Revista Brasileira de Geografia Física
ISSN:1984-2295
Homepage: https://periodicos.ufpe.br/revistas/rbgfe

Impacts of land cover policy on soil erosion in the Guandu river basin – Espírito Santo

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Artigo recebido em 22/01/2025 e aceito em 03/12/2025

A B S T R A C T
Land use can be considered one of the main variables influencing watershed erosion. Over the past decades, extensive native areas have been restored in the central-western region of Espírito Santo State, Brazil, while the agricultural pressures on land-cover–sedimentation dynamics remain, to some degree, unknown. This study aimed to analyze the effects of environmental restoration policies on soil erosion and sediment dynamics in the Guandu River Basin, Espírito Santo, based on the quantification of potential soil losses and sediment delivery rates in 2012 and 2021, integrating the RUSLE model with observed data from three sedimentological stations. In terms of sediment delivery ratio (SDR), the year 2012 presented the mean annual value of 8.5%, while 2021 showed a mean annual value of 6.2%. These results revealed a reduction in average potential erosion, highlighting the mitigating role of native vegetation expansion and the retreat of agricultural areas and exposed soils. A consistent decline in sediment production and SDR values was also observed, suggesting lower hydro-sedimentological connectivity and increased sediment retention capacity, both resulting from environmental restoration. The results showed that forest restoration and the strengthening of environmental regulatory instruments have promoted substantial improvements in erosion control and sediment transport reduction, reinforcing the relevance of policies based on ecological restoration for enhancing hydrological resilience and guiding sustainable land-management strategies.
Key-words: RUSLE, sediment transport, empirical model, hydrology.

Impactos da política de cobertura do solo na erosão hídrica da bacia do rio Guandu – Espírito Santo

Resumo
O uso do solo pode ser considerado uma das principais variáveis que influenciam a erosão das bacias hidrográficas. Nas últimas décadas, extensas áreas nativas foram restauradas na região centro-oeste do estado do Espírito Santo, Brasil, enquanto as pressões agrícolas sobre a dinâmica da cobertura do solo e sedimentação permanecem, em certa medida, desconhecidas. Este estudo teve como objetivo analisar os efeitos das políticas de restauração ambiental na erosão do solo e na dinâmica dos sedimentos na Bacia do Rio Guandu, Espírito Santo, com base na quantificação das perdas potenciais de solo e das taxas de entrega de sedimentos em 2012 e 2021, integrando o modelo RUSLE com dados observados em três estações sedimentológicas. Em termos de taxa de entrega de sedimentos (SDR), 2012 apresentou o valor médio anual de 8,5%, enquanto 2021 apresentou um valor médio anual de 6,2%. Os resultados revelaram uma redução na erosão potencial média, destacando o papel mitigador da expansão da vegetação nativa e do recuo das áreas agrícolas e solos expostos. Também foi observada uma diminuição consistente na produção de sedimentos e nos valores de SDR, sugerindo menor conectividade hidro-sedimentológica e maior capacidade de retenção de sedimentos, ambos resultantes da restauração ambiental. Os resultados mostraram que a restauração florestal e o fortalecimento dos instrumentos regulatórios ambientais promoveram melhorias substanciais no controle da erosão e na redução do transporte de sedimentos, reforçando a relevância das políticas baseadas na restauração ecológica para aumentar a resiliência hidrológica e orientar estratégias sustentáveis de gestão da terra.
Palavras-chave: RUSLE, transporte de sedimentos, modelo empírico, hidrologia.

Introduction

Soil erosion is a natural process of removal and transportation of soil material through the action of erosive agents (Panditharathne et al., 2019), and in recent years it has been accelerated by activities such as intensive agriculture, soil degradation, deforestation and other anthropogenic disturbances (Wang et al, 2023).

The erosive process leads directly to the loss of environmental services through the reduction of nutrients and organic carbon in the soil (Baldotto et al., 2021), reducing the biota of the eroded area, in addition to the loss of reservoir capacity and the silting of rivers through the transport of soil particles by water (Lira et al., 2020).

According to Guedes et al. (2025), predicting soil loss is an important tool for assessing risks and determining appropriate land use and management. However, obtaining local data to estimate soil losses at the watershed level is costly and time-consuming.

In this respect, several approaches can be used, ranging from empirical to conceptual and physical models (Raza et al., 2021), embedded in technologies that involve the use of geographic information systems (GIS), allowing the study of the spatio-temporal dynamics of the variables that affect sediment generation and transport.

Among the existing empirical models, the Revised Universal Soil Loss Equation (RUSLE), developed by the United States Department of Agriculture (Renard et al., 1997), is a flexible tool that has been adapted to watershed scales combined with GIS in soil erosion assessments (Duarte et al., 2021) and its results can aid the management and conservation of natural resources (Lense et al., 2022).

One of Brazil's main river basins is the Doce River, which has a history of land and water use degradation (Ramos et al., 2024), with high rates of deforestation followed by erosion, sedimentation and eutrophication of its rivers. The situation was exacerbated in November 2015 with the collapse of the Fundão dam in the city of Mariana in Minas Gerais. This environmental disaster spilled between 40 and 62 million m³ of mining waste into the Doce River (Meira et al., 2016).

Fernandes et al. (2016) believed that this incident, in addition to drastically polluting the waters, was the last straw in a long-lasting process of degradation, with losses of ecosystem services to the order of 5.21 billion US dollars (Garcia et al., 2017).

Following this incident, the Brazilian government committed itself to an ambitious environmental recovery plan through the reforestation of 12 million hectares by the year 2030 (Pires et al., 2017), and a set of national-level policies, which include: the Native Vegetation Protection Law (NVPL), the National Climate Change Adaptation Plan (NCCAP), focusing on water resource management, and the National Native Vegetation Restoration Policy (Brancalion et al., 2016; Scarano, 2017). These measures are expected to result in ecosystem benefits, especially in terms of water quality.

