

Adjustment of the conservation practices factor calculation in estimating soil loss

Adequação do cálculo do fator de práticas conservacionistas na estimativa da perda de solo

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Abstract

Agricultural intensification without conservation practices exacerbates soil loss, affecting productivity and ecosystems. In the Sorocabuçu River Basin, this issue is intensified by agricultural activities. The RUSLE method, used to estimate soil loss, has limitations in calculating the P factor, as it applies the default value of 1 in the absence of conservation practices, disregarding the area's actual conditions. This study proposes adjusting the calculation of the P factor based on data from the Brazilian Agricultural Census (IBGE), including information on conservation practices and property sizes. In 2006, 38.72% of the 935 establishments adopted conservation practices, with an average soil loss of 11.42 t/ha.year; using P equal to 1, this average was 12.0 t/ha.year. In 2017, only 12.5% of the 720 establishments followed such practices, resulting in an average soil loss of 11.44 t/ha.year, while P equal to 1 indicated 11.74 t/ha.year. Contour planting was predominant, adopted by 89% and 88% of properties in 2006 and 2017, respectively. Areas with higher soil loss were associated with large-scale farming without conservation measures. Despite minor numerical differences, the adjusted model proved more efficient, accurately representing the basin's reality. This study highlights the importance of incorporating conservation practices in the P factor calculation for better soil loss estimation and preserving soil and ecosystems in the Sorocabuçu River Basin, with potential applicability to other study areas.

Keywords:

Water erosion, Soil conservation, RUSLE, P factor.

Resumo

A intensificação agrícola sem práticas conservacionistas agrava a perda de solo, afetando a produtividade e os ecossistemas. Na Bacia do Rio Sorocabuçu, essa situação é intensificada pela atividade agrícola. O método RUSLE, usado para estimar a perda de solo, apresenta limitações no cálculo do fator P ao aplicar o valor padrão de 1 na ausência de práticas conservacionistas, desconsiderando as condições reais da área. Este estudo propõe ajustar o cálculo do fator P com

base em dados do Censo Agropecuário do IBGE, incluindo informações sobre práticas conservacionistas e o tamanho das propriedades. Em 2006, 38,72% dos 935 estabelecimentos adotavam práticas conservacionistas, com uma perda média de solo de 11,42 t/ha.ano; no P igual a 1, essa média foi de 12,0 t/ha.ano. Em 2017, apenas 12,5% dos 720 estabelecimentos seguiam essas práticas, resultando em uma perda média de 11,44 t/ha.ano, enquanto o P igual a 1 indicava 11,74 t/ha.ano. O plantio em curvas de nível foi predominante, adotado por 89% e 88% das propriedades em 2006 e 2017, respectivamente. As áreas com maior perda de solo foram associadas a grandes cultivos sem medidas conservacionistas. Apesar de diferenças numéricas pouco significativas, o modelo ajustado se mostrou mais eficiente, representando a realidade da bacia de forma coerente. Este trabalho destaca a importância de considerar práticas conservacionistas no cálculo do fator P para uma melhor estimativa da perda de solo e para a preservação do solo e dos ecossistemas na Bacia do Rio Sorocabuçu, podendo ser replicado em outras áreas de estudo.

Palavras-chave:

Erosão hídrica, Conservação dos solos, RUSLE, Fator P.

I. INTRODUCTION

The impacts of anthropogenic activities, such as deforestation and the lack of conservation techniques, compromise environmental quality (Santos et al., 2024). They generate problems such as loss of soil fertility, siltation of rivers, and damage to public health (FAO, 2017; Pendrill et al., 2022). It is estimated that a third of global soils are already in decline, with water erosion as the main cause of this degradation (Kassam et al., 2020; Meena et al., 2020).

Erosion transports nutrients and sediments into water bodies, contaminating ecosystems and aggravating problems such as the formation of gullies and reduced water quality (Dodd; Sharpley, 2016; Guo et al., 2023). In agricultural areas, the lack of proper management intensifies soil loss and sedimentation, which underscores the need for efficient planning and the adoption of conservation practices (Porto et al., 2014; Gaubi et al., 2017; Xiong; Leng; Tang, 2023).

The RUSLE (Revised Universal Soil Loss Equation) is widely used to estimate average annual soil losses as a function of factors such as rainfall erosivity (R), soil erodibility (K), topographic factor (LS), soil cover and management (C), and conservation practices (P). The evolution of Geographic Information Systems (GIS) and integration with geospatial data has increased the model's accuracy. Thus, it can be applied at different scales and contexts (Pandey et al., 2021).

In particular, the P factor reflects the effectiveness of conservation practices in reducing erosion. Therefore, it is crucial to consider agricultural practices that help to conserve soils, such as no-till with vegetation cover, contour planting, and terrace cultivation (Wischmeier; Smith, 1978; Gashaw et al., 2020). However, few

studies use the P factor in the RUSLE due to the lack of comprehensive information on conservation practices and specific agricultural techniques and the high cost of field monitoring (Guerra et al., 2014; Xiong et al., 2019). When these practices are not applied, the P factor is assigned a value of 1, representing the most critical scenario. Meanwhile, the lower its value, the more effective the given practice is (Bertoni; Lombardi Neto, 2005).

Furthermore, according to the literature, a value of 1 is also assigned to areas where it is difficult to identify the presence of conservation practices and which ones are used, resulting in an overestimation of soil loss estimates. For a more precise attribution, it is crucial to map the agricultural areas that adopt these practices and characterize their specific typologies (Didoné et al., 2021; Ebabu, 2022; Martins; Kaleita; Gelder, 2021).

Nevertheless, at larger scales, this identification faces significant challenges. Either due to the high cost of analysis with high-resolution satellite images, or the difficulty of making direct observations in situ (Zhu, 2015; Xiong et al., 2019; Santos et al., 2025). Once conservation practices have been identified, we can apply the P values available in the literature, allowing for a more detailed and reliable analysis of these practices' contribution to soil conservation (Ebabu, 2022).

Thus, few studies address this factor's quantitative impact on a regional scale, and they are based predominantly on secondary data. In many studies, the absence of planned information leads to using the C factor and existing literature (Borrelli et al., 2020). Therefore, current studies present specific priority soil conservation measures (Taye et al., 2018) and technical field investigations in plateau regions, which restricts the P factor's applicability and effectiveness in the RUSLE.

This study aimed to refine soil loss estimates using the Revised Universal Soil Loss Equation (RUSLE) in the Sorocabuçu River Basin (SRB). We analyzed data from the 2006 and 2017 Brazilian Agricultural Censuses to identify conservation agricultural practices and determine P factor values, comparing the estimates proposed here with $P = 1$. This approach seeks to fill gaps left by studies focused on individual soil conservation measures, often on isolated properties, as Xiong et al. (2019) pointed out. Or when it is not known which practices are being effectively applied, resulting in the adoption of a value of 1 for the P factor, as in Arantes (2023).

Moreover, there is a lack of feasible alternatives when the practices used are not known or implemented in an accessible way and at no additional cost to farmers. By using data from the Brazilian Agricultural Census (IBGE), the study improves the P factor's applicability in the RUSLE and contributes to more effective soil conservation management. Using this data offers a robust way of making federal government information more useful and visible, highlighting its relevance for environmental analysis at regional scales. Thus, this study aimed

to provide more accurate soil loss estimates, offering a model that can be replicated in other areas with an eye to practical reality and sustainable soil management

II. MATERIALS AND METHODS

Study area

The SRB is part of the Itupararanga Environmental Preservation Area (EPA) and protects important natural resources. It was created by State Law No. 10,100 of 1998. This EPA includes the Itupararanga reservoir, built in 1912 to supply cities such as Ibiúna, Sorocaba, and Votorantim and generate electricity. The SRB comprises the Sorocabuçu, Sorocamirim, and Una rivers (Moraes et al., 2014; Gomes et al., 2020) (Figure 1).

