0.95
L+2	502.13	112.94	58	119.59	0	0	119.59	0	602.73	1.06

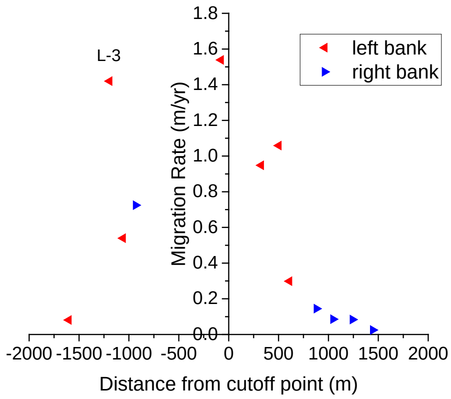
Label	Distance (m)	Length (m)	Block Number	Total area (m^2)	Area (m^2) for different erosion period (yr)				Bank Erosion (t)	Migration rate (m/yr)
					a>=4	a=3	a=2	a=1		
L+3	607.97	68.34	15	20.4	0	0	20.4	0	77.11	0.30
R+1	880.60	140.90	22	20.39	0	0	9.96	10.43	91.76	0.14
R+2	1046.95	181.11	27	31.03	0	0	13.88	17.15	117.29	0.09
R+3	1241.67	135.84	26	22.73	0	0	5.07	18.26	75.69	0.08
R+4	1445.18	271.19	21	13.42	0	0	0	13.42	56.77	0.02

Note: The width of cutoff section was 45m; Downstream was positive direction of distance; “Length” was the length of each bank erosion segment. Total blocks area and its distribution of different slump age stage was both listed in the table, and “a=2” mean that blocks slumped at the same year with meander cutoff; Bank erosion = Density*Total Area*Thickness, density of soil and distribution of layer thickness was also measured; Migration rate = Total Area/Length/erosion duration.

In accordance with data in Table 1, it was shown in Fig. 2 that slump blocks varied at different segments, and the migration rate varied along the river. Bank erosion caused by cutoff had different spatial characteristics. It was shown in Fig. 2(a) that total erosion quantity and the total number of slump blocks decreased with the distance from the cutoff. It was shown in Fig. 2(b) that retreat rate of bank also behaved a progressive decline trend as distance from the cutoff point increased for both the upstream and downstream. The erosion and migration at the left bank was much more intensive than the right bank: the migration rate of the left bank was 0.84m/yr on average, while it was 0.21m/yr for the right bank. And the erosion quantity and migration rate decreased slower in the upstream than in the downstream: from 1.54 to 0.08 m/yr in the 1.6 km long region of the upstream, while from 0.95 to 0.09 m/yr in the 1.4 km long region of the downstream.



(a) Erosion amount and slump block number at each segment;



(b) Migration rate along the river
Fig. 2 Statistics of bank erosion and slump blocks caused by neck cutoff
(Labels and data are connected with Fig.1 and Table 1; and distance=0 is at the meander cutoff in Fig. 2(b).)

In addition, bank erosion also showed temporal variation among segments. As shown in Fig. 2(a), the nearby segments (L-1, L+1~L+3 and R-1) were eroded only contemporaneous as the meander cutoff occurred. The further upstream segments (L-2~L-4) were also affected by another older cutoff, indicated by the slump blocks aged more than 2-year old in the segment L-4. While some bank failure at the lower reach segments (R+1~R+4) fell behind meander cutoff, indicated by 1-year old slump blocks. The response lag increased downstream, indicated by increase of the percentage of 1-year old blocks.

3. CONCLUSIONS

According to the field investigation of the meandering river on the alpine meadow, it was found that in addition to mass loss directly swashed away from the neck of meander, bank erosion propagated along both upstream and downstream from the cutoff. Enormous slump blocks were eroded from the vegetated bank owing to the propagated bank failure associated with the cutoff. Once occurred, meander cutoff immediately caused collapse of slump blocks in the nearby segments of the cutoff point and severe retrogressive erosion at the upstream neighbor bend. Meanwhile, bank erosion propagated downstream step by step until it was far away (**more than 30 times of the neck**) from the cutoff. The erosion and migration at the left bank was much more intensive than the right bank. And the erosion quantity and migration rate decreased slower in the upstream than in the downstream.

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Session VIII: Salinization and contamination of soils and waters

8.1.O

FOREST FIRES AS A SOIL CONTAMINATION DRIVER.

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1. INTRODUCTION

According to the International Panel on Climate Change, there is higher degree of confidence that meteorological conditions associated to climate change will be propitious to increasing extreme events (IPCC, 2014). Impacts on land degradation will be shown in bigger and more frequent wildfires. One aspect of fire-induced ecosystem degradation that has drawn scientific interest recently is the chemical pollution related to forest fires. This contamination is associated either to the use of chemicals to their extinction known as flame retardants, FRs (Pepper et al., 2011), or because incomplete combustion of vegetation favours the formation of polycyclic aromatic hydrocarbons, PAHs (Yuan et al., 2008). Brominated flame retardants account for a large group of FRs used for firefighting. Polybrominated diphenyl ethers (PBDEs) are the second highest production group of BFRs, and their presence has been reported in sediment and soil (Akortia et al., 2017). Some researchers have already studied the PAH production in relation to forest fires finding a significant increase in soil, but not so much as to reach harmful levels (Choi, 2014).

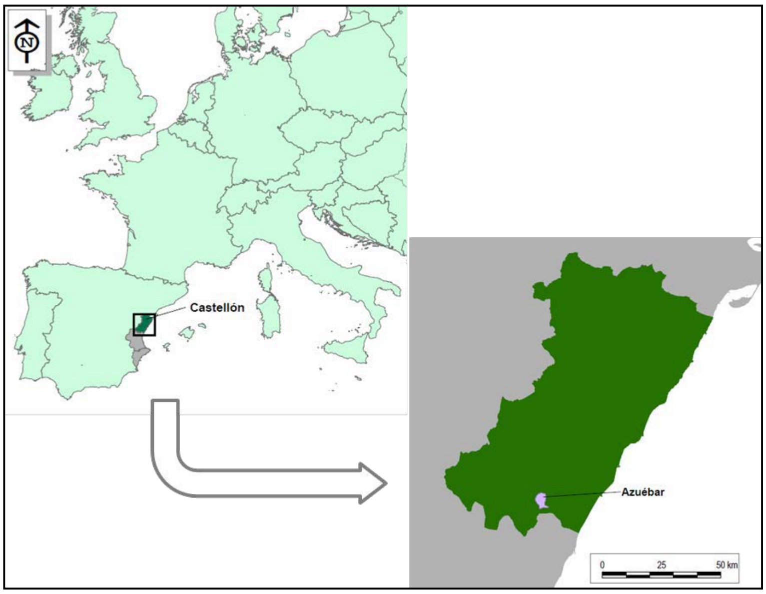
It has been found that compounds such as BFRs and PAHs are persistent, bio-accumulative and/or toxic to wildlife and humans, as well as potential endocrine disruptors (Eulaers et al., 2014). Consequently, the production and use of the most common PBDEs, penta- octa- and deca-BDE, is nowadays highly restricted after regulations that were introduced by USA and EU, and have been included as Persistent Organic Pollutants in the Stockholm Convention (UNEP, 2010). The hypotheses of the study are that the fire occurred, in 2014, in Azuébar, Castellón (Spain) added significant amounts of emerging contaminants to the soil, and that these concentrations are a function of the position on the hillslope, the soil depth and the presence of vegetation. It is also hypothesized that contaminants are transported downslope with increasing levels in the eroded sediment.

2. MATERIAL AND METHODS

2.1. Study site and sampling

This work was carried out in the municipality of Azuébar, Natural Park of Sierra de Espadán, in the Province of Castellón, Spain (Fig. 1). Coupled hillslopes (burned: BU, and control: CO) were selected (BU: 39°50'45.11"N, 0°22'20.52"W; CO: 39°51'08.7"N, 0°22'17.6"W). Both slopes are located on forested concave hillsides, with ENE aspect, 25-28° of slope and an altitude of 370 m a.s.l.. The climate is meso-Mediterranean. Vegetation cover is characterized by a Mediterranean shrubland developed after recurrent wildfires. Soils are classified as Luvisol Chromic Skeletic (FAO UNESCO, 2006). Last wildfire on Azuébar occurred on 28/8/2014 and burned 10.59 ha of forested area. Sampling was done on 19/09/2014: top of the hillslope (eroding), middle part (transport) and foot's slope (depositional). Two environments (under canopy soil: UC; inter-plants or bare soil: BS), and two depths (TS: 0-2 cm and SS: 2-5 cm) were considered. Samples (BU: n= 12; CO: n=12) were dried at room temperature and sieved to < 2 mm. Sediments were collected from four sediment fences at the foot of the burned slope.

Fig. 1. Location of the study area.



***2.2.** Sample extraction and determination*

All soil and sediment samples were analysed for their concentrations of some tetra-, penta-, hexa-, hepta-BDE congeners and of the 16 PAHs on the EPA's priority list. Extraction was done by pressurized liquid extraction using an ASE (Dionex ASE 200, Sunnywale, CA, USA). Sample (2 g) was weighed into extraction cells with cellulose filters and filled off with granulated silica. PBDE and PAH internal standards (50 µl) were added before extraction. The ASE was done using acetone/hexane (1:1) as solvent.

The resulting extracts were concentrated to 0.5 ml. The extracts were run through a clean-up column packed with 1 cm of folded glass wool at the bottom, 6.75 cm hydrated Al_2O_3 (11% water) and 1 cm granulated Na_2SO_4, and eluted with 25 ml hexane. After, the extracts were concentrated a second time with a Vigreux column to 1.0 ml. Finally, 100 µl of extracts were pipetted into a vial for GC-MS. The final analysis was done with a Thermo-Quest Trace GC 2000 gas chromatograph (Thermo Fischer Milan, 176 Italy), and the products were separated by a fused silica column (J&W, 60 m × 0.32 mm i.d.) coated with DB-5 (film thickness 0.50 µm) and Helium was used as carrier gas. Sample solutions (2 µl) was injected at 60°C, with a temperature programme reaching up to 320°C. The column was coupled to a Finnigan Trace mass spectrometer (MS). Target compounds were identified using Xcalibur Software. Quantification was done with internal standard methodology based on peak areas.

3. RESULTS AND DISCUSSION

The hepta- and hexa-BDEs 153, 154 and 183 were not found in any of the soil samples. Concentrations of the detected tri- to penta-BDEs were in general low (the highest value was for PBDE-85: 5.6 ng g^{-1} d.w.). PBDEs were observed in both, BU and CO soils with sums of the PBDE values ranging between 0.5 ng g^{-1} d.w. (CO) and 7.3 ng g^{-1} d.w. (BU) (Fig. 2). The occurrence of PBDEs on the CO might be related to atmospheric transport and deposition as stated by Eljarrat et al. (2008). The most frequent compound was PBDE-47, which was found in all the burned samples and in most of the control ones. Only the PBDE-85 showed significant differences between BU and CO. This might implicate that PBDE-85 could have been used in fire extinguisher. However, according to La Guardia et al. (2006) PBDE-85 accounts for only 2-3% of the most widely produced penta-BDE mixtures and it is questionable whether a penta-BDE mixture was present in the fire extinguisher applied to the studied soil. None of the variables, slope position, vegetation and depth, have a significant influence on PBDEs levels individually, however decreasing trends were observed from transport, to erosion and finally to deposition zones. Similarly, in TS compared to SS, and in UC compared to BS.

In general, all PAHs were found in BU and CO soils. Concentrations ranged from 0.2 ng g^{-1} d.w. to 803 ng g^{-1} d.w. Figure 3 shows the total concentration of 16 PAHs in soil samples with values between 133.5

and 1255.3 ng g^{-1} d.w., which are in agreement with the ones reported by Choi (2014). The proportion of light PAHs (2 and 3 aromatic rings) was higher in BU than in CO, indicating that these were the main ones produced by the fire (∑16 PAHs BU>CO). The heavy PAHs (4, 5 and 6 rings) were dominant in CO and in both treatments, differences based on the depth were detected (TS>SS).

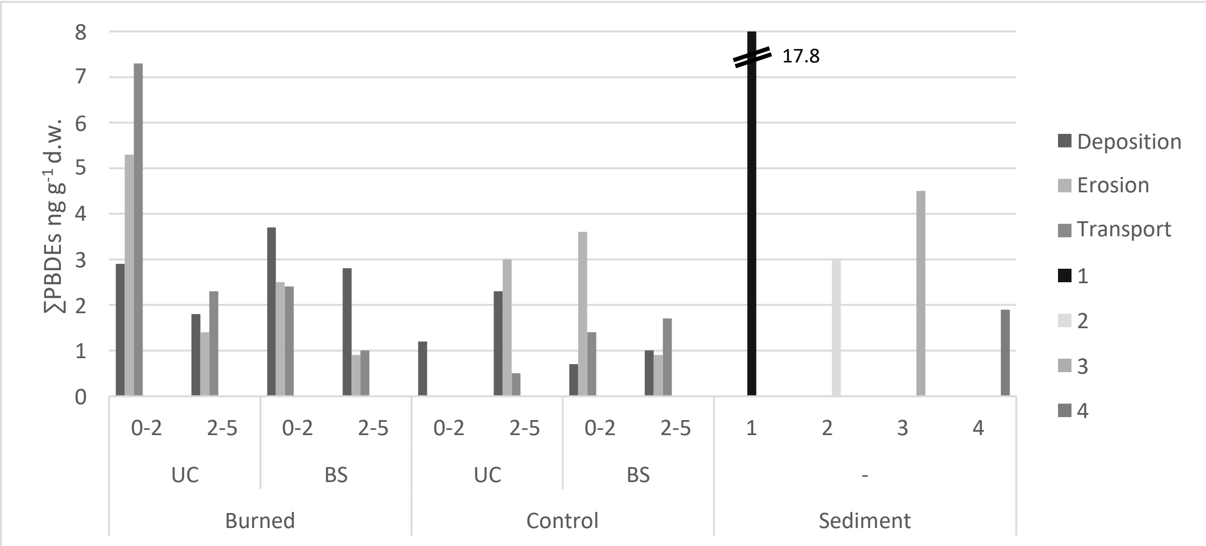


Fig 2. ∑PBDEs in soil and sediment samples.

Similar results were also presented by Kim et al. (2003) who found a range of 150-1600 ng g^{-1} in soils after several forest fires in the eastern coastal region of Korea, but higher than the findings of Vergnoux et al. (2011) in repeatedly burned sites of the South of France (Σ14 PAHs=77-157 ng g^{-1} d.w.), as well as the ones of Pizarro-Tobías et al. (2015) who measured up to 400 ng g^{-1} for Σ15 PAHs after controlled fires in the Parque Natural de los Montes de Málaga (Spain). PAH patterns in burned soils are remarkably constant across studies and very similar to the one described here.

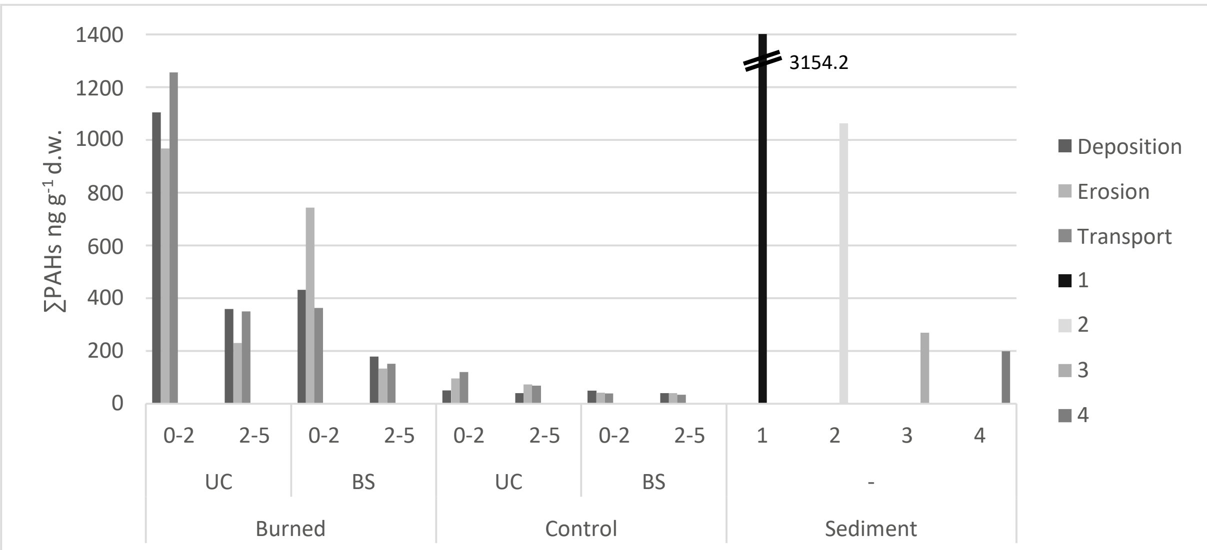


Fig 3. ∑PAHs in soil and sediment samples.

The position did not have a significant influence on the distribution of PAHs along the hillslope. However, a trend to accumulate in the middle of the hillslope was observed in both treatments. Despite significant differences were not found between TS and SS, there must be some downwards movement during or shortly after fire, because PAH levels increased at both depths.

Four erosive rain events were considered in this study. Based on the data of the closest pluviometer (Sot de Ferrer: 4.5 km), these were registered in 29/11/2014, 23/3/2015, 15-16/6/2015 and 2/11/2015, and produced 12.7, 143.6, 12.6 and 62.2 kg of sediment, respectively. These events did not produce any sediment in the CO hillslope. As it was expected, PBDE-153, -154 and -183 were not found in the sediment samples. Concentrations of the tri- to penta-BDEs were also low, but PBDE-85 presented the highest value detected (11.4 ng g^{-1} d.w.). Only the PBDE-47 was observed in all the sediment samples. The sum of the PBDE values was high in the sediment of the first erosive event (17.8 ng g^{-1} d.w., Fig. 2), being higher than in soils. Similarly, the PAH concentrations in the sediments from the first event (∑16 PAHs=3154.2 ng g^{-1} d.w., Fig. 3) were higher than those found in soil. Concentrations also decreased in the sediments of the following events. PAHs distribution in the first sediment was similar to the one in soil suggesting that erosion is important for contaminants transport.

4. CONCLUSIONS

Concentrations of PBDEs were low on the burned hillslope and most of the concentrations found can probably be related to atmospheric deposition, because these relatively low concentrations were also found on the control site. PBDEs seemed to be more abundant in the top layer of the soil. There was no clear pattern of PBDE concentrations over the different slope positions, highest concentrations were found on the erosion and transport zones. Although highest concentrations were found under canopy, there is no significant difference between concentrations under canopy and on bare soil. On the other hand, the fire in Azuébar actually added substantial amounts of PAHs to the soil. Mainly the upper soil layer was affected, and soil under vegetation generally had higher concentrations. PAHs tended to move downwards into the soil, possibly in gaseous form. The sum of concentrations of both contaminants in sediment of the first rainfall was much higher than in any soil sample. Erosion seems to be an important factor in the dynamics of PBDEs and PAHs transport on burned slopes. Levels of contamination could therefore be higher in places where sediment accumulates after erosive events. The concentrations of the different compounds were low and therefore the contamination from the studied fire is unlikely to eventually harm ecosystems. However, PBDEs and PAHs might accumulate in the soil leading to bioaccumulation and potentially hazardous levels in higher trophic levels. Such processes have to be considered when the effects of fire-related contaminants are further explored.

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8.2.O

KINETIC AND EQUILIBRIUM MODELLING OF METHYL TERT-BUTYL ETHER (MTBE) ON ZSM-5

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ABSTRACT
The intensive use of MTBE as a gasoline additive has resulted in serious groundwater and aquifer contamination due to its high solubility, volatility and recalcitrance, which has been receiving ever-increasing attention. In this study, batch adsorption and desorption tests were conducted to explore the detailed adsorption mechanism and the effect of pH on the MTBE adsorption onto ZSM-5. Results showed that the adsorption of MTBE onto ZSM-5 follows the Langmuir model and obeys the Pseudo second order model. Approximately 24 h is required for the adsorption process to reach equilibrium. Boyd film diffusion model and Weber and Morris intra-particle diffusion model indicated that the adsorption was mainly controlled by intra-particle diffusion for all MTBE initial concentrations. The PZC of ZSM-5 is 5.5 and initial solution pH has little impact on the MTBE adsorption on ZSM-5.

1. INTRODUCTION

MTBE, a widely used gasoline additive, can easily produce significant contamination in groundwater and aquifers and consequently be accessible to the public as a health risk due to its high solubility, volatility and recalcitrance. Since MTBE is resistant to biodegradation and chemical oxidation (Johnson et al., 2000), physical adsorption has proved to be one of the most efficient approaches for the removal of MTBE (Sacchetto et al., 2013). A few studies have revealed that ZSM-5, a high-silica MFI type zeolite, is an extremely effective adsorbent for MTBE due to its hydrophobicity and suitable pore size (Levchuk et al., 2014). However, most studies focused on the relationship between the properties of ZSM-5 and its adsorption capacity of MTBE (Gonzalez-Olmos et al., 2013), while there is a lack of studies on the detailed mass transfer mechanisms.

In this study, the adsorption equilibrium and kinetics that describe the detailed mass transfer mechanisms and the rate limiting step of the MTBE adsorption on ZSM-5 were determined. The effect of pH on the adsorption was also investigated.

2. METHODS

MTBE and ZSM-5 were purchased from Fisher scientific and Acros Organics, respectively. The specific surface area and particle size of ZSM-5 were 400 m^2/g and 2-8 μm, respectively, and the SiO_2/Al_2O_3 ratio was 469. The cation exchange capacity is determined to be 1.81 cmol/kg. All tests in this study were conducted in duplicate in a temperature-controlled lab (20±1 °C). Batch kinetic and isothermal studies were carried out by adding 0.1 g of ZSM-5 into 20 mL solution with different MTBE concentrations. After being agitated for a pre-determined time, the mixtures were filtered and the concentrations of MTBE were determined by GC-FID with headspace technique. The effect of initial solution pH on the MTBE adsorption was also evaluated. After the batch adsorption experiments, the filtrate was decanted and 20 mL deionized water was added to conduct desorption kinetic tests, the MTBE concentration was determined at certain time intervals.

Pseudo second order model was used to fit the kinetic data. Where q_t and q_e are the MTBE uptake ($mg \cdot g^{-1}$) at time t and equilibrium respectively, k ($mg \cdot g^{-1} \cdot h^{-1}$) is the rate constants.

$$q_t=q_e^2kt/(1+q_ekt)$$

The Boyd film diffusion model was expressed as follows (Boyd et al., 1947).

$$B^*t=\ln(6/\pi^2)-\ln(1-q_t/q_e) \qquad B=\pi^2D_f/r^2$$

The slope obtained from B*t vs t plot was used to calculate the film diffusion coefficient, D_f (cm^2/s), where r is the radius of absorbent particle in cm, assuming spherical particles with an average radius. Weber and Morris intra-particle diffusion model was used to describe the possibility of intra-particle diffusion (Furusawa and Smith, 1974).

$q_t=k_{in}t^{1/2}+c$ . . . .

Where k_{in} ($mg\cdot g^{-1}\cdot h^{-0.5}$) is the diffusion rate constant and c gives the information about the thickness of boundary layer.

3. RESULTS AND DISCUSSION

3.1 *Adsorption kinetics and isotherm*

As shown in Fig. 1a, the MTBE adsorption onto ZSM-5 followed the pseudo second order model, suggesting a mechanism of chemisorption. The half-adsorption time of MTBE, $t_{1/2}$, increased with the increase of MTBE concentration. The desorption reached equilibrium within 24 h at the MTBE concentration of 300 mg/L and only 2 % was desorbed, indicating that the adsorption bond strength was very strong between the MTBE molecule and ZSM-5. Therefore, ZSM-5 can be regarded as a suitable adsorbent due to its high sorption capacity and affinity for MTBE.

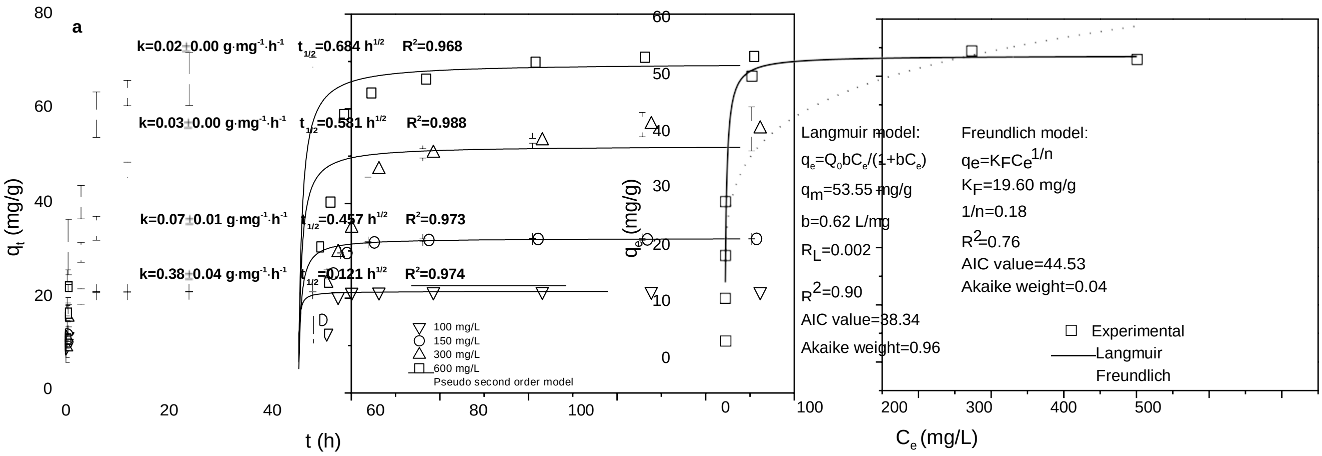

Figure 1 Adsorption kinetics (a) and isotherm (b) plots for MTBE adsorption onto ZSM-5

The experimental data was fit by Langmuir and Frendlich isotherm models. The MTBE adsorption follows Langmuir model (R^2=0.90) better compared with Frendlich model (R^2=0.76), indicating a monolayer and homogeneous adsorption process. In addition, The Langmuir model was 22.11 times more likely to be correct to describe the equilibrium data compared with the Freundlich model according to the AIC test. The R_L value (0.002) showed that the adsorption process is favourable, and the maximum adsorption capacity is 53.55 mg/g. Therefore, ZSM-5 had a strong affinity for the adsorption of MTBE and could be employed as an effective adsorbent for MTBE.

3.2 *Mass transfer mechanism*

Some researchers have studied the mechanism for dyes and metals adsorption on absorbents, and found that the adsorption process involves four steps (Weber, 1984): transport from bulk solution to the boundary layer, film diffusion (boundary layer), intra-particle (pore and surface) diffusion and adsorption on the interior surface of adsorbent. There is a consensus that the first and last steps are very fast and the overall adsorption process is controlled by film diffusion and/or intra-particle diffusion (Mahdavi and Amini, 2016).

To explore the film diffusion, the calculated B*t values in the initial period (0-12 h) of adsorption were plotted against t in Figure 2. The B*t vs t plots passed through the origin, indicating that the intra-particle diffusion took a part in rate controlling process for all initial MTBE concentrations

(Kalavathy et al., 2005) in this study. The values of D_f were calculated and listed in Table 1 and varied with the initial MTBE concentrations.

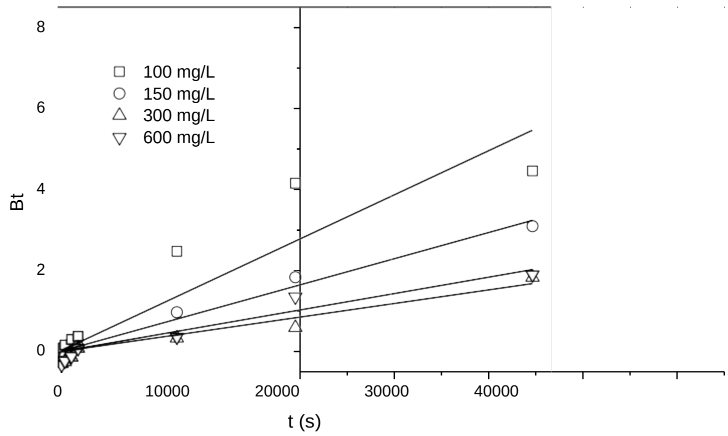

Figure 2 Boyd plots for MTBE adsorption on ZSM-5 at different initial concentrations

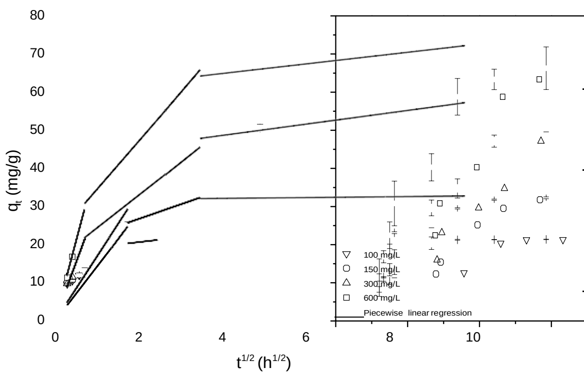

Figure 3 Intra-particle diffusion plot for the MTBE adsorption onto ZSM-5 at initial MTBE concentration of 100, 150, 300 and 600 mg/L, the lines represent the intra-particle diffusion periods

Table 1 Multilinear regression diffusion parameters for MTBE onto ZSM-5

	100 mg/L	150 mg/L	300 mg/L	600 mg/L
$K_{in,1}$	14.23 ±2.36	16.88±2.14	30.56±1.44	41.45±1.21
R_1^2	0.876	0.924	0.991	0.997
$K_{in,2}$	1.23	3.69±1.05	8.61±1.09	12.69±2.55
R_2^2	-	0.849	0.954	0.888
$K_{in,3}$	0.02±0.01	0.10±0.04	1.44±0.25	1.25±0.22
R_3^2	0.641	0.605	0.900	0.886
$D_f \times 10^{13}$	8.03	4.76	2.47	2.98

The intra-particle diffusion plot of MTBE on ZSM-5 described by Weber and Morris model was presented in Figure 3. There were three periods included. First, the ZSM-5 particles were surrounded by a boundary layer and MTBE molecules had to overcome the boundary resistance (McKay et al. 1985) in this stage. It was found that all the curved plots covering this initial phase passed through the origin. That is, the intercept, which means the thickness of the boundary layer, was close to zero, suggesting that the film diffusion was very fast in the early stage (Tütem et al., 1998) and intra-particle diffusion should be the rate-controlling step in the removal of the adsorbate. This was consistent with the conclusions from Boyd film diffusion model. In addition, as shown in Table 1, the $k_{in,1}$ values increased with increasing MTBE concentration, which indicated that film diffusion became faster at higher initial MTBE concentration. This may be due to that the increasing surface loading increased the mass transfer driving force and consequently the rate of film diffusion. The second was the gradual adsorption stage where intra-particle diffusion happened. The slope of the this linear portion has been defined to yield the intra-particle diffusion parameter $K_{in,2}$. The third stage was the final equilibrium stage.

3.4 The effect of pH

Solution pH controls the electrostatic interactions between the adsorbent and adsorbate. Therefore, it determines the adsorbent surface charge and the dissociation or protonation of organic weak electrolytes (Moreno-Castilla, 2004). As shown in Figure 4, the PZC (point of zero charge) of ZSM-5 was around 5.5. This means that when pH values were below 5.5, the surface of ZSM-5 was protonated and positively charged, which was favourable for cation exchange. It was also shown that the removal percentage of MTBE onto ZSM-5 remained at ~90 % and was hardly affected by the change of initial solution pH. This may be due to that ZSM-5 in this study had little potential for the ion exchange considering its high SiO_2/Al_2O_3 ratio and low cation exchange capacity. In addition, MTBE is a weakly polar molecule and the electrostatic interactions between ZSM-5 and MTBE might be weak.

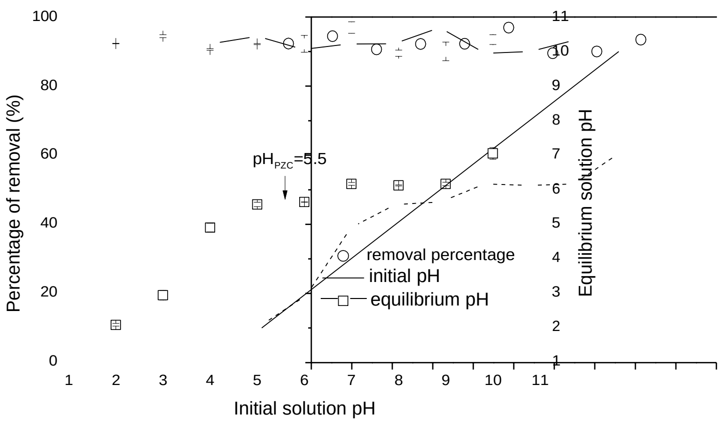

Figure 4 The influence of solution pH on the percentage of MTBE removal

4. CONCLUSIONS

Kinetic and isotherm studies indicate that ZSM-5 can be effectively employed for the adsorption of MTBE. The adsorption followed the Langmuir model and obeyed a Pseudo second order model, suggesting a monolayer and homogeneous chemisorption process. Both the adsorption and desorption processes reached equilibrium within 24 h. The adsorption process was mainly controlled by intra-particle diffusion, and film diffusion was very fast for all MTBE initial concentrations. Moreover, initial solution pH had little effect on the adsorption process.

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8.3.O

MAPPING EROSION AND SALINITY RISK CATEGORIES USING GIS AND THE RANGELAND HYDROLOGY EROSION MODEL

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INTRODUCTION

Up to fifteen percent of rangelands in the state of Utah in the United States are classified as being in severely eroding condition (Rasely et al., 1991). Some of these degraded lands are located on saline, erodible soils of the Mancos Shale formation, resulting in a disproportionate contribution of sediment, salinity, and selenium to the Colorado River which is a critical water resource for both the United States and Mexico. Prior work has documented a significant linear relationship between salinity and sediment in the runoff from experimental rainfall simulator plots on the Mancos Shale (Cadaret et al, 2016), and this demonstrates the potential for using erosion models to predict salinity loads under different conditions. Land management agencies in the U.S. are considering mitigation activities to reduce erosion on these saline rangelands, and an initial question is how to prioritize different locations for action. Such prioritization would benefit greatly from an efficient mapping strategy that could identify regions of elevated risk of erosion. It would be best if the estimate of risk used a physically based model of erosion processes, but full-scale implementation of an erosion model for a large watershed can consume a significant amount of time and resources. This paper presents a risk mapping method that classifies the landscape into a limited number of environmental types that can each be efficiently characterized with a plot-scale, process-based erosion model.

METHODS

This risk mapping effort used GIS datasets covering a sub-basin of the Upper Colorado River Basin. Four geospatial datasets were selected to characterize erosion risk:

1) slope length calculated from 30 m digital elevation model (DEM),
2) soil K factor from the U.S. Natural Resource Conservation Agency's (NRCS) SSURGO soils database,
3) enhanced vegetation index (EVI) from the satellite-based MODIS sensor, and
4) gridded average 15-minute precipitation rate for storms with a 10-year return frequency derived from the NOAA Atlas 14 database.

NRCS produces two soil map products, the more detailed SSURGO product that did not cover the entire sub-basin, and the much coarser STATSGO product that had complete coverage. For the purposed of developing this method we opted for the finer scale SSURGO product. Since multivariate analysis methods can be sensitive to the range of values in different input variables, each GIS variable was normalized to a common range by subtracting the mean and dividing by the standard deviation.

The four raster GIS datasets were composited together in order to run an unsupervised classification with the ISODATA routine in the ENVI image processing software package. The ISODATA algorithm iteratively clusters samples together or splits them apart based on user-specified parameters. A range of parameters were tested, with the goal of creating a manageable number classes that would express themselves as a coherent pattern on the landscape. With the normalized inputs, a maximum class standard deviation and minimum class distance of 1.0 resulted in six classes that divided the landscape in a reasonable manner.

In addition to assessing the spatial configuration of each mapped class, the distribution of the six classes within the measurement space of the four environmental variables was visualized using ENVI's n-Dimensional Visualizer (Figure 1). This allowed us to rotate the data space in order to understand the level of aggregation associated with each cluster and how the clusters related to each other. Based on this visualization, we learned that one cluster had encompassed a very large amount of variability in EVI. Given the importance of vegetation on erosion, we were able to manually split that cluster in two within the visualization environment and then propagate that edit into the map of classes.

A B

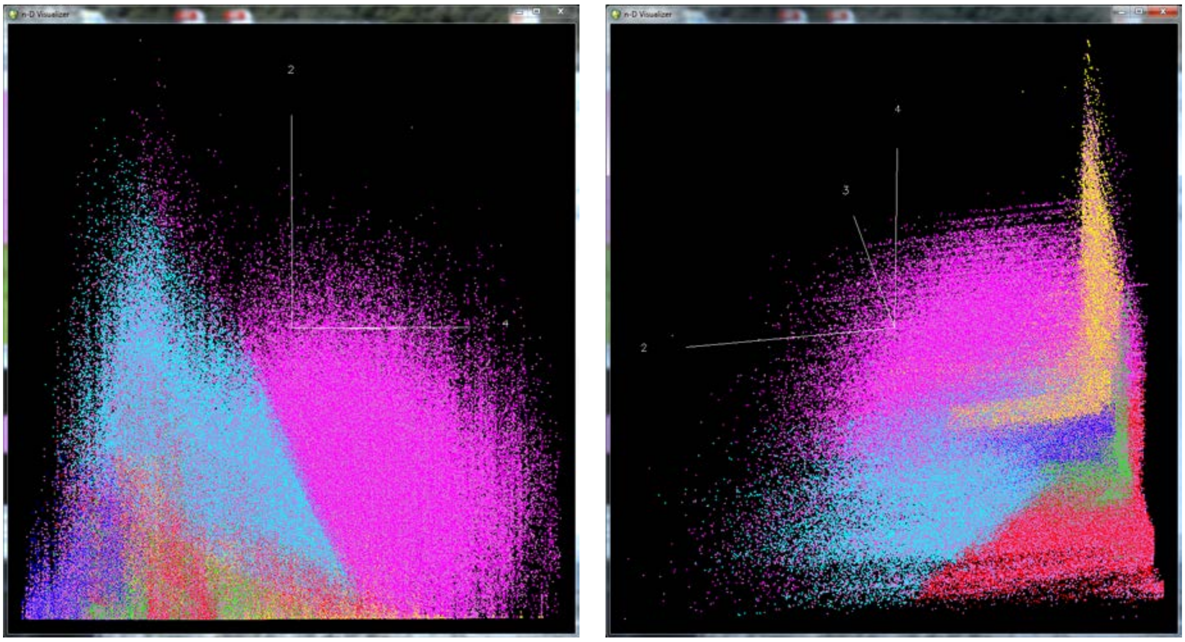

Figure 1: Interactive exploration of data space. The n-Dimensional Visualizer can rotate through mutliple combinations of variables (axis 2 = K factor; axis 3 = EVI; axis 4 = 15-min, 10-year rainfall).

Having defined seven environmental classes to characterize the predominant types of surface cover that would relate to erosion risk, the class map was intersected with a map of soil textures derived from the SSURGO product. The Rangeland Hydrology and Erosion Model (RHEM; Nearing et al., 2011) was parameterized for each class/texture using field data that had previously been collected by the NRCS National Resource Inventory Program. In the GIS, NRI survey locations within a radius of 100km were intersected with the four normalized environmental variables to determine which of the seven classes that they were most similar to, based on euclidean distance within the data space. A maximum euclidean distance was selected for each of the seven risk classes that resulted in approximately 10 NRI survey points (9 - 11) per class.

The RHEM model was run using data from each of the multiple NRI points within each combination of risk class and soil texture. The average erosion rate predicted from the multiple NRI points was calculated and assigned to each class/texture, resulting in a risk map for the sub-basin (Figure 2).

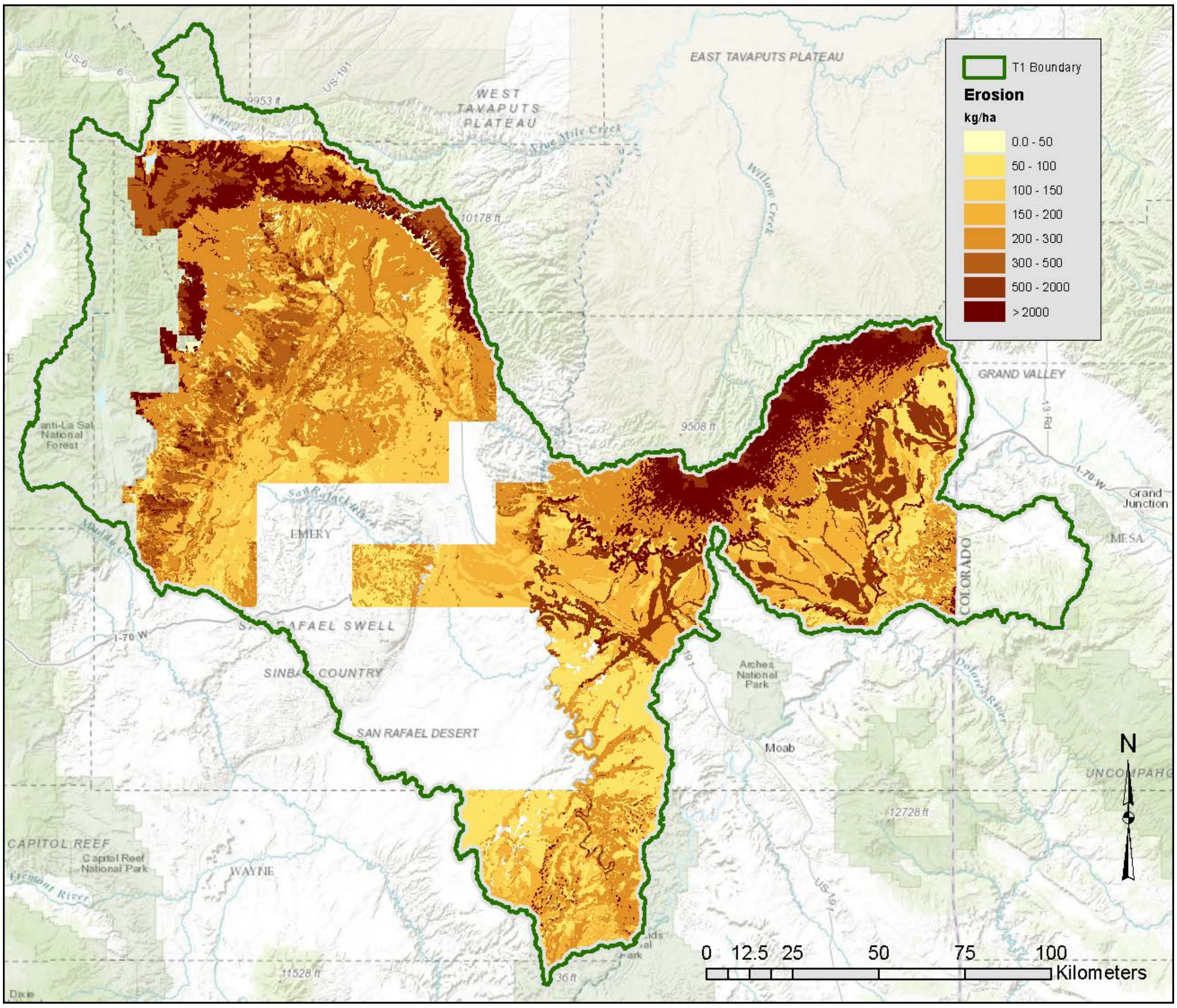

Figure 2: Risk based on plot-scale erosion estimates. Gaps are due to limited SSURGO coverage.

The RHEM model provides a plot-scale estimate of erosion risk, so the risk map in Figure 2 does not account for transport processes. However, it does provide a useful representation of which areas would be expected to be most in need of efforts to mitigate erosion. As seen in Figure 2, data availability can be a problem for this method, though a coarser map product still could have been developed from STATSGO. While the NRI data used here will not be generally available, the map of intersected risk classes and soil textures could be used to develop an efficient field data collection effort.

CONCLUSIONS

Classification of GIS variables with an unsupervised clustering routine like ISODATA provides a useful way of characterizing general patterns of erosion risk across the landscape. In addition to viewing the mapped results to ensure that the algorithm's parameters are appropriate, it is useful to visualize the distribution of the clustered classes within the data space. Having reduced the complexity of a large landscape to a limited number of classes, it is possible to estimate relative levels of risk using a plot-scale erosion model. The resulting risk map can help to ensure that planned mitigation activities target the most serious risks and are not overly biased by prior expectations.

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8.4.O

PHYOSTABILIZATION OF LEAD IN CONTAMINATED SOILS OF THE CENTRAL AREA OF CHILE BY ATRIPLEX HALIMUS

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To evaluate the phytostabilization and/or phytoextraction capacity of *Atriplex halimus*, chelating agents (CA, citric acid and FA, fulvic acids) and an organic amendment (OA) were added to a soil highly contaminated with lead (2,952±280 mg Pb kg^{-1} and pH 6.30±0.13), recollected in Valparaíso Region of Central Chile. The assays were established in pots and six treatments were evaluated: Control, CA, FA, OA, CA+OA and FA+OA. Chelating agents were applied to irrigation water at 4 g L^{-1}, while the OA was applied with a volumetric dose of 5%. After 180 days, A. *halimus* increased significantly the Pb concentrations with CA+OA reaching concentrations of 32, 42 y 360 mg Pb kg^{-1} in leaves, stems and roots, respectively. *A. halimus* with the FA application increased the total dry matter, while CA decreased the leaves dry matter. In general, the plant showed higher Pb concentration in roots than in the aerial part, presenting a high potential to phytostabilization of soils contaminated with Pb.

METHODS

Samples of soils were obtained from Puchuncavy Valley located at Valparaíso Region (273017 m E, 6386884 m S) in Central Chile. The origin of soil contamination with Pb is due to a mixture with construction residues that was used to level a slope. The assay was established during the months of December 2014 to May 2015 in the greenhouses sector of the Faculty of Agronomic Sciences of the University of Chile, in outdoor conditions and protected from rainfall. Soil analysis methodologies are described in Sadzawka et al. (2006). A sequential extraction of Pb in soils was carried out to estimate the bioavailability of the metal according to methodology based on Tessier (1979). The sequential extraction is divided into 5 fractions to which the metal is bound: 1) FI: soluble-exchangeable; 2) FII: acid-soluble carbonate associated; 3) FIII: Fe and Mn oxides; 4) FIV: organic matter and 5) FV: residual. As chelating agents, citric acid (CA, Sigma Aldrich) and fulvic acids (FA, Plus Fulvital®, HUMINTECH) were used and organic amendment (OA) a commercial substrate based on compost of pine bark, sawdust and algae residues (Acid soil, Roots®). The Atriplex genus (Chenopodiaceae), denominated saltbush, is one of the most important families of plants in the region of Antofagasta, Chile (Poblete et al., 1991). These shrubs xerohalophyte are dominant in many arid and semi-arid regions of the world, particularly in saline, arid soils, and they are used for rehabilitation of degraded lands (Lutts et al., 2004; Conesa et al., 2007), ornamental plants and revegetating sealed landfills (Ingelmo et al., 1998) and for animal feed. Certain assays have shown that plants of the genus *Atriplex* accumulate B, Cd, Mo and Se (Lutts et al., 2004; Tapia et al.,2011). *Atriplex halimus* is native of Mediterranean frequently encountered on marginal soils and degraded lands in southern Europe and North Africa (Lefèvre et al., 2009).

Fig.1. *Atriplex halimus* during in the assay in contaminated soil with lead.

The plants of *Atriplex halimus* were obtained from seeds and after 8 months the plants were transplanted in pots of 3 L. The irrigation was 250-300 mL twice a week. Six treatments were established: Control (No addition of chelating agents or amendment), CA (citric acid), FA (fulvic acid), OA (organic amendement), CA+OA (citric acid+organic amendement) and FA+OA (fulvic acid+organic amendement). Chelating agents were applied to irrigation water at 4 g L^{-1}, while the OA was applied at the beginning of the experiment with a volumetric dose of 5%. After 180 days, the dried and sieved root and aerial samples were digested with HNO_3 and H_2O_2 and the Pb was determined by atomic absorption spectrophotometry.

RESULTS AND DISCUSSION

The soil shows a high electrical conductivity, weakly acidic pH and high level of organic matter. Concerning the levels of N are high and the levels of P and K are within average range to adequate. The level of Pb, indicates high contamination that is established between the range 2000-10000 mg kg^{-1} (Kelly, 1979). Contaminant concentrations are also observed in Cr and Mn metals.

Table 1. Main chemical properties of soil collected in the Puchuncavy Valley, central Chile (values are mean ± standard deviation (n = 4).

$pH_{agua(1:2,5)}$	6.30 ± 0.13
Electrical conductivity$_{satured\ extract}$ dS m^{-1}	4.58 ± 0.53
Organic matter%	5.28 ± 0.05
$N_{available}$ mg kg^{-1}	55.2 ± 2.6
P_{Olsen} mg kg^{-1}	15.4 ± 0.4
$K_{available}$ mg kg^{-1}	0.59 ± 0.02
CEC $cmol_+$ kg^{-1}	18.0 ± 6.5
$_{total}$ Cu (mg kg^{-1})	117 ± 5.11
$_{total}$ Cr (mg kg^{-1})	297 ± 52
$_{tota}$Mn $_{l}$ (mg kg^{-1})	7275 ± 26
$_{total}$ Pb (mg kg^{-1})	2952± 279
$_{total}$ Zn (mg kg^{-1})	77.5 ± 5.00

The sequential extraction of Pb in the soil shows that Pb is associated with fractions II (associated with carbonates) and fraction III (associated with organic matter) (Fig. 2). When 1-10% of the metals present are associated with carbonate and with the exchangeable fraction, the bioavailability of these metals is low (Perin et al., 1985). Therefore, the Pb in this soil is of high bioavailability. Probably the fraction from which the Pb is uptake is associated with carbonates. The dissolution of carbonates occurs when water and CO_2 is present in the system, which may come from microbiological activity. The lower concentration of the metal in the residual fraction indicates that Pb is not an important part of the parent material. It is assumed that the Pb associated with the residual fraction is not available to the plants.

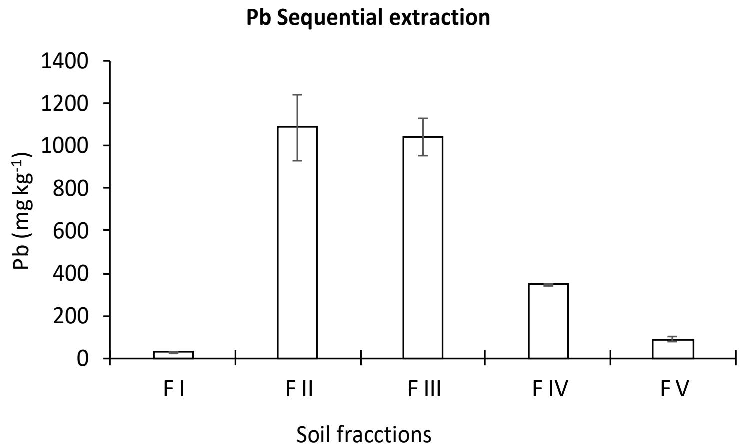

Fig.1. Distribution of Pb (mg kg^{-1}) in the different soil fractions FI: soluble-interchangeable FII: carbonates; FIII organic matter; FIV: Fe and Mn oxides and Fv: residual. Values are average (n = 4). The lines on the bars correspond to the standard deviation of the measurements.

The plants tolerated the high Pb concentration of these soils, and showed no symptoms of toxicity such as chlorosis or necrosis in young leaves. The concentration of Pb in the plants is shown in Table 2. The plant accumulates the Pb mainly in the root. The highest leaf Pb concentration was reached with CA + OA. Pb is one of the most immobile elements in soils, however, the continuous supply of citric acid together with the organic amendment, showed a chelating effect for this metal. In addition, it should be considered that Pb is associated with the fraction associated with carbonates that is available to plants. The concentration of Pb in leaves that reached *A. halimus* can be toxic for some plants, whose level is from 30 mg kg^{-1} (Kabata-Pendias, 2011). In other assays, lower levels of Pb (5.3 ± 0.9 mg kg-1) were found in *A. halimus* after 60 days of irrigation with citric acid in substrates, but with a lower total Pb concentration of 80.9 ± 6.5 mg kg^{-1} and mainly associated with organic matter (Tapia et al., 2013). Regarding growth, the plant generated significantly higher biomass with FA + OA (Table 3). Treatments that included fulvic acids (FA and FA + OA) stimulated growth, which is consistent with the function of humic substances. The dose to stimulate growth may range from 5-300 mg L^{-1} for humic acids and fulvic acids (Brady and Well, 2008). A significant decrease in dry weight was observed with the application of citric acid, which could be attributed to soil compaction, which affected root development. Other assays of application of citric acid to soil or substrates have also found compaction with the addition of citric acid that causes the dissolution of carbonates and thereby a change in physical and chemical properties (Lesage and Meers 2005; Tapia et al., 2013).

Table 2. Pb concentration (mg kg^{-1}) in tissues of *Atriplex halimus* after 180 days.

Tratamientos	Pb (mg kg^{-1})			Dry weight (g)		
	Leaves	Stems	Roots	Leaves	Stems	Roots
Control	17.2 ± 3.3b	12.5 ± 5.9b	63.5 ± 18.2b	3.9 ± 0.7a	3.6 ± 0.8a	2.1 ± 0.4a
CA	19.2 ± 4.2b	21.1 ± 4.1bc	143.3 ± 34.2b	2.9 ± 0.9a	3.5 ± 0.7a	1.8 ± 0.2a
FA	15.1 ± 8.0b	19.3 ± 6.7bc	107.5 ± 37.8b	6.1 ± 0.7b	7.0 ± 0.9b	3.2 ± 0.2b
OA	13.7 ± 3.2b	13.6 ± 6.7bc	95.4 ± 19.4b	3.1 ± 0.6a	4.4 ± 0.6a	2.5 ± 0.5a
CA+OA	32.0 ± 6.9a	41.5 ± 11.0a	360.1 ± 91.6a	3.4 ± 0.7a	3.0 ± 0.3a	3.2 ± 0.4b
FA+OA	23.2 ± 7.6ab	26.9 ± 15.1bc	146.6 ± 55.2b	6.5 ± 1.8b	8.2 ± 1.0b	3.6 ± 0.5b

Values correspond to an average ± standard deviation (n = 3). Different letters correspond to statistical differences between treatments according to Fisher's test $p \leq 0.05$.

CONCLUSIONS

Lead was found in a significant concentration in soils in bioavailable form. The application of citric acid significantly increased the level of lead in leaves of *Atriplex halimus*, reaching 32.0 ± 6.9 mg kg^{-1}, but decreased its growth, attributed to the effect of compaction in the soil that produced its application. In general, the plant showed higher lead concentration in roots than in the aerial part, presenting a high potential for phytostabilization of soils contaminated with lead.

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8.5.O

MITIGATION MEASURES FOR REDUCING NON-POINT SOURCE POLLUTION OF NITROGEN ON IRRIGATED MAIZE FIELDS IN THE MEDITERRANEAN ZONE OF CHILE

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INTRODUCTION

Maize production in Chile is predominantly for domestic consumption, and is located mainly in three central Regions: O'Higgins, Maule and Metropolitan. It is cultivated mainly on flat soils located in alluvial terraces (Casanova et al., 2013), under conventional irrigation systems during spring-summer (September-April). These production systems usually use high nitrogen (N) fertilisation rates between 350 and 560 kg N ha^{-1}, together with low irrigation efficiency (<45%) (Nájera et al., 2015).

Thus during the maize growing season a combination of two factors, N over-fertilisation and furrow irrigation systems favour N surplus losses from the maize fields into neighbouring water bodies that promote eutrophication. In addition, some recent studies evaluated the impact of agricultural activities on the level of NO_3-N in water bodies (Salazar et al. 2014; Fuentes et al. 2015). They showed that measured NO_3-N values in water bodies adjacent to agricultural areas in central Chile, where maize is the most common crop, usually exceed the Chilean water quality standard for drinking water (10 mg NO_3-N L^{-1}). Similarly, in other Mediterranean agroecosystems in the world, irrigated maize fields with high N doses have been highlighted as posing a high risk of creating diffuse N pollution areas (Quemada et al. 2013).

A broad range of mitigation measures have been used for maintaining surface and groundwater water quality, for instance applying an appropriate N rate, timely fertiliser application, incorporation of fertiliser, cover crops to scavenge dissolved inorganic N forms and appropriate cropping/residue management and vegetated buffer strips. The main aim of the manuscript is to resume the results of some studies of mitigation measures for reducing non-point source pollution of N carried out on maize fields in the Mediterranean zone of Chile between 2011 and 2016.

STUDY SITES AND METHODS

These studies has been carried out on maize fields located in the O'Higgins Region in central Chile. Maize is sown in spring (September-October) and harvested in autumn (March-April). Usually, a commercial hybrid maize adapted to this area is used for an anticipated stand of about 95,000 plants ha^{-1} (Nájera et al., 2015). In these fields the maize grain yields ranged from 11 to 19 t ha^{-1}. During the growing season the maize was irrigated using a furrow system with irrigation ranges between 10.000 and 20.000 m^3 ha^{-1} during the cropping cycle. All fields included in this study have a climate described as semi-arid Mediterranean, with hot summers and relatively cold winters, a mean annual air temperature of 15 ºC (29 ºC in January and 5ºC in July) and a mean annual precipitation of around 700 mm, mostly falling between May and October.

In 80 fields, the nutrient use efficiency (NUE) was calculated considering the ratio N output in harvested product/N input. N output in harvested product (maize grain) was calculated considering the grain yield and N concentration in grain, whereas N input considered only N fertilisation (Salazar et al., 2016). The NUE was used for comparison of the N fertilisation carried out by the farmers during the season 2014-2015 versus a scenario in which the recommended N rate by the extension service was applied for the season 2016-2017.

In a pilot field was carried out a study to determine if narrow vegetated buffer strips (BS) would effectively remove dissolved inorganic N from subsurface lateral flow (Salazar et al., 2015). Nitrate-N (NO_3-N) and ammonia-N (NH_3-N) concentrations in subsurface lateral flow were measured at 1 m depth in a BS system consisting of five treatments: G: strip of grass (*Fescue arundinacea*); GS: strip of grass and line of native shrubs (*Fuchsia magellanica*); GST1: strip of grass, line of shrubs and line of native trees 1 (*Luma chequen*); GST2: strip of grass, line of shrubs and line of native trees 2 (*Drimys winteri*); and C: bare soil as control. Water samples for the NO_3-N and NH_3-N measurements were collected between June 2012 and August 2014 in observation wells located at the inlet (input) and outlet (output) of each treatment.

Other study was carried out in four experimental fields considering different N management, irrigation system and crop rotation, such as: i) maize with furrow irrigation, mineral N fertilization and fallow ii) maize with furrow irrigation, pig slurry and fallow iii) maize with center pivot irrigation, pig slurry and cover crop; and iv) rainfed native annual prairie (SN) as natural reference site (Salazar et., 2017). In these sites were collected data for N balances between 2011 and 2013.

RESULTS

In the 80 fields was found that the mean NUE was 45%, ranging from 27% to 99%, where most of the NUE results (76% of the NUE values) were less than 50%. Therefore most farmer N rates exceeded yield potential indicating that residual N, mainly NO_3^-, may leach if there is sufficient water to percolate below the root zone. As a result, N over-fertilisation increase the risk of N soil accumulation and nitrate leaching. In addition, runoff from furrow irrigation can increase the risk of movement of residual N from the fields towards surface and subsurface waters during the growing season (Corradini et al., 2015). In contrast, using recommended N rate by the extension service was found a decrease in N fertilizer input for the crop season 2016-2017 compared to season 2014-2015 (Salazar et al., 2016). Under this scenario most of the NUE would decrease to values between the two NUE reference values of 50% and 90% (85% of the NUE values) (Figure 1). Clearly, most of maize fields were over-fertilised during season 2014-2015, and the N surplus ranged between 16 and 325 kg ha^{-1}.

In the vegetated BS study was found that the NO_3-N concentrations in lateral subsurface from maize fields were reduced after passage through all vegetated BS treatments (Table 1). The NO_3-N concentrations in the G treatment showed a significant concentration reduction (p<0.05) in comparison with the control (C). Vegetated BS (G, GS, GST1 and GST2) had a NO_3-N removal efficiency ranging from 33 to 67%, with a mean value of 52%. Unlike vegetated BS treatments, in the bare soil (C) the NO_3-N concentrations increased by 24%, showing the lowest removal efficiency and significant differences (p<0.05) with the BS treatments. No significant differences (p>0.05) in NH_3-N concentration were detected between treatments and NH_3-N removal efficiency of treatments with trees (GST1 and GST2) showed a higher NH_3-N retention capacity (p<0.05) than the other treatments.