However, according to Pires et al. (2017), such effects depend on the region of implementation and the ecological parameter under study, since it is not yet clear how compliance with the NVPL would lead to changes in water quality in the Doce River basin and how national commitments could promote the restoration of land use and land cover in the region.

Due to the lack of information on the reduction in soil loss that may occur with the implementation of these policies, this study aimed to identify the spatial and temporal distribution patterns of soil erosion and sediment transport in the Guandu River Basin (BRG), a tributary of the Doce River, using a combination of the RUSLE quantitative model and the SDR approach before and after the tailings spill (2012 and 2021, respectively), and 24 years of sediment records at three monitoring points along the basin, representing the upper, middle, and lower reaches of the watercourse. The results can serve as a source of information for decision-makers, facilitating the planning and prioritization of soil restoration projects and natural resource conservation programs.

Material and Methods

Study area

The Guandu river basin (GRB) is located in the central-western portion of the state of Espírito Santo at coordinates 19°26' and 20°20' south latitude and 40 °55' and 41°23' west longitude (Figure 1), covering the municipalities of Laranja da Terra, Brejetuba, Afonso Cláudio and Baixo Guandu, occupying an area of approximately 2,141km², with a population of 74,000 inhabitants (IGAM, 2010). It belongs to the Rio Doce hydrographic network, with its source in the municipality of Afonso Cláudio and its outlet in the municipality of Baixo Guandu. Its main watercourse is 160 km long.

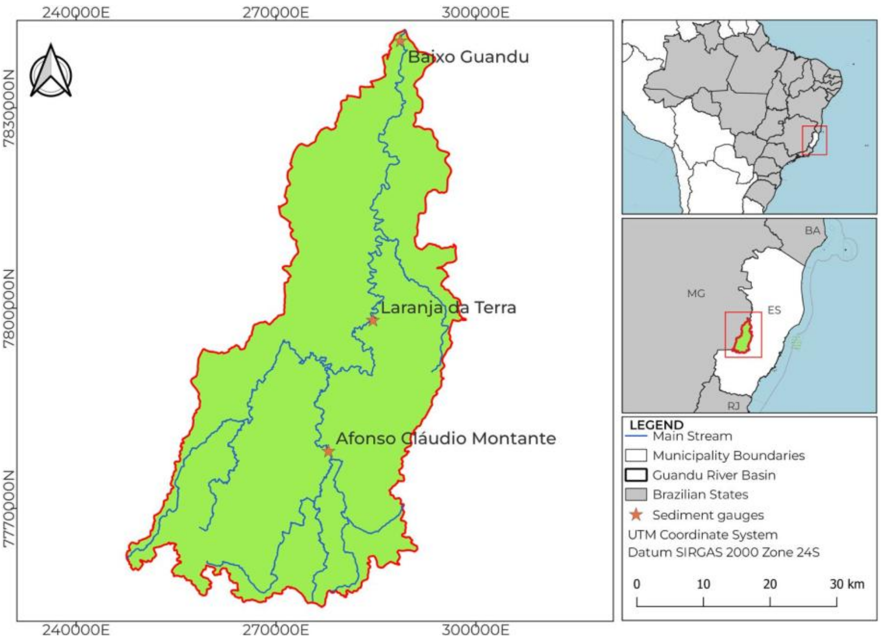


Figure 1. Geographical location of the area under study.

The topography of the GRB region is characterized by strongly undulating to mountainous terrain. The basin's soils were classified using the Brazilian soil map (EMBRAPA, 2018) on a scale of 1:5,000,000 with a predominance of red argisol – PvAd (53.31%), red-yellow oxisol – LVAd (29.09%), dystrophic haplic cambisol - CXd (16,14%), eutrophic haplic cambisol - CXe (1.06%), and lithic ustorthents - RLd (0.40%). Figure 2 shows the spatial distribution of the soil classes in the basin.

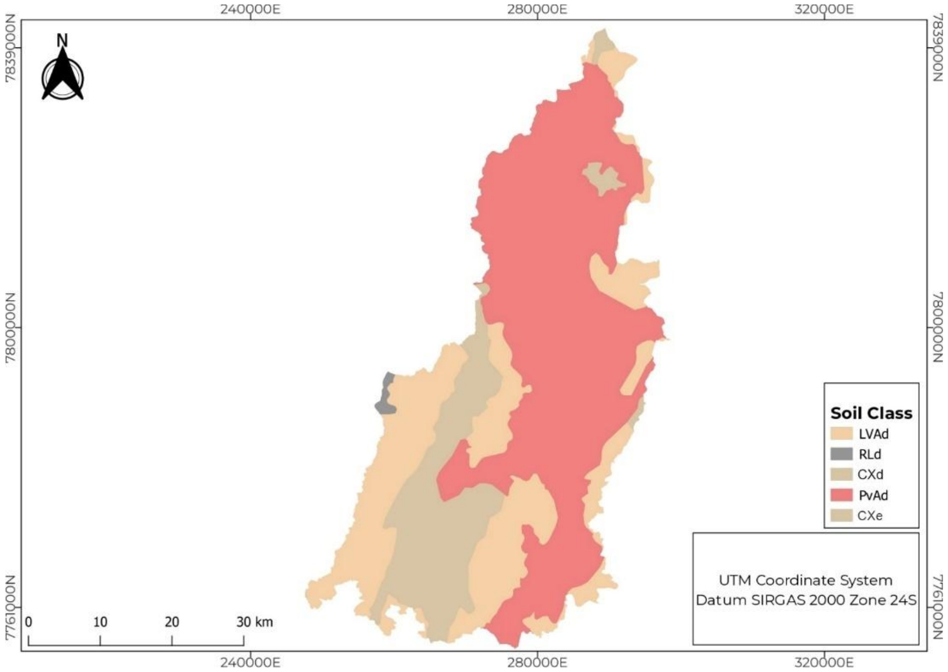


Figure 2. Soil spatial distribution at Guandu River Basin.

