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MATHURIN FRANÇOIS

EVALUATION OF ECOSYSTEM SERVICES RELATED TO NATURAL FORESTS IN THE CONTEXT OF EROSION CONTROL IN SOUTHERN BAHIA

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Supervisor: Prof. Dr. Eduardo Mariano Neto

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Didicated to:

My family, friends, former professors, supervisor, co-supervisors, and mentors who have supported me along this long scientific journey.

« S’il arrive que tu tombes, apprends vite à chevaucher la chute.
Que ta chute devienne ton cheval pour continuer ton voyage ».

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SUMMARY

L’espoir est un fluide nécessaire à l’homme comme l’eau à la terre. Il déclenche des forces insoupconnées de la nature humaine.

(Lauréanne Harvey)

General abstract

The Atlantic Forest is one of the most important biomes in the world due to its diversity, but it has suffered accelerated degradation since the colonization period. Southern Bahia is a special case where forests were, in part, replaced by cacao agroforestry systems. The Atlantic Forest has some positive aspects regarding the maintenance of a portion of biodiversity, ecosystem processes, and services, such as those linked to soil protection and erosion control; however, its roles in ecosystem services have not yet been measured. As forests and agroforestry play a large role in water availability in the watersheds of southern Bahia, it is important to estimate the services associated with erosion control of these forests. This research comprises seven parts: Firstly, the general introduction, which covers contextualization. Here, each chapter refers to an article. Secondly, a state-of-the-art review examining experiences in adopting cacao certification in Brazil (**Chapter I**). This chapter underscores that Rainforest Alliance (RA), Forest Stewardship Council (FSC), and *Instituto Biodinâmico* (IBD) are among the most commonly used certifications for cacao products in the country. Furthermore, this chapter outlines that bureaucracy, the high cost of certified products, and meeting FSC requirements are among the main barriers to certification in Brazil. Thirdly, Interactions between forest cover and watershed hydrology: A state-of-the-art review (**Chapter II**). In this paper, we analyzed the controversy that exists in the literature regarding the positive and negative effects of forest cover at the watershed level on soil water availability. In the hydrological cycle, trees act positively through water percolation, reduction in runoff, and other effects, but they can also reduce soil water availability through evapotranspiration. Controversy arises because this is a complex system with several nonlinear relationship factors, including tree density, geology, geomorphology, successional development, and tree characteristics. The fourth part of this study is the chapter entitled: Impacts of cacao agroforestry systems on climate change, soil conservation, and water resources: A review (**Chapter III**). This chapter showed the roles of cacao agroforestry systems in climate change mitigation, soil conservation, and water decontamination. Additionally, it showed that a cacao agroforestry system can be used as a phytoremediation technique for soil decontamination to reduce aluminum (Al), mercury (Hg), and cadmium (Cd). The fifth part of this thesis is: Assessing soil erosion and its drivers in agricultural landscapes: the case of southern Bahia, Brazil (**Chapter IV**). We hypothesize that areas with cacao plantations and forests suffer less erosion and soil loss than agricultural areas, and these ecological services in such a large area as southern Bahia can have considerable value. We found that areas with forests had soil loss approximately two times less than non-forested areas. The last chapter entitled: Assessing the global sensitivity of the Revised Universal Soil Loss Equation (RUSLE) factors: a case study of southern Bahia, Brazil (**Chapter V**). The results of this chapter showed that slope angle was the most influential factor impacting the soil loss assessment in our study when using the RUSLE model. Additionally, this chapter demonstrates that soil erosion assessment is only a nominal value. It depends on a set of factors, including the equation used for calculating the topographic factor. Finally, we presented the general conclusion of these five articles. The findings of this research can help planners and decision-makers to take suitable measures to mitigate the problems of water pollution, erosion, and the impact of natural forests on a watershed.

Keywords: Ecosystem services; water quality; water protection; reduce runoff, trees; soil loss