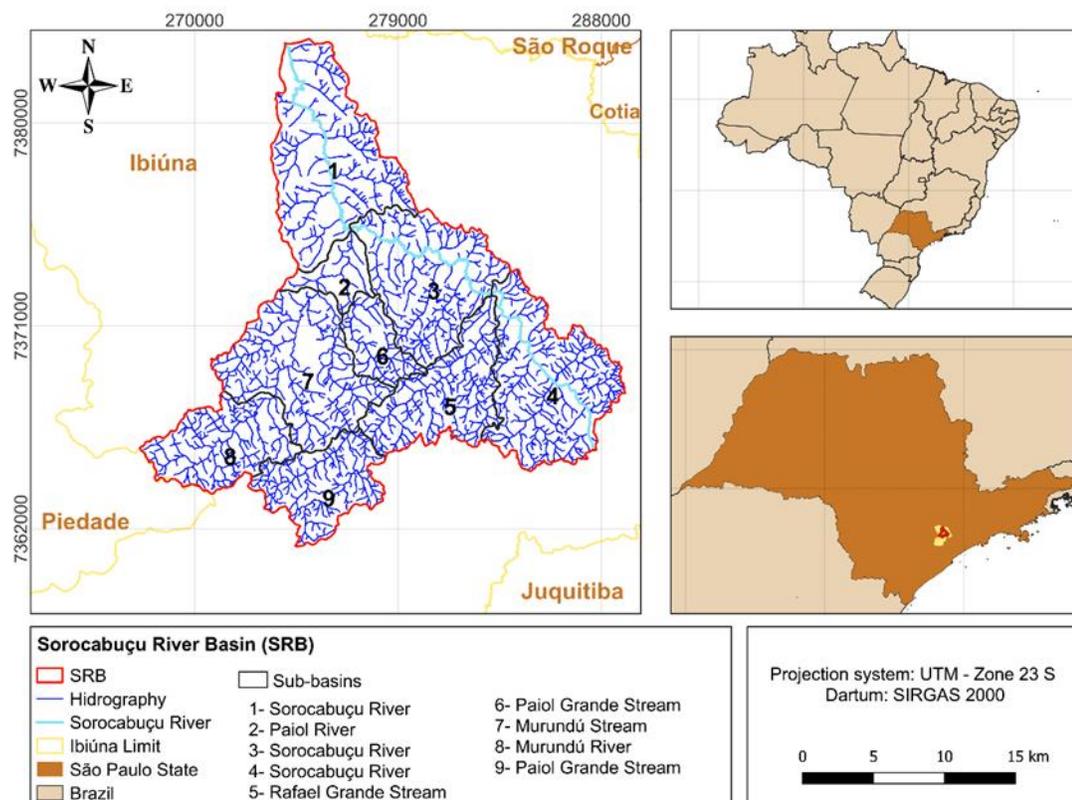


Figure 1 – SRB location map. Source: Authors.

Covering 202.67 km², the basin is dominated by agricultural and industrial activities, including wood processing and vegetable and fruit production (Maia; Lourenço, 2020; Vasques et al., 2021; IBGE, 2022; Arantes et al., 2024). The climate is temperate mountain (CWB), with annual rainfall of 1,330 mm and temperatures between 14.2 °C and 21.3 °C. The predominant soils are red-yellow latosols with gravel.

The vegetation is part of the Atlantic Forest biome, characterized by ombrophilous and seasonal forests with rich biodiversity, including species from the Myrtaceae, Fabaceae, and Lauraceae families (Andreoti, 2012; IBGE, 2019).

2.2. Methodological procedures

This study used data from the 2006 and 2017 Agricultural Censuses of the Brazilian Institute of Geography and Statistics (IBGE) to investigate the adoption of conservation practices and develop a method for calculating the P factor's values in the river basin. The Agricultural Census, conducted since 1920, is the main source of information on the structure and dynamics of Brazilian agriculture. Although its periodicity is more than ten years, which can be a challenge for more detailed temporal analysis, it is the most comprehensive and reliable database available, allowing the identification of large-scale patterns and trends.

This study chose 2006 and 2017 due to the need for a comparative analysis considering the availability of georeferenced data. Although the census has been conducted at different times since its creation, only the most recent editions have systematically included spatial information that enables more precise geographical analysis. Thus, the temporal analysis was based on the 2006 and 2017 censuses. Furthermore, the free availability of this data and its credibility as an official source reinforce its suitability for research.

Then, we applied the Universal Soil Loss Equation (USLE), according to Equation 1. We subdivided the area into sub-basins to ensure greater precision, which provided more detailed results. These cut-outs come from the Geoprocessing and Environmental Mathematical Modeling Laboratory (LABGEMM).

$$A = R \times K \times LS \times C \times P \quad (1)$$

Where:

A = average annual soil erosion rate per unit area (Tons/hectares.year);

R = average annual rainfall erosivity factor ($\text{MJ}\cdot\text{mm}\cdot\text{ha}^{-1}\cdot\text{h}^{-1}\cdot\text{year}^{-1}$);

K = soil erodibility factor ($\text{t}\cdot\text{ha}\cdot\text{MJ}^{-1}\cdot\text{mm}^{-1}$);

LS = topographic factor, where L is the length of the slope and S is the declivity (dimensionless);

C = soil cover factor (dimensionless);

P = conservation practices factor (dimensionless).

Figure 2 shows the methodological procedures adopted to calculate each RUSLE factor. It highlights the development of the methodological approach for the P factor, which considers conservation practices in the region.

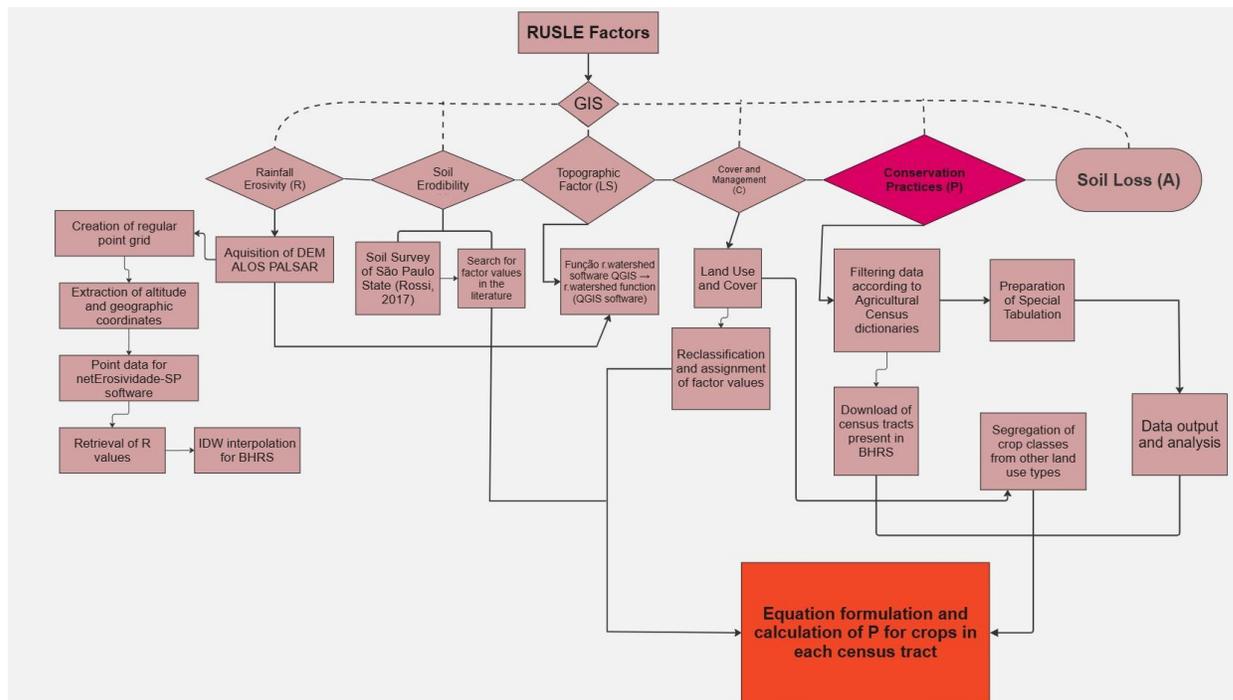


Figure 2 – Methodological flowchart. Source: Authors.