According to the Köppen climate system, most of the state of Espírito Santo has a humid tropical climate (Am), with average temperatures above 18ºC (Milagre et al., 2024). With maritime and continental influences, as well as a location and topography susceptible to desertification processes, the study area has a diverse climate with well-defined periods.

The land use map was developed from LandSat multispectral sensor images with a spatial resolution of 30 meters and referring to 2012 and 2021, made available by the Ministry of the

Environment (Figure 3). Its land cover classes are listed in Table 1. The classification of land use and management was carried out in two stages. First, a supervised classification was generated using the Support Vector Machine (SVM) image classification algorithm, with object-based orientation. It was then reclassified by manually editing the land use classes, with the aim of correcting any flaws in the algorithm and improving the accuracy of the mapping.

Table 1. Land use cover classes in the Guandu River Basin in 2012 and 2021.

Land Cover Class	2012	2021	Land Cover Class	2012	2021
	Area (km²)	Area (km²)		Area (km²)	Area (km²)
Pasture	985.22	1016.22	Wetland	5.42	3.82
Native vegetation	550.71	708.85	Mining	1.14	0.80
Urbanization	6.26	14.77	Campo rupestre	1.10	0.78
Annual crop	35.47	12.19	Pinus	1.21	0.85
Water	7.51	7.51	Rubber tree	0.10	0.07
Coffee	285.01	200.89	Eucalyptus tree	94.11	66.33
Bare soil	75.46	42.63	Rock outcrop	80.60	56.82
Sugar cane	0.26	0.18	Perennial crop	10.48	7.39

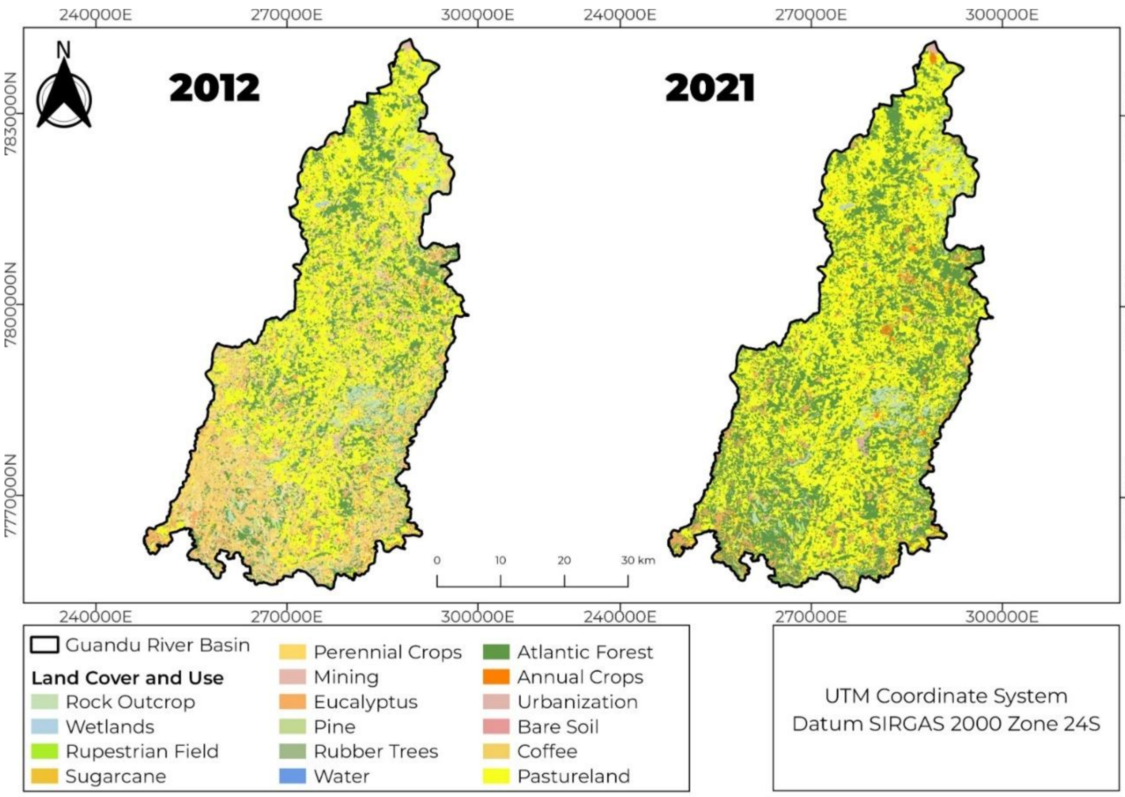


Figure 3. Spatial distribution of land cover classes for 2012 and 2021 in the Guandu River Basin.

RUSLE model

The RUSLE model was structured in a GIS environment, allowing individual maps of the variables to be generated in spatialized form, with the aim of applying map algebra to identify the areas most vulnerable to water erosion in each selected period. The RUSLE is part of an adaptation and refinement of the USLE developed by Renard et al. (1997) to allow it to be used at the river basin level by adjusting the calculation of the topographic factor. Its structure is as follows:

$$Y = R \cdot K \cdot LS \cdot C \cdot P$$

Where Y representes soil loss ($t \cdot ha^{-1} \cdot year^{-1}$), R is rainfall erosivity ($MJ \cdot mm \cdot ha^{-1} \cdot h^{-1} \cdot year^{-1}$), K is soil erodibility ($t \cdot h \cdot MJ^{-1} \cdot mm^{-1}$), LS represents the topographic factor (dimensionless), C denotes land use and management (dimensionless) and P represents conservation practices (dimensionless).

