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Soil and nutrient losses by runoff from farmlands in Southern Brazil

Luis Eduardo Akiyoshi Sanches Suzuki [1,*], Silvia Santin Bordin [2], Tiago Matieski [3], Pablo Rostirolla [1], Gilberto Strieder [1] e Márcio Renato Nunes [3]

[1] Universidade Federal de Pelotas, Pelotas, Rio Grande do Sul, Brasil.
[2] Universidade Estadual do Rio Grande do Sul, Erechim, Rio Grande do Sul, Brasil.
[3] USDA-Agricultural Research Service, National Laboratory for Agriculture and the Environment, USA.

* Autor Correspondente: dusuzuki@gmail.com

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Resumo: A erosão acelerada do solo é uma importante causa da sua degradação, levando a perdas de solo e nutrientes, aumentando os custos de produção e causando enormes danos ao ambiente. Nosso objetivo foi quantificar as perdas de solo e nutrientes por escoamento superficial e os custos da erosão em sete propriedades rurais (seis em plantio direto e uma em plantio convencional) no sul do Brasil. As perdas de solo e nutrientes por escoamento superficial foram medidas entre 31 de agosto de 2009 e 19 de novembro de 2010. O custo para reposição, através de fertilizantes minerais, dos nutrientes escoados superficialmente, dentro do período de avaliação, também foi quantificado. O total de solo erodido variou de 0,8 a 3,1 t ha^{-1}, com as maiores e menores taxas de erosão ocorrendo, respectivamente, nas propriedades em plantio convencional e plantio direto. O silte e a argila compõem mais de 50% do solo erodido, que também apresentou alta concentração de nutrientes. A concentração de nutrientes no solo escoado superficialmente foi maior do que na camada de 0 a 0,05 m do solo nas propriedades rurais. O custo estimado de fertilizantes para repor os nutrientes disponíveis P, K, Ca, Mg, e S foi de US$ 0,75 por hectare (menor taxa de erosão) e US$ 1,88 por hectare (maior taxa de erosão).

Palavras-chave: Erosão do solo; custo da erosão; redução da fertilidade; plantio direto; conservação do solo; manejo do solo.

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Abstract: Accelerated soil erosion is an important cause of soil degradation, conducting to soil and nutrients loss, increasing production costs, and causing damages to the environment. Our goal was to quantify soil and nutrient losses by runoff and the costs of soil erosion from seven farms (six under no-till and one under conventional tillage) in Southern Brazil. We measured soil and nutrient losses by runoff from those farms between August 31, 2009 and November 19, 2010. The cost to replace the nutrients lost with mineral fertilizer within that period was also quantified. Total soil runoff ranged from 0.8 to 3.1 t ha^{-1}, with the highest and the lowest erosion occurring within, respectively, the conventional tillage and no-till farms. Silt and clay composed more than 50% of the soil runoff, which also presented a high concentration of nutrients. The concentration of nutrients within the soil runoff was higher than that within the 0 to 0.05-m soil depth of the farms. The estimated fertilizer cost to replace available P, K, Ca, Mg, and S was US$ 0.75 per hectare (lowest erosion level) and US$ 1.88 per hectare (highest erosion level).

Key-words: Soil erosion; cost of erosion; fertility depletion; no-tillage; soil conservation; soil management.

1. INTRODUCTION

Geologic soil erosion by water (natural erosion) is the constant process of water detaching and transporting soil particles to the lowest point in a drainage basin. This is a natural process and its cumulative effects over billions of years has been responsible for the formation of the landscape (PIMENTEL & KOUNANG, 1998). These soil erosion processes, however, can be accelerated through human manipulation of soil resources, plant cover, or animal use. Therefore, accelerated soil erosion has become harmful to the environment and may impair land productivity.

Soil erosion is one of the biggest challenges within agricultural systems and has on-farm and off-farm impacts (LAL, 1985). Water that does not infiltrate within the soil becomes surface runoff, which carries soil particles, soil organic matter, fertilizers, and pesticides from agricultural fields, thereby reducing soil depth, promoting soil nutrients and organic matter losses, and degrading waterbodies (BAUMHARDT et al., 2015). By promoting soil nutrient losses and decreasing soil productivity, soil erosion increases production costs (SCHICK et al., 2000). A literature review by Lal (1985) showed that on-site, soil erosion decreases crop growth and yield due to poor seedling establishment, waterlogging, and crop burial (direct effect) and by altering the physical and nutritional properties of the soil by reducing the fertilizer use efficiency (indirect effect); meanwhile, the reservoir sedimentation is the major off-site damage caused by soil erosion. Therefore, soil erosion is a source of environmental, social, and economic issues, and can be considered a major setback for food security and a serious problem for sustainable development (TELLES et al., 2011).

Several practices have potential to accelerate soil erosion in agricultural lands, including (i) intensive soil tillage, (ii) removal of vegetation to create agricultural land, (iii) monoculture, (iv) insufficient soil protection by crops, (v) planting on a slope ("downhill"), (vi) burning or harvesting crop residues, (vii) overgrazing, (viii) disregard of the usability or the suitability of agricultural land, (ix) lack of vegetation to cover the soil, (x) cultivation in undulating terrains and without conservation practices, among others. Heathcote et al. (2013) showed that most of the sediment deposited in 32 lakes in the intensively agricultural region of the Midwestern United States was from the last 50 years and reflected agricultural intensification in that region rather than land clearance or predominance of agricultural lands. Although conservation agricultural practices, such as reduced tillage, have been adopted, Heathcote et al. (2013) suggested that traditional soil conservation programs have not decelerated downstream losses in the face of intensive agricultural practices, recommending new approaches to mitigate erosion and water degradation.

Polices should be established at regional and local land-use levels, according to the average rate of soil formation. Hundreds of crop species can be cultivated within the Brazilian territory, but plant and soil management should be chosen to reduce the soil loss to acceptable rates (SILVA et al., 2011). Thus, it is important to assess the effectiveness of recent improvements in land management, such as no-tillage and minimum-tillage, and the implementation of large-scale soil conservation programs, and their implications for the sustainability of the soil resource and future food security (WALLING, 2009).

Assessing the costs of water erosion is important because it reflects production expanses and land depreciation. However, analyzing these costs is difficult and complex because it involves direct and indirect questions, which may be difficult to quantify. Some may argue that the economic impacts of soil erosion belong to society understand the need to implement policies that encourage the adoption of soil conservation practices. Nevertheless, Alfsen et al. (1996) made the following question relative to the cost of soil erosion in Nicaragua: if there is excessive soil erosion, why do the affected farmers not engage more heavily in soil conservation? These authors also pointed out the slow process of soil degradation, resulting in small annual changes which are hard to verify when crop growth varies considerably due to management, disease, and rainfall. Technological improvements, such as fertilizer and pesticides inputs, may hide the impact of soil degradation.