The erosivity factor expresses the rainfall's capacity to cause erosion in unprotected soil (Bertoni; Lombardi Neto, 2005). We used a systematic sampling approach for the calculation, following Yamamoto and Landim's (2013) methodology.

We created a regular mesh with points distributed throughout the river basin using the “Create Fishnet” tool in ARCMAP 10.8. Each point was associated with metric coordinates and altitude from the digital elevation model. At this stage, we used the netErosivity SP system. This system uses an artificial neural network to estimate the R factor based on latitude, longitude, and altitude data, overcoming the lack of rainfall records (Moreira, 2006). We used the Inverse Distance Weighted (IDW) interpolation method to estimate the Erosivity values where no samples were taken, with an exponent of 2 (recommended in the literature), which provided reliable results (Landim, 1998; Bărbulescu; Şerban; Indrean, 2021).

Soil erodibility, represented by the K factor, is the natural tendency of some soils to erode more easily than others, even when other conditions are equal (Ahaneku et al., 2024). The K factor was determined based on

the pedological classes identified in the SRB. We used the Pedological Map of the State of São Paulo (Rossi, 2017). Each soil type was reclassified in QGIS 3.1.8. We considered the erodibility values available in the literature, applying the arithmetic mean for each pedological unit to obtain a simplified estimate of the average behavior of each soil class (Table 1).

Table 1 – Erodibility values present in the SRB.

| Pedological classes | erodibility - K (t. h/mj.mm) | | | | | Standard Deviation | K factor average |
|---------------------|------------------------------|-----------------------|-----------------------|-----------------------|-----------------------|--------------------|------------------|
| | | | | | | | |
| Haplic Cambisols | 0.05087 ⁽¹⁾ | 0.0697 ⁽²⁾ | 0.0254 ⁽³⁾ | 0.0311 ⁽⁴⁾ | 0.0254 ⁽⁵⁾ | 0.01436 | 0.0303 |
| Red-yellow Latosols | 0.0162 ⁽¹⁾ | 0.0184 ⁽²⁾ | 0.0112 ⁽³⁾ | 0.0156 ⁽⁴⁾ | 0.02 ⁽⁵⁾ | 0.01131 | 0.0162 |
| Melanic Gleisol | 0.0361 ⁽¹⁾ | 0.0605 ⁽²⁾ | 0.0044 ⁽³⁾ | 0.018 ⁽⁴⁾ | 0.0044 ⁽⁵⁾ | 0.00248 | 0.0246 |

Source: Authors- using data from (1) Silva and Alvares (2005); (2) Fiorio (1987); (3) Mannigel *et al.* (2002); (4) Bertoni and Lombardi Neto (1999); and (5) Ribeiro and Alves (2008).

The LS factor represents the influence of topography on soil loss, with the length of the slope (L) decreasing the distance between the erosion start point and the deposition site. Meanwhile, the declivity (S) is related to the speed of runoff and erosion processes (Ganasri; Ramesh, 2016; Sales *et al.*, 2024). When calculating the LS, we generated a digital elevation model with a 12.5-meter resolution from the Alos Palsar satellite. Then, we used the GRASS GIS system in QGIS 3.32.3 to apply the *r.watersheed* tool and calculate the LS values.

The soil cover and management factor represents regional variability in soil loss, influenced by soil type, management practices, and local climatic conditions. It is a dimensionless ratio, varying between 0 and 1, which compares soil loss in vegetated areas with that in exposed soil without cover (Wischmeier; Smith, 1978; Xie *et al.*, 2021).

When calculating the C factor for the RUSLE survey, we conducted a land use and land cover classification for 2006 and 2017 based on collection 8 of the MapBiomias project, which provides the Annual Mapping of Land Cover and Land Use in Brazil. The classes were aligned with the IBGE Land Use Technical Manual and reclassified to fit the values suggested in the literature for the C factor (Panagos *et al.*, 2015).

Finally, the conservation practices factor is a measure that compares soil loss in areas with specific conservation practices to soil loss in crops planted along the maximum slope of the land or in areas without these practices. This factor ranges from 0 to 1. Similar to the C factor, the P factor values were obtained from experimental data and complemented by scientific observations on cause and effect relationships, and models based on physical principles (Wischmeier; Smith, 1978; Galdino *et al.*, 2017; Tian *et al.*, 2021).

The methodology for estimating the P Factor was based on data from the 2006 and 2017 IBGE Agricultural Censuses. This choice was based on the fact that agriculture and livestock farming are the main sources of income in the area, which is reflected in the predominance of rural properties, and soil loss occurs significantly in areas where agricultural activities take place (Maia Júnior; Lourenço, 2020; Sousa et al., 2022).

First, the data was collected through special tabulations for all the census tracts in the municipality of Ibiúna through a formal request on the Federal Government's platform since it is only freely available at the municipal level. These sectors are territorial units created by the IBGE for statistical purposes, both urban and rural (IBGE, 2007). The raw data was filtered to identify the sectors comprising the SRB based on cartographic information available on the IBGE website.

Then, the agricultural practices self-declared by the establishments in the census tracts of interest were analyzed. These practices included water erosion control and other conservation measures. This study used values established in bibliographic sources, such as those by Bertoni and Lombardi Neto (2005), to estimate the P values for each agricultural practice.

The average per area group was calculated, considering the properties' size, to assess data overlap and the accuracy of the locations. Thus, we multiplied the proportion of properties adopting conservation practices by the total number of properties in the sector and by the value of the corresponding P sub-factor. Furthermore, we subtracted the value 1 to estimate the absence of conservation practices. This calculation considered the percentage of properties that adopt practices and those that do not (Equation 2).

$$P = \frac{\sum_{j=1}^n \left(A_j \left(\frac{\sum_{i=1}^3 npi \left(\left(\frac{npi}{npt} \right) \times Pi + \left(1 - \frac{npi}{npt} \right) \times 1 \right) \right)}{\sum_{i=1}^3 npi} \right)}{\sum_{j=1}^n A_j} \quad (2)$$

Where:

P = Conservation practices factor for the census tract;

A_j = weighted average of the size class of the properties áreas;

npi = number of properties adopting a given practice;

npt = total number of properties in the census tract;

Pi = P value for each conservation practice.

Then, we added up the P values and divided them by the sum of the weighted averages of the area classes, resulting in the calculation of the P factor for each census tract. These values were integrated into ArcGIS, where each sector was assigned a specific P value. Areas without conservation practices and other non-agricultural areas

have been assigned a default value of 1. This approach enabled a detailed analysis of the influence of area size on the distribution of conservation practices and their environmental impact.

III. RESULTS AND DISCUSSION

RUSLE Factors

Soil loss estimation maps were prepared using RUSLE based on Equation 1. This calculation involved multiplying all the raster maps corresponding to the factors of erosivity (R), erodibility (K), topographic factor (LS), soil cover and management (C), and conservation practices (P). This process involved using the “Map Algebra” tool and the “Raster Calculator” in ArcGIS, resulting in the annual soil loss map.