In the N management and irrigation study was found that the field with furrow irrigation system and excessive pig slurry applications showed the highest risk of non-point source N pollution. Particularly, the slurry application in the wet winter had an important effect in N leaching, since the excess pig slurry applied to the field could have been transported by leaching to groundwater. In other field, the cover crop acted as a large NO_3–N sink in autumn–winter, when the cover crop through transpiration reduces the water percolation and in consequence the N leaching. In addition, N uptake by cover crop was an important process in the N retention capacity. In addition, the center pivot used a 30% of the furrow irrigation volume used during maize cultivation. However, the cost of center pivot irrigation can not be afford for most small farmers yet. Therefore for small maize farmers it is necessary improvements in furrow irrigation efficiency, considering in the design: suitable soil type, gentle furrow slope, the stream size, the irrigation depth and the field length.

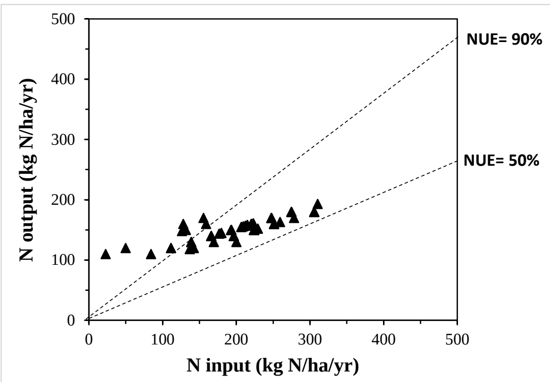

Figure 1. Nitrogen use efficiency (NUE) in the maize fields in the Chilean O'Higgins Region using a scenario of the recommended rate of nitrogen fertiliser by the extension service for the season 2016-2017 (n=80). The slope of the diagonal wedge represent a range of desired NUE between 50% and 90% (Salazar et al. 2016).

Table 1. Mean input and output values of nitrate-nitrogen (NO_3-N) and ammonia-nitrogen (NH_3-N) concentrations in the different treatments (Salazar et al., 2013).

Treatment	NO_3-N[a]			NH_3-N[a]		
	Input	Output	Removal	Input	Output	Removal
	--- mg NO_3-N L^{-1}---		**- % -**	--- mg NH_3-N L^{-1}---		- % -
C	5.33a	6.63a	-24d	1.98a	3.86a	-95c
G	5.56a	1.85b	67a	2.72a	2.90a	-7b
GS	5.77a	2.93ab	49b	2.02a	2.58a	-28b
GST1	8.05a	5.41ab	33c	3.36a	0.65a	81a
GST2	6.45a	2.61ab	60ab	1.62a	0.56a	65a

[a]Means (n=99) within columns with different letters are significantly different (Kruskal-Wallis test, *p*<0.05).

CONCLUSIONS

Overall, great emphasis must be placed on good agronomic management, including accurate calculation of N fertiliser rates, improving water use efficiency and establishment of suitable mitigation measures, such as vegetated buffer strips and cover crops. The recommended N rate should be calculated based on a mass N balance for assessing crop N fertiliser needs by considering N uptake at a specific dry matter yield level and N contributions from non-fertiliser sources, thus most fields would show N use efficiency (NUE) between 50% and 90%. The centre pivot irrigation applied 30% of the furrow irrigation volume used by most farmers during maize cultivation, which may reduce the risk of deep percolation and surface runoff and in consequence reduce N losses. The vegetated buffer strips located along the edge of maize field have shown NO_3–N removal efficiency ranging from 33 to 67 % (mean 52 %) of subsurface later flow reaching nearly open drainage channels. In a maize-cover crop rotation, the cover crop acted as a large N sink in autumn-winter, with transpiration of the cover crop reducing percolation and consequently N leaching.

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8.7.O

PARAQUAT AND GLYPHOSATE SOIL/WATER DISTRIBUTION USING THE MULTILINEAR MODEL TO DETERMINE THEIR ENVIRONMENTAL FATE

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ABSTRACT

In Colombia the environmental fate of pesticides has been poorly studied, and in 2017, the new environmental soil resource policy is coming. This situation has made that many farmer associations want to know what is the environmental state of their places. However, this kind of studies are very expensive and take too much time and effort. For this situation, we decided to use modelling tools, because they are cheap and with some analysis we can predict the state of the place. To model is soils, is well known that the organic matter is generally the most important soil constituent responsible for the sorption of organic compounds and this has led to the use of the organic carbon normalized partition coefficient, Koc (L/kg), the octanol-water partition coefficient, Kow (Lwater/L octanol), and fraction of organic carbon in the soil, foc to describe this phenomenon. These simplifications have been successfully employed to model the partitioning of many hydrophobic organic chemicals like PCBs and PAHs, but their application to pesticides results in order of magnitude errors. For this reason, we use the multilinear model. This model was developed to predict the environmental fate of munitions constituents and predict the partition coefficient (soil/water) with some soil properties. We conducted batch experiments near 1:1 soil to solution ratios reflecting field conditions using a Paraquat and Glyphosate agricultural solution concentrations of corn and coffee beans crops because in our Country these products are the ones in the agricultural production. We made an experiment that involved 2 days of adsorption in two soils. The most important observation was that for each pesticide, we successfully predicted the partition coefficients improving them by a factor of 2 in the accuracy compare with the carbon normalization traditional approach using only the organic carbon, clay content and extractable iron soil information.

INTRODUCTION

Gonzalez (2014) demonstrated the importance of including additional properties of the soil and clay-size particles in sorption modeling of nitro-compounds as explosives. This finding has also been observed by many authors as Sheng et al. (2001), Van Noort (2010) and Dontsova et al. (2009). In addition, Gonzalez (2014) developed the Multilinear Model. It calculates the partition coefficient (nitro-compound concentration into the soil/ nitro-compound concentration into the water) Kp from the sum of three linear terms, one for the organic carbon, the second one for the clay size particle and the third one as the extractable iron content as the fraction of each component times their corresponding sorption coefficients. Equation 1 shows the multilinear model:

$$K_{ps,m} = Koc_{\,m}(f_{oc})_s + K_{Clay\,m}\,(f_{Clay})_s + K_{Fe\,m}\,(f_{Fe})_s \qquad (1)$$

where:, $K_{oc\,m}$ = sorption coefficient to organic carbon in the soil and $K_{Clay\,m}$ = sorption coefficient to the clay in the soil, $K_{Fe\,m}$ = sorption coefficient to the extractable iron in the soil, f_{oc} = fraction of the organic carbon in the soil, f_{Clay} = fraction of the clay in the soil, f_{Fe} = fraction of the extractable iron in the soil and: s = soil, m = munitions constituents. Foc, fclay and f_{Fe} are properties of a soil and Koc_m, $Kclay_m$ and K_{Fe} are function of the nitro-compound used.

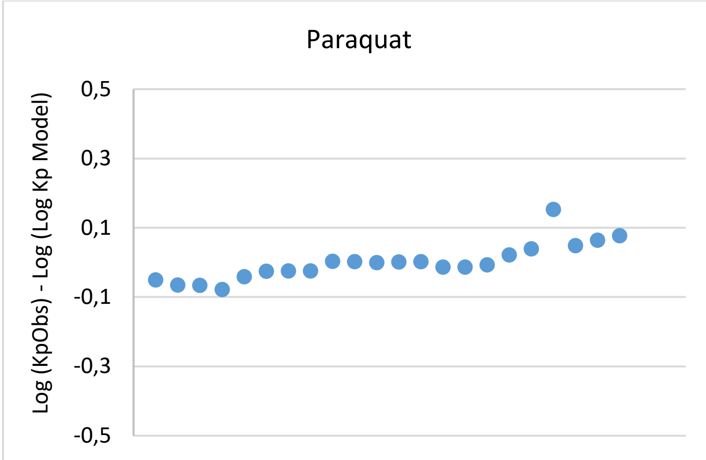
METHODS

This study employed 3 soils collected from agricultural zones of Colombia (Corn and coffee beans crops) and their properties were determined by the soil laboratory at the national authority of soils in Colombia IGAC (Agustin Codazzi Geographic Institute).

For the sorption experiment, 5 ± 0.0001 grams of soil were added to 12 mL borosilicate centrifuge tubes with phenolic caps and PTFE liners. Batch sorption experiments were conducted at 1:1 (w/v) soil to solution ratio, reflecting near field conditions and at room temperature. Soils were hydrated for 5 days prior to the addition of pesticides in a solution containing 0.01 M calcium chloride ($CaCl_2$) and 0.01 M sodium azide (NaN_3). $CaCl_2$ was added to prevent flocculation of soil components and NaN_3 was present as a microbial growth inhibitor. After the 5-day hydration time, 5 mL of pesticides were added. The concentration of each pesticide was determined by the typical agricultural application for each crop. Triplicate samples were vortex mixed for 15 seconds to suspend the soil and next they were shaken at 10 rpm in an end-over-end shaker for 2 days (this methodology was the same as Gonzalez, 2014 applied the multilinear model develop). Subsequently, the tubes were centrifuged for 30 min at 3000 rpm (750 G) and the decanted supernatant was filtered and the supernatant obtained from each adsorption was analyzed by COD (Chemical Demand Oxygen) as organic matter. A calibration curve relating COD and pesticide concentration was done to determine the pesticide remaining in the solution after the adsorption. After that, calculations were done and soil pesticide concentration and Kp were determine.

RESULTS

The partition coefficient K_p in L Kg^{-1} was calculated from the data as the relationship between the amount of MC sorbed per mass of soil and the concentration remaining in the solution after equilibration in the adsorption step. The parameters K_{OC} K_{Clay} and K_{Fe} were calculated for all the chemicals and soils. They were obtained by fitting the multilinear model by the minimization of the log residuals square between the K_p calculated from the experimental data and the K_p obtained by the model using the Excel solver tool. Figure 1. Shows the fitting between Kp Observed and Kp Model to the crops (corn and coffee beans) of each pesticide because the residual plot varying 0.2 and -0.2. As is observed, the closeness between data is very well.



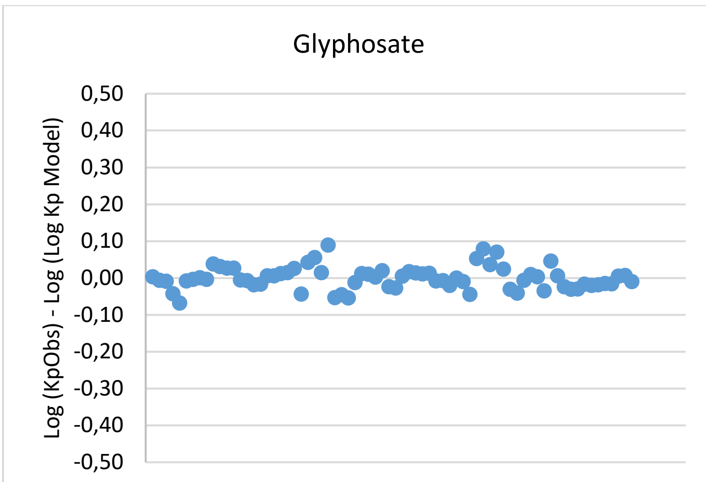


Figure 1. Residual plots Log (KpObs) - Log (Log Kp Model) of Paraquat and Glyphosate

In addition to that, the root mean square error RMSE of the predicted and measure K_p data was calculated and the results were: 0,008 (Paraquat) and 0,0095 (Glyphosate) indicating that the model can be used to predict the fate of these pesticides in those soils.

The parameters Koc, Kclay and Kfe of each pesticide and crop obtained are presented in Table 1 and the data suggest that Glyphosate has more affinity to clay than organic carbon and extractable iron. In the case of Paraquat the extractable iron has more influence in the adsorption of the chemical for each crop.

Table 1. Parameters of the multilinear Model of Glyphosate and Paraquat

Glyphosate			
Crop	Koc	Kclay	Kfe
Corn	1,04442581	1,20741098	0,18027903
Coffee Beans	1,06447209	1,3556232	0,15034615
Paraquat			
Corn	1,00019045	1,00317475	1,91673713

CONCLUSIONS

The multilinear model can be used to predict the fate of Glyphosate and Paraquat in the soils of corn and coffee beans because the RMSE obtained to compare the Kp measured and Kp modeled were very low (0,008 and 0,0095 respectively).

The parameters of the models Koc, Kclay and Kfe indicates that Glyphosate has more affinity to clay and organic carbon as soil properties rather than extractable iron, but for Paraquat the extractable iron is the main soil property that control the adsorption phenomena.

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8.1.P

PRESENCE OF PHARMACEUTICALS AND HEAVY METALS IN SOILS OF A MEDITERRANEAN COASTAL WETLAND: POTENTIAL INTERACTIONS.

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INTRODUCTION

Since the Industrial Revolution, one of the most important parameters in human development fingerprinting have been the heavy metal presence in the different environmental compartments. In the last decades, with the advance of analytical techniques, other compounds appeared to be relevant in environmental pollution because of human development. These last, mainly of organic nature, have been called “emerging contaminants”. Among them, pharmaceuticals are of increasing concern worldwide because of its potential effects on the fauna through soils, waters and sediments, and the large number of drugs used in human and veterinary medicine (Andreu et al., 2009). The number of this substances detected in the environment has shown a huge increase in the very last years covering a wide scope of almost all therapeutic classes. Usually, these compounds reach the soil by the treated wastewaters used in agriculture irrigation or by the use of sewage sludge as organic amendment.

By the other hand, wetlands are one of the most sensitive ecosystems in the world because are sustained by a very fragile equilibrium of its intrinsic conditions (level of waters, soils, vegetation), and support a very rich biodiversity. Among these ecosystems, the coastal wetlands present the greatest dynamism and biodiversity, but they show a great fragility, being particularly sensitive to alterations in their soil and water quality. In these sense, they are suffered the pressure of the exponential increase in population occurred in the Mediterranean coastal areas of Europe. Related to these pressures, pharmaceuticals and heavy metals have been considered as potential indicators of human development in the different environmental Media (Vystavna et al., 2013). Pharmaceuticals have been also considered as anthropogenic markers of social activity; meanwhile, heavy metals have been more related to industrial activities.

In this work, the presence of seven heavy metals and 17 pharmaceuticals was studied in soils of the Natural Park of L’Albufera (Valencia, SPAIN), a typical Mediterranean coastal wetland, which shows a constant pressure derived from the high human and industrial occupation. The potential interactions between heavy metals (total and extractable fractions) and pharmaceuticals were evaluated.

METHODOLOGY

Study Area

The target of this study is the L’Albufera Natural Park, which is located at the Eastern shore of Spain, in the Valencian Community. It was selected for its special characteristics that have favored their declaration as Natural Park –one of the most important in Europe. L’Albufera is a coastal lagoon, of almost 27,538 m^2 surface, surrounded by rice fields. It also comprises a coastal strip occupied by dunes and pinewoods in its not urbanized part. In spite of this great ecological value, among other multiple impacts, it suffers those derived from the high human and industrial occupation, and of the decrease of hydrological contributions in dry seasons.

In April 2012 and October 2013, soil samples were collected at 20 different locations (see Fig. 1) covering the most important land uses of the Natural Park.

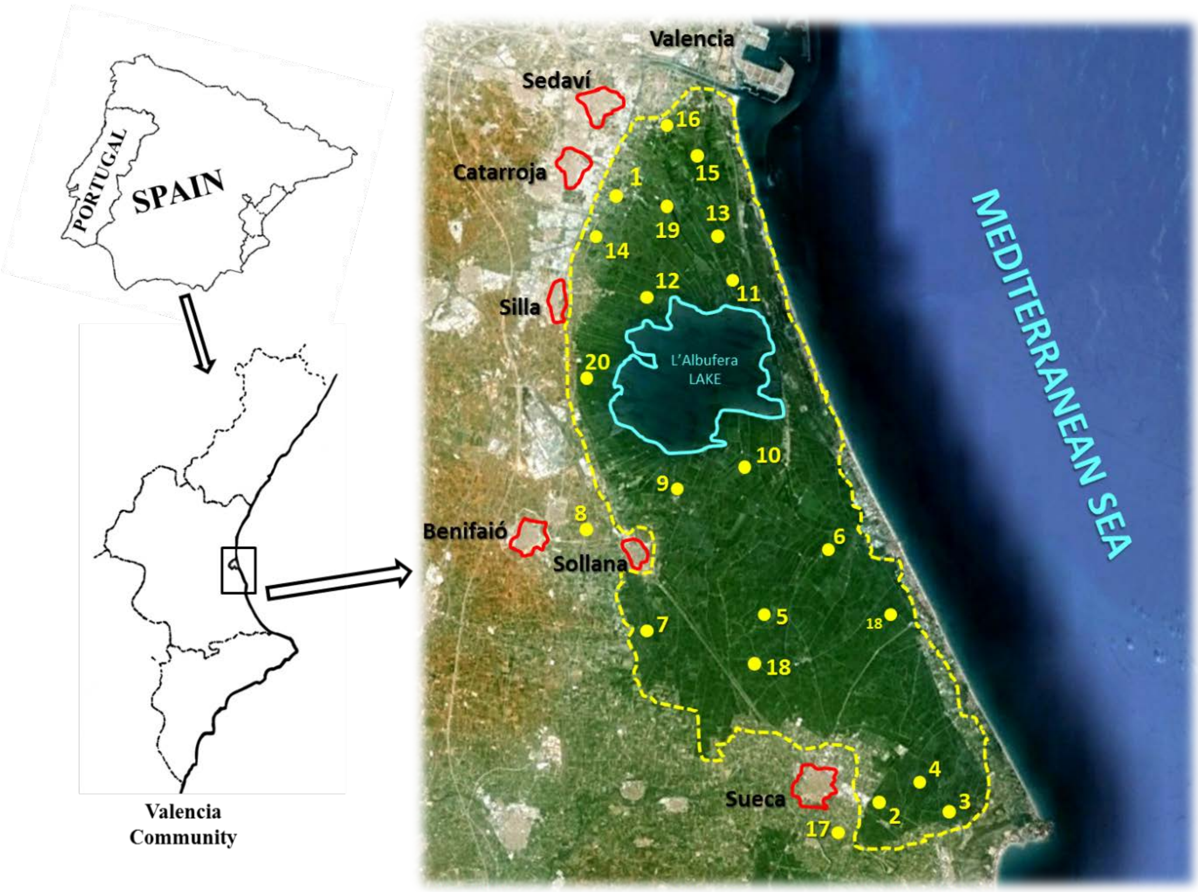


Figure 1. Study area and sampling zones in the Natural Park of L'albufera (Valencia, Spain)

Sampling

Soil samples of the upper 0-15 and 15-50 cm depth layers were collected. From each of the 20 sampling zones, of 25 m^2, 5 sub-samples were taken. Once in the laboratory, samples were dried and passed through a 2mm Ø sieve, and then, the sub-samples of each sampling point were homogenized to create a composite one. The composite soil samples were extended in a layer of approximately 1 cm thickness on polypropylene trays and air-dried in darkness at 20 ºC to moisture content of approximately 3% water. Then, samples were stored in sealed plastic bag at 4 ºC.

Analytical

Standard analytical methods were used to measure soil physical and chemical properties. Total content and extractable fraction of seven heavy metals (Cd, Co, Cr, Cu, Ni, Pb and Zn) in soil samples were extracted by microwave acid digestion, and the extractable fraction with EDTA, both were determined by ICP-OES.

Pharmaceuticals were selected covering an important and wide variety of different therapeutic classes: β-blockers (metoprolol-MPL and propranolol-PRL), antidepressants (diazepam-DZP), anti-epilectic drugs (carbamazepine-CBZ), analgesics (acetaminophen-ACM and codeine-CDN), nonsteroidal anti-inflammatory drugs (ibuprofen-IBF and diclofenac-DCF), lipid regulators (clofibric acid-CFA and fenofibrate-FNF) and seven of the most used antibacterials (ciprofloxacin-CPX, norfloxacin-NFX, ofloxacin-OFX, oxytetracycline-OTY, sulfamethoxazole-SMZ, tetracycline-TCY and trimethoprim-TMP). To determine the selected pharmaceuticals in soils, the samples were lyophilized, homogenized with EDTA-washed sea sand and extracted according to a previously reported procedure (Vazquez-Roig et al., 2010) by pressure liquid extraction (PLE) with water at 90 ºC and 1500 psi. The analytes from aqueous extract were isolated by solid-phase extraction (SPE) using a combination of SAX and Oasis HLB cartridges, and then, eluted with methanol. Determination was carried out by liquid chromatography-tandem mass spectrometry (LC-MS/MS) with electrospray (ESI) in both, positive and negative ionization modes.

RESULTS AND DISCUSSION

All analyzed pharmaceuticals were detected, in at least one sample, at concentrations from MDL to 13.2 ng g-1. The presence of pharmaceuticals in soils was sporadic, probably because they are not in permanent contact with water or by the high biological and chemical activity of soils. Only in five sampling zones, pharmaceuticals were not detected. Acetaminophen, carbamazepine, fenofibrate and diazepam were detected in the first 15 cm depth. However, it is between 15 and 50 cm depth where the largest number of these compounds was found, being found acetaminophen, carbamazepine, diazepam, fenofibrate, propranolol, sulfamethoxazole, and tetracycline. Carbamazepine is the only drug appears in 15 sampling points (15) followed by acetaminophen (8), while the sulfamethoxazole and propranolol only appear at one point each. The highest levels determined were for Simazine (252.32 ng/g) and Acetaminophen (116.47 ng/g). At the North of our studied area, values from 18.8-64.3 µg kg-1 and 15.7-105.4 µg kg-1 of tetracycline and oxytetracycline were found, respectively. This fact could be due to the higher pressure that suffer the agricultural soils close to a city like Valencia (900,000 inhabitants) compared with L'Albufera Natural Park that although is surrounded by towns, they are small ones with an average of 1500 inhabitants, only 3 of them reach the 15,000 inhabitants.

Higher levels and frequency of pharmaceuticals appears in the north area of the lagoon, which is justified by a larger concentration of population. However, the most contaminated sampling point was P8, located in the southwest of the lagoon. This point is located in the middle of an agricultural area but surrounding by several leisure points. Organic matter and available phosphorous are the soil parameters more significantly related to the pharmaceuticals.

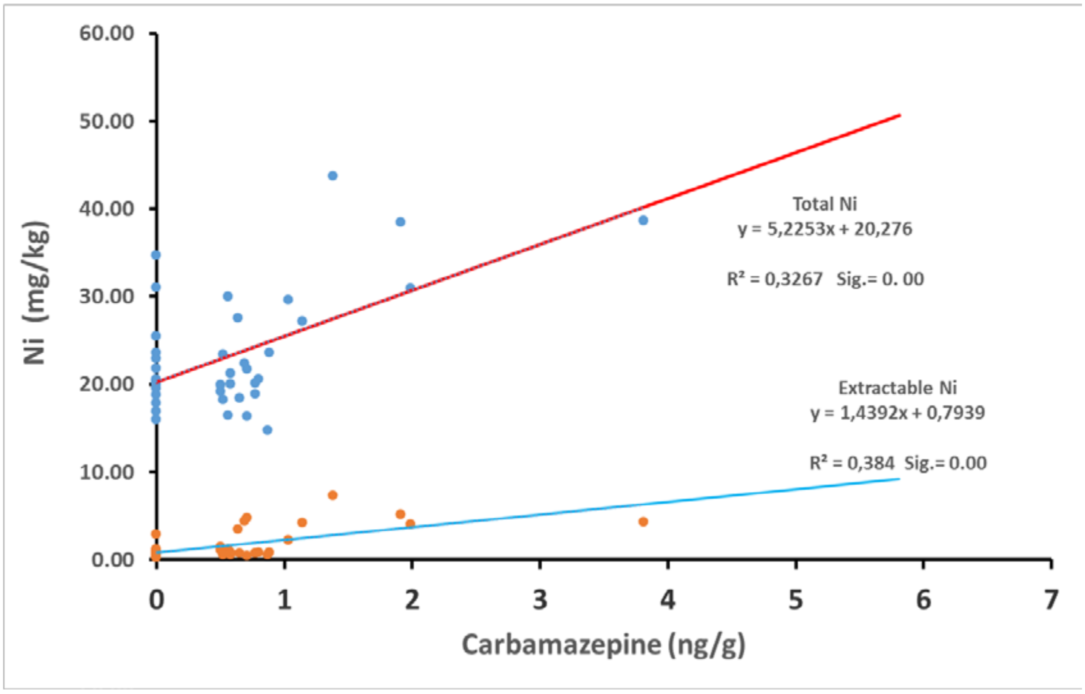
Regarding extractable content of metals, a similar pattern of distribution like total contents in the study area was observed. The highest values were for Cu (62.48 mg kg-1) y Zn (58.75 mg kg-1) in the north on the Park. From the study of the proportion of metal available regarding total content, has been observed that some zones showed very high values in toxic metal like Cd (79.26%, zone 16) and Pb (59.08%, zone 4), or Cu (58.78%, zone 11). It could indicate a potential toxicity state of these soils.

Cr, Cu and Zn show the highest concentrations in all land uses and zones. Cr is the metal that present maximum concentration in the studied area (254.93 mg kg-1), being almost the only metal studied that exceeds the limits established by the Spanish an EU legislation. Co and Ni, and the same for Acetaminophen and Diazepan, show a tendency to accumulate below the 30 cm depth, the other metals and pharmaceuticals studied continue with the cumulative trend in surface horizons. Carbamazepine, Diazepam and propranolol showed high significant correlations with all extractable metals, except Co and Zn.

Ni, in its different forms, shoed highly significant correlations with Carbamazepine and Diazepan. The northeastern zone of the Natural Park is the most contaminated one in all cases and soil uses for both pharmaceuticals and heavy metals. The overall trend shows that levels of metals in surface samples are higher in rice farming soils than in citrus crops, with the exception of Zn, where the trend was reversed. Soils of natural zones show the lower values, significantly different from those of the other land uses. All the studied metals, except Co, have highly significant correlations with the available phosphorous and K, which indicates a possible influence of fertilizers and organophosphorous pesticides as main input sources, mainly in the case of rice farming soils.

CONCLUSIONS

The human pressure on the environment results, in one of its aspects, in the emergence of products of common use in soils. In this work, it has been established the presence of pharmaceuticals, both in the surface horizon and depth in soils of the Natural Park of L'Albufera de Valencia (Spain). The northern area of the Park showed the presence of a greater number of the compounds and metals studied, which coincides with a greater density of population and the presence of important WWTPs. In this area, the sampling zones closer to the coast presented higher concentrations of metals, mainly sampling zones 11 and 15, in the NE of the Park. All the studied metals, except Co have highly significant correlations with the available phosphorous and K, which indicates a possible influence of fertilizers and organophosphorous pesticides as main input ways, mainly in the case of rice farming



soils. Organic matter and available phosphorous are the soil parameters more significantly related to the pharmaceuticals. Significant relationships between metals and pharmaceuticals were detected, mainly between Ni and diazepam, carbamazepine and propranolol.

Figure 2. Relationships found between carbamazepine and Ni in soils of the studied area.

The consequences derived from the incidence of these drugs in soils can induce to changes in its fauna and in his meso and microflora, although these effects are still sparsely studied, mostly in ecosystems as fragile as the Mediterranean wetlands.

ACKNOWLEDGEMENTS

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8.1.P

EFFECTS OF BIOCHAR ON COPPER IMMOBILIZATION AND SOIL MICROBIAL COMMUNITIES IN A METAL CONTAMINATED SOIL IN A TWO-YEAR TRIAL

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1 INTRODUCTION

Anthropogenic activities have been contaminating soils with metals (*Choppala et al.*, 2012). Several methods for remediation of metal polluted soils have been developed including land excavation, soil washing and stabilization to reduce metal bioavailability (Hakeem et al., 2015). Nonetheless, effective low cost practices are needed to solve this problem, emerging the phytoremediation as a cheaper method for phytoremediation (*McGrath and Zhao*, 2003). The success of the phytoremediation depends on the ability of plants to produce biomass, which is difficult in a soil with phytotoxic metal concentrations (*Ginocchio et al.*, 2004). In this sense, the use of BC to metal contaminated soil systems is being attempted in order to promote plant establishment (Meier et al., 2015). Nevertheless, mostly of experiments using BC as a soil amendment in metal contaminated soils has are focused on the reduction of metal availability paying less attention to simultaneous effects of BC on plant growth, soil microbial activity and changes on microbial communities measured in a long-term assay.

If biochar is able to reduce metal uptake by plants, will the biochar amendments protect the microbial communities in Cu polluted soils from harm and maintain its effect on time? We hypothesized that the BC amendments in soil would enhance the activity of bacterial and fungal communities in Cu contaminated soils using a metallophyte and those effects will be maintained on time. The objective of this study was to investigate the effects of BCs on the Cu immobilization and over soil microbial communities in a metal contaminated soil using the metallophyte *Oenothera picensis* in a two years assay.

2 MATERIALS AND METHODS

This soil was obtained from the vicinity of Ventanas Cu smelter, situated in Puchuncaví Valley of Central Chile (32°46’30” S, 71°28’17” W). The soil properties are as follows: pH 5.76, OM 3%, CEC 4.63 cmol (+) kg^{-1}, Cu 338 mg kg^{-1}, N 6 mg kg^{-1}, P 11 mg kg^{-1}, K 21 mg kg^{-1} and Zn 17 mg kg^{-1}.

Chicken manure biochar (CMB) and oat hull biochar (OHB) were produced at 500 and 300 ºC.

2.1 Plant growth experiment

The soil was mixed by hand with CMB or OHB (0, 1 and 5% w/w) until reach homogeneity. Accordingly the BC treatments are termed as CMB1, CMB5, OHB1 and OHB5. The natural soil (without biochar addition) was used as control (NS). Seeds of *O. picensis* were surface sterilized and transplanted to 1L pots and were allowed to grow for 6 months and then harvested (Harvest 1 –H1-). In each harvest 150g of soil were removed from each pot and used for physiochemical analysis. After that, new plantlets were put in the remaining soil and grown for 6 months (Harvest 2). The process was repeated two times more in order to obtain harvest 3 (H3) and harvest 4(H4).

2.2 Soil microbial activity

At the end of each harvest, soil samples from pots were analyzed for microbial activity. Soil basal respiration and dehydrogenase activity (DHA)

2.3 PCR-DGGE of the bacterial and fungi communities

The composition on microbial communities (bacteria and fungi) was analyzed by PCR-DGGE, according to the method described by Acuña et al. (*Acuña et al.*, 2013). For bacterial communities analysis, fragments in 16S rRNA gene were amplified by touchdown-PCR using specific primer set EUBf933-GC. Fungal ITS regions (18S rRNA gene) were amplified with nested-PCR using primer set ITS1F (Data not shown in this abstract)

2.4. Copper fractionation in the soil

For chemical analysis of soil samples (collected after each harvest) were dried and analyzed for pH, and metal fractionation using a sequential extraction technique (*Tessier et al.*, 1979).

3 RESULTS

The BC addition reduced easily exchangeable fraction and increased the organic matter and residual bound fraction of Cu in all treatments and harvest evaluated (Fig. 1).

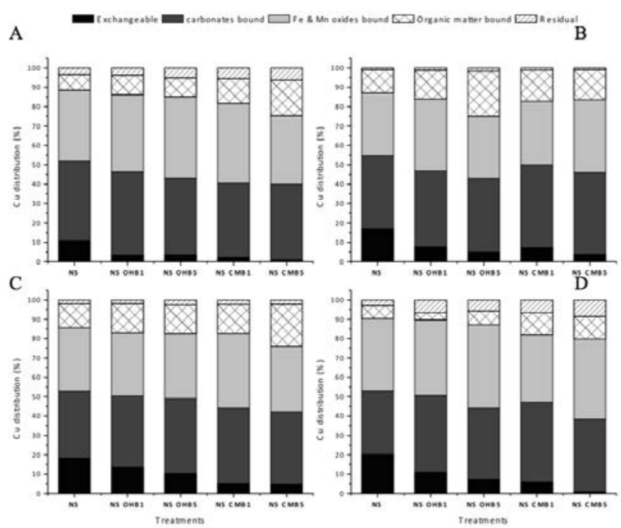

Fig. 1 Copper fractions in soil after pot experiment under the influence of chicken manure biochar (CMB) or oat hull biochar (OHB) applied at doses of 0, 1 and 5% w/w evaluated in four harvests. A) Harvest 1(H1); B) Harvest 2 (H2); C) Harvest 3 (H3); D) Harvest 4 (H4).

Copper was strongly bound in the control soil because ~50% of its total content was associated with Fe and Mn oxides, organic matter and residual bound fractions, which are no available for plants (Fig. 1). The CMB1 and CMB5 decreased the exchangeable fraction of Cu by 80% and 90% and its effect was maintained in time for all harvest evaluated.

Both biochars increased the plant biomass production and reduced Cu uptake by plant being its effect dependent of the doses applied (Table 1). In this sense, CMB1 treatment nearly doubled the shoot and root biomass of *O. picensis* in harvest 1 (H1) compared to that of the control i.e. NS (Table 1). This effect was generally increased in time (H2, H3, H4) even nearly quadrupling its effect in H4.

Table 1. Shoot and root biomass production Cu concentration in plant tissues of *Oenothera picensis* grown in soils amended with chicken manure biochar (CMB) and oat hull biochar (OHB) at doses of 0, 1 and 5% (w/w). Each value represents the mean of three replicates ± standard error. Different letters within a column indicate a significant difference at p < 0.05 according to Tukey's multiple range tests using SPSS 15 software.

Harvest 1					
		Biomass (g)		Cu concentration (µg g^{-1})	
Treatment	**BC (%)**	**Shoot**	**Root**	**Shoot**	**Root**
NS (Control)	0	2.23 ± 0.24^{c}	0.33 ± 0.05^{c}	72.7 ± 1.2^{b}	1664 ± 54^{a}
CMB	1	4.93 ± 0.04^{b}	0.61 ± 0.03^{bc}	74.1 ± 4.9^{b}	931 ± 203^{bc}
OHB	1	4.79 ± 0.79^{b}	0.82 ± 0.07^{ab}	90.1 ± 2.4^{a}	1494 ± 133^{ab}
CMB	5	6.87 ± 0.45^{a}	0.92 ± 0.06^{a}	71.2 ± 1.4^{b}	632 ± 25^{c}
OHB	5	4.85 ± 0.43^{b}	0.88 ± 0.12^{ab}	64.1 ± 3.0^{c}	1212 ± 150^{abc}
Harvest 2					
NS (Control)	0	1.16 ± 0.26^{b}	0.14 ± 0.08^{b}	43.5 ± 3.9^{a}	1295 ± 143^{a}
CMB	1	2.67 ± 0.07^{ab}	0.44 ± 0.07^{ab}	37.3 ± 7.7^{a}	826 ± 48.6^{ab}
OHB	1	1.86 ± 0.34^{b}	0.33 ± 0.02^{ab}	37.9 ± 2.8^{a}	851 ± 274^{ab}
CMB	5	4.99 ± 0.60^{a}	0.59 ± 0.10^{a}	37.0 ± 0.8^{a}	222 ± 30.8^{b}
OHB	5	2.18 ± 0.22^{b}	0.40 ± 0.03^{ab}	38.3 ± 7.2^{a}	818 ± 96.3^{ab}
Harvest 3					
NS (Control)	0	1.64 ± 0.33^{c}	0.31 ± 0.1^{c}	34.7 ± 3.53^{a}	643.3 ± 22.0^{a}
CMB	1	3.0 ± 0.27^{b}	0.98 ± 0.14^{b}	30.2 ± 0.7^{a}	288.9 ± 15.2^{b}
OHB	1	2.24 ± 0.46^{bc}	0.47 ± 0.07^{bc}	27.6 ± 1.68^{a}	429.3 ± 112^{ab}
CMB	5	7.75 ± 0.77^{a}	0.98 ± 0.14^{a}	27.4 ± 0.69^{a}	287.9 ± 26.7^{b}
OHB	5	3.1 ± 0.27^{b}	0.47 ± 0.09^{bc}	27.8 ± 3.1^{a}	396.6 ± 63.8^{ab}
Harvest 4					
NS (Control)	0	1.20 ± 0.3^{b}	0.18 ± 0.02^{b}	68.2 ± 12.91^{a}	1936 ± 105^{a}
CMB	1	3.25 ± 0.24^{ab}	0.61 ± 0.06^{a}	33.8 ± 7.83^{b}	1699 ± 114^{ab}
OHB	1	2.85 ± 0.47^{ab}	0.47 ± 0.07^{ab}	37.2 ± 5.71^{b}	1551 ± 69^{b}
CMB	5	4.70 ± 0.27^{a}	0.60 ± 0.06^{a}	30 ± 1.80^{b}	1161 ± 76.4^{bc}
OHB	5	3.1 ± 0.74^{ab}	0.37 ± 0.02^{ab}	40.8 ± 7.8^{ab}	1377 ± 71^{c}

The microbial activities (respiration and DHA) increased in all BC treatments compared to control (Table 2). In H1 the CMB increased the soil basal respiration by ~19 and ~37%, respectively for the low and high doses and these effects were increased in H2 (~42 and ~25%) and H3 (~61 and ~47%). On the other hand, the DHA activity was increased by both biochars in all harvest evaluated, being more effective in CMB treatments (Table 2). In H1 CMB increased DHA by ~62 and ~671% for the low and high doses respectively, and its positive effect remained and even increased in time.

Table 2 The pH, microbial activity, and dehydrogenase activity (DHA) in a Cu-contaminated soil spiked with additional Cu and amended with chicken-manure-derived biochar (CMB) and Oat Hull derived biochar (OHB) at doses of 0, 1 and 5% (w/w). Each value represents the mean of three replicates ± standard error. Different letters within a column indicate a significant difference at $p < 0.05$ according to Tukey's multiple range tests using SPSS 15 software

Harvest 1				
Treatment	**BC (%)**	**pH**	**Basal respiration (mg CO2 kg^{-1} h^{-1})**	**DHA (mg TPF kg^{-1} 24 h^{-1})**
NS (Control)	0	5.35 ± 0.01^{c}	4.42 ± 0.51^{c}	4.66 ± 0.09^{c}
CMB	1	6.29 ± 0.05^{b}	5.24 ± 0.6^{b}	7.63 ± 0.48^{b}
OHB	1	6.57 ± 0.12^{ab}	5.01 ± 0.47^{b}	7.82 ± 0.64^{a}
CMB	5	6.45 ± 0.21^{b}	6.05 ± 0.47^{a}	31.3 ± 3.39^{a}
OHB	5	7.04 ± 0.09^{a}	5.7 ± 0.42^{b}	8.0 ± 0.50^{b}
Harvest 2				
NS (Control)	0	5.21 ± 0.04^{c}	5.01 ± 0.31^{c}	2.03 ± 0.13^{d}
CMB	1	6.73 ± 0.09^{b}	6.29 ± 0.20^{ab}	4.67 ± 0.28^{c}
OHB	1	6.10 ± 0.10^{bc}	5.47 ± 0.20^{bc}	3.80 ± 0.61^{cd}
CMB	5	6.97 ± 0.06^{b}	7.10 ± 0.42^{a}	8.88 ± 0.63^{b}
OHB	5	7.31 ± 0.09^{a}	5.82 ± 0.12^{bc}	13.24 ± 0.63^{a}
Harvest 3				
NS (Control)	0	5.32 ± 0.04^{c}	4.19 ± 0.20^{c}	1.13 ± 0.08^{c}
CMB	1	6.45 ± 0.11^{bc}	6.17 ± 0.12^{ab}	3.47 ± 0.46^{b}
OHB	1	6.30 ± 0.10^{bc}	5.59 ± 0.20^{b}	2.17 ± 0.16^{bc}
CMB	5	6.99 ± 0.08^{b}	6.75 ± 0.23^{a}	8.39 ± 0.12^{a}
OHB	5	7.36 ± 0.12^{a}	6.05 ± 0.12^{ab}	7.08 ± 0.40^{a}
Harvest 4				
NS (Control)	0	5.30 ± 0.14^{d}	4.77 ± 0.23	2.95 ± 0.10^{c}
CMB	1	6.33 ± 0.20^{c}	6.40 ± 0.23^{ab}	4.29 ± 0.24^{b}
OHB	1	6.13 ± 0.15^{c}	7.33 ± 0.20^{a}	3.64 ± 0.1^{bc}
CMB	5	6.96 ± 0.06^{b}	5.70 ± 0.12^{bc}	6.67 ± 0.14^{a}
OHB	5	7.37 ± 0.12^{a}	5.94 ± 0.20^{bc}	6.17 ± 0.27^{a}

Analyses of DGGE profiles and non-metric multidimensional scale (nMDS) for all harvests (H1 to H4) are shown in Figures. 2,3,4 and 5. The result shows an unequal intensity of dominants bands in all DGGE profiles of BCs treated soils in respect to control, either for bacterial and fungal communities. According to this differences in clustering analysis of bacterial communities for harvest 1 revealed the existence of 65% similarity between CMB1, OHB5 and OHB1, whereas CMB5 shown 55% similarity with respect to

control (Fig. 2A). The above effect was lesser evident in H2 were, in general, no differences among groups were observed, with the exception of CMB at the highest doses applied with a separation of 50% (Fig. 2C). Nevertheless, H3 and H4 presented differences between NS and amended soils highlighting the soils amended with CMB5 with a separation of 50 and 20% for H3 and H4, respectively (Fig 3).
These results suggest that the type and dose of BC produce changes for fungal and bacterial community structure presenting a high variability and its impact could be major for bacteria community among the different BC treatments

4 CONCLUSIONS

The incorporation of biochar into Cu-contaminated soil has the potential to reduce the Cu availability in soil and it positive effect was dependent of kind of BC used and the doses applied. In this sense, both BCs reduced the bioavailable Cu of soils by 5 and 10 times at the highest doses and it effect was steadily on time. The microbial respiration and DHA activity were increased in all BC treatments and harvests evaluated, reaching maximum values of 7 and 6 times higher than control soils in CMB and OHB respectively. The above could be related with the changes in bacterial and fungal communities produced by BC, which influenced the soil biophysicochemical properties, improving the habitat for microorganisms. In this regard, both BC altered the fungal and bacterial communities, but the effect were more evident using CMB and evaluated in harvest 1. The changes of physochemical soil properties and microbial communities induced by BC resulted in a better plant grow in amended soils. The BC treated plants grew, in average, 4 and 3 times more in CMB5 and OHB5 respectively. The results show that the BCs decreased Cu available fractions, reduced Cu uptake by *O. picensis,* improved habitat for microorganisms, changed the microbial communities in soil and enhanced plant growth in Cu polluted soil, suggesting that biochars may be utilized to remediate Cu contaminated soils.

ACKNOWLEDGEMENTS: This research was supported by Fondecyt project number 11150480,

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8.6.P

COMBINATION EFFECT OF INORGANIC NITROGEN FERTILISATION AND COVER CROP-MAIZE ROTATION IN NITROGEN LEACHING

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INTRODUCTION

In Mediterranean zones of Chile there are further concerns because N over-fertilisation in irrigated maize fields can be associated with a high risk of diffuse pollution of water bodies (Nájera et al., 2015). Particularly important are coarse textured soils cultivated with maize that are more prone to N leaching due to the high percolation and low water retention capacity (Casanova et al., 2013). Particularly, dissolved inorganic N (DIN) forms, such as NO_3-N has been pointed out as the main sources of N leaching in agricultural systems that promote eutrophication of water bodies.

Similarly, in other Mediterranean areas in the world, irrigated maize fields with high N doses have been highlighted as a nitrate source in N diffuse pollution areas (Quemada et al., 2013). In these areas, the inclusion of cover crops during the intercropping period of maize has been proposed to counteract the negative impacts of DIN diffuse pollution from irrigated maize fields (Gabriel and Quemada, 2011). As in Central Chile it is a common practice to leave a fallow period between maize cash crops, we hypothesized that replacing the fallow with a cover crop could mitigate DIN leaching losses from the cropping system. The main objective of this study was to investigate the combined effect of inorganic N fertilisation and cover crop-maize rotation in DIN leaching.

STUDY SITES AND METHODS

This study was carried out in a temperature-controlled glasshouse with undisturbed soil columns packed in PVC tubes (0.2 m diameter, 0.5 m long) in August 2015. The columns were taken from the Antumapu Experimental Farm (Santiago, Chile) in a soil classified as Entic Haploxerolls, with loam to loamy sand textural class, pH_{water} = 8.6 and total organic C content of 0.65%. A combination of 13 treatments (Table 1) with four replications were evaluated considering combination of: N doses of 250 or 400 kg N ha^{-1} for maize (*Zea mays*) or 0 or 150 kg N ha^{-1} for cover crops, crop rotation (maize-fallow, maize-cover crop, cover crop or fallow) and cover crop species (*Lolium multriflorum* or *Trifolium repens*). For maize a dose of 250 kg N ha^{-1} corresponds to an optimum N dose, whereas 400 kg N ha^{-1} represent an excessive N dose usually applied by farmers in Central Chile. Treatments with fallow had bare soil during the autumn-winter period (April to September), while maize is cultivated during spring summer (October to March). The first season with maize was set from October 2015 to March 2016, but samples were not taken because the soil columns were in a setting stage. The maize was harvested in March 2016. From April

2016 to January 2017, water that percolated from the columns after irrigation events was measured and a subsample saved for analyses of NO_3-N concentrations using colorimetric methods. Nitrate loads were calculated as the volume of percolated water times the NO_3-N concentration and converted to kg NO_3-N ha^{-1} considering the column area.

RESULTS

During the autumn-winter period the treatment fallow (F) and fallow-maize with 400 kg N ha^{-1} (Zm–F (400 N)) showed the highest N loads, with significant differences ($p< 0.05$) with treatments with cover-crops an lower N applications (Table 1). This can be explained due to the higher water percolation in fallow treatments compared to cover-crops, with transpiration of the cover crop reducing percolation. In addition, cover crop acted as a large N sink in autumn-winter reducing NO_3-N in the soil solution.

Table 1. Mean nitrate–nitrogen (NO_3–N) loads in the different treatments

Treatment[1]	Season[2]			
	Autumn-winter		Spring-summer	
	-------------------- kg NO_3-N ha^{-1} --------------------			
F	154.85 ± 26.91	a	204.7 ± 88.84	ab
Lm (0 N)	3.88 ± 4.44	e	41.71 ± 29.12	b
Lm (150 N)	6.45 ± 4.58	e	74.1 ± 31.36	ab
Tr (0 N)	82.16 ± 11.30	c	107.94 ± 31.36	ab
Tr (150 N)	95.90 ± 13.31	abc	187.91 ± 88.84	ab
Zm–F (250 N)	118.36 ± 17.41	abc	41.46 ± 29.12	b
Zm–F (400 N)	157.37 ± 27.73	a	340.45 ± 144.4	a
Zm–Lm (250 N)	53.56 ± 8.03	d	42.81 ± 29.12	b
Zm–Lm (400 N)	99.91 ± 16.13	abc	245.76 ± 88.84	ab
Zm–Tr (250 N)	122.97 ± 18.39	abc	122.11 ± 29.12	ab
Zm–Tr (400 N)	148.38 ± 28.77	ab	325.64 ± 88.84	ab
Zm–Lm+Tr (250 N)	139.32 ± 22.36	ab	52.19 ± 29.12	ab
Zm–Lm+Tr (400 N)	89.31 ± 14.21	bc	115.94 ± 29.12	ab

[1] F = fallow; Lm = *Lolium multiflorum*; Tr = *Trifolium repens*; Zm = *Zea mays*.

[2] Means ± standard errors (autumn-winter, n = 12; spring-summer, n = 8) within columns with different letters are significantly different (ANOVA, $p<0.05$).

During the spring-summer period the treatment Zm–F (400 N) showed the highest N load, with significant differences ($p< 0.05$) with treatments with maize and lower N applications (Zm–F (250 N) and Zm–Lm (250 N)) and treatment with permanent *Lolium multiflorum* as cover-crop (Lm (0 N) (Table 1). Although the biomass production and water transpiration were similar in treatments with maize, the maize treatment that received an excessive N application (400 kg N ha^{-1}) had higher residual NO_3-N available for leaching.

CONCLUSIONS

During the autumn-winter period, the treatment with fallow and maize with excessive N application showed higher NO_3-N loads compared to cover-crop treatments due to the higher water percolation generated in a bare soil, and because cover crop acted as a large N sink in autumn-winter reducing NO_3-N leaching. During the spring-summer period, the treatment with fallow and maize with excessive N application showed higher NO_3-N loads compared to maize with optimum N dose and permanent cover-crop, because the N overfertilisation promoted higher residual NO_3-N available for leaching.

ACKNOWLEDGEMENTS

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Session IX: Socio-economical factors in soil and water conservation

9.1.O

IMPACT OF ADVISORY SERVICES ON WATER AND SOIL CONSERVATION – A CASE STUDY FROM HESSE GERMANY

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INTRODUCTION

Diffuse pollution with nitrogen and other pollutants from agricultural sources is a problem for water quality in many parts of the world. There are many measures along the agricultural production chain to reduce these emissions, of which several are relevant for soil conservation as well.

The environmental costs incurred by agricultural nitrogen emissions into the water bodies and other environmental media, such as biodiversity decline, are external costs to farmers. Therefore economic incentives for farmers to increase nitrogen efficiency are generally not sufficient as far as society as a whole is concerned. Hence policy interventions are necessary to internalize at least some of the environmental costs or to provide other incentives to reduce the use of nitrogen, or to improve its allocation. In addition to regulatory and economic measures, information and advisory services are used as essential policy instruments. In 2012 14 EU member states reported to have implemented advisory services in the context of the EU Water Framework Directive (EC, 2015) and the European Commission is promoting advisory services concerning sustainability in general (EC, 2012) and water protection specifically (Regulations (EU) No 1305/2013 and No 1306/2013). The question arises whether such advisory services are effectively reducing pollution and whether they are cost-effective. There are only a limited number of studies on this question with no comprehensive conclusive results.

Therefore, a case study of the Water Framework Directive (WFD) advisory service in the WFD river basins in Hesse, Germany was conducted. This investigated advisory service is restricted to measures on crop land and started in the different river basins between 2010 and 2011. The objective of the case study was to test the applicability of a framework of indicators for effectivity and cost-effectiveness of advisory services. Additionally the study adds to the few studies on the impact of agricultural advisory services on water protection and expands the basis for potential meta-analyses.

MATERIALS AND METHODS

A socio-economic analytical framework was combined with bio-physical indicators:

I, Aggregated nitrogen balances as representative target indicators. While in their available form they would have little meaning, in the context of the following indicators they could be evaluated.

II, Management measures taken by farmers as first-order process indicators.

III, Increase in farmers' problem awareness and knowledge as second-order process indicators.

IV, Different types of benefits farmers derive from taking part in the advisory service, and social norms that may be influenced by the advisory services.

The indicators were tested in a quasi-experimental research design (Rossi, 2004) with comparisons between an intervention and a control group. The quasi-field experiment was carried out by a selfadministered questionnaire survey with responses from 1477 advised and non-advised farmers (return rate 33 %). In addition to the questions on the process indicators, contextual questions such as the perceived quality of the advisory service and characteristics of the farms and farmers were addressed in the survey.

The quantitative data was analysed descriptively and statistical correlations between variables were determined with Cramer's V or Spearman's rank correlation coefficient, depending on the scale of measure. We used the significance level of $\alpha=0.05$.

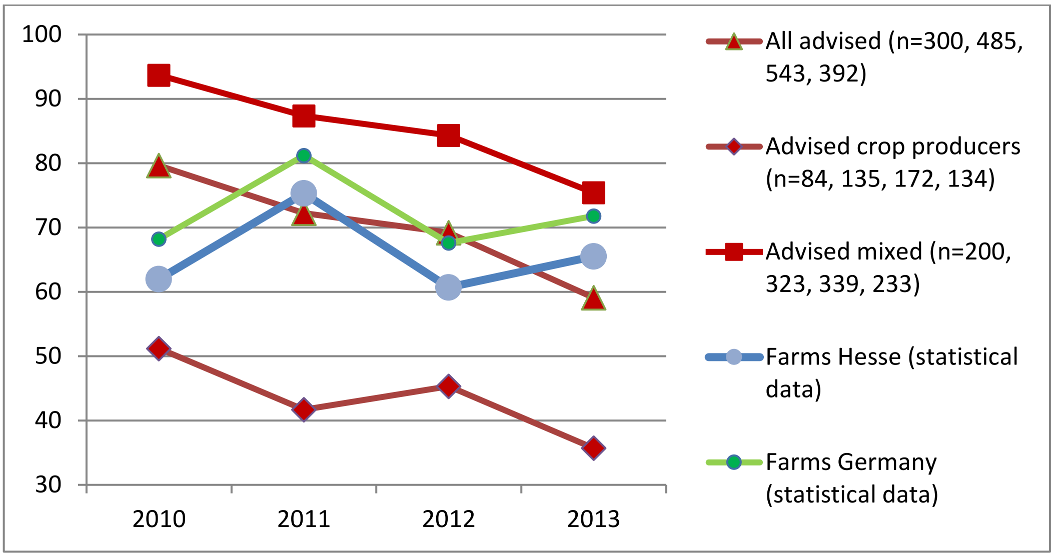
Important information for designing the questionnaire was gained from predominantly qualitative interviews with farmers, advisors and representatives of the authorities, alongside a theoretical model and other empirical studies. The theoretical model was an application of the concept of plural rationalities (Vatn, 2005) with regard to advisory services.

Furthermore, the qualitative interviews carried out before and after the written survey permitted more comprehensive interpretations of the questionnaire survey and a deeper understanding of the context. In addition, nitrogen balances of advised farmers aggregated by farm types were evaluated.

RESULTS

The aggregated nitrogen surpluses of intensely advised farms (HLNUG, 2016) showed a continuous downward trend from 2010 to 2013 from 80 to 60 kg N/ha (Figure 1). There is no data from comparable non-advised farms. At least, statistical estimates on the nitrogen surpluses of Hesse and Germany as a whole (LiKi, 2016) are available. Their absolute values are not directly comparable because of methodological differences, but their trends show that nitrogen balances are not declining in agriculture as a whole. This difference in N-surplus development of advised farms and the whole agricultural sector suggests, but does not prove, an influence of the advisory services.

Figure 1: Developments in the N-surpluses of intensively advised farmers in the WFD context (sample sizes of the WFD advised farmers for 2010, 2011, 2012, 2013 as "n = ..." in the legend) and statistically estimated N-surpluses for Hessian and German agriculture



Sources: HLNUG, 2016; LiKi, 2006

To narrow the influence of the WFD advisory services down, the following indicators along the functional chain of the advisory service were drawn into account.

Concerning **management measures** taken by farmers, for example 71 % of advised farmers indicated that they grow cover crops and that they use reduced tillage methods while this was only indicated

by 51 and 47 % of non-advised farmers respectively. Six to 14 percentage points more advised farmers said that they carried out respectively one out of seven different fertilization planning measures, such as taking Nmin samples from the soils. Advised farmers also performed the fertilization planning measures more frequently than the control group.
As expected, the data showed that the differences in whether or not these measures were implemented cannot be triggered exclusively by the WFD advice. However, the amount of added nutrients, plowing the soil, the choice of varieties and the share of land cultivated with catch crops were important management details influenced by the advisory service, according to farmers.
Concerning increased **knowledge** transferred by the advisors, a core result is that two-thirds of the responding advised farmers (n = 154) stated in the written survey that the advice gave them insights that are highly relevant for their competency and motives to act water-friendly. For example they learned about the "nitrogen release and loss potential of individual activities". This is reinforced by the fact that the vast majority of the advised farmers (86% of the advised, n = 162) say that they benefit from the advisory service in terms of new knowledge concerning water conservation.
Moreover, the great majority of the advised farmers reported different **benefits** from the advice, partly serving as incentives for adopting water conserving measures. In particular, reducing fertilizer spending, named by four out of five respondents, is directly attributable to water-friendly actions, because in this context it is only possible by reducing the amount of applied fertilizer.
Approximately one-third of the advised farmers believe that their benefit from the consultation will increase in the future and around half think that this might happen, suggesting that their actions will improve as well. The interviews showed that this is because **time** is needed for confidence building, observing successful trials over several years, and the diffusion of insights among farmers.
Finally, the case study provides evidence that advice can strengthen **social norms**, add to their internalization or even contribute to their new formation through group events. This became especially evident in regard to Nmin-test-results from the farmers' lands, which were discussed in groups more or less anonymously and motivated the farmers to avoid high values.

By analyzing the advisory impact on the different process indicators (actions taken, problem awareness and knowledge, benefits and norms), the picture shows that, on average, the nitrogen surpluses can be reduced by a few kilograms per hectare per year, given correspondingly high surpluses. This is all the more true in the long-term perspective when considering that the investigated advisory service had only been operating for one to three years when we took the data. On the basis of this assessment, it became possible to interpret the nitrogen surpluses of the intensely advised farmers as representative target indicator. To allow further data interpretations, in a final step of the analysis we developed scenarios of the possible impact of the advice on the nitrogen surpluses. The basic trend of the nitrogen surpluses of intensively advised farms was -6.85 kg N/ha per year; this is the weighted average decrease in these balances from 2010 to 2013. We assumed that the advisory services had a share of at least 30 % in this reduction (2 kg N/ha), considering that the overall balances of the agricultural farm land of Hesse and Germany did not decrease and the results for the process indicators imply that there has to be at least an impact in that size for the intensely advised. As the maximum scenario we assumed that the advisory services had 80 % share in the reductions, seeing how on the one hand these 5.5 kg N/ha are easily achievable through the actions indicated by the study and on the other hand a 100 % effect cannot be assumed because we did identify a small selection bias. That is, on average the advised farmers were a bit better educated and had more land (Cramer's V between 0.1 and 0.2). The medium scenario assumed the mean value of the shares of the advisory services in N-surplus reduction in the minimum and the maximum scenarios, namely 55 %. In addition, the scenarios assumed that respectively 500, 1000 and 2000 additionally advised farmers reduced their nitrogen balances by 1 kg N/ha to take account of the fact that the advisory service is by no means confined to the intensely advised: An additional 6827 farmers were advised extensively by the end of 2013 in Hesse.

Public costs are estimated at around 4.2 million € per year for the advisory services. This is the amount of funds used in 2013 (HMUKLV, 2016). The farm costs are not taken into account, only the public expenditure. However, the case study results imply that farmers have generally not incurred any major costs with these reductions, and they are also likely to achieve some cost savings. Also, the public costs already included sampling and analyses of soil, manure and plant material.

Finally, the cost effectiveness of the advisory service was calculated to be between 8 and 26 € per kilogram of nitrogen surplus prevented, while the medium scenario was 12 €/kg N.

CONCLUSIONS

The results of the estimated reductions in nitrogen balances and cost effectiveness have to be interpreted in the light of the short duration of the advisory service of one to three years at the time of the study and the indications of the results that the effects of an advisory service need more time to reach their maximum.

The estimated values of reduction of nitrogen surpluses can over several years significantly contribute to reducing the German agricultural nitrogen surplus of 95 kg N/ha (total nitrogen surplus, Federal Statistical Office 2016) towards the short-term sustainability target of 80 kg N/ha (ibid.).

Furthermore, the results in figure 1 show that the cost-effectiveness of the advisory service could be increased if its contents were to include the livestock production, as the reduction potential is still high in mixed farms with livestock while arable farms already move in the direction of inevitable surpluses.

The effects are not restricted to water conservation. Nitrogen balances are a good integrative indicator with implications also for biodiversity, greenhouse gas emissions and human health.

There are also synergies of the advisory service with other policy areas that go beyond the reduction of N emissions. For example, growing cover crops can improve the soil structure, reduce soil erosion, contribute to C sequestration and promote biodiversity, depending on the specific design.

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9.2.O

FORESIGHT FOR AGRICULTURAL SOIL MANAMGENT AND PRESSURES ON SOIL FUNCTIONS

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INTRODUCTION

Soil management is the key pressure that shapes agricultural soil quality, soil functions and soil services. Soil functions are suffering from degradation processes in many parts of the world. Taking appropriate actions for soil conservation requires the identification of drivers and trends of future soil management and the challenges and opportunities these offer for shaping a sustainable future.

There are both global and local/regional drivers of soil management, and global drivers evoke regionally differentiated trends on soil management. This is why we took a regional approach with Germany as example case to assess the drivers and trends of agricultural soil management and the resulting challenges and opportunities for soil functions.

MATERIALS AND METHOD

The conceptual starting point of the analysis was the DPSIR (Driver-Pressure-State-Impact-Response) framework introduced by the European Environment Agency (Smeets and Weterings, 1999). We analyzed firstly which factors drive soil management to what end in general and secondly how these drivers are coined in Germany. From this we derived assumptions for potential developments in soil management in Germany.