Rainfall erosivity fator (R)

Conceptually, erosivity represents the potential for rainfall to promote erosion by detaching solid particles from the soil due to the kinetic energy of rainfall with an intensity of up to 30 consecutive minutes (Wischmeier and Smith, 1978). Data from 13 rainfall stations, both inside and outside the area, were used to estimate the rainfall, weighting them using the Thiessen polygon method. The modified Fournier index was applied by using local coefficients (Oliveira et al., 2009) as described in Equations 2 and 3:

$$MFI = \frac{1}{N}\sum_{j=1}^{N}\sum_{i=1}^{12} p_{ij}^2/P_j \quad (2)$$

$$R = 129.39 \cdot MFI^{0.7982} \quad (3)$$

Where R is rainfall erosivity (MJ·mm·ha^{-1}·h^{-1}·year^{-1}), MFI is the modified Fournier index, p_{ij} is total monthly rainfall (mm), P_j is total annual rainfall (mm), and N is the number of years in the historical series.

Soil erodibility factor (K)

The soil erodibility factor represents the soil's susceptibility to water erosion, which can be determined directly in the field during rainfall events on experimental plots or through empirical methods using information on texture and organic matter content. The K values applied to this study and their respective citations are shown in Table 2.

Table 2. Soil erodibility classes occurring in the Guandu River Basin.

Soil	Code	K (t·h·MJ^{-1}·mm^{-1})	Source
Red argisol	PvAd	0.0130	Sá et al. (2004)
Dystrophic haplic cambisol	CXd	0.0240	EMBRAPA (1984)
Eutrophic haplic cambisol	CXe	0.0350	EMBRAPA (1984)
Red-yellow oxisol	LVAd	0.0112	Mannigel et al. (2002)
Lithic ustorthents	RLd	0.0440	Pereira and Cabral (2021)

Topographic factor (LS)

To calculate the LS factor, the methodology proposed by Hagras (2025) was used, replacing the accumulated area by the product between the accumulated flow and the cell size, according to Moore and Bruch (1986) procedures and shown in Equation 4:

$$LS = \left[FA \cdot \left(\frac{CS}{22.13}\right)\right]^{0.4} \cdot \left(\frac{sen(S)}{0.0896}\right)^{1.3} \quad (4)$$

Where FA represents the flow accumulation or contribution area, CS is the cell size of the digital elevation model, corresponding to 30 meters of spatial resolution of the flow accumulation image and S is the slope in radians calculated for each pixel.

It should be noted that the value of the denominator of the second member of Equation 4 refers to the sine of the angle corresponding to the slope of the standard plot from which the LS factor was initially derived (9 cm·m^{-1}).

Land use management (C) and Conservation practices factor (P)

According to Wischmeier and Smith (1978), the P factor represents the cultural practices that can contribute to water erosion management. However, given the impossibility of obtaining this factor from satellite images, a similar procedure to those of Oliveira et al. (2014) and Durães et al. (2016) was followed, where the P value was equal to 1.

On the other hand, the C values in this study were obtained from the available literature for the existing uses in the basin, considering that for the exposed soils class the C value is equal to 1. The values for the other land covers and land uses are shown with their respective sources in Table 3.

Table 3. C factor values for land use and land cover classes at Guandu River Basin.

Land Cover	Factor C	Source
Pasture	0.09	Martins et al. (2010)
Native vegetation	0.001	Beskow, Mello and Norton (2009)
Urbanization	0	-
Annual crop	0.25	Bertoni and Lombard Neto (2005)

Water	0	-
Coffee-growing	0.11	Prochnow et al. (2005)
Bare soil	1	-
Sugar cane	0.05	Donzeli et al. (1992)
Wetland	0	-
Mining	1	-
Campo rupestre	0.1	Quinto et al. (2013)
Pinus	0.3	Martins et al. (2010)
Rubber tree	0.032	Silva (2007)
Eucalyptus tree	0.3	Ozsoy et al. (2012)
Rock outcrop	0.001	Ribeiro and Alves (2008)
Perennial crop	0.29	Ruhoff et al. (2006)

Sediment delivery rate (SDR)

According to Walling (1983), the sediment delivery rate is defined as the ratio between the sediment transported (average value obtained by solid discharge or observed erosion) to the control section and the average potential water erosion estimated for the basin, expressed as follows:

$$SDR = \frac{Y_{obs}}{Y_{est}} \quad (5)$$

In which, SDR is the sediment delivery rate (dimensionless), Y_{obs} represents the average water erosion observed in the control section (t·ha^{-1}·year^{-1}) and Y_{est} is the average potential water erosion estimated from the RUSLE (t·ha^{-1}·year^{-1}).

To estimate the GRB's sediment yield and subsequently its SDR, it is necessary to determine the sediment rating curve, which is a fitted relationship between river discharge (Q) and suspended-sediment concentration (C_{ss}). This procedure is commonly used to assess patterns and trends in river water quality, and this study was based on the monitored sediment data provided by the National Water Agency and Sanitation (ANA) at 3 gauges stations in the basin (Table 4). Equations 6, 7 and 8 represent the Afonso Cláudio Montante, Laranja da Terra and Baixo Guandu stations, respectively (Figure 4).

$$Q_{ss} = 0.9115 \cdot Q^{1.8929} \quad R^2 = 0.89 \quad (6)$$
$$Q_{ss} = 0.2702 \cdot Q^{2.2639} \quad R^2 = 0.82 \quad (7)$$
$$Q_{ss} = 0.2826 \cdot Q^{2.2501} \quad R^2 = 0.79 \quad (8)$$

Where, Q_{ss} is the solid discharge (t·day^{-1}) and Q is the daily average flow (m^3·s^{-1}).

Table 4. Sediment gauges at Guandu River Basin.