The R, K, and LS factors were kept constant for 2006 and 2017 (Figure 3). Erosivity was calculated based on long-term climate data with no significant variations over the period. On the other hand, erodibility reflects soil properties that do not change in the short term and the topographic factor occurs unchanged since the relevance is considered stable on similar time scales, as it is the relief (Renard et al., 1991; Weiler et al., 2021).

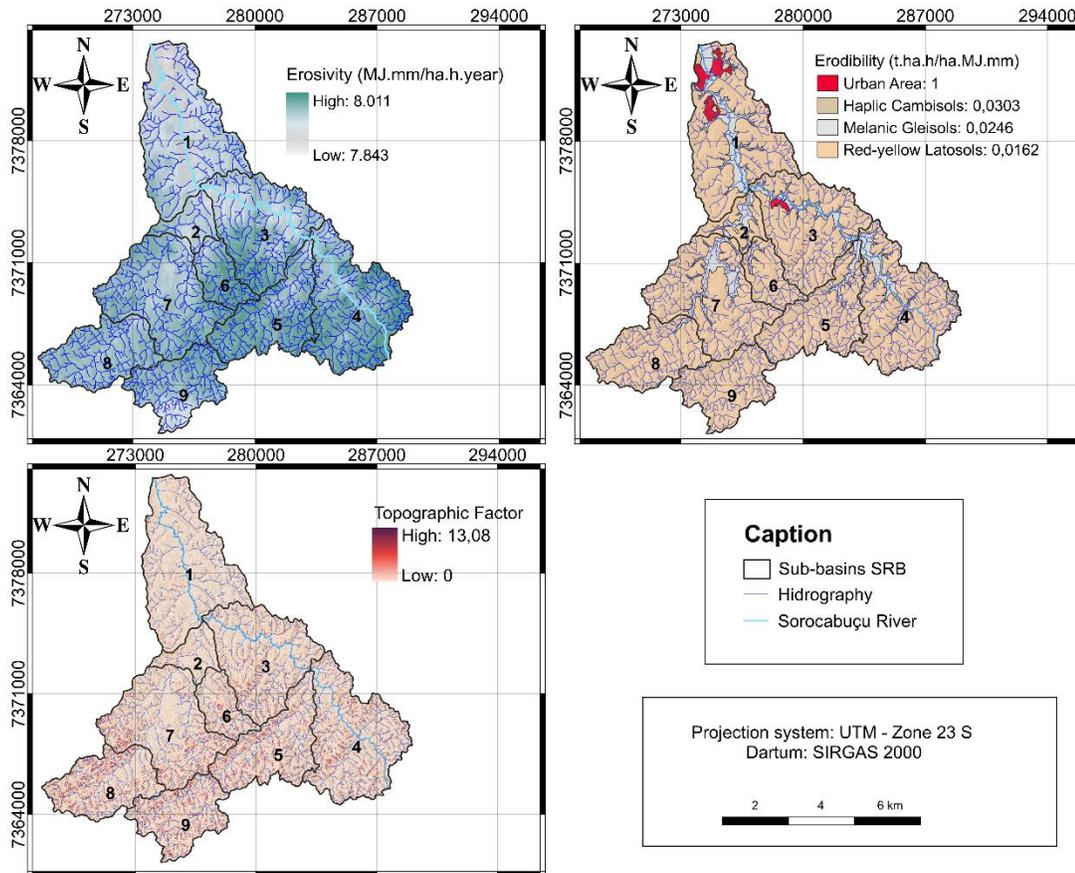


Figure 3 – Erosivity, erodibility, and topographic factors. Source: Authors.

Analysis of the R factor shows the influence of rainfall on erosion due to its intensity and kinetic energy (Dai et al., 2023). The average values in the river basin indicate strong erosivity (Table 2), with 7,961 MJ.mm/ha.h.year. There are the highest concentrations in the south, in areas where rainfall is more intense and where the slope is steeper, as pointed out by Arantes (2023) in the same basin.

Table 2 – Erosivity Classes.

| Classification of Erosivity | Erosivity (MJ.mm/ha.h.ano) |
|-----------------------------|----------------------------|
| Weak | $R \leq 2.452,0$ |
| Moderate | $2.452,0 < R \leq 4.905$ |
| Moderate to strong | $4.905,0 < R \leq 7.357,0$ |
| Strong | $7.357,0 < R \leq 9.810,0$ |
| Very strong | $R > 9.810,0$ |

Source: Foster et al. (1981).

In vulnerable areas, urbanization or inadequate soil management in agriculture can intensify erosion (Batista et al., 2021). Santos et al. (2025) and Arantes et al. (2024) point out that, in the SRB, it is essential to align agricultural practices with conservation measures to reduce surface runoff and water erosion, such as no-till with vegetation cover (Clement; Biolders; Degré, 2024). In this context, identifying areas with higher rainfall erosivity

is crucial to understanding environmental problems and assessing the movement, transport, and deposition of sediments (Bolledi et al., 2023).

The K factor reflects the pedological characteristics of the area, comprising predominantly Red-Yellow Latosols (88%), Melanic Gleisols (10.33%), and Haplic Cambisols (0.25%).

On the other hand, Latosols have low erodibility due to their high permeability and structuring. On the other hand, although they occupy a smaller area, Gleisols and Cambisols are more susceptible to erosion. Red-Yellow Latosols (RYLs) are deeply weathered soils with high total porosity, a high flocculation index, and a weighted average diameter, which indicates high permeability, especially under native vegetation. They occur in areas of flat or gently undulating terrain. These soils have low erodibility, as shown in Figure 3, on the erodibility map, and in Table 3, with the erodibility values for each soil typology presented. Its depth, good drainage, permeability, and absence of boulders favor agriculture (EMBRAPA, 2018). These characteristics, coupled with local relief features (De Aguiar et al., 2024), justify the lower susceptibility of RYLs to erosion compared to other soils (Table 3).

Table 3 – Soil erodibility.

| Erodibility values (t. h/mj.mm) | Classification of erodibility |
|------------------------------------|-------------------------------|
| <0,01 | Very low |
| 0,01 – 0,02 | Low |
| 0,02 – 0,03 | Medium |
| 0,03 – 0,04 | High |
| >0,04 | Very high |

Source: Adapted from Paraíba (2006).

Gleisols, formed in flooded areas with a water table close to the surface, have poor drainage conditions that favor gleization. Although limited for agriculture due to flooding, they become suitable after drainage and chemical correction (Embrapa, 2022). With medium erodibility, these soils are susceptible to erosion due to the degradation of the structure caused by constant saturation.

Haplic Cambisols vary in depth and drainage, and have high erodibility, being more vulnerable to erosion in steep areas with inadequate management (IBGE, 2007; EMBRAPA, 2018). Together, Gleisols and Cambisols occupy only 11% of the river basin, a smaller area than the Red-Yellow Latosols.

The LS factor (slope length and inclination) reveals the relationship between topography and erosion, with steeper declivities to the south, favoring greater surface runoff and increased soil loss (Schmidt; Tresch; Meusburger, 2019). The LS values range from 0 to 13.08, with higher concentrations in the south of the river basin, showing topographic vulnerability.

We obtained an average of 0.71, with the highest values concentrated in the southern section of the river basin (Figure 3). According to Panditharathne (2019), the LS factor is crucial when analyzing erosion processes since topographical factors such as slope length and declivity are directly related to soil loss.

Furthermore, we can see that higher values are concentrated in the south, while lower values are concentrated in the north. Moreover, the lowest values are concentrated near the main river of the basin, which is where the flattest reliefs occur (Santos et al., 2025), implying low interference in the erosion process (Demarchi et al., 2019).

These factors reinforce the need for sustainable practices to reduce the impacts of water erosion, especially in critical areas identified by the erosivity, erodibility, and topographic factor maps.