Resumo geral

A Mata Atlântica é um dos biomas mais importantes do mundo devido à sua diversidade, mas sua degradação começou desde o período da colonização. O sul da Bahia, particularmente a Mata Atlântica, é um caso especial onde as florestas foram, em parte, substituídas por sistemas agroflorestais cacaueiros. Vale ressaltar que a Mata Atlântica apresenta alguns aspectos positivos no que diz respeito à manutenção de uma parcela da biodiversidade, dos processos e serviços ecossistêmicos, como os ligados à proteção do solo e à contenção da erosão; no entanto, o seu papel nos serviços ecossistêmicos ainda não foi medido. Como as florestas e a agrofloresta desempenham um papel importante na disponibilidade hídrica nas bacias hidrográficas do sul da Bahia, é importante estimar os serviços associados à contenção da erosão desta floresta. Assim, esta pesquisa tem sete partes: Primeiramente, a introdução geral, que abrange a contextualização. Nesta tese, cada capítulo é um artigo. Em segundo lugar, uma revisão do estado da arte sobre o exame das experiências na adoção da certificação de cacau no Brasil (**Capítulo I**). Este capítulo destacou que *Rainforest Alliance* (RA), *Forest Stewardship Council* (FSC) e instituto biodinâmico (IBD) estão entre os tipos de certificação mais utilizados para produtos de cacau no país. Além disso, este capítulo destacou que a burocracia, o alto custo dos produtos certificados e o cumprimento dos requisitos do FSC estão entre as principais barreiras à certificação no Brasil. Em terceiro lugar, interações entre a cobertura florestal e a hidrologia de bacias hidrográficas : Uma revisão do estado da arte (**Capítulo II**). Neste artigo, analisamos a controvérsia existente na literatura sobre os efeitos positivos e negativos da cobertura florestal no nível da bacia hidrográfica sobre a disponibilidade de água no solo. No ciclo hidrológico as árvores atuam positivamente através da percolação da água, redução do escoamento superficial e outros efeitos, mas também podem reduzir a disponibilidade de água no solo através da evapotranspiração. A controvérsia surge porque este é um sistema complexo com vários fatores de relacionamento não linear, incluindo densidade das árvores, geologia, geomorfologia,

desenvolvimento sucessional e as características da árvore. A quarta parte deste estudo é um artigo intitulado: Impactos dos sistemas agroflorestais de cacau nas mudanças climáticas, conservação do solo e recursos hídricos: uma revisão (**Capítulo III**). Este capítulo mostrou o papel dos sistemas agroflorestais de cacau na mitigação das mudanças climáticas, na conservação do solo e na descontaminação da água. Além disso, mostrou que o sistema agroflorestal de cacau pode ser utilizado como técnica de fitorremediação para descontaminação do solo para redução de alumínio (Al), mercúrio (Hg) e cádmio (Cd). O quarto capítulo desta tese foi intitulado : Avaliação da erosão do solo e seus fatores determinantes em paisagens agrícolas : o caso do sul da Bahia, Brasil (**Capítulo IV**). Nossa hipótese é que áreas com plantações de cacau e florestas sofrem menos erosão e perda de solo do que áreas agrícolas, e esses serviços ecológicos em uma área tão grande como o sul da Bahia podem ter um valor considerável. Descobrimos que as áreas com florestas tiveram uma perda de solo aproximadamente duas vezes menor do que as áreas não florestadas. O último capítulo foi intitulado : Avaliação da sensibilidade global dos fatores da Equação Universal Revisada de Perda de Solo (RUSLE): um estudo de caso do sul da Bahia, Brasil (**Capítulo V**). Os resultados deste capítulo mostram que a inclinação angular é o fator de maior influência que impacta a avaliação da perda de solo do nosso resultado quando utilizado o modelo da RUSLE. Além disso, este capítulo demonstrou que a avaliação da erosão do solo é apenas um valor nominal que depende de um conjunto de fatores, incluindo a equação utilizada para o cálculo do factor topográfico. Por fim, a conclusão geral dos cinco artigos foram apresentado no final da tese. As conclusões desta pesquisa podem ajudar planejadores e tomadores de decisão a tomar medidas adequadas para mitigar os problemas de poluição da água, erosão e o impacto das florestas naturais em uma bacia hidrográfica.

Palavras-chave: Serviços ecossistêmicos; qualidade da água; proteção da água; redução do escoamento superficial, árvores; perda de solo

1. General introduction

Ecosystem services (ES) are defined as "the conditions and processes through which ecosystems and biodiversity sustain and fulfill human life" (Brauman and Daily, 2014). From the viewpoint of the Millennium Ecosystem Assessment, ES are the benefits that humans obtain from natural ecosystems (Cardona, 2012), but they are limited and threatened by overexploitation (Harrison Hester, 2010). Services provided by the ecosystems allow the conservation of water supply, air purification, soil formation, pollination, food and fruit production, and climate change control. In this case, ES are particularly linked to the economy (Aziz and Cappellen, 2019), supporting sustainable development. Several techniques, such as flood control, pollination of crops, purification of water, and reduction of the climate have been engineered for the substitution of ES, but no technologies have been developed for their replacement yet. ES are capable of maintaining biodiversity in the habitat and reducing degradation linked to environmental problems, including soil erosion, which can be calculated using the Revised Universal Soil Loss Equation (RUSLE). It is noted that RUSLE is constituted of the following factors: the rainfall erosivity factor (R), the soil erodibility factor (K), the topographic factors (L and S), and the cropping management factors (C and P). It is noteworthy that RUSLE has a set of uncertainties and sensitivities for each factor that constitutes it.