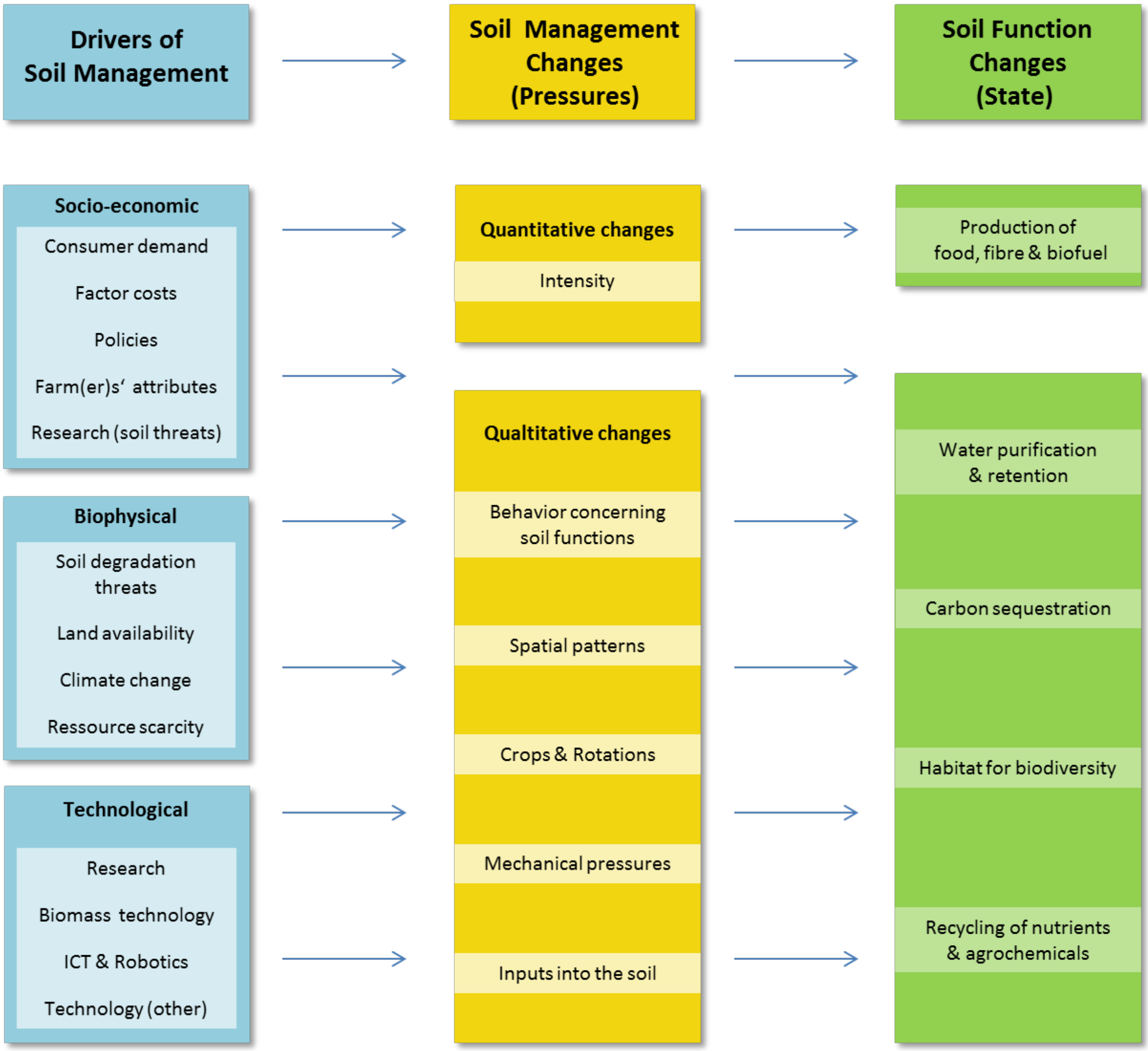
Between January and November 2016 we reviewed a total of 259 sources. The sources included scientific evidence with priority given to peer-reviewed publications. Governmental sources, such as farming statistics and laws, were used to analyze the characteristics of the drivers, e.g. the age structure of farmers. For public debate on a driver or pressure, three relevant German magazines (Top Agrar, DLG-Mitteilungen, Agra-Europe), directed at different stakeholder groups, were sighted and selected according to the analytics of the readers of several magazines (Schleyerbach, 2009), their own descriptions of their target groups and the authors' experiences. Other online information for different stakeholders was taken into account to analyze either the public debate, e.g., a statement of an important lobby group, or to analyze market and industry trends when there was no current scientific analysis available.

During February and March 2017 we have conducted 14 semi-structured interviews with experts from science (soil science including soil ecology and biodiversity, agronomy, agricultural systems technology, plant sciences), agriculture and authorities to validate and complement our review results. The interviews are being transcribed. They contain some structured questions that are evaluated quantitatively and some open questions that are evaluated with qualitative content analysis. First results from this have been included here.

RESULTS

A complex net of the drivers and trends of soil management was identified, shown simplified in Figure 1. We analyzed socio-economic, biophysical and technological drivers and identified two modes of future management changes: quantitative and qualitative changes.

Figure 1: Overview of the identified drivers of soil management, categories of soil management (pressures) changes and soil functions (state) that are affected by soil management



Quantitative changes affect soil management in terms of decreased or increased biomass production. Ceteris paribus, the quantities of production factors are changed, e.g., adding less/more fertilizers or de-/increasing harvest frequencies. Globally a need for intensification of production is anticipated mainly due to population growth and changing diets. For Germany this can be expected to a much lesser degree, especially because production intensities and technological standards are already high and marginal utilities of input factors are correspondingly low.

We assigned qualitative changes of soil management to five categories of agricultural practices:
I, Behavior concerning soil functions in general:
Improved recognition of soil multifunctionality in soil management can expected, mainly due to the change in farmers' attributes, including age structure and qualification, consumer demand in combination with new technologies and research.
II, Spatial field patterns:

A trend towards larger fields and fewer transition zones is still visible. If in the coming decades small, autonomous machines become implemented, for which the technology is currently being developed, this will open opportunities for more small-scaled field patterns. This would also open up new opportunities for policy incentives such as agri-environmental measures which already promote landscape elements.

Spatial patterns of crops within fields may change towards more intercropping, e.g. of cereal and legumes or of agricultural crops with woody plants (agroforestry). Both are seen by some experts to have potential for a relevant extent of implementation due to increasing research results that show better use of soil functions with these systems and the likely adoption of smaller, autonomous machines with high precision abilities in the future facilitating these systems. Yet, some experts are rather skeptical that intercropping and/or agroforestry will gain relevant distribution in Germany.

<u>III, Crops and rotations:</u>

Integration of lignocellulosic crops is likely to gain relevance with technological improvements to use them for energy and fiber. Yet, some experts assume that this will only gain a relevant extent if policies set incentives for this. Crops are being bred continuously towards improved performance under changing conditions like climate change while, according to experts, interactions with the soil are not sufficiently addressed. Crop rotations are already becoming slightly more diversified with increased shares of cover crops and this trend is expected to keep going on. Main reasons for some experts are policy incentives due to citizens' demand. Others assume that in German regions in which crop rotations are narrow, such as rapeseed-wheat-wheat, problems with pests and their resistance towards pesticides are already initiating an emerging trend towards diversified crop rotations, supported by a diversifying consumers' demand.

<u>IV, Mechanical pressures:</u>

While there is still an increasing pressure on soils from heavy machines, some emerging technologies are likely to reduce weight and contact stresses. It becomes possible to do tillage more site-specific as a function of soil texture and water content. More reduced tillage and specific forms of it like strip-tillage and controlled traffic farming are expected by some experts, possibly being hampered by restrictive pesticide policies. Improved subsoil management, including the cultivation of deep rooted crops, is still in the research stage but gaining recognition. Technology and algorithms for the automated optimization of routes on the fields, loads and tire pressure will become available soon and likely to be wide-spread within the next 20 years. Within 20 years it can also be expected that small autonomous machines will become relevant in German agriculture. While these machines are already in development, and the automation would decouple machine size from labor costs, their implementation is still insecure.

<u>V, Inputs into the soil:</u>

Application of fertilizers and pesticides is expected to become more precise, following an ongoing trend and further technological development. This includes (on-the-go) sensors for soil characteristics, data fusion algorithms, translation into decision-support systems, infrastructure with GNSS (global navigation satellite system) and mobile networks, robots and drones, which are all under development. Concerning pesticides aside from their application, an ongoing further development and more specific pesticides are expected while regulation on pesticide use is very insecure. The use of natural enemies of pests may reduce pesticide needs in the future. Concerning organic matter input, even though there are drivers pointing to opposing directions, it does seem like the relevance of organic input is gaining broader awareness and this may lead to actions counteracting organic matter decline in places where it is relevant. There is a clear trend towards recycling nutrients for fertilizers while it is still unsure when and what kind of change exactly will occur. The inoculation of soils and seed with microbes and natural enemies of pests are likely to become marketed as sustainable solutions for soils management and crop production. While their impact cannot yet be sufficiently anticipated it seems probable that positive impact like reduced pesticide needs, will depend on other practices promoting the inoculated and native organisms, like increasing soil organic matter. It is likely that irrigation will be increased in Germany as a reaction to climate change in the coming decades.

CONCLUSION

There are great opportunities for agricultural soils posed especially by the technological development towards including smaller and lighter machines and more precise management, by research that is increasingly uncovering positive effects of some soil improving production methods on soil quality and yield development, and by a societal will to support sustainable production methods or subsidiary aspects such as the SOC content of agricultural soils. At the same time challenges stem largely from costs of soil-friendly measures and new technologies, as well as possibly from a moderate agricultural intensification in Germany.

Some challenges will be met and opportunities seized by farmers and the farming industry. However, the realization of opportunities is not self-evident. Some options need to be further developed and some changes need to be initiated or supported by policy measures. Agricultural and soil research can play a vital role, on the one hand, in developing sustainable and effective methods of soil management in this framework of drivers, for example, to manage the subsoil sustainably and avoid subsoil compaction or to use inoculation with microbes and natural enemies of pests to reduce pesticide use. On the other hand, the analysis showed that for many upcoming management practices little scientific evidence exists about their effect on soil functions, especially on the habitat for biodiversity function. Thus it is crucial for researchers from different disciplines involving soil scientists, agricultural scientists, natural scientists, and socio-economic scientists to cooperate in analyzing the unfolding management changes, their alternatives and their impacts on soil functions so that they can provide evidence for sound suggestions to farmers, authorities, politicians and society. Basic and applied research must also work together to understand the value of soil functions for societal value systems, particularly in terms of ecosystem services, resource efficiency and ethical and equity considerations.

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9.4.O

BARRIERS AND DRIVERS FOR ADOPTION OF NON-INVERSION TILLAGE IN FOUR EUROPEAN COUNTRIES

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INTRODUCTION

Researchers largely agree on the benefits of soil conservation measures for soil quality and erosion reduction. However, adoption rates of some of these measures, including non-inversion tillage (NIT), remain low in many European regions. To promote adoption, we need a better understanding of the drivers and barriers that motivate or prevent farmers to implement these practices. The drivers and barriers can be very diverse in nature ranging from natural and technical to human and socio-economical influencing factors. The EU FP7 Catch-C project has, therefore, conducted a large scale farm survey to identify the drivers and barriers of several agricultural practices across seven European countries. In this contribution, we present barriers and drivers for implementing NIT as found in eight zones with different farm structures and pedo-climatic conditions (Farm Type Zone; FTZ) from four countries, i.e., Belgium, the Netherlands, Germany and Italy. More details of the study can be found in Bijttebier *et al*. (in preparation).

SUMMARY OF METHODS

The study areas were chosen from the European Farm Type Zones (FTZs) as identified by Hijbeek et al., 2013. These FTZs are a combination of land use, farm specialization and agri-environmental zones. Agri-environmental zones are defined based on climate, slope and soil texture. Selection of FTZs in each country to be included in the farm survey was done by experts using criteria such as the area of the FTZ, the economic importance of agriculture in the FTZ and soil degradation problems caused by farming activities in the FTZ. Characteristics of the eight selected FTZs are summarized in Table 1.

The methodology used is based on the Theory of Planned Behaviour (TPB) that states that individual beliefs about a behavior or practice are believed to determine the intention and behavior (Ajzen, 1988; Ajzen, 1991). Our approach consisted of a qualitative step followed by a quantitative step and was applied in each FTZ independently from the other FTZs as it was believed that potential drivers and barriers are specific for the biophysical and farming conditions in the region.

In the first step, 5 to 9 farmers were interviewed in each FTZ in order to make a complete list of potential drivers and barriers for the application of non-inversion tillage. Non-inversion tillage (NIT) was defined by the researchers as 'a tillage system in which the soil is not turned. Alternatively, it is called ploughless cultivation. A farmer applies non inversion tillage if he does not turn the soil on a particular field plot for a least an entire year while he sows at least one crop during that year.' Only in Italy, the definition was deviating, it stated that it was tillage at reduced depth without the requirement that there was no ploughing for an entire year.

***Table 1**. Summary of the characteristics of the eight Farm Type Zones used in the study*

	Farm specialization	**Land use**	**Soil texture (JRC classification)**	**Slope (%)**	**Climate**
BE_A	arable	specialised crops	medium fine	2%	Atlantic Central
BE_D	dairy cattle	maize, grassland	coarse	0%	Atlantic Central
GE_A1	arable+mixed	maize, grassland, cereals, potato	coarse	0%	Atlantic North
GE_A2	arable+mixed	specialised crops	medium fine	2-6%	Continental
IT_A1	arable	cereals and other arable crops	coarse, medium fine	0%	Mediteranean North
IT_A2	arable	cereals, other arable crops, grassland	medium, medium fine	3-14%	Mediteranean North
NL_A	arable	specialised crops	coarse	0%	Atlantic North and Atlantic Central
NL_D	dairy cattle	maize, grassland	coarse	0%	Atlantic North and Atlantic Central

After the qualitative phase, a quantitative questionnaire was derived from this list using the TPB framework. This framework groups potential drivers and barriers in three groups, being outcomes, referents and control factors. For each group two questions were asked. For each outcome, its likelihood (belief strength) was assessed using the question, e.g., "What is the likelihood that NIT reduces soil erosion?", which was scored by the farmer on a scale from 1 (very unlikely) until 5 (very likely). Its subjective evaluation was assessed by letting the farmers evaluate the outcome (i.e., less erosion) on a scale from 1 (very bad) to 5 (very good) (outcome evaluation). For each referent (e.g., other farmers), the farmer was asked to which degree the referent was positive or negative towards a practice, on a scale from 1 (very disagree) to 5 (very agree) (normative belief) and to which degree he values the judgment of the referent, on a scale from 1 (very unimportant) to 5 (very important) (motivation to comply). For each control factor (e.g., availability of the appropriate machinery), farmers were asked to which degree it makes NIT attractive/difficult, on a scale from 1 (unattractive/difficult) to 5 (attractive/easy) (control power) and to which degree that control factor is valid on the farm, on a scale 1 (not true) to 5 (true) (control belief strength). Besides more general information on the farm (e.g., farm size, crop rotation), farmers were also asked if they adopt NIT or if they have the intention to apply the practice in the near future. The questionnaires were sent in 2013 to a large number of farmers (150-1100/FTZ), selected randomly from a complete database (Belgium, the Netherlands) or sent through farmers' associations, farmers' extension services or other contacts (Germany, Italy).

RESULTS

The response rate of the questionnaires varied largely between countries, i.e., from 8-9% in the Netherlands, 13-21% in Germany, 27-28% in Belgium, to 59% in Italy. The number of completed questionnaires received per FTZ was between 57 and 173. Adoption rates of non-inversion tillage vary greatly between FTZs, but even more between countries (Table 2). The highest adoption rates were found in Germany (68-84%), followed by Italy (41%) and Belgium (19-23%). The adoption rate in the Netherlands was more variable between the studied FTZs; the dairy farmers had a lower adoption rate (26%) than the arable farmers (54%). The positive intention rate to apply NIT in the near future was higher than the adoption rate in Belgium and the Netherlands but lower in Italy and Germany.

Table 2: *Adoption rates and rates of positive intenders* to adopt non-inversion tillage in the near future*

	BE_A	BE_D	GE_A1	GE_A2	IT_A1	IT_A2	NL_A	NL_D
Adopters (%)	23	19	68	84	41	41	54	26
Positive intenders (%)	36	32	60	76	38	38	67	43

*having an average score of > 3 on three similar questions whether the farmer intends to apply non-inversion tillage in the near future on a scale of 1 to 5 (unlikely/not true-likely/true)

Beneficial effects of NIT as observed by farmers across FTZs.

In all countries, farmers are aware of beneficial effects of NIT. In Italy, the number of perceived positive outcomes of NIT is rather limited and is related to less labour needed, reduced tillage costs and reduced risks for water logging. Less labour/higher working efficiency and reduction of tillage costs/lower use of fuel were also perceived as potential drivers in Germany, the Netherlands and Belgium. In these countries farmers also believe that NIT can have positive effects towards soil quality and the prevention of soil degradation. More soil fauna, better soil structure/increased physical soil quality, increased moisture holding capacity, prevention of plough pans, nutrients in the top layer and less erosion are mentioned as positive outcomes. Dutch farmers also agreed on increased soil carbon contents as a beneficial outcome, but this opinion was not shared by Belgian (BE_A) and German (GE_A1) farmers. The freezing of remaining potatoes during winter was another benefit mentioned by Belgian and Dutch arable farmers. Belgian farmers believe that adopting NIT is more easy when subsidies are provided and when they can sow cover crops in August. Despite the fact that subsidies can be a driver for adoption, Belgian farmers also mention that the prerequisites for obtaining the subsidies are a barrier.

Negative effects of NIT as observed by farmers across FTZs.

A shared concern across the European FTZs, except for GE_A2 where it was not mentioned, is more weeds/increased herbicide use/more difficult elimination of weeds. In several FTZs, except in Italy, higher risks for transfer of crop diseases and pests were also mentioned as negative outcomes of NIT. Lower yields are a concern in the Belgian FTZs and IT_A2, but lower yields were not regarded to be very likely in the Netherlands, IT_A1 and GE_A1. Farmers in Germany and/or Belgium mentioned additional negative outcomes related to cultivation techniques (bad seedbed/crop germination, slow warming up of the soil in spring) and soil physical quality (more soil compaction). Belgian and German (GE_A2) farmers perceive the fact that NIT leaves a less esthetic beautiful or uneven field also as being a negative outcome, which reveals that also more cultural factors can play a role in behavior. Other farmers (IT_A2) on the other hand mention that they do not aim to have a nice looking field (IT_A2) or they do not agree that NIT leads to 'uneven' fields (GE_A1). In line with the lowest adoption rates in this European study, Belgian farmers additionally mention another set of barriers, such as having no experience or not enough technical knowledge, having an inappropriate crop rotation and grass as preceding crop which would need to be destroyed chemically when applying NIT, and having good results with mouldboard ploughing. Finally, Italian farmers (IT_A2) belief that it is more difficult to apply NIT on clay soils.

Influence of social environment.

Like any other citizen, farmers are embedded in a social environment, influencing their decisions and their beliefs. Except in Italy, farmers have a high motivation to comply with other farmers (average score of 3.3-3.7). In dairy farms in Belgium and the Netherlands and the German FTZs, other farmers are perceived to be rather negative towards applying NIT and can therefore act as a barrier for adoption. The opposite is true for the arable farmers in the Netherlands who perceive other farmers to be rather positive towards the practice. The high motivation to comply with other farmers illustrates the importance of farmer-to-farmer interactions with adopters who successfully apply the practice. In Belgium and the Netherlands, farmers value literature and/or the results of experimental fields as well as the opinion of researchers and experts. Research and experts are in these countries perceived to be rather positive towards NIT and in the Netherlands, arable farmers feel also supported by literature. That is in contrast with the Belgian farmers who do on average not agree to have seen positive results in literature (BE_A) and on experimental field trials (BE_A+D). Despite several (demonstration) projects supported by the provinces and the Flemish Government in the FTZ and financial support through agro-environmental schemes (mainly aimed at using NIT for erosion reduction), Belgian arable farmers on average feel that the Flemish government and extension from the province are not positive towards NIT. Farmers in all countries have a high motivation to comply with extension services/agents (average score > 3.5). In the Netherlands (NL_A) and Italy (IT_A1), extension is perceived to be slightly positive towards NIT (average score of 3.1) while the opposite was the case for the Belgian FTZs (BE_A:2.9-BE_D: 1.72) and GE_A1 (2.84).

CONCLUSIONS

This study, based on the theory of planned behavior, gained insight in the adoption rate and main motivations and barriers for adopting non-inversion tillage (NIT) in 8 farm type zones across 4 European countries. Widely shared positive outcomes include the belief that NIT leads to less labor and fuel needs and thus reduced costs, while widely accepted barriers are related to more problems with weeds. Other drivers and barriers mentioned are diverse varying from natural and technical to human, cultural and socio-economical influencing factors and are region and context specific. This study shows that motivations of farmers whether or not to adopt a practice is dependent on many influencing factors and the insight in these factors should allow policy makers, extension services and researchers to better target their work if they want to increase adoption rates in a given region.

ACKNOWLEDGEMENTS

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9.6.O

ASSESSMENT OF CRITICAL THINKING SKILL IN SOIL GENESIS SUBJECTS BY MEANS OF LESSONS TOOL (SAKAI): ACTIVITIES FOR IMPROVEMENT

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INTRODUCTION

The main objective of the Universitat Politècnica de València (UPV) is to ensure that its students acquire the necessary skills to be able to develop correctly in the performance of their work as engineers or graduates (UPV, 2015). This is the main reason why it is developing an intense work to apply and evaluate transversal skills in their degrees. Thirteen transversal skills have been selected and applied at all university levels. Critical thinking is one of them, and an institutional team has developed it. The authors of this article belong to this team who began to work in the 2012. Professors make up the team from different disciplinary fields who share common objectives of innovation and educational research and who are an engine of change and generation of knowledge about education in the UPV. This group knows that students are the active protagonist of their learning process and professors become mere guides. The introduction of this new educational paradigm has led to restructuring subjects, methodology and materials that are used by professors. Critical thinking is the process of discrimination and judgment process of "the truth" that each individual performs according to its previous knowledge and opinions (Norris and Ennis, 1989). Currently the higher education system establishes that transversal skills are an indispensable requirement to obtain the corresponding degree. By this reason, professors must work, evaluate and encourage transversal and specific skills. The main goal tries to help the students to work and develop these skills. Several studies have been carried out in the university about skills: Learning based on problems (Saiz and Rivas, 2012), work in group (Andreu-Andrés and Garcia-Casas, 2015) or critical thinking applied in medicine and nursing among others. However, the studies on the application of this skill in subjects of soil genesis are practically non-existent. Due to this, the following study aims to facilitate the assessment of critical thinking skill by means of two types of activities, included in an advanced web tool developed by Sakai (Lessons).

MATERIAL AND METHODS

The experience was carried out on 220 students of the Degree in Agri-Food Engineering and Rural Environment, and the Degree of Forestry and Natural Engineering. More specifically, they were enrolled in the Geology, Soil science and Climatology subject that it is taught at the first course of the Higher Technical School of Agronomic Engineering and Natural Environment (ETSIAMN) at the UPV. To reach the acquisition of the skill, professors developed several activities that were submitted in an institutional online platform: *PoliformaT*. This platform allows professors to use *Lessons* Tool. A *Sakai* tool that facilitates the sequencing of concepts and teaching tasks. In addition, Lessons allows introducing a test for auto evaluation by students about their impressions of skill achievement.

Professors structured the tasks according to theory concepts and explained what the students had to do with a task submitted in *Lessons.* In addition, the students had the possibility to see the rubric that the professors had developed for correct the activity. This rubric was submitted in *Lessons* at the same time that the activity.

The first activity "Soil Genesis" consisted of identifying the concepts developed in class that a soil expert explained in a lecture. Students should attend the conference and after it, they must fill a sheet with theory concepts and the conference ideas. The goal of this activity was focussed on the student, who had to show a critical attitude towards real phenomena and situations. The rubric for this first activity is described in table 1:

Table 1 – Rubric for activity 1

A. Excellent
Students reflect and investigate the why of things, and they find answers and argue objectively ***Students find contradictions and They can give a judgment based on evidences of other authors that They use as a bibliographic source***
B. Sufficient/good
Students ask themselves the reason of things, and investigate to get answers autonomously. They can be influenced on the judgments. ***They find contradictions and they usually give a judgment based on the majority opinions of others***
C. Developing the skill
They ask about certain situations in which they live. They are not capable of making judgments and evaluations according to their ideas. Needs help from others to get answers ***Explain the ideas of the lecturer, but they ind contradictions. However, They do not give any own judgment***
D. Deficient/Not reached
Students do not manifest a critical spirit: the situation is not questioned. Assumes as true any information that they receive ***They only write down the ideas of the lecturer***

The second activity "Geological Materials" showed a discussion between two students in class. Students should choose one of the two options and justify which was the correct one and which were the reference sources to confirm the solution selected. Two rubrics were developed for this activity (tables 2 and 3).

Table 2 – Rubric to assess the critical opinion

A. Excellent
Students detect inconsistencies and contradictions, give arguments, and they reformulate coherently the contradictory statements ***Students Identify the correct option and argue through evidences why it is correct. In addition They also reformulate the one that is wrong in the correct way***
B. Sufficient/ good
Students detect inconsistencies and contradictions, and hey provide arguments to demonstrate the error ***They identify and argue the right choice through evidence***
C. Developing the skill
Students can detect some inconsistencies, but they cannot explain why ***They Identify and detect the right option but they do not explain which is the reason which corroborate the statement***
D. Deficient/Not reached
Students cannot detect inconsistencies or contradictions in a text or speech ***They do not detect which is the correct argument***

Table 3 – Rubric to assess the quality of bibliographic sources

A. Excellent
Students consult scientifically reliable sources. They contrast the information and contribute with their personal valuations. ***Students delve into the subject and justify their reasons. They work with more than one bibliographic source, and these sources have proven their scientific reliability. In addition, students also contribute with a coherent opinion.***
B. Sufficient/ good
Students consult different sources and contrast or verify their reliability. They delve into the subject and justify it. ***They delve into the subject and justify the reason. They work with more than one reliable bibliographic source.***
C. Developing the skill
Students consult different sources, but they do not verify their reliability. They delve slightly into the subject. ***They work with more than one bibliographic source, but without contrast.***
D. Deficient/Not reached
Students are not able to delve into the subject. They use only a single source and they do not contrast the information. ***They do not delve into the subject and only work with one non-checked bibliographic source***

Both activities were planned with sufficient time to develop correctly.

RESULTS

The outcomes of the application of both activities on critical thinking evidenced that 63% of the students presented an adequate or excellent level of acquisition of the competition. 30 % had a developing level and only 7% did not reach the appropriate level of acquisition for their course.

From an individual point of view of each activity, note that in the first, more than 50% had not reached the skill or were in development (Figure 1). Possibly this was due to two reasons, mainly attributable to the design of the task:

- The level of soil expert was too high for students of the first course.
- The task was done at the beginning of the course and therefore the students had a partial view of the subject.

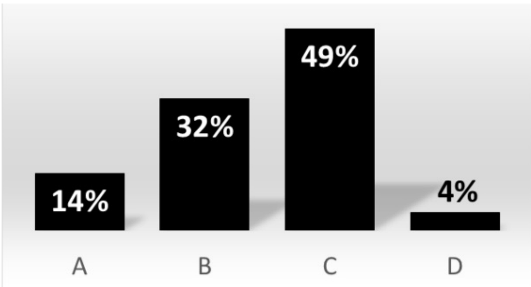

Figure 1 – Outcomes of activity 1. Levels of achievement of skill; A: Excellent; B: Sufficient/ good; C: Developing the skill; D: Deficient/Not reached

Regarding activity 2, note that 57% of the students showed an adequate or excellent level of skill achievement (Figure 2). This outcome indicates that students have developed critical thinking skill more easily to their level of proficiency. The activity was directly designed with two options, and therefore it showed a clear structure for the student.

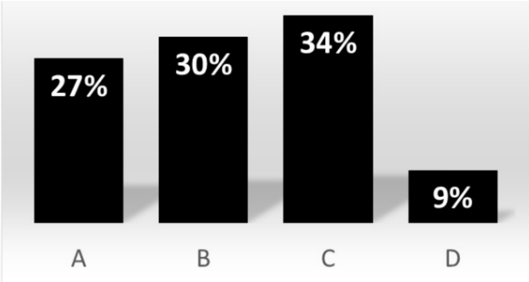

Figure 2– Outcomes of activity 2. Levels of skill achievement; A: Excellent; B: Sufficient/ good; C: Developing the skill; D: Deficient/Not reached

More than 70% of the students expressed that they had achieved a critical attitude during the auto evaluation test. However, only 46% and 57% in activity 1 and 2 respectively.

CONCLUSIONS

The main conclusion indicates that the design of activities is essential in the acquisition and improvement of critical thinking skill. Moreover, the use of *Lessons* can help students and professors to develop this crucial item in the learning process. For students of the first courses, it would be advisable to use directed activities in which there are more than two visions. It could be interesting to use doubts or questions that arise in class. Students of last courses would use the conference activity. In addition, it is necessary that students know their perception about the level of acquisition of skills before carrying out the activity and the final marks assessed by the professors. At the end, developing critical thinking skills in soil genesis is feasible with the combination of well-developed activities and the use of web tool platforms.

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9.1.P

MULTIFUNCTIONAL CHARACTERIZATION OF TROPICAL LIVESTOCK SYSTEMS IN MEXICO: IMPLICATIONS FOR PRODUCTION SUSTAINABILITY

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INTRODUCTION

Multifunctionality denotes a phenomenon associated with currently or potentially offers multiple material and immaterial "goods" that satisfy needs of society, is a concept with economic and social implications. The basis of multifunctional analysis is to understand how land uses affects functions of system and how multiple demands of society are to solve. For this reason, sustainable development of livestock production system based on perspective of multifunctionality, must integrate elements of environmental protection, social welfare and economic growth (Wiggering et al., 2006). Livestock is an important global activity that has a drastic impact on social, economic and environmental spheres (Thornton, 2010; Herrero et al., 2016). The global demand for livestock products is growing rapidly as a result of rising incomes, population growth and urbanization (Steinfeld et al., 2009; Alexandratos & Bruinsma, 2012), and combination of these factors, involved an unprecedented food and agricultural systems challenges (Steinfeld et al., 2009). Therefore, it was determined to perform a multifunctional characterization of livestock production systems in tropical ecosystems, in order to understand their potential as a source or sink of nutrients (C, N and P net contents), and to identify them explanatory association with climatic and physical-chemical properties and how it maintain ecosystem multifunctionality.

METHODS

Study site.
The study area was located in tropical dry forests of the Yucatan Peninsula. Three sites were selected in order to evaluate different livestock practices: Monoculture pastures (extensive pastures) (MCp), induced silvopastoral systems (monospecific) (SPind) and natural silvopastoral systems (SPn). Each site was named based on near locality (X'matkuil (XM), Tizimín (TI) and Tzucacab (TZ)), present a gradient of mean annual precipitation (MAP) (995 - 1250 mm), but similar values of mean annual temperatura (MAT), topography, geology and soils, and share the same vegetation development type and state (Cuevas et al., 2013). Field campaigns were carried out in May 2016 (drought season).

Nutrient concentration and physical-chemical characteristics.
The soil samples were sieved and a sub-sample was dried at 65°C for 24 hours. 0.05 g of soil was then weighed for subsequent determination of total Nitrogen (Nt) and total Phosphorus (Pt). That

determination was performed by acid digestion in concentrated $H2_SO_4$ thorough Kjeldahl method. Soil organic carbon (SOC) was obtained by the Walkley and Black method based on the oxidation of the active forms of SOC of soil organic matter (SOM). Soil pH was determined in sub-sample (10 g) of soil which was mixed with deionized water in a 1:5 ratio. To determine soil water content (SWC), 10 g of soil were dried at 65°C for 24 hours and expressed in % of moisture.

Structural Equations Model (SEM).

The SEM was performed to infer direct and indirect effects of climate (MAP and MAT), and soil properties (SWC, pH and clays) on ecosystem multifunctionality (EMF). For this, data on the following ecosystem functions were taken: carbon and nitrogen in microbial biomass (MBC and MBN, respectively), SOC, CO_2 emissions, mineral N, total N and P, which were determined for same season. The EMF index described by Maestre et al., 2012 was used. Z-scores were calculated for each function and then was averaged in order to obtain EMF (Jing et al. 2015).

Statistical analysis.

Analysis was carried out with R statistical package (R Core Team, 2016). Normality was contrasted with Shapiro - Wilk test and homoskedasticity with Bartlett test. Analysis of nested variance (nested ANOVA) was performed to determine differences within municipalities, among treatments (livestock practices), and vice versa. Statistical significance was set at a level of 95% (0.05α). The SEM was performed using the Lavaan package.

RESULTS

Nutrient concentration and physical-chemical characteristics.

The SOC for livestock in MCp was 20%, average in SPind was 19.3%, and in SPn was 19.6%; these differences were not statistically significant (Table 1). In contrast, SOC in the municipality XM was 30.58%, in TI was 21%, and in TZ was 20.8% (TI and TZ ≈ 10% < SOC), these differences were significant within XM and TI (Table 2). The total N concentration in MCp was 3843 $\mu g\ N\ g^{-1}$, in SPind was 3095 $\mu g\ N\ g^{-1}$ and in SPn was 3195 $\mu g\ N\ g^{-1}$, these differences were significant only within the MCp and SPind (Table 2), and within municipalities XM (4782 $\mu g\ N\ g^{-1}$) and TI (3054 $\mu g\ N\ g^{-1}$), but not within TZ (2952 $\mu g\ N\ g^{-1}$). The total P concentration in MCp was on average 286.6 $\mu g\ P\ g^{-1}$, in SPind it was 294.1 $\mu g\ P\ g^{-1}$ and in SPn it was 293.9 $\mu g\ P\ g^{-1}$, these differences were significant within all practices (Table 1), and in the municipalities XM (351.23 $\mu g\ P\ g^{-1}$) and TI (122.14 $\mu g\ P\ g^{-1}$), but not in TZ (247.67 $\mu g\ P\ g^{-1}$) (Table 2).

Table 1. P-Values obtained for nutrient concentration (SOC, total N and total P) and for physical-chemical characteristics of the soil (pH, SWC and Clay (%)), through nested ANOVA – Within livestock practices, among the municipalities.

	Monoculture Pastures (MCp)	**Induced Silvopastoral (SPind)**	**Natural Silvopastoral (SPn)**
	p-value	**p-value**	**p-value**
SOC	0.05004	0.2952	0.5141
Total N	$3.27x10^{-5}$ ***	$2.91x10^{-5}$ ***	0.4523
Total P	0.005795 **	$3.421x10^{-5}$ ***	0.0038 **
pH	$1.54x10^{-7}$ ***	$7.05x10^{-7}$ ***	0.0391 *

SWC	0.0469 *	$2.42x10^{-5}$ ***	0.0132 *
Clay	$4.18x10^{-8}$ ***	$9.37x10^{-5}$ ***	0.0003 ***

Soil pH, SWC and clays followed same patterns of variance. Soil pH in MCp was 7.91, in SPind was 7.70 and in SPn was 7.64, these differences were significant within all practices (Table 1), and within all municipalities (XM = 8.04, TI = 7.46 and TZ = 7.73) (Table 2). SWC in MCp was 20% average, in SPind was 26% and in SPn was 20.3%, these differences were significant within all practices (Table 1), and within all municipalities (XM = 29% TI = 21% and TZ = 19%) (Table 2). Finally, percentage of clays in MCp was 12.7% average, in SPind was 13.8% and in SPn of 16%, these differences were significant within all practices (Table 1), and within all municipalities (XM = 12.5%, TI = 12.8% and TZ = 17.16%) (Table 2).

Table 2. P-Values obtained for nutrient concentration (SOC, total N and total P) and for physical-chemical characteristics of the soil (pH, SWC and Clay (%)), through nested ANOVA - Within municipalities, among the livestock practices.

	X'matkuil (MAP: 995 mm)	**Tzucacab** (MAP: 1090 mm)	**Tizimín** (MAP: 1.250 mm)
	p-value	**p-value**	**p-value**
SOC	$2.82x10^{-8}$ ***	0.3271	0.0013 **
Total N	0.00018 ***	0.4129	0.0012 **
Total P	0.00256 **	0.2825	$6.03x10^{-6}$ ***
pH	0.0001 ***	0.0014 **	$3.07x10^{-5}$ ***
SWC	$1.64x10^{-5}$ ***	0.0304 *	0.0109 *
Clay	$4.20x10^{-5}$ ***	0.0001 ***	$1.22x10^{-7}$ ***

Structural Equation Model (SEM).

The SEM showed that the influence of the clays on EMF (β=0.395, standardized coefficient) was mediated through available precipitation (PMA). Temperature (TMA) had an influence mediated through pH on EMF (β=0.746, standardized coefficient), and this factor was the strongest relationship observed in SEM analysis (Figure 1). SWC was not related to precipitation (no significant relationship) (p-value= 0.212). However, the most significant, but negative, parameter that directly influenced on EMF was precipitation (MAP) (β=-0.527, standardized coefficient), a result that suggests great relevance of this factor in the ecosystem multifunctionality (Figure 1).

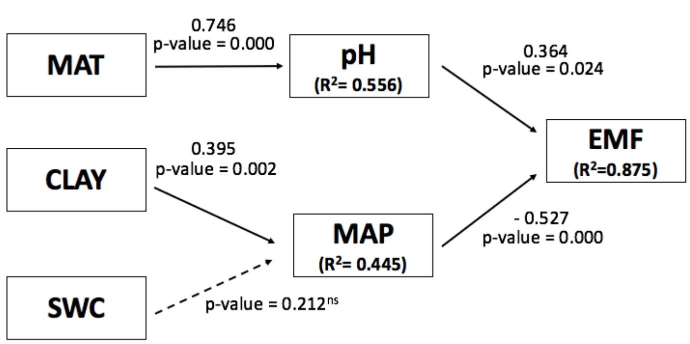

Figure 1. Structural equation model of climate and soil as predictors of ecosystem multifunctionality (EMF). Solid arrows represent significant positive and negative paths (p-value <0.05) and dotted arrow represents a non-significant paths (p-value >0.05). The path coefficients are report as standardized effect

sizes. MAT: mean annual temperature, MAP: mean annual precipitation, SWC: Soil water contents, and EMF: Ecosystem multifunctionality.

CONCLUSIONS

- The implementation of silvopastoral systems (SPind) increases water retention during dry season, increasing biogeochemical cycles dynamics that usually are to diminished during this season.
- Municipality with the lowest MAP (X'matkuil: 995 mm) showed the highest SOC, total N and total P concentration, as well and the highest pH and SWC. This highlights the influence of soil water contents (precipitation) on ecosystem multifunctionality (nutrient concentration) under sub-humid conditions.
- The climate had a great influence on the EMF, indirectly by MAT through the soil pH, and directly by the MAP that had the highest value of the standardized coefficient observed within the factors that directly influenced ecosystem multifunctionality in the structural equation model analysis.
- Cattle raising with silvopastoral management provides a variety of goods and services to society, is a means of adaptation and mitigation to climate change. Increasing production sustainability and integration of silvopastoral systems implies commitment and responsibility by the various social actors involved, as well as significant changes in state and national livestock raising policies.

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9.2.P

A COMPARISON OF SLOPE EROSION SEDIMENT YIELD CHARACTERISTICS ON YELLOW SOIL IN SOUTHWEST CHINA AND LOESS IN NORTHWEST CHINA

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1 INTRODUCTION

In recent decades, many researchers have done research on soil erosion in karst regions, which have made important contributions to understanding the characteristics of soil and water loss in karst areas. However, most of those studies focus on the characteristics of regional soil and water loss (Xiong et al., 2012), the classification of soil erosion intensity (Zhang et al., 2007) and the relation between soil erosion and rocky desertification (Long et al., 2006). Rainfall simulation (Liu et al., 2013) and flume (Guo et al., 2013) experiments, as well as the use of remote sensing (RS), geographical information systems (GIS) and other technologies for regional scale monitoring have been done (Shokr et al. 2016). Chen et al. (1994) compared the benefits of different soil and water conservation methods through experimental plot data. However, studies of soil and water loss on the slope scale are limited, especially with respect to characteristics of soil erosion.

As for characteristics of runoff and sediment yield on the slope scale, most of those studies have been done on the Loess plateau. For example, Liu et al. (2002) proposed a soil loss prediction model, which is suitable for China. The objective of this study was to evaluate the characteristics and relations among rainfall, runoff, and sediment yield under natural rainfall events through statistical methods, and to compare the differences between two stations based on the data of field plots in Bijie of Guizhou Province and Ansai of Shaanxi Province.

2 MATERIALS AND METHODS

2.1 Study sites and data

The study area of yellow soil is located in the Shiqiao Watershed in Guizhou Province, which has a total area of 3593 ha. The overall climate can be classified as humid subtropical monsoon, and the bedrock is limestone, with an elevation ranging from 1400 to 1742 m. The watershed receives an average annual rainfall of 863 mm, and about 52.4% of the rainfall occurs during the monsoon season (July-September). The average annual temperature is 14.03℃ and the main vegetation type is evergreen deciduous broad-leaved mixed forest. The study site of loess is located in the Ansai Soil and Water Conservation Station of Chinese Academy of Science and Ministry of Water Resources, which belongs to the Loess Hilly region. The region has a semi-arid, continental climate with an annual average temperature of 8.8℃. The watershed receives an average annual rainfall of 549.1 mm, and about 61.1% of the rainfall occurs during the monsoon season (July-September). The elevation ranges from 1040 to 1425 m.

In each of the study areas, a unit plot with a slope length of 20 m and a slope of 15° was constructed (Zhang et al., 2009). The soil depth of the plot in the Shiqiao Watershed was 21 cm, and the soil particle distribution was 22.2% clay, 32.5% silt, and 45.3% sand. The water stable aggregates (> 0.25 mm) comprised 61.6% and the bulk density was 0.94 g·cm^{-3} of yellow soil. The total porosity was

44.31% and the steady infiltration rate was 7.75 mm·min^{-1} for the Shiqiao Watershed plot. As for the loess, the soil is a silt loam with 4.3% clay, 54.1% silt, and 41.6% sand. The water stable aggregates (> 0.25 mm) comprised 29.6%, and the bulk density was 1.1 g·cm^{-3}(Yu et al., 2014). The steady infiltration rate for the loess was 0.67 mm·min^{-1} (Zhu et al., 2012).
Suspended load and bed load were sampled and sediment yield was calculated using the oven-drying method. The rainfall process was recorded as a pluviograph, and rainfall kinetic energy and erosivity were obtained. The data of rainfall, runoff, and sediment yield for the yellow soil plot was collected during 2012-2014 for 43 rainfall events. The data for the loess was obtained during 1985-1989 for 38 rainfall events.

2.2 Data analysis

The statistical method uses the correlation analysis routines in the SPSS18.0 software to do bivariate correlation analysis and applies a significance test. Linear and exponential regression analysis was applied to study the relations among rainfall, runoff, and sediment yield. All figures were mapped using Origin 8.5.

3 RESULTS

3.1 Characteristics of rainfall, runoff, and sediment yield

Rainfall, runoff, and sediment yield data for both the yellow soil plot and loess plot were analyzed using statistical methods (Table 1). Rainfall, runoff, and sediment yield characteristics of the yellow soil were less than those in the loess area. The average precipitation, runoff, and sediment yield of storms in the karst area accounts for 66.8, 57.8, and 40.6% of those in the loess region, respectively. In addition to the runoff coefficient, the coefficients of varitation of precipitation, runoff depth, and sediment yield in the karst region were greater than those in the loess area. These results show that erosive rainfall were more evenly distributed in the karst region.

Table 1 Basic characteristics of rainfall, runoff, and sediment yield on the yellow soil slope and loess slope (Ansai)

	Site	Minimum	Maximum	Average	Standard deviation	Coefficient of variation
Precipitation/mm	Bijie	4.10	67.30	19.21	17.68	0.92
	Ansa	5.00	110.60	28.75	21.32	0.74
Runoff/mm	Bijie	0.16	23.04	2.77	5.20	1.88
	Ansa	0.13	37.22	4.79	8.09	1.69
Runoff coefficient	Bijie	0.02	0.37	0.11	0.08	0.75
	Ansa	0.01	0.55	0.15	0.14	0.93
Sediment yield/（t·km^{-2}）	Bijie	0.07	4118.	243.03	821.87	3.38
	Ansa	0.20	7804.	597.97	1386.59	2.32

Based on meteorological classification method of Ke (1995), rainfall is divided into light (P < 10 mm), moderate (10 ≤ P < 25 mm), heavy (25 ≤ P < 50 mm), storm (50 ≤ P < 100 mm), and extraordinary storm (P ≥ 100 mm), using precipitation (P) as an index. The total of 43 and 38 rainfall events in the karst region and the loess area were classified during the study period (Table 2). For the yellow soil plot (Bijie), light and moderate rainfall accounted for 74.4% of the total rainfall events, and contributed 21.1% of the total precipitation. As for the loess plot (Ansai), 57.9% of the rainfall events were light and moderate rainfall, and they contributed about 10.7% of the total precipitation. 89.7% of the total runoff came from the heavy and storm rainfall for the yellow soil plot, and this number for the loess plot was 85.4%. The main rainfall types in Karst region were light and moderate rainfall, and in the loess area, heavy rainfall and storm were the major types. However, in both regions, heavy rain and storm contributed most of the precipitation in all years. With the increase in rainfall level, runoff and sediment yield for the yellow soil plot increased gradually range to range, significantly up to 10 times over the full range. Those increases for the loess area were more steady.

Rainfall which results in a sediment modulus greater than 100 t·km^{-2} for the yellow soil plot happened 5 times during the study period, including 1 time of heavy rain, and 4 times of storm, accounting for 97.49 and 64% of the total sediment yield and runoff, respectively. As for the loess plot, there were 6 times of rainfall with a sediment modulus greater than 1000 t·km^{-2}. All of those rainfall events contributed about 78.2% of the total sediment yield and 66.3% of the runoff in all. These results show that for both areas, most of the runoff and sediment yield came from heavy rain and storms.

Table 2 Classification statistics of rainfall, runoff, and sediment yield of individual rainfall events

	Bijie				Ansai			
Precipitation classification	Rainfall	Average rainfall/(mm)	Average runoff /(mm)	Average sediment yields/ (t·km^{-2})	Rainfall	Average rainfall/(mm)	Average runoff/(mm)	Average sediment yield/(t·km^{-2})
＜10	23	7.51	0.67	2.0	5	8.06	0.84	33.18
10～25	9	17.33	1.32	14.69	17	16.92	2.28	283.28
25～50	6	33.0	2.32	117.55	11	37.88	4.04	415.49
50～100	5	59.88	15.56	1913.02	4	59.28	14.32	1342.58
≥100	0	0	0	0	1	110.60	37.22	7804.50

3.2 Relations between rainfall and sediment yield

The results of the correlation analysis among rainfall characteristics and sediment yield are listed in Table 3. The correlation sequence of rainfall characteristics and runoff depth in the karst region was $I_{60} > P > I_{30}$, and in the loess area, the result was $I_{60} > I_{30} > P$ (where I_{60} and I_{30} are the 60 min and 30 min intensities, respectively). Sediment yield was less related to rainfall properties on the yellow soil slope than those on the loess slope. Figure 2 shows the quantitative relations between rainfall characteristics and sediment yield. With the increase in rainfall intensity, sediment yield on the yellow soil slope increased exponentially, which is consistent with the study of red soil slope erosion by Zhang et al. (2010). However, the sediment yield on the loess slope increased following a power function, which was similar to the research by Zhang et al. (2009) in the loess hilly region. The relation between sediment yield and precipitation on the yellow soil followed an exponential function, shown in fig. 2(c), similar to the conclusion of Gu et al. (2015). The fitted equation is: $M = 0.54e^{0.11P}$, where M and P refer to the sediment yield modulus, t·km^{-2}, and precipitation, mm, respectively. In areas lacking of erosion data, this equation can be used to predict soil erosion on slopes.

There is a linear relation between sediment yield and precipitation in the loess area, as Yang et al. (2010) also found.

Table 3 Correlation of rainfall characteristics and sediment yields

Area	P	I_{30}	I_{60}
Bijie	0.640**	0.600**	0.662**
Ansai	0.692**	0.776**	0.903**

Note: ** refers to significance at 0.01 level.

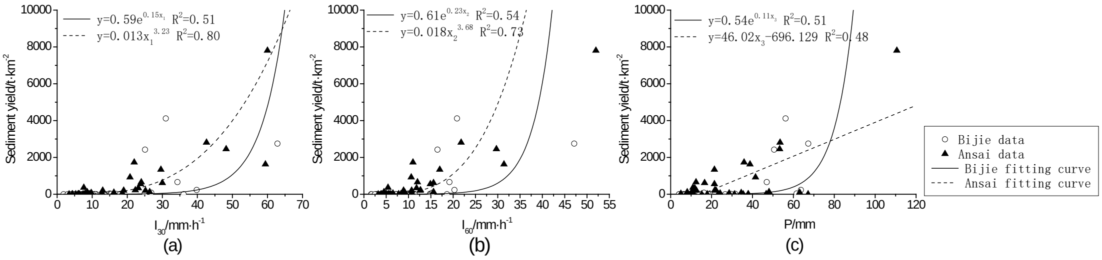

Fig.2 Relations between sediment yield and rainfall characteristics

Note: x_1, x_2, x_3, refer to I_{30}, I_{60}, and P, respectively.

The study also considered the rainfall erosivity index, using the product of E_n (kinetic energy) and I_n (rainfall intensity), in particular the EI_{30} index as Wischmeier (19xx) proposed stated [cite a reference by filling in the xx, don't forget to also mention Wischmeier co-authors, if any]. Using the measured runoff data, Fig. 3 shows the result of regression analysis between erosivity index and runoff and sediment yield. Erosivity index shows linear relations with runoff depth and sediment yield. It shows that loess is more prone to erosion than yellow soils. The greater rainfall erosivity, the greater differences between runoff and sediment yield.

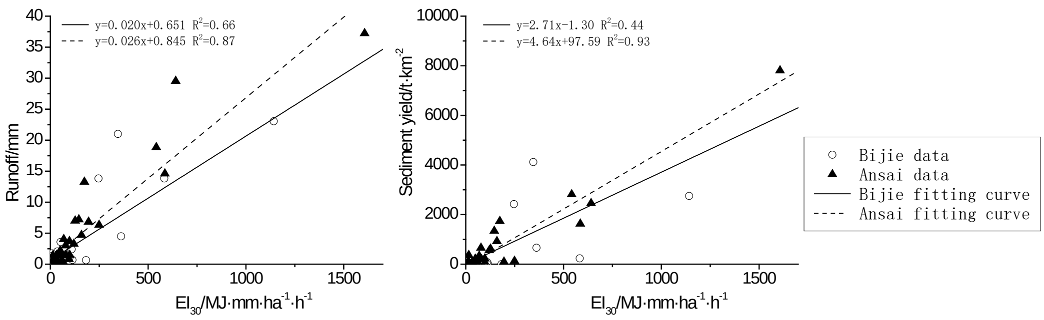

Fig. 3 Relations between erosivity index and runoff and sediment yield

3.3 Relations between slope erosion sediment yield and runoff

The regression analysis of runoff depth and sediment yield modulus was done for the measured data for the field plots (Fig. 4). The results show that the sediment yield on the loess slope was slightly greater than it on the yellow soil slope. Because of the lack of soil resources in the southwest karst region the degree of damage to the soil resources of the yellow soil slope erosion is much greater than that of the loess area.

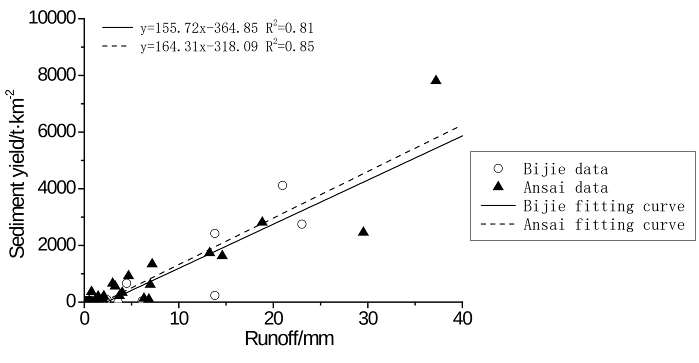

Fig.4 Relations between sediment yields and runoff depths

4 CONCLUSIONS

1) Runoff and sediment yield mainly resulted from heavy rains and storms, and the erosional rainfall, runoff, and sediment yield in the yellow soil area were all lower than those in the loess area.

2) Runoff was more responsive to precipitation on the yellow soil slope, and runoff was more responsive to I_{60} on the loess slope. For increased rainfall levels, the relation between runoff and rainfall characteristics of the yellow soil slope changed from irregular to linear. There is a linear relation between the runoff from the loess slope and the rainfall intensity of the study period.

3) The rainfall erosivity index (EI_{30} index) is more applicable for the yellow soil slope as a response to rainfall and slope erosion index [This sentence is unclear and does not seem to be supported by the data, please clarify]. When lacking erosion data, the equation M = $0.54e^{0.11P}$ can be used to predict soil loss on the yellow soil for individual rainfall events.

4) The sediment yield for the yellow soil slope is exponentially related to rainfall intensity, and the relation between sediment yield and rainfall intensity for the loess slope follows a power function. The sediment yield from both yellow soil and loess slopes were linear functions of the runoff.

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9.3.P

CREATING CONDITIONS FOR KNOWLEDGE EXCHANGE: A PROJECT ON PRESERVATION OF ECOSYSTEM SERVICES IN FLUVIO-LITTORAL LANDSCAPES

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1. INTRODUCTION

The management of the natural capital of fluvio-littoral Mediterranean landscapes, and the flow of ecosystem services (ES) that are generated, are key concepts for the functionality and sustainability of these areas. The sustainable management of these resources has been a topic of continued concern, in part due to the complex and dynamic socio-ecological systems (SES) in which they are imbedded. As a result the challenge for managers is to ensure the sustainability of these areas in the long term by preserving its ecological and cultural values against pressures (anthropogenic and not anthropogenic, predictable and unpredictable) and at the same time to guarantee the efficient use of environmental resources (Millenium Ecosystem Assessment, 2005).

In some sites, the conservation objective is actually dependent on the maintenance of human practices that shaped the ecosystems and actively strive to include all stakeholders in the management of the sites (Bouamrahe *et al*., 2016). Under this philosophy, the Eco^{2}Tools project we present in this paper, in an innovative approach, integrates the concepts of natural capital, flow of ecosystem services derived from this capital, contaminants, ecological risk assessment, environmentalomics, environmental fingerprint economic analysis, and participatory process in conservation and environmental restoration programs. Eco^{2}Tools aims to join local knowledge and practices with technical expertise and innovation, related to context and location. By doing so we may develop all together (scientists, nature conservation managers, producers, and users) a decision support tool for these habitats management and the mitigation of their vulnerabilities, so the entire project is not to remain a purely theoretical objective.

Indeed there is an emerging realization that effective research uptake in policy and practice may be built upon a foundation of active knowledge exchange (KE) and stakeholder engagement during the process of knowledge production itself (Phillipson *et al*., 2012). KE describes the interchange of knowledge between research users and "scientific" producers (Mitton *et al*., 2007). It is a process aimed at building relations and opening the lines of communication between researchers and their nonresearch audiences, to facilitate the translation of research findings into action (Graham *et al*., 2006).

Building dialogue among the stakeholders then appears to be one of the preliminary conditions needed to manage these landscapes from a sustainable development perspective. In this context, the goal of this paper is to describe and analyze the process of enabling conditions to improve the knowledge flow in a multi-directional process among experts, managers and users. Moreover, main difficulties detected over a range of contextual factors will be analyzed.

2. METHODS

An experimental approach to knowledge exchange has been adopted. It involved the initiation of knowledge co-operatives engaging researchers with managers and local users.

2.1. Study area

The selected case study is located between the Turia (to the North) and Jucar (to the South) rivers that flow into the sea along a broad coastal plain. This landscape structure has a dense network of canals and ditches for irrigation, the waters of which come from catchment systems built on both rivers some centuries ago. Such landscape is mixed with the original organization of the flood plain, where natural environments in a fluvio-littoral coastal wetland type (Albufera) are still distinguished. Agriculture is the main activity of the area. However, there are also a large population and urban development pressure, surpassing population densities of 2000 inhabitants per km^2, (above the rates given for other areas in the Valencia Community).

The selection of the study area has been based on two key aspects: the great ecological value of the services generated (its high biodiversity, water quality improvement, carbon sequestration, and also the aesthetic, recreational and touristic value they provide, among others), and their fragility under the pressures they suffer (urban, industrial expansion, etc.). The environmental importance of the selected area and, therefore, the need to reconcile protectionist interests with different socio-economic demands is emphasized because more than 50% of its surface has some level of protection as natural space.

2.2. Engagement process

The stages of the engagement process were:

(1) Stakeholder analysis based on Reed *et al*. (2009). These techniques are used to help identify and prioritize actors who constitute the network of potential suppliers and demanders of ecosystem services in the studied area, and systematically consider their behavior, interests, agendas, influence on decision-making and the framework conditions.

(2) A local stakeholder network establishment. It included a *Snow-ball* sampling approach, where first contacted stakeholders provided suggestions of other participants to be contacted.

2.3. *Instruments used to generate information and knowledge*

Communication between stakeholders was firstly mediated through a focus group, composed of members of the scientific and bureaucratic organizations initiating the project. The aim of the focus group and subsequent techniques is to evaluate and adapt the proposed aims of the project in order to ensure it is focussing on relevant issues.

Different participatory tools were used:

(1) Find and review secondary data: such as files, reports, maps, aerial photographs, satellite imagery, articles and books;

(2) *Transect walks*: Researchers systematically walked with key informants through a selected area, observing, meeting people, asking, listening, discussing, identifying different zones, local technologies, seeking problems, and mapping findings as proposed by Mascarenhas (1990). Everyone who wanted to contribute was included. Since the area was large enough it was divided up into smaller transect segments that were combined later.

(3) Participative GIS: to document human activities, uses and issues in the selected sites. As stated by Abbot et al. (1998) and Cinderby et al. (1999), these techniques encourage community participation and involvement in the production of GIS data. During transect walks, participants were asked to mark the relevant points and to describe each of them. GPS readings were taken for each of these pegged points enabling a link to be made between the local knowledge data and data from a scientific soil, water and sediment survey carried out in parallel. Within an agriculturally dominated case study area a more detailed land use classification, which included differentiated information on agricultural management practices, was utilized.

3. FINDINGS

This section reflects on what we are learning within this participatory process:

(1) Stakeholder analysis led to the following outcomes: i) a list of stakeholders and their stakes; ii) a list of stakeholder categories; and iii) information about how these categories of stakeholder related to one another. Target groups included local-level stakeholder (land users, local technicians, representatives of local authorities, interest groups). Researchers were also considered stakeholders.

(2) We were able to connect with stakeholders with widely different interests, and engage a broad range of opinions and comments in a learning conversation about local circumstances and needs. At this point, identifying the very diverse needs of the network members required a flexible approach and an array of technical and interpersonal skills within the focus group. From academia we need to extend our communication and networking abilities to engage citizens with projects related to preservation of ecosystem services. This included the ability to listen to and translate the opinions and knowledge of end users into scientific understanding of processes.

(3) Stakeholders are contributing to a variety of inputs to Eco^{2}Tools. Most make tangible contributions to knowledge production itself, providing access to facilities, materials or study sites. Transect walks have generated the highest level of detail of locals' knowledge as they are closely linked to visual observations and management practices. The on-site visits provide then opportunities for learning about land users' integrated knowledge about the environment and

their perception of challenges and solutions. In this regard, among the groups of local actors involved in the rice sector of the area, a growing concern is related to the appearance of the *Akiochi* disease of rice, also known as 'hydrogen **sulfide** toxicity', in the surroundings of the Albufera Natural Park (Valencia).

(4) Main difficulties detected over a range of contextual factors included: (i) scientific literature is not freely available to decision-makers, due to scientific journals requiring subscription to access the contents. Therefore, there is a clear need to develop a feasible knowledge feedback platform, providing clear and relevant outcomes; (ii) because stakeholders hold different perspectives, values, motivations and timing this leads to negotiations and struggles in the creation of knowledge that is to be used; and (iii) institutional barriers that need innovative 'adaptive' solutions.

4. CONCLUSIONS

The stakeholder approach implies that a common ground has to be built between stakeholders concerning their perceptions and expectations on interaction platforms before technical knowledge can be exchanged.

The facilitation and translation of information and meanings among stakeholders can lead to the co- production of knowledge, more informed decision making, and in a very pragmatic way, more effective use of scientific discovery. Dialogue and stakeholders' engagement is the necessary starting point.

5. ACKNOWLEDGEMENTS

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9.4.P

THE ROLE OF FARMER FIELD SCHOOLS IN SOIL AND WATER CONSERVATION: A PARTICULAR CASE IN THE VENEZUELAN ANDES

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Hundreds of experiences show that the lack of success of many natural resource conservation programs has been due to the lack of effective coordination between institutions related to this task and the low participation of resource users, both in the design of proposals, as in the execution and evaluation of results; and on the other hand, because many of these programs do not consider the non-formal education or rural extension as a strategy in the programs and projects execution. This would enable people to be trained with knowledge acquisition, skills and favorable attitudes for the conscious participation in the execution of adequate practices for soil and water management, leading not only to their conservation, but also to generate greater profitability in their production systems. So the soil and water conservation, foundations for agricultural development and rural development highly depends on the family participation who use these resources, especially the farming families, duly trained for this purpose. On the other hand, the participation of institutions responsible for the resources conservation, use and promotion of rural development processes is fundamental.

In this context, the case here presented refers to the "Conservation project of the productive landscape of the community of *La Laguna,* Mérida, Venezuela", located in the Arzobispo Chacón Municipality, Mérida State, Venezuela". Finance resources where granted from the *Small Grants Program* of the Global Environment Facility (SGP / GEF), implemented by UNDP on behalf of GEF-UNDP, UNEP, through the local NGO FUNDA-EPÉKEINA. The project is part of a inter-institutional cooperation agreement between FUNDA-EPÉKINA, the farmers council "La Laguna" and the University of Los Andes (ULA) through CORPOULA, in order to establish social, educational, academic, scientific and cultural cooperation relations in the execution of the project. For this purpose, the University of Los Andes (ULA) was in charge of the coordination and execution of the "Farmers Training" component of "La Laguna" farmers council. The main objective of the project is to contribute with the productive landscape conservation through the implementation of agro-ecological management system, which leads to the improvement of yields and product quality, the income and finally the living conditions of people, using a educational approach.

