LARGE-SCALE MODELING OF SOIL EROSION WITH RUSLE FOR CONSERVATIONIST PLANNING OF DEGRADED CULTIVATED BRAZILIAN PASTURES

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ABSTRACT

Pasture degradation is one of the major environmental and economic problems of Brazilian livestock production. Based on the estimates of soil loss in 140,297 km^2 pasturelands of Goiás State and the Federal District, the effects of land use and management and conservation practices on soil erosion by water were evaluated. Soil loss was estimated with the empirical revised universal soil loss equation model under four scenarios of land use and management of pastures and the implementation of terraces. The effects of converting hilly areas into permanently preserved areas were also evaluated. Well-managed pastures planted in terraces and highly sloped areas with native vegetation featured a reduction of 709% in the rate of soil erosion by water in comparison with poorly managed pastures without soil conservation practices. The environmental consequences of pasture degradation become particularly important when they are associated with land use in the Amazon and Cerrado biomes because farmers in these regions are forced to open new natural areas. In addition, the high indices of bare soil in degraded pastures contribute significantly to sediment deposition in rivers and dams. The modeling of soil loss is demonstrated to be an important tool for land use planning and supporting public policies focused on the sustainable use of natural resources. The results of these simulations demonstrate the importance of actions to recover degraded pastures, such as the Federal Government's program known as ABC (Low Carbon Agriculture). Copyright © 2015 John Wiley & Sons, Ltd.

KEY WORDS: soil loss; remote sensing; geographic information system; USLE; RUSLE; Brazilian savanna

INTRODUCTION

Soil erosion by water has been very accelerated by human activity and is a major concern affecting all continents of the world. As an example, high erosion rates can be found in Eastern Spain (García-Orenes *et al*., 2009) in the western Mediterranean badlands (Martínez-Murillo *et al*., 2013), in the Loess Plateau of China (Zhao *et al*., 2013), in the eastern Himalayas of India (Mandal & Sharda, 2013), in the western Brazilian Amazon (Lu *et al*., 2007), and Slovakia (Lieskovský & Kenderessy, 2014). Pasture and grazing land has also presented high soil losses, mainly because of mismanagement and overgrazing (Cerdà & Lavee, 1999; Dickie & Parsons, 2012; Galdino, 2012; Martínez-Garza *et al*., 2012; Mekuria & Aynekulu, 2013; Souza Braz *et al*., 2013). The impact of overgrazing on soil erosion is because vegetation controls the soil losses, and this was found at pedon scale and watershed scales (Cerdà, 1998; Keesstra, 2007; Keesstra *et al*., 2009; Gabarrón-Galeote *et al*., 2013; Bochet, 2015) and also in the river channels (Keesstra *et al*., 2012).

A survey led by the Food and Agriculture Organization of the United Nations (Steinfeld *et al*., 2006) indicated that pastures occupy 26% of world's ice-free land. Livestock also provides employment to ~ 1·3 billion people around the world and is an important source of subsistence to another billion people. The activity contributes to approximately 40% of gross domestic product of the world's agriculture. Brazil holds the largest commercial cattle herd in the world (Steinfeld *et al*., 2006). According to the 2006 Brazilian Agricultural Census (IBGE, 2013), Brazil has approximately 176 million cattle herds and 102 million hectares of cultivated pastures.

Ninety-three percent of cattle in Brazil are pasture fed. Its costs are low, representing only approximately 60% and 50% of the costs associated with livestock in Australia and the USA, respectively (Ferraz & Felício, 2010). The small investments are related to the lack of proper handling, which leads to pasture degradation, one of the biggest problems of the sector. Pasture degradation leads to economic and environmental losses.

It is estimated that more than 70% of pastures in Brazil are degraded to some degree. This degradation can be perceived as a degeneration process or as a loss of productivity (Macedo, 2000). Approximately 80% of the cultivated pastures in the Brazilian savanna (Cerrado) biome are degraded

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to some level degree (Macedo, 2000). Soil erosion is the most critical step in the pasture degradation process. In fact, soil erosion is present in all phases and tends to increase with reductions in the soil vegetation cover of pasture, weeds, or surface residues (Galdino, 2012). To minimize soil erosion in pasturelands, the use of correct management and conservation practices is mandatory (Macedo, 1997; Macedo *et al.*, 2014).

According to IBGE (1998), the investments in soil conservation practices in Brazil were adopted in only 15% of rural establishments; 11% of farms use contour lines, and only 3% adopted terraces. The reclamation of degraded pastures is currently one of the Brazilian government's top priorities. The Ministry of Agriculture and Food Supply (MAPA) established a program for reduction of greenhouse effect gases in agriculture, known as the ABC Program (Low Carbon Agriculture), which includes funds for projects related to the recovery of degraded pastures (Cooper *et al.*, 2013).

In the 1989–2009 period, Goiás State, of which 97% is covered by the acidic, low-fertility soils of the Cerrado biome, had a 20% growth in its cattle herd. According to IBGE (2013), 37% of land in Goiás is cultivated pastures. In 2010, Goiás had 21·3 million animals, representing 10% of the total Brazilian cattle herd (IBGE, 2013).

Models that predict soil loss based on geotechnology are important tools for identifying areas with high erosion risks and for evaluating the impacts of different land use scenarios, as well as the effects of soil conservation practices on soil erosion by water in agricultural areas. A recent example in Brazil was the study conducted by Galdino (2012) that measured sandy soil erosion in pastures of the Alto Taquari Hydrographic Basin (which includes the states of Mato Grosso and Mato Grosso do Sul). The high production of sediments, mainly related to cattle raising activities, intensified the soil sediment accumulation in the Taquari River basin plains, leading to major socioeconomic and environmental impacts in the Brazilian Pantanal (Galdino, 2012).

Based on the spatial estimates of soil loss in different land use scenarios and on the conservationist land use practices in pasturelands of Goiás State (GO) and the Federal District (FD), the objective of this study was to evaluate the impacts of conservationist planning on the reduction of soil erosion by water. This information will support public policies focused on sustainable use of natural resources.