From a biodiversity conservation standpoint, there is a need to preserve the Brazilian Atlantic Forest, particularly in the southern Bahia (Delabie et al., 2011). In the Atlantic Forest, particularly in southern Bahia, cacao is the most commercialized crop, and around 70% of the plantation is conducted under an agroforestry system known as cabruca (Morato, 2020), which is a type of cultivation system within the forestry complex that favors greater maintenance of biodiversity (Vieira et al., 2021). Part of this cocoa is in cabrucas, which plays an important role in biodiversity conservation, particularly fauna and flora (Cassano et al., 2009). In a traditional system known as cabrucas– cacao scrubs grow under the shade of a few native trees, and have a potential economic (Delabie et al., 2011) and provide a high diversity of organisms (Bisseleua et al., 2009). The native forests and these agroforests comprise a large extent (~600.000ha) of the remaining forest cover and they provide habitat and landscape-scale connectivity for many species of the regional biota (Landau et al., 2003). Despite the importance of Southern Bahia for conservation as well as diversity, the Atlantic Forest is reduced to 7% of its original native area due to deforestation (FAO and UNEP, 2020). Despite the great importance of cacao cultivation in the southern region of Bahia, there are few studies in the literature highlighting the certification types used for cacao products in Brazil. Also, the barriers to the certification of cacao products are still unclear.

Vegetation around rivers and springs have a fundamental role in controlling erosion processes, mitigating the contribution of sediments and toxic substances to rivers and springs, increasing the percolation of water into the subsoil, and decreasing surface runoff, among other effects on water regimes (Makarieva et al., 2006; Ditt et al., 2010). To cope with pesticides in the river, erosion and others, it is possible to recourse to ecological restoration, which has become essential to recover not only biological diversity but also critical ecosystem services (De Groot et al., 2013). The wood vegetation, for example, in riparian zones improves the groundwater quality in the no-tillage system in temperate climates (Aguiar et al., 2015) and has the ability to retain contaminants (Silva et al., 2020).

Water services are essential for maintaining biodiversity as climate change mitigation is essential for maintaining water services. The control of silting up is related to a decrease in the cost of water treatment and maintenance and the longevity of supply reservoirs, and an economy that has calculated values may not be neglected (Strassburg et al., 2016). The East and Southeast Atlantic basins are already vulnerable in terms of human supply water storage capacity (ANA, 2013).

Recently, SOS Mata Atlantic reported that only 6% of the rivers in the Atlantic Forest biome have good water quality (SOS Mata Atlantic, 2019). The type of land use and soil type are among the factors that influence water quality at the watershed scale (Jabbar et al., 2019). In such a case, researchers should turn their attention to seeking sustainable solutions based on ES.

The role of the forests in the hydrologic cycle is complex; the type of trees, their density, the climate, and the type of soil have great influence on water availability. Studies showed that forests could either increase or decrease water availability (Ilstedt et al., 2016), depending on the influence of the climate and type of the soils (Cao et al., 2011). Tree plantations lsuch as *Populus* and *Salix* may decrease the water yield (Stromberg et al., 2007). Therefore, it is necessary to investigate more on the types of certifications utilized in Brazil for cacao products, the role of cacao agroforestry systems for soil conservation and climate mitigation, the relationship between tropical forests and water cycles, soil loss assessment in forested and non-forested areas in the southern part of Bahia, and finally determine the most influential factor of the soil loss assessment.

1.1 General objective

- To assess ecosystem services related to forest certification and natural forests in the context of environmental sustainability.

1.1.1. Specific objectives

- To provide an overview of the experiences with forest certification in Brazil and the types of certifications used for cacao products (Paper I);

- To create a conceptual model for understanding the role of trees in the hydrologic cycle and examine the conditions under which they can be an element that increases or decreases the water supply in tropical regions (Paper II);

- To revise the influence of agroforestry systems on climate change, soil conservation, and groundwater quality (Paper III);

- To assess soil loss in landscapes of Cachoeira River watershed, in southern Bahia, Brazil (Paper IV);

- To do a global sensitivity analysis of the RUSLE as part of a soil erosion assessment in southern Bahia, Brazil (Paper V).

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Chapter I

Experiences in adopting cacao certification in Brazil: A state-of-the-art review

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Experiences in adopting cacao certification in Brazil: A state-of-the-art review

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Abstract

Forest certification is a mechanism for guaranteeing the quality of products on the market and maintaining environmental sustainability. This state-of-the-art review provided an overview of experiences with forest certification in Brazil and the types of certifications used for cacao products. For this, published documents were searched in international scientific databases, including Google Scholar, Web of Science, and Scopus. The findings showed that even though forest certification was introduced in Brazil long ago, Brazilian enterprises still lag behind. Several types of certifications, including universal trade zone (UTZ), the Rainforest Alliance (RA), Forest Stewardship Council (FSC), and Instituto Biodinâmico (IBD), are used to certify cacao products. However, high costs, bureaucracy, and the country's laws are the main obstacles to the implementation of certification. Additionally, this review found that Brazilian enterprises have positive attitudes and perceptions toward the benefits of certification and are willing to pursue recertification. The research findings could help stakeholders better understand the willingness of Brazilian companies to sell certified products. Further research is indeed required to present the required processes and evaluation processes of the aforementioned certifications used for cacao products.