The educational approach was framed in the characteristics of "Non-formal Education" or in the approaches of the "Modern Rural Extension" with procedures and requirements different from those required by Formal Education, which is based on a dialogical communication that allows the Interaction, participation and knowledge exchange and meanings for the concretion of development proposals in the search for the collective well-being of the communities where it is carried out, motivating people to increase their knowledge, understanding useful information, gain new concepts meanings and information; acquisition of skills and adoption of desirable attitudes and ideals such as cooperation, natural resources conservation or the importance of farmers' organization to achieve better living conditions for their communities (Molina, 2013). In this case, the extension process was carried out in accordance with the FAO Field Farmers' Schools (ECA). These schools introduce technological innovations while taking advantage of local knowledge, where the teaching strategies are based on meaningful learning through discussion and practice, with emphasis on workshops, field trips and practical work (http://www.agriculturesnetwork.org/magazines/latin-america/1-aprendiendo-con-las-ecas/elementos-fundamentales-de-una-escuela-de-campo:received online in December 2015).

In addition, the whole training process use the participatory approach, making intensive use of participatory tools, considering that participation ensures the permanence of the process, since people learn to teach others, to organize work groups, to plan and to find solutions to their problems (Molina 2006, Molina et al., 2008). At the beginning of the training process, a participatory diagnosis was made, in order to adapt the training programme to the participant needs and expectations, being validated and feedback with them to go forward with the entire training process.

La Laguna Community is located at south Merida State, Venezuela, forming part of the Venezuelan Andes mountain range; between UTM coordinates: 898,300 to 902,800 m north latitude and 229,000 to 232,500 m east longitude. The area is approximately 570 ha. Altitude varies from 1400 to 2300 m.a.s l.; in terms of climate, it has an average annual precipitation of 1406 mm and 12,5 °C to 24 °C of temperature oscillations with an annual mean average of 18.25 °C. The population is made up of 49 families, with the same number of agriculture production units.

Participatory diagnosis was done applying a set of tools: community history, community map, representative farm map, flows diagram and agricultural calendars of the community. It is determined that production systems are highly diverse with a total of 18 agriculture items, distributed in: permanent crops, semi-permanent crops and short-cycle crops, as shown by the flow diagram performed by a community group in the participatory diagnostic process (Figure 1) Some of these items are for trade and Others exclusively for self-consumption.

Figure 1. Flows Diagram of the community of La Laguna, carried out in the participatory diagnosis process.

Coffee is the main item for the commerce and source of income of the community. The animal component is equally diverse: poultry, rabbits, bovine, sheep, goats and pigs are common. The bovine cattle is characterized by being extensive with grazing. Some of the pastures are wooded and allow them to be supplied with firewood and wood. Pigs have two forms of maintenance: grazing and confined. Poultry and rabbits keep them loose around the houses which they feed with maize, leftover food and sugar cane.

As a result of the educational process, La Laguna Farmers' Field School was created, with the following main themes: "Conservation of the productive landscape of La Laguna" and "Agro-ecological management of coffee crop" as a basis for sustainable rural development and friendly environment. The process begins with the familiarization and motivation of the community, through visits, meetings and interviews that allowed a preliminary design of a training programme. This was analyzed, modified and validated by the community in the process of participatory diagnosis. Thus, a study programme was carried out with 11 theoretical-practical workshops, whose themes and participation of community members are shown in Table 1.

Table 1. Training workshops and community participation held at ECA *La Laguna*

Main theme	Participants
1. The agroecosystem of the community La Laguna	22
2. Pest and disease management in coffee crop	28
3. Agronomic coffee crop management	29
4. Field day to areas with conservation practices and agroecological coffee crop	25
5. Organic fertilizer production using local materials.	20
6. The seedbed	20
7. The coffee nursery and shade trees	31
8. Importance of coffee plantations in natural resources and watersheds conservation	42
9. Soil management and conservation	31
10. Farm administration	20
11. Synthesis workshop "Integrating learning"	29
12. Establishment of coffee plantation and shade trees on contour lines and other soil conservation practices	27

The results of the training process exceeded the activities of the original proposal, which initially included only 3 training activities, meeting the requirements for a diploma level required by the University of Los Andes. This diploma certifies the acquisition of knowledge, skills and attitudes of the participants in "coffee crop agroecological management" within the project "Conservation of the productive landscape". Thus, 31 people obtained the diploma, in addition 4 certificates were given to people who attended 50% or more of the workshops or who had a motivating performance in the training process; 52 people attended more than 2 workshops at the farmers' field school, which indicates that if only the three training activities originally planned in the project had been carried out, the "participation" indicator would probably be higher, but would not be valued the constancy, interest and motivation that has kept the participants until the end of the training process, which culminated with a special practical activity in their own production units. The end-of-course work led to self-training, which allowed them to make presentations, in which they observed knowledge of the subject and didactic skills; showing in some participants their potential as future facilitators of other training processes.

The above table shows that the participation was maintained throughout the year of the training process, some of these workshops attracted special interest in the participants such as: Soil management (Figure 2). Most of them are organic fertilizers producers even though the participation in the workshop was not so high, they are applying the techniques to add

Figure 2. Workshop culmination moment. Practical module on Soil management and conservation.

nutrient value to process coffee residues, and finally it is worth emphasizing the interest aroused in soil conservation, showing great motivation to apply contour lines, shade trees and other conservation practices that allow them to avoid erosion and thus preserve the nutrients for coffee plantations.

An important result is the decision to install three model plots located in the upper middle and lower part of the community area of La Laguna in which the agroecological practices learned in the training process will be applied and tested in their end-of-course work , such as: ethological pest control using handcrafted traps, shade tree nurseries, organic fertilizers such as *bocashi* and compost taking advantage of coffee pulp residues and other local materials in the community, contour lines and triangulation frame for coffee plantation, shade trees and other conservation practices, which is intended to extend to 59 plots with potential for coffee establishment, which were raised during the training period and which project the community of La Laguna as a demonstration community of sustainable agricultural practices.

The production and acquisition of around 20,000 coffee plants with certified seeds were also produced and acquired in the model plots and in some plots of successful coffee producers and participants in the training process at the ECA-La Laguna, which would be destined to Seed production.

In conclusion, people participation and education contributes to change the way community members think, make, and feel, that would help to make more successful natural resource conservation and sustainable rural development programs.

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9.5.P

ANTHROPOGENIC SOIL SEALING AS A DIRECT PRESSURE IN AGRO-ECOLOGICAL PROTECTED AREAS: A SPATIAL AND TEMPORAL ANALYSIS IN L'ALBUFERA DE VALÉNCIA NATURAL PARK, SPAIN

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INTRODUCTION

Soil sealing is defined as the covering of soil due to urbanisation and infrastructure construction, such that soil is no longer able to perform the range of functions associated with it (European Environment Agency, 2002). It is the major direct impact on the environment and, consequently, it is considered one of the most threatening soil degradation processes worldwide, which especially affects the pre-littoral and littoral areas of the Mediterranean ecosystems where an intensive urbanization has been produced in the period 1970-2010 (UNEP/MAP, 2012).

It has two main characteristics: it is cumulative in time and extensive in space. According to Wilby and Perry (2006), there is a continuous trend to intensification of population living in cities and, hence, the process is likely to continue in near future, which in turn makes the need of obtaining precise data on soil sealing spatial structures at local level more relevant.

The aim of this work was the development of an integral methodology, based on the analytical facilities provided by Geographical Information Systems, to evaluate direct and indirect pressures due to artificial surfaces expansion on those spaces that should not be present, or at least they should be minimised. Therefore soil sealing process has been analysed. The methodology was applied to the Natural Park of L'Albufera in Valencia, eastern Mediterranean Spain, and surrounding municipalities to obtain background on how artificial surfaces extend from previously existing built up areas.

The main objectives were to determine rates and trends of artificial surfaces expansion represented by soil sealing and fragmentation, and to established spatial differences for soil sealing degradation according to incidence of growth.

METHODOLOGY

Direct (on-site) impacts on soils have been addressed using a Geographical Information Systems (GIS) approach for both synchronic (at one given date) and diachronic (time span) spatial analysis. Two detailed concrete and asphalt artificial covers for the year 1991 and 2011 were created using ARCGIS (v. 10.1) GIS vector structure. For the year 1991, an information layer was produced from a panchromatic aerial photograph of 1991. A present date artificial surfaces layer was constructed using the 2011-year colour orthophoto (provided by the Spanish National Geographical Institute).

The resulting layers were further integrated into a GIS analytical structure together with a coverage containing municipal boundaries of the administrative study area and another one including the limits of the Natural Park. Based on existing metrics (Cushman et al., 2008), landscape (spatial) structure was analyzed to determine the extent and trends of anthropogenic soil sealing and the degree of fragmentation and patchiness for both the municipalities of the administrative area and the protected land of the Natural Park. Soil sealing and fragmentation values were extracted for each year as absolute and relative surfaces.

Map overlay techniques (Gao, 2008) between the artificial surfaces and administrative entities and Natural Park layers were performed to obtain synthetic soil sealing values for each municipality and the total protected land respectively. Cartographic Diachronic spatial analysis was undertaken at administrative level using municipal temporal trends of soil sealing between 2011 and the reference date (1991) and, for the Natural Park, with the creation of a 500 metres buffer density map that represented neighbouring and expansion effects of artificial surfaces towards the protected agro-environmental area.

Spatial fragmentation was analysed deriving synthetic metrics from both the total administrative surface (without considering municipal division to avoid non desirable added land fragmentation imposed by administrative boundaries) and the Natural Park land.

RESULTS

Geographical concentrations of soil sealing processes are well identified. Rates of growth in both dates are higher in Northern municipalities (Fig. 1(A), with diachronic percentages of growth above 3%. The higher densities may be associated to intrinsic urban dynamics that will produce spatial differences depending on the proximity to urbanisation axis or main city centre (Salvati, 2014).

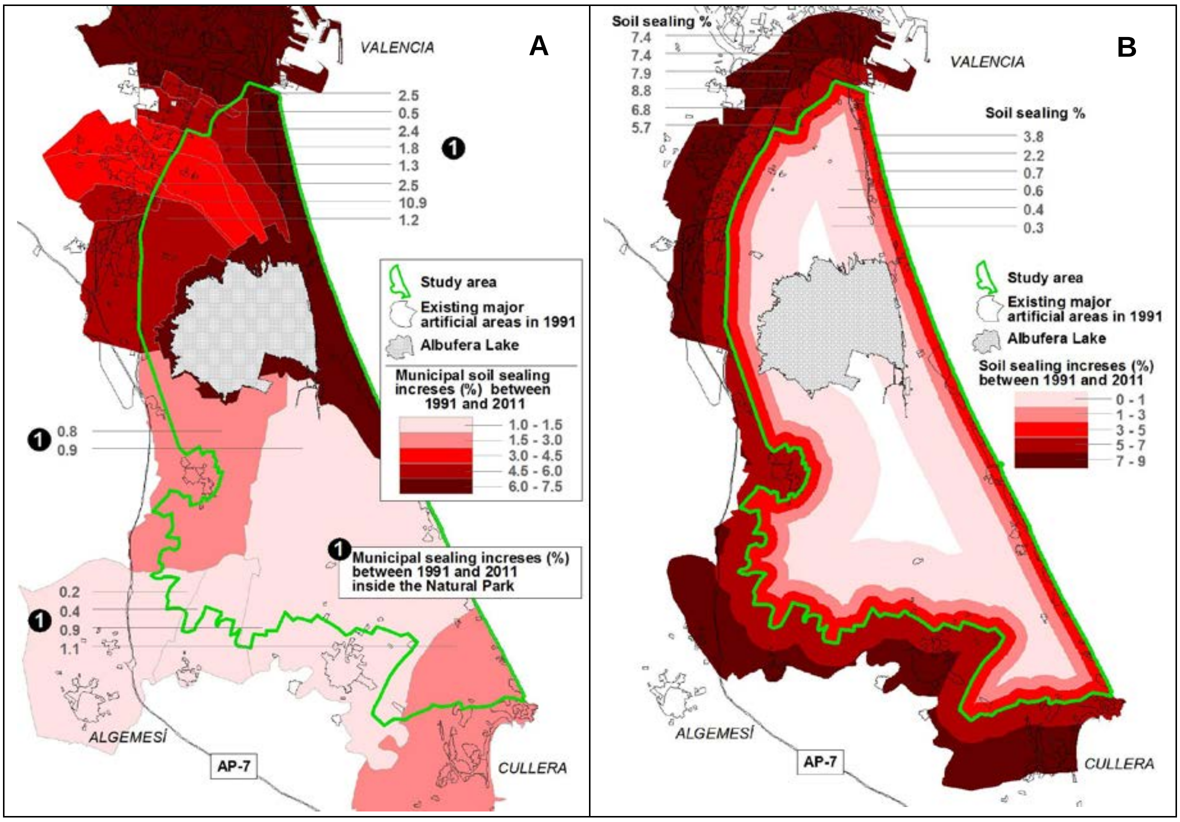

Figure 1. Soil sealing spatial representation: (A) municipal rates of change, (B) Density buffers in the protected Natural Park.

Soil sealing in this sector is related to artificial surfaces increase of a larger metropolitan area entity in which the city of Valencia acts as engine of growth through outsourcing of functions which, in turn, contributes to the expansion of neighbouring surfaces already consolidated, giving a double phenomenon of extension of the already existing land uses -neighbourhood effect-, and expansion of activities to other places -spatial externalities- (Hagoort et. al., 2008).
The consequence of that peri-urban trend is the continuity of the land-take process inside the limits of the Natural Park. Figure 1(B) shows a map of 500 m parallel buffers with percentages of soil sealing increases in each of them. Neighbourhood effect is produced in the contact between the protected land and other municipal landscape units, where rates of increment are between 1 and 5%.
Land degradation due to land use changes involves the interaction of the natural and the socio-economic systems (Bajocco et al., 2012), with predominance of the later. Whenever urban activities follow continuous expansion open space landscapes will be affected and will produce the unsustainable (economical and/or environmental use) of rural and peri-urban areas that cannot overcome the new socio-economic conditions. This is also applicable to the land of the Natural Park which is in connection to the non-protected municipal sectors, where socio-economic growth factors may impede total effectiveness of environmental policies in the protected area (Cabral et al., 2012).
Soil sealing expansion has produced a further fragmentation of soil compartments (Table 1) with a reduction of the average patch size as the number of patches increases. Other statistics also indicates the above mentioned trend showing a sort of homogenization showed by the variance and standard deviation.

Table 1: Landscape fragmentation trends (1991-2011 in the total administrative surface and agro-environmental Natural Park

Metrics	Total administrative area		Natural Park	
	1991	2011	1991	2011
Number of patches	1507	2256	331	558
Mean Patch Size (ha)	23.6	15.0	53.0	30.9
Maximum Patch Size (ha)	4682.3	2696.0	1649.3	1517.5
Patchiness Variance	18017.1	5352.0	22553.0	9477.0
Patchiness Standard Deviation	134.2	73.2	150.2	97.4

CONCLUSIONS
Anthropogenic soil sealing increase has occurred in all territorial units (municipalities and Natural Park) analysed. The enlargement of artificial surfaces has produced an increment of soil fragmentation in the study zones.
It is characterised by the reduction of the maximum and mean patch size, and the homogenisation of the path size reflected by the reduction of the patch variance and standard deviation.
Artificial surfaces constant increase may be seen as one of the most relevant direct factors of land use/cover change in the study area. It is included in a larger structure (at least to metropolitan scale that has long continuity in time.

Neighbouring and externalization are processes that would explain the greater incidence of artificialisation in Northern municipalities and the penetration in the outer 1000 metres stripe of the Natural Park.

ACKNOWLEDGEMENTS

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9.6.P

THE PRICE OF SOIL N DEPLETION IN KENYA AND ZIMBABWE: A COST-BENEFIT ANALYSIS

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INTRODUCTION

Sub-Saharan Africa is facing a long-term food security crisis due to continued land degradation (Sanchez, 2002). The application of fertilizer can remediate these soils and increase crop yields, but the high cost of fertilizer in Africa calls into question the practicality of this solution (Vanlauwe and Giller, 2006). It is, therefore, crucial to evaluate economics of applying fertilizer in low nitrogen (N) environments. From 2010-2015, field experiments coordinated by CIMMYT in Embu and Kiboko, Kenya and in Harare, Zimbabwe, investigated maize yield and N uptake responses of 6 hybrids varying in N use efficiency (NUE) per site/season to 4 N fertilizer rates ranging from 0 to 160 kg N/ha in continuous maize production systems. In 2015, we assessed the impact of hybrids and fertilizer on soil nutrient content in the upper 90 cm of soil. Total plant N uptake was measured in the last season in Zimbabwe and the last 4 seasons in Kenya. In addition to detailing the hybrid-fertilizer-soil interactions in low N environments, the results of this study provide foundational data for an in-depth cost-benefit analysis for soil fertility remediation.

RESEARCH DESIGN AND METHODOLOGY

The sites for this study were Harare in Zimbabwe, and Kiboko and Embu in Kenya. The experimental design for each site was a split-plot arrangement in a randomized complete block design with 4 replications and N-rate as the main plot and maize hybrid as the sub-plot. At each of the sites, 6 maize hybrids (3 hybrids bred by CIMMYT to have superior NUE (HNUE) and 3 hybrids that were locally commercially available (COMM)) were grown with the same N rate each season. N rates ranged from 0 to 160 kg N/ha. The actual HNUE and COMM hybrids used each season varied between and within sites. The N fertilizer source in Kenya was calcium ammonium nitrate (26 %N) and in Zimbabwe, it was ammonium nitrate (33.5 %N). At all sites, 30% of fertilizer was applied at planting. In Kenya, the main fertilizer application occurred when the maize was knee high. In Zimbabwe, 35% of the N fertilizer was applied at 4 and 8 weeks after planting. No crop residue was returned in Embu and Harare and a third of residue was returned in Kiboko. At the Kiboko and Harare sites, prior to the initial planting (2011 for Kiboko and 2010 for Harare), the soil was depleted of N for 5 seasons with continuous sorghum in Kiboko and maize in Harare. The Embu experiment was initiated in 2010 on soil that had not been intentionally depleted of N.

At all sites and in most seasons, grain was analyzed for protein concentration by NIR. Protein concentration was converted to %N using the protein:N ratio 6.25:1 (Jones, 1931). Grain and stover samples were taken from the middle rows at the 2015 harvest in Embu and Harare and at the 2013 harvests in Kiboko. At all sites, soil was sampled post-harvest at the end of the 2015 season to a depth of 90 cm in 5 depth increments. Bulk density was measured in all replications and at all depths. Plant and soil samples were analyzed for total C and N using a Flash 2000 CHN Analyzer (ThermoFisher Scientific Inc). Soil samples were analyzed for ammonium and nitrate using an AQ2 Discrete Analyzer (SEAL Analytical). Grain and stover N content was used to calculate N balance:

$$N\ balance = N\ applied - Total\ Plant\ N\ Uptake_{N\ rate}$$

For seasons where stover samples were not analyzed for N, stover N content was calculated using the corresponding grain N content. The known N harvest index values were regressed with N rate for each hybrid and grain N content values were substituted into the regression equation to estimate the unknown stover N content values. The estimated stover N contents for the seasons where grain N data was available was then used to calculate N balances.

In most environments, a substantial portion of N applied would be lost and not taken up by plants. Therefore, our calculated N balance was an underestimation of potential N depletion. This study also looked at the indirect costs/benefits of conservation agriculture practices to maximize N return to soil such as returning plant residue to soil rather than burning the residue.

The soil N remediation cost was the cost of fertilizer needed to negate any soil N depletion i.e. yield a net zero N balance. Like the N balance calculation, the remediation cost assumed that none of the N applied was lost from the system. This cost was based on the soil N balance on an annual and cumulative basis using the average annual local cost of fertilizer (544 USD/ton in Kiboko, 538 USD/ton in Embu (africafertilizer.org), and 700 USD/ton in Harare (Omnia Fertilizer Zimbabwe Ltd)). Gross income was calculated using the average annual price of maize in the area surrounding the site (295 USD/ton in Kenya (FAO, 2014) and 390 USD/ton in Zimbabwe (USAID, 2015)). Net income was the difference between the cost and gross income. Data was analyzed using SAS 9.4 PROC Mixed ANOVA and differences in least squared means were significant at α=0.05. Blocks could not be pooled due to significant variance in residuals. Grain yield, grain %N, and biomass data was combined over seasons for seasons with consistent management (hybrid and N rate) and where Bartlett's test for homogeneity could be satisfied. A 2-tailed LSD (α=0.05) was used to compare least squared means to a constant.

RESULTS

There were mixed results when looking at the N rate effect on soil N remediation cost and net income. Increases in N rate did not necessarily decrease how much more N was needed to reach a net zero N balance due to a simultaneous increase in plant N uptake with N rate (Figure 1). In Harare, the soil N remediation cost decreased only at 160 kg N/ha, otherwise there was no N rate effect on cost. In Embu, cost decreased with N rate at 60 and 90 kg N/ha, but there was no effect at 30 kg N/ha. In Kiboko, cost increased at 40 and 80 kg N/ha, but then plateaued for rates above 80 kg N/ha.

In the final season of the study, averaged over all hybrids, the N balance was -20 to -75 kg N/ha in Embu,
-50 to -100 kg N/ha in Kiboko, and -50 to -100 kg N/ha in Harare (data not shown). There was 25-35 kg N/ha in the removed stover in Embu, 20-60 kg N/ha in the removed stover in Kiboko, and 30-75 kg N/ha in the removed stover in Harare (data not shown). When stover was burned rather than reincorporated in the soil, up to 80% of the N was lost (Wortmann and Kaizzi, 1998).

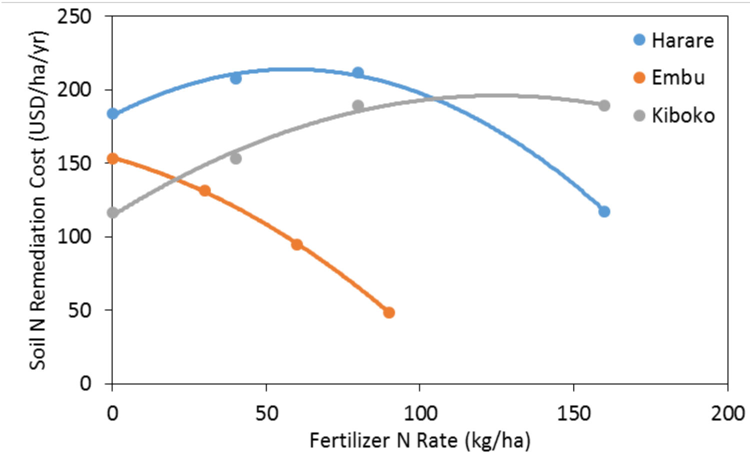

Figure 1 Averaged across all hybrids, soil remediation cost changed with N rate in the most recent seasons. There was a significant N rate effect on soil remediation cost all seasons it was measured in all sites (data not shown) ($p<0.01$).

Total soil N content in the top 90 cm increased with N rate in Embu at 30 and 90 kg N/ha only and in Harare, but not in Kiboko (data not shown, $p<0.05$). Soil inorganic N content in the top 90 cm increased in Embu at 90 kg N/ha only ($p<0.05$) and in Harare at both 80 and 160 kg N/ha ($p<0.01$), but not in Kiboko (data not shown). Assuming all grain was sold at market price, the net income increased with N rate in Kiboko and Embu, but did not change in Harare (Figure 2). Grain yield also did not change with N rate in Harare (data not shown). In Embu, net income increased with N rate at 30 and 60 kg N/ha, but plateaued at 90 kg N/ha. Grain yield plateaued at 30 kg N/ha in Embu (data not shown). In Kiboko, net income and grain yield increased with N at all N rates (data not shown).

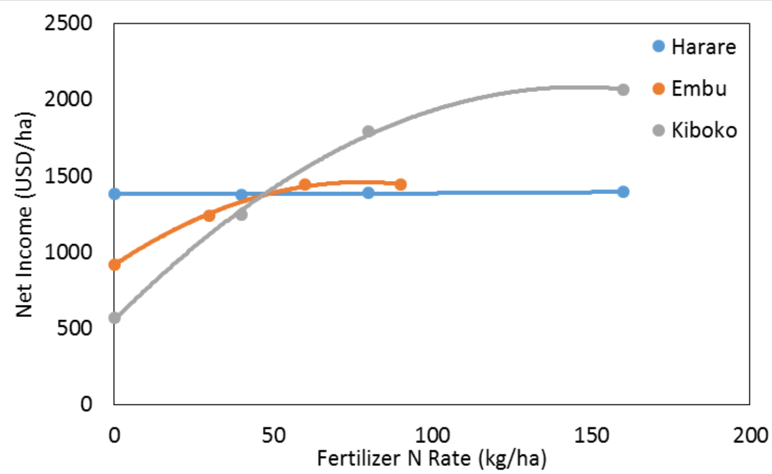

Figure 2 Averaged across all hybrids, net income in Embu and Kiboko changed with N rate in the most recent season ($p<0.01$). There was no N rate effect on net income in Harare during this season. Overall, potential net income increased with N rate in Kiboko and Embu.

DISCUSSION

The contrast in hybrids' response to N rate in different sites as reflected in this cost-benefit analysis shows that the addition of fertilizer may have limited impact on grain yield and profit, providing little incentive at some sites to remediate soil N with inorganic fertilizer. For this reason, this study also looked at the benefit of returning crop residue to the soil as an alternative method for remediating soil N with minimal input cost. In sites where fertilizer increased grain yield, however, applying additional fertilizer to remediate the soil N pool may not have a significant impact on profit.

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Miscellaneous

M.2.P

LOESS DEPOSITS IN THE LOWER EBRO BASIN (NE IBERIAN PENINSULA)

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INTRODUCTION

Sole Sabaris et al. (1957) was one of the first to mention the presence of loess in the Iberian Peninsula. After that little attention has been paid to these materials and even the Loess map of Europe from Haase et al. (2007) do not mention them. Several areas with loess deposits have been described recently in the Iberian Peninsula: in the River Tajo Valley (central-western Spain) by García-Giménez et al. (2012), in the Southern Spanish Plateau by González et al. (2000), in Andalucía (southern Spain) by Günster et al. (2001), and in Girona by Mücher et al. (1990).Within the Ebro Basin DARP (1987) and Boixadera et al. (2015) described them in the lower valley.

OBJECTIVES

The objective of this paper is to identify and characterize the distribution pattern of loess, in a large area, close to the one studied by Boixadera et al (2015), and to advance in the understanding of their origin, conditions of formation and pedogenesis.

MATERIALS AND METHODS

A systematic field survey was undertaken in Bajo Aragón and Matarraña counties (Aragon) and Terra Alta county(Catalunya) in the lower Ebro Valley (Figure 1), an area westwards to the one studied by Boixadera et al. (2015) to locate and characterize aeolian deposits. Four sampling sites were selected (Figure 1) from the loess deposits and at each site, land use and geomorphology were described, together with a macromorphological characterization of the deposit following FAO (1990). Samples were obtained for physico-chemical, particle-size, and micromorphological analyses. Soils were classified according to Soil Taxonomy (SSS, 2014).

Particle-size analyses were performed by the pipette method without removal of carbonates following the full procedure (Porta et al., 1986), obtaining the clay, fine silt, coarse silt, and five sand fractions (USDA). Samples were also analysed for calcium carbonate, gypsum, organic carbon, pH (1:2.5), and electrical conductivity (EC, 1:5) (Porta et al., 1986). Thin sections, 5x12 cm, were produced from undisturbed blocks from selected horizons of the deposits following Benyarku and Stoops (2005). These sections were described with a polarizing microscope according to Stoops (2003).

RESULTS AND DISCUSION

In the surveyed area, the loess deposits cover some 230 km^2 in an irregular and discontinuous pattern within an area of 1900 km^2. Most of them show the characteristics of sandy loess (Coudé-Gaussen, 1990), but others have properties of loess-like deposits. They appear on different landforms, such as flat summits, slopes, higher than T1 terraces of the Ebro River and its tributaries, abandoned meanders from Matarraña and Algars rivers and endorheic blowout basins. The location and thickness of the loess are, in general, related to the presence of transverse topographical obstacles that acted as traps for the material transported by the predominantly North-Westerly winds of the region. Different deposition areas may therefore be defined according to their location and distance from those geographical structures (Figure 1). Two centres of sedimentation, one in Batea and the other in Mora d'Ebre, are suggested.

The thickness of these deposits is, in general, 3-4 m, but can reach up to 10 m. They are very homogeneous in terms of particle size and are mostly composed of very fine sand and silt. They do not show sedimentary structures, and have a light yellowish brown colour. The risers are very stable despite the low cohesion among the particles. The main characteristics of some of the deposits studied are:

- Casserres (X: 0272548; Y: 4548817; Z: 475 m): located at the centre of the area on the flat summit of Batea-La Fatarella. This deposit has a thickness up to 530 cm and is composed of primary loess. The underlying rocks are Oligocene lutites and sandstones.
- Chiprana (X: 0733246; Y: 4570830; Z: 197 m): located at the North-Western corner of the area, in an endorheic blowout basin surrounded by sandstone paleochannels. It has a thickness up to of 325 cm, and includes both windblown-colluvium and purely windblown layers. The underlying rocks are Oligocene lutites and sandstones.
- Coll de Maella (X: 0269764; Y: 4554581; Z: 377 m): similar to the Casserres deposit but with thickness up to 505 cm.
- Fayón (X: 0276795; Y: 4569412; Z: 188 m): in the North of the area on an abandoned meander of the river Matarraña on the leeward side of a slope. It has a thickness up to 572 cm of primary loess. The underlying rocks are Oligocene lutites, limestones, and sandstones.

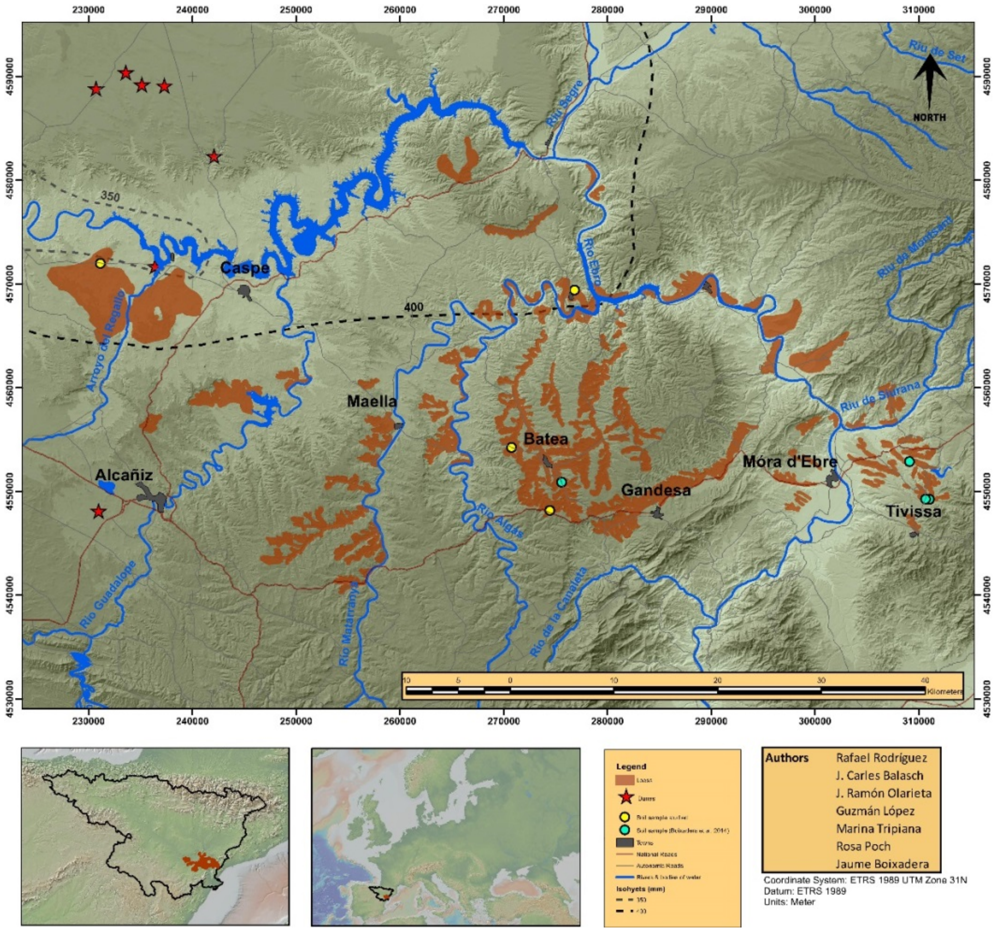

Figure 1. Location of loess deposits and dunes in the lower Ebro Valley.

The soils are mostly Calcic Haploxerept, and Typic Xerorthent (SSS, 2014) also appears. The most frequent Munsell colours are 10YR 5/6, 10YR 4/6, and 7.5YR 4/6. No mottling was observed except in one horizon. Rock fragments only appeared in a few horizons and in a very low proportion. Structure was mostly weak or moderate, subangular blocky, medium. Biological activity was frequent or abundant, as infilled galleries and worm casts. Secondary accumulations of $CaCO_3$ were present in all the deposits as friable and hard nodules or soft powdery lime. Secondary accumulations of gypsum also appeared in all the deposits except in Coll de Maella as vermiform gypsum and hard coatings of root channels. Gypsum rhizocretions also appear in the Fayón deposit.

The deposits mostly belong to the sandy loam, loamy, and silty loam USDA textural classes, but they vary vertically and laterally in all the studied deposit. The most common size fractions are (Table 1) very fine sand (50-100 µm; 32±11%), coarse silt (20-50 µm, 20±8%), fine silt (2-20 µm; 17±7%), and clay (<2 m; 18±4%).

Table 1. Mean, standard deviation, maximum, and minimum values of the various textural fractions USDA (n=61).

	Clay (%)	Fine Silt (%)	Coarse Silt (%)	Very Fine Sand (%)	Fine Sand (%)	Medium Sand (%)	Coarse Sand (%)	Very Coarse Sand (%)
Mean	17.7	17.5	20.0	32.0	9.7	2.1	0.6	0.4
Stan. Dev.	4	7.3	8.3	11.3	8	3.8	0.7	0.8
Maximum	24.4	45.8	37	53.1	37.3	20.7	3.9	3.4
Minimum	10.7	7.1	6.1	6.9	1.1	0	0	0

Soil pH (1:2.5) is basic due to the presence of $CaCO_3$ (Table 2). In 12% of the cases pH values reached up to 9.0. The mean value of EC (1:5) is 0.61 $dS{\cdot}m^{-1}$ a 25ºC; horizons with secondary gypsum have slightly lower pH (8.0-8.5), and EC (1:5) (2.0-2.6 $dS.m^{-1}$ at 25ºC).

Table 2. Mean, standard deviation, maximum, and minimum values of the main chemical characteristics of the studied soils (n=61).

	pH (1:2.5)	EC (1/5) at 25ºC (dS/m)	$CaCO_3$ (g/100g)	Oxidizable carbon (%)	Organic matter (%)	Gypsum (%)
Mean	8.5	0.61	40.5	0.2	0.4	3.1
Std. Dev.	0.3	0.69	7.0	0.2	0.3	4.9
Maximum	9.1	2.26	62.2	0.9	1.6	19.2
Minimum	8.0	0.11	29.1	0.1	0.1	0

These soils have a high content of $CaCO_3$ (mean value of 41% and most samples are in the range of 33-47%. The mean gypsum content is 3.1% with most samples between 0% and 8%. Chiprana and Fayón profiles have horizons with 4.5±5.4% gypsum, and higher than 10% in some cases. The organic matter concentration in surface horizons is between 1.0 and 1.8 %, and between 0.1 and 0.5%, in subsurface horizons. Microscopical observations showed thin sections to be very homogeneous. The groundmass is mostly composed of very fine sand of similar proportions of subangular quartz and more rounded calcite grains. Feldspars, micas, and opaque minerals appear in smaller proportions. Porosity is between 20-40%, and is made up of highly connected packing porosity. The c/f (coarse / fine) ratio is about 3/1 and the birefringent fabric of the micromass is crystallitic. The c/f related distributions are enaulic and close porphyric.

Soil development is very incipient, with just some redistribution of carbonates as orthic or disorthic impregnative micritic nodules, 500 to 1500 µm Ø. Infillings of sparitic microcrystals and calcite root pseudomorphs also appear in small proportions. Channel infillings of lenticular and xenotopic gypsum crystals were identified in various horizons in all deposits except in Coll de Maella. In the Fayón deposit a complex accumulation of gypsum was identified, made up of root pseudomorphs, xenotopic areas, and interspersed organic material. These deposits generally appear on the leeward side of the prevalent WNW winds in the area (Figure 1). Among the materials available for wind erosion, the fluvial sediments of the river Ebro were possibly those that could most easily be removed and were available in highest quantities, as they were annually replenished from the ice melting in the Pyrenees. A dry periglacial origin is proposed for these loess deposits, as other deposits in the Ebro Valley have been dated at 17-34 k-years (Boixadera et al., 2015). Their paleoenvironmental conditions correspond to the Heinrich cold periods 1, 2, and 3 (H1, H2, H3) of the Last Glacial Maximum (LGM). The distribution of the identified deposits also supports the hypothesis of a main source area located in the fluvial deposits of the river Ebro. The findings of Luzón et al. (2012) goes also in this direction. Nevertheless, the ~300 endorheic blowout basins in Escatrón, Caspe, Alcañiz, and Calanda (to the West of the study area), and the attrition of sedimentary rocks under dry and cold paleoenvironmental conditions might have also acted as source areas for these deposits. In the surveyed area (Figure 1) in very few cases, we have found very small sand dunes pointing out that possible source. The fibrous gypsum frequently present in the Tertiary rocks of the area is proposed as the source of the gypsum in the loess. The particle size, depth, and mineralogical composition of the deposits point to a relatively close source of particles, less than 100 km (Pye,1995).

CONCLUSIONS

We identified and mapped various loess deposits that discontinuously cover about 230 km^2 in the lower valley of the river Ebro (in the Bajo Aragón-Caspe and Matarraña areas in Aragón, and in Terra Alta in Catalunya). These deposits correspond to sandy loess, as they are rich in very fine sand and silt, coarser than the typical loess deposits. They have a gypsum content of 0-20% and that of calcium carbonate is 20-50%. We suggest that their development took place during the Heinrich cold periods H1, H2, and H3 of the Last Glacial Maximum, from the fluvial sediments of the river Ebro, the blowout materials from the endorheic basins of Caspe-Chiprana-Calanda and the attrition of local sedimentary rocks.

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M.4.P

LABILITY THE PHOSPHORUS IN DIFFERENT MANAGEMENT SYSTEMS

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The phosphorus (P) is essential for the crop production and your availability is necessary, in special for Brazilian Savanah, where the soils are of low fertility, including low P. One way of to increase the P in soils are the soil systems of management and conservation.
The objective of this study was to evaluate the lability of P in a dystroferric Red Latosol under three soil tillage systems: conventional tillage, no - tillage and integration of livestock farming.
Samples were collected in an experimental area of Embrapa Agropecuária Oeste (22 ° 14 'S - 54 ° 49' W and altitude of 430 meters) in the municipality of Dourados (MS), Brazil, of a typical dystrophic Red Latosol, kaolinite, very textured (Amaral et al., 2000) and with a climate classified as Cwa - moist mesothermic climate, with a predominance of hot summers and dry winters (FIETZ & FISCH , 2006).
The experiment consisted of three soil tillage systems - crop-livestock integration (CLI), no-tillage system (NTL) and conventional planting system (CTL), with soil occupation (Table 1) being performed with crops used in the region and The chemical and physical soil analysis presented in Table 2, collected at depth 0-0.10 m.

Table 1. Sequence of crops over time in different management systems in the experimental area.

Soil systens	1995	1995/96	1996	1996/97	1997	1997/98	1998	1998/99	1999	1999/20	2000	2000/01	2001	2001/02	2002	2002/03	2003	2003/04	2004	2004/05	2005	2005/06	2006	2006/07	2007	2007/08	2008	2008/09	2009	2009/10	2010	2010/11
CTL	C	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S	O	S
NTL	C	S	W	S	T	C	O	S	W	S	T	C	O	S	W	S	T	C	O	S	T	S	T	C	O	S	W	S	T	C	O	S
CLI	C	U.d.				S	O	S	O	U.d.				S	O	S	O	U.d.				S	O	S	O	U.d.				S	O	S

C: corn, S: soybean, O: oat, W: wheat, T: turnip, U.d.: *Urochloa decumbens*.

Table 2. Chemical and physical soil attributes in the experimental area.

	pH	P	K	Ca	Mg	Al	Areia	Silte	Argila
		mg dm^{-3}		cmol$_c$ dm^{-3}				g kg^{-1}	
CTL	6,8	21,9	0,42	8,7	2,1	0,0	229	131	640
NTL	6,8	37,8	0,82	7,5	4,0	0,0	162	148	690
CLI	6,4	14,3	0,38	6,2	3,2	0,1	179	131	690

Throughout 16 years of experiment, management systems received, on average, 61, 85 and 30 kg of P_2O_5 ha^{-1} year^{-1} for CTL, NTL and CLI, respectively. The contents of SOM in the 0-0.10 m layer were 15 , 50 and 30 g kg $^{-1}$, and in the 0.10-0.20 m layer were 10, 35 and 15 g kg^{-1} for CTL, NTL and CLI, respectively. The average values of dry mass under the soil (vegetal cover) were 945, 4143 and 3379 kg ha^{-1}, for CTL, NTL and CLI, respectively. Planting fertilization was performed with triple superphosphate, providing 120 kg ha^{-1} of P_2O_5. This was done in November 2010, and the fertilizer was incorporated after soybean planting in all management systems.

After the harvest (March / 2011) the soil was collected with the aid of a Dutch soil, at 0-0.10 m depths and 0.10-0.20 m, in the line of each plot. Each sample composed of the line originated from three subamotras that were then homogenized. This collection gave rise to 24 samples (3 systems x 2 depths x 4 replicates).

With the material collected, sequential fractionation of soil P was performed as described by Hedley et al. (1982), with modifications of Condron et al. (1985).

The fractions were separated by the lability described by Rotta (2012), where the labile P is composed of fractions of Pi extracted by resin and the contents of Po and Pi extracted by $NaHCO_3$; The moderately labile P is composed of the contents of NaOH 0,1 and 0,5 mol L^{-1} in the fractions of Pi and Po, and; The low labile P being the contents extracted with HCl 1.0 mol L^{-1} and H_2SO_4 (residual P). A table was made where the separation by P lability was as follows: total labile fraction, inorganic labile fraction, total moderately labile fraction, moderately labile inorganic fraction, moderately labile fraction, low labile fraction, inorganic fraction, total organic fraction of the soil fractions.

With the results obtained from the P fractionation, in association with the management, depths and collection sites, non-parametric statistical analysis using the Bonferroni test, by SAS® software. For the statistical analyzes, a randomized block design with subdivided plots was considered.

RESULTS

The fractions of P labil at both depths (0-0.1 and 0.1-0.2 m) did not differ for soil treatments (Table 3), except for the Po NaHCO3 fraction in depth 0,10-0,20 m. The differences between the depths were evidenced in all the managements for all extractions of the labile fraction, occurring reductions of the contents of P in the depth of 0.1-0.2 m, except for the NaHCO3 Pi of NTL and CTL.

Table 3. P (mg kg-1) in the soil fraction, collected in the 0-0.10 layer and 0.10-0.20 m, in a dystroferric Red Latosol.

P fracion	**Delph (m)**	**Soil system**		
		CLI	**NTL**	**CTL**
Pi Resin	0-0,10	59,0 aA	65,0 aA	64,6 aA
	0,10-0,20	34,8 aB	30,6 aB	32,5 aB
Pi $NaHCO_3$	0-0,10	59,9 aA	59,7 aA	39,8 aA
	0,10-0,20	24,9 aB	51,4 aA	25,1 aA
Po $NaHCO_3$	0-0,10	405,5 aA	754,4 aA	271,7 aA
	0,10-0,20	118,0 bB	475,8 aB	136,9 bB

Letters of the same type, do not differ among the managements within each depth. Letters that are the same as the upper case, do not differ in the same depth. Both analyzes were performed by the Bonferroni test

In the 0-0.10 m layer there was no significant difference in Po content in the management systems, however, in the 0,10-0,20 m layer there was no difference between CLI and CTL, but both differed from the NTL content (73% higher). In the management systems, there was a significant difference between NTL and the others, which did not differ in the 0.10-0.20 m layer in all four extractors of the moderately labile fraction (Table 4).

The lower amounts of moderately labile P in CLI and CTL systems in the more superficial layer (0-0.1 m) can be justified by the smaller amount of fertilization performed. For Pi extractors it is possible to observe that there is no difference between the depths.

The organic P at the depth of 0.10-0.20 m reduced in comparison to the depth of 0-0.10 m (Table 4). Organic P extracted by 0.1 mol L-1 NaOH had a 25, 49 and 42% depth reduction for CLI, NTL and CTL, respectively. The organic P extracted by NaOH 0.5 mol L-1 had a reduction to the lower layer of 7, 77 and 58% in the respective management systems. The levels of organic P extracted by NaOH 0,1 and 0,5 mol L-1 may have relation with the organic matter found in the soil.

The low labile fraction of P in the soil is presented in Table 5. The values found by the HCl extractor in the CLI management were higher than those found in the NTL and CTL systems, being 0-0.10 m 37 and 52% higher and In the 0.10-0.20 m layer 60 and 94% higher than the managements in question. The residual P of the soil presented about 95% of the little labile fraction of the soil.

Table 4. P (mg kg-1) in the moderately labile fraction (Pi NaOH 0.1 and 0.5 mol L-1; Po NaOH 0.1 and 0.5 mol L-1) -0.10 and 0.10-0.20 m, in the line in a Distroferric Red Latosol.

P fracion	Delph (m)	Soil system		
		CLI	CLI	CLI
Pi NaOH 0,1	0-0,10	141,6 aA	216,7 aA	149,4 aA
	0,10-0,20	141,2 bA	217,3 aA	142,5 bA
Pi NaOH 0,5	0-0,10	268,4 aA	317,0 aA	262,5 aA
	0,10-0,20	229,6 bA	332,7 aA	258,5 abA
Po NaOH 0,1	0-0,10	957,8 aA	883,0 aA	585,0 aA
	0,10-0,20	718,6 aB	457,1 aA	339,7 bB
Po NaOH 0,5	0-0,10	240,4 aA	597,5 aA	553,1 aA
	0,10-0,20	224,4 aA	139,6 bB	234,1 aB

Letters of the same type, do not differ among the managements within each depth. Letters that are the same as the upper case, do not differ in the same depth. Both analyzes were performed by the Bonferroni test

Table 5. P (mg kg-1) content in the low-labile fraction (Pi HCl and P residual) of the soil, collected in the 0-0.10 layer and 0.10-0.20 m, in the line on a Distroferric Red Latosol.

P fracion	Delph (m)	Soil system		
		CLI	NTL	CTL
HCl	0-0,10	35,6 aA	25,9 bA	23,4 bA
	0,10-0,20	25,5 aB	15,9 bB	13,1 bB
Residual	0-0,10	747,5 aA	852,5 aA	815,0 aA
	0,10-0,20	859,9 aA	632,4 aA	841,2 aA

Letters of the same type, do not differ among the managements within each depth. Letters that are the same as the upper case, do not differ in the same depth. Both analyzes were performed by the Bonferroni test

Although there were no management differences in each fraction alone (Table 3), there is evidence of difference in labile forms when added together (Table 6). The total labile OP was higher in NTL in both layers, being higher in the 0-0.1 m layer (67% for CLI and 129% for CTL), and in the 0.1-0.2 m layer this difference is (214% for CLI and 187% for CTL).

Table 6. P fractions collected In the 0-0.10 and 0.10-0.20 m layer,on a Distroferric Red Latosol

	Profundidade (0-0,10 m)			Profundidade (0,10-0,20 m)		
	Sistemas de Manejos			Sistemas de Manejos		
	CLI	CTL	NTL	CLI	CTL	NTL
TLAB	524.43 b	383.13 b	879.11 a	177.73 b	194.49 b	557.77 a
ILAB	118.90 a	111.45 a	124.72 a	59.75 a	57.63 a	81.95 a
TMOD	1608.24 b	1549.98 b	2014.14 a	1313.81 a	974.73 b	1146.71 ab
IMOD	410.02 b	411.87 b	533.70 a	370.84 b	400.94 b	549.98 a
OMOD	1198.22 b	1138.11 b	1480.45 a	942.97 a	596.73 b	573.79 b
POUCL	783.07 b	838.34 ab	878.39 a	885.39 a	854.28 a	648.28 b
INOR	528.92 b	523.31 b	658.42 a	430.59 b	458.57 b	631.93 a
ORG	1603.76 b	1409.79 b	2234.84 a	1060.95 a	710.64 b	1072.55 a
TOTAL	2915.7 b	2771.4 b	3771.6 a	2376.9 a	2023.5 b	2352.8 a

Letters of the same type, do not differ among the managements within each depth by Bonferroni test

The different systems of land use and management may interfere with the dynamics of P, and may promote changes of the P compartments. When comparing the sum of all the extractions (NaOH at 0.1 and 0.5 mol L-1), we found that at the most superficial depth (0-0,1 m) the NTL presented higher levels of P both adding only the organic forms , As adding the inorganic forms or adding all the fractions (organic and inorganic together). At the depth of 0.1-0.2 m we found increase in the organic forms for the CLI, while the inorganic forms had higher values in the NTL.

Total soil PO (sum of all fractions of P) presented higher values in the 0-0.1 m depth for the NTL, while in the 0.1-0.2 m layer these values were also higher in the CLI system besides Of the NTL (Table 6). The great contribution of the organic P for this purpose is verified. In this way, it is verified that the total P content will be increased in depth in both the NTL and the CLI system. Conservationist systems tend to improve the distribution of soil fertility attributes, as reported by Lourente et al. (2011). However, it can be observed that in the NTL the labile fraction of P presents a higher percentage in relation to the other systems, in both depths.

CONCLUSION

The results showed that the no-tillage system provides better conditions for availability of phosphorus. The moderately labile P fraction was the largest fraction of the total P in the three management systems. The P of greater lability is the depth 0-0.10 m soil for all managements. Organic P was higher in the no-tillage system regarding the integration crop farming and conventional tillage systems.

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M.7.P

EFFECTS ON SOIL OF FOREST PLANTATIONS OF HIGH DENSITY AND SHORT ROTATION.

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ABSTRACT

Costa Rica in search of carbon neutrality for 2021 has opted for an energy matrix based on renewable energies, in this interest forest biomass has been considered as one of the likely sources to replace fossil fuels. From this perspective, short-rotation forest plantations are a business opportunity for foresters in Costa Rica, but at the same time an opportunity in the recovery of degraded soils. Joining the interests of private companies dedicated to the industrialization of sugar and coffee, and the Technological Institute of Costa Rica, the first high-density forest plantations of Costa Rica were established since 2012. At least 2 species (*Eucalyptus tereticornis*, E*. saligna, Gmelina arborea, Tectona grandis,* and *Dipteryx panamensis*) were tested per site at densities of 5,000, 10,000 and 20,000 trees per hectare at each of the 5 sites in the Zone Huetar Norte, Zona Sur and Central Zone including low and high zones. Results of the first harvests show productivity levels above 50 Mg ha^{-1} in three years, revenues of \$ 5000 USD ha^{-1}, these results are promising for the industrial sector in Costa Rica that markets more than 200 Mg per day of Forest biomass. In addition, an increase in the levels of organic matter was found, without causing a nutritional loss in the soil; the previous one of the most promising results from the soil and environmental point of view. For now, research continues to improve forestry adapted to this type of plantations.

INTRODUCTION

Costa Rica is known worldwide for being a mega-diverse country, with a very small extension, only 51 000 km^2, and recently for having an energy matrix with more than 90% of renewable energy sources, which in the rainy season caters to almost 5 million people with renewable sources, among them mainly hydro, geothermal and wind energy.

Sugar mills and coffee mills saw interest in producing forest biomass energy for two different types of problems, in the case of the Sugar mills that cane harvest extends only 6 months a year, which has the consequence of having an industry 6 months off; and on the other hand the coffee mills of receiving a variability biomass for the boilers that makes it difficult to achieve the temperatures required for the processing of the coffee and keep them constant.

In 2012, high-density forest plantations were started in Costa Rica in order to produce energy on marginal land for agricultural production. This implies that these lands have physical or chemical limitations that limit production levels.

METHODOLOGY

Characteristics and location of the tests

The trials were established in the years 2013, 2014 and 2015 on land belonging to private companies (Taboga, Coopetarrazú and Puro Verde) and in higher education institutions, TEC, UNA and CATIE, seeking a representativeness of climates and potential interested parties. The country's coastline representing both dry and wet lowland areas of the Pacific slope as well as areas on the Caribbean slope including High zones; with Inceptisols (Udepts, Usteps) and Ultisols (Udults, Humults). The landscape includes slopes of 1% up to 60%. The test areas have an average extension of 1.5 ha. In each of the sites, weed control was performed until the closure of the canopy.

Experimental design, species and densities

In the trials, randomized complete block designs were established with 3 replicates. The dimensions of the experimental units varied between sites because they depended on the availability of land from the owners, the experimental units varied from 100 to 900 m^2. Each experimental unit considered a buffer zone and an internal treatment plot of 49 trees for all cases. The treatments considered planting densities of 5,000 trees ha^{-1} (1,52 x 1,32 m), 10,000 trees ha^{-1} (1,07 x 0,93 m) and 20,000 ha^{-1} trees (0,76 x 0 , 66 m). All staggered and the species evaluated were *Gmelina arborea* and *Gliricidia sepium* in Taboga; *Eucalyptus tereticornis* and *E. saligna* in Coopetarrazú; *Tectona grandis, Gmelina arborea, Erythrina poepigiana, Dipteryx panamensis* and *Gliricidia sepium* in Puro Verde; *Eucalyptus tereticornis* in TEC; *Gmelina arborea* at UNA and finally *Eucalyptus tereticornis, Gmelina arborea* and *Gliricidia sepium* at CATIE.

Tree growth was evaluated in an internal plot of 49 trees within the experimental unit measuring the height and diameter of the base at 10 cm from the soil and diameter of chest height after reaching 1.3 meters in height.

Sampling and laboratory analysis.

Mineral soil samples were obtained at depths of 0-20 cm and 20-40 cm prior to planting the trees (baseline) and at 12 and 24 months of age by means of a composite sampling (8 sampling points) in each unit experimental. The samples were transported to the laboratory and dried in the air for at least 24 hours, sieved to 2 mm (ATMS mesh No. 10) to determine the percentages of soil, gravel and biomass in the sample.

Complete chemical analyzes were performed on samples in a Laboratorio de Suelos y Foliares del Centro de Investigaciones Agronómicas at the Universidad de Costa Rica.

Statistical analysis

All analyzes were performed by depth. The effect of density was analyzed using ANOVA, and then separated the means using the Tukey test with 95% confidence using the statistical software SAS Version 9.1.

RESULTS AND DISCUSSION

Table 1. Content biomass production and biological utilization coefficient (BUC) of nutrients in the stem in dendroenergy plantations of *Eucalyptus saligna* and *E. teriticornis* two years and three stocking.

Specie	Trees stocking	Stem Biomass	Stem Nutrients					
	tree ha $^{-1}$	Mg ha $^{-1}$	N	P	K	Ca	Mg	BUC
E. saligna	5 000	3,56	13,9	1,4	20,3	11,8	1,4	73
E. saligna	10 000	9,95	31,8	4	53,7	21,9	3	87
E. saligna	20 000	14,42	66,3	7,2	90,8	34,6	4,3	71
E. teriticornis	5 000	2,36	30,9	5	28,1	24,4	2,3	26
E. teriticornis	10 000	6,94	23,6	2,8	34	18,7	1,4	86
E. teriticornis	20 000	10,37	37,3	5,2	60,1	30,1	3,1	76

Table 2. Distribution of soil nutrients in two different depths and in three stocking, dendroenergy plantations of *Eucalyptus saligna* and *E. teriticornis*, two years old.

	pH	Acidity	Ca	Mg	K	CEC	SA	P	Zn	Cu	Fe	Mn	EC	C	N
	H2O			cmol(+)/L			%			mg/L			mS/cm	%	
Depth															
0-20	4,7 a	4,90 b	0,24 a	0,33 b	0,15 b	5,8 b	84,51 a	2,94 b	1,53 a	2,17 a	417,72 b	18,67 b	0,14 b	5,53 b	0,42 b
20-40	4,89 b	4,07 a	0,42 b	0,17 a	0,11 a	4,6 a	88,38 b	2,22 a	2,09 b	1,71 a	302,00 a	8,83 a	0,10 a	3,48 a	0,28 a
Tree ha^{-1}															
5000	4,88 b	4,52 a	0,4 b	0,21 a	0,14 a	5,33 a	85,23 a	2,42 a	1,76 ab	2,07 a	328,75 a	15,67 a	0,11 a	4,29 a	0,34 a
10000	4,82 ab	4,34 a	0,26 a	0,27 a	0,13 a	4,94 a	87,83 a	2,67 a	1,64 a	1,67 a	341,83 a	10,17 a	0,13 a	4,80 a	0,37 a
20000	4,7 a	4,59 a	0,34 ab	0,27 a	0,14 a	5,33 a	86,27 a	2,67 a	2,04 b	2,08 a	409,0 a	15,42 a	0,13 a	4,43 a	0,35 a

Different letters indicate significant differences of Tukey test, $\alpha = 0.05$

The results showed low pH values, characteristic of the study area. The cation exchange capacity (CEC) showed values close to the general critical level for the extractive solution (5.0 cmol (+) / L), this characteristic is closely linked to the soil fertility and the availability of nutrients for the plants. (Krull et al., 2004). This is evidenced by the result of the analysis that indicated low concentration of P in the soil, and high concentrations of Fe and Mn.

Eucalyptus species extract different amounts of nutrients and accumulate them in different ways in the tissues. In juvenile ages of Eucalyptus plantations, da Silva et al. (2001), determined that 7.5% of the stem corresponds to the same bark (functional phloem) that accumulates the greatest amount of N, P, K, Ca and Mg as the branches. In the case of the present study the wood was evaluated as: wood + bark, where a similar behavior of absorption is observed in both species, however, it was mentioned previously that *E. saligna* produced greater amount of biomass than *E. teriticornis*, could indicate that the latter has a higher demand for nutrients per unit of biomass produced.

The explanation for such a difference may be related to the absorption capacity of the species. Alvarado and Raigosa (2012) present different values of biological utilization coefficient (BUC) for eucalyptus species of different ages, where it is notable that the behavior among species varies as a result of the distribution or utilization of the nutrients and the amount of biomass Produced and / or its absorption capacity. For this study, the CUB of both species behaved very similarly in the evaluated densities with the exception of *E. teriticornis* in the treatment of 5,000 ha^{-1} trees, related to the low biomass production of this treatment.

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M.8.P

THE SITUATION OF ANIMAL-POWERED LOGGING IN STATE-OWNED FORESTS OF HUNGARY

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INTRODUCTION

Skidding/logging means the process of dragging the wood from the felling site to the landing area, in other words to the concentration yard (http1). The process of logging requires high attention to the soil; the wood stand and the saplings (Firbás, 1996). Skidding could happen manually, by draught animals or machines (Andrésiné & Andrési, 2008). In 1949 the only known and used skidding method was horse logging, however the mechanization of forestry developed quickly in Hungary (Keresztesi, 1971, 1982; Hegyi, 1978). By the time of 1979 more than the half of the skidding was mechanized in the Hungarian forests (Keresztesi, 1982). In the 1930's machines begin to replace animal power and by the 1960's totally undertook its role in the Southern United States (Russell, F. and Mortimer, D. 2005). Intermittently, since the 1970's, horse logging has been returning to small-scale forestry worldwide (Shrestha et al., 2005). Nowadays mechanized skidding is the most common in practice, however there are still future facilities for the draught animals too (Firbás, 1996). Namely, animal-powered logging is considered to be gentle to the soil (Zimmermann, 1994), reduces damage to residual trees and seedlings (Rodriguez and Fellow, 1986; Wang, 1997).

Extraction routes or skid trails allows people, animals and machines to reach the felling site of the forests until the landing. (Pankotai – Madas, 1956). Throughout these ways, the soil could be easily compacted under the weight of the machines (Varga, 2001). It is stated that the extraction routes have a great emphasis on the productivity of horse logging (Melemez et al., 2013). According to Wágner (1986) and Oprea (2008, cit. in Borz and Ciobanu, 2013) horse logging could work in the widest range in the fields, although it is not profitable compare to the machines when the distances are longer than 100 meters. In distances greater than 50 m skidders are cheaper than horses, if the skid trails have already been made. However, if new skid trails need to be built, one-horse logging group is cheaper than skidders in distances up to 200 m (Magagnotti and Spinelli, 2011).

Horse skidding is an appropriate practice for small-scale forestry (Bray et al., 2016) and for local farmers because of the small investments and operating costs (Oprea, 2008, cit. in Borz and Ciobanu, 2013). In addition, Bray et al. (2016) proved that it has significance in large-scale industrial community forestries too within Chihuahua, Mexico.