Name	Code ANA	Area (km²)	Lat.	Long.	Time series
Afonso Cláudio Montante (ACM)	56990990	435	-20.0775	-41.1242	1998-2022
Laranja da Terra (LT)	56991500	1330	-19.9011	-41.0581	1998-2022
Baixo Guandu (BG)	56992000	2141	-19.5236	-41.0142	1998-2022

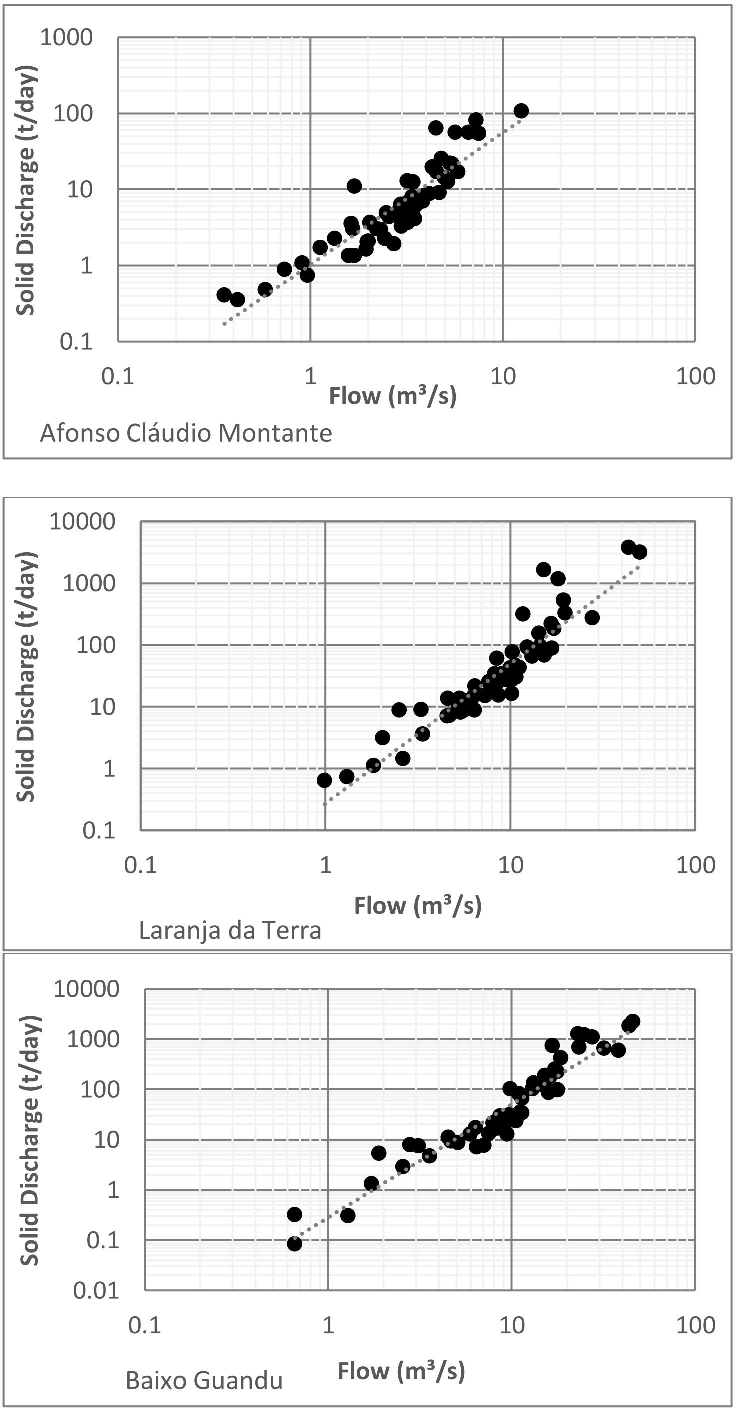


Figure 4. Sediment rating curve

Results and Discussion

The Guandu river basin has two well-defined seasons: rainy and dry. The rainy season begins in October and lasts until March, while the dry season lasts from April to September. Average annual rainfall in the basin ranged from 808 to 1150 mm for the years 2012 and 2021, respectively. These average annual rainfall values are close to those found in the work of Schwamback et al. (2020), which ranged from 770 mm to 1297 mm.

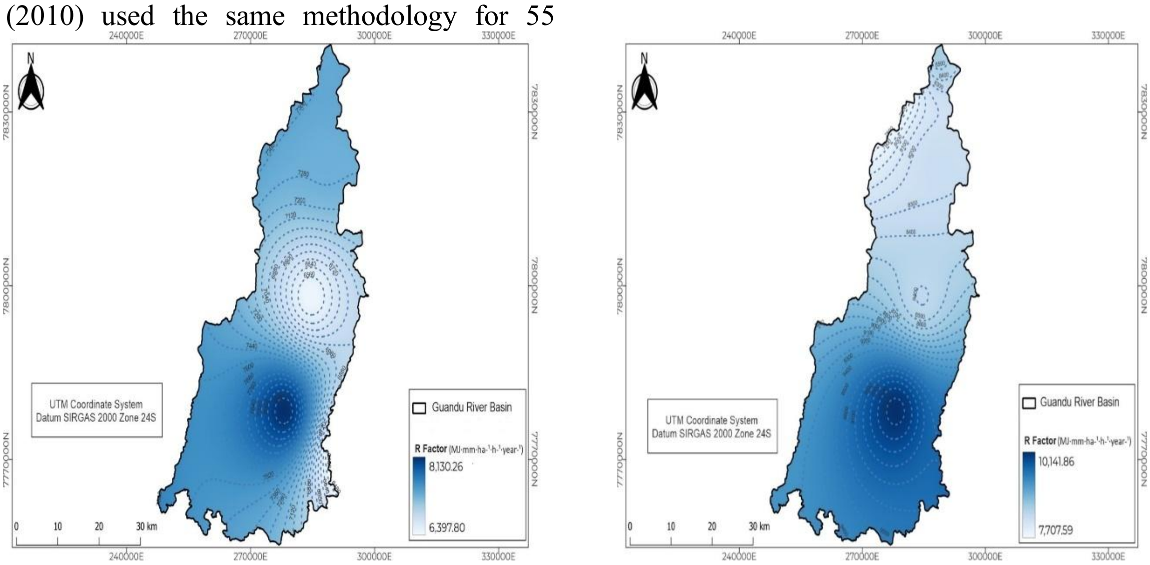
Applying Equation 3 to the rainfall data allowed the erosivity to be estimated for each station, ranging from 6,397 to 8,130 $MJ \cdot mm \cdot ha^{-1} \cdot h^{-1} \cdot year^{-1}$ for the year 2012 and from 7,707 to 10,141 $MJ \cdot mm \cdot ha^{-1} \cdot h^{-1} \cdot year^{-1}$ for the year 2021.