MATERIAL AND METHODS

Study Area

The study area was the areas occupied by cultivated pastures in Goiás and the FD (in the year 2009), between the latitudes 12°23′30″S and 19°29′52″S and the longitudes 45°54′30″W and 53°15′14″W (Figure 1). Goiás and the FD cover areas of $340,112 \, km^2$ and $5,780 \, km^2$, respectively, in the central part of the country. The Landsat Thematic Mapper image interpretation showed that, in 2009, 22% of FD territory ($\sim1,246 \, km^2$), was covered with pastures, while 41% of Goiás State was occupied by pastures ($\sim140,297 \, km^2$) (Sano *et al.*, 2010). According to IBGE (2013), in the 1989–2009
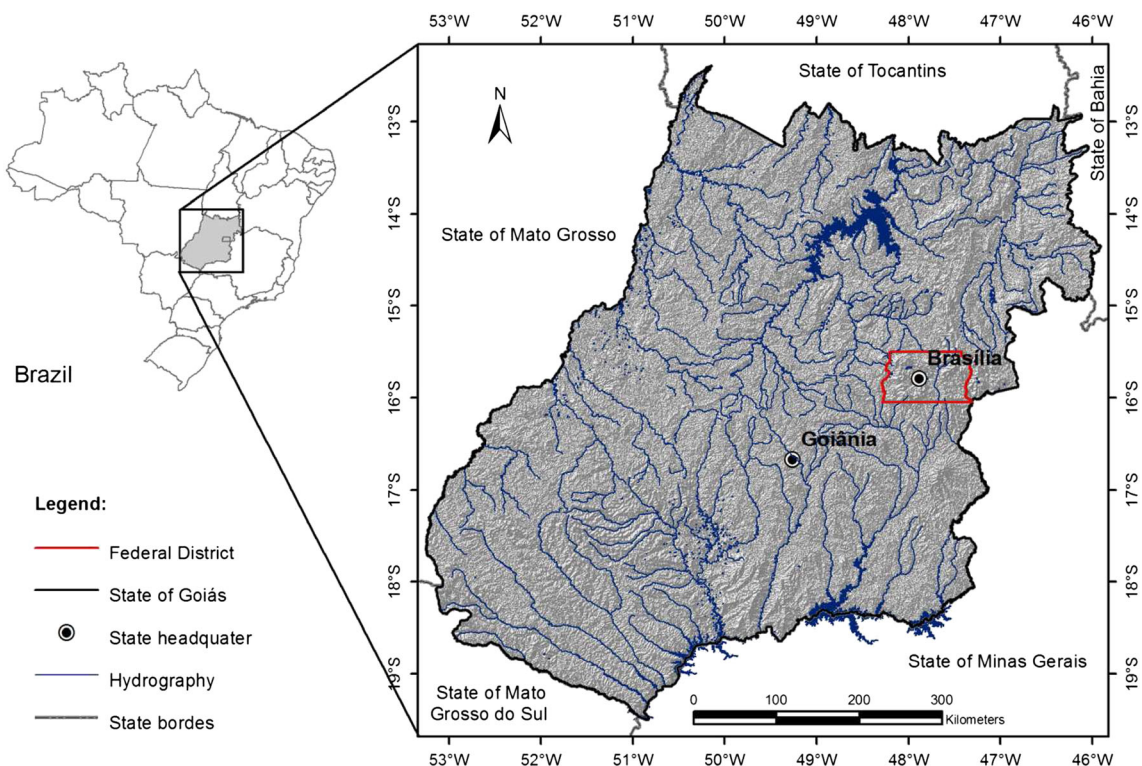

Figure 1. Location of Goiás State and the Federal District in Brazil. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr

period, Goiás State had a 20% growth in its cattle herd. In 2010, Goiás had 21·3 million animals, representing 10% of the total Brazilian cattle herd.

Topography of the study area is gentle: 31% of its land is flat (0% to 3% slope), 43% is gently undulating (3% to 8% slope), and 20% is undulating (8% a 20% slope). Slopes higher than 20% are present in only 6% of the total area. The climate in the study area is classified as tropical continental with dry winter (Aw in the Köppen classification system) (Peel *et al.*, 2007). Average monthly temperatures vary from 22–26 °C. According to the rainfall data recorded by the National Institute of Meteorology (INMET, 2014), more specifically by the rain gage stations located in 13 municipalities, the mean annual rainfall in the study area is 1,550 mm (standard deviation = 274 mm), with two well-defined seasons: the dry season, from mid-April to late September, and the wet season, from early October to mid-April. The total rainfall in the months of June, July, and August is often lower than 1 mm.

Goiás and FD lies entirely within the Central Highlands, which are located in the central part of the country. The large plateau, that is, the vast and flat terrain with elevation varying from 750 to 900 m, hosts the headwaters of three largest Brazilian river systems: the Paraná River (southern part of Goiás), the São Francisco River (eastern part of Goiás), and the Araguaia/Tocantins River (northern part of Goiás).

Approximately 97% of study area is covered by the acidic, low-fertility soils. Ferrasols (IUSS Working Group WRB, 2006), corresponding to the Latossolo Vermelho and Latossolo Vermelho-Amarelo in the Brazilian soil classification system (EMBRAPA, 2006), occur in 43% of the study area. The other two important classes are the Cambisols (Cambissolos) and Acrisols (Argissolos), which represent 31% of Goiás and FD together.

Estimation of Soil Loss

The universal soil loss equation (USLE) (Wischmeier & Smith, 1978) and the revised universal soil loss equation (RUSLE) were used to estimate soil loss in the study area (Equation 1).

$$A = R*K*L*S*C*P \qquad (1)$$

where A = soil loss per area and time (in $\text{Mg ha}^{-1}\text{y}^{-1}$); R = a rainfall erosivity factor (in $\text{MJ mm ha}^{-1}\text{h}^{-1}\text{y}^{-1}$); K = a soil erodibility factor (in $\text{Mg h MJ}^{-1}\text{mm}^{-1}$); L = a slope length factor; S = a slope steepness factor; C = a cover-management factor; and P = a support practice factor. The last four parameters are unitless.

Renard *et al.* (1997) conducted an extensive revision of USLE, leading to the development of RUSLE. Although the equation structure is the same, the methods to determine the model factors were modified. The RUSLE is an empirical model as the USLE; the added valued (compared with the USLE) is mainly linked with its implementation, and the possibility to integrate new and more flexible equations in the model. Both USLE and RUSLE are widely utilized

for erosion prediction in Brazil (Beskow *et al.*, 2009; Lu *et al.*, 2007; Galdino, 2012) and worldwide (Lal, 2001; Fu *et al.*, 2005; López-Vicente & Navas, 2009; Omuto & Vargas, 2009; Haile & Fetene, 2012; Lieskovský & Kenderessy, 2014; Ligonja & Shrestha, 2013).

The USLE/RUSLE has been also widely applied for regional scales. For example, Spaeth *et al.* (2003) evaluated these equations using simulated rainfall data from varying set of United States rangeland vegetation (8 states, 22 sites, and 132 plots). Fu *et al.* (2005) assessed the soil erosion using RUSLE in the Chinese Loess Plateau, one the areas in the world presenting the highest soil erosion rates. Chen *et al.* (2011) conducted a regional soil erosion risk mapping combining RUSLE, geographical information system, and remote sensing data over ~ 16,000 km^2 Miyun reservoir watershed located in the northern part of Beijing, China.