The FAO-led Global Soil Partnership reports that 75 billion tonnes of soil are eroded every year from arable lands worldwide, which equates to an estimated financial loss of US$ 400 billion per year (GSP, 2016). Telles et al. (2011) estimated that soil erosion cost is US$ 44 billion per year in the United States, while in Paraná state, Brazil, this cost was around US$ 242 million per year. The same study showed that the cost of soil erosion in the European Union vary from US$ 5, to remove each ton of sediment to keep navigation activities in operation, to US$ 45.4 billion per year to offset the effects of loss of soil fertility and sedimentation of water resources. In a comprehensive literature review, Pimentel et al. (1995) estimated that the United States would have to invest US$ 6.4 billion per year (US$ 40 per hectare) for conservation practices to reduce erosion rates from 17 tonnes ha^{-1} year^{-1} to a sustainable rate of 1 ton ha^{-1} year^{-1} on most cropland, and an additional of US$ 2 billion per year to reduce erosion on pastureland. That is a small investment considering that erosion causes about US$ 44 billion in damages every year. It means that for each US$ 1 invested, US$ 5.24 would be saved, reducing soil loss by about 4×10^9 tonnes. Estimating soil erosion costs can assist conservation agencies in making decisions about soil conservation programs and provides information to governments to adopt public policies to ensure the sustainable development of economic activities linked to the farming sector (TELLES et al., 2011).

The requirements for food, fiber, and other resources for the human population are expanding, thus food security for the world population depends on the conservation of soil, water, energy, and biological resources (PIMENTEL & KOUNANG, 1998). Although sustainable agricultural practices have been promoted and adopted around the world, their benefits might be site-specific and cannot be generalized (NUNES et al., 2020). For example, the magnitude of those effects on soil health and crop yield depends on the adopted cropping system, soil type, weather, topography, and other inherent factors (NUNES et al., 2018, 2020). Thus, case studies based on individual farms and/or watersheds are still important and can provide relevant information to regional policy development (DISSART et al., 2000).

Considering the soil erosion impact on soil and nutrient loss, which contribute to soil fertility depletion and production cost increases, our goals were: (i) to quantify soil and nutrients losses by surface runoff; and (ii) to quantify the costs of soil erosion from farmlands under different use and management in the Erechim county, Rio Grande do Sul (RS) state, Brazil.

2. MATERIAL AND METHODS

2.1. Characterization of the area under study

The study was conducted in commercial farms located in Erechim county, Rio Grande do Sul state, Brazil. The county is located between 27º28'53" and 27º47'03" S and 52º 20'27" and 52º08'53" W, at an altitude of 768 m. Geologically it is in the Paraná Intracratonic Basin, located stratigraphically in the Serra Geral formation, which is derived from mafic rocks (basalt/diabase).

The region is characterized by two topographical areas: (i) the plateau with soft relief on the south and valleys on north; and (ii) abrupt slopes with basaltic outcrops, intensifying the coast and water erosion and the gravel and cobble contents. In that region, the relief is strong, predominantly consisting of Oxisol and Entisol (SOIL SURVEY STAFF, 2014) or, respectively, Latossolo and Neossolo soils, according to the Brazilian System Soil Classification (SANTOS et al., 2018). In the Erechim region, corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.) are the major grain crops and accounted for 68.7% of the planted area in 2005, while the permanent crops accounted for 4.2%. In addition to corn and soybean, cattle are relevant, with 25% of the rural population, approximately 400 families, producing milk for the regional market (ROSA & RODRIGUES, 2008). Recently, the agricultural census realized in 2017 (IBGE, 2019) indicated 20,009 hectares with temporary crops (929 farms), 1,020 hectares with permanent crops (388 farms), and 2,656 hectares with pastures (835 farms), being 620 farms with cattle. Considering the temporary crops, corn and soybean represent, respectively, 64% (594 farms) and 66% (612 farms) of the farms.

For this study, seven farms within the Dourado River watershed were chosen and, within these farms, representative agricultural areas were used for evaluation. Each farm was smaller than 50 ha and utilized family-based labor. The farms consisted largely of Entisol soils with different periods of no-tillage adoption. One farm remained under conventional tillage (Table 1). During the research, the areas evaluated on the farms were primarily growing corn in the summer and forage with cattle in the winter.

Table 1. Soil management, crop, and slope of farms during the study period (2009 to 2010).

Farm	Soil management[1]	Years		Slope, %[2]
		2009	2010	
1 (NT12)	no-tillage for 12 years	Cattle in *Lolium multiflorum*	*Zea mays*	30
2 (NT15)	no-tillage for 15 years	Cattle in *Avena sativa*	*Zea mays*	25
3 (NT8)	no-tillage for 8 years	Cattle in *Lolium multiflorum*	*Zea mays* + grazing for twice	30
4 (NT10)	no-tillage for 10 years	Cattle in *Lolium multiflorum*	*Zea mays*	30
5 (NT4)	no-tillage for 4 years	*Zea mays* harvested + Fallow	Fallow	18
6 (NT10)	no-tillage for 10 years	Cattle in *Avena sativa* + *Lolium multiflorum*	Fallow	36
7 (CT)	conventional tillage	Soil tilled	*Zea mays*	18

[1]Soil management history within each farm until August 2008. [2]Assessed by a clinometer Abney level.

2.2. Soil sampling and analyses

From August 31 to September 02 of 2009, disturbed soil samples were collected within the 0 to 0.05 m depth. Those samples were analyzed for particle size distribution, dispersible clay in water, wet aggregate diameter pondered, and chemical characterization. At the same time, undisturbed soil samples were collected in 0 to 0.05, 0.05 to 0.10, 0.10 to 0.15, and 0.15 to 0.20 m depth increments to evaluate soil porosity, bulk density, and saturated hydraulic conductivity. For this sampling, were used cylinders of 0.0470 m diameter and 0.0300 m height. Four replicates were sampled in the field for the cited analyses.

2.2.1. Particle size distribution, dispersible clay in water, and degree of flocculation

Particle size distribution was analyzed by the pipette method (GEE & OR, 2002). The dispersion of the soil samples was performed according to Suzuki et al. (2015) by horizontal shaker with 120 rpm lasting four hours, using glass bottles of 100 mL containing 20 g soil, 10 ml of 6% NaOH (chemical dispersant), 50 mL of distilled water and two nylon spheres weighing 3.04 g, 0.0171 m diameter and 1.11 Mg m^{-3} density each one. The particle sizes were separated in the following diameter: cobble (20 to 200 mm); gravel (2 to 20 mm); coarse sand (2 to 0.2 mm); fine sand (0.2 to 0.05 mm); silt (0.05 to 0.002 mm); clay (< 0.002 mm). The sand fraction was separated by sieving, clay was extracted using a pipette and the silt content was calculated.

Dispersible clay in water was quantified following the same procedure used for total clay evaluation but without using the chemical dispersant.

The calculus of the degree of flocculation (DF, %) followed the equation (1):

$$\mathrm{DF} = \frac{total\ clay - clay\ disperse\ in\ water}{\text{total clay}} x\ 100 \quad (1)$$

The textural class was determined using a soil texture triangle available from the National Resource Conservation Service/United States Department of Agriculture (NRCS/USDA, n.d.)