The spatial distribution of erosivity (Figure 6) shows that the highest values were found in the upper Guandu region, influenced by the topographic factor and the higher altitude. According to the interpretation of the erosivity index adopted by Carvalho (2008) and Foster et al.

(1981), GRB's erosivity is classified as "moderate" to "very high".

Applying the RUSLE model to evaluate the water erosion in the Espírito Santo state, Teodoro et al. (2023) observed that the erosivity varied between 4,843 and 7,783 MJ·mm·ha^{-1}·h^{-1}·year^{-1} with a mean value of 6,012 MJ·mm·ha^{-1}·h^{-1}·year^{-1}, which agrees with the values found by this study. According to these authors, the greater erosivity of some points can be associated with higher altitude, indicating areas of more intense rainfall.

Ribeiro (2012) applied the methodology proposed by Bertoni and Lombardi Neto (2005) to seven rainfall stations in the GRB, with a historical series of 64 years, observing that erosivity varied from 5220 to 6986 MJ·mm·ha^{-1}·h^{-1}·year^{-1}. Silva et al. (2010) used the same methodology for 55 rainfall stations in the state of Espírito Santo, with a minimum historical series of 33 years. The authors found that the average variation in rainfall erosivity was between 5665 and 6813 MJ·mm·ha^{-1}·h^{-1}·year^{-1}. On the other hand, Schwamback et al. (2020) applied a methodology based on the equation developed by Oliveira et al. (2009), suitable for the Doce River region, and found values between 7297 and 8948 MJ·mm·ha^{-1}·h^{-1}·year^{-1}. The difference between the values obtained in the works by Schwamback et al. (2020), Ribeiro (2012) and Silva et al. (2010) and this study can be attributed to the number of years used to obtain the average annual erosivity and the increase in rainfall volume in the southeast region in the summer months over the last decade (Ferreira, 2023).



(a) (b)

Figure 5. Rainfall erosivity for 2012 (a) and 2021 (b) in Guandu River Basin.

Once the RUSLE variables were established and geospatialized, the potential water erosion in the basin was then calculated, also considering the estimates for the Upper, Middle and Lower Guandu, through the map algebra technique represented in Equation 1. The spatial distribution of water erosion vulnerability as estimated using RUSLE is presented in Figure 6 and classified ranging from Slight to Extremely High, according to Durães et al. (2016). Table 5 shows the final distribution of soil erosion classes in the GRB.

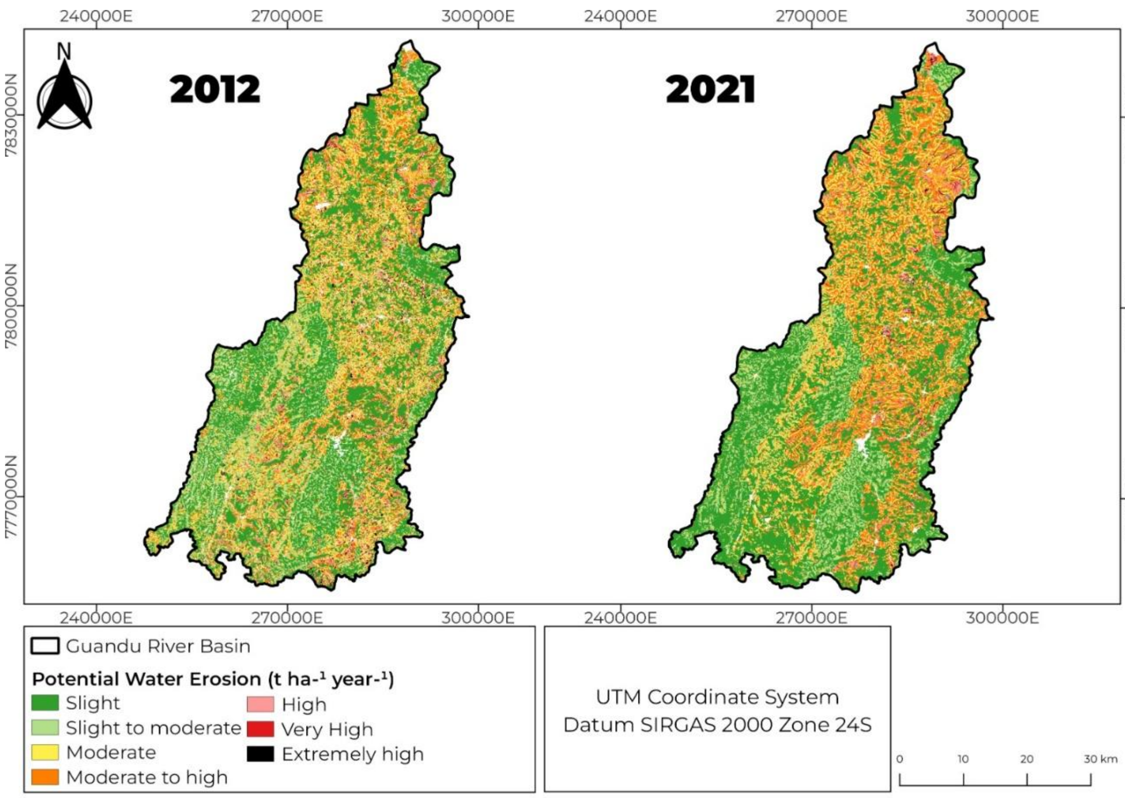


Figure 6. Water erosion distribution at Guandu River Basin in 2012 (left) and 2021 (right).