The WaTEM/SEDEM model (Van Oost *et al.*, 2000; Van Rompaey *et al.*, 2001) represents a simple spatially distributed erosion and sediment delivery model based on the RUSLE. The WaTEM/SEDEM, in addition to simulate soil loss, is able to simulate also the deposition of eroded material by using additional input parameters. The model has been applied and validated under various geographical conditions principally in Europe: Belgium (Van Rompaey *et al.*, 2001), Italy (Van Rompaey *et al.*, 2005), Slovenia (Keesstra *et al.*, 2009), Czech Republic (Krasa *et al.*, 2010), Spain (Alatorre *et al.*, 2010), the Slovakia (Lieskovský & Kenderessy, 2014), and other countries.

According to Wischmeier and Smith (1978), rainfall erosivity is directly proportional to the kinetic energy and the maximum rainfall intensity within 30 min. However in Brazil and worldwide, there is a lack of rain-gage-based rainfall data. Thus, several studies using modified fournier index were developed for estimating the R factor based on the pluviometric data (Oliveira *et al.*, 2012; Taguas *et al.*, 2013). One of the most used models in Brazil was developed by Lombardi Neto and Moldenhauer (1992) using 1954–1975 time series data of precipitation in Campinas, Sao Paulo. They found a high correlation between the monthly average of the erosivity index and the monthly average rainfall coefficient (Equation 2).

$$EI = 68.73\left(\frac{p^2}{P}\right)^{0.841} \qquad (2)$$

where EI = the monthly average erosion index (in $\text{MJ mm ha}^{-1}\text{h}^{-1}\text{y}^{-1}$); p = the monthly average rainfall (in mm); and P = the annual average rainfall (in mm).

Considering the lack of long rainfall data series for the study area and that rainfall data from the Brazilian National Water Agency (ANA) is the main source of data available, the applicability of Equation 2 was verified using rainfall data from 88 pluviometric stations with at least 22 years of complete annual records. Afterwards, the total annual rainfall averages were compared with the monthly rainfall data collected by those 88 stations from Campinas.

Simple linear correlation coefficients (*r* of Pearson) were determined between the monthly rainfall averages from the ANA stations and the rainfall data from Campinas for the period 1954–1975. The calculated *r* values were high (average = 0·93; range: 0·87–0·96). The Campinas annual average rainfall was 1,359 mm, and the annual average rainfall of the ANA stations was 1,531 mm. The results were considered satisfactory for the use of Equation 2 for the Goiás and FD data. To generate a rainfall erosivity map, the values of *R* from the 88 stations were interpolated. Kriging was used for this interpolation because it allows interpolation with minimum variance and without bias.

Some preliminary tests were performed. An exploratory analysis was first considered using descriptive statistics to identify outliers and the normality of the distribution of erosivity data frequency. The parameters of skewness, kurtosis, and the significance level of Kolmogorov–Smirnov test (*KS* test) for normality was obtained according Steinskog *et al.* (2007).

Geostatistical analysis was used to identify the presence of spatial variability and generate a rainfall erosivity map. The calculation of semivariance and the adjustments in the semivariogram were performed to verify the existence of a spatial dependence of erosivity, assuming the stationarity of the intrinsic hypothesis and calculation of semivariance $\gamma(h)$, estimated by Equation 3:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N} [Z(x_i) - Z(x_i + h)]^2 \qquad (3)$$

where $N(h)$ = the number of measured pairs of values $Z_{(xi)}$ and $Z_{(xi+h)}$, separated by a vector *h*. As stated by Vieira (2000), it is expected that measurements located closer to each other tend to be more similar than those separated by larger distances. In other words, $\gamma(h)$ increases with the distance *h* until a maximum value, becoming stable at a level that corresponds to the limit of spatial dependency, which is the range. Measurements located at distances larger than the range present random distribution because they are independent of each other.

For the analysis of spatial variability, the semivariogram was adjusted with the mathematical model that featured better correspondence (exponential) and the nugget effect (C_0); partial sill (C_1) and range (*a*) were obtained. The semivariogram adjustment generates some uncertainty about the parameters that were adjusted to models. This uncertainty is the estimated error that can be evaluated using the auto validation procedure called jack-knifing, and it involves an estimate of each measured point considering its inexistence during the estimation, as detailed by Vieira *et al.* (2010). The software and procedures for the build-up and adjustments of the semivariogram model were based on Vieira *et al.* (2002). The degree of spatial dependence (GD) was calculated (Equation 4) and categorized into the following classes (Zimback, 2001): weak (<25%), moderate (26–75%), and strong (>75%).

$$GD = \left(\frac{C_1}{C_0 + C_1} \right) \cdot 100 \qquad (4)$$

where C_0 is the nugget effect, that is, the semivariance at zero distance and C_1 is the structural variance. When spatial dependency is demonstrated by the semivariogram, it allows the estimation of values at any other unsampled place by using ordinary kriging.

The values of *R* from the 88 stations were interpolated. Kriging was used for this interpolation because it allows interpolation with minimum variance and without bias. The type used was ordinary kriging, one of most used by its simplicity and results (Vieira, 2000; Yamamoto & Landim, 2013). It is a method for estimation of values for locations that were not sampled considering the linear combination of the values for immediate neighborhood.

The data was processed by the software GEOTEST, a geostatistcal package for data analysis. (Vieira *et al.*, 1983).

The estimated values were then used to develop rainfall erosivity maps with the ArcGIS 10·1 software (ESRI, 2013).

The soil erodibility map was obtained from the soil map of Goiás and the FD, as well as from literature review of *K* values for soil classes present or similar to the ones found in the region (Nogueira, 2000; Mannigel *et al.*, 2002; Bertoni & Lombardi Neto, 2005; Galdino, 2012). The vector-format Goiás and FD soil maps at 1:1,000,000 scale were obtained from the homepage of *Sistema Estadual de Estatística e Informações Geográficas do Governo de Goiás* (SIEG-GO).

The mapping of the topography factor (*LS* factor) was performed using a digital elevation model (DEM) obtained by the NASA Shuttle Radar Topography Mission Global 1 arc second (SRTMGL1). The DEM cell size was approximately 30 m. The SRTMGL1 DEM was obtained from the Land Processes Distributed Active Archive Center (LP DAAC) through the site "http://lpdaac.usgs.gov/."

The USLE and RUSLE were conceived to calculate average soil losses in agricultural areas. Several modifications were applied in the model to obtain automatic estimates of soil loss on complex slopes, such as those found in the hydrographic basins, based on GIS and DEM. One of the biggest limitations of the USLE/RUSLE method is the identification of areas where the dominant process is deposition rather than erosion. This is critical to identifying the beginning and the end of a long slope, which is directly related to the *L* estimation. Hickey *et al.* (1994) and Hickey (2000) developed a mechanism, called the cutoff slope angle, to define the depositional areas. The calculation of *L* from upslope drainage area has the potential of accounting for the flow divergence and convergence patterns (van Remortel *et al.*, 2004).