Keywords: Brazil, Brazilian enterprises, Brazilian laws, deforestation, forest certification

1. Introduction

Forest certification is a private and voluntary market-driven instrument crucial for environmental management to reduce the costs associated with ecosystem services, including water quality. Certification can also be used to certify green products (Gulbrandsen, 2005). The Forest Stewardship Council (FSC), a worldwide accrediting certifier (Vargas, 2010), is an alternative for environmental law enforcement (Ulybina and Fennell, 2013), particularly in countries where governance capacities are insufficient to manage natural resources (Ebeling and Yasué, 2009). It is also an option for the regulation of wood extraction (Polisar et al., 2017), with the purpose of taking social and environmental responsibility and allowing good practice in society (Nysten-Haarala, 2013). Initially, in 1990, the purpose of forest certification was to deal with degradation and deforestation (Rametsteiner and Simula, 2003) and to promote socially beneficial, environmentally friendly, and economically viable forest management worldwide (Boström, 2012; Wolff and Schweinle, 2022). Since its creation, 3.2% of worldwide forests have been certified (Rametsteiner and Simula, 2003). Today, community forests represent just over 1% of the worldwide FSC-certified area (Charnley et al., 2022). This percentage may increase due to the need for the protection of biodiversity, guarantee of consumed products, and importance of achieving the sustainable development goals associated with forests.

Importantly, forest certification is associated with some fundamental factors, including awareness of chain of custody certification and belief in competitiveness (Jayasinghe et al., 2007), and resources (Tikina et al., 2010), and some requirements, including the willingness and education of locals to efficiently take part in achieving these goals. Several aspects, such as practices in forest management, environmental protection, and public affairs, should be clearly improved to receive forest certification (Cubbage et al., 2010; Taha and Jusoff, 2008). Moreover, mapping and respecting high-conservation-value forests, international labor codes, and management of non-timber forest products are also required (Cerutti et al., 2011). Of note, there are some inconsistencies (for example, path-dependent social institutions and local practices) in the outcomes associated with forest certification (Ulybina and Fennell, 2013). Notably, the effects of forest certification are limited in some countries. For example, the utilization of forest certification is limited in Brazil (Spathelf et al., 2004) due to the country's legislation. Currently, Brazil is using several types of certifications, such as FSC and the Brazilian Forest Certification Program (Cerflor) systems. Remarkably, 7,010,025.64 and 4,218,827.15 hectares of Brazilian territory were certified by the FSC and Cerflor, respectively (Meijueiro et al., 2020). Cerflor certifies the sustainability of any species of forest management and promotes social, environmental, and economic development of enterprises (de Almeida, 2015).

There is a gap in the scientific literature regarding the type of product certification used for products, particularly cacao, which is predominantly cultivated in the southern part of the state of Bahia. In addition, there are few papers that review the challenges of implementing of forest certification in developing countries. Therefore, this work presents an overview of the experiences of adopting the certification for cacao in Brazil. Besides, this review seeks to understand the experience of certification of cacao products in the southern state of Bahia. In this review, cacao and forest certification are discussed together because cacao trees are considered part of the region's agroforestry systems. This review is structured as follows: after the introduction, Section 2 presents the methodology, while Section 3 covers experiences with certification. Section 4 focuses on cases of certified cacao products in Brazil, and Section 5 discusses community-based forest enterprises (CFEs) in Brazil. Section 6 addresses future challenges and the way forward, and Section 7 presents the conclusions.

2. Methodology

This state-of-the-art review had a twofold objective: (i) to present an overview of the experiences of adopting certification for cacao in Brazil, and (ii) to understand the experience of cacao product certification in the southern state of Bahia. A set of keywords, such as 'forest certification', 'community-based forest enterprise', 'Certflor', 'southern Bahia', 'Brazil', and 'Forest Stewardship Council', was used to search for published documents in English and Portuguese (e.g., articles, reports, and book chapters) from 1996 to 2023 in international scientific databases, including Scopus, Web of Science, and Google Scholar. The keywords were connected using the Boolean operators 'AND' and 'OR' to refine the search results. Only papers related to at least one of the previously mentioned keywords that met the preestablished criteria were included in this review. It is noted that the reference lists of these documents were also used to search for additional research. After screening the titles and abstracts, information was extracted from the abstracts and conclusions of the selected documents. Finally, Zotero was used for managing data and ensuring the accuracy of the references.