2.2.2. Wet aggregate stability

The soil aggregates were pushed through a 9.52-mm mesh sieve, breaking aggregates larger than 9.52-mm in their weak planes. Then, aggregates (>9.52 mm) were used to quantify wet aggregate stability using a method adapted from Kemper & Chepil (1965) using a Yoder (1936) vertical oscillation apparatus with sieves of 4.76, 2.00, 1.00, 0.25- and 0.106-mm mesh.

The pondered medium diameter of aggregates (PMDA, mm) was calculated following the equation (2):

$$PMDA = \frac{\sum_{i=1}(\text{mAGRi x ci})}{\sum_{i=1} mtAGRi} \quad (2)$$

where mAGRi = mass of aggregates (grams) in the class i (excluded the inert material); ci = medium value of the class of aggregates i (mm); mtAGRi = total mass of aggregates (grams) (excluded the inert material).

2.2.3. Soil chemical characterization

Soil samples were also analyzed for organic matter (OM), phosphorus (P), potassium (K), sulfur (S), calcium (Ca), magnesium (Mg), iron (Fe), copper (Cu), zinc (Zn), manganese (Mn), sodium (Na), aluminum (Al), and potential acidity (H + Al) as described by Tedesco et al. (1995). Soil pH was measured using a 1:1 soil and water ratio. Through these determinations were calculated the cation exchange capacity effective and pH 7.0 (CECeffective and CECpH7.0, respectively), base saturation (V), and aluminum saturation (m). The H + Al was determined for the SMP index, while the extractant KCl 1 mol L^{-1} was used to determine Ca, Mg, Mn, and Al, and the Mehlich I extractant was used for P, K, Na, Zn and Cu. Moist digestion was used to determine organic matter.

2.2.4. Soil porosity, bulk density, and saturated hydraulic conductivity

Samples with preserved soil structure were used to quantify the saturated hydraulic conductivity of soil, using a permeameter of constant charge (LIBARDI, 2005). Next, the samples were analyzed using a tension table (6 kPa tension) for two days to determine macroporosity, and then sent to the drying oven around 105°C to determine microporosity, total porosity (TEIXEIRA et al., 2017) and bulk density (BLAKE & HARTGE, 1986).

2.3. Soil and nutrient losses by surface runoff

The assessment of soil losses by surface runoff conducted between August 31, 2009 to November 19, 2010. To measure soil losses, four plots were placed within each farm. The plots were constructed using polyvinyl chloride (PVC), with strips of 1 m length and 0.15 m height, delimiting an area of 1 m^2, and were fitted using PVC pipe with a height of 0.25 m (Figure 1). The strips of the plots are linked through slots, to facilitate assembly, disassembly, and transport of such material.

Figure. 1. A meter square plot of PVC strips (1 m length and 0.15 m height) with a PET bottle cut in the down extremity of the plot.

A polyethylene tereflalato (PET) bottle was cut in half and placed in the lower edge of the plot to sample the soil loss by surface runoff. In the field, a hole was opened in the ground for fixing the PET bottle, where its border remained close to the ground surface and the PVC strips connected to the border of the PET bottle. The soil loss by runoff in the delimited area (1 m^2) was captured in the PET bottle. The management used in each farm was also applied in the plots.

To measure the plot slopes (Figure 2), fine iron stems were fixed in the soil at the superior (higher side – point A) and inferior (lower side – point B) extremes of the plots. A string was stretched from point A to B and a device was used to ensure and maintain the string as level. A mark was made in the stem of point B to measure the difference in height between the mark and the string. The horizontal distance (HD) between points A and B was measured along with the vertical distance (VD) at point B. The plot slope (S, %) was calculated using equation (3). The average plot slopes within each farm were: 27.1% - farm 1 (NT12); 28.5% - farm 2 (NT15); 22.7% - farm 3 (NT8); 30.1% - farm 4 (NT10); 33.3% - farm 5 (NT4); 33.4% - farm 6 (NT10); 27.5% - farm 7 (CT).

$$S = \frac{VD}{HD} x\ 100 \tag{3}$$

where VD = vertical distance, m; HD = horizontal distance between the extremities of the plot, m.

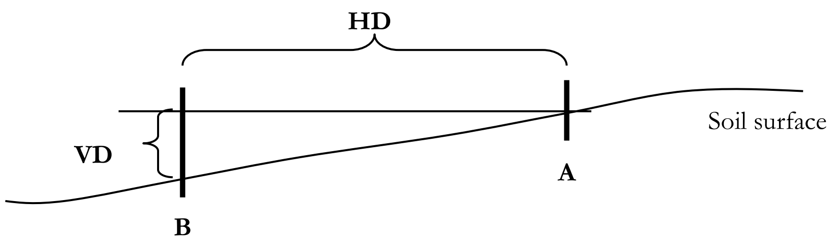

Figure 2. Scheme demonstrating the slope evaluation of the plots.

Between August 31, 2009 to November 19, 2010, after each rain event, the material collected in the PET bottle fixed in the lower edge of each plot were delivered to the laboratory and dried. At the end of the study period, all soil lost by surface runoff of each farm was combined to a composite sample, which was analyzed for particle size distribution and chemical characterization, following the same procedures described within subsections 2.2.1 and 2.2.3, respectively.

The total soil runoff collected from August 31, 2009 to November 19, 2010 was calculated using equation (4) and presented in kg ha^{-1}.

$$V = \frac{m}{BD\ x\ 1000} \tag{4}$$

where: V is the volume of soil runoff (m^3 ha^{-1}); m is soil runoff (kg ha^{-1}); BD is soil bulk density (Mg m^{-3}) at 0 to 0.05-m depth; the 1000-value is used to change Mg m^{-3} to kg m^{-3}.

By multiplying the soil (< 2 mm) runoff (g ha^{-1}), and the nutrient concentration (g dm^{-3}), the available nutrients runoff in one hectare (g ha^{-1}) was determined.

The cost of soil erosion (from August 31, 2009 to November 19, 2010) was calculated in consideration of the replacement of the available nutrients lost by runoff in one-hectare P, K, Ca, Mg and S, through commercial mineral fertilizer. Specifically, superphosphate (18% P_2O_5 + 10% S) was considered for P and S, potassium chloride (50% K) for K, and dolomitic limestone (32% CaO + 6% MgO) for Ca and Mg replacement. The information about the % content of each nutrient in the fertilizer was obtained in the "Instrução normativa nº 39", of August 8, 2018 (Instrução normativa do Ministério da Agricultura, Pecuária e Abastecimento do Brasil) (BRASIL, 2018), and the fertilizer cost was obtained using current market values.

The rainfall for the period of study was obtained from the National Institute of Meteorology, available at http://www.inmet.gov.br/portal/index.php?r=home/page&page=rede_estacoes_auto_graf, and the sum of rainfall in each month was plotted. The historical average of rainfall (1976-2005) was also plotted (WREGE et al., n.d.).