Table 5. Simulated soil loss interval for current soil use in the Guandu River Basin.

Soil loss (t·ha^{-1}·year^{-1})	Classification	Area (%)					
		ACM		LT		BG	
		2012	2021	2012	2021	2012	2021
0-2.5	Slight	44.29	54.89	43.29	51.55	40.72	43.99
2.5-5	Slight to moderate	17.55	15.56	18.87	14.67	15.14	11.53
5-10	Moderate	16.44	11.01	17.74	13.16	19.33	15.73
10-15	Moderate to high	8.32	9.04	8.39	9.97	14.20	19.17
15-25	High	8.25	4.95	7.01	5.63	7.39	7.21
25-100	Very high	4.02	4.36	3.92	4.79	2.31	2.03
>100	Extremely high	1.13	0.19	0.78	0.23	0.91	0.34

Modelled soil losses across the basin for the year 2012 presented an average of 12.38 t·ha^{-1}·year^{-1} ranging from 0.022 t·ha^{-1}·year^{-1} to 2281.48 t·ha^{-1}·year^{-1}. In comparison, the estimated soil losses for the year 2021 showed a reduction of approximately 11.4% in the average potential water erosion due to the changes in land cover. An increase of approximately 3% in pasture class and 28% in forest class was observed, combined with a reduction in annual and perennial agriculture, bare soil and other land cover classes which resulted in an average value of 10.97 t·ha^{-1}·year^{-1}, ranging from 0.028 to 829.8 t·ha^{-1}·year^{-1}.

In terms of potential erosion for the Afonso Cláudio subbasin, an average value of 12.5 t·ha^{-1}·year^{-1} was estimated for the year 2012, while for the year 2021 this value was 7.9 t·ha^{-1}·year^{-1}. For the Laranja da Terra subbasin, the estimated average values for 2012 and 2021 were 11.34 t·ha^{-1}·year^{-1} and 8.91 t·ha^{-1}·year^{-1}, respectively.

This reduction in potential erosion in the sub-basins can be correlated with the change in land cover over the period in question. This change in land use, especially in the exposed soil and agriculture classes, shows that even with the increase in rainfall erosivity, afforestation and grassland restoration play a major role in erosion control (Han et al., 2023) by reducing rain splash effects, covering the soil and decreasing soil particulates to detachment (Hong et al., 2020).

For Lense et al. (2021), management practices, changes in land use, and vegetation cover maintenance are important to reduce erosion rates, especially in the most vulnerable soil classes according to their erodibility.

According to Milagre et al. (2024), native vegetation coverage (forest and non-forest natural formation) increased approximately 12% in GRB, between 1985 and 2020. One of the reasons for this scenario was the establishment of the reforestation program in Espirito Santo, which began in 2012. This initiative aimed to recover and protect forest fragments to ensure water availability and soil and biodiversity conservation by compensating farmers for the conservation and restoration of over 2100 ha from 2015 to 2019 (SEAMA, 2021).

In addition, it should be noted that the expansion of native vegetation areas has occurred predominantly in areas used for agricultural activities, due to the increased requirements for the environmental regularization of rural properties after the implementation of the new Brazilian forest code in 2012 (Parras et al., 2020).

In terms of sediment yield calculated using the sediment rating curve, it is important to highlight that this procedure stems from the fact that daily records and sampling are economically unfeasible in large basins. However, even assuming that flow discharge is a good indicator of sediment concentration, this technique can be inaccurate (Hicks et al., 2004) due to the influence of some variables in the sediment transport process that vary in space and time. On the other hand, this technique is widely used in larger basins because the fluctuations between the maximum and minimum daily average flows are not significant (Córdova and González, 1997).

Considering the assumptions mentioned above, the sediment yield observed from the monitoring stations ranged from 0.26 $t \cdot ha^{-1} \cdot year^{-1}$ to 0.40 $t \cdot ha^{-1} \cdot year^{-1}$ in the years 2012 and 2021, respectively, in Afonso Cláudio Montante. The main characteristics of the basin delimited up to this point are agriculture and stretches with higher relief, contributing to sediment transport in these places.

In the middle Guandu, represented by the Laranja da Terra station, sediment yield was 1.14 $t \cdot ha^{-1} \cdot year^{-1}$ in 2012 and 0.58 $t \cdot ha^{-1} \cdot year^{-1}$ in 2021. In this section, there was a decrease in the pasture area and an increase in the forest class. According to Martins et al. (2021), an increase in forest cover can contribute to a reduction of more than 30% in sediment transport, either by reducing the effect of rain erosion or direct surface runoff.

For the Baixo Guandu station, there was also a reduction in sediment production. In 2012, sediment yield was 1.68 $t \cdot ha^{-1} \cdot year^{-1}$, while in 2021 it was 0.76 $t \cdot ha^{-1} \cdot year^{-1}$, a reduction of 45%. At this point, there is a smoother terrain and lower rainfall erosivity factor that contribute to lower soil and nutrient loss (Wang et al., 2023), as well as the restoration of the forest area. These reductions in solid discharge observed for the middle and lower GRB in 2021 compared with 2012 indicate that the actions proposed under the national environmental recovery plan through the NVPL have had a positive effect.

According to Martins et al. (2021), agricultural land use increases soil erosion, as well as contributing to the pollution of water by the sediments transported from upstream. On the other hand, Bendito et al. (2023) showed that in different conservation scenarios, ecological restoration and conservation practices in a watershed in the Brazilian savanna reduced soil loss and sediment export. In this respect, the well-managed afforestation of degraded land contributes to soil conservation (Valente et al., 2021), while the recovery and management of pastureland, although requiring appropriate management practices, conserves the soil and, above all, water resources by reducing their turbidity and siltation.