Its advantages lie on that it is more cost-effective than mechanized skidding, creates more employment and has less impact on the environment due to decreased greenhouse gas emissions (Bray et al., 2016, Engel et al., 2012). More than the half (ca. 60%) of its greenhouse gas emissions is caused by the transportation of horses to the extraction site (Engel et al., 2012). Besides that, fossil fuel consumption when skidding is made by horses is from 8 to 20 times less than with the skidders (Magagnotti and Spinelli, 2011). Environment- and nature- friendly aspects turn animal power in favorable position (Rajczi, 2010). There will be growing demand for the work of horses to spare the saplings (Káldy, 1968 and Wágner, 1970). Valló (2012) and Dudás (2013) state that horse logging has been living its renaissance these days. Although, there is only a few literature and research about this environment-friendly log transportation method.

METHODS

The goals of the research include, among others, the following: determine the frequency of horse logging/skidding in state-owned forests of Hungary, collect most of its advantages and disadvantages, furthermore compare its effects on the vegetation and on the soil to the skidders' (forestry machines used for skidding). To determine its frequency in state-owned forests all of the Hungarian forestries - altogether 116 - were surveyed by phone. Semi-structured interviews (Héra és Ligeti, 2010) were carried out with the contractors who apply horses for logging (horse loggers) in the fields. The main aim of the questions was to point out the most important attributes of horse logging. Those plots of the interviews were selected and visited personally which have high forest cover and with the purpose to collect representative data all across Hungary. Altogether 11 forestries, 16 horse loggers were observed in the 17 different fields, among them 2 were in privately owned forests and 1 was investigated in 2 different places.

To exactly define and quantify what kind of advantages and disadvantages the horse logging has and what is the degree of them investigations of the vegetation and the soil are essential. Before the surveys correct sample plots must be selected according to the forest management plan and to the description sheet describing the actual state of an exact forest subcompartment. The sample plots with the same factors - defined in the above-mentioned documents - need to be arranged into pairs. So that the effects of both skidding method become comparable with each other in different subcompartments too. The following factors need to be taken into consideration while selecting and pairing the sample plots: the average age of wood stand, the dominant tree species, the genetic soil types and the physical characteristics of the soil, the degree of the slope and the weather conditions during skidding. The vegetation survey defines the Braun-Blanquet (1951) cover-abundance scale, Ecology Indicators (Borhidi 1993, Simon 1988), Social Behaviour Types (1993), Raunkiaer (1943) and Pignatti (2005) life-forms. Values of Sociability of the considered individuals could be added to the vegetation survey too. The degree of soil compaction is an important indicator and it is easy to examine throughout the penetration resistance and soil moisture with a cone penetrometer, furthermore determine the depth of the compacted layer with a hand probe is also necessary (Duiker, 2002). These investigations are put into effect along 2 meter longs and 0,5 m wide transects in every second month.

RESULTS

The number of those forestries who employed horse loggers during the interval of the research is 30 out of 116 forestries which is 25.86% of all forestries. They often apply horses every year. Besides them, 9 forestries (7.76 % of all) reported that horse logging is used only occasionally, approximately twice or three times yearly, which means 100-200 m^3 wood getting logged by horses. 3 forestries apply horses for game feeding and hunting with horse carriages but not for skidding. Other 3 forestries have their own horses and use them for log transportation. 66.38% of all forestries do not apply horses in any way, they substitute their work for mechanized units. Some of them associated the usage of horses with an ancient forestry proccess.

Altogether 16 contractors were observed personally, examining how they use animal power in practice. One of the private contractor was visited in two different areas, because of this his work method was calculated to the averages only once, except the extinction of the given area. Among the visited 17 places there are 15 which are protected natural areas and 3 of them are highly protected natural areas. Horse logging is frequent in the highlands which are not unfolded and have continuous forest cover. The biggest concerned area by horse logging is 23 ha, the smallest area is 0.5 ha. The average extension of the concerned area by horse logging is 6.05 ha. The most common occasion when horses are used is during thinning. Borz and Ciobanu (2013) also reported that horses are mostly applied in very young and dense stands where thinning operations are done. In these stands, horses could work efficiently with low impact and manoeuver easily as least damage as possible. The most commonly used horse type is the cross-breed with a rate of 37%. Those contractors who apply cross-breeds prefer their usage because of their fast pace and smaller size. Others swear on the cold-blooded (in other words: draft) types because of their huge capacity and calm temperature. The capacity of horses depends on many factors, for example the relief, the weather, the consumed forage and the team-work. The average daily capacity of a single horse is 15.5 m^3. The average quantity logged by a horse in one round is 0.81 m^3. The most common way to make the log move is to drag it with a chain or a rope to the horse.

CONCLUSIONS

The main advantage of horse logging is that it is harmless for the topsoil, the wood stand and saplings. It does not hinder the grove of the saplings since it makes less damage to the topsoil. In those forests, which has dense stand, horses could maneuver easier than machines and leave more wood untouched. Its environmental significance is that no harmful substance emission occurs, no harmful fuel presents and the forage consumed by the horse can be produced by the contractors or the locals as well. This fact correlates with the principles of sustainable development. Another advantage of this method is that it provides possibilities for the native horse breeds and types. The horse logging is a nature- and eco-friendly method of transportation of logs which is required mainly in protected natural areas.

The main disadvantage of this method is that the capacity of horse logging is less than that of the machines. The contractors have to find other jobs during the vegetation period while the timber extraction is paused and have to feed their horses without a reasonable income by them. It takes rather significant time to travel to the working area with horses and it creates more liability. Other significant difficulty is that working with them requires a specifically strict and hard way of life. Developed equipment would increase the efficiency of horse logging and decrease the differences of output values between machines and horses. Horse logging is overall a still viable and evermore sustainable method within forestry, it has benefits for the forests, and gives possibilities for the people who work in the field of timber extraction.

Vegetation and soil surveys written in the methods below are an essential part of this topic and need to be accomplish in the practice in different subcompartments as many pairs as possible to collect enough evaluable data. The methods of the surveys could be developed still according to the suggestions and the practice.

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M.9.P

SPATIAL VARIABILITY OF SOIL AGGREGATE STABILITY IN A DISTURBED RIVER WATERSHED IN KENYA.

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ABSTRACT

Analysis of spatial distribution of soil properties like soil aggregate stability presents an important outset for precision agriculture. The study area was classified into different landscape units according to physiographic features namely: mountains, plateaus, uplands, valleys, pen plains, alluvial plains, lacustrine plains and hills and a map was drawn. The objectives of this study were to evaluate the effects of landscape and land use interaction on the spatial variability of aggregate stability. The variability of aggregate stability exhibited spatial dependence (SDP) which helped in the generation of a spatial dependence index (SDI) that was described using semivariogram models. SPD $_{Gaussian}$(%) ≤ 25% gave a weak spatial dependence, moderate spatial dependence was given by 25% < SDP (%) ≤ 75% and strong spatial dependence by SDP (%) > 75%, while SDI $_{Gaussian}$ (%) ≤ 25% gave a strong spatial dependence index while moderate spatial dependence index was indicated by 25% < SDI (%) ≤ 75%, and weak spatial dependence index SDI (%) > 75%. Mean Weight Diameters (MWD) of 0.25 – 0.45 represented unstable soils mostly found in wetlands occurring in valleys, mountains, plains, and depressions in hills, 0 55 –0.62 represented moderately stable soils mostly in agricultural and grassland areas which include plateaus, uplands, and plains, while 0.62 – 0.92 represented stable and very stable soils being found in forested areas, mountains and hills. Various interpolation (kriging) techniques capitalized on the spatial correlation between observations to predict attribute values at unsampled locations using information related to one or several attributes that helped in the construction of an aggregate stability prediction map using Empirical Bayesian kriging (EBK) technique.

KEYWORDS: land scape, Spatial Variability, semivariogram, Geostatitics, aggregate stability, kriging

M.10.P

NUTRIENT DISTRIBUTIONS AFFECTED BY LANDFORM IN TROPICAL INLAND PEAT IN RIAU, INDONESIA

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Tropical peatlands in Indonesia with dome formation are generally oligotrophic due to acidic condition and lack of mineral enrichment. Furthermore, drainage of peatland for intensive cultivation accelerates decomposition of peat and leads to release of acid that restricts mineral nutrients availability for crops. The aim of this study is to determine mineral distributions (Ca, Mg, K and Na) in active layer of peatland and the potential of total Ca, Mg, K, Na, Fe, Al in relation to peat thickness and distance to mineral soil. The ultimate goal of the present study is to progress toward effective fertilizer management based on mineral distribution and to optimize crop productivity.

The study site was an inland basin peat (Anshari et al., 2010) under oil palm plantation situated between Siak River and a small hill in Siak-Riau, Sumatera, Indonesia (Fig 1). The site was at approx. 11 m a.s.l with about 60 km from the nearest coastline. Current fertilizer was applied equally for whole farm but not based on nutrient distributions. The landscape of peat dome was analyzed based on 32 systematically plotted points (Fig 1). We collected peat samples from the active layers at three depths (0–15 cm, 15–30 cm and 30–50 cm). In order to investigate chemical composition through the peat profile, coring was also conducted on a transect with six profiles from the river toward the adjacent hill (Fig 1). Water table data were also collected during sampling, ranging 80–120 cm below surface. Electrical conductivity (EC) and pH were determined based on a sample: water ratio of 1:10. Based on 1 M ammonium acetate (pH 7) extraction, exchangeable Ca and Mg were determined using atomic absorption spectroscopy, whereas exchangeable K and Na were determined using flame emission spectroscopy. An aliquot of sample was digested with HF and $HClO_4$ then total Ca, Mg, K, Na, Fe, and Al were determined by an intercoupled plasma atomic emission spectrometer.

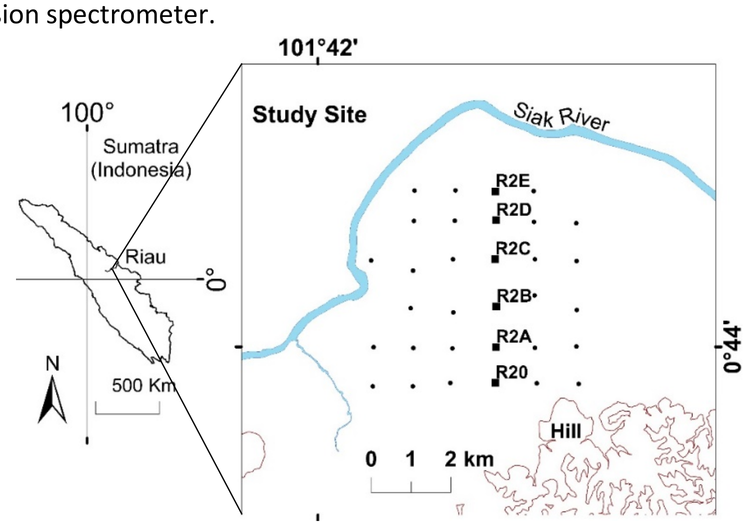


Fig 1. Sampling design and the profile transect (marked with square)

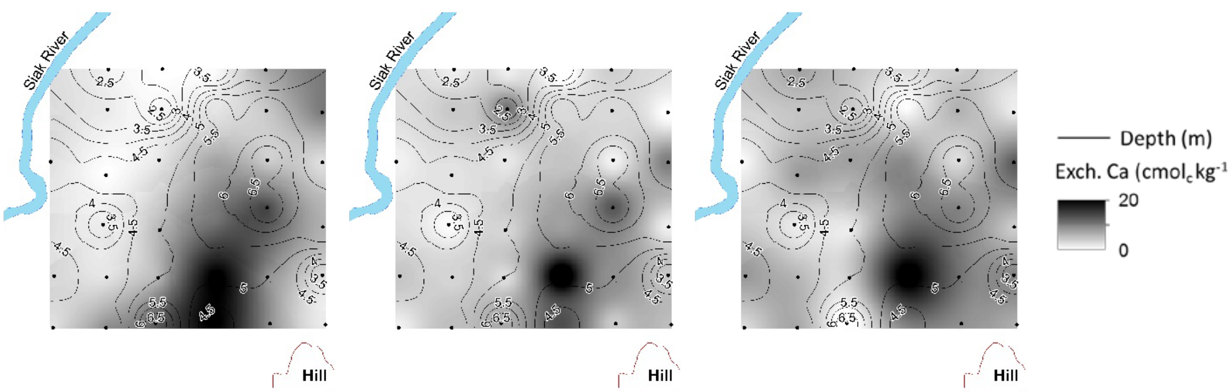
Elevation ranged from 7.5 to 16 m a.s.l where higher elevations were near the hill with peat thickness of > 3 m. The mean peat thickness and distance to the the hill were 4.6 m and 2.7 km, respectively. Although EC and pH were similar, general physico-chemical analyses revealed that exchangeable Ca, Mg, and K were high at 0–15 cm depth and tended to decline at the lower layers (Table 1).

Table 1. General physico-chemical properties of active layers

Layer (cm)		BD[a]	pH	EC	Exchangeable cations ($cmol_c\ kg^{-1}$)			
		($g\ cm^{-3}$)		($\mu S\ cm^{-1}$)	Ca	Mg	K	Na
0–15	Mean	0.26	3.7	185	8.29	2.73	0.65	0.13
	SD[b]	0.04	0.3	57	5.08	1.10	0.34	0.07
15–30	Mean	0.19	3.7	179	6.51	2.51	0.50	0.10
	SD[b]	0.04	0.3	92	3.85	1.21	0.33	0.19
30–50	Mean	0.16	3.7	184	3.71	2.12	0.42	0.10
	SD[b]	0.02	0.2	90	2.20	0.85	0.37	0.12

[a] bulk density; [b] standard deviation

Interestingly, exchangeable Ca in 0–15 cm layer (surface) tended to be higher on deep peat. Up to 30 cm depth, exchangeable Ca were considerably higher, particularly on peat thickness > 3 m (Fig 2). This pattern, however, contradicted with the previous finding studied by Watanabe et al. (2013) where lower content of nutrients were on deep peat.



a) b) c)

Fig 2. Distribution of exchangeable Ca in different peat thickness (m) at a) 0–15cm, b) 15–30 cm, and c) 30–50 cm

Growing concern on factors of the finding above leads us to investigate the influence of the surrounding landscape on chemical composition. We examined the relationship between exchangeable Ca and the adjacent hill. We found that higher exchangeable Ca up to 30 cm depth that were on higher points (20, 2A, and 2B) were also concentrated near the hill (0–4 km) (Fig 3). The farther points from the hill had lower contents of exchangeable Ca. Higher elevations were surrounded by the hill indicating the configuration of study site were influenced by the hill.

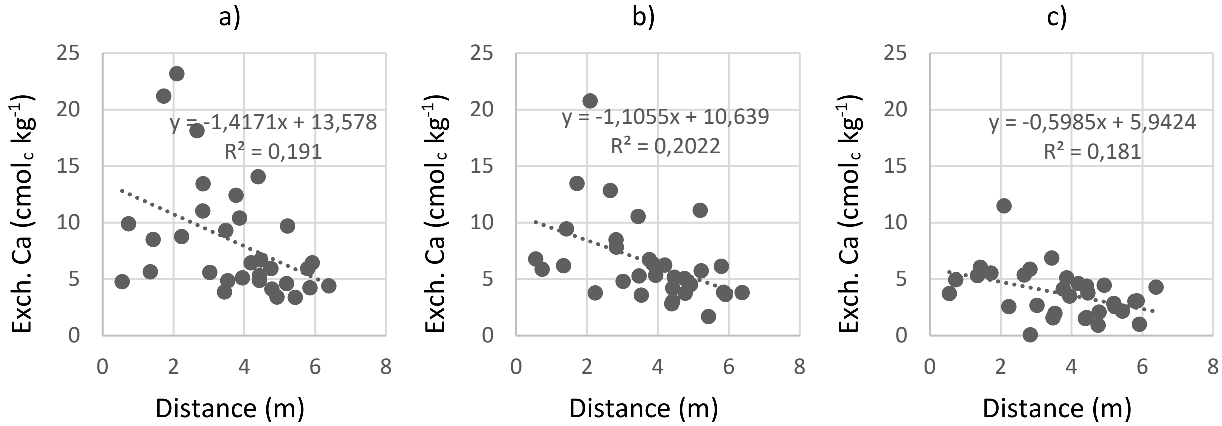


Fig 3. Correlations between distance to the hill and exchangeable Ca at
a) 0–15cm, b) 15–30 cm, and c) 30–50 cm.

In order to confirm the influence of hill on peat, profile analyses were also done. In the surface, total Al and Fe as two of major elements of mineral soil were higher near the hill. In 20, 2A, and 2B, total Al and Fe were considerably higher at surface, ranging from 2.2–3.8 g kg^{-1} and 1.7–2.3 g kg^{-1} ,respectively. They tended to be lower in the middle layer and abundant in the bottom layer (Fig 4).

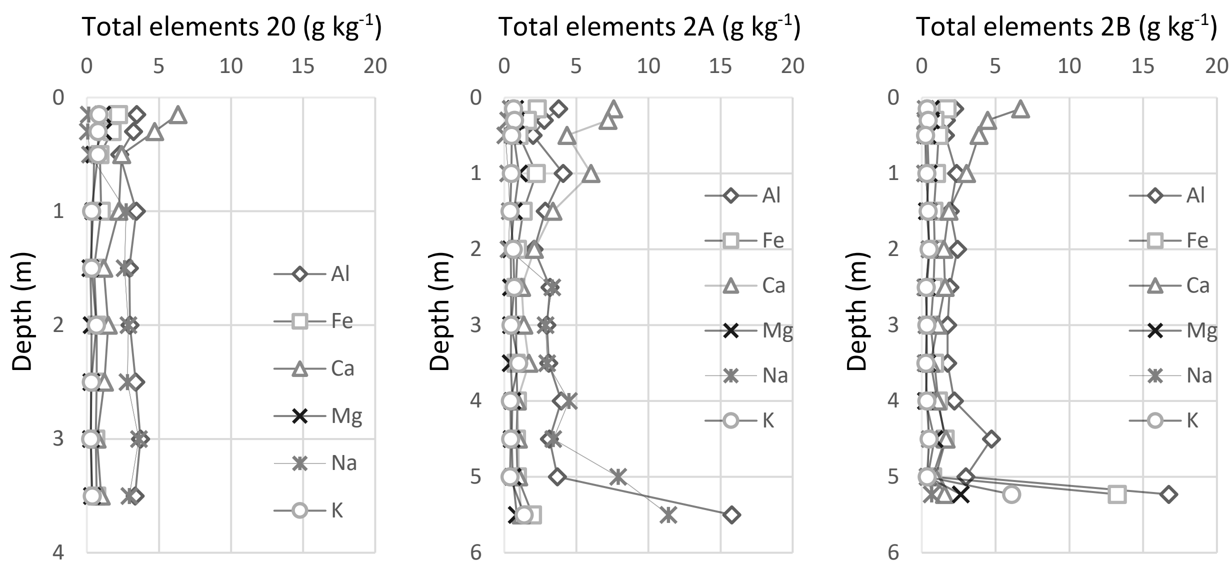


Fig 4. Mineral contents of peat near the hill

Differently, total Al and Fe at the surface of 2C, 2D, and 2E were approximately one third of those were near the hill (Fig 5). Instead, they were determined high in the middle of profiles. These patterns implied that tidal and riverbank were considered as the factors since relevant points were in the lower elevation and close to Siak River with frequent inundation during peat formation.

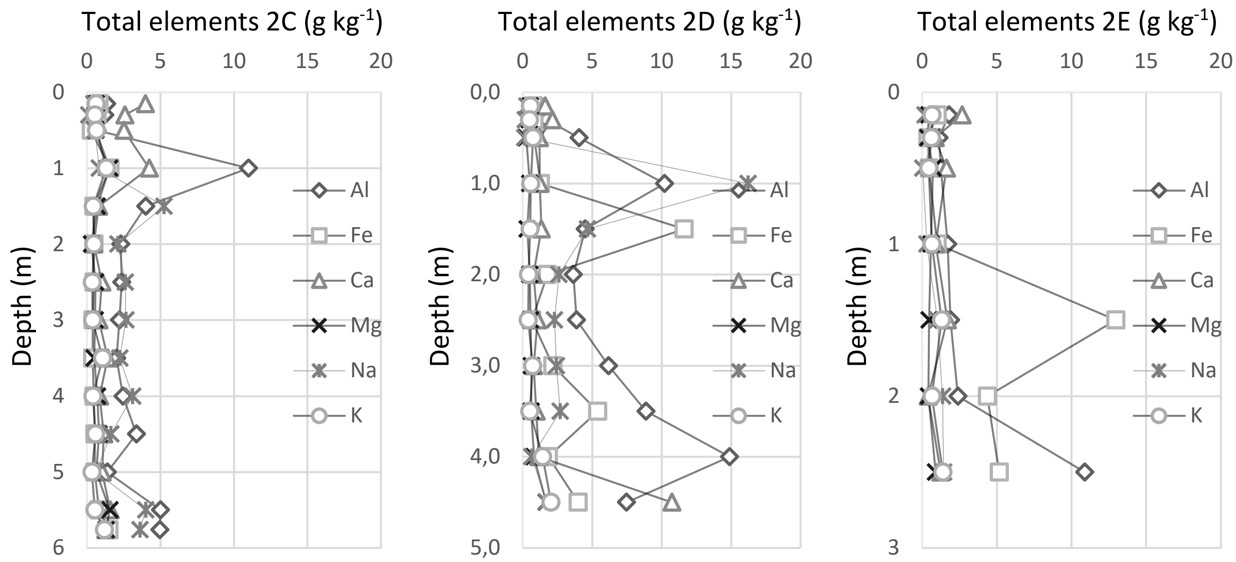


Fig 5. Mineral contents of peat far from the hill.

CONCLUSION

Exchangeable Ca, Mg, K and were high on the surface (0-15 cm) and slightly declined with depth. For exchangeable Ca, the content were distinctly higher on deep peat (peat thickness > 3 m) that is close to the hill, implying mineral enrichment to the study site. Total Al and Fe were higher at the surface close to the hill, confirming that the hill influenced mineral distributions on the study site. Based on each mineral distributions, the composition of fertilizer may be adjusted especially to the area > 4 km from the hill to optimize crop productivity.

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M.11.P

SOIL MONOLITHS OF THE PYRENEES. A LONG TERM PROJECT OF THE CARTOGRAPHIC AND GEOLOGIC INSTITUTE OF CATALONIA

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INTRODUCTION

With the vision to be a reference center for the information and dissemination of mountain soils, the Territorial Support Center of the Pyrenees (CSTP) of the ICGC (Institut Cartogràfic i Geològic de Catalunya), leads several projects related to geology and to pedology. The project "Soil monoliths of the Pyrenees" which began in 2014, has been strengthen with the opening of the embryo of the Center of Interpretation of Soils of the Pyrenees (CISP) in December 2016, as a permanent preservation and exhibition of soil monoliths and related information.

Soil monoliths are used as an element for dissemination and education about soil, its use and the interaction of the environment, climate and humans. After a laboratory process, which includes in some way the artistic skills of the experts, field collected samples of soil profiles are preserved in almost their natural undisturbed conditions. Each monolith also tells a story through its morphological features (e.g., texture, structure, color, horizon thickness) and many of the soil-related limitations to land use are often apparent (Haddad et al., 2009; Krzic et al, 2013, Torres and Madero, 2007). On display, they attract considerable attention. Moreover, well-constructed soil monoliths are beautiful and most of them look like real works of art (Giencke 2014).

The advantages that offers a collection of monoliths of soil in comparison with other elements or activities has been discussed, among others by Lawrie and Enman (2010): they are transportable elements, can be used many times, allow easily compare different soil profiles or group them, can be observed indoor, are very suitable for groups, can be exposed for long periods (monoliths have been made for over 100 years; i.e. Vanderford, 1897). In addition, the use of monoliths is complementary to the field work, and to expand and compare information obtained from direct observation. Conversely, they have also some disadvantages compared to open pits, as for instance that properties as moisture, hardpans or active biological activity among other, cannot be displayed when the monolith has been separated from its natural setting (Lawrie and Enman, 2010).

SUMMARY OF METHODS

The project began in 2014 by searching and comparing related bibliography (Van Baren and Bomer, 1979; Haddad et al., 2009; Más-Martinez et al., 2010 and Torres et al, 2013, among others) and visiting the ISRIC World Soil Information in Wagenningen in order to share and discuss the methodology for the elaboration of soil monoliths, and the requirements for their preservation and display. All this actions permitted to define the methodology to be followed, which in several aspects is in close relation to that used for the extraction, transportation and preparation of big to mid-sized fossil bones for paleontological studies.

In 2015 the implementation of the defined methodology resulted in the first 4 monoliths. Those monoliths were extracted from different locations of Pallars Jussà (Southern Pyrenees, Lleida, Spain).

In our experience, the methodology for making accurate soil monoliths requires multiple processes for each type of soil, but they are contained in the two known main phases: the sample extraction and its preservation. Figure 1 is a summary of them.
Just to mention one of the innovations of the preservation methodology used in respect to that defined by Torres (2003) and Van Baren (1979), is the successful use of polyvinyl acetate replacing the cellulose lacquer and the organic solvent as consolidation agent.

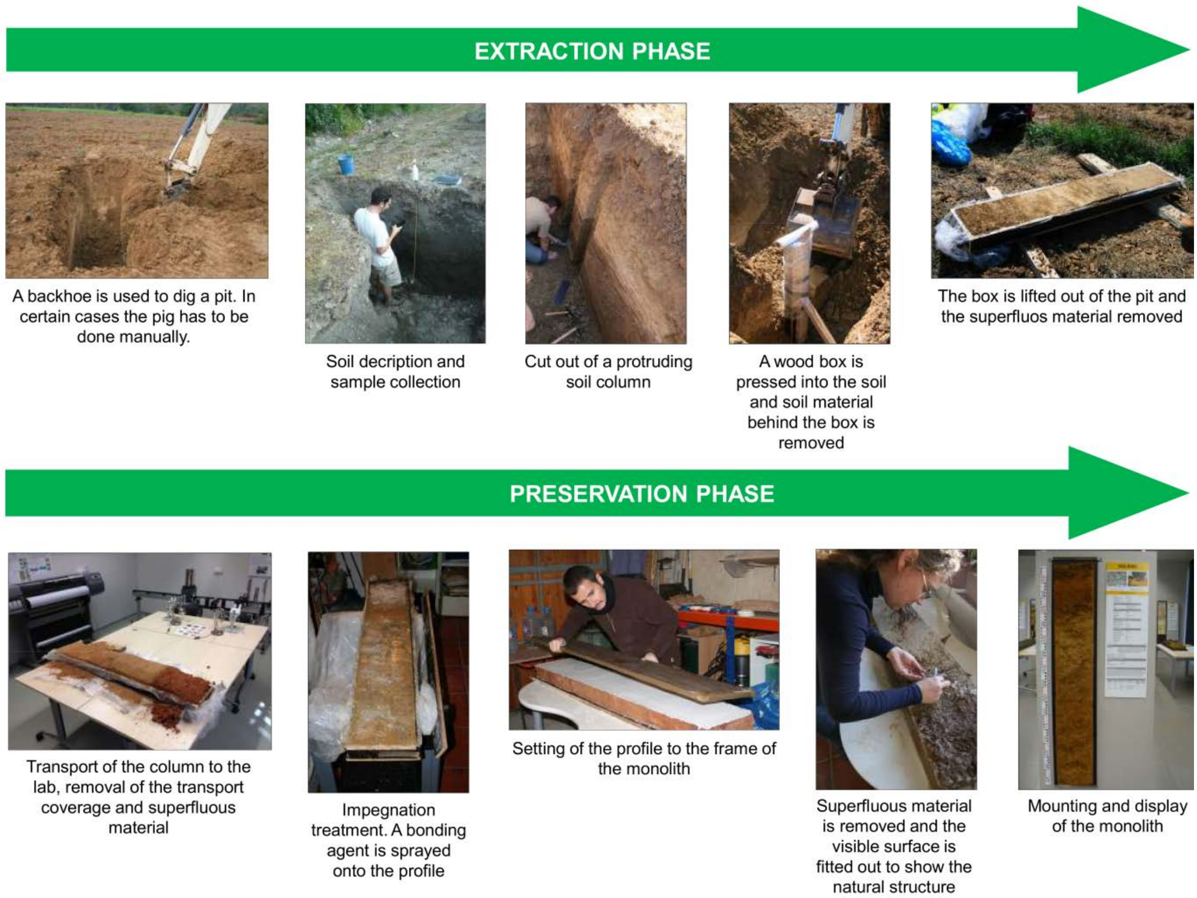

Figure 1. Overview of the two main phases for preparing a soil monolith

After the consolidation there are two other phases: the exhibition and display and the monolith preservation. The chosen display system for exhibition is a 180cm large, 80cm width wooden structure with a support system which provides a slightly tilted position of 80 degrees, so that the monolith weighs down on the wood mount more equitably. The monolith is positioned to one side of the board. This leaves space at the other side to include additional information as soil profile description, maps, images of the landscape, etc. this information can be modified depending on the intended audience (researchers, students, farmers, families, etc.).
The preservation procedure has been defined taking into account the requirements for the long-term preservation of the monoliths. Since the chemical products used on the impregnation usually decompose under the influence of ultraviolet light (Van Baren, 1981) the monoliths should not be exposed to full daylight, using artificial illumination. Although it seems that environmental conditions requirements are not very restrictive, a data logger has been installed in the exhibition room to have a constant record of humidity and temperature.

RESULTS

During 2015 and 2016 nine soil monoliths 25 cm width, approximately 5 cm thick and 150 cm wide as maximum have been elaborated in the project. Table 1 summarizes their main characteristics.

Soil Monolith	WRB Classification	SSS Classification	Land Use
SOLS001	Haplic Calcisol	Typic Calcixerept	Vineyard
SOLS002	Calcic Luvisol	Calcic Haploxeralf	Quarry
SOLS003	Haplic Vertisol	Typic Calcixerert	Cereal crop
SOLS004	Haplic Calcisol	Typic Calcixerept	Vineyard
SOLS005	Petric Calcisol	Petrocalcic Calcixerept	Vineyard
SOLS006	Haplic Fluvisol	Typic Ustifluvent	Walnut crop
SOLS007	Oxyaqüic Fluvisol	Oxyaquic Cryofluvent	Grassland
SOLS008	Fibric Histosol	Haplofibrist Fluvaquentic	Grassland
SOLS009	Albic Podzol	Typic Haplocryod	Fir forest

Table 1. List of the to date monoliths made by ICGC.

The methodology for soil monoliths elaboration and conservation has been defined, including extraction, consolidation or preservation, display and exhibition and conservation. It solves some difficulties that can occur during the process of extraction (i.e. the use of a wooden box has been proved as useful in most of the cases, and the use of a polyurethane foam casing has proven to be a very successful solution for stony soils).

The analysis of the consistency and hardness of the resulting monoliths demonstrate that polyvinyl acetate is a very suitable product for the impregnation and consolidation of soil profiles, but this has to be tested after 10 years, and to minimize the cracks frequency, a strict control of the drying period is advisable.

An individual display system has been designed in order to assure the stability of the samples and their conservation. This system allows the exhibition of the profiles regardless of the structure of the hall. As Figure 2 shows, monoliths can be grouped around a particular theme as soil formation factors, physical characteristics (e.g. texture, color), agricultural behavior, etc.

Figure 2. Partial view of the current exhibition of the CISP at the CSTP

Finally, the conservation conditions have been established. The use of Tyvek fabric has proven to be very useful in order to avoid UV light exposure and to prevent fungal growth due to the limitation of transpiration.

CONCLUSIONS

An improved methodology for the elaboration of soil monoliths has been defined, comprising the extraction of the sample, the preservation, display in individual support systems and conservation. However, it is not possible to confirm at this moment the adequacy of the conservation methodology as the monoliths are only 1 and 2 years old and a minimum of 10 years is needed.

Today, the CISP holds nine soil monoliths and during 2017 5 new monoliths will be elaborated. It is planned to annually increase their number up to 52 samples in 2021, in order to be able to represent the maximum variety of soils of the Pyrenees and pre-Pyrenees.

As an exhibition of the most representative soil types in the Pyrenees, the CISP will be a very useful tool to raise awareness of the soil importance, both for agricultural and forest production and for environment protection. They also have proven to be a powerful tool for teaching and demonstration purposes. Furthermore, their beauty and visual impact catch people's attention either their age or knowledge.

Soil monoliths are a very reliable testimony of the composition and structure of soil profiles in the moment when have been extracted. In the case that a soil monolith can be extracted on the same place 50 years later it will be possible to determine differences between both monoliths and some climate change consequences will be observed. In this sense, CISP will be a reference for the Pyrenees and Pre- Pyrenees soils in order to determine their condition in the moment of their extraction.

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M.13.P

PHYSICAL AND CHEMICAL CHARACTERISTICS OF GRASSLANDS IN ALTIPLANO POTOSINO OESTE, MEXICO

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Grasslands in northern Mexico are degraded by overgrazing, limiting their productivity and nutrient supply to livestock. The objective was to evaluate some physical and chemical characteristics of the grasslands of the Altiplano Potosino, Oeste. We take soil samples from six private production units (PU) and four of commonly used (Ejidos). We determined pH, electrical conductivity (EC), organic matter (OM), cation exchange capacity (CEC), Ca, P, Na, K and Mg. The pH in the private PU (7.7) was higher than in the Ejidos (7.1) and were considered as non-limiting to grass growth, EC in private UP (1.2 dS m^{-1}) and ejidos (1.2 dS m^{-1}) was similar; the OM in the soils of the Ejidos was higher (2.2 %) than in the private UP (1.4%), both limiting the growth of grass; However, the CEC classified as mean and similar in the private PU (23.1 Cmol + kg^{-1}) and Ejidos (24.4 Cmol + kg^{-1}), so that the nutrient loss due to leaching is low. The contents of Ca and K were similar in the private UP (28.5 and 2.1 ppm) and Ejidos (29.1, 1.2 ppm) respectively, being considered high and not limiting for the optimal grass growth. Soils of the private PU contain more Mg and Na (3.7 and 2.3 ppm) than the Ejidos (1.6 and 0.4 ppm); on the other hand, P was higher in Ejidos (3.8 ppm) than in Private (2.4 ppm), considered low and limiting for optimal grass development. Organic matter, salts and P in grasslands soils are one of the major constraints, mainly in Ejidos for MO, for an optimal growth of grass plants.

INTRODUCTION

In Mexico, arid zones represent about 60% of the territory and one of the primordial productive activities that has been developed is extensive livestock farming, principally of cattle, sheep, and goats. The tenancy of the land is divided into Ejidos (communal property) and smallholdings (private) with differential land management of pastureland between both types of tenancy. It is known that the ecosystems of arid zones are some of the most endangered in the world, due to factors such as overgrazing, the opening of new areas to irrigation agriculture and climate change, among others. Overgrazing has led to lower vegetation cover, less availability of high value foraging pastureland, and a greater presence of species of shrubs (Wilson and Macleod, 1991), which in turn have been favored by a higher concentration of atmospheric CO2 (Gordon and Prins, 2008). In soil, overgrazing causes a lower infiltration of water, greater water and wind erosion, a smaller store of water and nutrients for plants, and a decrease in animal production (Holechek *et al*., 1989; Whitehead, 2000). The objective was to evaluate some physical and chemical characteristics of communally used pastureland (Ejidos) and smallholdings (private) in Altiplano Potosino Oeste.

METHODS AND MATERIALS

The study was carried out in central Mexico; it comprised the Economic Microeregion Altiplano Potosino Oeste, bounded by the extreme coordinates 101.22° – 102.3° west longitude and 22.47° - 23.8° north latitude. The area of the study corresponds to the municipalities of Salinas, Villa de Ramos and Santo Domingo (Figure 1), whose principal economic activity is extensive livestock farming (cow-calf production system) and seasonal agriculture. The climate is dry desert or desert steppe (BS_0kw, 82%; BS_1kw, 18%) with rains in the summer; the precipitation varies from 157 to 369 mm and it has an average annual temperature of 15.8 to 17.7 °C (Servicio Meteorológico Nacional, 2015).

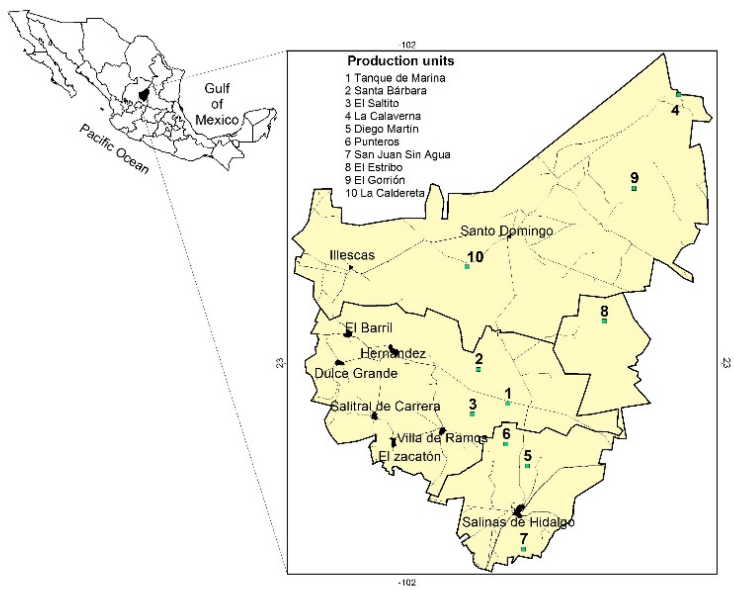

Figure 1. Study area, Altiplano Potosino, Oeste, Mexico.

The principal types of soil (%) are: Calcisols (24.11), Phaeozems (21.5), Kastanozem (18.1), Chernozem (15.6), Leptosol (7.8), Luvisols (4.8) (INEGI, 2007). The dominant vegetation is microclimate desert scrub (54.3 %), desert rosetophilous (3.5%), crasicaule (3.4%), induced grassland (1.98%), halophyte vegetation (0.7%), natural grassland (0.2%) and gypsophile grassland (0.1%).

Soil samples of approximately one kg were taken from six UP and from four Ejidos from the first 20 cm of depth. In the laboratory, the samples were air dried, sieved with mesh, and stored until their analysis. The analysis of the soil samples was undertaken according to the Official Mexican Standard (SEMARNAT, 2002); they were: 1) organic material (MO), through the Walkley and Black method; 2) electric conductivity (EC), obtained with a conductivity meter (Hanna, HI98311), in a soil-water ratio of 1:5; 3) pH was measured with a potentiometer (Corning, 340); 4) cation exchange capacity (CEC) and interchangeable bases –calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K)-, with 1N ammonium acetate at pH 7.0 as a saturating solution; 5) phosphorous (P) through the Olsen procedure *et al.* (1954). The contents of Ca, Mg, Cu, Fe, Zn, Mn and Co were determined through spectrophotometry of atomic absorption (Aurora AI-1200) and P was determined by colorimetry in an ultraviolet light spectrophotometer (Thermo Scientific Genesys 105 vis). The statistical analysis was undertaken using the GLM procedure of the SAS statistical packet (2012); in the model, the effect of UP was considered and for the comparison of averages, Tukey's test was used (Steel and Torrie, 1997).

RESULTS AND DISCUSSION

The results of the physical and chemical characteristics are showing in Table 1. The pH of the UP Ejidos soils (7.1) and PP (7.7), varying between neutral and fairly alkaline, does not limit the absorption of nutrients nor the development of forage. The content of MO was less in the PP (1.4%) than in the Ejidos (2.4), varying from low to medium class (SEMARNAT, 2002), and can be considered inappropriate for the adequate or optimum development of vegetation; in addition, it can limit the absorption of Cu and nitrogen by the forages. The EC indicates slightly saline soils and this can limit the development of vegetation. In general, the soils studied are of middle to high class depending on the CEC (SEMARNAT, 2002), and so the loss of nutrients by leachate is considered low.

Table 1. Physical and chemical properties of soils of private production units and Ejidos in the Altiplano Potosino, Mexico.

	Private production units	Ejido	*P*
pH	7.7 (MA)	7.1 (N)	0.1016
OM, %	1.4 (B)	2.2 (M)	0.0137
EC, dS m^{-1}	1.2 (Mls)	1.2 (Mls)	0.9756
CEC, Cmol + kg^{-1}	23.1 (M)	24.4 (M)	0.7251
Ca, ppm	28.5 (A)	29.1 (A)	0.9438
P, ppm	2.4 (B)	3.8 (B)	0.0242
Na, ppm	2.3 (MA)	0.4 (B)	0.032
K, ppm	2.1 (A)	1.2 (A)	0.1117
Mg, ppm	3.7 (A)	1.6 (M)	0.0375

pH: MA= Medium alkaline, N= Neutral.
OM: B= low, M= Middle, MB= Very low.
P: A= High, M= Middle, B=Low.
CEC, Ca, Mg, K, Na: MA= Very High, A= High, M= Middle, B=Low.
EC: N= Not saline, Mls= Very slightly saline.

The contents of Ca and K were similar in the private UP (23.1 and 1.8 ppm) and Ejidos (24.4, 1.2 ppm) respectively, being considered high and not limiting for optimal grass growth. The Ejidos soil had more P (3.8 ppm) than those of the PP (2.4 ppm), probably due to the fact that in the PP, more intensive cattle extraction is done than in the Ejidos. The values of P are considered low and limited by an optimal growth of plants (Castellanos *et al*., 2000) and the quality of forage for cattle (Benavides *et al*., 2001;

Underwood and Suttle, 2003). The PP soils contain more Mg and Na (3.7 y 2.3 ppm) than the Ejidos (1.6 y 0.41 ppm) respectively. The values of Na in the PP are considered high and can limit the growth of plants, however in the Ejidos, they are considered low. Meanwhile, Mg in the PP is considered high while in the Ejidos it is considered low.

CONCLUSION

Organic material, salts, and phosphorous in the pasture soils are the greatest limitations, principally in PP, for optimum plant growth and, thus, for sustainable long-term livestock production. Additionally, they can be considered indications of degradation of pastureland soils.

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M.14.P

NITROGEN FERTILIZATION IN TOPDRESSING IN SOYBEAN INTERCROPPED WITH FORAGES

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The crop livestock integration (CLI) is a system that integrates agricultural and livestock activities in the same area rotating the same or in a intercropping. Thus, one of the agricultural alternatives to integrate with livestock is the soybean crop, because it has great demand and economic value. According to National Supply Company, Brazil harvested approximately 96 million tons of soybeans in 2015, making Brazil the second largest producer of grain (CONAB, 2015).

The main objective of the forage in a intercropping is to provide straw for soil cover, however it is possible to take into account the forage quantity and quality as food for animals, since the system aims to benefit both agronomic activities. In this sense, one of the crops that stood out as a mode of crop livestock integration is the intercropping with *Brachiaria brizantha* that allows plant cover for a long time on the soil due to its high C/N ratio (PORTES et al., 2000).

The *Brachiaria brizantha* cv. Marandu is a forage that presents high productivity, which has a wide climatic adaptation, high forage value and production of green mass and tolerance to shading, which provides good shoot dry matter production and consequently good fattening of the animals. The *Panicum maximum* cv. Tanzania stands out for its high productive potential compared to buffalo grass (ABREU, 1999). Among genus *Panicum* cultivars, *Panicum maximum* cv. Tanzania is widely indicated for use in grazing because of its high productive potential and proven nutritional quality (PATÊS et al., 2008).

In this context, the experiment aims to evaluate soybean performance cultivated in different production systems (monocropping; intercropping with *Panicum maximum* cv. Tanzania; intercropping with *Brachiaria brizantha* cv. Marandú), with nitrogen fertilization in topdressing (0; 50; 100; 150 kg ha^{-1}).

MATERIALS AND METHODS

The experiment was carried out in the experimental area of the Mato Grosso do Sul State University – Aquidauana University Unit, located in Aquidauana county (MS) - Brazil, comprising the following geographic coordinates 20º27'S and 55º40'W with 170m altitude. The region climate, according to Köppen-Geiger classification is Aw (Tropical of Savana) with 1200 mm annual precipitation average and maximum and minimum temperatures of 33 and 19ºC, respectively.

The experimental design was a randomized complete block design in a split plot scheme, with the treatments constitued by three production systems (soybean monocropping, soybean intercropping with *Brachiaria brizantha* cv. Marandu, soybean intercropping with *Panicum maximum* cv. Tanzania) and four N doses (0; 50; 100; 150 kg ha^{-1}) top-dressed, with four replications.

Soybean sowing occurred in November 2014, using SYN 1059 RR cultivar, 0.45 m row spacing and 14 plants m^{-1} plant density. For the cover fertilization, 0, 50, 100 and 150 kg ha^{-1} N doses (ammonium sulphate: 21% N), were applied at same time, in V5 growth stage. Already forages sowing was done manually, when the soybean crop was in R6/R7 growth stage.

RESULTS AND DISCUSSION

The stand was not influenced by the production systems, as well as by the nitrogen topdressing fertlization (Table 1), and this is related to the forages sowing time, which occurred around the R6/R7 growth stage. At this growth stage, soybean is already with 100% of grains formed and, therefore, all forages influence was negligible. In general, the plant stand approached near the ideal.

The plant height was positively influenced by nitrogen topdressed application (Table 1). The data fit the equation $Y = 61.79 + 0.1614\ x - 0.00088\ x^2$, demonstrating that the application of 91.8 kg ha^{-1} N provided the maximum value for plant height (69.2 cm). Already Silva et al. (2011) and Vieira Neto et al. (2008), working with nitrogen topdressed fertilization in soybean crop, observed that the application of 24 and 200 kg ha^{-1} of N provided plant height between 73.6 and 89.52 cm, respectively.

Table 1. Mean values for soybean final stand (SFS), plant height (PA), first pod insertion height (FPIH) and number of pod per plant (NPP) in function of different production systems and topdressed N doses fertilization. Aquidauana (MS), 2015.

Treatments		SFS (plant ha^{-1})	PA (cm)	FPIH (cm)	NPP
Production systems	Soybean	275926 a	65,2 a	15,9 a	46 a
	Soybean + *B. brizantha*	305093 a	70,5 a	16,2 a	42 a
	Soybean + *P. maximum*	271528 a	62,9 a	15,2 a	37 a
Nitrogen doses kg ha^{-1}	0	273457	* 61,9	14,9	40
	50	287654	67,4	16,0	42
	100	286728	69,4	16,8	40
	150	288889	66,1	15,4	46

* $y = 61{,}79 + 0{,}1614\ x - 0{,}00088\ x^2$ ($R^2 = 0{,}9953$). Means followed by common letters in the columns do not differ by Tukey test (0.05).

The first pod insertion height was not influenced by the production systems, as well as for nitrogen topdressed application (Table 1). For all treatments, the soybean first pod height average is within the recommended range (BONETTI, 1983), which, according to the author, minimizes possible losses in mechanized harvest.

For the number of pods per plant there was no influence of the treatments (Table 1). In this sense, Peixoto (2000) evaluating soybean behavior as a function of sowing times and seed density, showed that the stand increase causes a decrease in number of pods per plant, which did not occur in the present experiment.

No influence of production systems or nitrogen topdressed fertilization was observed for number of grains per pod (Table 2). According to Mundstock and Thomas (2005), the number of grains per pod is the variable that presents the lowest variation among the production components; The authors also point out that this variable is a characteristic that generally remains close to two grains per pod, which corroborates the data obtained in this experiment that maintained an average of between 2.2 and 2.5 grains per plant.

The production systems and the nitrogen topdresses fertilization did not provide significant effects for 100 grains mass, corroborating data presented by Aratani et al. (2008), that working with different times of topdressing N, did not observe significant effects for 1000 grains mass. The values presented in the present experiment are below those obtained by Aratani et al. (2008) and Jendiroba and Câmara (1994), with the varieties Conquista and IAC 8, respectively. However, it should be noted that the region where the present experiment was conducted showed a high average night temperature, which may have contributed negatively to grain filling.

The production systems (soybean intercropping with *Panicum maximum* cv. Tanzania and soybean intercropping with *Brachiaria brizantha* cv. Marandú) did not influence grain yield (Table 2), evidencing that the forage used in this experiment did not affect crop productivity, probably because were seeded when soybean was in R6/R7 growth stage, which favored soybean because it was already close to the complete grain filling, corroborating Silva et al. (2004), which worked with the suppression of *Brachiaria brizantha* forage in a intercropping with soybeans sown simultaneously (different doses of fluazifop-p-butyl) at different times (21 and 28 days after emergence) observed higher soybean yield when the herbicide was applied 28 days after emergence, because at this time there was grass control and it did not infest the area, evidencing that grass control improved soybean yield. However, the herbicide use can be avoided by sowing the grass when the soybean is already in the R6/R7 growth stage.

Tabela 2. Mean values for number of grains per pod (NGPO), number of grains per plant (NGPA), mass of 100 grains (M100) and grain yield (GY) in function of different production systems and topdressed N doses fertilization. Aquidauana (MS), 2015.

Treatments		NGPO	NGPA	M100 (g)	GY (kg ha^{-1})
Production systems	Soybean	2,13 a	98 a	12,92 a	2418 a
	Soybean + *B. brizantha*	2,08 a	89 a	12,81 a	2464 a
	Soybean + *P. maximum*	2,17 a	80 a	14,05 a	2448 a
Nitrogen doses kg ha^{-1}	0	2,08	83	13,90	*2081
	50	2,05	86	12,79	2546
	100	2,17	87	13,33	2702
	150	2,21	101	13,01	2446

* $y = 2075 + 13{,}33\,x – 0{,}07217\,x^2$ ($R^2 = 0{,}9974$). Means followed by common letters in the columns do not differ by Tukey test (0.05).

Nitrogen had a positive influence on soybean grain yield (Table 2). The data corresponded to the equation $y = 2075.6 + 1230.33\,x – 0.07217\,x^2$, showing that the application of 92 kg ha^{-1} N provided the highest result (2690 kg ha^{-1}), differing from the presented results by Mendes et al. (2008), which evaluated the N application with different sources (urea, ammonium sulfate, ammonium nitrate and control) with up to 200 kg ha^{-1} N at different times (R1 and R5 growth stages). Nitrogen had no advantage over the traditional inoculation regarding grain yield. Similarly, in an experiment involving the N doses application (0, 30, 60, 90 and 120 kg ha^{-1}) at different times, Bahry et al. (2013) observed that N application to soybeans, regardless of reproductive stage, did not provide positive increases in grain yield.

CONCLUSIONS

The intercropping between soybean and *Panicum maximum* cv. Tanzania or *Brachiaria brizantha* cv. Marandú is promising since it does not negatively affect any soybean crop vegetative or reproductive components, with the benefit of shoot dry matter production by the grasses for soil cover.

The application of 91.5 kg ha^{-1} of N in V5 growth stage provided positive results for plant height (69 cm) and maximum soybean grain yield (2691 kg ha^{-1}).

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M.15.P

WATERLOGGING OF THE SOIL: EFFECT ON THE PARTITION OF ASSIMILATES AND YIELD GENOTYPES BEAN SEEDS

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INTRODUCTION

Beans are among of the most important sources of protein, complex carbohydrates, fibre, iron, minerals and vitamins.

The occurrence of water stress, either by excessive reduction or excessive elevation of soil moisture, is a factor that compromises sharply the development of bean cultures. Water stress can result in translocation rate reduction from the leaves to the root system in organic compounds, such as carbohydrates, which results in decreased activity and growth. Moreover, it can affect the efficiency of solar energy conversion and partition of dry matter (PEDÓ et al., 2015).

Growth and yield of a species result from the interaction that occurs between the culture used and its environment. For a better understanding of the performance of plants in certain environment conditions, evaluating variables related to plant growth can be an important tool.

The purpose of this study was to evaluate the effect of soil waterlogging on dry matter partition and yield attributes of bean genotypes.

MATERIAL AND METHODS

The research was carried out in a chapel greenhouse, at Federal University of Pelotas (UFPel), Brazil. The experimental design used was a completely randomized design in factorial scheme 2 x 2 (two bean genotypes and two environmental conditions, waterlogged and non-waterlogged).

Bean seeds of the IPR Jabiru and Carioca genotypes were used, which belong to black bean and pinto bean commercial groups, respectively. The seeding was done manually, in black polyethylene vessels with 10 liters volume capacity. The soil used was previously limed.

Soil field capacity was determined with table tension methodology, the volume of water needed to keep the flooding for a period of 12 hours was defined at the vegetative stage V4. A 20 mm blade of water was held on the soil surface.

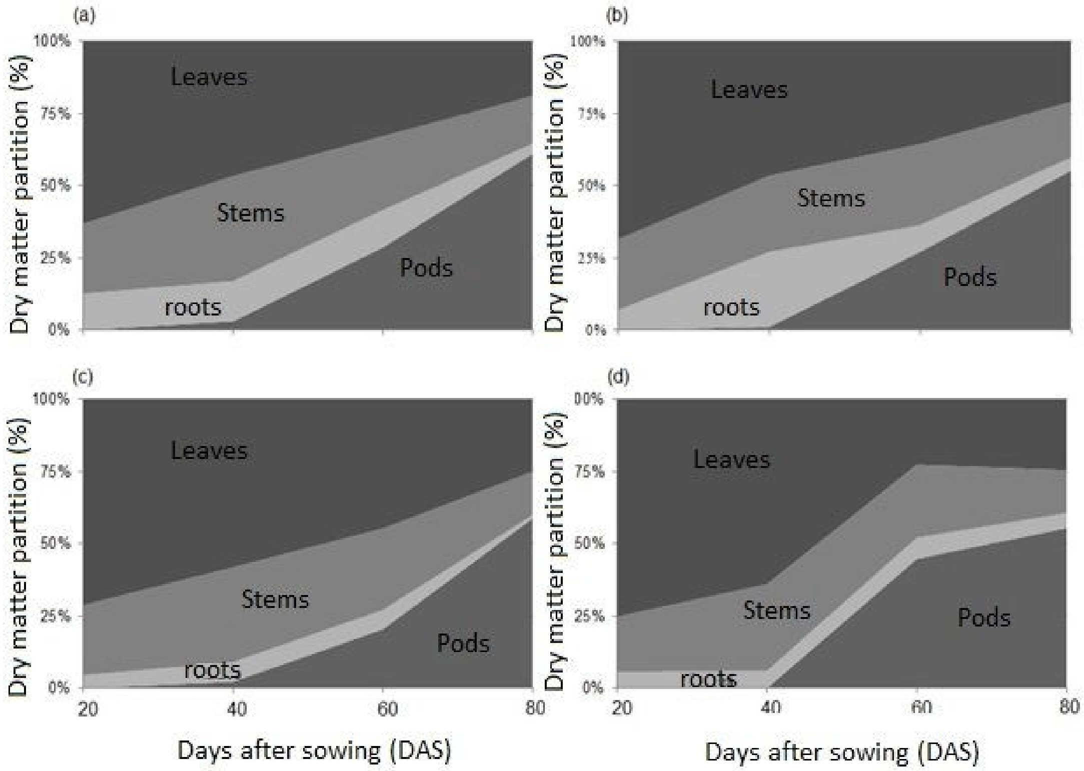
For determination of dry matter partition data, plants were collected at regular intervals of 20 days during their full development cycle. The first collection was held 20 days after sowing (DAS). In each collection, plants were cut low to the ground, separated into leaves, stems, roots and beans and wrapped in brown paper envelopes, separately. To obtain dry matter, the material was transferred to forced ventilation oven, at a 70 ± 2° C temperature for 72 hours. The growth variables composing the plants assimilated partition analysis (dry weight of roots, stems, leaves and pods) were converted to percentage and expressed through orthogonal polynomials.

The number of pods (Nvg) and seeds (Ns) was obtained by direct count and expressed in number of pods and seeds plant^{-1}. The variables data were subjected to the F test analysis of variance ($p \leq 0.05$). If significance was found, the effects of genotype and environment condition factors were compared by the Duncan test (p ≤ 0.05).

RESULTS AND DISCUSSION

The beans of genotype IPR Jabiru subjected to waterlogging showed differences in the distribution of assimilated throughout their development, compared to those kept at field capacity (Figure 1a, 1b). In plants of this genotype the maximum dry matter accumulation in leaves occurred on 20 DAS both for beans subjected to flooding (Figure 1a), and for beans kept at field capacity (Figure 1b). The plants of genotype IPR Jabiru subject to flooding showed significant increment in the accumulation of dry matter of the stem, between 20 and 40 DAS (Figure 1a). In that same period of time, it is observed that beans of this genotype, when not subjected to flooding, showed significant increase in root dry matter (Figure 1b). Until 37 and 39 DAS the leaves were the structures where 50% of the total dry matter of IPR Jabiru was allocated, on waterlogging and field capacity conditions, respectively.

For the IPR Jabiru genotype it was observed that the distribution of assimilated took the following order: leaves, stems, roots and pods, up to 60 DAS, be it subjected to waterlogging or maintained in the capacity field (Figure 1a and 1b). For this same genotype, beans subjected to flooding showed change in preferred metabolic drain about five days before those kept at field capacity. For plants subjected to flooding, pods are preferred as the metabolic drain 65 DAS, while in those maintained at field capacity the pods become the main drain only 70 DAS.



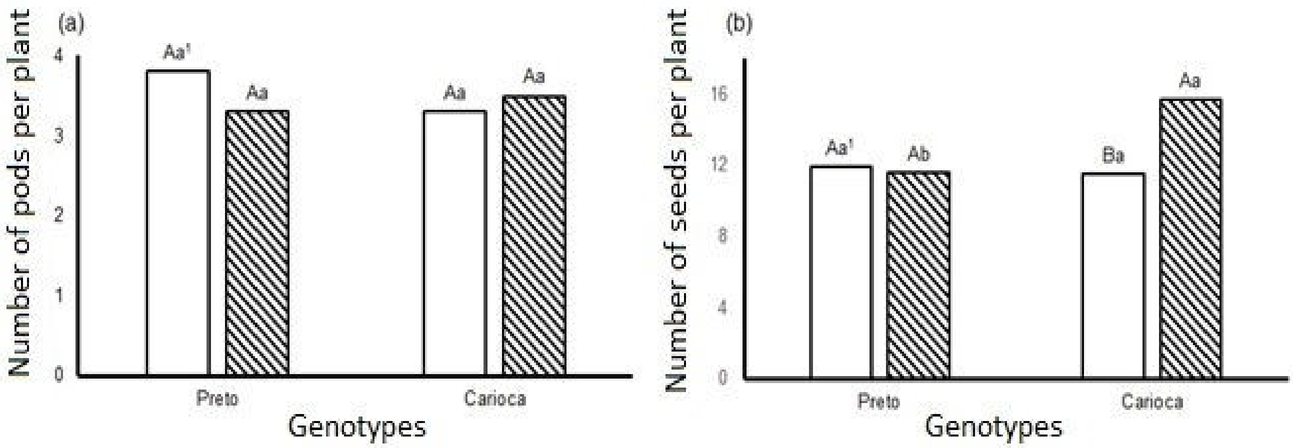
Picture 1. Partition of dry matter of plants of two bean genotypes, either not subjected to waterlogging or kept under 12 hours of waterlogging, as follows: Jabiru IPR in waterlogged soil (a); Jabiru IPR at field capacity (b); Carioca in waterlogged soil (c) and Carioca at field capacity (d).

Carioca genotype beans subjected to soil waterlogging or kept at field capacity allocated greater dry weight on leaves 20 DAS (Figure 1 c and 1 d). Beans of this genotype, kept in the field or subjected to waterlogging, showed greater distribution of assimilated among the leaves, stems, roots and pods, until 58 and 70 DAS, respectively. After these dates, the pods were the preferred plant metabolic drain on both waterlogging and field capacity conditions. From 40 DAS on, an expressive increase was observed in the dry matter of the pods of Carioca genotype beans maintained at field capacity (Figure 1 d). However, when subjected to waterlogging, significant increment of dry matter occurred only 60 DAS (Figure 1 c).

Differences in dry matter partition between plants maintained at soil capacity and waterlogging conditions may occur due to limitations in the absorption of nutrients and water by the roots, as well as due to the reduction of beans' photosynthetic rate, caused by excessive water in the soil. In waterlogged environment, plants exhibit marked variability of morphological, anatomical and physiological characteristics, as they try to acclimatize against the negative effects of stress (SENA et al., 2007).

The genotypes present distinct behavior when subjected to soil waterlogging. The most sensitive is Jabiru, IPR. The negative effects caused by soil waterlogging in beans' morphological attributes resulted in changes in dry matter partition, culminating with a reduction in the number of seeds per plant, which is related to performance.

Regarding the number of pods per plant, there was no significant difference among genotypes and between different environmental conditions (Figure 2a). Regarding the number of seeds per plant, there was no difference between black beans maintained at field capacity and those subjected to soil waterlogging (Figure 2b). On the other hand, as for pinto beans, the plants not subjected to soil waterlogging presented a 25% higher number of seeds per plant when compared to those exposed to this condition.



Picture 2. (a) Number of pods per plant of two bean genotypes: Preto (black bean) and Carioca (pinto bean). White bar represents waterlogged conditions, while the striped bar represents not waterlogged conditions.

(b) Number of seeds per plant of two bean genotypes: Preto (black bean) and Carioca (pinto bean). White bar represents waterlogged conditions, while the striped bar represents not waterlogged conditions.