3. Experience of certification in Brazil

Brazil is the second-largest producer of tropical wood in the world, thanks to the Amazonian Forest, which accounts for more than 80% of its timber production (Drigo et al., 2009). Currently, Brazil has two forest certification systems: FCS and Cerflor (Araújo, 2008). However, forest certification is difficult to institutionalize in Brazil (Cashore et al., 2007) because the majority of the country's forests, except in Amazonia, are on private land. Such a situation often causes social conflicts associated with forest exploitation across the country (Rafael, 2019). In the case of product certification, the instances work on green seal projects to boost the competitiveness of the industries in terms of exportation. The mechanism of environmental certification is mainly based on two assumptions, with the purpose of essentially protecting the environment and consumers. The first postulates that there are different ways to produce goods in terms of environmental impacts, whereas the second assumes that cleaner production or certified products are generally more expensive compared to others (Moura, 2013).

3.1.Certified areas in Brazil

Brazil adopted forest certification in 1994, and since then, a total of 5.5 million hectares have been certified by the FSC across 12 states (Keppe et al., 2008). In the country, 2.78 million hectares of native forest were certified, whereas the area of forest plantations was approximately 4 million hectares. A recent study indicated that there were 1363 companies certified by the FSC system and 60 companies certified by Cerflor in the chain of custody (Meijueiro et al., 2020). Studies have pointed out that most of the companies were located in the Southeast region, which had the largest number of certified companies, and also concluded that FSC was the type of certification most commonly used in Brazilian states (Meijueiro et al., 2020; Vargas, 2010). The influence of FSC and Cerflor varied from one state to another. Of note, the influence of FSC and Cerflor in Brazil depends on multiple factors, including external market pressures and industry lobbying (Sundstrom and Henry, 2017).

3.2.The requirement for FSC and Cerflor

3.2.1. The requirement for FSC

Forest certification is an alternative that provides information on the impacts of products from forest management (Islam and Siwar, 2009), enhances sustainable forest management, and creates employment, income, and wealth for the local population. It is noted that the principles are essential rules aiming to ascertain that forest management occurs in an environmentally correct, socially beneficial, and economically feasible manner, whereas the criteria are ways to judge if a principle is reached or not (FSC, 2018). The principles of FSC are described in **Table 1.**

Table 1 The 10 principles of FSC (FSC, 1996)

Principles	Requirements
1	Respect of the laws and principles of the FSC;
2	Tenure and use rights and responsibilities;
3	Rights of indigenous peoples;
4	Community relations and workers' rights;
5	Benefits from the forest;
6	Taking into account the environmental impacts;
7	Management plan ;
8	Monitoring and evaluation;
9	Maintenance of high conservation value of forests;
10	Tree plantations

The requirements are based on international agreements and the national legislation of the country. Notably, forest certification increases forest managers' awareness of their visions of conservation values, management measures, and conservation status monitoring (Buliga and Nichiforel, 2019) and includes an indicator of wood used for energy (Stupak et al., 2007). The greatest barriers to the implementation of certification in Brazil include the capacity to demonstrate compliance with the national legislation of the country on forest management, labor, health, and safety, which are prerequisites for forest certification (Viana et al., 2002). Besides, challenges to the implementation of certification in developing countries can be either direct (i.e., those that can independently prevent the establishment of forest certification) or indirect (i.e., those that can negatively influence forest certification) barriers (Becker and Laaksonen-Craig, 2006). Direct barriers include a lack of land and/or tenure rights, ineffective legislation or policies, poor governance, a weak institutional environment, and the inability to sell certified products. Indirect barriers mainly include forest operation size, institutional influence, political will, consumer buy-in, and others (Becker and Laaksonen-Craig, 2006).

Similarly, Xu and Lu (2021) documented conventional institutions and governance are the main barriers to the implementation of forest certification in developing countries. Besides, a recent study has reported that Principles 4 and 6 are the main challenges in the implementation of the FSC certification process (FSC, 1996; **Table 1**). Forest certification can indirectly offer social benefits to humans, such as ecosystem services and improved air quality. Research carried out in the state of Minas Gerais, Brazil, concluded that forest certification can be considered a mechanism enabling the reinforcement of social and environmental legislation (Basso et al., 2012). New criteria are being developed to protect intact forest landscapes and adapt the standards to current realities (Kleinschroth et al., 2019).

3.2.2. Procedures for obtaining FSC certification

Importantly, certification requires a training program, efforts, and skills from all the employees and contractors who work in the enterprises (Leite et al., 2017), and its obtaining requires several steps. For example, the company should first fill out an online form before being contacted by the staff member to sign the formal application with a confidentiality agreement (Hak, 2007), as shown in **Fig. 1.** In addition, the green seal signs a confidentiality agreement, and once the application is completed, the staff member contacts to clearly explain the evaluation process to the company and provide a checklist of the required information (Hak, 2007), as shown in **Fig.1**.