3. RESULTS AND DISCUSSION

Rainfall occurred in all months between August 31, 2009 to November 19, 2010 in the study region (Erechim county), but the total precipitation varied from one month to another (Figure 3). It should be noted that among the climatic factors responsible for starting the runoff, rainfall intensity is more important than the total rainfall. Thus, even the relatively low total precipitation within some months (i.e., total precipitation > 150 mm month^{-1}) may have a high potential to promote runoff and soil erosion since precipitation concentrated within a short period (i.e., minutes or hours) and antecedent soil moisture.

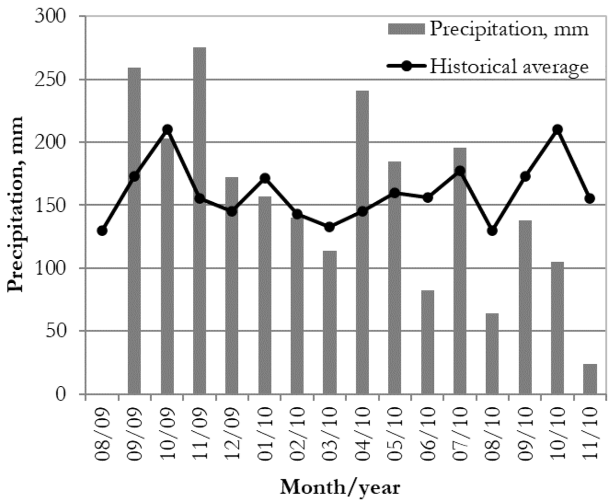

Figure 3. Month precipitation volumes in the study region (Erechim county) used to measure soil and nutrient losses by runoff and historical average (1976-2005). In August 2009, was only considered the day 31 (beginning of the measurements of soil runoff), and in November 2010 was considered the precipitation until day 19 (last day of measurement of soil runoff).

In addition to the climatic factors, several inherent soil characteristics drive runoff occurrence. Herein, soil hydraulic and physical characterization was intended to verify how those soil variables may influence runoff and thus soil and nutrient loss from farmlands. Independent of the farm or soil depth, soil bulk density was low (0.98 to 1.27 Mg m^{-3}) and soil macroporosity was high (0.106 to 0.264 m^3 m^{-3}), which contributed to the high saturated hydraulic conductivity for those soils (Table 2).

Low bulk density and high macroporosity reflected both fresh soil tillage in farm 7 (conventional tillage) and presence of cobble and gravel in the other farms (Table 3). Loose soil may increase soil hydraulic conductivity but, if the soil surface does not have the protection from cover (i.e., mulching) against rain impact, the soil may disaggregate, and further erosion may occur. Furthermore, the studied Entisols have a subsurface rock layer that tends to decrease water infiltration rates, making those soils more susceptible to water erosion. Within all farms, the soils were silty (soil texture classified as loam, silt loam, and silty clay loam) with a high amount of gravel (Table 3). Therefore, despite the high soil hydraulic conductivity, the soil's physical characteristics combined with high rainfall intensity suggest that the studied soils are susceptible to runoff.

Cobble was found on four farms (Table 3) and it was runoff from three of those farms (Table 4) that can be attributed to two factors: (i) cobble presence within the topsoil and (ii) high rain erosive force. On farm 5, tillage and runoff effects exposed cobble from soil depths deeper than 0.05-m (data not shown) and gravel runoff from all seven farms (Table 4). The total soil loss (cobble + gravel + sand + silt + clay) ranged from 0.9 to 3.9 Mg ha^{-1} and 0.89 to 3.60 m^3 ha^{-1}, while the loss of soil < 2 mm varied from 0.8 to 3.1 t ha^{-1} and 0.74 to 2.86 m^3 ha^{-1} (Table 4). Farms under conventional tillage or short-term (4 years) no-till had greater soil runoff than farms under long-term (> 8 years) no-till systems (Table 4), which reflect poor soil structure stability in those farms. These results agree with past studies. For example, Kaufmann et al. (2014) applied simulated rainfall on volumetric drainage lysimeters in soils from Southern Brazil and found that turning the soil over increases the time before surface runoff and internal drainage began. They have also shown that the generation of surface runoff was influenced by both surface sealing and plant development stage.

Soil loss from runoff must be lower than the soil formation rate. In addition, the nutrient loss and polluting water resources from soil loss must also be accounted for to improve land use and management. Herein, the rate of erosion can be associated with rain intensity, area slope, land use and management, soil texture and mineralogy. Selby (1973) verified that the variables of maximum rainfall occurring in 1 hour, slope angle, total rainfall and soil particles size together explain 61% of the runoff in undeveloped areas, while in pasture areas, maximum rain in 0.5 hours, the highest temperature and duration of no rainfall in each time (low preexisting soil moisture increases runoff) together explain 63% of the runoff. In this study, it was possible to verify the erosion process such as the splash, break up of soil aggregates, and transportation of individual soil particles in the plots. However, due to the short plot length (1 m), the effect of slope in the runoff process was minimal. Thus, if the actual length of the slope in the field was considered, the sediment loss could be even larger than that observed (Table 4), since the size and amount of sediment suspended in water is driven by the speed at which water runs and this speed depends on runoff length and slope. In soil under no-till with 0.105 m m^{-1} slope, Morais & Cogo (2001) verified that the critical runoff length ranged from 29 to 58 m to 3.9 t ha^{-1} with soybean residues, and 152 to 164 m to 6.2 t ha^{-1} with corn residues. Their study suggested that there are length limits on the gradient in no-till systems, but those limits depend on rainfall regime, soil, slope, and management conditions. Furthermore, plant spacing and architecture, plant establishment, soil tillage, weeds and crop residues can also influence erosion (PUTTHACHAROEN et al., 1998).

Table 2. Soil physical and hydraulic characterization by soil depth and farms.