However, the C factor was calculated for both years due to the changes in land use and cover between the years studied. Table 4 compares the use classes and their respective C values.

Table 4 – Coverage classes and C factor values.

| Classes | Area (km ²) | Area (%) | Area (km ²) | Area (%) |
|---------------------------|-------------------------|----------|-------------------------|----------|
| | 2006 | 2006 | 2017 | 2017 |
| Coffee | - | - | 0,01 | 0,00003 |
| Pasture | 27 | 13,36 | 10,61 | 5,25 |
| Forest Formation | 99 | 48,94 | 100,77 | 49,82 |
| River, Lake, and Ocean | 0,65 | 0,32 | 0,42 | 0,21 |
| Mosaic of Uses | 55 | 27,19 | 55,85 | 27,61 |
| Urbanized Area | 3 | 1,48 | 4,32 | 2,14 |
| Soybeans | 1 | 0,49 | 2,68 | 1,32 |
| Forestry | 0,43 | 0,21 | 1 | 0,49 |
| Other Non-vegetated Areas | 0,15 | 0,08 | 0,08 | 0,04 |
| Other Temporary Crops | 16 | 7,91 | 26,78 | 13,24 |
| Wetlands and marshes | 0,05 | 0,02 | 0,01 | - |

Source: Adaptado de Silva et al., 2010¹; Bertoni; Lombardi Neto, 2005²; Tomazoni; Guimarães³; Stein,1987⁴; Bertoni e Lombardi Neto, 2012⁵.

Temporary crops, characterized by short vegetative cycles, have expanded in the river basin, especially in steep areas near the Sorocabuçu River. According to Mahala (2018), these areas, along with barren land, are more susceptible to erosion due to the high values of the C factor. Its values increased from 8.4% of the total

area in 2006 to 14.49% in 2017, highlighting the advancement of agriculture and its potential to intensify erosion processes.

On the other hand, regions with dense vegetation cover showed low C values, providing greater protection for the soil. These predominantly forested areas mitigate the impact of rain, favor water infiltration, and reduce surface runoff. Figure 4 shows that areas with C equal to zero, such as water bodies and urbanized areas, are practically free of erosion due to the soil's impermeability or saturation (Bhandari et al., 2021; Zhang et al., 2021).

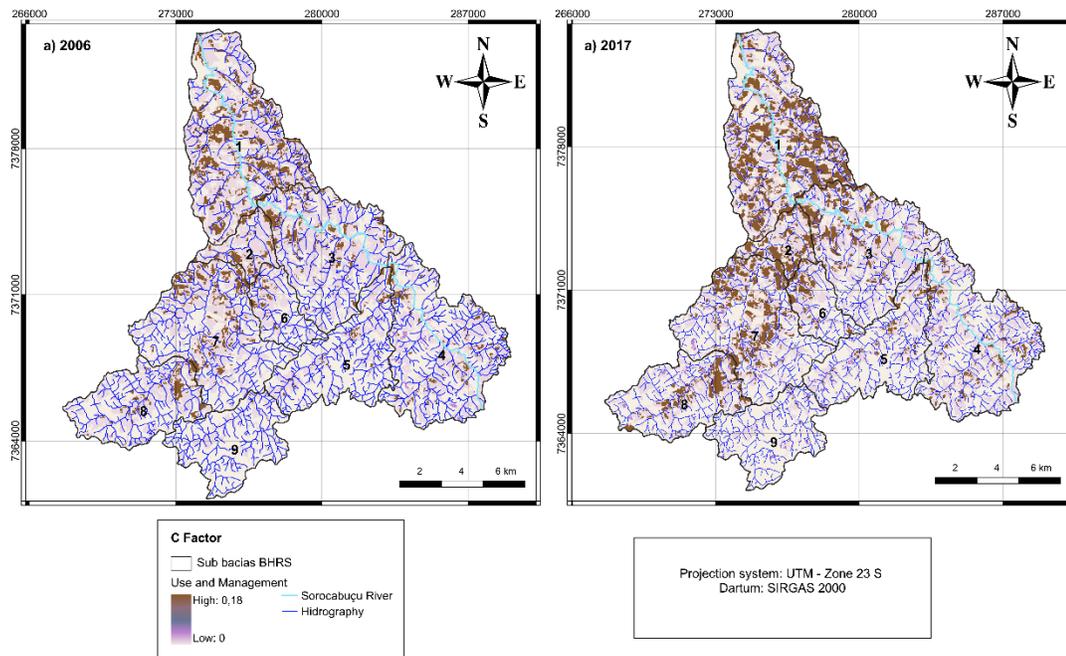


Figure 4 – Land Use and Management Map. Source: Authors.

Regarding P in the river basin under study, the total number of agricultural establishments decreased from 935 in 2006 to 720 in 2017. At the same time, the number of census tracts increased from 51 to 80 due to demographic and administrative adjustments made by the IBGE. This reduction in the number of establishments may be associated with factors such as the reclassification of production units, land concentration, and changes in land use.

Crop areas were analyzed in size ranges varying from less than 1 hectare to more than 500 hectares (Table 5).

Table 5 – Number of establishments by crop area class.

| Crop Area (in hectares) | Number of establishments in 2006 | Establishments in 2006 (%) | Number of establishments in 2017 | Establishments in 2017 (%) |
|-------------------------|----------------------------------|----------------------------|----------------------------------|----------------------------|
|-------------------------|----------------------------------|----------------------------|----------------------------------|----------------------------|

| | | | | |
|---------------------------------------|------------|------------|------------|------------|
| > 0 < 1 | 207 | 22,14 | 147 | 20,42 |
| 1 < 2 | 178 | 19,04 | 133 | 18,47 |
| 2 < 5 | 282 | 30,16 | 242 | 33,61 |
| 5 < 10 | 6 | 6,52 | 91 | 12,64 |
| 10 < 20 | 34 | 3,64 | 43 | 5,97 |
| 20 < 50 | 12 | 1,28 | 12 | 1,67 |
| 50 < 100 | 5 | 0,53 | 5 | 0,69 |
| 100 < 200 | 2 | 0,21 | - | - |
| 200 < 500 | - | - | - | - |
| > 500 | 9 | 0,96 | - | - |
| Undeclared/Producer with no crop area | 145 | 15,51 | 47 | 6,53 |
| Total | 935 | 100 | 720 | 100 |

Source: Authors.

The group of areas between 2 and 5 hectares was the most significant in both years. In 2017, the number of properties with areas between 5 and 10 hectares increased significantly, reflecting a possible redistribution of land to smaller categories. Notably, crop areas of more than 100 hectares disappeared in 2017, which may be related to changes in land use or conversion to pasture and other purposes.

In Ibiúna, each fiscal module corresponds to 16 hectares, according to SEMIL (2014). The establishment must have up to four fiscal modules to be recognized as a family farm under Law No. 11,326/2006. Given the size of the crops, it indicates that the river basin has great potential to be characterized by family farming. The law also states that family farmers must work with family labor and obtain most of their income from their establishment. You must exceed this area to be considered a large producer (Brazil, 2006).

Law 8,629/1993 classifies medium-sized properties as those with an area between 4 and 15 fiscal modules, and large properties generally have more than 15 fiscal modules (Brazil, 1993). Only 1.17% of establishments could be classified as medium or large producers in 2006. However, the whole river basin was classified as family farming in 2017.

Regarding conservation practices, the SRB included contour planting, terrace cultivation, and no-till. In 2006, 38.72% of establishments adopted these practices. Meanwhile, in 2017, this number dropped dramatically to 12.5%, indicating a significant reduction in the use of conservation practices. Among the practices evaluated, contour planting was the most frequent in both periods analyzed, with 89% in 2006 and 88% in 2017. Regarding no-till, in 2006, it accounted for 10%. In 2017, it was 12%. As a result, terrace cultivation only accounted for 1% of the total and was not adopted by any property in 2017. Therefore, these results indicate low diversification in the adoption of conservation technologies.