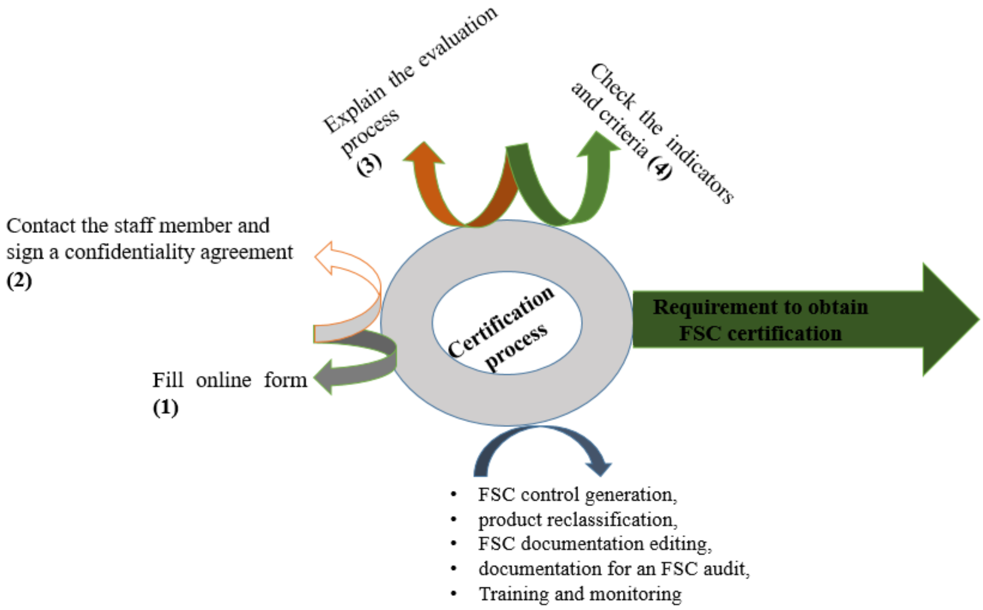

Fig. 1: Steps to obtain FSC certification

Authorities can develop their national FSC standards as long as they meet FSC principles and criteria; characteristics of the standard-developing group (e.g., research and expertise; socio-economic context; stakeholder dynamics; forest history; natural conditions; and attitudes toward certification); FSC international (checking the indicators and criteria); and finally, national FSC standards (Lehtonen et al., 2021), as shown in **Fig.1**. Similarly, another study outlined the following steps—FSC control generation, product reclassification, FSC documentation editing, documentation for an FSC audit, and finally training and monitoring are required to obtain FSC certification for an enterprise or forest (Solana et al., 2013), as shown **Fig. 1**. Certification can cover a range of aspects, including the geographical designation of origin, fair trade (FT) certification, certification of goods produced using ecologically sustainable techniques (organic, rain forest), and the production of better quality agricultural goods (Cogueto, 2014; Frederico, 2015). Studies have highlighted that FSC does not emit certificates but accredits independent organizations to qualify via audits (Silva, 2013; Solana et al., 2013).

3.2.3. The requirement for the Cerflor

Cerflor is a type of certification used by Brazilian companies. Similar to the FSC, there are some criteria adopted by Cerflor. Cerflor uses principles and criteria based on the FSC while adapting them to the national reality. Cerflor indicator verification processes require a set of principles, including compliance with legislation; zeal for biological diversity; respect for water, soil and air; Environmental, economic and social development of regions in which falls within the forestry activity; rationality in the use of forest resources in the short, medium and long term, in search of its sustainability (Faria, 2009). Notably, Cerflor is a national scheme accepted by the Program for Endorsement of Forest Certification. In such cases, the adoption of FSC or Cerflor may depend on the company or the individual interested.

Forest certification receives much attention due to its ability to promote export markets and offers a set of advantages, including the valorization of forest products, enhancement of competitiveness between companies, and improvement of management and forest quality. Moreover, forest certification assures customers that label-carrying products are from certified forests (Espinoza et al., 2012). Both FSC certification and Cerflor are used in Brazil for the production and utilization of wood as a raw material. A previous study claimed that forest certification enables the planning and monitoring of the environment (Islam and Siwar, 2009).

4. Case of cacao products

The utilization of certification for payment for ecosystem services is a promising strategy to preserve biodiversity-friendly or environmentally friendly cacao systems (Waldron et al., 2015). There are different types of cacao plantation systems, including shade and unshade plantations. It is noted that the main certifications for cacao are FT, UTZ, IBD, and RA (Marrocos et al., 2018; Saravia-Matus et al., 2020; Opoku, 2024), which represented 22% of worldwide cacao certification in 2012 (Marrocos et al., 2018). The type of certification most commonly used in Brazil depends on the Brazilian state. For example, organic certified cacao was the only type of cacao certification that existed in the state of Pará (Cesario, 2012), while

RA certification is largely used in southern Bahia, Brazil (Schroth et al., 2011). Certification aims to add value to cacao products. For example, a previous study conducted in Bahia reported that certified organic cocoa marketed by the Povos da Mata Network sold for R$270.00 per kilo (Boto Xavier et al., 2021).