Depth	*Macro	Micro	TP	BD	Ks	PMDA
m		$m^3 m^3$		$Mg\ m^{-3}$	$mm\ h^{-1}$	mm
			Farm 1 (no-tillage for 12 years)			
0-0.05	0.1911	0.3958	0.5869	1.17	86.46	2.48
0.05-0.10	0.1543	0.3994	0.5537	1.27	57.12	3.79
0.10-0.15	0.1633	0.3879	0.5512	1.18	66.54	3.56
0.15-0.20	0.1706	0.3950	0.5655	1.15	45.95	3.40
Average	**0.1698**	**0.3945**	**0.5643**	**1.19**	**64.02**	**3.31**
			Farm 2 (no-tillage for 15 years)			
0-0.05	0.1768	0.3968	0.5736	1.13	106.81	2.54
0.05-0.10	0.1552	0.3908	0.5460	1.19	45.42	1.75
0.10-0.15	0.1719	0.3730	0.5449	1.18	38.66	1.89
0.15-0.20	0.1413	0.3969	0.5382	1.17	53.69	1.76
Average	**0.1613**	**0.3894**	**0.5507**	**1.17**	**61.14**	**1.99**
			Farm 3 (no-tillage for 8 years)			
0-0.05	0.1346	0.4279	0.5626	1.26	22.42	8.22
0.05-0.10	0.1553	0.3950	0.5503	1.24	25.48	8.80
0.10-0.15	0.1471	0.4158	0.5629	1.24	48.93	8.98
0.15-0.20	0.1065	0.4390	0.5455	1.21	34.87	8.45
Average	**0.1359**	**0.4195**	**0.5553**	**1.24**	**32.92**	**8.61**
			Farm 4 (no-tillage for 10 years)			
0-0.05	0.1820	0.4200	0.6020	1.11	68.45	7.53
0.05-0.10	0.1237	0.4422	0.5658	1.14	41.47	7.58
0.10-0.15	0.1325	0.4379	0.5705	1.15	112.30	7.94
0.15-0.20	0.1290	0.4525	0.5815	1.11	35.51	8.04
Average	**0.1418**	**0.4382**	**0.5800**	**1.13**	**64.43**	**7.77**
			Farm 5 (no-tillage for 4 years)			
0-0.05	0.2235	0.4094	0.6329	0.98	209.47	8.93
0.05-0.10	0.1600	0.4329	0.5929	1.14	49.24	9.83
0.10-0.15	0.1647	0.4220	0.5867	1.21	76.78	9.98
0.15-0.20	0.1657	0.4414	0.6071	1.20	65.36	9.75
Average	**0.1785**	**0.4264**	**0.6049**	**1.13**	**100.21**	**9.62**
			Farm 6 (no-tillage for 10 years)			
0-0.05	0.1760	0.4147	0.5907	1.14	74.82	8.05
0.05-0.10	0.1526	0.4095	0.5621	1.23	32.05	7.64
0.10-0.15	0.2125	0.3875	0.6000	1.12	87.85	7.93
0.15-0.20	0.1565	0.3894	0.5459	1.22	50.38	8.16
Average	**0.1744**	**0.4003**	**0.5747**	**1.18**	**61.28**	**7.95**
			Farm 7 (conventional tillage)			
0-0.05	0.2643	0.3387	0.6030	1.08	207.07	8.02
0.05-0.10	0.2892	0.3441	0.6333	0.98	182.67	7.64
0.10-0.15	0.1965	0.3671	0.5636	1.17	212.32	7.78
0.15-0.20	0.1892	0.3713	0.5606	1.21	107.71	7.24
Average	**0.2348**	**0.3553**	**0.5901**	**1.11**	**177.44**	**7.67**

*Macro: macroporosity; Micro: microporosity; TP: total porosity; BD: bulk density; Ks: saturated hydraulic conductivity; PMDA: pondered medium diameter of aggregates.

Table 3. Particle size distribution, dispersible clay (DC), degree of flocculation (DF), and textural class within the 0 to 0.05 m soil depth for each farm.

Farm	*Cobble	Gravel	Sand			Silt	Clay	DC	DF	Textural class
			Total	Coarse	Fine					
	%									
1 (NT12)	1.73	19.20	29.25	11.20	18 .05	43.82	26.93	10.74	60.00	Loam
2 (NT15)	3.55	14.93	38.38	10.45	27.93	39.71	21.92	8.39	61.78	Loam
3 (NT8)	0.00	6.00	13.13	5.68	7.45	54.45	32.42	15.87	50.91	Silty clay loam
4 (NT10)	3.74	11.90	23.85	6.50	17.35	57.06	19.09	11.27	39.90	Silt loam
5 (NT4)	0.00	11.52	23.40	8.93	14.48	50.57	26.03	6.03	76.86	Silt loam
6 (NT10)	2.03	16.03	26.28	6.87	19.40	54.02	19.71	11.40	42.04	Silt loam
7 (CT)	0.00	14.92	22.98	12.85	10.13	51.52	25.50	15.72	38.28	Silt loam

*The sand, silt and clay particles refer to the fine soil fraction, whereas cobble and gravel percentages refer to the whole soil.

Table 4. Soil loss in runoff by farm between August 31, 2009 to November 19, 2010.

Farm	Cobble	Gravel	*Soil	Total	Cobble	Gravel	Soil	Total
	kg ha^{-1}				m^3 ha^{-1}			
1 (NT12)	0.00	369.20	898.63	1267.83	0.00	0.48	0.77	1.25
2 (NT15)	59.80	292.32	1087.58	1439.70	0.20	0.30	0.96	1.46
3 (NT8)	0.00	105.33	1800.73	1906.07	0.00	0.07	1.43	1.50
4 (NT10)	43.50	176.78	1472.11	1692.40	0.33	0.13	1.33	1.79
5 (NT4)	235.73	644.38	2392.42	3272.52	0.89	0.26	2.44	3.60
6 (NT10)	0.00	109.88	839.93	949.80	0.00	0.15	0.74	0.89
7 (CT)	0.00	802.58	3091.07	3893.65	0.00	0.28	2.86	3.14

*Soil: particle of diameter < 2 mm; total: sum of cobble + gravel + soil.

In this study, the highest soil runoff rate was observed under conventional tillage (Farm 7), while the farms under long-term no-till had the lowest rates. It is well known that the absence of crop residues on the soil surface, low surface roughness, disaggregated soil particles due to intensive tillage (plowing and harrowing) result in reduced water infiltration into the soil and increased runoff (GUADAGNIN et al., 2005). Crop residues on the soil surface can dissipate the kinetic energy of raindrops, virtually eliminating soil displacement by splash, increasing water infiltration, reducing ponding and the ability to breakdown and sediment transport (GUADAGNIN et al., 2005). Studies have also suggested that converting from conventional tillage to no-tillage can decrease sediment and nutrients losses from farmlands (RICHARDSON & KING, 1995). The effect of tillage intensity on soil and nutrient losses was also observed by several other studies around the world (HUSSEIN & OTHMAN, 1988). For example, Ampofo et al. (2002) who, using the Water Erosion Prediction Project (WEPP) model, verified that disc plow (conservation tillage) reduced soil loss by water erosion relative to conventional tillage plow in a semiarid region. Later, Zhou et al. (2009) also used the WEPP model and predicted sediment yield of 22.5, 17.7, and 3.3 t ha^{-1} year^{-1}, respectively for chisel plow, disk tillage and no-tillage in Iowa watershed (IA, USA). This predicted sediment yield was larger than that observed in our study, which can be associated with the rain intensity, slope, farm management, soil type, or even gravel and cobbles that create an uneven surface, reducing runoff speed and sediment transport in the farms studied here. Although the establishment and maintenance of conservation practices can be costly, their benefits in reducing on-site soil loss and the adverse off-site damages are significant (ZHOU et al., 2009).