Soares et al. (2024) modeled soil loss using InVEST software in a watershed located in the state of Minas Gerais, considering the effect of the impacts of three conservation scenarios in relation to current land use, and according to the authors, while the current scenario showed an average annual erosion of 2.75$t \cdot ha^{-1}$, all the other scenarios that featured reforestation and preservation of riparian forests, together with soil conservation practices, showed a reduction in soil loss, transport and an increase in sediment deposition, contributing as an important alternative in the management of areas degraded by erosion, as also seen in this study.

The results in terms of potential water erosion and sediment yield have made it possible to calculate the SDR and explain the spatial and temporal heterogeneity of the sediment transportation and its interaction with precipitation and the terrain's features across the GRB. The SDR is a scale factor used to accommodate average differences in sediment yield, ranging from 0 to 1. The values found in this study are shown in Table 6.

Table 6. Sediment delivery rate calculated for Guandu River Basin.

Sediment gauge	Y_{obs} ($t \cdot ha^{-1} \cdot year^{-1}$)		Y_{est} ($t \cdot ha^{-1} \cdot year^{-1}$)		SDR	
	2012	2021	2012	2021	2012	2021

ACM	0.26	0.40	12.5	7.9	0.021	0.051
LT	1.14	0.58	11.34	8.91	0.100	0.065
BG	1.68	0.76	12.38	10.97	0.136	0.069

The mean annual value for the SDR was 8.5% for 2012, showing that only 13.6% of the soil loss in the basin (estimated using the RUSLE model) were transported and exported beyond the outflow, even with the lower mean annual rainfall depths and erosivity compared with the year 2021 that presented a mean annual value for the SDR of 6.2% and with up to 6.9% of the sediments reaching the outflow. Therefore, this may indicate conditions of low hydrological connectivity (Santos et al., 2017) and high influence of land cover, considering overgrazing identified in 2012.

The values calculated for the GRB vary in dimension between subbasins of 2.1% to 13.6% for 2012 and 5.1 to 6.9% in 2021. These spatial variabilities have been observed in other studies, such as that of Vente et al. (2008), who found an SDR ranging from 0.03 to 55% in 61 subbasins in Spain, and Verstraeten et al. (2007), who estimated SDR values of 20% and 39% for basins from 167 up to 2173 km² in Australia.

In relation to Brazilian watersheds, it is important to highlight the findings of Silva and Schulz (2007), who studied the hydrosedimentological dynamic in the Água Fria River Basin in the state of Tocantins and obtained an average SDR value of 6.2%. Beskow et al. (2009), examining the Rio Grande River Basin in the state of Minas Gerais, found an average value of 1.62% for a drainage area greater than 6,000 km². Meanwhile, Durães et al. (2016) carried out a study to provide important information for measures associated with the management, conservation and planning of land use in the state Paraná, Brazil and estimated SDR values varying from 6.6% to 88%.

The natural factors that control the sediment transport process can be summarized in seven categories, namely, hydrology, topography, basin shape, soil texture, geomorphology, vegetation cover and catchment area (Wu et al., 2017). According to Lin et al. (2018), vegetation cover has mitigating effects on sediment transport in a catchment. For this reason, Silva et al. (2021) claimed that the selection of suitable forage for land susceptible to erosion is crucial when it comes to reducing the environmental footprint of the livestock sector and improving soil and water conservation.

Furthermore, as the catchment area is inversely proportional to the SDR value (Wang et al., 2016) and depends on its landforms (Jing et al., 2023), the probability of sediment deposition in a large catchment is greater than in a small catchment under the same circumstances (Zhang et al., 2015).

In this context, Jovino et al. (2022) mentioned that payment for environmental services (PES) is intended to increase farmers' income by replacing their agricultural and livestock activities with less aggressive agriculture through the implementation of programs and public policies to promote environmental restoration. Consequently, this reduces sediment transport, as established in the NVPL and achieved to a certain degree in the GRB.

The results found here show that alterations in land use policies can significantly affect the hydrological dynamic of drainage basins in terms of surface runoff and its sediment transport process. This alteration of the runoff can lead to better control of sediment transportation due the reforestation and mitigation plan, with consequences in a reduced erosion process and increase in water storage in deep soil layers and the replenishment of aquifers.

Conclusions

This study explored the efficiency of implementing an environmental recovery plan for agricultural areas in controlling soil erosion, based on mathematical modeling of water erosion and the rate of sediment input. We found that estimated soil erosion showed a decreasing trend, while rainfall erosivity was at its highest in the year 2021, with an average value of 8,578 $MJ \cdot mm \cdot ha^{-1} \cdot h^{-1} \cdot year^{-1}$.

The spatial distribution of water erosion showed that there was a reduction in the areas most prone to erosion, especially in the southwestern regions of the GRB, with the slight and slight to moderate classes being predominant, comprising more than 55% of the entire area.

In relation to sediment transport observed from the sediment gauges analyzed, a downward trend was observed in all of them, showing the positive effect of the recovery of degraded areas and the policy of payment for environmental services adopted in the basin during the period in question, demonstrating the positive benefits of ecological restoration on water erosion and showing that the benefits of afforestation were stronger than grassland restoration.

We suggest that future ecological restoration focus more on areas where dominant soil erosion types are changing, as these areas are sensitive to climate change and anthropogenic activities. The decision makers can optimize ecological restoration strategies by selecting appropriate vegetation restoration measures and coordinating vegetation restoration with water balance, human well-being, and climate change to alleviate land degradation and promote the sustainable development of the region following public policies that improve farmers' income and nature preservation and restoration.

The results of this study may contribute to the development of policies based on ecological restoration at different scale, guiding sustainable land-management strategies that encourage conservation and sustainable land use in the Espírito Santo state.

Acknowledgements

The authors would like to thank the Laboratory of Climatology and Water Resources (LCRH) at the Universidade Federal de Uberlândia – UFU, for their assistance in analyzing the data and providing the physical space for conducting this research.

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