Considering the water flow carrying sediments from the land surface, the speed will considerably slow in certain places, and the soil deposition will occur. The value of the limiting slope angle proposed by van Remortel *et al.* (2004) is 0·5 for slopes ≥ 5%. For slopes < 5%, the proposed

Table I. Rainfall distribution in Goiás State and the Federal District

Month	Average (mm)	Standard deviation (mm)	Percentage
January	280·6	37·1	18·30
February	214·3	26·1	13·98
March	218·7	24·3	14·26
April	98·7	15·9	6·43
May	30·5	10·7	1·99
June	9·0	5·5	0·59
July	5·7	3·0	0·37
August	12·9	6·0	0·84
September	46·3	12·9	3·02
October	129·0	15·5	8·41
November	210·6	19·3	13·73
December	277·1	32·4	18·07
Total	1,533·6	147·2	100

limiting slope angle is 0·7 because, in general, deposition is likely to occur on smaller slope gradients.

In this paper, the *LS* factors of RUSLE for Goiás, the FD, and neighboring areas were obtained through the program "lsfac_c.exe" (version 4), developed in C++ language (van Remortel *et al.*, 2004) with the DEM as entered data.

The cultivated pastures maps of Goiás and FD from 2009 at the 1:250,000 scale were obtained through Landsat Thematic Mapper image interpretation, complemented with municipality-based data obtained by in 2006 (IBGE, 2013). The use of census data was particularly important for separating cultivated and natural pastures, which is not always evident using only spectral data.

For mapping degraded and non-degraded pastures, the normalized difference vegetation index (NDVI) from the moderate resolution imaging spectroradiometer for the period between January 2002 and December 2012 was used. The NDVI time series was processed

according to the following sequence: pre-processing, calculation of time series alteration rate, and linear regression analysis.

The linear regression analysis was used to simulate the tendency of change of each grid. Stow *et al.* (2003) adopted this method to simulate the rate of vegetation greenness change. For each pixel, the maximum annual NDVI linear tendency during the period of the study was estimated through the application of ordinary regression of minimum squares. The NDVI slope (Equation 5) is the slope coefficient of the regression line adjusted for each pixel. Positive NDVI slopes indicate that vegetation is in process of regeneration, while negative NDVI slopes indicate the presence of some degradation process in pasture (Liu *et al.*, 2010).

$$NDVI - slope = \frac{\sum_{i=1}^{n} (Y_i - \overline{Y}) \ (Y_{NDVI_i} - \overline{Y_{NDVI}})}{\sum_{i=1}^{n} (Y_i - \overline{Y})^2} \qquad (5)$$

where $n = 10$ due to the utilization of one 10-year series of NDVI data (2002–2012); $i =$ year 1 for 2002, year 2 for 2003, through year 10 for 2012; and $Y_{NDVI_i}{}^{i} =$ the NDVI maximum value on year i.

The slope coefficients were used to evaluate the NDVI changes throughout the time series. Indications of degradation are observed when the NDVI slope is less than -0.001 (Liu *et al.*, 2010). Thus, two classes indicative of degradation processes were established for cultivated pastures. Pastures were classified as degraded when the NDVI slope is less than -0.001 and non-degraded for NDVI-slopes greater than or equal to -0.001.

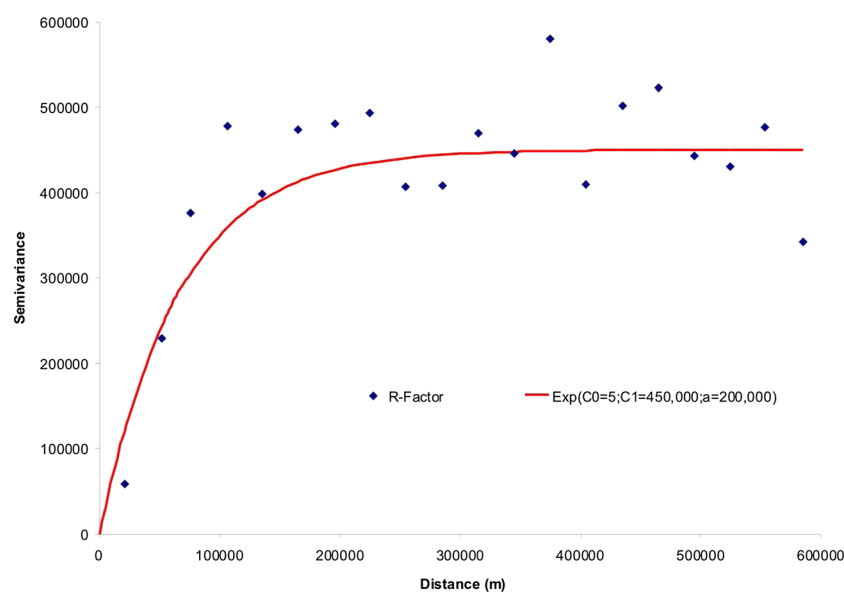The *C* factor for degraded pastures used in this study was proposed by Vázquez-Fernández *et al.* (1996): non-

Figure 2. Semivariograms of rainfall erosivity (*R* factor) (in MJ mm ha^{-1} h^{-1} y^{-1}) fitted by exponential model and correspond to the parameters C$_0$, C$_1$, and a. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr

degraded pasture = 0·0080 and degraded pasture = 0·0300. These values are similar to *C* factor estimates obtained by Galdino (2012) using RUSLE for pasture areas of Brazilian Cerrado planted in Arenosols. These estimates were 0·0070 for non-degraded pasture and 0·0330 for degraded pasture. Pasture areas considered degraded by Galdino (2012) were, in average, more than 8 years old, 41% of soil was uncovered and 10% of vegetation was weeds. On the other hand, non-degraded pasture was 2 to 4 years old, presented 8% uncovered soil and 2% covered with weeds.

For mapping *P* factor in degraded pastures, the maximum terrace horizontal interval (*HI*) was estimated and subsequently *P* factor values were used as proposed by RUSLE for different *HI* (Renard *et al.*, 1997).

The Natural Resouces Conservation Service of the United States Department of Agriculture (USDA-NRCS, 2011) recommends an empiric equation (Equation 6) for calculating the vertical spacing between terraces.

$$VI = XS + Y \qquad (6)$$

where *VI* is the maximum vertical spacing between terrace channels, ft; *S* is the slope, in percentage; *X* is a variable with values from 0·4 to 0·8 for graded terraces and 0·8 for levels terraces; and *Y* is a variable with values from 1·0 to 4·0 ft, as influenced by soil erodibility, cropping systems, and crop management practices. For this study, the value for *X* was 0·8 (levels terraces), and *Y* was 2·5, value recommended when one of the factors is favorable and the other is unfavorable (USDA-NRCS, 2011).