In short, although contour planting is widely adopted, the lack of information on the repetition of agricultural practices and the limited distribution of multiple practices in the census tracts underline limitations in the application and interpretation of the agricultural census. The conservation practices identified include contour planting, terrace cultivation, and no-till, with values assigned based on empirical studies by the Agronomic Institute of Campinas (IAC) (Bertoni; Lombardi Neto, 2005).

According to Lombardi Neto (2005), contour planting can reduce soil loss by up to 47%, with a sub-factor (P) value of 0.53. Terraces vary in effectiveness depending on their typology and the distance between them. An average P value of 0.50 should reduce soil loss by up to 50% (EMBRAPA, 2017). No-till can reduce soil loss by up to 53%, resulting in a P value of 0.47. It is worth noting that the effectiveness of conservation practices depends on local conditions and the characteristics of the land, such as declivity and the distance between terraces (Bertoni; Lombardi Neto, 2005).

Figure 5 shows the P factor for the years analyzed, considering that they were calculated for the agricultural areas of the SRB for each census tract. Furthermore, a value of 1 was assigned to the areas where there were no practices within the crop areas and to the other areas as well.

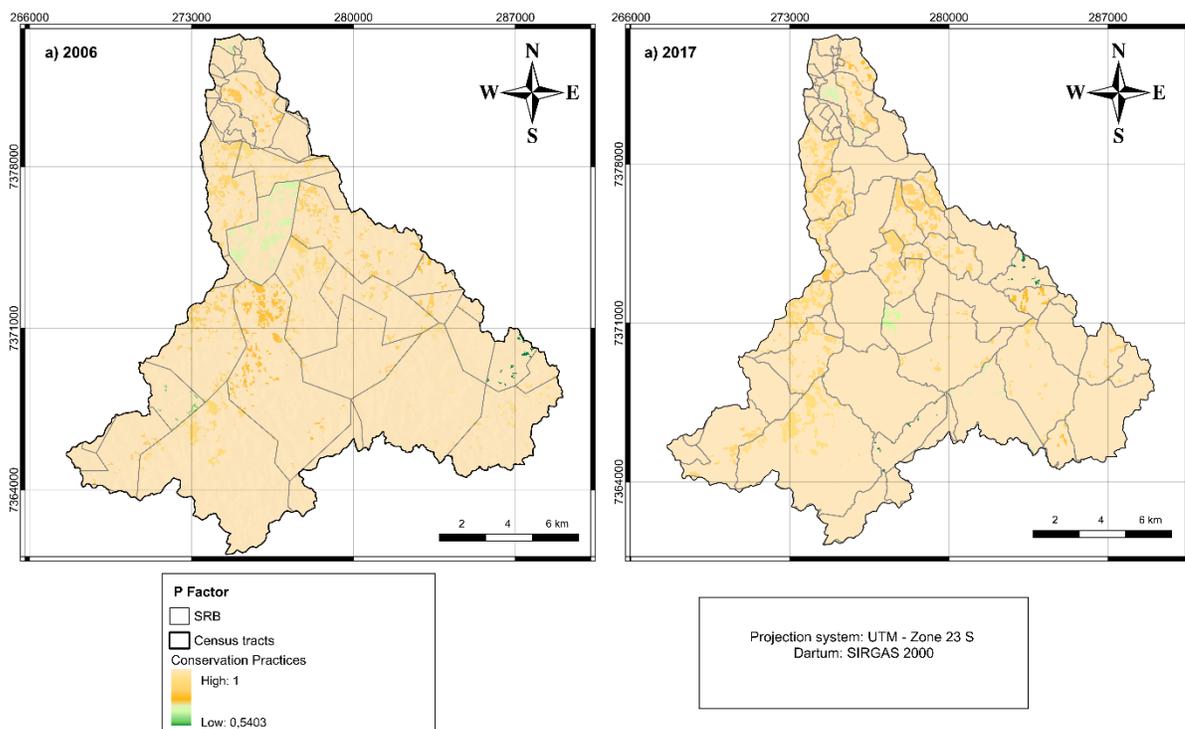


Figure 5 – Values of conservation practices calculated for the SRB. Source: Authors.

The P factor in the RUSLE shows significant variations according to agricultural practices and types of land use in the sub-basins. In the two years analyzed, the lowest P value recorded was 0.5403, with an overall average of 0.99. Meanwhile, the maximum value reached was 1, reducing areas without conservation practices or not destined for agriculture. Focusing on areas of work, the average P was 0.94 in 2006 and increased to 0.97 in 2017. Therefore, it is worth noting that even though there are no major discrepancies between the P factor equal to 1 and the values found by the methodology adopted, the variations are still significant and indicative of trends and changes in agricultural practices over time. It reflects a transformation in land use, especially due to the expansion of agricultural areas, which went from 8.4% of the total river basin in 2006 to 14.49% in 2017. This growth, although considerable, still represents a small fraction of the river basin as a whole, which justifies the difficulty in implementing conservation practices since agricultural areas are still a minority.

Sub-basins with a higher intensity of agricultural activities generally have higher P values, highlighting a distinct pattern in areas where agricultural practices are predominant. Factors such as contour planning directly influence these higher P values, reflecting the conservation profile of each location.

The higher average P values are related to large clusters of farmland (Table 6), which often have a lower proportion of properties adopting soil conservation practices. This characteristic contributes directly to the higher P levels, evidencing the relationship between intensive land use and less application of conservation practices.

Table 6 – Adopters of conservation practices by crop area group.

| Crop Area (in hectares) | Adopting establishments in 2006 (%) | Adopting establishments in 2017 (%) |
|--------------------------------|--|--|
| > 0 < 1 | 27,9 | 16,25 |
| 1 < 2 | 20,72 | 12,5 |
| 2 < 5 | 35,08 | 42,5 |
| 5 < 10 | 8,29 | 16,25 |
| 10 < 20 | 6,35 | 10 |
| 20 < 50 | 0,55 | - |
| 50 < 100 | 0,83 | 2,5 |
| 100 < 200 | 0,28 | - |

Source: Authors.

The lack of government support and the structural barriers faced by small farmers partly explain this low adoption. As discussed by Tosakana et al. (2010), small producers face difficulties related to limited access to technologies, credit, and markets, factors that restrict the implementation of effective conservation practices. On the other hand, medium and large establishments, with greater financial capacity and access to technological resources, tend to adopt conservation practices more easily, which is reflected in the lower P values in these areas.

Therefore, even though the values found do not show a significant discrepancy regarding the P factor equal to 1, they show a trend toward greater intensity of conservation practices in smaller agricultural areas. As we move toward larger areas where intensive agriculture predominates, there is a progressive reduction in the adoption of these practices. These results are consistent with the observed land use and reinforce the relationship between agricultural planning and conservation practices, suggesting that the P factor obtained is representative of the river basin's reality.

These variables, such as the expansion of agricultural areas and the decrease in adherence to conservation practices, have direct implications for soil management and environmental impacts (Tufa et al., 2023). Therefore, the adopted methodology, which considers these particularities, offers a more precise and contextualized analysis. Thus, it contributes significantly to understanding the dynamics of land use and the impacts of agricultural practices in the studied river basin.

Despite its relevance, the development of the methodology used here had some limitations. One of the main limitations involved the time needed to obtain and process the data, given the large amount of information required for analysis in the municipality of Ibiúna. Furthermore, the lack of more detailed data, such as the exact location of the properties, hindered the precise identification of which practices were implemented on each property or whether the same property might have been replicating different conservation practices over time. This restriction is due to the need to preserve personal and sensitive information under the General Data Protection Law (LGPD), which restricts the availability of precise geospatial data on agricultural establishments.