4.1. IBD certification

Instituto Biodinâmico (IBD), the oldest (Fonseca, 2002) and most important Brazilian organic certifier (Hora et al., 2021), is a Brazilian non-profit enterprise that inspects and certifies agricultural activities, especially in the field of organic cultivation (Gomes and Casagrande Junior, 2018). Cocoa-growing in the south of Bahia covers an area of approximately 92,000 km² (Setenta and Lobão, 2012). The main certifications used for the culture of cacao are: certification of sustainability, IBD, and IG (Hora et al., 2021). The first IBD certification occurred in Ilheus, Bahia, in an area of 2,000 hectares of cacao (da Silva and Petterson Neto, 1997). A recent study reported that IBD was used to certify 37 properties, and the area of conserved forest fragments corresponded to 764.95 km² (Marrocos et al., 2018). Notably, this type of certification is globally accepted (IBD, 2014). Cacao was the first crop certified with the IBD organic seal (Nascimento, 2011) and contributed the most to the conservation of the Atlantic Forest (Marrocos et al., 2018). Similarly, universal trade zone (UTZ) and the Rainforest Alliance (RA) were also used for the certification of cacao production in Brazil (Silva, 2018). Organic certification and rainforest alliance actions have been developed to boost the production of fine and organic cacao in the south of Bahia (Baiard, 2015; Viana, 2015).

Importantly, IBD is made up of a team of inspectors who visit the properties and verify the agricultural production process and compliance with established norms (Gomes and Casagrande Junior, 2018). The certification process of IBD can be done via checklists based on international standards while taking into account the different realities of each client and through periodically reviewed reports (Gomes and Casagrande Junior, 2018). Various studies have outlined that the Geographical Indication (GI), implemented in 2018, is the most recent certification used for cacao in southern Bahia (Christ, 2020; Sant'ana et al., 2020). A consumer market for products with differentiated quality and standards that are distinct from those of products already existing in the markets should be developed for the delivery of certification (da Silva and Petterson Neto, 1997).

4.2. Project of law on cacao-cabruca in Brazil

The cabruca is known as a type of cultivation system within the forestry complex that favors greater maintenance of biodiversity (Vieira et al., 2021). It is noted that the Law No. 14.877 of 2024, in its Article 1, addressed the creation of cacao cabruca with the goal of certifying sustainability, social, and environmental interests in Brazilian cacao farming.[1] This law, in its Article 2, stipulates that the green seal of cacao cabruca may be awarded to the cacao grower who meets the following criteria: I. complies with all national, state, and municipal environmental and labor laws; II. cultivates cocoa in the cabruca agroforestry modality in the Atlantic Forest to conserve ecological diversity and its associated values, water resources, soils, ecosystems, and fragile and unique landscapes, maintaining the ecological functions of the forest as much as possible, among others. In other words, this project of the law encourages and valorizes environmental protection through environmental certification.

4.3. Impact of cacao certification in the economy

Forest certification enables consumers to purchase products coming from managed forests, thus improving market access for their products (Deniz, 2023; Humphries, 2005; Rantala, 2020). Certified products are generally welcomed by consumers but are more expensive than others already existed on the market. For example, a recent study has pointed out that the price of

[1] https://www2.camara.leg.br/legin/fed/lei/2024/lei-14877-4-junho-2024-795712-publicacaooriginal-171958-pl.html

certified cacao increased by 4% to 20% in the international market (Brenes et al., 2023). Similarly, a study reported that there was a situation where a buyer paid a 10% premium on a bag of UTZ certified cacao (Neto et al., 2018). Likewise, another study revealed that buyers were willing to pay an average premium of 10% for coffee certified by FT (Smith, 2016). In the case of chocolate, producers that achieve the certification process have the legitimate right to charge more for their products, not only because the qualities are better but also because of the environmental and social services they provide (Neves, 2019). Consumers have a positive attitude toward purchasing products from certified forests, although the cost of certification affects the sale price (Deniz, 2023).