The maximum terrace *HI* in ft was calculated by Equation 7:

$$HI = VI * \left(\frac{100}{S} \right) \qquad (7)$$

Raster files of slope (in percentage) and of vertical spacing (*VI*) and horizontal (*HI*) between terraces were generated using SRTM DEM in ft and ArcGIS. Subsequently, the *HI* estimates were reclassified into six value classes of *P* factor
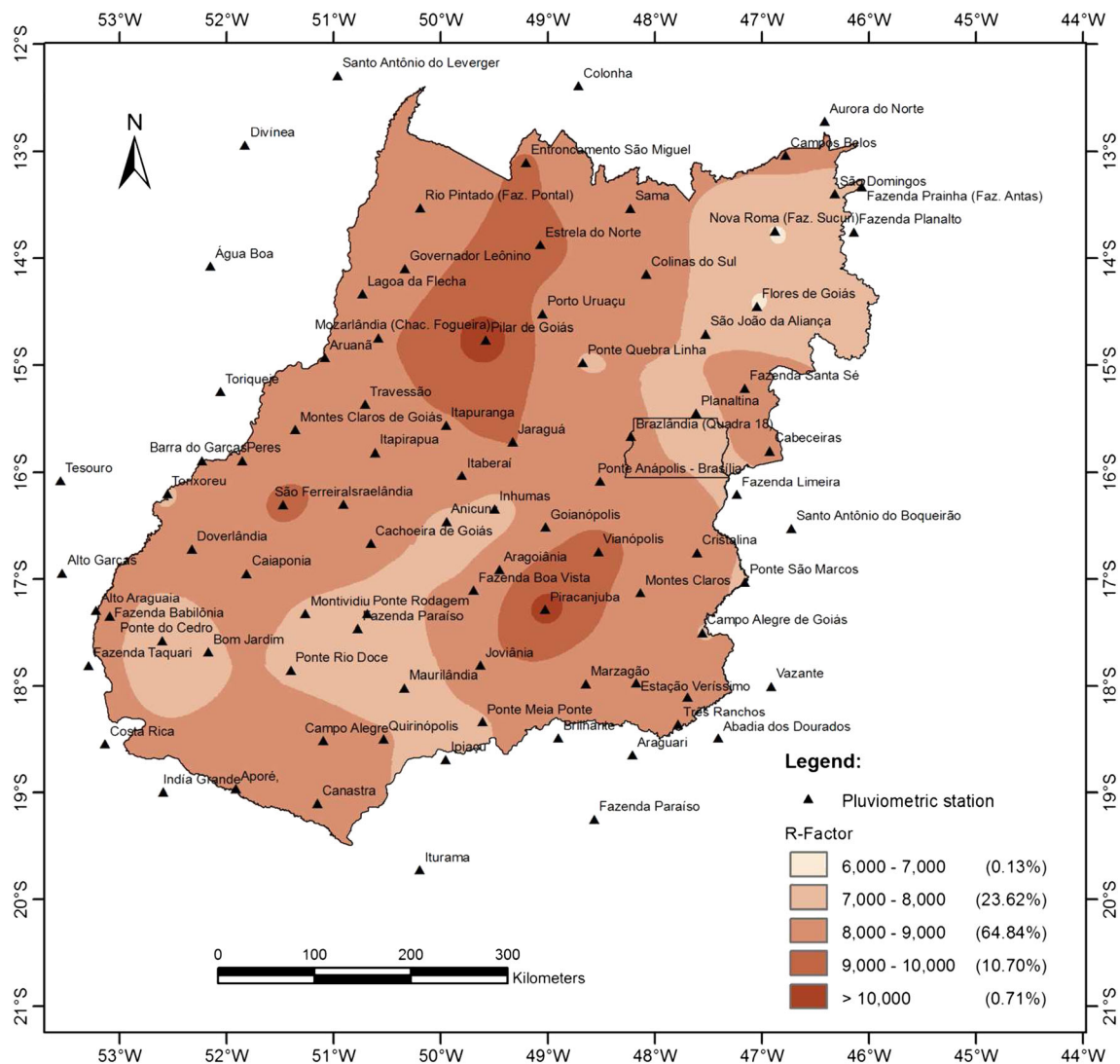

Figure 3. Location of pluviometric stations and spatial distribution of rainfall erosivity (*R* factor) (in MJ mm ha^{-1} h^{-1} y^{-1}) in Goiás and the Federal District. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr

(Renard *et al.*, 1997): for $HI < 110$ ft, $P = 0.5$; for HI between 110 and 140 ft, $P = 0.6$; for HI between 140 e 180 ft, $P = 0.7$; for HI between 180 and 225 ft, $P = 0.8$; for HI between 225 and 300 ft, $P = 0.9$; and for $HI > 300$ ft, $P = 1$.

Terrain with slopes > 20% is unsuitable for the implementation of terraces (Bertoni & Lombardi Neto, 2005). An option for these areas is the maintenance or regrowth of native vegetation. The *C* factor for areas covered with Cerrado's native vegetation is 0·0001 (Vázquez-Fernández *et al.*, 1996).

Evaluation of Soil Loss in Different Scenarios

Soil losses in areas covered with pastures in Goiás and the FD were estimated for four scenarios based on differences in management and conservation practices: (A) degraded and non-degraded pastures (in 2009) without terraces; (B) degraded pastures without terraces (critical condition); (C) non-degraded pastures without terraces; and (D) non-degraded pastures, planted with terraces when slopes are ≤ 20%, and for areas with pasture in 2009 and with slopes > 20% would be destined to regrowth of native vegetation.

Most of the cultivated pastures, mainly those presenting high levels of degradation, were most likely not planted in terraces. Consequently, scenario A was considered to have the highest level of soil loss. Scenario B simulates a critical condition of erosion that occurs when pastures are poorly managed, with excess animal load and without corrections to soil fertility. Consequently, there is an increase of bare soil exposed in the pasture fields. Scenarios C and D represent the desirable and optimum conditions, respectively.