The reliance on self-reported data from the Census presents some uncertainties since the absence of field checks means that the data obtained is based on the owners' perception. In other words, self-reported data. It implies that, although the information is valuable, it cannot accurately reflect the reality of the management conducted on farms, especially when considering the diversity of approaches employed within the same census tract.

Moreover, although the adoption of practices such as contour planting was significant, the lack of information provided on the reproduction of agricultural practices and the distribution of multiple practices within the same census tract still represents one of the limitations in interpreting the data. The methodology was based on the information provided by the establishments on the practices adopted. However, this information does not make it possible to distinguish, for example, which practices were applied to which crops or how often they were renewed, which limits the accuracy of the obtained results.

Despite these limitations, this methodology is of great importance for the advancement of analysis related to soil conservation. Although it is worth considering the uncertainties regarding this data's accuracy, the model's application with the adjusted P factor using data from the Agricultural Census offers a valuable perspective on the adoption of conservation practices. This study contributes to a better soil loss estimate and to an understanding of the agricultural practices adopted by farms, especially in family farming. It represents an advance in the approach to conservation practices and the strengthening of sustainable agriculture.

Soil Loss

This factor (A) was calculated only for crop areas for a better soil loss representation. Soil loss is more intense in agricultural areas due to practices such as plowing and seeding, which expose and degrade the soil (Lal, 2020; Ash, 2021). Applying the estimated calculation in these regions is methodologically appropriate since the reduction in vegetation cover and intensive soil use increase its susceptibility to erosion. By excluding areas of low erosion, such as forests, water bodies, and urban areas, we have a priority analysis of agricultural areas, where the impact of erosion prevails (El Jazouli et al., 2017).

Figure 6 shows the soil loss maps in the SRB crop areas for 2006 and 2017, classified according to the methodology of the Food and Agriculture Organization of the United Nations (FAO, 1967).

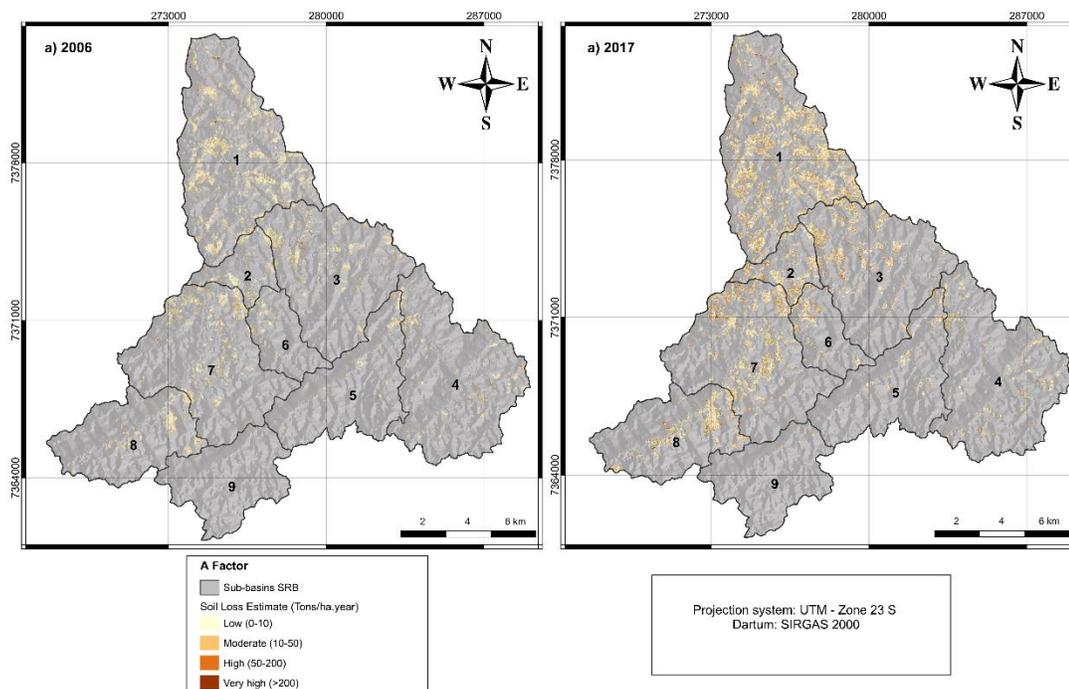


Figure 6 – Soil loss with adjusted P in crop áreas. Source: Authors.

In the context of geoprocessing, the RUSLE model, applied to agricultural areas, enables accurate mapping of soil loss rates. Integrated with GIS, it analyzes geospatial data to identify erosion patterns and the most vulnerable areas (Ostovari et al., 2017). The methodology focuses on agricultural regions, eliminating the influence of areas with low erosion, which provides a more detailed analysis.

The aim was to compare soil loss estimates between the model with the P developed by the study in question and the model with P equal to 1, conventionally used in the literature. In both years, the soil loss estimation category with the largest area is Low (Tables 7 and 8).

Table 7 – Soil loss in agricultural areas with adjusted P.

| Sub-basins | Low (0-10) | | Moderate (10-50) | | High (50-200) | | Very high (>200) | | Crop area (km ²) by sub-basins in 2006 | Crop area (km ²) by sub-basins in 2017 |
|------------|------------|-------|------------------|-------|---------------|-------|------------------|------|--|--|
| | 2006 | 2017 | 2006 | 2017 | 2006 | 2017 | 2006 | 2017 | | |
| 1 | 64,55 | 65,19 | 32,82 | 33,74 | 0,80 | 0,85 | 0,59 | 0,23 | 7,20 | 11,73 |
| 2 | 72,78 | 69,39 | 25,72 | 28,34 | 1,50 | 2,27 | - | - | 1,13 | 2,21 |
| 3 | 62,62 | 62,24 | 34,62 | 34,59 | 1,97 | 3,05 | 0,79 | 0,24 | 2,44 | 3,88 |
| 4 | 64,80 | 64,96 | 30,80 | 32,05 | 4,31 | 2,99 | 0,09 | - | 1,30 | 1,76 |
| 5 | 66,28 | 66,07 | 30,54 | 30,78 | 3,02 | 3,16 | 0,17 | - | 0,63 | 0,89 |
| 6 | 65,62 | 62,89 | 33,33 | 34,51 | 33,33 | 34,51 | 1,05 | 2,60 | 0,46 | 1,01 |
| 7 | 66,02 | 66,34 | 32,12 | 31,10 | 1,86 | 2,56 | - | - | 2,93 | 5,43 |
| 8 | 65,93 | 64,36 | 29,92 | 30,70 | 4,15 | 4,94 | - | - | 1,24 | 2,39 |
| 9 | 75,00 | 63,77 | 21,88 | 24,64 | 3,13 | 11,59 | - | - | 0,03 | 0,11 |

Source: Authors.