There was no significant difference in the number of seeds per plant between the genotypes evaluated, when in waterlogged conditions. The reduction in the number of seeds per plant in waterlogged conditions observed in black beans when compared with pinto beans may be due to its reduction in the number of leaves.

A similar production of black bean seeds in waterlogged soil conditions and field capacity conditions indicates a probable greater tolerance to soil waterlogging. This greater tolerance may possibly be related to the effect of temperature on the development and the formation of new leaves.

CONCLUSIONS

The beans of IPR Tuiuiú genotype submitted to waterlogging achieved lower production of total dry matter, comparatively to those not exposed to this condition. For the Carioca genotype there was similarity in the production of total dry matter between plants subjected to waterlogging and those maintained at field capacity. Therefore, waterlogging affects the partition of assimilates and performance variables of beans, depending on genotype chosen.

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Keynote Presentations

SOIL AND WATER CONSERVATION TO MITIGATE CLIMATE CHANGE AND ADVANCE FOOD AND NUTRITIONAL SECURITY

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EXTENDED ABSTRACT

Soil is the largest reservoir of the terrestrial carbon (C) stock. The soil organic C (SOC) stock in world soils is 1500 Pg (petagram=Gt=10^{15}g=I million metric ton) to 1-m depth, 2060 Pg to 2-m depth, and 3000 Pg to 3-m depth. In addition, SOC stock in soils under permafrost(Cryosols or frozen soils) may be as much as 1670 Pg. In total and to 3-m depth, soil C stocks (both organic and inorganic) in world soils(6000 Pg) is 7.5 times that of the atmospheric stock (800 Pg) and 9.7times that of the vegetation stock (620 Pg). Therefore, even slight changes in soil C stock globally can lead to large changes in atmospheric concentration of CO_2. The soil C stock can be a source or sink of atmospheric CO_2 depending on landuse, management, soil and climate factors. Soil erosion, by water and wind, leads to preferential removal of the SOC fraction because it has low bulk density (0.5-0.8g/cm^3) and is concentrated in close proximity to the soil surface. Pathways over the landscape and fate of SOC transported by erosional processes are not understood. However, accelerated erosion depletes SOC stock on-site, and the SOC transported by water and wind is prone to mineralization en-route to depositional site(s) because of disruption of aggregates and change in moisture and temperature regimes. Erosion-induced global emission of CO_2 may be as much as 1.1 PgC/yr. The erosion-induced emissions may include CO_2 and N_2O under aerobic conditions and CO_2, CH_4, and N_2O under anaerobic environments. The global warming potential, relative to that of CO_2, is 21 for CH_4, and 310 for that of N_2O. Thus, the net effect of erosion on radiative forcing must be assessed in terms of CO_2 equivalent. Over and above the on-site effects of erosion on soil health and agronomic productivity, important among the off-site effects are gaseous emissions and eutrophication of natural waters (e.g., algal bloom). Thus, adoption of conservation-effective measures are essential to numerous agronomic and ecologic benefits including the avoidance of erosion-induced gaseous emissions. Further, conservation of soil and water in agroecosystems, which enhance biomass productivity and increase the input of C to the soil, is essential to re-carbonization of the terrestrial biosphere (soil and vegetation), adaptation and mitigation of climate change, provisioning of numerous ecosystem services, and restoration of degraded soils and desertified ecosystems. Conservation of soil and water resources is also essential to advancing the Sustainable Development Goals (SDGs) of the United Nations. Controlling soil erosion and restoring eroded/degraded soils are pertinent to SDGs #1 (end poverty and improve health and wellbeing), 2 (end hunger and achieve food security), 6 (provide clean water), 13 (mitigate climate change), and 15 (enhance and improve life on land). The SDG #2 of ending hunger is important because 795 million people are prone to hunger and 2 billion to malnutrition and protein deficiency. The SDG #13 is important because atmospheric concentration of greenhouse gases is increasing at the rate of 2.3 ppm/yr for CO_2 (400 ppm), 11 ppb for CH_4 (1845 ppb) and 1.0 ppb for N_2O (328 ppb). Improving soil health through erosion management and conservation is essential to achieving these SDGs. The Therefore, adoption of conservation-effective measure and restoration of eroded soils and landscape are essential to numerous ecosystem services of relevance to human wellbeing and nature conservancy. These strategies are also pertinent to national and international security.Depleting soil organic C pool, degrading soils, recurring drought, marginal use efficiency of fertilizers and other inputs, low crop yields, perpetual poverty and hunger,

high infant mortality due to hunger and malnutrition are as real threats to global peace and security as are ICBMs and nuclear weapon proliferation because the health of soil, plants, animals, people and ecosystems are one and indivisible. Therefore, soil stewardship and care must be embedded in every fruit and vegetable eaten, in each grain ground into the bread consumed, in every cup of water used, in every breath of air inhaled, and in every scenic landscape cherished.

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THREATS TO SOIL AND WATER CONSERVATION – GENERAL DEVELOPMENTS AND FUTURE SCENARIOS – A WORLDWIDE PERSPECTIVE

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EXTENDED ABSTRACT

1. INTRODUCTION

Soils are providing important goods and services to humankind and the environment, through the production of biomass, especially food, fodder, renewable energy and raw materials, but also through filtering, buffering and transformation between the atmosphere, the ground water and the plant cover, protecting the environment, especially the ground water and drinking water resources. – Moreover, soil is the largest biological habitat and gene reserve on earth.

Soil is also used as a physical basis for technical, industrial and socio-economic structures and their development, e.g. industry, housing, transport, sports, dumping of refuse and others. In order to build such structures, soil is providing materials, e.g. clay, sand and gravel for construction. – Finally soil is a geogenic and cultural heritage, forming an essential part of the landscape and protecting palaeontologicaland archaeological remnants.

2. GLOBAL LAND AND SOIL RESOURCES

For prioritizing our efforts in soil and water conservation, the inherent quality of land and soil is of paramount importance, becauseonly 12 % of the land surface is suitable for the production of food and fiber, 24% can be used as pasture, 31% produce forests and 33 % are unsuitable for any kind of sustainable use, due to the type of relief,insufficient temperature, no water availability or because of the lack of soils.

About 2 thirds of the most productive soils occur in the northern hemisphere and only about one third in the southern hemisphere , what explains to some extent the worldwide pressures on land and soil.

Based on this worldwide distribution of inherent land and soil quality and analyzing the worldwide use of these resources reveals 8 main global developments,which will be described as future scenarios in sustaining the world population and the environment, based on soil and water conservation.

3. GENERAL DEVELOPMENTS AND FUTURE SCENARIOS

3.1. Increase of world population and changes in its spatial distribution

Every year, about 80-85 million more people need more food, more space and more energy. Even more important is that every year, between 100 and 150 million people move from rural into urban areas, or are born there,abandoning subsistence farming and increasing pressure on soil through unsustainable land use.

3.2. Loss of fertile soils through urbanization, industrialization and further human impacts

The best agricultural soils are increasingly lost through urbanization, industrialization and further human impacts. Our ancestors were searching the best soils, before they started to settle, and these initial settlements have grown in the meantime into urban and industrial agglomerations of great extension. Therefore, until today urban sprawl occurs still on the most productive soils.- At a world-wide level, we are losing every day about 300-400 km² of fertile land.

Further soil losses occur through erosion, compaction, contamination, salinization and other adverse human impacts.

3.3. Changing human life style and increasing demands for food

With increasing social and economic wealth, the life style of people is changing and the demand for food is increasing.

Besides the worldwide demand for more individual living space, in most of the industrialized countries people are wasting up to one third of their purchased food, most of it unpacked, and eat excessively, which leads to obesity. – For combating obesity, billions of US $ are annually spent, which could nurture to a great extent hungry people; at the moment, nearly 1 billion.at the worldwide level.

Moreover, the increase in the consumption of animal protein (meat) increases the demand for grain. and other vegetable food.,- For the production of 1 kg of chicken meat about 2-3 kg of grain are needed, for 1 kg of pork 4-5 kg, and for 1 kg of beef 7-10 kg of grain respectively.

3.4. Increasing demands for bioenergy

Increasing demands for bioenergy (biogas, biofuel or solid material,e.g.straw or wood)can be observed on a world-wide level.

The production of ethanol from agricultural plants, e.g. from grain or sugar cane, or biodiesel from oil plants (e.g. rape, soy beans or oil palms) is still increasing. In 2011 worldwide 13% of all grain and 35% of all sugar cane were used for ethanol production, 16% of all vegetable oil for biodiesel.

In many world regions the competition for water in the production of food and/or bioenergy is increasing. In those areas where food production is already limited by a lack of water, any increase of bioenergy production is no longer possible.

3.5. Changes in world economy

Changes in world economy and emerging new financial instruments in the marketing of agricultural commodities , allowing for speculative performances, e.g. derivatives , are volatizing the prizes ,and cause land and soil degradation, e.g. through pressures on agricultural production, especially in the case of an increase of production costs (energy, fertilizers, pesticides, agricultural machinery etc.) and the decline of market prizes.

The land take in foreign countries , called "land grabbing" and the subsequent change in agricultural and forest land use is additionally causing in most of the cases new land and soil degradation , especially in the tropics and subtropics.

3.6. Climate change

One of the most important global impacts on land and soil degradation derives from climate change, which means the change in temperature and precipitation intensity and variability, thus influencing soil and land management.

Climate change is not only causing global warming, partly exceeding temperature thresholds for agricultural production, increased crop water requirements, and increased incidences of pests and diseases, but also land and soil erosion through changes in the precipitation pattern (quantity and intensity of rainfall) and surface runoff.

The increased occurrence of extreme weather incidents and increased climate variability is also increasing soil sedimentation and other adverse hydrological impacts. - However, those climatic changes are not uniformly distributed at a world-wide level but vary in different regions of the world. They are predominantly increasingsoil and water conservation problems in tropical and subtropical countries.

3.7. Decrease of fresh water supply

The worldwide existing lack of fresh water reserves is also a result of land and soil degradation, because of increased surface water runoff and insufficient rainwater infiltration, due to soil erosion and compaction.

In view of the fact that agricultural production consumes worldwide about 70 % of all fresh water resources, this means that the global food security is endangered.

3.8. Globally increasing spread of invasive alien species

Through the invasion of new plant and animal species, also new methods in soil and water conservation have to be developed, in order to protect agricultural and forest production and to maintain soil productivity and soil health.

4. CONCLUSIONS AND OUTLOOK

-. Soil and water conservation at a global level is a complex task, because of its different ecological, technical, social, economic and cultural dimensions;

- The presented future scenarios were developed on the basis of visible developments. However, uncertainties remain because of changes in the social and cultural environment, changes in world economy and climate change;

- Scientists cannot take decisions but are able to support politics and decision making by creating scenarios based on indicators developed by interdisciplinary and multidisciplinary scientific research.

NEW PERSPECTIVES FOR SOIL AND WATER CONSERVATION IN TODAY GLOBAL TRANSITION SCENARIOS

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INTRODUCTION

Soil conservation and the sustainable land management have a decisive contribution to most of the global environmental and socio-economic problems we are facing today. Specifically there are three main global issues tightly linked to soil conservation. These are: a) the need to increasing food production through a more ecological agriculture; b) the application of ecological regulation functions to biodiversity and landscape maintenance and c) reinforcing the soil regulatory capacity for climate change mitigation.The demands for sustaining biodiversity, carbon cycle regulation, combating desertification, food production, water resource regulation, landscape maintenance and conflicts-security aspects require new soil protection paradigms that should be developed under soil multifunctional and multiuse approaches. The concept of soil conservation in its wider application has undergone important changes through history. For centuries, agrarian production and the provision of food for humankind remained the main and crucial vision of the interaction of societies with soil. Development of soil conservation is strongly connected to schemes of socio-economic and technological changes, demographic growth and territorial expansion but always linked to crop production. Nevertheless and slowly, new visions and perceptions of soil conservation were expanding and evolving. A quick reference to some milestones/periods of this conceptual evolution is the following: a) Early intuitive, not organized knowledge b) Perception of mistakes. Trial and error. c) Practicability. Knowledge based on experience. d) Adaptation to suitable land. e) Food production. Famines. f) Age of enlightenment: pioneers of holistic vision. g) Mechanization and Industrial agriculture: Yields. h) Green revolution, leveling: Economic development. i) Environmental concerns. j) Rio Summit 92, UN Conventions. k) EU Soil Protection Strategy. l) Soil as a multifunctional medium. m) Bio-Eco engineering for soil conservation and n) Agroecosystem Serviceswhich reinforced the biological connection of soil and soil conservation to socio-economic aspects and framed soil as crucial link in the many interconnected processes and domains comprising the biosphere.

Security aspects related to mismanagement of the land and failure of soil conservation approaches are of particular concern. Land degradation and desertification, water scarcity, impacts of drought, crops failure and the effects of extreme weather events exacerbated by the tendency of climate change are factors which have direct negative effects on the functioning of the terrestrial ecosystem and on the people who live in those ecosystem. The decrease of available resources such as soil, water and food and the competition to get access to scarce or degraded land is closely linked to the security of populations. There is a general consensus that healthy soils are pivotal for food security. Food production is one of the main ecosystem services provided by and thus dependent on well-conserved and well-functioning soils. There are also intrinsic connections between the four pillars of food security – food availability, access, utilization, and stability – with how soils are managed, accessed and secured, in particular by food insecure and vulnerable populations. On the other hand, socio-political and economic processes that precipitate inequalities and heighten vulnerabilities among poor populations often increase pressure on soils due to unsustainable forms of land use, poor agricultural practices and lack or improper soil conservation approaches. The disruption of the soil capacity to provide good and benefits implies societal, economic and environmental security dimensions. The land degradation/desertification risk is an environmental problem with implications on security issues of worldwide scope that affects the five continents. The scarcity or degradation of soil-land resources and the collapse of social structures can increase subsistence crises, conflicts and violence menacing basic security dimensions. A crucial approach to secure populations under the above menaces is the development and implementation of a true sustainable development which ties together concern for the carrying capacity of natural systems to provide food security and the social challenges facing human development. The general objective of the paper is to attempt to higthligth the central role of land use management and soil conservation in the overall objectives of sustainable development and poverty reduction. It includes abetter understanding of the connections between land degradation and human wellbeing, specificallyon the implications of soil/land degradation on socio-economic consequences linked to social instability, poverty, migration and the development of conflict and war. Also the identification of needed institutional and societal changes, and the proposal of best strategies and actions to combat severe socio-economic consequences of land degradation and prevent future conflicts. Modern society demands new ideas, new information schemes and new conceptual development on soil conservation to support the enabling functioning of the global and local terrestrial ecosystem

DISCUSSION

Today's soil conservation frameworks should expand to include new perspectives and interlinkages in the context of the integrated functioning of terrestrial ecosystems meanwhile maintaining basic aspects from previous schemes including soil fertility and improving soil-water interactions. The emerging field ofAgroecosystem Services (UNEP,2005)applied to soil conservation offers important options for soil restoration and stabilization, for keeping land productivity and for preserving the quality of the landscape in terms of biodiversity and ecological functioning. The delivery of agro-ecosystems services (Supporting, Provisioning, Regulating and Cultural) should be linked to soil and water conservation schemes. Ecosystems services are connected both to the soil structure and the structure of the landscape. Agricultural landscapes should be managed through soil conservation approaches that enhance resilience and fertility of the landembedded in the context of the natural landscape.The formidable challenge of increasing agricultural production, keeping soil environmental requirements and restoring soil ecological functions and services demands better land management and conservation practices to enhancing the biological component of soil. Soil conservation should includeselected factors affecting soil functions and the provision of ecosystem services, including best management practices, improvement of soil properties, adequate selection of crops/plant materials, enhancingsoil biological activity and promoting suitable adaptation to the climatic potential and limitation of the land. This is a much needed general strategy because its lowers the risk of land degradation, improves food production; improves water efficiency; protect nature ecosystems and biodiversity; is crucial for restoration of degraded land; to combat desertification and to reduce the impact of climate change.In this context increasing agricultural efficiency is one of the most important keys to minimizing the destruction of terrestrial agro-ecosystem and natural habitats. However there is considerable debate concerning the margin for increasing agricultural efficiency and some recent socioeconomic scenarios require no net increase in land under cultivation at the global scale over the 21st century. Attaining this goal would require limited population growth, substantial increases in agricultural productivity and efficient use of primary production (e.g., reduction of post-harvest losses, limited meat consumption). Negative impacts of agricultural intensification on biodiversity can be minimized by appropriate agricultural practices (Leadley et al., 2010). The current situation of scenarios in transition is enhancing the continuous development and incorporation of new conceptual vision and perspectives in the field of soil conservation. As an example Dumansky (2015) discusses some new diving forces, new international programs and new methodologies including institutional, legislative and policy aspects which demonstrate the adaptive potential of soil conservation strategies. Also recently we have increasing soil initiatives that were just unthinkable few years ago. Some of them are: Global Soil Map (2009), Global Soil Forum (2011), Global Soil Partnership (2011), Global Soil Biodiversity Initiative (2011), Global Soil Week (2012), Soil references in Rio+20(2012), World Soil Charter (2014), International Year of Soils 82015), International Technical Panel on Soils (2015), Soil initiatives in UN environmental conventions (UNCCD, CBD,UNFCCC), Post 2015 Sustainable Development Goals (2015) and Voluntary Guidelines for Sustainable Soil Management (2016). We could say that besides the current complicated momentun for soil issues theoretically we have today more opportunities than ever before to promote soil conservation. The launching of the UN Sustainable Development Goals in September 2015 that are aimed at ending poverty and improving the lives of the poor, are tightly connected to protected and increase the productivity of soils. Yet only four of the seventeen targets specifically mention soil: Goal 2 (to end hunger, achieve food security and improved nutrition, and promote sustainable agriculture (Target 2.4), Goal 3 (to ensure healthy lives and promote well-being for all at all ages (Target 3.9), Goal 12 (to ensure sustainable consumption and production patterns (Target 12.4), and Goal 15 (to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation, and halt biodiversity loss (Target 15.3). To meet these goals Keestra et al. (2016)recommend the following steps to be taken by the soil science community as a whole: (i)embrace the UN SDGs, as they provide a platform that allows soil science to demonstrate its relevance for realizing a sustainable society by 2030; (ii)show the specific value of soil science: research should explicitly show how using modern soil information can improve the results of inter- and transdisciplinary studies on SDGs related to food security, water scarcity, climate change, biodiversity loss and health threats; (iii)take leadership in overarching system analysis of ecosystems, as soils and soil scientists have an integrated nature and this places soil scientists in a unique position; (iii)raise awareness of soil organic matter as a key attribute of soils to illustrate its importance for soil functions and ecosystem services; (iv) improve the transfer of knowledge through knowledge brokers with a soil background; (v) start at the basis: educational programs are needed at all levels, starting in primary schools, and emphasizing practical, down-to-earth examples; (vi) facilitate communication with the policy arena by framing research in terms that resonate with politicians in terms of the policy cycle or by considering drivers, pressures and responses affecting impacts of land use change; and finally (vii) all this is only possible if researchers, with soil scientists in the front lines, look over the hedge towards other disciplines, to the world at large and to the policy arena, reaching over to listen first, as a basis for genuine collaboration.

CONCLUSIONS

With a growing global population expected to reach 9.1 billion in 2050 and the increasing impacts of climate change, sustainable use of soil and ecosystems for food security is a crucial challenge.

The synthesis of a broad range of global land use scenarios, arable land availability trends, new socio-economic scenarios, earth system models and models of climate changeimpacts project great human induced transformations of terrestrial biomes

Lags in the underlying socio-economic, climate and global biogeochemical drivers make acceleration in land transformations and degradation inevitable over the next several decades and require that mitigation and adaptation measures must be taken well before unacceptably large impacts on soil resources are observed

Thresholds, amplifying feedbacks and time-lag effects leading to "tipping points" are widespread and make the impacts of global change on soil resources hard to predict, difficult to control once they begin, and slow and expensive to reverse once they have occurred. Also in some instances sudden catastrophic shifts could lead to intense, rapid and nonlinear land degradation responses

The formidable challenge of increasing agricultural production, keeping soil environmental requirements and restoring soil ecological functions and services requires better soil conservation practices to enhancing the biological component of soil. This is a win-win strategy because its lowers the risk of land degradation, improves food production; improves water efficiency; protect nature ecosystems thus biodiversity; is crucial for restoration of degraded land; to combat desertification and to reduce the impact on human health (Rice, 2014)

Enhancing soil conservation and its biological dimension should be place as a cornerstone for the future of the Earth enable to provide food and peace for mankind.

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DIAGNOSTIC CRITERIA FOR SOIL DEGRADATION
NECESSARY DISTINCTIONS FOR TROPICAL ENVIRONMENTS

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INTRODUCTION

The tropical (or inter-tropical) zone contains regions with highly diverse climatic, biological, geological, geo-morphological and demographic properties. The International Union for Conservation of Nature (IUCN, www.iucn.org/) lists 160 countries and island groups that have all or parts of their landmass in the tropics. Demographically, population densities in tropical regions are the highest and show little tendency to decline with time. Farming is the primary occupation, and because productive lands are no longer available, expanding population needs have forced agriculture to “encroach” onto marginal lands previously in forests or other natural vegetation. Inappropriate use and management of these fragile areas have often led to rampant degradation. Desertification abounds in arid and semiarid areas with soil salinization of irrigated lands and accelerated erosion in most climatic sub-zones are, respectively, the most dominant chemical and physical degradation processes.

THE SOILS

It has long been recognized that soils in the tropics are so highly diverse that most *Orders* in the U.S. Soil Taxonomy are found in the region. The Hawaiian Islands are a microcosm whereby most soil orders are present. However, *the highly weathered, red, acidic, well-aggregated but nutrient poor **Oxisols** are the most distinctive soils because they are largely restricted to the tropics* (Figure 1). To be optimally productive, these soils often require substantial costly inputs, including irrigation, plant nutrient supplements, and enrichment with organic matter. Their structural and hydrologic behaviors are highly determined by strong structure, high aggregate stability and mineralogical composition that is dominated by variable charge characteristics.

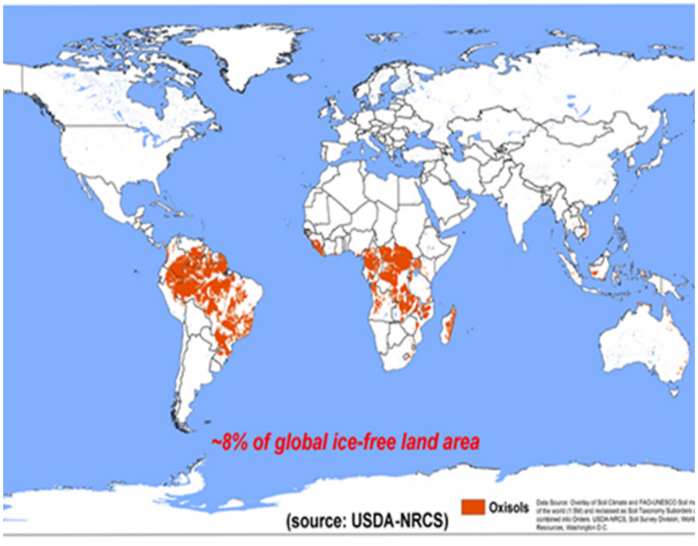

Figure 1. Global Distribution of Oxisols

VARIABLE CHARGE CHARACTERISTICS AND CONSEQUENCES

A typical matrix of a highly weathered tropical soil includes primarily 1:1 silicate clay minerls and a substantial content of metallic sesquioxides (up to 50% in certain gibbsite- and goethite-rich soils). With such high contents, it is important to consider their roles as fundamental soil constituents, not merely “cementing agents”. While silicate clays possess fixed negative charges; sesquioxides are amphoteric and may display negative or positive charges depending on pH (figure 2).

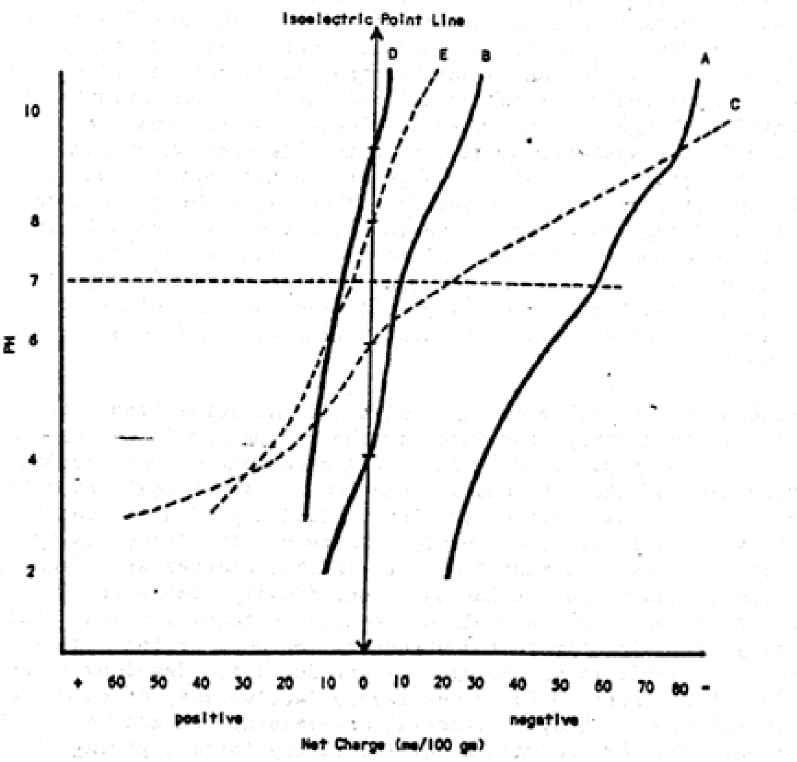

Figure 2. Dependence of net charge on pH for major soil mineral constituents: A - 2:1 layer silicates (e.g. montmorillonite), B - 1:1 layer silicates (e.g. kaolinite), C - amorphous/poorly chrystallized minerals (e.g. allophane or ferrihydrite), D - hematite (crystalline iron oxide), E - cryslalline metal hydroxides (e.g. gibbsite and geothite). Source: ***http://www.ctahr.hawaii.edu/huen/tpss435/El-Swaify.pdf***

Measuring soil pH in 1:1 suspensions with distilled water and 1M KCl yields a value for **ΔpH**, an important index of the soil's net charge:

$$\Delta pH = pHKCl - pHH2O$$

Negative, zero and positive **ΔpH** values indicate that the soil carries a net negative, zero or positive charge, respectively. The implications of 1:1 mineral and sesquioxide abundance, and variable charge attributess to the chemical and physical behavior of highly weathered tropical soil systems include:

- possessing more postive than negative charges at normally encountered or modified soil pH values, and therefore a high capacity for retaining anions than excluding them,
- displaying less selectivity for Na sorption than do typicall temperate zone soils,
- even at high exchangeable sodium percentages, the soils display strong resistance to structural breakdown ,

- water retention and release patterns resemble those of sands but with significanly more retention of (partly accessible) water by aggregates at higher tensions, and
- low inherent susceptibility to erosion as reflected by low soil erodibilty values.

SOIL SALINITY AND SODICTY CASE STUDY

El-Swaify *et al* (1977)) confirmed the following needs for standardizing the methods of diagnosing soil salinity and sodicity in a group of irrigated soils in Hawaii:

a. Water content for saturated soils: Soil Due to the wide variation in human judgements of what constitues a prepared saturated soil paste, it is recommended that saturation water content of a particular soil be determined by capillarity. This saturation value can be used for preparing reproducible saturated water extracts, with a time saving "bonus".

b. Adjusting expected changes in soil solution salinity after irrigation: Strong aggregation of highly weathered soils results in achieving the *"field capacity"* status at lower water tensions than the conventional 0.3 Atm. As a consequence, the encountered concentration of soil solution after irrigation is considerably lower than conventionally assumed. This is another dimension of salinity "tolerance" by these soils when irrigated with marginal quality waters.

c. Predicting exchangeable sodium values from soil solution composition: As figure 3 shows, highly weathered soils have less affinity for exchangeable sodium than predicted by commonly used models. Since certain crops (and most soils) are negatively affected by exchangeable Na, it may be concluded that highly weathered tropical soils can "tolerate" irrigation with high salinity and sodium water more than expected for their temperate counterparts.

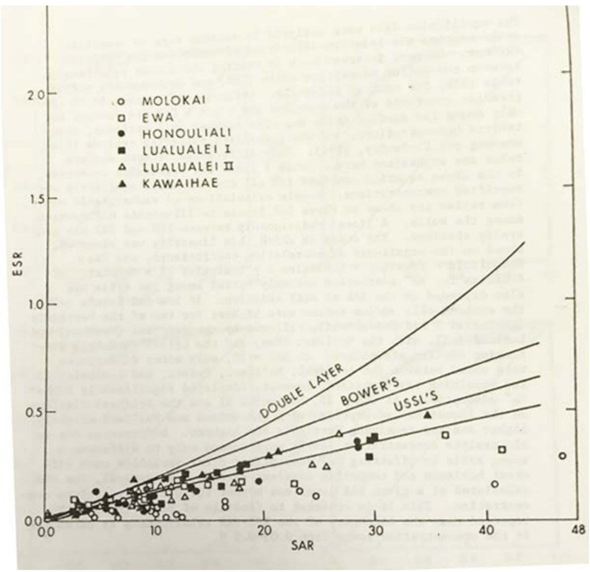

Figure 3. Scatter diagram for Hawaii soils relating exchangeable sodium ratios (Na/Ca+Mg) to sodium adorption ratios of saturation extracts, in comparison with 3 predictive models

d. Hydrologic and structural response to exchangeable sodium: An impressive cosequence of the mineralogical and structural attributes of highly weathered soils of the tropics is their high resistance to structural breakdown due to exchangeable sodium. This suggests allowing more flexibility when judging the quality of marginal water resources for irrigation (El-Swaify et al, 1977).

SOIL EROSION CASE STUDY

Watershed and field-plot monitoring, as well as rainfall simulation studies on the 10 important agricultural soils of Hawaii have shown that:

a. For the erosivity of rainfall: Based on the higher values for both the quantity and encountered intensities of tropical rainfall, several workers have found it necessary to create new predictive algorithms and maps for rainfall erosivity which expand rainfall erosion hazards beyond those established for temperate regions (Lo and El-Swaify , 1985).

b. For soil erodibility: Soil *structural* integrity, reflected by the percent of unstable soil aggregates, proved to be the primary determinant of soil erodibility in highly weathered soils. This is in contrast to earlier conclusions on temperate zone soils (e.g. Wischmeier and Mannering, 1969 cited by Roth *et al*, 1974)where *textural* properties were the dominant predictive parameters. Median annual rainfall at the study sites, a logical surrogate for the extent of soil weathering, was well correlated with measured soil erodibility values, although the correlation was distinctly different for residual and volcanic ash soils.

c. For erosion's impacts on soil productivity and water and nutrient use efficiency: **"**Conventional wisdom" in *developed* countries has often considered off-site impacts of soil erosion to be far more important than on-site impacts. The latter is deemed to be long-term and easily reversed or masked by external inputs. Neither observation is true for resource-poor families dominating farming communities in developing tropical countries. El-Swaify (2001) summarized his team's work on assessing erosion impacts on a variety of crops, and provided four additional reasons in support of short-term erosion impacts on highly weathered tropical soils. These are the lack of affordable external inputs, difficulty in restoring lost soil productivity, and the significant declines in both water and nutrient use efficiency.

CONCLUSION

Several decades of research in Hawaii and other tropical areas have demonstrated the necessity of using applicable and relevant diagnostic criteria for assessing soil degradation. To be sure, the provided examples also demonstrate a broad need for tailoring diagnostic methods and interpretations to many other specific locals.

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ANTHROPOGENIC SOILS AND SOIL SECURITY: ENVIRONMENTAL AND ECONOMIC CONSIDERATION

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1. INTRODUCTION

Long before the 20th century and up until the '60s of the last century, soil with its diversity, was considered almost exclusively in the context of farm management and food production.

In the second half of the last century, the remarkable increase of the global impact of humankind on the natural resources has lead in the public opinion an equal increase of the awareness of the importance of the environmental resources. The soil was no longer considered only a domain of agriculture but it was projected onto a broader and more appropriate scale, much more suitable to its environmental importance.

This started in the 1960s, and in conjunction with the beginning of the Anthropocene (Crutzen, 2002), an era influenced by agricultural, industrial and urban developments, that has affected the ability of soils to produce goods and services in greater quantity and better quality. In many cases, the human pressure on land was of such intensity as to transform the original soilscape pattern to a total disorganization, a pedological chaos that consequently led to the annihilation of the soil diversity (Lo Papa et al., 2011). The scientific debate that has developed in recent years on matters relating to the role of soil-systems in the environmental balance, allowed us to attribute to soils, properties and concepts that commonly characterize the living beings, such as that of genetic erosion (Dazzi, 1995) or diversity (Dazzi, 1995; Ibañez et al., 1995). These new ideas have led in time to the definition of the concept of Soil Security (McBratney et al., 2014).

2. ANTHROPOGENIC SOILS AND SOIL SECURITY: ENVIRONMENTAL AND ECONOMIC CONSIDERATION

Soil scientists have long been writing of the importance of soil to provide for growing human demand for food, water and energy, also expecting soil to provide ecosystem services that affect climate change, human health and maintain biodiversity.

A number of large existential environmental challenges have been recognized for the sustainable development of humanity and planet Earth. These are Food Security, Water Security, Energy Security, Climate Change Abatement, Biodiversity Protection and Ecosystem Service Delivery (Bouma and McBratney, 2013). They all have similar characteristics; namely, they are global, they are complex and difficult to resolve, and they are inter-related.

The inter-relationships between soils and social issues – such as food safety, sustainability, climate change, carbon sequestration, greenhouse gas emissions, degradation by erosion, loss of organic matter and nutrients – are fundamental elements of the recent proposition of the concept of "soil security" (Figure 1).

"Soil security" was define as the maintenance and improvement of the world's soil resource to produce food, fibre and freshwater, contribute to energy and climate sustainability, and maintain the biodiversity and the overall protection of the ecosystem (Bouma and McBratney, 2013).

Figure 1 – Food safety, sustainability, climate change, carbon sequestration, greenhouse gas emissions, degradation by erosion, loss of organic matter and nutrients – are fundamental elements of the "*soil security*" concept.

Soil security is linked to the six global societal challenges through the soil's Ecosystem Services (ES). These last are defined as the benefit that people derive from soils (Dominati et al., 2010) and, according to the widely adopted Millennium Ecosystem Assessment (MA) framework for ES (MEA, 2005), could be grouped in four categories: provisioning services; regulating services; cultural services; supporting functions.

Soil security also requires a value to be placed on soil. This value is aligned with the need for policy to aid in securing soils by encouraging correct soil management and protecting against mismanagement. Correct soil management form is also one of the main goal of pedotechniques, a new and interdisciplinary branch of soil science, which tries to understand and integrate the effect of soil handling on the soil qualities (Van Ouwerkerk and Koolen, 1988; Dazzi et al., 2009) and, obviously, on the soil's ecosystem services. In spite of this, in applying pedotechnique, farmers frequently do not take into account its fundamental aim: i.e. to satisfy the human needs avoiding any undesirable environmental threats that might occur during the handling of earthy materials. In agricultural management, the main aim of pedotechniques is to make agriculture more profitable.

Therefore, we are on the horns of a dilemma: how can assure soil security and save soil's ecosystem services in the Anthropocene era characterized by an always and always diffuse influence of humans on soil and an increasing spreading of the anthropogenic soils originated from pedotechnique?

Trying to give a solution to this dilemma, we could take into consideration a particularly meaningful case study that recently happened in Southern Sicily (Italy).

In a farm located in this area, in 2011 the landlords planned to change the land-use of their soils, used since long time to grow cereals (mainly durum wheat) and vegetables (such as eggplant, tomato, pepper, water melon). Therefore, they started to grow vines for table grape production. In 2012 using trucks, they started to cover 3 hectares of Vertisols with heaps of marly limestones. The marly limestones, that is to be considered as a human transported material (HTM), was extracted from a nearby hill making use of a Caterpillar that worked for several days. After this, In October 2012, the heaps of marly limestones were levelled above the soil surface. Three hectares of Vertisols were completely buried under a layer of marly limestone from 80 to 100 cm thick.

In summer 2013, this area was deeply ploughed at 90 - 100 cm depth, with a mouldboard one-furrow plough. This provided complete overturning of the HTM and a relatively stirring up of the topsoil of the Vertisols. Farmers, on this transformed soil, start to build a greenhouse completely equipped with a fog production system and an irrigation system. Inside the greenhouse, 4900 plants of vines were planted. At the end of these operations, the total cost was more than 120,000 euros per hectare. In 2014, vines grew rapidly and in 2015, they start to produce high quality table grape. In 2016, the greenhouse was in full production.

Therefore, we should ask a fundamental question: "What drives a farmer to do all this?" We believe that "profit" is the answer!

The profitability analysis of the table grape crop under greenhouse on anthropogenic soils, shows (from the second year of production), a high value of the Net income that is equal to € 34,561 per hectares. This is due to the higher average production per plant that amounts to 20 kilograms, sold directly on the plant at a price of €1.50 per kilo. This allows achieving a revenue of € 49,000 per hectare. The net income of the firm studied, deviates significantly from the profitability of other crops which are traditionally grown in Vertisol and in particular durum wheat, whose profitability is equal to € 488,00 per hectare. The cost/benefit analysis to evaluate the feasibility of the project highlights the greater convenience of the pedotechnique application, considering that all the economic performance indicators are highly positive.

Going back to the soil security issue, we stressed that also soil security would require a value to be placed on soil ecosystem services. Therefore, if we try to compare, even in a "qualitative" way, the ecosystem services provided by the two kinds of soil, we obtain the following information:

Soil service category	**Vertisol**	**Anthrosol**
Support (Biodiversity pool; Nutrient cycling; Soil formation; Water cycling, etc.)		
Regulating (Biological control of pest & diseases; Climate regulation; Hydrological control; Recycling of wastes and detoxification; Filtering of nutrients and contaminants, etc.)		
Provisioning (Biomass production; Clean water provision; Raw materials; Physical environment; etc.)		
Cultural (Heritage; Recreation; Cognitive; etc.)		

3. CONCLUSIONS

In the coming years, the socio-economic development of humankind and the maintenance of its prosperity will largely depend on its ability to ensure the sustainable use of the natural resources. This is a very complex task due to the impact of the human activities on the environment and particularly on soils. Soil Science will play a crucial role to achieve the need for an increasingly global and technological society. Soils must be considered in a new perspective: no longer limited to the agronomy and/or forestry issues but considered as important part of the environment and considerable element of the social and cultural systems. We believe that, in the near future, one of the issues that should be considered as a new frontier in soil science will concern the assignment of an "**economic value**" to the services offered by the soils.

Farmers consider soils as source of income, i.e. an economic resource! In large-scale farming for growing high-income crops using pedotechnique to tailor suitable soils for table grape cultivation, farmers are able to get a net income of more than 34,000 euro per hectare!

We are convinced that in overcoming the present limitations for studying and researching soil degradation processes and in the application of prevention and remediation practices, we need to change the soil paradigm, providing a new definition of soil:

"Soil is an economic resource! It is finite and non-renewable on the human time-scale since it does not regenerate at a significant rate within this time". Such revised definition stresses the economic value of the soils (the only one aspect that shake the attention of politicians and administrators!).

Being able to attribute an economic value to the processes of soil degradation by defining an algorithm to be considered in the calculation of the total GDP of a nation, would be a matter of major importance for the human society. It would cause the development of a strong perception of the importance of the soil as a resource and would increase awareness of soil importance in the environmental equilibria.

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PRELIMINARY FUNCTIONS OF SOIL AND WATER CONSERVATION PRACTICES FOR CLIMATE CHANGE MITIGATION AND ADAPTATIONIN CHINA

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Soil erosion, desertification, deforestation and the salinization and depletion of surface and underground water resources essential for agricultural production, threaten the future of food security and thus the national security of countries. Climate change threatens to accelerate all of these impacts. Fortunately, soil and water conservation practices can potentially conserve soil and water resources. Thus, WASWAC is releasing this position statement about the critical need to implement policies and management practices that conserve soil and water across the world for climate change mitigation and adaptation (*WASWAC, Position Statement on Climate Change and Soil and Water Conservation, ISWCR Vol.3 No.4, 2015*)。

In the past century, temperature rise in China has basically kept pace with global warming, with the average surface temperature increasing by 1.1℃, slightly higher than the global average.China has actively implemented policies and actions to adapt to climate change, particularly those to enhance the ability of the agriculture, forestry and other natural ecological systems, water resources, as well as ecologically fragile areas to adapt to climate change, and has achieved positive effects（*China's Policies and Actions for Addressing Climate Change, 2009*）.

Under the background of global climate Change changes in driving factors of soil erosion has been detected over the past century in China. Some comprehensive practices have been taken successfully to reduce the impacts of climate change on soil and water resources. The Grain for Green Project (GGP) is one of the most important projects for soil and water conservation in China, which was launched in 1999. GGP is aimed to focuses addressing soil and water conservation throughconverting the slope crop-land to grass and forest. The financial assistance and compensation from governmentare supplied to farmers and encourage them tostop cultivate on the slope-land and plant tree or grass instead. Till the end of 2013, more than9.3 million ha of slope farmland has been converted into forest or grassland and about 17.5 million ha of barren mountains and hills were re-vegetated in China. The coverage of forest and grass increases by about 4.5% for the whole China after GGP. The area of grain-cropping landin mountains and hills regions decreased significantly, however, the net income of each farmer increases significantly.A case study of 6 provinces, distributing different regions of China,has been conducted and the results showed that the GGP has many obvious ecological impacts. It is estimated to conserve water 18.3 billion M^3/year, to reduce soil loss 0.2 billion *tons/year*, to maintain soil nutrient 4.4 million *tons/year*, to fix C 14.0 million *tons/year*, to release $O^2$32 million M^3*/year*, to increase substance of wood accumulation 0.4 million M^3*/year.*The ecological benefit of GGP in this case study area every year total value is about 75 billion US$ according to price in 2013. Meanwhile, the run-off and sediment load decreased significantly in

11 rivers of China. Comparing with 1998-2002 the run-off and sediment load in 2003-2007 decreased by 18% and 45.4% respectively.

China will continued to implemented National Climate Change Program, intensified the effort to build capacities to address climate change. In December 2009, the Chinese government published the Action Plan of the Forestry Industry for Addressing Climate Change.This Action Plan stipulates the goals of three stages: Till 2010, more than 4 million hectares of land will be afforested annually, the national forest coverage rate will reach 20%, the forest reserves will reach 13.2 billion cubic meters, and the carbon-sequestering capacity of forests of the country will grow substantially; till 2020, over 5 million hectares of land will be afforested annually, the national forest coverage rate will reach 23%, the forest reserves will reach 14 billion cubic meters, and the carbon-sequestering capacity of forests of the country will achieve further growth; till 2050, the forest area will realized a net growth of 47 million hectares compared with 2020, the forest coverage rate will reach beyond 26%, and the carbon-sequestering ability of forests of the country will reach a stable level（China’s Policies and Actions for Addressing Climate Change, 2009）. The subsidies to GGP was extended in 2013 because of its obvious ecological functions, furthermore, there are some evidences show that GGP has improved the awareness of local farmers and changed the structure of regional economy. popularized nationwide.

In the Position Statement on Climate Change and Soil and Water Conservation, WASWAC recommends that governments and institutions should: (1)Develop policies that improve soil management to achieve a balance between increasing productivity and maintaining soil organic matter, reducing soil losses, and improving soil health and soil security. The carbon cycle is integral to how we manage soils, and soil carbon is one of the larger pools in the carbon cycle and increases soil productivity. (2)Encourage communication of soil and water conservation programs by communication systems that connect science to land managers and the public; teaching the value of soil carbon; increasing training; and enhancing exchange (e.g. at meetings, forums, etc.). (3)Develop, maintain, and/or expand programs for soil and water conservation practices for climate change mitigation and adaptation that maintain residue covers on soil surfaces, promote no-till systems, improve soil function with soil carbon; use multiple conservation practices at the field level and off-site; use precision conservation; promote energy efficiency; value water more; minimize greenhouse gas losses; improve nutrient cycling, nitrogen use efficiency and soil health.(4)Fund research in soil and water conservation that pays long-term dividends (*WASWAC, Position Statement on Climate Change and Soil and Water Conservation, ISWCR Vol.3 No.4, 2015*). In summary, it is necessary and practicable to apply soil and water conservation principles and practices to improve global sustainability and ecosystem services for Climate Change Mitigation and Adaptation in the world.

Keywords: Soil erosion, ecological restoration, adaptation, climate change

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TWO-THOUSAND YEARS DEBATES AND PRACTICES OF SEDIMENTATION MANAGEMENT OF THE YELLOW RIVER

Zhaoyin Wang

EXTENDED ABSTRACT

Throughout the history of China, the Yellow River has been associated with frequent flood disasters and changes of the lower reach course. The river carries sediment produced by soil erosion from the Loess Plateau and Qinghai-Tibet plateau and deposit the sediment on the channel bed and in the estuary. The erosion rate accelerated in the past 2000 years due to climate changes. The sedimentation rate increased from 1-3 mm per year to 30 mm per year from 00 AD to 1855 mainly due to human activities. From 1855-1985 human activities accelerated the sedimentation at an extremely high degree and the sedimentation rate rose to 50-100 mm per year. Over time, a perched river formed that frequently breached its levees. From 602BC to 1949 the river experienced 1,593 levee bursts, flooding vast areas in 543 years and claiming millions of human lives. The river shifted its major course (600-700 km long) by avulsion 26 times with the apex around Zhengzhou resulting in devastating calamities and numerous old channels, including 8 major shifts (5 natural and 3 human-caused) with the river mouth alternating between the Bohai Sea and the Yellow Sea. Because of its wild behavior, the lower Yellow River was dubbed “the sorrow of China”. The 700 km long lower reaches have swept throughout the north China plain and left numerous old channels (Fig.1).

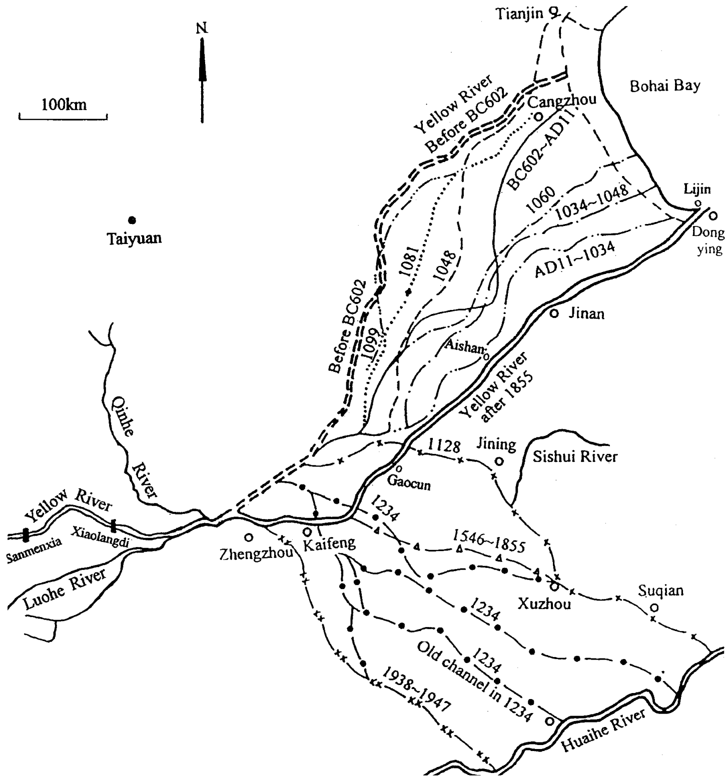

Fig. 1 Migration of the Yellow River and the abandoned channels (major avulsions)

Training of the Yellow River has a history of more than 3,000 years. Levee construction was the major strategy of flood control. Two extremely different strategies were proposed and practiced in the past 2000 years, i.e. wide river and depositing sediment strategy and narrow river and scouring sediment strategy. **Wang Jing** implemented a large-scale training project in 69AD. He completed and enhanced the levees and built many diversion channels and weirs. He constructed many gates about 5 km apart on the grand levees. The river was confined by the enhanced levees tens of kilometers apart, giving enough space for sediment depostion. The riverbed silted up at a low speed of less than 1-cm per year. During great events water and sediment were diverted through the water gates into diversion basins. In the following 800 years the river was calmed and no big flood disasters occurred.

From 850AD to 1500AD the river woke again and became very active. Sedimentation on the riverbed resulted in high flood stage approaching the crest of the grand levee. The Grand Levee was breached once per 2 years during the period. Closing the breached levee was a hard job for the river training engineers and the technology of levee defence was developed. Because of the population growth, the flood diversion strategy was more difficult to implement. **Pan Jixun** proposed the strategy of narrowing the river and confining the flood within the stem channel in order to raise the velocity and keep high the carrying capacity of the flood, preventing sediment from depositing and even promoting bed sediment scouring. He regulated the levee system, blocked many branches of the river and made the river flow in a single channel in the lower reaches in the period 1565-1592. After Pan Jixun the sediment deposition in the lower Yellow River channel sped up to 5-10 cm per year. The river migrated from south to north and captured the Daqing River in 1855 due to the levee breach at Tongwaxiang.

Shortly after Pan Jixun the river became very unrest and numerous levee breaches and flood disasters occurred. The third river training master Jin Fu took to the stage of history**. Jin Fu** (1633～1692) was appointed as the Viceroy of Yellow River Training at age of 45. The emperor Kangxi (Qing dynasty) entrusted him of flood control of the Yellow River and honored him the title of Minister of Industry. Chen Huang was the main assistant and adviser of Jin Fu. The Yellow River breached the levee frequently and caused great catastrophes as Jin and Chen took over the job of river training. Chen and Jin believed that the narrow channel and scouring sediment theory was suitable for the Yellow River training. They applied Pan's theory and practiced "converging flow with narrow channel and scouring sediment with high velocity of flow".

Since the 1950s the Yellow River Water Conservancy Commission (YRCC) has been the leading institute for river training. **Wang Huayun**, the chief of YRCC, proposed and implemented his training strategies in the period. The main strategies are to reduce flood discharge with reservoirs, enhance the capacity of the river channel by enhancing and reinforcing the levees, and retending floodwater with detention basins. These strategies are referred to in short as: *build a wide river and reinforce the levees, upper reaches storing, lower reaches discharging and two sides retaining.* Wang's strategy is almost the same as that proposed by Wang Jing.

In summary two conflict river training strategies have been proposed and applied in the past 2000 years: depositing sediment with wide river channel and scouring sediment with narrow river channel. Four great masters of Yellow River training have applied the two theories: Wang Jing applied the wide river strategies about 1,950 years ago, Pan Jixun applied the narrow river strategy about 450 years ago, Jin Fu applied the narrow river strategy about 350 years ago, and Wang Huayun applied the wide river strategy about 50 years ago.

Although the scouring sediment strategy looks much more sophisticated and received fulsome praise the long term effect of hazard mitigation was much lower than wide river strategy. Data of 543 levee breaches in the past 2000 years, which resulted in great flood disasters and death tolls, were collected and the number of levee breaches per century was calculated. Figure 2 shows the frequency of levee breaches as a function of time from AD 00 to 2016, in which the times of river training projects launched by Wang Jing, Pan Jixun, Jin Fu and Wang Huayun were indicated.

Wang Jing applied the wide river and deposing sediment strategy and brought about a long term (800 years) flood security after him. Pan Jixun applied the narrowing river and scouring sediment strategy and reduced the frequency of levee breaches quickly. Nevertheless, soon after his project the number of levee breaches increased to even higher level than before him. The same story occurred for Jin Fu. Jin brought down the frequency of levee breaches very quickly. Soon after his death the frequency of levee breaches reached an extremely high level (300 levee breaches per century, which is not shown in the figure because the point is out of the range of the figure). Wang Huayun applied the wide river and depositing sediment strategy from 1950. In the past 65 years the frequency of levee breaches reduced to zero. This is the only period of zero levee breaches in the past 4000 years history of the Yellow River. It can be clearly concluded that the wide river and depositing sediment strategy is much better than the narrow river and scouring sediment strategy.

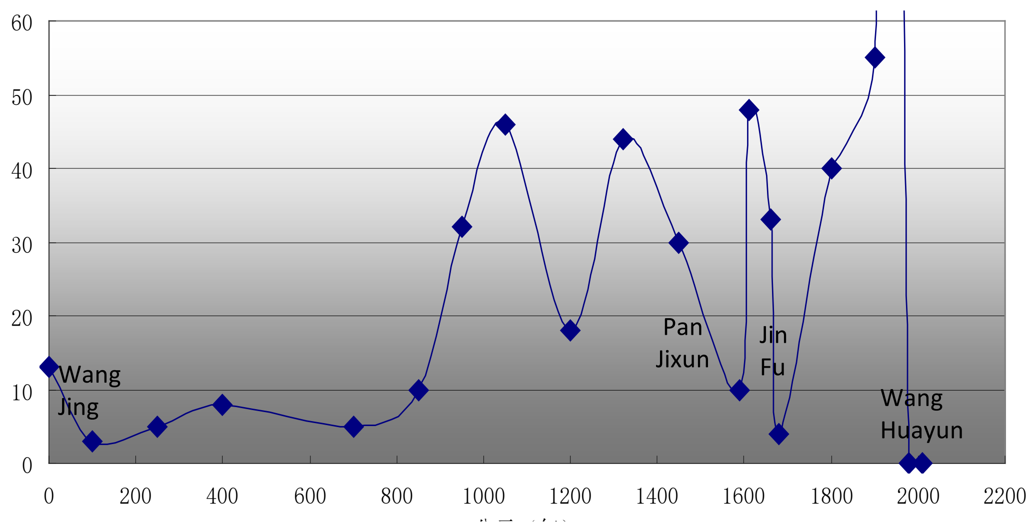

Fig. 2 Number of levee breaches per century as a function of time from AD 00 to 2016

The strategy of narrow river and scouring sediment is effective in sedimentation control, which can be used for local sedimentation control but not for the river training and flood control. In fact the narrow river strategy transferred the sediment problem to the downstream reaches and late time. There is evidence proving that the narrow river and scouring sediment strategy can cause long term consequence. In 1680 a flood from the Huaihe River carried a lot of sediment from the Hongze lake and buried the Sizhou Town and killed thousands of people, which is regarded a consequence of the narrow river strategy practiced by Pan Jixun and Jin Fu.

The debate on wide or narrow river strategies became international in the 20th century. Three international scientists should be mentioned who worked on the training of the Yellow River: Engels, Freeman and Franzius. American scientist Freeman visited China in 1917 and proposed to build cross dikes extending from the existing levees of the lower Yellow River, which were more than 6 km apart, and to build new levees near the tips of the dikes, 800 m apart. Freeman’s suggestion rekindled the century’s debate on whether the levees should be close or far apart as they were at the time. German scientist Engles conducted physical model experiments, authorized by the Chinese National Economic Council in 1931-1934. The test results indicated that with the levees set far apart, a

somewhat better scour was produced in the main channel than when the levees were close to the main channel edges. The result did not agree with the idea of further narrowing the river channel. Chinese government authorized Franzius, a student of Engles, conducted another physical model experiment and obtained different results. Yen (1999) indicated that Franzius' experiments were conducted without tail gate regulation and the results are not reliable as those of Engels.
The paper also discussed the new challenges and new strategies for the Yellow River training and management. Although the sediment load has been greatly reduced and there has being no levee breaches since 1950 the Yellow River delta has stopped land creation and the coastal line is retreating. Moreover, the river experienced flow cut off in the lower reaches due to water diversions. New strategies are proposed and practiced to solve these new problems. Sanmenxia Project was the first large dam on the Yellow River, which was regarded as a failure in the modern river training because extremely high rate of sedimentation caused flood in the Weihe River and the benefit from power generation had reduced to the minimum. Because the problems of siltation and induced flooding risk to the lower Weihe river have not been solved, decommissioning of the Sanmenxia dam has been under discussion for a long time as an alternative strategy to eventually solve the problem. The merit of the project and its role in the river training strategy are discussed and the future fate of the dam is outlooked.

KEYWORDS Yellow River, Levee breaches, Avulsion, Wide river and deposting sediment strategy, Narrow river and scouring sediment strategy, Sanmenxia Reservoir, Land creation

DYNAMIC ENVIRONMENTAL CONTROLS ON RAINFALL TRIGGERED LANDSLIDES

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EXTENDED ABSTRACT

Landslides have and continue to cause heavy damage to populations, property, and the environment, particularly in mountainous regions. A recent global survey of non-seismic triggered landslides reported an average of 4617 deaths per year over a 7-yr period (Petley, 2012). Rainfall triggers the majority of the world's landslides in steep terrain, but the complex interactions of hydrological, pedological, geomorphological, seismic, geotechnical, geological, hydrochemical, and biological processes that affect the initiation and propagation of these slope failures remains elusive. Much of the landslide research is focused on only one of these domains (particularly geotechnical and geomorphological investigations), and even though precipitation and hydrology are recognized as dynamic influences, the earth system and ecological effects are often assumed to be static. This paper attempts to synthesize major developments in hydro-eco-geomorphic processes that affect rainfall-initiated landslides with the ultimate objective to describe niches where new research is needed that will advance understanding of the dynamics of both hillslope and earth system processes linked to these slope failures.

In this paper, some of the underlying interactive processes are elucidated, including the dynamic role of vegetation in supporting slope stability (e.g., root strength, evapotranspiration, and effects on subsurface flow pathways); interactions of earthquake generated fissures on later landslide occurrence; effects of heterogeneous soils (particularly macropores); and the influence of topography. The interaction between soil moisture and cohesion is also outlined. These process interactions are discussed in the context of rainfall-initiated landslides triggered by both positive pore water accretion and loss of soil suction. A conceptual framework is presented to better understand these dynamic and interactive processes that provides improved knowledge to important questions, such as why certain hillsides fail during storms while others do not.

Tectonic activity can predispose hillslopes to landslidesby fracturing, folding, and other bedrock deformations that promote weathering and weakening of the substrate. Weathering of the regolith affects mineralogical composition, regolith strength, and hydrologic pathways, and can destabilize hillslopes. Geomorphic hollows (concave slopes) promote subsurface flow accumulation and are often sites of shallow, rainfall-initiated landslides. In contrast, during earthquakes, seismic waves tend to propagate to ridgelines and on convex slopes, lending these sites susceptible to landslides. Furthermore, fissures that develop on convex slopes and near ridgelines during earthquakes can promote rapid ingress of water that may influence the location and depth of future landslides initiated by extended rainfall or snowmelt.

Vegetation affects landslide occurrence in various ways, including root reinforcement, evaporation, transpiration, and root architecture that influences subsurface flow pathways. Because most of these processes are linked to root distribution, they more strongly affect shallow landslides where the root systems are dense and penetrate the entire soil mantle.Thus, root systems of woody species exert a

much greater positive influence than shallow rooted grasses. The role of forest canopies on the interception of incident rainfall typically plays a minor role on modifying shallow soil water and pore pressure propagation (Dhakal and Sullivan, 2014; Sidle and Ziegler, 2016). Slope reinforcement by roots has been extensively documented in field and modeling studies that demonstrate an increase in shallow landslide activity following clearance of woody vegetation (e.g., O'Loughlin and Pearce, 1976; Megahan et al., 1978; Sidle and Wu, 1999; Imaizumi et al., 2008).These studies indicate that there is a 'window' of approximately 3 to 15-20 years after forest clearing that coincides with an increase in landslide rate of about 2 to 10-fold compared to undisturbed forests.

The amount, timing, and pathways of water migration through soils and bedrock to potential failure surfaces are important for determining where and when a landslide will occur. These phenomenainvolve a complex set of broader-scale (e.g., hillslope shape and position, rainfall distribution), moderate-scale (e.g., location and connectivity of preferential flow paths), and small-scale (e.g., dynamic geotechnical and soil moisture) factors (Iverson, 2000; Sidle et al., 2000; Bogaard and Greco, 2015; Sidle and Bogaard, 2016).Preferential flow has been cited both as a factor that may cause pore water pressure accretion (where flow paths converge and possibly truncate) andmay also dissipate pore water pressure (where macropores act as efficient drains), thus having detrimental and beneficial effects, respectively, on slope stability (Uchida et al. 2001; Sidle and Ochiai, 2006; Bogaard and Greco, 2015). Most landslides occur due to a build-up of positive pore water pressure in the soil or regolith that occurs just above a hydrologic restrictive layer (e.g., bedrock, till) in response to rainfall infiltration, causing soil shear strength to decline to the point where the slope fails (e.g., Harp et al., 1990; Matsushi et al., 2006). Exfiltration from bedrock up into the soil mantle can also initiate landslides (e.g., Montgomery et al., 1997). The extent of pore water accretion is also influenced by antecedent moisture, with wetter conditions promoting more rapid pore pressure response during storms compared to drier conditions (Tsuboyama et al., 2000; Kuriakose et al., 2008).Rapid pore pressure accretion appears to be the most common and simplest triggering mechanism for landslides that occurs during an individual storm or prolonged rainfall period of several days. Slower, deeper-seated landslides are normally initiated by pore water pressure accretion, but infiltrating rain water or snowmelt is often buffered by thicker soil mantles, requiring cumulative rain and snow melt events to trigger a slope failure (Bogaard and van Asch, 2002; Handwerger et al., 2013).