RESULTS AND DISCUSSION

Rainfall Erosivity (R Factor)

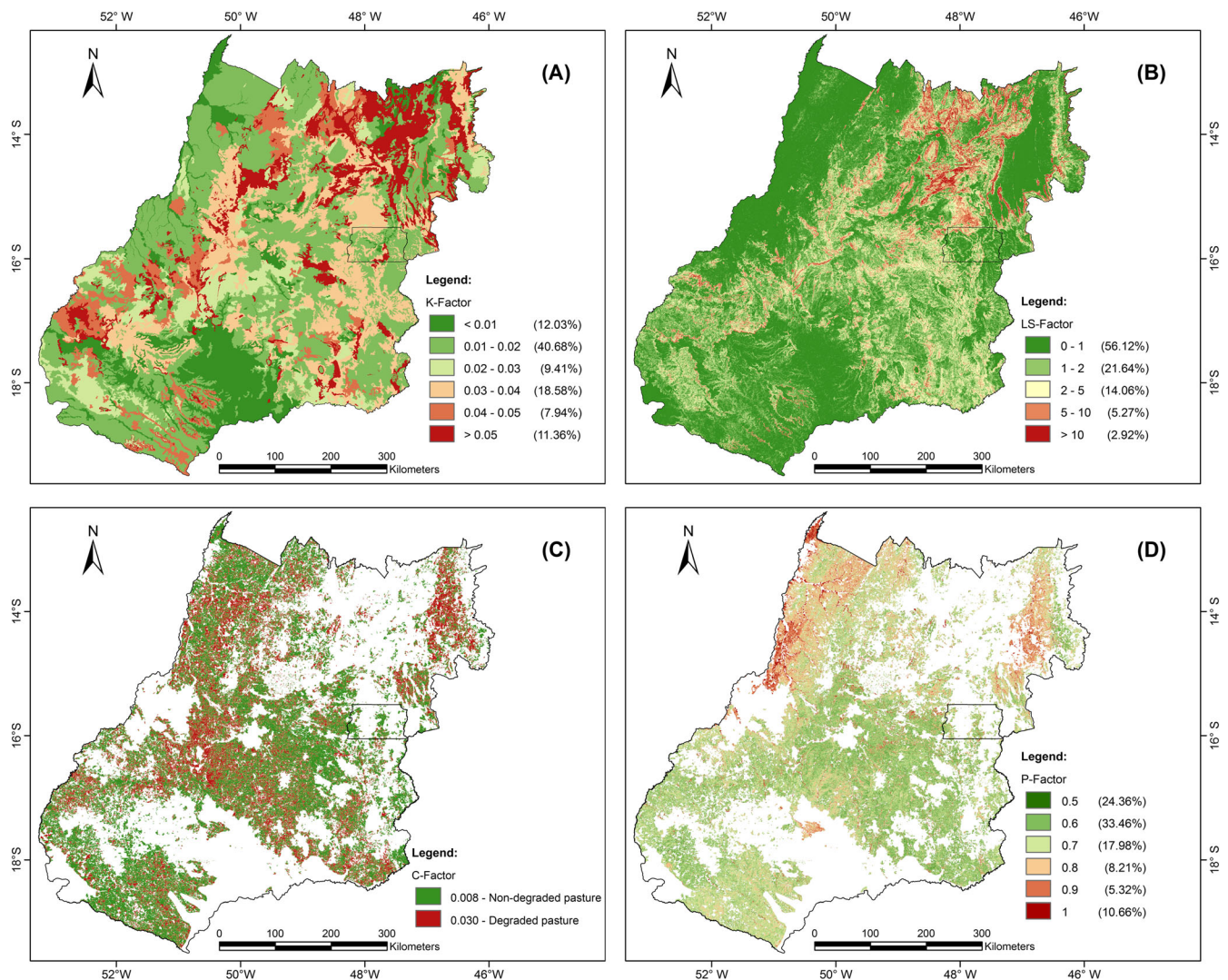The annual average precipitation is ~1,500 mm. Between October and March, 80% of annual rainfall is concentrated

Figure 4. Soil loss estimation factors: (A) distribution of soil erodibility (in Mg h MJ^{-1} mm^{-1}); (B) topographic factor (*LS*); (C) degraded and non-degraded pasturelands in Goiás State and Federal District in year of 2009 (*C* factor); and (D) *P* factor distribution in pasture lands planted in terraces (D scenario). This figure is available in colour online at wileyonlinelibrary.com/journal/ldr

(Table I). The descriptive statistics of rainfall erosivity indicated little variation. The average R factor was 8,324 MJ mm ha^{-1} h^{-1} y^{-1}, with a coefficient of variation (CV) of 8%. According to Warrick and Nielsen (1980), values of $CV < 12\%$, 12–60%, and $> 60\%$ are considered low, average, and high, respectively. Therefore, the erosivity CV the study area is low. To values asymmetry of 0·777 and a kurtosis of 2·374, in terms KS test, it is normal distribution (for significance level 0·05, $n = 88$, and KS value of 0·073).

The average R factor is the expected for climatic conditions of Brazilian territory. Silva (2004) estimated that rainfall erosivity in Brazil varies from 3,116 to 20,035 MJ mm ha^{-1} h^{-1} y^{-1}. Similar results were also obtained by Oliveira *et al.* (2012), finding erosivity values in Brazil from 1,672 to 22,452 MJ mm ha^{-1} h^{-1} y^{-1}, and average ($\pm$ sd) of 8,403 $\pm$ 4,090 MJ mm ha^{-1} h^{-1} y^{-1}.

The erosivity was also analyzed from the geostatistics point-of-view to identify spatial variability. The semivariance was calculated and adjusted the semivariogram (Figure 2). Semivariogram was fitted with exponential model and validated by jack knifing technique according to Vieira *et al.* (2010). The jack knifing parameters analyzed were the intercept (0·62), slope (0·26), and correlation coefficient (0·48) between measured and estimated values and the mean ($-0·0026$) and variance (1·054) of the errors calculated by the difference between measured and estimated values, divided by the square root of the estimation variances. The ideal numbers of neighbors for use in the kriging estimation was 12 also studied using the jack knifing procedure. A strong spatial dependency, with a range of 200,000 m, was found ($GD = 99·99\%$) (Zimback, 2001).

The semivariogram adjustment parameters were then utilized by the ordinary kriging interpolation, and the interpolated erosivity values were generated in a regular grid of 100×100 m. The location of pluviometric stations and the spatial distribution of rainfall erosivity (R factor) in Goiás and the FD, obtained from the Kriging interpolation, are shown in Figure 3.

The rainfall erosivity map for the entire Goiás State and FD exhibited an annual average of 8,361 MJ mm ha^{-1} h^{-1} y^{-1}

and a range from 6,882 to 10,744 MJ mm ha^{-1} h^{-1} y^{-1}. A well-defined spatial variability was observed, where locations are more similar to neighboring values than more distant values, supporting the hypothesis that the R factor does not occur randomly in the space. A tendency for rainfall erosivity to be higher in the northwest and southeast of Goiás State and lower in the northeast and southwest of Goiás is also noted.

The R factor in pasturelands displayed an annual average of 8,449 MJ mm ha^{-1} h^{-1} y^{-1}, ranging from 6,882 to 10,744 MJ mm ha^{-1} h^{-1} y^{-1}. The average value of rainfall erosivity was slightly higher (125 MJ mm ha^{-1} h^{-1} y^{-1}) than that obtained for the whole area of Goiás and the FD.