The particle size distribution of soil (< 2 mm) runoff indicated a greater proportion of silt (36 to 55 %), followed by sand (16 to 48%) and clay (16 to 29%) (Table 5). The textural class changed from silt loam in the 0 to 0.05-m soil depth of the farms 4, 5, 6 and 7 (Table 3) to loam in the soil runoff (Table 5). In general, there was an increase in total sand, especially fine sand, and a decrease in silt and clay in the soil runoff, compared with the soil depth 0 to 0.05-m in the farms.

The greater proportion of clay and silt in the soil runoff (silt + clay varying from 52 to 84%), compared to sand particle size, may be associated to the larger amount of these particles in the soil (silt + clay varying from 62 to 87%) due to them being thinner and lighter, which facilitates their transport. Furthermore, due to their electric characteristics, clay and silt particles tend to be transported together, especially in soil with the presence of oxides, like those in this study. Especially in oxidic soils, clay and other cementing agents form microaggregates named pseudo-silt (KUNZE & DIXON, 1986; GALVÃO & SCHULZE, 1996; PINHEIRO-DICK & SCHWERTMANN, 1996). Although fine sand was also transported in relatively large amounts (Table 5), silt and clay particles deserve more attention since they are more important in terms of nutrients losses. Compared to sand, silt, and clay particles have a greater surface specific area and greater anion and cation exchange capacity. Therefore, losing high content of clay and silt by runoff leads to soil fertility depletion and contamination of water resources, as they may adsorb chemical elements.

Table 5. Particle size distribution and textural class of the soil runoff followed by the percentage change (in parentheses) comparing its presence in the soil runoff and 0-0.05 m soil depth, calculated as % = (soil runoff) – (soil depth 0-0.05 m).

Farm	Sand			Silt	Clay	Textural class
	Total	Coarse	Fine			
			%			
1 (NT12)	39.55 (+10.30)	11.42 (+0.22)	28.14 (+10.09)	40.20 (-3.62)	20.25 (-6.68)	Loam
2 (NT15)	47.58 (+9.21)	11.80 (+1.35)	35.78 (+7.86)	36.02 (-3.69)	16.41 (-5.51)	Loam
3 (NT8)	16.45 (+3.32)	5.80 (+0.12)	10.66 (+3.21)	55.01 (+0.56)	28.54 (-3.88)	Silty clay loam
4 (NT10)	30.60 (+6.75)	6.40 (-0.10)	24.20 (+6.85)	44.85 (-12.21)	24.12 (+5.11)	Loam
5 (NT4)	28.88 (+5.49)	8.46 (-0.46)	20.43 (+5.96)	44.83 (-5.74)	26.29 (+0.26)	Loam
6 (NT10)	28.21 (+1.94)	4.87 (-2.00)	23.34 (+3.94)	52.87 (-1.15)	18.93 (-0.78)	Loam
7 (CT)	26.60 (+3.63)	5.81 (-7.04)	20.79 (+10.67)	49.80 (-1.72)	23.60 (-1.90)	Loam

Runoff from the farms in this study carries sediments to the Dourado's river. Depending on its flow, these sediments (especially clay) can remain suspended in the river water and change its turbidity and color (Figure 4). A fraction of the sediment runoff does not reach the river. Durán et al. (2012) analyzed runoff and sediment yield from a semi-arid watershed (669.7 ha) in Southeastern Spain spanning three hydrological years (2007/08, 2008/09, and 2009/10) of rainfall. They found that a large proportion of soil loss from farmlands did not reach the watershed outlet, and the sediment delivery to the watershed outlet was determined by the spatial distribution of land-use types, as well as by the connection of sediment-producing areas and the runoff.

Figure 4. Images of Dourado´s river before and after the rain (May-7, 2010), showing the input of sediment from the watershed after rainfall.

Compared to the farm topsoil (0 to 0.05-m depth) chemical characterization (Tables 6 and 7), high concentrations of nutrients in the soil runoff (Tables 8 and 9) was verified. Following the "Manual de adubação e calagem para os estados do Rio Grande do Sul e Santa Catarina" (Fertilization and liming manual for the states of Rio Grande do Sul and Santa Catarina) (SBCS/CQFS, 2004), which is used for soil nutrient interpretation in that region, in general, the soil runoff pH, CECpH7.0 and base saturation (V) classification ranged from medium to high, aluminum saturation (m) ranged from low to very low, and organic matter (OM) content was classified as medium (Table 8). The P levels ranged from very low to very high, K, Ca, Mg, S, Fe, Mn, and Zn were considered high, while Cu content ranged from medium to high (Table 9). These results show that a high amount of soil nutrients was removed from the farmlands by the water (runoff), which aligns with past studies (HUSSEIN et al., 1999).

Table 6. Chemical characterization of the 0 to 0.05 m soil depth of the farms.

Farm	*pH	Al	H+Al	$CEC_{effective}$	$CEC_{pH7.0}$	m	V	OM
		$cmol_c\ dm^{-3}$				%		
1 (NT12)	5.8	0.2	5.7	28.4	34.8	0.6	84.2	3.1
2 (NT15)	6.5	0.1	5.1	16.3	21.3	0.4	75.9	3.0
3 (NT8)	6.1	0.2	6.0	14.6	20.5	1.4	70.5	2.9
4 (NT10)	6.9	0.0	3.8	37.1	40.8	0.1	90.7	3.2
5 (NT4)	5.2	0.5	10.5	22.9	32.9	2.4	67.5	5.2
6 (NT10)	6.0	0.1	4.8	13.2	16.9	1.0	67.0	2.8
7 (CT)	5.0	0.6	6.2	10.5	15.6	6.1	60.4	1.8

*pH: pH in water 1:1, Al: aluminum, CEC: cation exchange capacity, m: Al saturation, V: base saturation, OM: organic matter.

Table 7. Chemical characterization of the 0 to 0.05 m soil depth of the farms.

Farm	*P	K	Ca	Mg	Mn	Zn	Cu	Na
	$mg\ dm^{-3}$		$cmol_c\ dm^{-3}$		$mg\ dm^{-3}$			
1 (NT12)	4.6	35.5	13.4	9.0	65.6	5.0	2.1	5.7
2 (NT15)	5.5	19.3	7.9	6.3	65.6	7.2	2.5	2.0
3 (NT8)	3.8	17.7	6.4	5.0	168.7	8.3	1.9	2.9
4 (NT10)	7.5	162.0	17.3	12.3	61.1	4.3	1.1	6.8
5 (NT4)	8.9	46.1	8.8	6.4	170.6	14.1	1.2	7.0
6 (NT10)	1.9	158.8	6.8	4.3	59.5	8.8	2.4	1.9
7 (CT)	3.6	127.4	4.3	2.8	150.1	3.5	1.9	1.8

*P: phosphorus, K: potassium, Ca: calcium, Mg: magnesium, Mn: manganese, Zn: zinc, Cu: copper, Na: sodium.

Table 8. Chemical characterization of the soil runoff and, in parentheses, the percentage comparing its presence in the soil runoff and 0-0.05 m soil depth, calculated as % = (soil runoff) – (0-0.05 m soil depth).