Another trigger mechanism for shallow landslides involves a decrease in soil strength associated with a reduction in soil matric suction during wetting (Fredlund and Rahardjo, 1993). Debate persists concerning the importance of this mechanism in the initiation of natural landslides. A considerable volume of field evidence suggests that although this mechanism is an important contributor to the initiation of some landslides, it is rarely the primary or sole contributing factor in well-aggregated (structured) or heterogeneous soils (e.g., Sidle and Swanston, 1982; Terlien, 1997; Vieira and Fernandes, 2004; Lacerda, 2007; Hencher and Malone, 2012).In addition to the loss of shear strength during progressive wetting, the concurrent increase in soil weight during a rain event, as well as any external loading (surcharge) promotes instability of steep slopes (Sidle and Ochiai, 2006; Ray et al., 2010). The influence of increased surcharge due to rainfall is most effective in destabilizing very steep hillslopes with highly porous soils.

A better understanding of the complex interplay amongst meteorological forcing, subsurface hydrology, geotechnical soil properties, geomorphology, and vegetation is needed to develop more realistic predictive models for landslides and real-time hazard assessments.Advances in geotechnical behavior of soils at small-scales needs to be expanded and coupled with field-scale geomorphic and hydrologic studies.Generalizations of uniform pore pressure (both positive and negative) across hillslopes can miss conditions that may trigger a landslide or may overestimate instability. One approach to address this problem is to link dynamic pore pressure phenomenon with hydrogeomorphic processes in the vadose zone coupled with distributed effects of vegetation (particularly root systems). The manifestation of root systems at spatial scales equivalent to landslides needs to be investigated to better elucidate their effects on mechanical reinforcement and subsurface water routing associated with different biomes, vegetation management practices, and impending species adaptations due to climate change.More progress is needed related to hydrologic response to rainfall in unstable landscape units – e.g., geomorphic hollows, deeply weathered and altered landforms, terraced and urbanized landscapes, and areas with complex subsurface topography – and how these units may evolve over time. This involves a better understanding of feedbacks among temporally variable geomorphic infilling processes (including weathering), soil evacuation processes, and subsurface hydrologic dynamics during different rainfall conditions. Many of these challenges require a combination of skills that include process understanding through field investigations, theoretical developments, experimental approaches, analytics, and modeling.

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THE RANGELAND HYDROLOGY AND EROSION MODEL

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EXTENDED ABSTRACT

RHEM is a newly conceptualized model that was specifically designed to address rangelands conditions for estimating runoff, erosion, and sediment delivery rates and volumes at the spatial scale of the hillslope and the temporal scale of a single rainfall event. RHEM links the model's hydrologic and erosion parameters with rangeland plant community by providing a new system of parameter estimation equations based on diverse rangeland datasets through a simple web-enabled interface. Model inputs are surface soil texture, slope length, slope steepness, slope shape, dominant plant life form, percentage of canopy cover, and percentage of ground cover by component. Climate (precipitation intensity, duration, and frequency) is estimated for sites within the United States with the CLIGEN stochastic weather generator. RHEM uses this information to estimate the average annual soil loss during a 300-year time span and to estimate the vulnerability of a site to soil erosion based on the risk of experiencing a runoff event with a given magnitude (e.g., 10-, 25-, or 50-year return period storm events). Results are partitioned into probability distribution of soil erosion. RHEM uses the 50th, 80th, and 95th percentiles of the reference state erosion rates. This enables comparison of yearly soil losses of alternative states for different severity levels. They represent four soil erosion severity levels: low, medium, high, and very high and are useful for assessing soil health and sustainability of a site. RHEM model inputs and outputs are displayed in tabular and graphical form and multiple runs can be compared to assess how changes in cover characteristics from management practices will influence runoff, soil erosion and sediment yield.

The USDA-Agricultural Research Service (USDA-ARS) developed RHEM V1.0 originally from the Water Erosion Prediction Project (WEPP) model [Flanagan and Nearing, 1995]. However, the equations used in WEPP are based primarily on our understanding of erosion on croplands. Though many of the basic hydrologic and erosion equations used in WEPP are equally applicable to rangelands, there were some aspects of the erosion process as modeled in WEPP that were not appropriate for rangeland application. Many of these differences were addressed in the original RHEM version [Nearing et al., 2011].

As RHEM V1.0 was initially developed it did not have the capability to deal with applications to disturbed rangeland conditions such as those occurring post-fire. The primary difference between the disturbed and undisturbed state for erosion is related to the occurrence of erosion in small micro-channels on the hillslope, whereas concentrated flow erosion is limited on undisturbed sites and most soil loss occurs by rain splash and sheet erosion. A new splash and sheet equation developed by Wei et al. [2009] based on rangeland rainfall simulation data collected from the WEPP and IRWET [IRWET and NRST, 1998] projects, which included 49 sites in 15 western states.

Versions of RHEM V2.+ are much improved and essentially different from previous versions. These versions utilize a dynamic form of the stream-powered sediment continuity equation [Bennett, 1974] rather than the steady state approach driven by shear stress for modeling concentrated flow erosion. Many previous studies have shown that using streampower as the predictor for unit sediment load is preferable to shear stress (or unit stream power) (Nearing et al., 1997 and others). Al-Hamdan et al. [2012] conducted experiments with small concentrated, overland flow simulations on both disturbed and undisturbed soils to develop soil erodibility parameters and to evaluate relationships between erosion and hydraulic parameters. Those results indicated that stream power gave the best linear fit for these soils. RHEM V2.+ also uses a stream-power based empirical equation (Nearing et al., 1997) to estimate sediment transport capacity. In versions of RHEM leading to the current V2.3, parameter estimation equations for describing and quantifying disturbed rangelands, and several other minor changes have been incorporated.

RHEM parameters are based in part on the types of plant vegetation present at a site, based on 4 plant lifeforms of bunch grass, sod grass, shrub and trees, or annual grasses and forbs. It also is tested for application to disturbed condition such as fire, climate change, and rangeland management practices, which impact infiltration, runoff, and erosion processes on rangelands. The model has a user-friendly web-based interface that allows one to use RHEM with very common measured attributes of the land, such as slope steepness, vegetation cover, climate location, and soil texture. The interface allows the user to manage alternative scenarios, view results, compare results, and output tabular and graphical results.

The model is able to use climate information provided by the user, but easy-to-use climate data through the interface is available only for the United States and four stations in Lebanon. There is a current emphasis to build and make available climate data files from around the world. The RHEM team welcomes collaboration with scientists and users outside the United States and will work with users to develop climate files for their application. Currently there are plans to develop a climate database for Kazakhstan for assessing grazing lands across that country.

RHEM has been and is being used for a variety of application in the US. It was used in a report to Congress on the state of rangelands in the western U.S., where it was applied to measure site data from more than 10,000 locations as part of the US National Resource Inventory [Weltz et al., 2014]. It is being used to help inform the development of Ecological Site Descriptions across the U.S. as part of a formal project within the USDA [Williams et al., 2016]. It has also been incorporated into the workflow technology for conservation offices across the U.S, via the Conservation Desktop and the Integrated Erosion Tool being developed by NRCS. It is also being incorporated into the HEC RAS modeling system at EPA.

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IMPACTS OF RE-VEGETATION ON SURFACE SOIL MOISTURE OVER THE CHINESE LOESS PLATEAU AND NEW CHALLENGES OF SOIL AND WATER CONSERVATION

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ABSTRACT
A large-scale re-vegetation project of the Grain for Green Project (GGP) has greatly changed local eco-hydrological systems, with an impact on soil moisture conditions for the Chinese Loess Plateau. It is important to know how, exactly, re-vegetation influences soil moisture conditions, which not only crucially constrain growth and distribution of vegetation, and hence, further re-vegetation, but also determine the degree of soil desiccation and, thus, erosion risk in the region. In this study, three eco-environmental factors, which are Soil Water Index (SWI), the Normalized Difference Vegetation Index (NDVI), and precipitation, were used to investigate the response of soil moisture in the one-meter layer of top soil to the re-vegetation during the GGP. Two separate periods, which are 1998–2000 and 2008–2010, were selected to examine the spatiotemporal pattern of the chosen eco-environmental factors. From 1998–2000 to 2008–2010, the average annual NDVI increased for 80.99%, while the SWI decreased for 72.64% of the area on the Loess Plateau. The average SWI decreased by 4.37% for two-thirds of the area. More specifically, 57.65% of the area on the Loess Plateau experienced an increased NDVI and decreased SWI, 23.34% of the area had an increased NDVI and SWI. NDVI and SWI decreased simultaneously for 14.99% of the area, and the decreased NDVI and increased SWI occurred at the same time for 4.02% of the area. These results indicate that re-vegetationmay be the major factor for soil moisture change in most areas of the Loess Plateau and that challenges the on-going soil and water conservation.
KEYWORDS: the Loess Plateau; Grain for Green Project (GGP); Soil Water Index (SWI); Normalized Difference Vegetation Index (NDVI); remote sensing; precipitation; China

EXTENDED ABSTRACT

Water is recognized as a critical natural resource that restricts vegetation development. The water use of plants mainly comes from soil moisture on the Loess Plateau (Li and Duan 2012). The variation and regional differences in surface soil moisture play a significant role in the water and energy exchanges which not only affects the individual vegetation growth but also influences the vegetation type and the distribution (Wang 2002). Natural precipitation is the only source of soil moisture in arid and semiarid areas of northwest China (Shao et al. 2004).

The "Grain for Green Project" (GGP) was initiated by the Chinese government in 1999 to control the severe soil erosion and flash floods on the slopes, to reduce the sedimentation of the water engineering and river beds and to improve the sustainability of the Loess Plateau (Chen et al. 2007). It has shown remarkable success in ecological restoration and agricultural production.

A semi-empirical model (Wagner 1998) has been proposed to estimate the soil moisture (the soil water index, SWI) derived from the Europe Remote Sensing Satellite (ERS 1-2) by the Institute for Photogrammetry and Remote Sensing, Vienna University of Technology. The SWI data in this study represent the soil moisture in the surface soil layer, which is highly dynamic and shows the high correlation with precipitation and its relationship to vegetation cover (Wagner 1998).

The objective of the present study is to understand the impact of the large-scale, re-vegetation by the "Grain for Green Project of China". Analysis of vegetation, soil moisture and precipitation will help us better understand the complex interactions of the relationship among the hydrosphere, biosphere and atmosphere.

MATERIALS AND METHODS

Study site

The Loess Plateau lies in the upper and middle reaches of the Yellow River of China, covering an area of approximately 622,009 km^2. The Loess Plateau covers arid, semi-arid and semi-humid areas, and the average annual precipitation was approximately 400 mm, with a large range from more than 600 mm in the southeast to less than 200 mm in the northwest. The seasonal distribution of the precipitation is very uneven due to the great influence of the monsoon season (Wang 2015), and approximately 69.33% of the precipitation as intensive storms. The average annual evaporation ranges from 820 to 1650 mm (Yang and Yu 1992).

Data

***Profile soil moisture**:* The soil moisture data were downloaded from the Essential Climate Variable (ECV) Soil Moisture dataset. The spatial resolution of the SWI data was 50 km, and the temporal resolution was approximately ten days (Wagner et al. 1999a).The soil moisture content in the soil profile, indicated by the soil water index (SWI), represents the soil moisture content in the 0-1 m layer of the soil in relative units ranging between wilting level and field capacity level. SWI is influenced by vegetation and soil surface roughness and the change detection algorithm developed by TU-Wien has been applied to eliminate these effects (Wagner et al., 1999b; Scipal, 2002; Fontaine, 2007).

***NDVI data**:* The NDVI (normalized difference vegetation index) data linked to green vegetation photosynthetic activity (Zribi et al. 2009) is often used as a monitoring tool for the vegetation health and dynamics, enabling easy temporal and spatial comparisons.

Precipitation data: The monthly precipitation data of 101 weather stations within and near the Loess Plateau were obtained from the China Precipitation Administration National Precipitation Information Center (http://cdc.cma.gov.cn/home.do). Because the implementation of GGP encompasses more than ten years on the Loess Plateau, we chose data for analysis in 2 stages: the beginning (Stage 1: 1998-2000) and ten years later (Stage 2: 2008-2010).

METHODS

Scale harmonization

The precipitation, NDVI and SWI data, which have different spatial resolutions, were transformed into the same scale of 25 km * 25 km.

Seasonal variation of precipitation, NDVI and SWI

Thespatial distribution characteristics of the three factors, precipitation, NDVI and SWI, in different seasons are discussed in the paper.

Statistical characteristics of precipitation, SWI and NDVI within seasons

The basic statistical characteristics, such as maximum, minimum, mean, and Standard Deviation (STD), the coefficient of variation (CV), are analyzed.

Cross Analysis process of three factors

Firstly, the annual NDVI values were computed from monthly data using the maximum value composite (MVC) method. Secondly, mean annual value in 2 stages (1998-2000 and 2008-2010) were calculated as averages from multiyear values. The NDVI mean annual values were matched with each corresponding rectangle cell. Thirdly, the annual SWI values were obtained through computing the arithmetic average of monthly values, and the mean annual value of this SWI processing is the same as NDVI. Then, the two layers of SWI and NDVI, which have the same resolution, are reclassified by the spatial analyst tools and, together, yield the intersection results in space.

MAIN RESULT

Thecomplex effects of precipitation on the SWI and NDVI: The complex changes of precipitation in the 4 change types of the SWI and NDVI are shown in Table 1.When the area of the SWI decreased and the NDVI increased, precipitation had a different trend; it accounted for a 32.59% increase in the northern Loess Plateau and a 33.41% decrease in the southeast of the Loess Plateau. Another interesting result was in the west of the Loess Plateau; here, the precipitation decreased by 5.03% in Jingyuan County and the Baiyin District; southwest of the Loess Plateau and increased by 1.61% in the northwest Loess Plateau.

Table 1 The statistical changes of precipitationand changes (in % area) of the 4 NDVI-SWI types

Precipitation			A	B	C	D
Change	Range in mm	NDVI	Decrease	Increase	Decrease	Increase
		SWI	Decrease	Decrease	Increase	Increase
Decrease	(-364, -100)		0.70	5.33	0.00	1.21
	[-100, -80)		1.01	4.13	0.10	0.50
	[-80, -60)		0.91	4.23	0.00	0.81
	[-60, -40)		1.21	4.02	0.00	0.70
	[-40, -20)		0.50	7.95	0.10	0.70
	[-20, 0)		0.70	7.75	0.00	0.20
Increase	[0, 20)		0.20	6.94	0.31	3.12
	[20, 40)		0.20	8.05	0.20	3.62
	[40, 60)		0.70	7.24	0.50	7.04
	[60, 80)		0.31	6.54	0.40	4.23
	[80, 100)		0.20	2.41	0.00	1.91
	[100, 140)		0.00	1.41	0.00	1.71
Decrease	E (-364, 0)		5.03	33.41	0.20	4.12
Increase	F [0, 140)		1.61	32.59	1.41	21.63
	(-364, 140)		6.64	66.00	1.61	25.75

Note: A, B, C and D means the NDVI-SWI type, respectively; E and F represent the precipitation's total decrease and increase interval, respectively.

CONCLUSIONS

The effect of vegetation restoration is remarkable that produced obvious ecological benefits after the implementation of GGP.Compared the NDVI, extracted from remote sensing data, in 2 stages of the GGP: the beginning (1998-2000, Stage 1) and ten years later (2008-2010, Stage 2), NDVI significantly increased by 91.75% on the entire Loess Plateau. A decade later, the NDVI in spring, summer and autumn increased in the Loess Plateau by 14.7%, 13.5% and 19.1% respectively.

SWIshowed that soil moisture decreased by 72.64% on the entire Loess Plateau. Soil moisture conditions present a tendency of drought in the future. There are about 66% of areas where the vegetation coverage increased whereas the surface soil moisture decreased, of which near a half experienced increasing precipitation.

The results suggest that we must focus on the surface soil moisture conditions in large scale construction of re-vegetation. In particular, it should take into account to select appropriate vegetation species and proper planting density according to the soil moisture conditions in arid and semi-arid areas. Therefore, the water cycle and ecological recovery will become sustainable.

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MINING SOILS IN THE ARGENTINEAN PAMPAS: HIDDEN COSTS DERIVED OF TECHNOLOGICAL INTENSIFICATION IN INDUSTRIAL AGRICULTURAL MODELS

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EXTENDED ABSTRACT

A RURAL LANDSCAPE TRANSFORMATION

Argentina has been a large food and biomass supplier to the world for more than a century. In 2013 it represented the third largest soybean, third largest lemon and forth largest maize producer worldwide, while in 2015 it was the second top supplier of dietary energy as food available for human use with more than 3500 of kcal/cap/day.

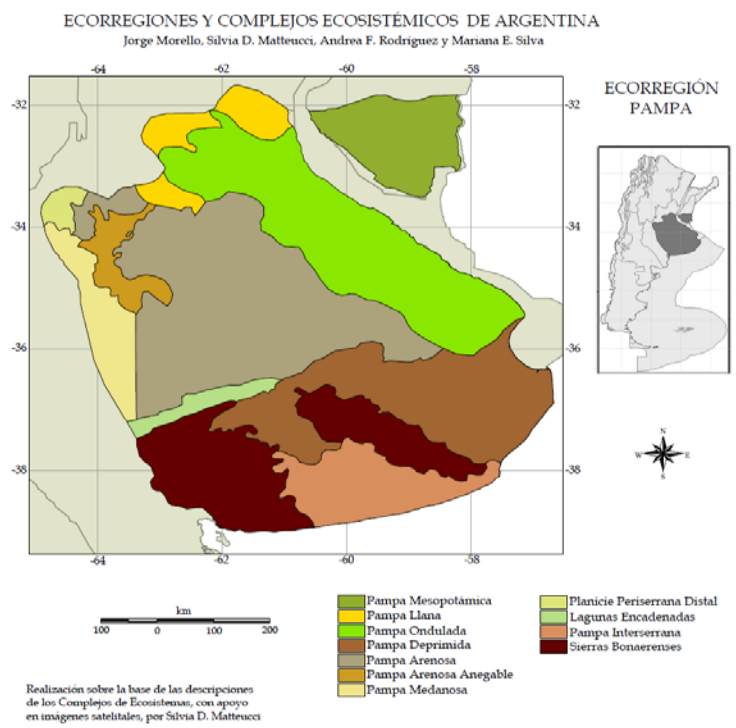
With a territory of more than 2.7 million km2, it was positioned fourth in arable land per capita and represents the second largest country in South America that accounts a large agricultural region (the Pampas) with fertile soils and a favorable climate, surge of commodity flows to the rest of the world over the last 40 years.

Soils in Argentina are classified in eight orders from the Soil Taxonomy classification. The most abundant order is the Molisols, on which occur most of the cash crop agriculture and intensive cattle operations (mainly cattle and dairy). Dryland agriculture is significant in most humid areas, where this group comprises the most fertile soils of the Pampas. Entisols and Aridisols are undeveloped soils with low water holding capacity, important in arid and semiarid areas. The fourth group in terms of the occupied area is the Alfisols, located mainly in the humid subtropical areas of north-eastern Argentina. These four groups account for more than 80% of the country's lands, most of them are going through an intense process of agriculturization.

The soil resources in Argentina have been the main support of economic development in the country. The main area of the country is the Pampas (See Ecorregion Pampas Map, Morello et al, GEPAMA, FADU, UBA). The Humid Pampas occupies somewhat less than one third of the territory, where plains are dominant, formed by modern unconsolidated sediments, with natural grasslands and temperate climate. Highly contrasting are the remaining two thirds of the surface of Argentina, most of which are dominated by arid climate.

Pampas's soils were the basket of food of Argentina and part of the world. Its potentiality for agriculture production was considerer relevant. This sediment is known as Pampean loess because of its similarities with loess materials and deposits in other parts of the world. The physical characteristics of the Pampean loess favour the formation of well structured, deep, dark surface horizons, adequate for root development.

Since twenty years ago, "industrial agriculture" in hands of an intensification of agrochemicals and transgenic crops were the main factor of transformation of the rural environment so in The Pampas such as in the Chaco ecorregion. Process such as agriculturization (soybeanzation) and pampeanization were the economic, financial and agronomic factor that put under pressure land and soils in Argentina and Latin America



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No Tillage practices, in the way that they have been implemented in the country (monoculture), supported by millions of tones of Glyphosate and others chemicals produced a nature reaction that produces several impacts. In the current situation, the industrial agriculture model in Argentina shows two Damocles' swords: the appearance of weed resistance (superweeds), deforestation on marginal borders and nutrients depletion in terms of extraction of vital chemicals in biomass exported.

SOCIAL METABOLISM AND INTERNATIONAL TRADE

Land and Soils are relevant natural resources that allocate Latin American and Argentina potentialities among the big players of the global food system. International trade and global social metabolism is rising drastically. Material flows, virtual soils and nutrients flows, virtual water and other embodied materials in trade are being considered as key indicators for the global interaction and sustainability of the process.

Several sustainable development goals are directly related with material flows, natural resources transferences embodied materials.

In the case of developing countries, exportation of commodities means opportunities and risks. Land and soils, as the main issue for food production, agroindustry, biomaterials and biofuels (biomass in general), are the core for sustain the supply and stability in the global and regional food security and activities involved.

For the global context, the circulation of commodities, particularly Agrifoods (cereals, grains, meat, milk, eggs) has been growing sharply in recent decades and are now part important to the global flow of goods in the world economy.

Technological changes in agricultural models have been one of the significant issues that sustain these trends and transformations.

MODERN AGRICULTURE AND NUTRIENTS DEPLETION

The introduction of transgenic crops, No tillage practices and herbicides control (called "The technological Package") in Argentina, where the main elements that transformed the landscape of the country. Agricultural landscape were transformed so in the Pampas as in the Chaco Ecorregiones. The mentioned process of *agriculturization, soybeanzation* and *pampeanization* where relevant process that can be detected in a single view. But other process, that involve soils depletion, in terms of stability, quality and nutrients extraction.

While exporting countries such as Argentina, are selling their products, they are not incorporating their intangibles values (e.g. soils nutrients), in terms of overexploitation (soils, water, biodiversity), and the cancellation of relevant environmental services (biogeochemical cycles, water cycling, pollination, e.g.). There is a metabolism of materials that needs to be evaluated, particularly those materials that are relevant for the stabilization of the food chain, such as the soils, particularly their nutrients, in terms of nutrients flows and balances (**nutrients footprint**).

The real flow of these agroindustrial products could be sustained as the result of not incorporating the externalities in terms of ecological and social costs. In each exported product, we send a part of relevant "environment" that developing countries are selling at a very low value.

If water means life, soils are the basket that contains it. Soils are and will be unless during the coming fifty years **the support of food production and the main substrate that contains the life and nutrients cycles of ecosystems**. Land degradation in the form of soil erosion, nutrient depletion, water scarcity, salinity and disruption of biological cycles is a fundamental and persistent problem. Latin America is losing its best soils in hands of the exportation of it with no an adequate management.

METHODOLOGICAL APPROACH

The long-term performance of biomass production in Argentina can be analyzed from a biophysical perspective and as a metabolism approach for agricultural production, used to investigate the physical base of socioeconomic systems in the fields of ecological economics and industrial ecology.

We devised a simplified conceptual model to explain nutrient flows in biomass production from a sink and source perspective, considering those major and most relevant import and export fluxes for which reliable and long-term data sets were available. This was used to identify trends in nutrient input and output flows and induces soil's nutrient stock variation from an indirect approach. Even though scaling-down estimations may lead to error, the advantage of this approach lays on the fact it focus on global nutrient quantities in sources and sinks that are easier to estimate, when gains and losses are highly soil-, climate- and crop-specific. Following a mass balance principle, all the material that enters into the system must be equal to the outputs plus material accumulation.

This allows the construction and maintenance (or depletion) of the system's stocks compartments, which are considered essential to the system's performance. Nutrients studied were nitrogen (N), phosphorus (P) and potassium (K), due to their critical relevance for biomass production worldwide

The intensification of Latin American and particularly, Argentine agriculture is related to the reduction of nutrients in the soil, land use change, water consumption and the deforestation process, that are causing relevant environmental and social costs for the region.

In this Lecture, we will show the numbers and revise the original view of Raul Prebisch for the Latin American model under a core-periphery theory XXI with physical flows as a new focus, and particularly its relationship with the current situation from a regional and historical perspective under the light of the new situation with the industrial agriculture, the agronomic and economic results of this modernization and the ecological and social costs of the whole process.

NPK flows were analyzed considering major nutrient input and output flows over a 55-year period of biomass production in Argentina. NPK nutrients in fertilizer production (NFP), imports (NIF) and exports (NEF) were used to determine the NPK nutrient apparent consumption from fertilizer source (NFC). Nutrient actual consumption by the plants was then measured indirectly by estimating nutrient losses (NL) from fertilizers consumed yearly.

We estimated N, P and K nutrient flow accounts for biomass production of 74 crops, from 1961 to 2015, to analyze long-term nutrient intensity use and better understand spatial and temporal variability of nutrient dynamics in Argentina. Estimated nutrient harvested accumulated 113 Tg (76 Tg N, 11 Tg P, 26 Tg K), equal to an annual extraction of 64 kg N ha^{-1}, 9 kg P ha^{-1} and 22 kg K ha^{-1}. Nutrient supplied in fertilizers explained 19% of total removed, or 13.5 kg N ha^{-1} (13%), 3.8 kg P ha^{-1} (36%) and 0.8 kg K ha^{-1} (3%). 65 Tg (69%) of NPK collected came from sources other than fertilizer or BFN, implying soil depletion. Nutrient balances are -46 Tg N, -7 Tg P and -21 Tg K, equal to -38 kg N ha^{-1} yr^{-1}, -6 kg P ha^{-1} yr^{-1} and -21 kg K ha^{-1} yr^{-1}. Soybean was the most extractive crop, with 52% of total NPK removed, or 173 kg NPK ha^{-1}.

Unfortunately, global trends and the original conditions of the Pampas soils, hide the environmental and agronomical costs, that the intensification of agriculture is promoting the rural landscape.

FINAL COMMENTS

The Pampas are a relevant prairie for helping the world in the SDGs 2015-2030. Main transformations go through biomass production with an important extraction of soils nutrients.

This soil exhaustion represents a 'hidden cost' or environmental intangible, since nutrients exported from soils as natural capital remains unaccounted for.

The relative low input of fertilizer use in Argentina and the high fertility of the soils lead the way to an export of nutrient capital as embodied material in harvested organs, or 'virtual soil'. When assessing the performance of the past century agricultural impact in Argentina, the reduction of nutrient capital is evidenced

For farmers and rural society, nutrients depletion, highly costs of fertilizers and increasing costs for weed control are putting under discussion the real benefits of the transgenic package and No Tillage system that arrives to the country two decades ago, and that has been presented such as reaching the heaven. Today, the whole system is facing a nightmare that is being tried again, regrettably, with the similar tools that created the problem.

CREDITS

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SOYBEAN EXPANSION IN BRAZIL: LAND USE CHANGES AND SOIL MANAGEMENT CHALLENGES

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EXTENDED ABSTRACT

1. Soybeans in Brazil

The area cultivated under soybeans in Brazil has grown exponentially over the past 40 years, reaching 32 million ha for the 2015/16 season, turning Brazil into the second largest producer of soybeans worldwide (97 x 10^6 metric ton). With the expansion of the pork and poultry industries since the 1960's, the international market has increasingly demanded soy meal as a source of animal fodder. Prior to this, wheat was the primary crop in southern Brazil due to this region's favorable winter climate, and soybeans became a popular summer succession crop. With the increase in soy prices on the international market in the 1970's, the Brazilian government began subsidizing the purchase of crop amendments, fertilizers and machinery to facilitate its production. The coincidence of the harvest period in Brazil with the North-American winter, when international soy prices reach their peak values (EMBRAPA, 2017), is another factor that propelled the growth of soy production in Brazil.

2. Environmental consequences of the expansion of soy in the South and Cerrado regions of Brazil

Throughout the 1970s and 80s, soy occupied about 10 million hectares and was concentrated in the southern states of Brazil, predominantly in basalt soils (oxisols) with deep profiles, good drainage, low natural fertility, and limited phosphorus availability. In this region, soybean cultivation did not lead to significant changes in land use, since these areas had previously been cultivated with wheat. Only a few areas, such as the west of Parana state, were deforested to accommodate soybeans. But the expansion of the agricultural frontier was soon underway, pushing from the south into the south-central portion of the country (the Cerrado region), where conditions were favorable for production of subtropical varieties of soybeans. In this setting, there were significant changes in land use, as soy displaced rangeland. The soil management adopted to produce grains in this region caused substantial environmental impact due to high rates of soil erosion (> 20 metric ton yr^{-1}) and consequent sediment yield (Sorrenson & Montoya, 1989;Merten, 1994). This situation was worrisome to the agricultural sector as well as to hydroelectric producers. Increased sediment loads quickly reached the Pantanal Biome due to its proximity to the agricultural frontier (Bordas, 1996).

Burning wheat straw was a common means of crop sanitation, and soil was prepared using disk plows and heavy off-set disk harrows, leaving the soil surface uncovered and compacted at the subsurface, which reduced its water infiltration capacity, facilitating surface runoff. To make matters worse, the period of greatest rainfall erosivity in these regions coincided with the period of soil preparation, when soil was uncovered.

3. Expansion of no-till in Southern Brazil

With the goal of remediating the negative impacts of the wheat-soy crop sequence in Southern Brazil, in the 1970's the scientific community began studying the problem of water erosion. At that time, the focus of rural extension services was to promote the use of terraces to control erosion, although these frequently burst under the high volumes of runoff produced due to poor infiltration. To address this problem, extension agents would design massive terraces, up to 2 m high, which became known as "terraces Murunduns". The difficulty in controlling the surface runoff led to a conceptual change, with the focus gradually shifting away from structural controls (terracing) to improving the infiltration capacity of the soil through agronomic techniques such as conservation tillage, use of cover crops, crop rotation, and ending the practice of burning crop residues. Research confirmed that this shift in focus not only reduced erosion, but also improved soil quality (Derpsch et al. 1986; Roth et al.1988).

During the early 1990's, research into no-till and the availability of herbicides and no-till seeders (Casão Jr. et al., 2012) permitted steady expansion of no-till in the South and Cerrado regions (Bollinger et al., 2006). Between 1989 and 1996, the government of Parana state, with support of the World Bank, carried out a successful program of rural development, called Parana Rural, which spread conservation agriculture throughout the state. The program used watersheds as planning units and sought to organize and motivate farmers to adopt conservationist technologies, including minimum tillage, use of cover crops, crop rotation, and implantation of structures such as terraces and straightening of rural unpaved roads to collect and direct surface runoff. This experience became a reference worldwide for large-scale diffusion of conservation agriculture.

4. Soybean expansion in the Brazilian Cerrado and its environmental implications

Beginning in the 1980's, Brazilian scientists worked to develop soy cultivars adapted to tropical regions and to overcome the acidity and low phosphorus availability of soils in the Cerrado, permitting the expansion of soy into the central-west regions of Brazil (Camargo et al. 2017). Today, more than 20 million ha are under soy in the Cerrado region. This biome of 200 million ha is the largest tropical savanna in the new world and the second largest biome in Latin America (Furley, 1999). The expansion of soy into this region has provoked large-scale changes in land use, both in areas with native vegetation (Morton et al 2006) and especially in low-density pastureland (Brandao et al. 2005; Barona et al. 2010;Carneiro Filho & Costa, 2016; Strassburg's et al. 2014). Due to is complex biodiversity and wealth in water resources, the Cerrado plays important ecological services, including the headwaters of three major watersheds (the Paraná, São Francisco and Tocantins-Araguaia) and half of the recharge area for the Guarani aquifer (one of the largest in the world). As land use changes lead to important hydrological modifications (Tucci, 2007), it is reasonable to assume that the expansion of agriculture into the Cerrado would also provoke changes in the water cycle, which must be fully understood in order to minimize effects on strategic sectors, such as drinking water supply, energy production, and navigation. Studies on small areas (Oliveira et al, 2005) and in small (Dias et al, 2015) and large watersheds (Costa et al, 2003) show that the substitution of the Cerrado's native vegetation bycommercial crops reduces evapotranspiration, increases overland flow and flow rate in rivers. These hydrologic changes increase the sediment yield, with negative impacts to the hydroelectric sector and degradation of water quality, among other consequences. Another aspect is the deforestation of tropical rainforests as pastureland is pushed north into the

Amazon (Brandão et al., 2005). This, in turn, increases greenhouse gas emissions (Cerri et. al, 2007; Batlle-Bayer, et al., 2010).

5. Challenges and opportunities for sustainable agriculture

Given the availability of arable land, the consensus is that Brazil can accommodate part of the world's growing demand for foodstuffs. However, how will the country reconcile agricultural expansion while safeguarding the greatest area of tropical vegetation on the planet? The scientific community also agrees that the expansion of the cultivated area can occur without destroying native vegetation of either the Cerrado or the Amazon. Of a total of 170 million ha under grazing, 40 million of these are considered degraded, making these areas an obvious choice to accommodate agricultural expansion. According to Dias Filho (2014), applying existing technology to reclaim degraded areas reduces the need for extensive land areas for grazing. Current stocking density is, on average, just 0.6 animals/ha. Reclaiming even part of these areas would permit increased stocking density and reduce the area dedicated to grazing in Brazil, while the remaining area could be converted into cropland without further impingement on native vegetation (Strassburg et al., 2017). The current environmental crisis requires innovative solutions. A conservation agriculture program in Brazil that seeks to reclaim areas of degraded pasture for grain production and technological improvement of pastures is an intelligent response to this situation. However, a program such as this requires a broad consensus among the various stakeholders, and the adoption of transformative public policies. It is important to note that the foundation for a conservation program in Brazil already exists: the Brazilian Forest Code, which requires each agricultural property to maintain a minimum area of native vegetation (proportional to the cultivated area) and to preserve riparian zones. It is also crucial for researchers and extensionists to work together, whether at the level of the individual landowner or the watershed, to implant conservation technologies that have been tested and validated in that region. Strengthening projects and organizations that seek responsible soybean production, such as the Round Table on Responsible Soy and the Institute for Responsible Agribusiness (ARES) are also fundamental for this goal. The government's ABC Plan for Low Carbon Agriculture, which funds sustainable technologies to reduce greenhouse gas emissions by agriculture and meat producers, is another initiative which could be expanded. Its implementation is based on the elaboration of conservation plans for rural properties with farmers' participation and support from ANATER (the National Agency for Technical Assistance and Rural Extension) to train extensionists. The different sectors that benefit from a program like this, such as hydroelectric producers and grain and meat exporters, could contribute to a fund to finance the program. It is important to emphasize that farmers' participation in civil society organizations determines the efficacy of rural development projects, as was demonstrated in Parana, since they are much more likely to commit to a program when they participate in decision-making processes. Lastly, complementary policies such as territorial planning, improved law enforcement, monitoring and tenure security must be considered as appointed by Strassburg et al. (2014).

CONCLUSIONS

The availability and aptitude of natural resources for agriculture and meat production have given Brazil an important role to play in meeting future global demand for food. Nonetheless, the responsibility for preserving one of the greatest areas of tropical forest and savanna vegetation in the world obliges Brazil to steward these resources and utilize them to generate wealth, reduce poverty and promote a sustainable economy, safeguarding the environmental services provided by these ecosystems for the benefit of future generations. For this to take place, a program of conservation agriculture that incorporates the technological stock generated by the Brazilian scientific community, establishes appropriate public policies, and brings together the diverse stakeholders involvedmust be developed and implemented.

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SOCIO-ECONOMIC ISSUES OF TORRENTIAL FLOODING PREVENTION

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EXTENDED ABSTRACT

1. INTRODUCTION

According to the natural characteristics, Serbia is predisposed to erosion processes. But in the world and in Serbia, a large percentage of erosion processes are contributed by anthropogenic factors. The activity of man can be both negative and positive, depending on the degree of awareness of the importance of using natural resources on the principles of sustainability. Preventing the degradation of torrential floods and erosion processes contained in the sustainable management of land resources. The paper presents a model of sustainable management of land resources, adapted to the conditions of hilly areas of Serbia, which includes the planning of production on sloping terrain from the aspect of soil protection, then the needs of the population for certain localities particular production, and profitability of planned production.

2. NATURAL FACTORS OF EROSION PROCESSES AND TORRENTIAL FLOODS

Specific and variable terrain features, geological substrate, microclimate, soil and vegetation cover, represents a wide range of natural conditions and factors of torrential floods and erosion processes. Relief of Serbia provides favorable conditions for flash floods and water erosion. The fields/areas with a drop to 5% occupy about 30% of country area, while 70% of the land belongs to wavy, hilly and mountainous region that has great energy relief. In Serbia there are about 12,000 torrential flows The average annual sediment yield in Serbia is around 37 million m^3, while the specific production is 422 m^3/km^2. As a result of torrential rains and flooding, erosion and sediment are causing a number of problems in agriculture, forestry, water management, energy sector, transport and similar.

3. NEGATIVE HUMAN ACTIVITY

Negative human activity, in terms of use of land resources, is reduced to the disproportion arising between the agricultural population and land area or methods of agricultural business. Land farming on surfaces that by the slope and agro-morphological characteristics are not suitable for land farming, tillage down the slope, logging on steep slopes and unstable geological surface, pruning the trees for winter forage, caused a process of accelerated erosion (Zlatić, 1998).

4. POSITIVE HUMAN ACTIVITY

Positive activity is reduced to the sustainable management of land and water resources of the local population. There are numerous examples of the application of conservation land resources by farmers, especially in mountainous areas, since they are aware that their conservation of soil is of life importance. Some of these examples have also entered into the world database of conservation approaches and technologies through a program WOCAT - the World Overview of Conservation

Approaches and Technologies (Zlatić 2008). It’s certain that mentioned have influence on decreasing floods, due to reduced sediment transport from basin into the river-recipients.

5. SUSTAINABLE LAND MANAGEMENT – PREVENTION OF TORRENTIAL FLOODING AND EROSION

Production model from the aspect of sustainable land management based on an assessment of erosion processes according to "USLE" method is established on the base of the degree of erosion threat and slope on agricultural land (Zlatić, M., 1994). The degree of erosion vulnerability is defined as the ratio of actual and the tolerance of losses, which are fortified based on soil depth.
On the slopes and under certain specific vulnerability of land is projected such production that will bring down the losses of land below the tolerance limits, and it will be based on the needs of the population and the possible economic effects, or profits.

6. CONCLUSION

Prevention of degradation of torrential floods and erosion processes is contained in sustainable management of land resources. Model of sustainable management of land resources, that is adapted to the conditions of the hilly areas of Belgrade surrounding, had emphasized socio-economic and environmental impact of the local population (Zlatić, 1994). Highlights are: (1) the environmental aspect, and production planning on slopes in terms of conservation of soil resources, (2) the social aspect - the need of the population of some sites for a particular production, and (3) the economic aspect - profitable effects of planned production.

SOIL EROSION AT EUROPEAN SCALE: STATUS AND THE WAY FORWARD

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ABSTRAT

The implementation of RUSLE2015 for modelling soil loss by water erosion at European scale has introduced important aspects related to management practices. The policy measurements such as reduced tillage, crop residues, cover crops, grass margins, stone walls and contouring have been incorporated in the RUSLE2015 modelling platform. The recent policy interventions introduced in Good Agricultural Environmental Conditions of Common Agricultural Policy have reduced the rate of soil loss in the EU by an average of 9.5% overall, and by 20% for arable lands (NATURE, 526, 195). However, further economic and political action should rebrand the value of soil as part of ecosystem services, increase the income of rural land owners, involve young farmers and organize regional services for licensing land use changes (Land Degradation and Development, 27 (6): 1547-1551). RUSLE2015 is combining the future policy scenarios and land use changes introduced by predictions of LUISA Territorial Modelling Platform. Latest developments in RUSLE2015 allow also incorporating the climate change scenarios and the forthcoming intensification of rainfall in North and Central Europe contrary to mixed trends in Mediterranean basin. The rainfall erosivity predictions estimate a mean increase considerably in European Union by 2050. Recently, a module of CENTURY model was coupled with the RUSLE2015 for estimating the effect of erosion in current carbon balance in European agricultural lands (Global Change Biology, 22(5), 1976-1984; 2016). Finally, the monthly erosivity datasets (Science of the Total Environment, 579: 1298-1315) introduce a dynamic component in RUSLE2015 and it is a step towards spatio-temporal soil erosion mapping at continental scale.

KEYWORDS: Soil loss, land degradation, Water erosion, Environmental policies, management practices, RUSLE2015

INTRODUCTION

Erosion can be defined as the wearing away of the land surface by physical forces such as rainfall, flowing water, wind, ice, temperature change, gravity or other natural or anthropogenic agents that abrade, detach and remove soil or geological material from one point on the earth's surface to be deposited elsewhere. When used in the context of pressures on soil, erosion refers to accelerated loss of soil as a result of anthropogenic activity, in excess of accepted rates of natural soil formation (Huber et al, 2008).

The loss of soil leads to a decline in organic matter and nutrient content, the breakdown of soil structure, a reduction of the available soil water stored, which can lead to an enhanced risk of flooding and landslides in adjacent areas. Nutrient and carbon cycling can be significantly altered by mobilization and deposition of soil (Quinton et al., 2010), as eroded soil may lose 75-80 per cent of its carbon content, with consequent release of carbon to the atmosphere (Morgan, 2005). Soil erosion impacts strongly on the environment and has high economic costs; to mitigate these effects, soil and water conservation strategies are required.

Recent evaluations have shown that nearly all of Europe is affected by considerable soil erosion by water (Panagos et al., 2016a) and wind (Borrelli et al., 2016a). Soil erosion by water is one of the most widespread forms of soil degradation in the European Union (EU).During the period 2013-2016, in the Joint Research Centre(JRC) of the European Commission, the Research group in relation to “Soil Degradation by erosion” has developed a framework (Fig.1) for developing models to assess soil erosion by wind (Borrelli et al., 2014, Borrelli et al., 2015; Borrelli et al., 2016a), soil erosion by water (Panagos et al., 2015a, Panagos et al., 2016a) and specific focus on erosion in forestlands (Borrelli et al. , 2016b). In this framework, it should be added the coupling of erosion with soil organic carbon in a recent modelling application of CENTURY to erosion assessment of Europe (Lugato et al., 2016).

In the present article, we focus on soil erosion modelling by water at European scale. The On-site modelling and prediction of soil erosion by water has a long history with first studies published in international journals more than four decades ago (Li, 1974). During the last 3 decades an increasing number of erosion models have been developed. In a recent inventory, Karydas et al. (2014) identified 82 water-erosion models classified on different spatial/temporal scales with various levels of complexity. The most commonly usederosion model is the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) and its revised version (RUSLE) (Renard et al., 1997) which estimates long-term average annualsoil loss by sheet and rill erosion.

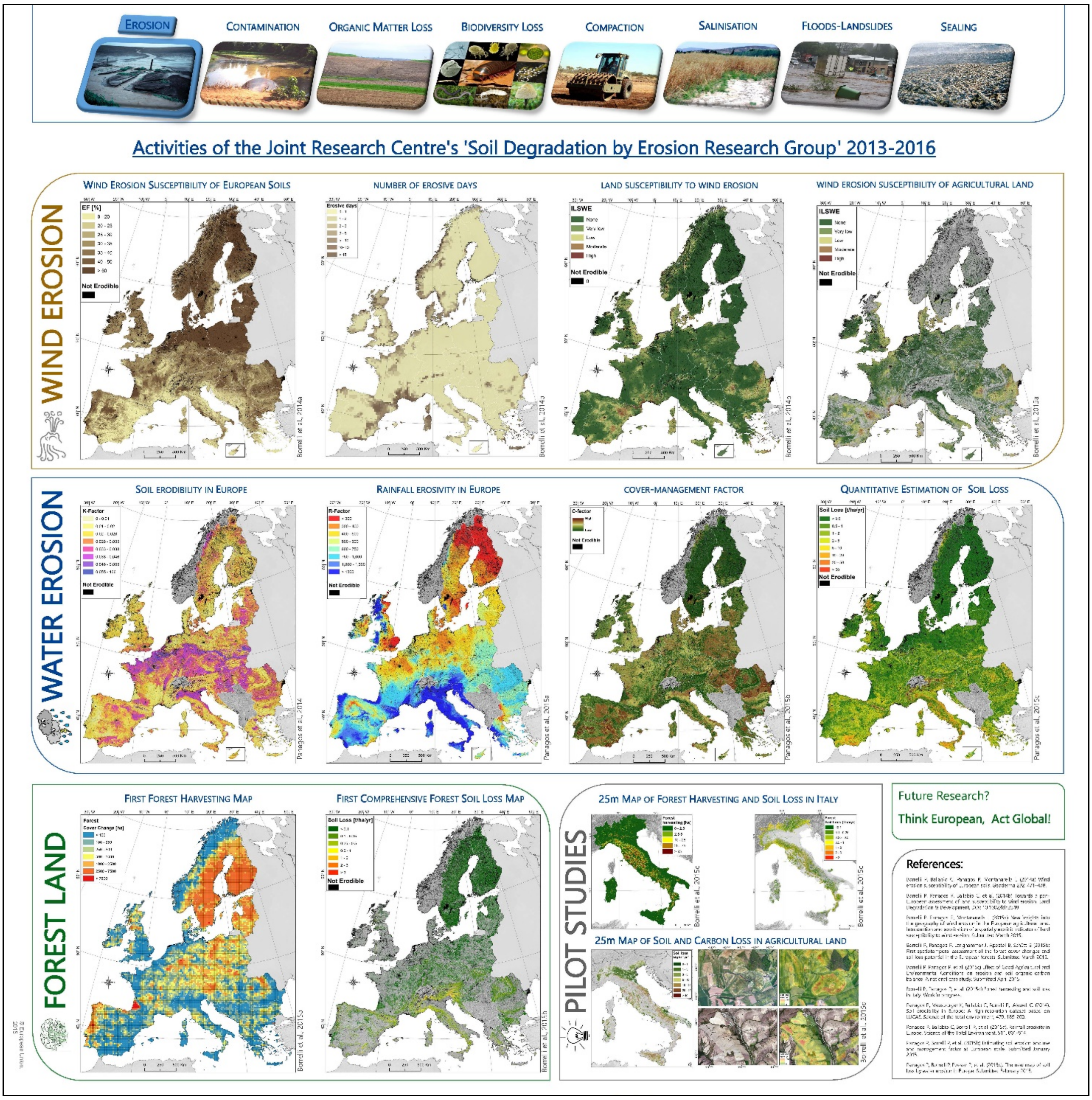

Fig. 1: Activities of Joint Research Centre' Research group in relation to "Soil Degradation by erosion" in the period 2013-2016.

The application of the Revised Universal Soil Loss Equation (RUSLE) on European scale has been discussed very controversially (Panagos et al., 2015a;Panagos et al., 2016a;Evans and Boardman, 2016;Fiener and Auerswald, 2016;Panagos et al., 2016b, ;Panagos et al., 2016c). Modelling in general and large-scale modelling specifically can per senot aim at an accurate prediction of point measurements, but tests our hypothesis on processunderstanding, relative spatial and temporal variations, scenario development and controlling factors(Oreskes et al., 1994). As such, our approach can be offered as a helpful tool to policy makers at pan-European scale. We are confident that the simple transparent structure of RUSLE as well as the discussionof the uncertainties of each modelling factor will help to supply objective guidance to policy makers.

RUSLE2015: model description

The main factors affecting the rates of soil erosion by water are precipitation, soil type, topography, land use and land management. As stated above, the most regularly used erosion model is the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) and its revised version (RUSLE) (Renard et al., 1997) which estimates long-term average annual soil loss by sheet and rill erosion. It should be noted that soil loss caused by (ephemeral) gully erosion is not predicted by RUSLE (Poesen et al., 2003). RUSLE is still the most frequently used model at large scales (Renschler and Harbor, 2002; Kinnell, 2010) as it can process data input for large regions, and provides a basis for carrying out scenario analysis and taking measures against erosion (Lu et al., 2003). In addition, a recent collection of soil loss data in Europe by the European Environmental Information and Observation Network (EIONET) found that all participating countries used USLE/RUSLE (Panagos et al., 2014b) to model soil loss.

The revised version of the RUSLE is an empirical model that calculates soil loss due to sheet and rill erosion. The model considers six main factors controlling soil erosion: the erosivity of the eroding agents (water), the erodibility of the soil (including stoniness), the slope steepness and the slope length of the land, the land cover and management, and the human practices designed to control erosion.

The model estimates erosion by means of an empirical equation:

$$Er = R\ K\ L\ S\ C\ P$$

Where:

Er = (annual) soil loss (t ha^{-1} yr^{-1}).

R = rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}).

K = soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}).

L = slope length factor (dimensionless).

S = slope factor (dimensionless).

C = cover management factor (dimensionless).

P = human practices aimed at erosion control (dimensionless).

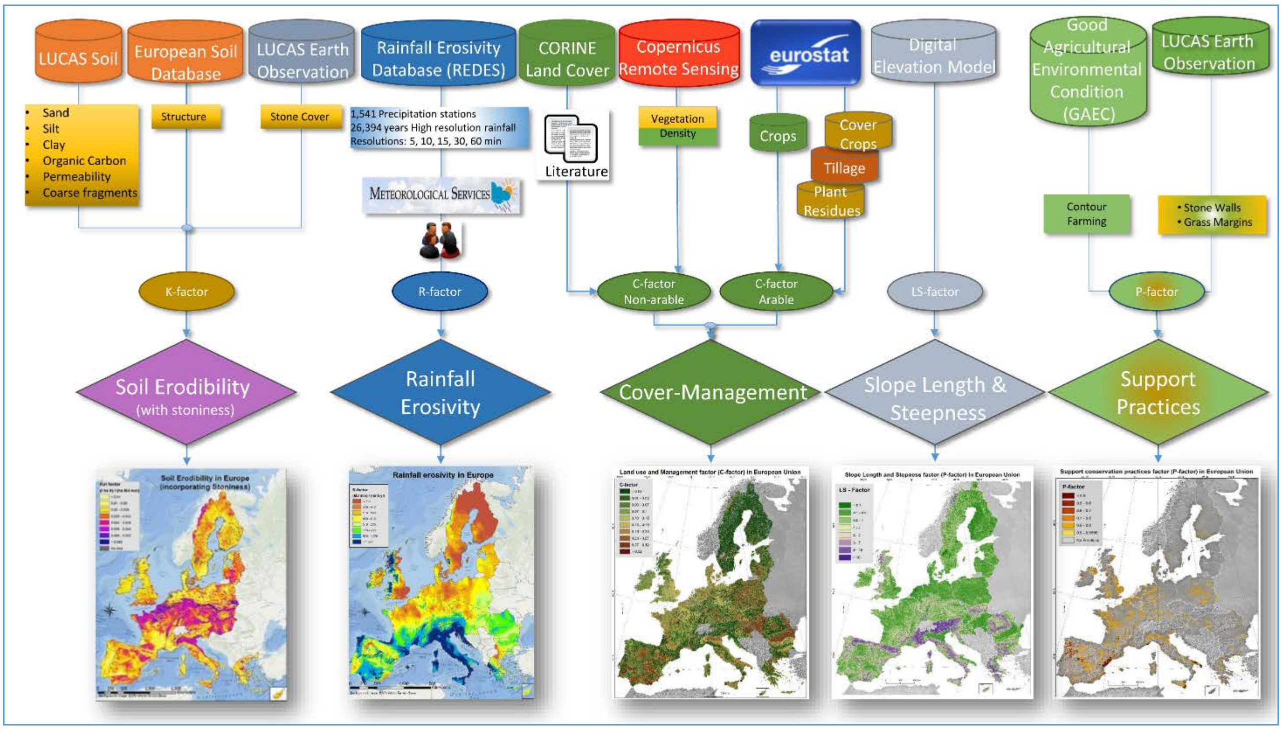

Fig.2: RUSLE2015 soil erosion model structure

The model equation is the same as the original one; however RUSLE2015 proposed important developments in each factor. The RUSLE2015 model introduces some improvements to each ofthe soil loss factors, adapting them to the latest state-of-the-art datacurrently available at the European scale. The main difference frompprevious studies that modelled soil loss at the European scale usingRUSLE (e.g. Van der Knijff et al., 2000; Bosco et al., 2015) is theimproved quality of input layers. Each input factor has beenestimated in a transparent way. The assessment procedures for thesoil erodibility factor (Panagos et al., 2014a), the rainfall erosivity(Panagos et al., 2015b), the cover-management factor (Panagos et al.,2015c), the topographic factor (Panagos et al., 2015d) and supportpractices factor (Panagos et al., 2015e) have recently been published,and the corresponding datasets are available from the European SoilData Centre (ESDAC).

The K-factor is estimated for the 20,000 field sampling pointsincluded in the Land Use/Cover Area frame (LUCAS) survey (Tothet al., 2013) and then interpolated with a Cubist regression modelusing spatial covariates such as remotely sensed data and terrainfeatures to produce a 500 m resolution K-factor map of Europe(Panagos et al., 2014a). The dataset was verified against 21 regional and national studies from 21 countries. Besides the inclusion of measured texture attributes, the K-factor model included as well the soil structure, permeability, coarsefragments andstone cover. The importance of soil erodibility layer has been recognized with at least 55 citations during the last 28 months (among them Carvalho-Santos et al, 2015; Guerra et al., 2016; Virto et al, 2015).

The R-factor is calculated based on highresolutiontemporal rainfall data (5, 10, 15, 30 and 60 min) collected from 1,541 well-distributed precipitation stations acrossEurope (Panagos et al., 2015b). The R-factor model combined the influence of precipitation duration, magnitude and intensity as it was based on high temporal resolution rainfall data covering long period. The data collection included 29,000 years of high Temporal resolution precipitation records and the variability of time series ranged between 7 and 56 Years with a Mean: 17.1 years; however, 75% of time series include the period 2000-2010. The data collection was done between March 2013 and June 2014 with very systematic way in participatory approach asalmost all Environmental & Meteorological Services from

EU Member States contributed to this. For the first time, Rainfall Erosivity Database on the European Scale (REDES) has been compiled and the Erosivity map of Europe is available to the public(Panagos et al., 2015b). According to scholar, the R-factor has been cited 45 times in 16 months (among them Bezak et al., 2015; Cervasco et al., 2015; Dominguez et al, 2015).

At European scale, C-factor was modelled with a newly developed model land use and management (LANDUM) model. LANDUM distinguishes between non-arablelands where it applies a combination of land-use class and vegetationdensity while in arable lands C-factor is based on crop compositionand land management practices (reduced/no tillage, cover cropsand plant residues) (Panagos et al., 2015c).For all non-arable lands, the C-factor is determined mainly by vegetation. Non-arable lands cover ca. 75% of the EU. The CORINE Land Cover database (CLC, 2012) was used to derive the different land-use classes of non-arable lands in Europe, and to assign a range of C-values for each class. Using biophysical attributes such as vegetation-coverage density (derived from remote-sensing datasets of the Copernicus Programme), a C-value was assigned to each pixel based on the combination of land-use class and vegetation density. For arable lands, the C-factor was estimated using crop statistics from the EU's Statistical service (Eurostat) and assigning the C-factor values per crop type based on an extensive literature. In addition, the effect of some management practices on soil loss rates was quantified at the European scale for the first time ever. The calculation of the C-factor of arable lands included soil-tillage practices, cover crops and plant residues. These datasets are available from Eurostat (Eurostat, 2014). Land management practices (reduced/no tillage, cover crops and plant residues) decrease the C-factor for Europe by an average of 19.1% in arable lands, with reduced tillage having the largest impact on soil loss rates due to the large areas of application. The successful application of LANDUM model and the C-factor results are also used in relevant studies (Miura et al.,2016; Napoli et al., 2016; Tzilivakis et al., 2016).

The LS-factor (Panagoset al., 2015d) is calculated using the recent Digital Elevation Model(DEM) at 25 m and applying the equations proposed by Desmetand Govers (1996). The use of high-resolution (25 m) Digital Elevation Model (DEM) for the whole European Union (Eurostat, 2014), resulted in an improved delineation of areas at risk of soil erosion as compared to lower-resolution datasets. The LS-factor data are available for download in 2 resolutions (25m, 100m) from the European Soil Data Centre (ESDAC). The high resolution of the LS-factor dataset allows the use of this dataset even at local and regional scale (Karamesouti et al. 2016; Vallebona et al., 2016).

The P-factor takes into account a) contourfarming implemented in EU agro-environmental policies, and theprotection against soil loss provided by (b) stone walls and (c) grass margins (Panagos et al., 2015e).The P-factor was estimated using the latest developments in the EU's Common Agricultural Policy (CAP) and applying the rules set for contour farming over a certain slope gradient derived from the Good Agricultural Environmental Condition (GAEC) requirements. The 270,000 field observations of the Land use/cover area frame statistical survey (LUCAS, 2012) were used to model the presence and density of stone walls and grass margins (van den Zanden et al., 2013). The mean P-value for the EU was estimated at 0.97, while in agricultural lands it was estimated at around 0.95. The use of stone walls and the impact of P-factor is much appreciated and used in the literature (Napoli et al., 2016; Agnoletti et al., 2015; Pacheco et al., 2015; Schiefer et al., 2016).

RESULTS

A map of soil loss in the European Union was produced usingRUSLE2015 at 100 m resolution (Fig. 3). This resolution depends onthe data availability of the input factors. The scale of 100 m pixelsize was selected as being the most appropriate because the C-factorlayer (at 100 m resolution) can be altered as a result of policyinterventions that affect land use.

Only 0.4% of EU land suffers from extreme erosion (> 50 t ha^{-1} yr^{-1}) and around 4.8% of EU land is subject to severe erosion (10 -50t ha^{-1} yr^{-1}). In a statistical analysis, 11% of total area subject to moderate/high erosion risk (>5 t ha^{-1} yr^{-1}) contributes to almost 70% of total soil loss which is estimated around 970 billion tones (Panagos et al., 2015f). This estimate is lower compared to the previous estimations that 16% of Europe's land area is affected by soil erosion (EEA, 2003) due to incorporation of CAP measurements. Focusing only in arable lands, around 12.7% of arable lands in EU is estimated to suffer from moderate to high erosion (>5 t ha^{-1} yr^{-1}). This equates to an area of 140,373 km^2 (More than entire surface area of Greece). Using conservative estimates of wheat yields of 3 tonnes / ha in Europe (Gardi et al., 2015) and a market price of €300 / tonne of wheat), in an area of arable land affected by moderate to severe soil erosion, agricultural production in the European Union of €12.633 million could be under threat – if the economic value is placed on the loss of soil carbon (currently CO2-credits are around €20 / tonne), the figure would be even higher.

Considering the average of soil water erosion rate by country, several European countries (most of them in North Europe) appear not to be significantly affected by notable soil erosion susceptibility when compared to a 'continental mean' of around 2.46 t ha^{-1} yr^{-1}. Spain, Greece, Italy, Cyprus, Croatia, Slovenia, Austria and Romania have average rates higher than the mean European one. However, such values can be misleading as they mask the fact that erosion rates in many areas can be much higher, even for those countries that have a low mean rate of erosion. The converse is also true for countries with high values. On the other hand, some countries, mainly in the southern part of Europe, are clearly characterised as being particularly susceptible to erosion.

The major sources of uncertainty for the soil erosion map of Europe are found in some highly erosion-prone CORINE land-cover classes (e.g. sparsely vegetated areas – class: 3.3.3) that demonstrate high variability between Mediterranean regions (bad-lands) and northern Europe (mixed vegetation with rocks). For example some of the areas in Northern Scotland are classified as sparsely vegetated areas while probably those areas are not prone to soil erosion (mixed vegetation with rocks).