Spatial Distribution of K, LS, C, and P Factors

The spatial distribution of K, LS, C, and P factors is shown in Figure 4.

Table II presents the distribution of major soil classes under pasturelands, according to EMBRAPA (2006) and IUSS Working Group WRB (2006). The World Reference Base for Soil Resources (WRB) is the soil classification system endorsed by the International Union of Soil Science (IUSS).

In 2009, 45% of pastures were cultivated in Ferralsols (IUSS Working Group WRB, 2006), corresponding to the Latossolo Vermelho and Latossolo Vermelho-Amarelo in the Brazilian soil classification system (EMBRAPA, 2006). The other two important classes are the Acrisols

Table III. Distribution of classes of slope in pasturelands

Class	Slope (%)	Area (ha)	(%)	Accumulated (%)
Flat	0–3	3,520,126	25·25	25·2
Gently undulating	3–8	6,288,259	45·10	70·4
Undulating	8–20	3,639,617	26·11	96·5
Strongly undulating	20–45	447,164	3·21	99·7
Hilly	45–75	45,492	0·33	100
Steep	>75	1,120	0·01	100

Table II. Distribution of major soil classes of pasturelands, according to EMBRAPA (2006) and IUSS Working Group WRB (2006) and factor K value (Mg h MJ^{-1} mm^{-1})

Embrapa	WRB	K factor	Area (ha)	%
Argissolo Vermelho	Acrisols	0·0228*	270,918	1·91
Argissolo Vermelho-Amarelo	Acrisols	0·0466*	2,445,605	17·28
Cambissolo Háplico	Cambisols	0·0346*	2,555,407	18·05
Latossolo Vermelho	Ferralsols	0·0193**	3,402,691	24·04
Latossolo Vermelho-Amarelo	Ferralsols	0·0145**	2,977,933	21·04
Neossolo Litólico	Leptsols	0·0551**	873,106	6·17
Neossolo Quartzarênico órtico	Arenosols	0·0292***	655,802	4·63
Others	—		972,872	6·87

Source:
*Mannigel *et al.* (2002);
**Nogueira (2000);
***Galdino (2012).

(Argissolos) and Cambisols (Cambissolos), which represent 37% of the cultivated pastures.

The estimated K values (Figure 4A) varied from 0·0047 to 0·0551 Mg h MJ^{-1} mm^{-1}, with an average of 0·0266 Mg h MJ^{-1} mm^{-1}. The average value for soil erodibility was only 2·6%, lower than the other land cover types in Goiás and the FD. In 62·12% of Goiás and FD lands, the K factor was lower than 0·03, indicating that most of the soils in this region are resistant to soil erosion by water.

The distribution of slope classes (Table III) showed that almost half of the pastures were not cultivated on steep slopes (3 to 8% slope), and 81% of pastures were exposed to slopes < 8%. Only 1·7% of pastures (237·119 ha) were cultivated on slopes > 20%. Planting in terraces in these areas was limited because of the steep slope (Bertoni & Lombardi Neto, 2005).

The average slope of pasturelands was 6·8% and median was 5·27%. For the entire area of Goiás and the FD, the average slope is 8·5%, indicating that the large majority of steeper areas are covered with other types of vegetation.

The distribution of the LS factor is shown in Figure 4B. The LS values varied from 0 to 114, with an average value of 1·80. Approximately 56% of the surface has an LS values of < 1; approximately 78% of the area has an LS value of < 2; and only in 2·92% LS of > 10.

The RUSLE LS factor in cultivated pasturelands varied from zero to 95, with an average of 1·25. Approximately 60% presented LS < 1. In approximately 84% of the area, the LS values were < 2, and only 0·7% of the area had LS > 10. The average LS of pasturelands was 31% less than the average LS for Goiás and the FD, most likely because most of the pasturelands are not cultivated on steeper slopes, leading to a reduction in the steepness of the slope factor (S factor) and consequently in the topographic factor (LS).

The distribution of pasturelands, both with and without indications of degradation (based on Liu *et al.* (2010) method), together with C factor of USLE, is shown in
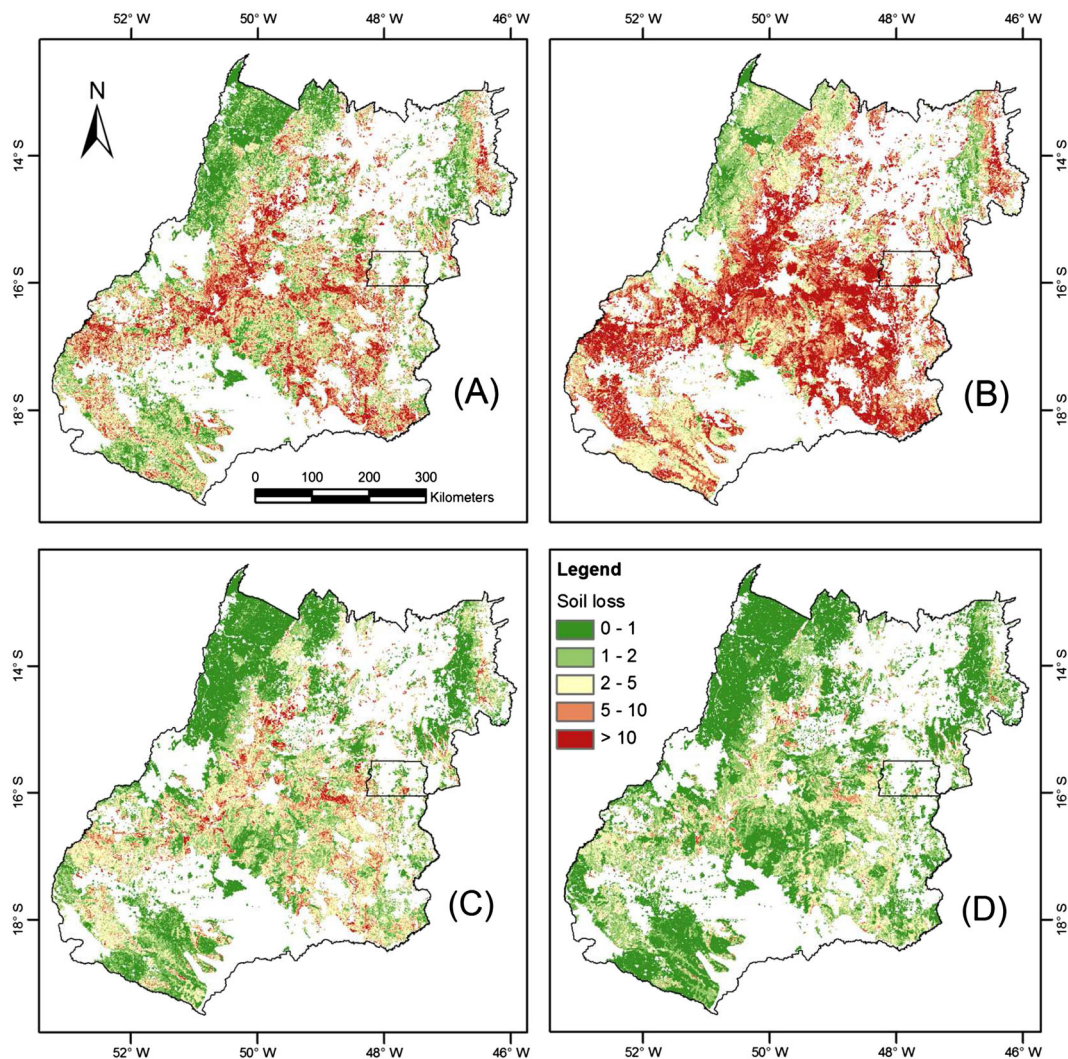