Table 8 – Soil loss in agricultural areas with P equal to 1.

| Sub-basins | Low (0-10) | | Moderate (10-50) | | High (50-200) | | Very high (>200) | | Crop area (km ²) by sub-basins in 2006 | Crop area (km ²) by sub-basins in 2017 |
|------------|------------|-------|------------------|-------|---------------|-------|------------------|------|--|--|
| | 2006 | 2017 | 2006 | 2017 | 2006 | 2017 | 2006 | 2017 | | |
| 1 | 65,19 | 64,92 | 32,82 | 33,74 | 1,02 | 1,02 | 0,59 | 0,23 | 7,20 | 11,73 |
| 2 | 70,89 | 67,85 | 27,42 | 29,19 | 1,69 | 2,96 | - | - | 1,13 | 2,21 |
| 3 | 62,09 | 62,24 | 34,58 | 33,95 | 2,54 | 3,45 | 0,79 | 0,36 | 2,44 | 3,88 |
| 4 | 63,31 | 64,11 | 32,15 | 32,51 | 4,46 | 3,32 | 0,08 | 0,06 | 1,30 | 1,76 |
| 5 | 65,10 | 64,26 | 31,54 | 32,48 | 3,19 | 3,26 | 0,17 | - | 0,63 | 0,89 |
| 6 | 61,71 | 59,21 | 36,94 | 37,62 | 1,35 | 3,17 | - | - | 0,46 | 1,01 |
| 7 | 64,98 | 65,63 | 32,42 | 31,79 | 2,59 | 2,58 | - | - | 2,93 | 5,43 |
| 8 | 64,87 | 63,71 | 30,36 | 31,09 | 4,77 | 5,20 | - | - | 1,24 | 2,39 |
| 9 | 75,00 | 61,90 | 21,88 | 26,19 | 3,13 | 11,90 | - | - | 0,03 | 0,11 |

Source: Authors.

In 2006, the model with P equal to 1 recorded a maximum soil loss value of 3,227.58 t/ha.year. Meanwhile, the presented model showed a lower maximum value of 2,975.83 t/ha.year. The averages were 12.0

t/ha.year and 11.42 t/ha.year, respectively. In 2017, the model with P equal to 1 showed maximum and average values of 2,484.48 t/ha.year and 11.74 t/ha.year. Meanwhile, the adjusted model recorded 2,250.93 t/ha.year and 11.44 t/ha.year. These results indicate that the model with P equal to 1 has a higher estimated soil loss value than the adjusted model.

The high maximum soil loss values observed in the different modeled scenarios indicate the presence of outliers. These may be the result of several factors, such as natural terrain features, the influence of the RUSLE survey parameters, and the limitations associated with the spatial resolution of the raster data used in the modeling. It is worth noting that areas with hilly terrain, highly erodible soils, or a lack of conservation practices tend to have much higher erosion rates, which explains the occurrence of these extreme values.

Furthermore, pixel-based modeling can amplify these variations since each cell in the raster represents an average soil loss within a given area and can capture abrupt changes in relief and land use. Since the RUSLE innovation does not consider regional deposition processes, these high values may reflect where there is a strong removal of sediment without registering its redistribution, which directly influences the modeling results.

These values were kept to preserve the representativeness of areas susceptible to intense erosion processes. Previous studies, such as Arantes (2023), have reported similar values, providing consistency to the obtained data. Future studies should evaluate additional methodologies for treating these values, including statistical techniques for filtering out outliers and adjustments to data resolution.

Sub-basins 1 and 9 stood out in 2006, showing different characteristics to the others in both models. Sub-basin 1, which is larger in area, had between 70% and 75% of its area classified as low soil loss. Meanwhile, sub-basin 9, which is smaller in area and has fewer agricultural activities, had a similar trend. However, in the model with P equal to 1, there was a slight decrease in low soil loss areas in both sub-basins, mainly due to the increase in the high loss category. This pattern was more evident in agricultural areas, related to the use of P values equal to 1, which increase the concentration of area.

The analysis also revealed that the low soil loss category remains predominant in both models, covering more than 60% of the total area. However, in 2017, areas with high soil loss were more prominent in the model with P equal to 1, suggesting that the lack of conservation practices in this model favored a redistribution of categories that does not reflect reality. The most significant difference occurred in the very high soil loss category, where the model with P equal to 1 showed a larger area compared to the adjusted model.

Between the years studied, the distributions of soil loss categories showed slight variations, reflecting the structural characteristics of the river basin, such as topography, land use, and land cover (Rajbanshi;

Bhattacharya, 2020; Ayele et al., 2022). The high P values can be explained by the very low intensity of adoption of conservation agricultural practices. In 2006, there were only three practices, and in 2017, two.

In 2006, when comparing the calculations with adjusted P and P equal to 1, the average soil loss was 0.47 t/ha.year, equivalent to approximately 500 kg per hectare. This value can be significant for environmental impact and agricultural sustainability, indicating high susceptibility to erosion, as discussed by Maltsev and Yermolaev (2018). The maximum value recorded was 251.75 t/ha.year, possibly due to intensive agricultural practices and exposed soils or anomalous outliers. This loss can contribute to the siltation of water bodies such as the Itupararanga Reservoir.

In 2017, the average loss was 0.23 t/ha.year, reflecting a reduction in level. It was probably due to a reduction in the number of establishments adopting agricultural practices that increased the P factor, bringing it closer to the results of the model with P equal to 1. Despite the increase in the working area, from 8.4% to 14.56% of the river basin between 2006 and 2017, the changes in soil loss were small and may be influenced by the river basin's biophysical characteristics.

The differences between the models were relatively small. However, when considered cumulatively over the years and on a larger spatial scale, these variations represent a significant amount of soil lost. Even differences of less than 1 t/ha.year can have an impact on soil fertility qualities, agricultural productivity, and the use of water bodies, especially in areas with high susceptibility to erosion. Thus, these values indicate that, in the absence of effective conservation practices, the erosion process can intensify over time, aggravating environmental manipulation and hindering soil recovery. This scenario is compatible with this study since average soil loss was higher in 2017.

Analysis of the transition map (obtained from the difference between the soil loss maps with (P equal to 1 and with adjusted P) shows that in 2017, soil loss was closer to the values estimated by the model with P equal to 1 than in 2006. It suggests that, over time, P factor values have increased, reflecting a reduction in the number of properties adopting conservation practices. This scenario points to a greater susceptibility to erosion and reinforces the need for continuous monitoring and soil conservation strategies.

In addition to representing a more realistic erosion estimate, this approach also provides a diagnosis of the conservation character of the river basin. The use of data from the Brazilian Agricultural Census (IBGE) enables a detailed assessment of the degree of implementation of soil management practices, highlighting which areas require greater incentives for conservation. Unlike many studies, which assume that conservation practices do not exist ($P = 1$), the adjusted model enables a more contextualized analysis, contributing to more effective

environmental management strategies. By incorporating secondary data to map the impact of agricultural practices on soil conservation, this method offers a valuable tool for subsidizing public policies and sustainable landscape planning.

The implementation of conservation practices is associated with aspects such as educational level, income, access to extension services, and property size. In this sense, there are challenges in adopting these practices due to the costs and lack of technical support (Read; Wainger, 2021). However, the adjusted model shows that the adoption of these practices, even in smaller areas, can result in a significant reduction in soil loss. Thus, it contributes to the mitigation of erosion and promotes more sustainable soil use in the region, with the potential to be applicable in other regions.

IV. CONCLUSIONS

Conservation practices play an important role in reducing soil loss, especially in agricultural areas. The P factor can be overestimated by assuming that agricultural areas do not adopt such practices. Therefore, it is essential to consider the local reality when calculating this factor, identifying existing practices with data such as that from the IBGE.

Although the model was effective, there were limitations due to the quality of the data available, such as overlap, number of repetitions, and the location of establishments. When comparing the soil loss calculations using the two P values, the average and maximum differences were small since the agricultural areas occupy a small proportion of the river basin and there are some low conservation practices.

However, when analyzing the differences in soil loss in agricultural areas between the models with the two P used, we noticed a difference in the average soil loss value, emphasizing the importance of conservation practices. Even with small variations between soil loss models, the defined P factor is efficient, especially in small-scale farming river basins, which underlines the need to provide incentives for these practices.

These results reinforce the importance of public policies that encourage conservation practices and the use of data such as the IBGE's agricultural censuses to plan sustainable strategies, promoting soil conservation and mitigating erosion.

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