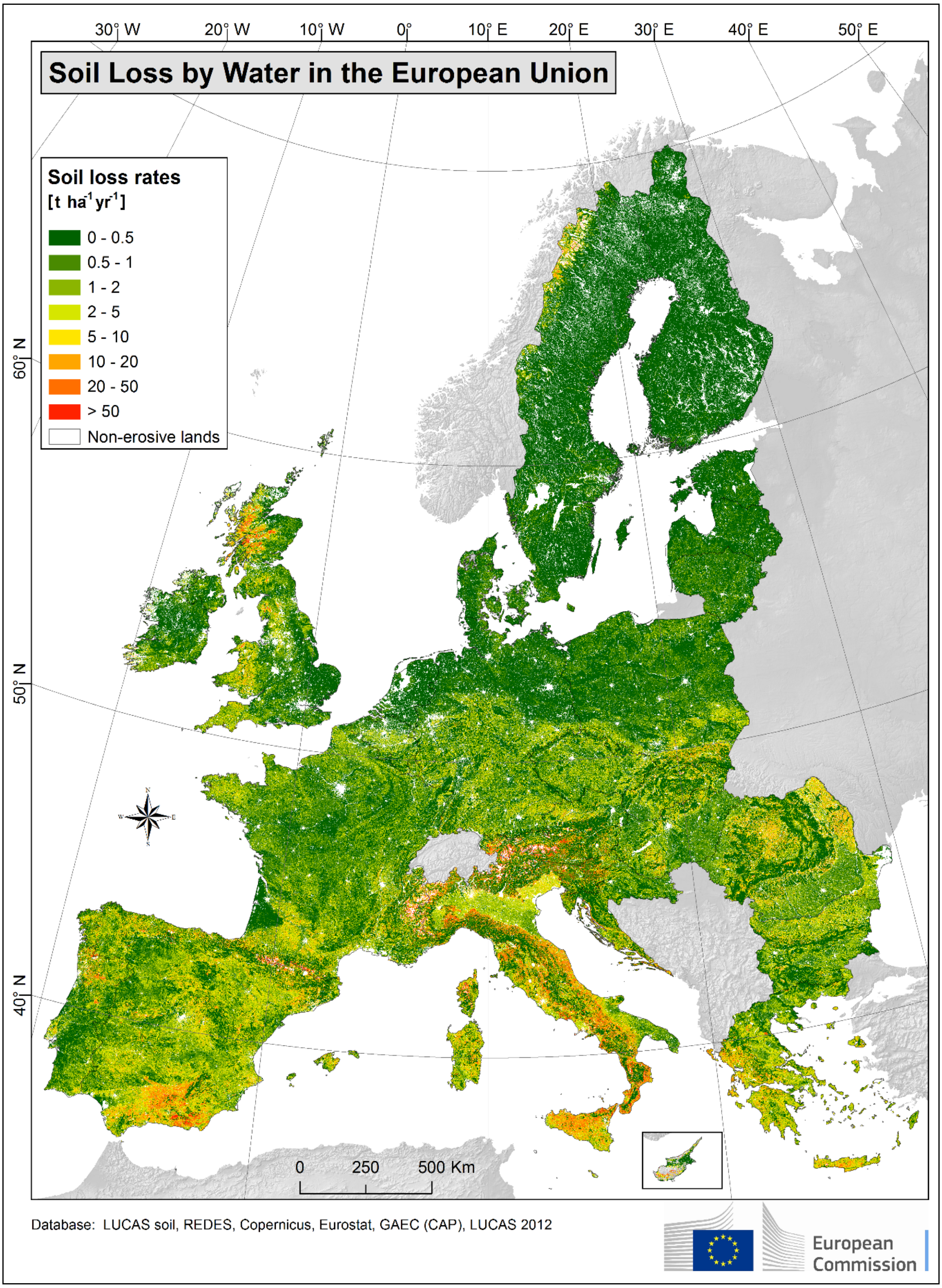

Fig. 3. Soil water erosion in t ha^{-1} yr^{-1}(cell size: 100 m) across erosive land surfaces in EU. Non-erosive lands: surfaces that are not prone to soil erosion, such as urban areas, bare rocks, glaciers, wetlands, lakes, rivers, inland waters and marine waters.

Climate change – Land use and Policy Scenarios

The RUSLE2015 model structure can simulate scenarios of landmanagement, land use change, and climate change. As such, themodel becomes a useful tool for policy makers to both assess pastperformance and estimate soil loss changes based on futurescenarios.

RUSLE2015 quantifies the impact of management and conservation practices (mainly introduced in GAEC) such as reduced tillage, cover crops, plant residues, maintenance of stone walls, contour farming and grass margins. The implementation of GAEC in the agricultural lands of Member States reduced soil loss rates. Since no statistical data were available about the conservation and management practices before the GAEC implementation (2003), we make the hypothesis that those management practices were previously not applied or were only applied to a very limited extent. If no GAEC requirements had been applied in the EU, the mean soil loss rate in the European Union study area (agricultural lands, forests and semi-natural areas) would have been 2.71 t ha^{-1} yr^{-1}. Compared to the current estimated mean annual rate of 2.46 t ha^{-1} yr^{-1}, this implies that overall soil loss in the EU was reduced by 9.5% during the past decade due to policy measurements (GAEC). Among the management practices, the reduced tillage had the greatest impact in reducing soil loss in arable lands (Panagos et al., 2015c). The relatively high frequency of grass margins and the high impact of stone walls in reducing soil loss have contributed in 3% overall soil erosion reduction (Panagos et al, 2015e). Cover crops, plant residues and contouring have a limited contribution to soil erosion reduction due to their limited application.

Soil erosion trends resulting from changes in land cover, rainfall erosivity or management practices. Past assessments showed that land cover changes identified by CORINE Land cover data (2000, 2006) are very limited. The results do not show any particular trend in the erosion of soil by water at a time interval of 6 years (2000-2006). RUSLE2015 model allows to incorporate land use and climate change scenarios.

Using the pan-European Land Use Modelling Platform (LUISA) (Lavalle et al., 2013), it is possible to predict that soil loss will decrease mainly due to increase of forest areas in the expense of semi-natural ones. We selected the projections of land use change for the year2050 based on the pan-European Land Use Modelling Platform(LUISA) (Lavalle et al., 2013; Baranzelli et al., 2014). LUISA translates policy scenarios intoland-use changes such as afforestation and deforestation, pressureson natural areas, abandonment of productive agriculturalareas, and urbanisation. According to LUISA, all agricultural landuses will be reduced by 2050 (croplands will decrease by 1.2%,permanent crops by 0.2% and pastures by 0.6%), and semi-naturalareas will also decrease by 1%. Urban areas will increase by 0.7%and forest areas by 2.2%. Forest lands, which are the least erosionprone(with mean annual soil loss of 0.065 t/ha), will replaceerosion-sensitive land uses (permanent crops, arable, pastures andsemi-natural). In total soil loss terms, the future land use changesprojected by LUISA will result in a 5.8% reduction in soil loss.However, LUISA should take into consideration the imminentthreat of peak phosphorous levels, with the only noteworthy Presources left in the Western Sahara and Morocco after 2013 (Elserand Bennett, 2011). Given this threat, the EU Member States willmost likely start to increase their area of arable land considerablyin the near future.

The rainfall erosivity prediction for 2050 is under review as this much depends on the decrease of precipitation based on WorldClim's future scenarios for Europe (Hijmans et al., 2005) combined with an increased intensity of events. The mean rainfall erosivity for the European Union and Switzerland is projected to be around 857 MJ mm ha^{-1} h^{-1} yr^{-1} by 2050 (Panagos et al. 2017), showing a relative increase of 18% compared to baseline data (2010) when the mean rainfall erosivity is estimated to 722 MJ mm ha^{-1} h^{-1} yr^{-1}. The changes are heterogeneous in the European continent with regard to the future projections both in the time of the year (increase of erosivity during summer months) and in space. The pan-European projection of future rainfall erosivity takes into account the high uncertainty of the climatic models.

Finally, the new CAP 2014-2020 is certainly promoting the increase of grass margins due to application of the ecological focus areas and the maintenance of stone walls which reduce soil loss. A possible duplication of areas with e grass margins and the application of contour farming in arable lands having slopes more than 5%, will further reduce soil erosion by 5%.A second contradictory example is the EU Biofuels Directive (BFD) which may pushfor the transformation of cereal croplands (C-factor: 0.20) intoenergy croplands such as sugar beet, sunflowers and maize (Cfactor:0.38), and will also result in reducing crop residues.Changing 10% of cereals to energy crops as a result of the BFDrequirements (Frondel and Peters, 2007) would lead to an increasein the C-factor of 3.8% in arable lands and a 2.2% increase in meansoil loss rates.

The soil organic carbon (SOC) cycle is affected by erosion, since large quantities of sediments and SOC are moved and re-deposited downhill especially in agricultural areas. RUSLE2015 has improved the scientific knowledge of one component of the global carbon fluxes, which has to date often been neglected in the past due to lack of data. To investigate this issue, we coupled soil erosion into a biogeochemistry model CENTURY, running at 1 km^2 resolution across the agricultural soils of the European Union (EU). Based on data-driven assumptions, the simulation took into account also soil deposition within grid cells and the potential C export to riverine systems, in a way to be conservative in a mass balance. We estimated that 143 of 187 Mha have C erosion rates <0.05 Mg C ha^{-1} yr^{-1}, although some hot-spot areas showed eroded SOC>0.45 Mg C ha^{-1} yr^{-1}. In comparison with a baseline without erosion, the model suggested an erosion-induced sink of atmospheric C consistent with previous empirical-based studies. Integrating all C fluxes for the EU agricultural soils, we estimated a net C loss or gain of −2.28 and +0.79 Tg yr^{-1} of CO2eq, respectively, depending on the value for the short-term enhancement of soil C mineralization due to soil disruption and displacement/transport with erosion (Lugato et al., 2016). We concluded that erosion fluxes were in the same order of current carbon gains from improved management. Even if erosion could potentially induce a sink for atmospheric CO2, strong agricultural policies are needed to prevent or reduce soil erosion, in order to maintain soil health and productivity.

Finally, RUSLE2015 incorporates the dynamic component of monthly and seasonal erosivity (Ballabio et al., 2017) and shows a high intra-annual variability of erosivity across Europe. Summer is the period with the highest R-factor and it is remarkable that around 55% of total rainfall erosivity in Europe takes place within only 4 months (June–September). However, the intra-annual distribution of erosivity and the concentration of extreme events have a high spatial variability in Europe. The spatio-temporal rainfall erosivity analysis at European scale is a first step towards developing dynamic (monthly, seasonal) maps of soil loss by water erosion.

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SOIL-WATER-CLIMATE MANAGEMENT AND CONSERVATION SYSTEMS IN ANCIENT CULTURES OF TROPICAL LATIN AMERICA

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EXTENDED ABSTRACT

INTRODUCTION

By parallel and independent paths, since ancient timesthe indigenous people of the American continent, have formed very complex and advanced cultures. These ancestral people developed organizational forms that allowed the flourishing of sophisticated civilizations, leading not only arts, architecture, astronomy and the knowledge of nature, but also the surprising development of agricultural engineering, using highly efficient technologies for sustainable land uses. Since the beginnings of the American settlement (35,000 to 40,000 years ago), the primitive inhabitants of this continent had to adapt to a diversity and often adverse natural conditions, to settle progressively on lands withdifferenttypes and limitation degrees. Starting from the knowledge accumulated during centuries, they gradually succeeded in developing highly sustainable land use systems, with ingenious techniques for soil, water and climate management. This allowed themto domesticate a great variety of crops that currently represents a fundamental part of the mankind diet, such as corn, beans, potatoes, tomatoes, squash, amaranth and quinoa,as well asa variety of grains, fruits, textiles, roots, tubers and other items widely distributed in the five continents. This domestication was achieved in parallel with the use of highly efficient crop management systems,ecologically adjusted to tropical conditions: multi-strata polycultures, short-cycle crop associations, cut and burning or cut and cover systems, and improved fallow systems, among others. The dissemination of such crop management systems waspossible by the development of ingenious infrastructure works that allowed the progressive use of extensive areas with natural limitations. This lecture will show examples of diverse technologies developed by ancestral people of tropical America to use successfully marginal lands with physical restrictions, with emphasis on works and agricultural engineering systems adapted to particular climate, topography, soil and drainage conditions.

SOIL-WATER-CLIMATE MANAGEMENT SYSTEMS FOR SUSTAINABLE LAND USE, THROUGH RELEVANT INFRASTRUCTURE WORKS

These works are grouped according to the main land limitation to be solved, showing examples for different tropical Latin America regions:

1. Lands with water excess limitations:

Hydraulic engineering works for lands affected by seasonal or permanentwater excesses, in different climatic regions.

1.1 Seasonal water excess:

Includes infrastructure works, depending on the climate conditions, for agricultureuse of poorly drained landsand water-saturated soils during part of the year:

1.1.1 In tropical coldclimate regions: raised fields or *waru-waru* (the *Altiplano*: Peru and Bolivian high-plain)

These hydraulic works are soil excavations with multiple forms and dimensions (generally elongated), following the landscape runoff patterns. They form an intricate network of raised beds and channels that allow cropping in flood areas, improving drainage in the rainy season or the uptake and water supply for sub-irrigation of the raised fields in the dry season. In addition, this ancient technology helps control the frost effects on crops during the coldest weather season, taking advantage of the caloric properties of the surrounding water.

1.1.2 In tropical warmclimateregions: high ridges (*El Beni* region in the Bolivian Amazon, Momposina depression in Colombia and low plains in Venezuela)

The high ridges are elongated mounds or embankments to allow cropping of extensive areas of seasonally flooded plains, improving drainage during the maximum rainfall season. They are combined with sustainable soil-water-crops systems and aquatic wildlife exploitation, to conform a more floodplain integrated management system.

1.2. Permanent water excess, in tropical cold climate regions: *chinampas* (Mexico):

These are the so called "floating orchards" or "floating gardens" that were gradually expanding in the Texcoco Lake (at the present time Mexico City). The system consists of small areas for grains, vegetables and flowers cultivation into the lake. Construction of these small islands is made by frameworks of large trunks tied with ropes of native fibers, completed and reinforced with a rectangular framework of branches and thinner trunks. Subsequently they are completely filled with muddy material extracted from the bottom of the lake. Finally, willow plants are sowing in the border, in order to provide more stability to the infrastructure.

2. Lands with water shortages:

Works designed to capture and store rainwater or runoff,to be used in crop production, under different climate conditions:

2.1 Water harvesting in tropical cold climate conditions: *qochas* (The *Altiplano*: Peru and Bolivian high-plain)

They are systems of temporary lagoons interconnected as a network, for storing and supplying water to crops in dry and cold weather areas, taking advantage of the caloric properties of water to control the frost effects on crops. Seeding is done at the circular edge of the lagoons, performing such plantings in temporal sequences as the wet edge is concentrically reduced by the progressive decrease of the volume of the stored water in the lagoon.

2.2 Water harvesting in different tropical climate regions: *amunas*, *puquios* and *mahamaes* (Peruvian sierra and littoral).

These structures include a variety of techniques to collect runoff or groundwater. The *amunas* are long open channels or ditches across the sloping lands with fractured rock underneath, allowing water percolation from the highland ditches to water springs located in the middle or at the lower part of the hillside(used mainly for irrigation). The *puquios* and *mahamaes i*nclude filtering galleries, sunken fields or built-up springs for groundwater collection in tropical dry and warm climate, taking advantage of underground water flows from contiguous highlands.

3. Lands with topographic limitations: terraces, platforms or *andenes* (Central Andes: Ecuador, Peru and Bolivia)

Andenes or terraces are constructed to modify the original terrain slope for sustainable uses of steeplands, in tropical cold climate highland regions. In addition to transforming sloping lands into horizontal terrains to facilitate agriculture (irrigation included), terraces also modify micro-climatic conditions by improving the uptake of radiant energy during the day and reducing radiative losses at night. They also allow air drafts mixing along the terraced sloping terrains, contributing to reduce the adverse effects of frost on crops.

CONCLUSIONS

It is highlighted that in tropical Latin America there are still examples of successful pre-Hispanic soil-water-climate management systems, which are part of the world's agricultural heritage. These microcosms of ancestral agriculture are promising models that promote ecological stability and self-dependency in a context of sustained productivity, offering valid alternatives that deserve to be studied, revalued and promoted as an important contribution to the challenges of contemporary agriculture, immersed each day more in the complex global change

SOIL EROSION, SOIL QUALITY AND CROP YIELD IN THE CHINESE MOLLISOL REGION

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INTRODUCTION

With the growth of economic and population in the world, agricultural sustainability has been considered as crucial for meeting food demand and economic development in developing countries. However, the increase rate at which corn, soybean and rice yields has slowed since 1995 (FAOSTAT, 2007). The contiguous areas of northeast China with black soil, chernozem and meadow soil are called the north-eastern black soil region, including three provinces (Heilongjiang, Jilin, and Liaoning) and the eastern part of the Inner Mongolian autonomous region, which is considered important for Chinese crop production (Xu et al., 2010), which provides one fourth to one third food supply in China. However, since large-scale cultivation of the region began in the 1950s, severe soil erosion has occurred and the thickness of Mollisol soils has decreased from 60–70 cm in the 1950s to 20–30 cm at present (Fan et al., 2005). In some places the loess parent material of Mollisol soil has been exposed to the surface, which reduces soil productivity (Wang et al., 2009). Consequently, the above changes further threaten the food security in China (Liu et al., 2010). Therefore, it is important to quantify how soil erosion affect soil quality and crop yield. The specific aims of this study were to analyze the impacts of soil erosion/deposition on soil characteristics and to select retainable soil quality assessment indicators, to discuss the corresponding relations of corn yield to soil quality and soil erosion/deposition rate, to fit equations among crop yield, soil quality, and soil erosion.

METHODS

The study was conducted in the Binzhouhe catchment, located in Bin County in the north of Heilongjiang Province in China. The catchment has a temperate continental monsoon climate, which is hot and rainy in summer and cold and arid in winter. The mean annual precipitation is 548.5 mm, and 80% of which is received from June to September. The mean monthly temperature is 3.9°C. The dominant soil association in this study catchment is classified as Mollisol based on USDA Taxonomy. As regards soil use, the field devotes to agriculture with corn [Zea mays L.] being the dominant crop for several decades and irrigation is not used in the study area. According to field investigation, 168 soil samples in the collected based on a 200*200 m grid of the research watershed. Meanwhile, four years of corn yields and 15 soil indicators, including physical, chemical and biological were measured. Moreover, ^{137}Cs tracing technique was used to estimate soil erosion/deposition rate.

RESULTS

1) The soil erosion and deposition rates on the watershed ranged from -7122 to 5471 t km^{-2} yr^{-1}. Soil erosion was dominated in the upstream area, erosion and deposition was coexisted in the midstream region, and soil deposition was dominated in the downstream area. At hillslope scale, severe soil erosion occurred at the midslope, and deposition occurred at the footslope with 20-30 cm depth.

2) Soil erosion significantly influenced soil characteristics, especially affected soil organic matter, total nitrogen, available phosphorus, urease, alkaline phosphatase, microbial biomass nitrogen. According to correlation analysis and principal component analysis, the thickness of mollic soil layer, mean weight diameter of soil aggregate, soil organic matter, soil total nitrogen, pH, soil sucrase and soil microbial biomass nitrogen were chose as the minimum data set indices for soil quality

evaluation. The mean soil quality index (SQI) at the catchment and the hillslope scales were 0.453 and 0.471, respectively. And the spatial distribution of soil quality index at both scales of watershed and sloping were reverse with soil erosion rate.

3) The spatial distribution characteristic of crop yield at the watershed was as follows: downstream > midstream > upstream. At hillslope scale, the minimum corn yield occurred at the midslope, which was the corresponding to soil quality and while the reverse with the soil erosion rate.

4) Corn yield declined as mollic thickness decreased. When the mollic was completely lost, the corn yield reduced by 24.2%. Particularly, within 20 cm mollic thickness, reductions in the corn yield ranged from 46.2 to120.8 kg ha^{-1} when 1 cm of mollic thickness was lost. In the extreme rain years, deposition depth at the slope foot greatly influenced corn yield, which could reduce 11.0%-31.7% of corn production, depending the deposition thickness.

5) Corn yield had a highly significant positive correlation with soil quality index and negative correlation with soil erosion rate. The equations among crop yield, soil quality, and soil erosion rate were established and validated.

CONCLUSIONS

This study analyzed the impacts of soil erosion and deposition on soil quality and crop yield. The results showed whether the spatial distribution of soil quality index or and crop yield was reverse with soil erosion rate. Corn yield declined as mollic thickness decreased. When the mollic thickness was less than 20 cm, corn yield was reduced by 8.2%-24.2% with decreasing mollic thickness. Particularly, within 20 cm mollic thickness, reductions in the corn yield ranged from 46.2 to120.8 kg ha-1 when 1 cm of mollic thickness was lost. In the extreme rain years, deposition depth at the slope foot greatly influenced corn yield, which could reduce 11.0%-31.7% of corn production. Corn yield had a highly significant positive correlation with soil quality and negative correlation with soil erosion rate. The equations among crop yield, soil quality, and soil erosion rate were established and validated. It indicated that preventing soil erosion in the Chinese Mollisol region is important for feeding Chinese people.

Acknowledgments

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SOIL CONSERVATION RELATED TO ORGANIC OR MINERAL FERTILISATION. NOTES ABOUT SOME CASES OF SOIL AND FERTILISATION MANAGEMENT IN CATALONIA

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EXTENDED ABSTRACT

A.- Some notes about fertilizers and soil fertility

Soil conservation is, among others practices, related to the aim of keeping soils with a correct fertility status. However, fertility definition is often imprecise. We accept it refers to “the status of soil related to its ability to supply essential elements for plant growth” (Foth and Ellis, 1988). Other authors refer explicitly the diffuse notion of the concept “fertility” (Sebillotte, 1989).

The soil fertility approach is quite different depending upon the user’s point of view. Farmer is mostly interested in “high production”, environmentalist is interested “is keeping soil quality” and food company is perhaps interested in “quality of the product” and “traceability”. This different practical focus implies different criteria in soil management practices. Recently, FAO (2017) defines soil fertility as the “capacity to receive, store and trasnmit energy tu support plant growth referering to soil nutrient availability and its relation to external soil nutrient application”.

The challenge of maintaining a proper status of soil nutrients in the root zone is essential to maintain soil quality in the perspective of future making demands of food, environmental quantity, climatic change horizon, etc.

Fertilizers are supposed to restore soil fertility by means of supplying the nutrients exported by the cultures. In figure 1 (Pellerin et al, 2014) we observe the typical yield response to nutrient availability (Phosphorus).

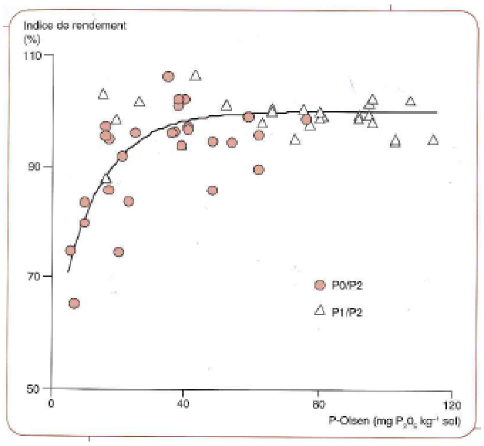

Figure 1.- Plant response to soil P avalilability (Source: Pellerin et al)

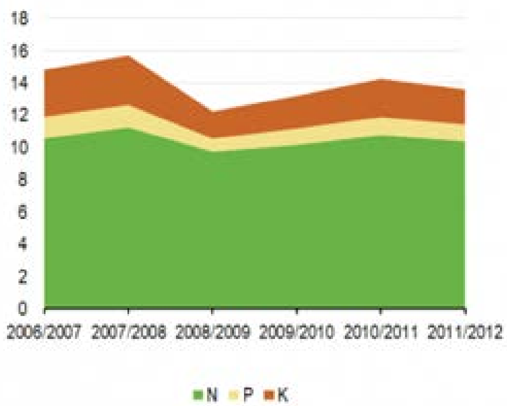

Figure 2.- Fertilizer nutrient consumption in the UE-27. (Source: Eurostat)

In Figure 1 two facts are evident: the increase of production upon increasing of P content in the soil, and a plateau tendency when P levels reach a sufficiency level. We could introduce more complexity adding the interaction with soil properties, crop management and environment. Although the precise response of soil-plant system is not known in a wide range of soils, the routine application of fertilizers in essential for maintaining crop production (Figure 2).

Considering the UE-27 data the total consumption of mineral fertilizers in 2012 year was of about 17 M of tones for N, P and K nutrients (Eurostat, 2017). Fertilizer application is a relevant an important agricultural activity in terms of inputs to the crops and associated costs. Fertilizers costs in new irrigated land in Catalonia (Lloveras and Cabasés, 2014) range from 18% to 34% of routine crop costs, considering extensive crops like wheat, barley, maize or alfalfa. Therefore fertilizers impact heavily in the cost of crop production, although the use of mechanisms of soil fertility control remains scarce or irregular.

Are fertilizers impacting soil quality? FAO (2011) recognizes several factors which affect negatively soil quality: erosion, salinization, acidification, contamination and soil nutrient depletion. UE rapports (Jones et al, 2012) include the following soil threads: loss of organic matter, erosion, compaction, soil sealing, salinization, acidification, loss of biodiversity, desertification, landslides and soil contamination. Excess of nutrients and fertilizer impurities are related to soil contamination and a misuse of either mineral or organic fertilization. The most insidious and specific problems of over-fertilization (mineral and organic) in Europe are the excess nitrogen and phosphorus contents as well as heavy metal accumulation in some areas.

Farmer's cases reveal frequent situations of non-balanced soil fertilizer application. With consequences in the degradation of soil fertility status. The question concerned is the possibility of reaching a good equilibrium between plants nutritional requirements and sound fertilizer application.

B.- Cases of fertilizer and soil management

Three cases related to soil management and fertilizer practices related to soil nutrient status are considered. They are situated in the Catalonia in the Ebro valley and they are concerned with the use of either mineral or organic fertilizers or soil erosion.

The agrosystem in Catalonia is often mentioned as a paradigm on the problems associated with high production of organic by-products, normally considered organic fertilizers. These products of organic origin are presented in a wide variety of formats and mixtures: sludge, slurry, composted organic matter, by-products of agro-food industry, etc. Concurrence is quite evident especially in the regions where intensive livestock is developed. Organic fertilizer from intensive livestock applied to soil is either a fertilizer or a contaminant considering the soil status and crop nutrient requirement status. The use of well-known criteria of soils status, crop requirement and organic fertilizer composition data and knowledge of soil status should allow a sound use of fertilizers.

In Table 1 three cases of fertilizer management are compared in Catalonia, in the Ebro valley area. They are situated in a semi-arid environment, with irrigation and intensive crop management.

Table 1. - Soil analytical results obtained in similar soils with different fertilizer management

Soil Ref.	pH	CE 1.5 (dos/m)	OM (%) (WB)	Carb. (%)	Texture class	N- NO_3 ppm	P (Olsen) ppm	K (am. ac.) ppm
A	8,2	0,21	2,31	22	loam	11	7	105
B	8,2	0,29	2,94	21	loam	41	65	509
C	8,5	0,76	0,67	39	Silty-clay-loam	3	<5	82

Both soils A and B soils are irrigated, moderately deep, well drained, and quite productive (Calcixerept tipic, SSS 1999), (Haplic calcisol, WRB 2006). They support an alfalfa-maize-wheat crop rotation. The status of nutrients is quite different in both cases.

Farmer cultivating soil A is only using mineral fertilizers; farmer cultivating soil B is applying slurry organic residues and some mineral fertilizers as well.

Nutritional mineral contents in soil A are quite low. A reinforcement of soil fertilizers applications could be envisaged in order to improve nutrient plants supply.

Soil B analytical results reflect a situation of overloading nutritional status; farmer's balance is not accurate enough and the result in the soil nutrients status reflects an unbalanced situation. A reduced fertilization supply ought to be considered in order to reduce the nutritional contents.

Nutrients status is not well adjusted neither in case A or in case B. Case A shows an example of a highly potentially productive soil but nutrient status shows a deficit status concerning P and K elements. The study of the agronomic rotation shows a deficit in fertilizer supply according to potential and yield average in recent years. The soil analyses reveals this situation, whereas a reinforcement of fertilization should be considered.

Case B is a quite frequent example of over-fertilisation. Soil is receiving a routine organic fertilization which is increasing the nutrient status content in the soil as a result of an unbalanced nutrient supply. Soil quality is in risk due to over nutrient supply and could be associated to nutrient transfer to water percolation in this area with a flow irrigation scheme (other soil quality problems are also possible (Puigpinós et al, 2016).

Both situations are quite frequent in intensive irrigated soils of the Ebro valley if a sound scheme of soil nutrient content in the soil is not carried out.

A third example (soil C) is given to emphasize the soil degradation due to soil in a field parcel affected by laminar and rill erosion.

This C soil if a soil which has suffered a process of severe erosion due to an unappropriated soil management associated to levelling for new irrigation implementation. Dry stone walls were eliminated for land and regrouping of parcel plots. Original, dry stone walls fragmented slope was transformed to its original natural physiography. Heavy rains in periods of uncovered surface did the rest and the properties original Ap horizon properties were downloaded. Soil analysis reflects a poorer soil nutrient status, although the most significant analytical values refer to low soil organic matter status. Increase value of electrical conductivity and pH reflect the Ap degradation. Status of soil nutrients is low while the problem is less relevant compared with erosion problems. Soil erosion control measures should be considered in this parcel. These tasks imply a complex focus in redefinition of soil management.

Conclusions and some proposals

An improved use of fertilizers might be reached by means of extension programs, administrative control of fertilizer uses, soil testing, model prediction, adjustment of crop requirements, crop rotation, catch crops, precision agriculture, landscape management, etc. The scope of possible actions to decrease the impact of fertilizers malfunctions is diverse. Furthermore, new technologies related to water and nutrient management (Mueller et al, 2012), nitrate leaching control (Quemada et al, 2013), remote or aircraft imagery with high resolution, contribute to a sustainable focus in a correct fertility management in the complete set of parcels to be managed, or even within the parcel unit.

To sum up:

- A sound crop nutrition action should start in the knowledge of soils in the field (soil observation and soil sampling routines (Figures 3)

- An approach by means of systematic evaluation of soil nutrient status and other properties using soil analysis (Figure 4) is an effective practice (Aran and Porta, 2016)

- An evaluation of field and laboratory data with the use of stablished soil references (Figure 5) allows the decision for measures to be implemented

- Wherever possible the completion of data with the use of additional sensors. It is the case of moisture sensors (Figure 6) or soil spectral variability in the field (Figure 7). This information integrates the data with the essential soil moisture evolution and in-field soil variability

Figure 3.- Soil observation in the field.

Figure 4.- Soil analysis, an useful tool for soil diagnostics (Source: Eurofins)

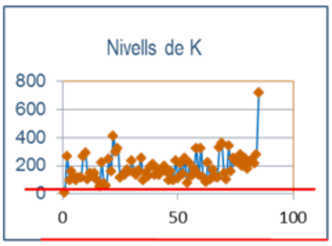

Figure 5.- Data evaluation (Source: Eurofins) sampling)

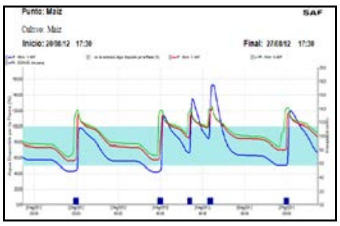

Figure 6.- Soil moisture status (Soure: SAF

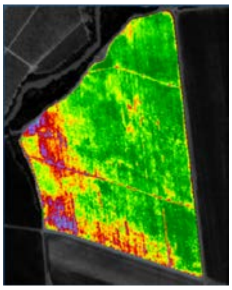

Figure 8.- Parcel variabity obtained by means of treated satellity imagery (Source Airbus DS, Terranis)

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NEW ADVANCES IN THE EVALUATION OF SALT-AFFECTED SOILS UNDER DRYLAND AND IRRIGATED CONDITIONS

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EXTENDED ABSTRACT

INTRODUCTION

Salinization and sodification are the main processes of soil degradation affecting irrigated lands. The growing development of irrigated agriculture is necessary for the sustainable production of the food required by the increasing World´s population. Such development is limited by the increasing scarcity and low quality of the available water resources and by the competitive use of those resources for other purposes. There are also increasing problems of contamination of surface and ground-waters to be used for other purposes, by the drainage effluents of irrigated lands. Taking into consideration the great investments required for the development of irrigated agriculture, the high contribution of this kind of agriculture to the World´s food production and the increasing scarcity and cost of available water resources, the degradation of irrigated lands through soil salinization and sodification becomes very important from the economic, social and environmental points of view. Besides, large and increasing proportions of the World´s irrigated land (25-50% depending on the evaluations) are affected by excessive salinity and sodicity.

In general, the soil salinity problems are a consequence of salt accumulation in zones and depths where the soil moisture regime is characterized by strong losses of water by evaporation and transpiration, and by reduced leaching of the remaining salts. The soil sodicity problems appear as a consequence of changes in the composition and concentration of those salts, with changes in the equilibrium exchangeable cations, leading to higher relative accumulation of exchangeable sodium percentage (ESP). Both processes of salinization and sodification are therefore influenced by soil water and solute balances (Pla, 1997). Irrigation and drainage may cause drastic changes in the regime and balance of salts and sodium in the soil profile, resulting in problems of salinization or sodification depending on climate, crops, soils, ground-water depth, irrigation and ground-water composition, and irrigation and drainage management. Restricted drainage may be due to low permeability of the soil or to the presence of shallow groundwater. Furthermore, the drainage waters coming from irrigated lands may contain not only natural salts, but also residues of fertilizers and pesticides, which are generally used in large amounts in intensive irrigated agriculture, and other pollutants derived from animal wastes and composted materials used as amendments and from partially or non treated urban or industrial waste-waters increasingly being used for irrigation.

DEVELOPMENT OF SALT-AFFECTED SOILS

Salt-affected soils, both saline and sodic, may develop both under dryland and irrigated conditions**,** affecting negatively the physical and chemical soil properties, the crop production and the animal and human health (Fig.1). Among the development processes of salt-affected soils, the processes of sodification have generally received less attention and are less understood than the development of saline soils. Although in both of them, hydrological processes are involved in their development, in the case of sodic soils we have to consider some additional chemical and physicochemical reactions, making more difficult their modeling and prediction.

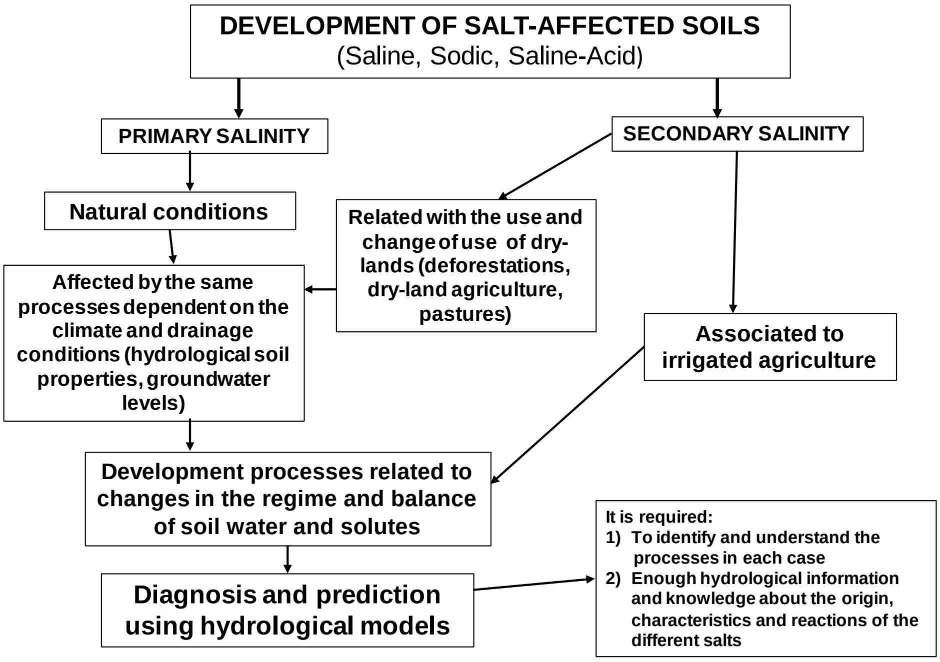


Figure 1. Common factors in the development of salt-affected soils under dryland and irrigated conditions

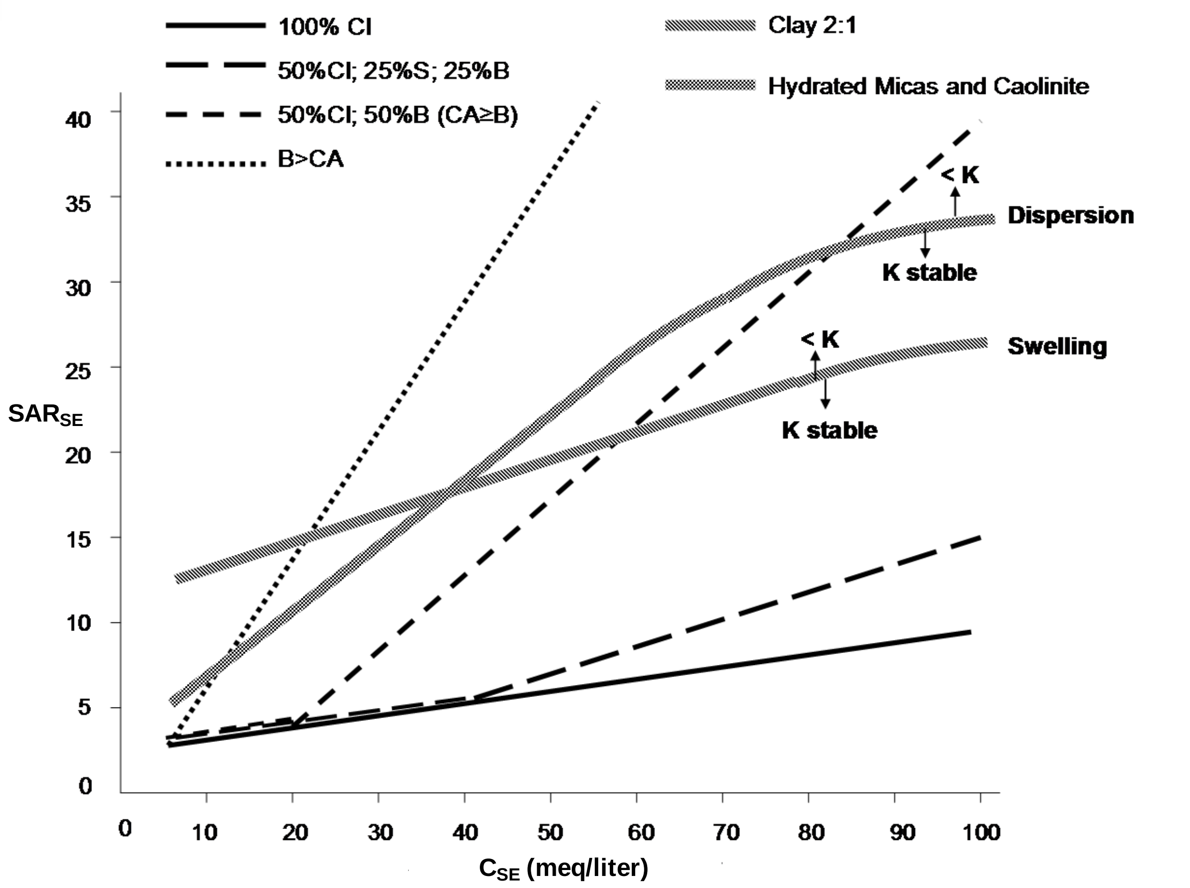
Sodic soils are dominated by Na (and by Mg in some cases) on their cation exchange sites. The sodicity status in the soil is expressed by the Sodium Adsorption Ratio (SAR):

$$SAR = Na / ((Ca + Mg)/2)^{1/2} \quad (meq/liter)^{1/2}$$

or by the Exchangeable Sodium Percentage (ESP), both related one to the other. Depending on different factors, soils are considered sodic if the ESP is 5-40%. The general lack of understanding of sodicity is in part due to the considerable variation of sodicity definitions, with variable ESP (or SAR) values reported to pose a sodicity problem, due to different textures and mineralogy of the soils and lack of consideration of the accompanying soil solution electrolyte concentration.

Sodicity produce changes on the soil´s physical properties, both by dispersion and plugging of soil pores by the moving clay particles and soil pore blockage by swelling clays. When surface soil disperses, the clay and silt particles clog surface pores, resulting on soil sealing, reduced infiltration and surface water-logging. This affects land use and plant growth by decreasing the permeability of water and air through the derived soil water-logging, and impeding root penetration. The impacts of these mechanisms are affected by several soil factors, mainly texture, clay mineralogy, total salinity and pH. Dispersion affects more soils with illite and kaolinite clays, at very low values of SAR if the salinity levels are also low, while swelling effects are more common in soils with smectites

The relationship between soil salinity and its flocculating effects, and sodicity and its dispersive or swelling effects on soil physical properties, specially the ones related with soil infiltration rates and hydraulic conductivity, is required to predict how specific soils will behave under different predicted combinations of salinity(C_{SE}) and sodicity (SAR_{SE}) (Fig. 2). Those relationships, depends highly on clay type, soil chemical reactions and soil texture. Among the main soil chemical reactions affecting those salinity/sodicity relationships are the ones involving bicarbonates and carbonates of Ca, Mg and Na, and Ca sulphates, leading to salt precipitation or dissolution, with changes in the soil salinity and sodicity levels (Pla,1968). Many errors in the evaluation and prediction of sodicity problems and effects are due to the non correct consideration of those chemical reactions under different relations among those cations and anions (Fig. 2). The main mistakes are done not considering the effects of sodium bicarbonate accumulation in soil solution, coming from irrigation waters or ground- waters, or produced by reactions under anaerobic conditions (Change in composition: $2Na^+ + SO_4^= + 2C + 2H_2O = S^= + 2NaHCO_3$).



SAR_{SE} values ($Na^+/(Ca^{++}+Mg^{++})^{1/2}$ (mmoles/liter)$^{1/2}$) with increasing salt concentration (C_{SE} meq/liter) in soil saturation extract (SE) from an original water with 5 meq/liter of salts (2,5 meq/liter Na^+; 1,25 meq/liter Ca^{++}; 1,25 meq/liter Mg^{++}) and different anion composition and concentrations (Cl:Chlorides; S:Sulphates; B: Bicarbonates) (CA:Ca + Mg; K: Saturated soil hydraulic conductivity)

Figure 2. Relationships between salinity and sodicity in the soil solution, and their effects on clay dispersion or swelling and in the soil hydraulic conductivity, for different anionic composition of the original water and different soil clays

MODELING SOIL SALINIZATION AND SODIFICATION

Both the addition of irrigation water and the changes in the depth and composition of groundwater may cause drastic changes in the water and solute balances in the soil profile. Modeling may be very useful for

the diagnosis and prediction of such changes, and in the selection of the best practices and systems of irrigation and drainage for a more efficient use of irrigation water and for reducing the losses and contamination of surface and groundwater, and controlling the soil salinization and sodification.

Use of more saline and sodic irrigation waters, such as treated municipal waste-waters and irrigation drainage waters, can be expected to increase in the future. This will require to modify existing soil and crop management practices, taking specially in consideration the interaction between soil management and soil sodicity under different levels of salinity. The main final objective has to be increased production with less water, reducing and controlling at the same time the negative environmental impacts on surface and ground-waters. Current available computer models provide only a limited ability to predict correctly those impacts of salinity and sodicity under different management conditions. It is necessary to include in them the interaction of many physical and chemical processes, for predicting short and long term consequences of varied management practices.

BARE SOIL (NO IRRIGATION)

$H_G = H_E - H_P$ (If : $H_E - H_P \geq 0$) ; $L_G = H_P / (H_E - H_P)$

At equilibrium: **If : $CA_G \geq B_G$**

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G \geq 30$

$CA_{SE} = ((CA_G - CAB_G - CaS_G) / L_G) + 40$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$; $CaS_p = (CaS_G / L_G) - 30$

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G < 30$

$CA_{SE} = ((CA_G - CAB_G) / L_G) + 10$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$

If: $CAB_G / L_G < 10$ and $CaS_G / L_G < 30$

$CA_{SE} = CA_G) / L_G$; $Na_{SE} = Na_G / L_G$

If : $CA_G < B_G$

$CA_{SE} = CA_G$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - CA_G$

$$C_{SE} = CA_{SE} + Na_{SE}$$
$$SAR_{SE} = Na_{SE} / (CA_{SE} / 2)^{1/2}$$

SOIL COVER WITH VEGETATION (NO IRRIGATION)

$H_G = H_{ET} - H_T - H_P$ (If : $H_{ET} - H_T - H_P \geq 0$) ;

$L_G = H_P / (H_E - H_T - H_P)$

At equilibrium: **If : $CA_G \geq B_G$**

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G \geq 30$

$CA_{SE} = ((CA_G - CAB_G - CaS_G) / L_G) + 40$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$; $CaS_p = (CaS_G / L_G) - 30$

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G < 30$

$CA_{SE} = ((CA_G - CAB_G) / L_G) + 10$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$

If: $CAB_G / L_G < 10$ and $CaS_G / L_G < 30$

$CA_{SE} = CA_G) / L_G$; $Na_{SE} = Na_G / L_G$

If : $CA_G < B_G$

$CA_{SE} = CA_G$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - CA_G$

$$C_{SE} = CA_{SE} + Na_{SE}$$
$$SAR_{SE} = Na_{SE} / (CA_{SE} / 2)^{1/2}$$

SOIL COVER WITH VEGETATION (IRRIGATION)

(NET UPWARD FLOW)

$H_G = H_{ET} - H_T - H_P - H_R$ (If : $H_{ET} - H_T - H_P - H_R \geq 0$) ;

$L_G = (H_P + H_R) / (H_E - H_T - H_P - H_R)$

At equilibrium: **If : $CA_G \geq B_G$**

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G \geq 30$

$CA_{SE} = ((CA_G - CAB_G - CaS_G) / L_G) + 40$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$; $CaS_p = (CaS_G / L_G) - 30$

If: $CAB_G / L_G \geq 10$ and $CaS_G / L_G < 30$

$CA_{SE} = ((CA_G - CAB_G) / L_G) + 10$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - 10$

If: $CAB_G / L_G < 10$ and $CaS_G / L_G < 30$

$CA_{SE} = CA_G) / L_G$; $Na_{SE} = Na_G / L_G$

If : $CA_G < B_G$

$CA_{SE} = CA_G$; $Na_{SE} = Na_G / L_G$

$CAC_p = (CAB_G / L_G) - CA_G$

$$C_{SE} = CA_{SE} + Na_{SE}$$
$$SAR_{SE} = Na_{SE} / (CA_{SE} / 2)^{1/2}$$

(NET DOWNWARD FLOW)

If : $H_{ET} - H_T - H_P - H_R < 0$

$H_D = H_P + H_R - H_{ET}$; $L_D = (H_P + H_R - H_{ET}) / H_R$

At equilibrium: **If : $CA_R \geq B_R$**

If: $CAB_R / L_D E \geq 10$ and $CaS_R / L_D E \geq 30$

$CA_{SE} = ((CA_R - CAB_R - CaS_R) / L_D E) + 40$; $Na_{SE} = Na_R / L_D E$

$CAC_p = (CAB_R / L_D E) - 10$; $CaS_p = (CaS_R / L_D E) - 30$

If: $CAB_R / L_D E \geq 10$ and $CaS_R / L_D E < 30$

$CA_{SE} = ((CA_R - CAB_R) / L_D E) + 10$; $Na_{SE} = Na_R / L_D E$

$CAC_p = (CAB_R / L_D E) - 10$

If: $CAB_R / L_D E < 10$ and $CaS_R / L_D E < 30$

$CA_{SE} = CA_R) / L_D E$; $Na_{SE} = Na_R / L_D E$

If : $CA_R < B_R$

$CA_{SE} = CA_R$; $Na_{SE} = Na_R / L_D E$

$CAC_p = (CAB_R / L_D E) - CA_R$

$$C_{SE} = CA_{SE} + Na_{SE}$$
$$SAR_{SE} = Na_{SE} / (CA_{SE} / 2)^{1/2}$$

H_G: Water coming from the water table, reaching the surface soil by capillary rise (mm)
H_E: Water loss by potential evaporation in a bare soil surface (mm)
H_{ET}: Water loss by potential evapo-transpiration in a soil surface covered with vegetation (mm)
H_T: Water loss by transpiration of the vegetation or crop covering the soil surface (mm)
H_P: Effective rainfall (infiltrated in situ) (mm)
H_R: Water applied by irrigation (mm)
C_G: Salt concentration in the ground-water (meq/liter)
C_R: Salt concentration in the irrigation water (meq/liter)
C_D: Salt concentration in the drainage water (meq/liter)
C_{SE}: Salt concentration in the saturation extract of the surface soil (meq/liter)
SAR: Sodium Adsorption Ratio (Na / $(CA/2)^{1/2}$) $(meq/liter)^{1/2}$
SE: Saturation extract of the soil
E: Leaching efficiency
G: Ground-water
CA: $Ca^{++} + Mg^{++}$ (meq/liter)
Na: Na^+ (meq/liter)
B: HCO_3^- (meq/liter)
S : $SO_4^=$ (meq/liter)
CAB = CA if B ≥ CA
CAB = B if B < CA
CaS = Ca – B – CACl if CaS ≥ 0
and (CaCl = Ca –B –S if CaCl ≥0)
CAC_p: Precipitated (Ca + Mg) carbonates (meq/liter)
CaS_p: Precipitated Ca sulphates (gypsum) (meq/liter)

Table 1. Equations to calculate the water and salt balances in the surface soil to predict salinization and sodification processes under different conditions, including the effects of ground water,

irrigation, and soil cover. (There are not shown the cases where $H_E – H_P < 0$ (NO COVER) or $H_{ET} – H_T – H_P < 0$ (VEGETATION COVER), and the only surface water input is rainfall water (NO IRRIGATION), because no soil salinization or sodification problems would be expected in such cases, unless the ground water table reaches the surface soil).

Most of the presently concepts and used approaches (static water quality indices, fixed limits for soil salinity and sodicity critical levels, empirical predictive models based on statistical relations or in laboratory methods not reflecting field conditions), although can characterize a particular system, can only be applicable to the conditions under which they were developed, but cannot be reliable if extrapolated to different situations. Models predictions would have to incorporate the effect of both the soil chemical (solution and exchange composition) changes, and soil and water management, as a function of irrigation and ground-water composition; and also the balance of water and solutes in the soil as influenced by climate, irrigation system, ground-water depth and crop water uptake. This model requirements are particularly important for soil sodification, where to reach an equilibrium level of exchangeable sodium may take a long time (up to several decades) to occur, but at the same time the resulting effects are very difficult to reverse.

A modeling approach, understanding and based on hydrological processes and associated impacts, may assist with decision making, and may provide an ability to test the impact of a variety of remedial or preventive options difficult to test previously in the field. The model SOMORE (Pla, 1997; 2006) and their present version) may be used for the initial evaluation of the soil water balances related to soil salinization and sodification processes. It has been found that vegetation changes can have strong effects on water dynamics, and on salt and sodium accumulation and distribution at different temporal and spatial scales under shallow ground-water. Therefore, hydrological studies, including water and salt balances, and water table fluctuation analysis, will be needed to preview the consequences of land use and vegetation changes on soil salinization and specially sodification processes, and to guide the requirements of irrigation and drainage management to prevent them.

The proposed model “SALSODIMAR” (Pla,1968;1977;1988;1997) is based on an independent balance of the salts and ions more common in irrigation waters, in ground-waters and in soil solution, and takes into consideration the processes and effects derived of the interaction among the compositions of the irrigation and groundwater, the evapotranspiration, the reactions of solution, precipitation and cation exchange, the soil hydrological properties and the effective leaching fraction. It was initially developed (Pla, 1967; 1968) to include the effect of hydrological aspects of irrigation and drainage on the salinization, and especially on the sodification processes in soils, associated to reactions of solution and precipitation of Ca and Mg carbonates and Ca sulphates occurring in the field. Subsequently, the effects of hydrological factors affecting the water and solute balances, the soil characteristics and the irrigation management were included in the model (Pla, 1977; 1983; 1986), and all was programmed in EXCEL for practical use. As such, it has been successfully used and tested at different levels, under tropical, subtropical and Mediterranean climate conditions

In this paper there is presented an adaptation of the model SALSODIMAR, including new specific hydrological components of the water and solute balances, to make it useful to predict the processes of both the dryland salinization and sodification processes originated in the ground-water, and the combined effects of irrigation and ground-water, with or without vegetation or mulch cover (Table 1)

CONCLUSIONS

There is shown how the model SALSODIMAR, based on the balance of water and soluble components of both the irrigation water and groundwater under different water and land management conditions, may be adapted for the diagnosis and prediction of the selected salinity and sodicity problems, and for the selection of alternatives for their management and amelioration. The prediction of soil salinity and sodicity problems, derived of increased use of low quality irrigation waters, including more or less treated waste-waters, in poorly drained soils, with shallow fluctuating groundwater levels, require adequate simulation modeling. These models have to be based on modeling hydrological processes responsible of water and solute balances in the soil, as influenced by climate, crops and irrigation and drainage management. Soil chemical and physicochemical reactions affecting the relationships of salinity and sodicity levels, have also to be included in modeling. The proposed adaptation of the model SALSODIMAR, that includes all those requirements, resulted to give reasonably good predictions in the preliminary evaluations of salinity and sodicity in the different case studies included in this paper. Additional research under field conditions would be needed for further improvement of the model predictions

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FOREST FIRES EFFECT ON SOIL EROSION PROCESSES

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INTRODUCTION

Soil erosion is one of the key processes to understand the World Land Degradation (Montgomery et al., 1988) due to the impact on soil and humankind sustainability (Brevik et al., 2015). This has been shown clearly by the rolesoils play in the United Nations Sustainable Development Goals (Keesstra et al., 2016a). It is well accepted that the threat of soil erosion also forms a long-term crop production issue (Larson et al., 1983), but the off site effects of soil erosionimpacts alsothreaten the land due to sedimentation. Moreover, soil erosion is changing the fate of the hydrological, erosional and biological cycles due to the transport and reallocation of seeds, nutrients, and water. The interaction between soil erosion and climate change will determine the fate of the Earth System (Nearing et al., 2004). As the drying trend caused by climate change in the Mediterranean the conditions are increasingly favorable for forest fires.

Forest fires results in the removal of the vegetation cover and changes in the soil surface cover and the soil properties (Lucas-Borja et al., 2016). As a consequence, the surface wash increase in forest fire affected land. Runoff is generated faster and with higher discharges, and sediment detachment is more efficient due to the lack of litter and vegetation (Martínez-Murillo et al., 2016a). This general trend that results in the degradation of soils is determined by the lack of vegetation. The vegetation recovery after forest fires favours that erosion rates will return to the previous situation. A key factor of the immediate post-fire effects on soils is the ash layer that can act as mulch. We found that black ash is water repellent but still allows infiltration in macro-pores and cracks, but white asks is hydrophilic and results in the formation of crusts (Bodí et al., 2011). This extended abstract will give examples of recent and long-term research about the effects of forest fires on soil erosion with special attention to the Mediterranean Ecosystems. We review here the main impact of forest fires on soil erosion through a review of scientific literature and we suggest sustainable restoration and rehabilitation management on fire-affected land.

SOIL EROSION CHANGES AFTER FOREST FIRES

The research carried out about the changes in soil erosion rates after forest fires shown that there is a sudden increase in soil losses the year after. Moody and Martin (2002) introduced the concept of ***Window of Disturbance*** as a consequence of forest fires temporal modification of the soil erosion process. The changes in soil erosion after forest fire where already measured by Cerdà (1998) in Eastern Spain using rainfall simulated experiments, and by Cerdà and Lasanta (2005) and Lasanta et al., (2005) under natural rainfall. There is a sudden increase in soil erosion rates that can be between 2 and 6 orders of magnitude the first year, but after five years is reduced to values similar to the previous ones, which confirm the ephemeral impact of forest fires on soil erosion. However, other researchers found that forest fires did not alter the soil erosion processes (Inbar et al., 1998). Morris and Moses (1987) found that the erosion rates were increased by three orders of magnitude, similar to the findings of Spigel and Robichaud (2007) or Martínez-Murillo et al., (2016b).

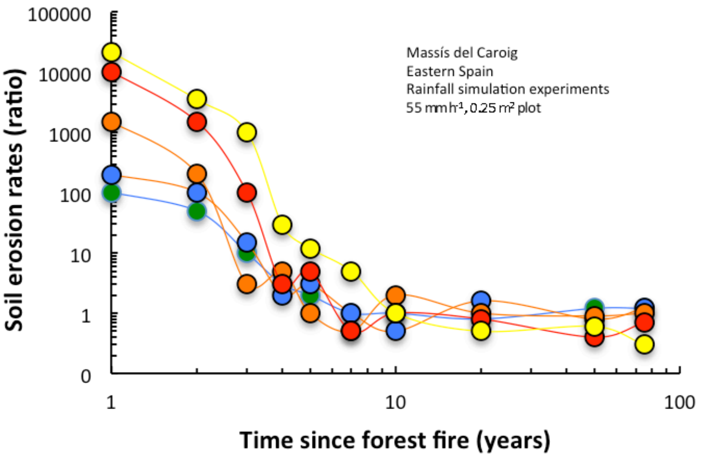

Figure 1. Soil erosion rates in areas with different fire history (time since last fire) in the Massís del Caroig SEDER study sites. Original data obtained by means of rainfall simulation experiments at 55 mm h^{-1} on miniature plots (0.25 m^2) which show that the detachment of material is very efficient after forest fires.

THE ASH ISSUE

Although the window of disturbance issue is fully accepted by the scientific community, there are clear evidences that immediately after forest fires soil erosion is controlled by the ash cover that act as a mulch and increase the infiltration rates and reduce soil losses as it prohibits runoff generation. The findings by Cerdà (1998) and Cerdà and Doerr (2008) of the effect of the ash on runoff generation and soil erosion was confirmed by (Woods and Balfour, 2008) and has been a key issue in understanding the effect of fire on the soil system and runoff generation (Zavala et al., 2009, Bodí et al., 2014). Figure 2 show how the ask cover is the key factor that determine the soil erosion rates after forest fires.

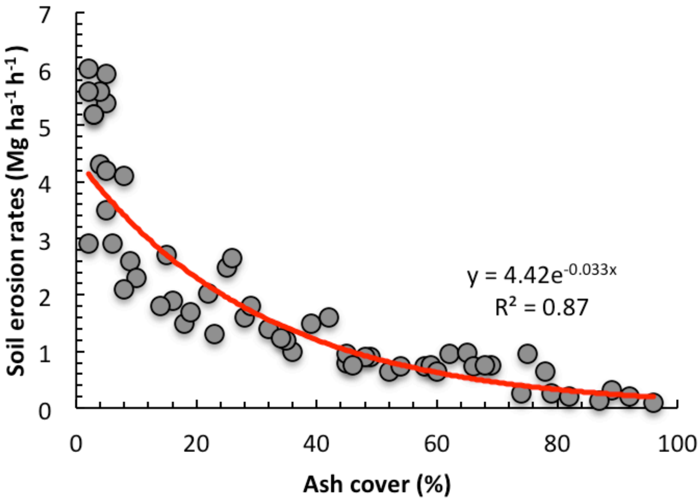

Figure 2. The effect of the ass cover immediately after a forest fire in Siete Aguas forest fire (2012, 25000 ha burned) measured by means of 0.25 m2 plots under simulated rainfall following the methodology of Cerdà and Doerr (2008). Original data

THE WATER REPELLENCY DEBATE

DeBano (2000) highlighted that after forest fires there is a reallocation of water repellent substances at 2 cm depth and this was researched by other colleagues (Doerr et al., 2000; MacDonald and Huffman, 2004). The water repellency is triggered after the forest fire due to the volatilization and condensation of plant substances at different soil depth, which is highly dependent on soil properties and fire temperature (Jordán et al., 2009). There is a debate about whether fire reduces or enhances soil water repellency. This is related to fire characteristics and type of plants (Bodí et al., 2011, Keesstra et al., 2017). Some plant species produce substances that are highly repellent, meanwhile others do not; and high temperature volatize those substances. The debate is open, and there is a need of more research about how the water repellent substances are redistributed during the fire. What is widely accepted is the importance of organic matter on soil water repellency such it was found in the Sierra de Enguera study site in Eastern Spain (Figure 3).

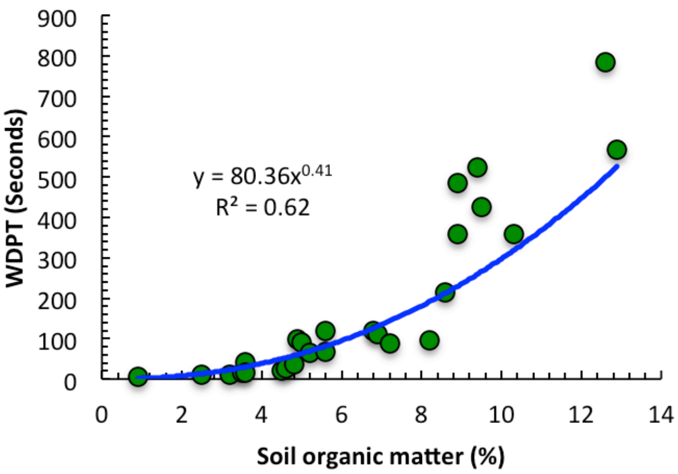

Figure 3. The effect of organic matter on soil water repellency in the Sierra de Enguerameasured under different plant species. Original data

Figure 4. Recovery of vegetation immediately after the forest fire. *Euphorbia sp*. to the left and *Dorycnium Pentaphyllum* to the right.

ARE FOREST FIRES SO BAD?

Forest fires were understood as the cause of Desertification and Land Degradation around the world. However, the positive influence of studies in Ecology have shown that forest fires are part of the ecosystem and that fire is part of nature (Pausas and Keeley, 2009). The recovery of vegetation after forest fire is a natural response of the ecosystem. The recovery of the biomass will end in the lowering of soil erosion rates (Cerdà, 1998). The comparison of fire affected land with other human disturbed land such as agriculture land shows that fire affected land does not show the largest erosion rates, meanwhile the citrus and olive plantations, and vineyards show extremely high erosion rates (Cerdà et al., 2016; Keesstra et al., 2016b, Rodrigo Comino et al., 2016). Moreover, after forest fires there is recovery of the vegetation that should be a key tool to manage the forest (Francos et al., 2016). Then, forest fires are not so bad.

CONCLUSIONS

Forest fires are part of the Ecosystem. The increase in soil erosion rates after a fire are temporally controlled by ash cover but soon,after few weeks or months, the bare soil yields large quantities of water and sediment. The recovery of the pre-fire conditions in terms of soil erosion rates takes some years (2-10). We need to do more research on water repellency changes and in the role of plants and ash in the fate of soil and water yield. Definitively, forest fires are not as bad as we thought.

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