Figure 5. Distribution of soil losses (in Mg ha^{-1} y^{-1}) for the following scenarios: (A) land use and management of pastures in 2009 without terraces; (B) degraded pastures without terraces; (C) non-degraded pastures without terraces; (D) non-degraded pastures, planted in terraces in areas with slopes ≤ 20%, and left for regrowth of native vegetation in areas with slopes > 20%. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr

Table IV. Estimation of soil losses (in Mg ha^{-1}y^{-1}) in pasturelands with different land uses and management and soil conservation practices (scenarios)

Scenario	Minimum	Average	Maximum	Standard deviation
A – Use and management of pastures without terraces	0	5·119	580·8	10·19
B – Degraded pastures without terraces	0	9·194	738·0	14·61
C – Non-degraded pastures without terraces	0	2·452	196·8	3·89
D – Non-degraded pastures with terraces	0	1·137	41·9	1·07

Figure 4C. Pastures covered 14,154,338 ha of the study area (41% of Goiás and the FD). It was estimated that 9,092,747 ha (64% of pasturelands) were not degraded. The remaining areas of pasture (5,061,591 ha or 36% of study area) were degraded to some degree.

Distribution of the *P* factor in pasture lands of Goiás and FD, planted in terraces (D scenario), is shown in Figure 4D.

When leveled terraces were used in pasture land with slopes ≤20%, the *P* factor was, in average, 0·67. In about half of cultivated pasture areas of Goiás and FD (57·82%), the *P* factor value was between 0·5 and 0·6, highlighting the importance of this conservation practice for reducing erosion. The slope in these areas varied between 4·25% and 20%.

Soil Loss in Different Scenarios

The results of the evaluation of different land use scenarios and management and conservation practices in Goiás and FD pastures are shown in Figure 5 and Tables IV and V.

When simulating a critical condition where pastures are poorly managed (degraded) and without terraces (scenario B), the average soil loss would increase by 4·075 Mg ha^{-1} y^{-1}, corresponding to an increase of 80% in comparison with the current estimation of soil loss (scenario A). Therefore, it becomes evident that inadequate management of pastures by cattle raisers will result in even larger rates of soil erosion. The importance of pasture restoration is emphasized in the analysis of scenarios A and C. If the degraded pasture areas present in 2009 were restored by correcting soil fertility and acidity, by replanting the grass (pasture reform), and experienced proper management (scenario C), the average soil loss would be as low as 2·667 Mg ha^{-1} y^{-1}, the equivalent of a 52% reduction in soil erosion compared with scenario A.

In a scenario where all pastures are well managed and terraces are implemented, excluding areas with slopes > 20% (destined to native vegetation), the average soil loss would be lowered to 3·982 Mg ha^{-1} y^{-1}, which corresponds to a reduction of 78% of the erosion rate compared with scenario A.

The importance of adequate pasture management and use of soil conservation practices for the reduction of soil erosion rates becomes extremely evident when comparing soil losses between scenario B (critical) and D (ideal). Going from the ideal to the critical scenario, the average soil loss would increase by 8·057 Mg ha^{-1} y^{-1}, the equivalent of an erosion rate increase of 709%.

The values of soil loss class distribution in pasture areas of Goiás and FD in various scenarios (Table V) explain the soil loss averages presented in Table IV, excepting when erosion rate is zero. In this case, soil loss class distribution was the same for all four scenarios, probably because in these areas, the *LS* factor estimate was null.

The correlation of factors *R*, *K*, *LS*, and *C* with soil loss estimates in 2009 were, respectively, 0·05, 0·34, 0·61, and 0·34, indicating that the topographic factor was the one that better explained the variation of soil loss in pasture areas of Goias and FD ($r^2 = 37·7\%$).

Estimates of soil loss obtained by empirical models, like USLE/RUSLE, should be qualitative and quantitatively analyzed. However, a large magnitude of difference in the rates of soil loss among different scenarios showed the importance of recovering degraded pasture and implementing soil conservation practices, such as terracing, to control soil erosion by water in the Brazilian Cerrado. This fact reinforces the importance of actions focused on the recovery of degraded pasture, such as the ABC Program.

Table V. Distribution of soil loss classes in pasture areas of Goiás and Federal District (in hectares and percentage) for the various scenarios

	Erosion rates					
	Without (soil loss equals zero)		Hight (10 and 40 Mg ha^{-1} y^{-1})		Very hight (>40 Mg ha^{-1} y^{-1})	
Scenario	ha	%	ha	%	ha	%
A	606,338	4·29	1,603,217	11·33	182,313	1·29
B	606,338	4·29	3,322,132	23·49	430,590	3·04
C	606,338	4·29	2,851,565	20·16	110,954	0·78
D	606,338	4·29	28,538	0·20	10	0

CONCLUSIONS

The relief was the factor that most influenced in the variation of soil loss in 2009 in the pasture areas of Goiás and FD. The analysis of the effects of different scenarios of land use and management and conservation practices on the erosion rates in pasture areas of Goiás and the FD highlight the importance of actions focused on the recovery of degraded pasture in the Cerrado biome. An increase of 709% in the soil erosion by water rate can be expected if pastures are poorly managed, and no conservation practices are adopted (the critical scenario) in comparison with a condition in which pastures are well managed, terraces are used and steeper slopes are saved for natural vegetation (the ideal scenario). The adoption of good management practices will improve the quality of the degraded pastures, increase productivity of livestock, and reduce the pressure to open new areas for food and energy crop expansion. The modeling of soil loss using geotechnologies has been demonstrated to be an important tool for land use planning and supporting public policies for the sustainable use of natural resources.

ACKNOWLEDGEMENTS

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