Farm	*pH	Al	H+Al	$CEC_{effective}$	$CEC_{pH7.0}$	m	V	OM
		$cmol_c\ dm^{-3}$				%		
1 (NT12)	6.8 (+1.0)	0.2 (0.0)	2.5 (-3.2)	25.3 (-3.1)	27.6 (-7.2)	0.8 (+0.4)	90.9 (+6.8)	4.6 (+1.5)
2 (NT15)	6.4 (-0.1)	0.1 (0.0)	2.8 (-2.3)	16.8 (+0.5)	19.5 (-1.8)	1.0 (+0.6)	86.0 (+10.1)	3.3 (+0.3)
3 (NT8)	6.0 (-0.1)	0.0 (-0.2)	3.2 (-2.8)	12.3 (-2.3)	15.5 (-5.0)	0.0 (-1.4)	79.5 (+9.5)	3.3 (+0.4)
4 (NT10)	6.1 (-0.8)	0.0 (0.0)	3.1 (-0.7)	25.6 (-11.5)	28.7 (-12.1)	0.0 (-0.1)	89.0 (-1.7)	3.6 (+0.4)
5 (NT4)	5.2 (0.0)	0.4 (-0.1)	6.2 (-4.3)	12.6 (-10.3)	18.4 (-14.5)	3.0 (+0.6)	66.0 (-1.5)	4.0 (-1.2)
6 (NT10)	6.1 (+0.1)	0.0 (-0.1)	3.5 (-1.3)	14.7 (+1.5)	18.2 (+1.3)	0.0 (-1.0)	81.0 (+14.0)	3.2 (+0.4)
7 (CT)	5.5 (+0.5)	0.3 (-0.3)	4.8 (-1.4)	8.9 (-1.6)	13.4 (-2.2)	3.0 (-3.1)	65.0 (+4.6)	2.8 (+1.0)

*pH: pH in water 1:1; Al: aluminum; CEC: cation exchange capacity; m: Al saturation; V: base saturation; OM: organic matter.

The high concentration of nutrients in soil runoff can be associated with the high concentration of nutrient in the farmland soils (Tables 6 and 7) and with the high concentration of silt and clay in the soil runoff (Table 5). Within the studied region, farms tend to have a steep slope and thus soil runoff tends to be deposited in the lower parts of the relief, enriching soil down-slope, and decreasing soil fertility in the highlands. Although this process of erosion and deposition are responsible for soil formation, sediments can be deposited in rivers, streams, or lakes where they may cause siltation, eutrophication, or contamination of water resources.

Soil runoff is one of biggest factors for the need to replace soil nutrients through fertilizers and, consequently, increased costs. Calcium is adsorbed on soil colloids, facilitating its runoff. Ca and Mg loss by the erosion process tends to speed up soil reacidification, even when considering these elements are added in relatively large quantities during liming (SCHICK et al., 2000). P and K adsorbed to mineral colloids and organic matter may be responsible for the water eutrophication process. Schick et al. (2000) verified greater loss of K relative to P in sediment and linked this fact to its higher solubility and higher content in the soil compared to phosphorus.

Table 9. Chemical characterization of the soil runoff and, in parentheses, the percentage comparing its presence in the soil runoff and 0-0.05 m soil depth, calculated as % = (soil runoff) – (0-0.05 m soil depth).

Farm	*P	K	Ca	Mg	S	Fe	Mn	Zn	Cu	Na
	mg dm^{-3}		cmol$_c$ dm^{-3}		mg dm^{-3}					
1 (NT12)	11.3 (+6.7)	375.0 (+339.5)	16.7 (+3.3)	7.4 (-1.6)	nd	nd	nd	nd	nd	34.0 (+28.3)
2 (NT15)	17.2 (+11.7)	639.0 (+619.7)	11.1 (+3.2)	4.0 (-2.3)	87.5 (nd)	10500.0 (nd)	409.0 (+343.4)	3.9 (-3.3)	0.3 (-2.2)	26.0 (+24.0)
3 (NT8)	16.0 (+12.2)	403.5 (+386.3)	7.8 (+1.4)	3.5 (-1.5)	58.5 (nd)	7600.0 (nd)	765.0 (+596.3)	14.3 (+6.0)	7.6 (+5.7)	37.5 (+35.1)
4 (NT10)	28.5 (+21.0)	564.0 (+402.0)	15.9 (-1.4)	8.3 (-4.0)	96.2 (nd)	11800.0 (nd)	961.0 (+899.9)	10.4 (+6.1)	7.6 (+6.5)	49.0 (+42.2)
5 (NT4)	19.0 (+10.1)	402.0 (+355.9)	7.8 (-1.0)	3.4 (-3.0)	93.6 (nd)	9500.0 (nd)	985.0 (+814.4)	10.9 (-3.2)	5.7 (+4.5)	21.0 (+14.0)
6 (NT10)	4.2 (+2.3)	291.0 (+132.2)	11.0 (+4.2)	3.0 (-1.3)	25.3 (nd)	7300.0 (nd)	561.0 (+501.5)	0.8 (-8.0)	0.8 (-1.6)	27.0 (+25.1)
7 (CT)	11.9 (+8.3)	245.7 (+118.6)	5.8 (+1.5)	2.2 (-0.6)	55.3 (nd)	7566.7 (nd)	828.0 (+677.9)	7.4 (+3.9)	9.1 (+7.2)	33.0 (+31.2)

*P: phosphorus; K: potassium; Ca: calcium; Mg: magnesium; S: súlfur; Fe: iron; Mn: manganese; Zn: zinc; Cu: copper; Na: sodium; nd: not determined.

Soil organic matter (SOM) is also prevalent in runoff and, unlike the mineral nutrients, SOM replacement is more complex and depends on the amount and quality of the organic material added to the soil, its decomposition rate, climate, and soil type, among several other factors. Due to the low density of organic matter, it is the first constituent to be removed from the soil (SCHICK et al., 2000) during the erosion process. The same authors observed that plant residues and the absence of tillage in no-till systems reduced carbon runoff relative to tilled soils.

Quantifying soil nutrients loss by runoff is important for soil fertility management because these values must be considered in the nutrients reposition. In this study, the runoff of nutrients from the farms was high (Table 10). Soil macronutrients losses by runoff ranged from 3.53 to 45.46 g ha^{-1} of P; 244.42 to 961.75 g ha^{-1} of K; 1,851.54 to 4,690.68 g ha^{-1} of Ca; 306.15 to 1,484.55 g ha^{-1} of Mg and 21.25 to 223.93 g ha^{-1} of S. Soil micronutrients losses ranged from 6,131.48 to 23,389.11 g ha^{-1} of Fe; 444.82 to 2,559.41 g ha^{-1} of Mn; 0.67 to 26.08 g ha^{-1} of Zn; 0.33 to 28.03 g ha^{-1} of Cu and 22.68 to 102.01 g ha^{-1} of Na (Table 10).

Table 10. Available nutrients loss in the soil runoff, considering its nutrients concentration.

Farm	*P	K	Ca	Mg	S	Fe	Mn	Zn	Cu	Na
	g ha^{-1}									
1 (NT12)	10.15	336.99	3,007.43	807.96	nd	nd	nd	nd	nd	30.55
2 (NT15)	18.71	694.96	2,419.26	528.56	95.16	11,419.60	444.82	4.24	0.33	28.28
3 (NT8)	28.81	726.60	2,814.76	754.82	105.34	13,685.56	1,377.56	25.66	13.60	67.53
4 (NT10)	41.96	830.27	4,690.68	1484.55	141.62	17,370.92	1,414.70	15.31	11.19	72.13
5 (NT4)	45.46	961.75	3,739.63	988.31	223.93	22,727.94	2,356.53	26.08	13.64	50.24
6 (NT10)	3.53	244.42	1,851.54	306.15	21.25	6,131.48	471.20	0.67	0.67	22.68
7 (CT)	36.68	759.37	3,613.46	813.72	170.83	23,389.11	2,559.41	22.87	28.03	102.01

*P: phosphorus; K: potassium; Ca: calcium; Mg: magnesium; S: súlfur; Fe: iron; Mn: manganese; Zn: zinc; Cu: copper; Na: sodium; nd: not determined. The available nutrients runoff (g ha^{-1}) was obtained multiplying the soil (< 2 mm) runoff (data presented in Table 4), in g ha^{-1}, and nutrient concentration in that soil (data presented in Table 9), in g dm^{-3}.

Nutrient loss from one hectare during one single year may look minimal, but within several years and larger farmland areas, the cumulative costs of fertilizers to replace nutrients lost by runoff can be significant. This cost may be even larger if the indirect erosion effects on crop yield, waste of energy to product fertilizers, land depreciation, and environmental impacts are considered. The estimated amount of fertilizers to replace the amount of nutrients losses by runoff from farm 6 were 213 g ha^{-1} of superphosphate (18% P_2O_5 + 10% S) to replace P and S, 406 g ha^{-1} of potassium chloride (50% K) to replace K, and 4,137 g ha^{-1} of dolomitic limestone (32% CaO + 6% MgO) to replace Ca and Mg in farm 6 (less eroded soil), while in farm 7 (the most eroded soil), 1,708 g ha^{-1} of superphosphate, 1,261 g ha^{-1} of potassium chloride and 8,178 g ha^{-1} of dolomitic limestone would be necessary.

Following the local market fertilizer prices, the cost was US$ 240.00/ton of superphosphate, US$ 202.50/ton of potassium chloride and US$ 150.00/ton of dolomitic limestone, with an estimated total cost equal to US$ 0.75 per hectare from farm 6 (less erosion) and US$ 1.88 per hectare from farm 7 (more erosion), considering the available nutrients runoff from August 31, 2009 to November 19, 2010. Farms 6 and 7 were considered because they were, respectively, the lower and higher eroded farms, among all seven farms. Although the other farms presented an intermediate soil runoff, the cost of soil erosion may vary according to the nutrient concentration in the soil runoff. Other variables such as transport of fertilizer until the farms were not considered in our calculations.

Soil erosion exported 14.9 kg ha^{-1} of N and 11.5 kg ha^{-1} of total P from vineyard fields in Spain, which represented 6 and 26.1% of the annual intakes and 2.4 and 1.2% of the annual income from the sale of the grapes, respectively (MARTÍNEZ-CASASNOVAS & RAMOS, 2006). Another study found that soil loss is higher under annual crops than under the perennial crops of banana or banana-coffee (respectively 38.5, 6.6 and 0.87 t ha^{-1} year^{-1}) with a cumulative cost to replace NPK losses by erosion equal to US$ 16,663, 4,404 and 442 ha^{-1} year^{-1}, respectively (ONESIMUS et al., 2012). The same study showed that the total cost to replace nutrients is higher (US$ 15,451 ha^{-1} year^{-1}) in areas without conservation practices (terraces) than in areas with terraces (6,058 ha^{-1} year^{-1}). In a wheat – maize rotation, in a Cambic chernozem of Romania, Bucur et al. (2007) verified mean annual losses of 10.24 kg ha^{-1} N, 0.62 kg ha^{-1} P_2O_5, 1.38 kg ha^{-1} K_2O, 0.66 kg ha^{-1} Ca^{2+}, 0.19 kg ha^{-1} Mg^{2+} and 195.95 kg ha^{-1} humus. Those values, however, decreased with the increase in crop rotation (i.e., the inclusion of pea, wheat, alfalfa, and perennial grasses into the cropping system) to protect the soil against erosion.

Relative to the results found in this study, the costs of soil erosion reported by past studies are larger. This reflects the smaller soil runoff due to the short gradient length of the plots and the presence of soil fractions > 2 mm in the soil in this study. Furthermore, herein, only the available P was considered rather than total P; in addition, the cost of reposition of SOM, the fertilizer transportation and application cost, and the environmental cost of soil erosion, were not considered. Externalities such as environmental costs are difficult to evaluate. However, this study results highlighted that it is important to monitor long-term weather and soil use and management effects on soil erosion costs. Soil conservation costs do not stop after these practices are installed but continue due to the cost of maintenance of the practices (TROEH et al., 1980). For example, terrace channels and grassed waterways need to be cleaned and reshaped over time, drainage systems must be kept open, and the vegetation must be fertilized, mowed, reestablished.

4. CONCLUSIONS

Between August 31, 2009 to November 19, 2010, this study quantified soil and nutrients losses by surface runoff and the erosion costs in farmlands under conventional tillage and no-till systems in the Erechim county, Rio Grande do Sul State, Brazil. Soil runoff occurred from all farms, independent of soil management, and the soil loss ranged from 0.8 to 3.1 t ha^{-1}. However, the greatest soil loss occurred within the conventional tillage system, which is due to the unprotected soil surface and the poor soil aggregation in that system. More than 50% of the soil runoff is composed of silt and clay. Moreover, the concentration of nutrients within the soil runoff was higher than that within the 0 to 0.05-m soil depth of the farms. This suggests high nutrient losses from the farmlands due to the erosion, which in turn may led to soil fertility depletion, land depreciation, environmental damages (i.e., eutrophication, siltation, and contamination of the water bodies), and higher crop production costs.

The cost to replace the available P, K, Ca, Mg, and S with mineral fertilizer was estimated at US$ 0.75 per hectare in the lowest eroded soil (farm 6) and US$ 1.88 per hectare in the most eroded soil (farm 7). This estimation did not consider other costs such as fuel, fertilizer transportation, land depreciation, and environmental impacts. Thus, we assume that the real cost of the soil erosion in those farms is even larger than that estimated herein.

Finally, we suggest that long-term studies are still needed to monitor the response of soil erosion to different soil and weather and soil and crop management. Results from this kind of study can support the adoption of new approaches to mitigate erosion and prevent soil and water degradation and to improve soil and water conservation programs.

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