Certification is associated with local economies and environmental impacts (Acharya et al., 2015) and is a way to ensure the economic and environmental sustainability of the production process while enabling premium prices for producers (Nguyen et al., 2023). However, due to the increased cost of certified products, certification can become a trade barrier for developing countries (Chen et al., 2020), despite its advantages, such as ensuring standards, creating a new basis of value and competition in the markets (Stringer, 2006), and ascertaining to interested outsiders that the products from enterprises that achieve the requirements established by FSC are acceptable and appropriate (Meidinger, 2001). Obviously, FSC is a commercial marketing strategy that contributes to the preservation of the environment (Solana et al., 2013). It was projected that the premium for Brazilian UTZ-certified cacao would be around 10% of the market price (Nieburg, 2013). Similarly, according to the Quality Manager of the Cocoa Innovation Center (CIC), cacao certified by the GI of Southern Bahia has an average added value of 50% above the conventional market price.[2]

[2] Cacau certificado pela IG Sul da Bahia já movimentou R$ 2 milhões – CACAU & CHOCOLATE (cacauechocolate.com.br)

5. Community-based forest enterprises (CFEs) in Brazil: certification

The acceptance of forest certification depends on enterprises, as their perceptions vary along with a set of factors. Findings from research on CFEs in Brazil's western Amazon showed that positive aspects were economic and social, whereas negative factors included the certification process and the associated economic expenditures (Humphries and Kainer, 2006). Certified community forests enormously contribute to positively influencing human wellbeing while enhancing equitable governance and forest restoration (Loveridge et al., 2023). However, bureaucratic regulations (e.g., long processes for obtaining certification), difficulties in complying with state law, certification costs (Fagundes et al., 2021; Nambiar, 2019), and low quality and volume produced, high cost of certification, and demanding standards (Hajjar, 2013) are the main disadvantages associated with the certification process (Silva, 2013). The direct costs encompass field audits, annual monitoring costs, and an annual certification fee, whereas the indirect costs include costs of forest management, costs in the chain of custody, and payment of taxes (Solana et al., 2013). A previous study documented that a producer spent at least between R$ 1,500 and R$ 2,000 to receive a technical visit from an auditor of the certification company and have their establishment inspected (Penteado, 2010). Moreover, fiscal incentives and the lack of scientific knowledge reaching producers can be challenges to the widespread adoption and acceptance of cacao certification across the country.

Besides, a reduction of bureaucratic delays and expenses can improve decentralization and special rates for CFEs (Humphries et al., 2012). A recent study indicates that Brazilian companies do not view certification as beneficial for achieving better prices for certified products (Araujo et al., 2009). Economic (Fagundes et al., 2021) and political factors also present challenges for forest certification in the country (Stringer, 2006). However, enterprises evince that FSC has some benefits (e.g., access to international markets and positive impacts for the enterprise) for the private sector (Mayr et al., 2020). Certification helps generate inputs and enhance the image of enterprises (De Haes et al., 2008), provided that the international criteria of importing countries are scrupulously met. For example, certified cacao produced in Brazil is primarily destined for the European market (FVPP, 2011), while non-certified cacao is sold only in the local market. Other advantages of certification include the creation of new markets and the ethical improvement of companies (Vogt et al., 2006).

6. Future challenges and way forward

The acceptance of forest certification is crucial for promoting environmental sustainability and ensuring the quality of green products. The benefits offered by certification are significant for underdeveloped countries with economies based on commodities. However, Brazilian enterprises still face challenges in adopting certification due to national legislation and the high costs associated with certified products in their markets. Moreover, Vogt et al. (2006) underscored a set of disadvantages for certification, such as internal economics that are transparent, loss of management options, and the requirement for annual reviews. These challenges can be overcome through the adoption of a new approach aimed at financing certification in developing countries. For example, future survey research can be carried out in the state of Bahia to investigate the willingness of private owners' enterprises to accept the FSC if local authorities accept to finance it at 25% and reduce the bureaucratic process. The limitations of this state-of-the-art review include a deep overview of how laws assess certification and the process to obtain each type of certification mentioned throughout this work. Going beyond this review, we would like to suggest future research to address the challenges previously mentioned.

7. Conclusions

There is an urgent need to promote environmental sustainability and choose certified products while preserving biodiversity and conserving natural resources. These challenges have attracted researchers' interest in the use of forest certification. This state-of-the-art review examines Brazil's experiences with adopting forest certification and the types of certifications used for cacao products. This review shows that UTZ, IBD, FSC, RA, and GI are types of certifications used for cacao products. It is noted that the GI is the most recent certification used for cacao in southern Bahia. Bureaucracy, the high cost of certified products, and the achievement of FSC requirements are among the main barriers to certification in Brazil. As a complementary strategy to the existing certification models, there is a pressing need to establish certification focused on monitoring and ensuring deforestation-free areas. This certification would specifically target regions like the southern part of Bahia, which encompasses valuable areas of the Atlantic Forest and degraded lands where agroforestry systems have been implemented. By incorporating satellite monitoring technology, this certification can effectively track and verify compliance with deforestation-free practices, ensuring the protection of crucial ecosystems and promoting sustainable land use.

The state of Bahia stands to greatly benefit from the adoption of this certification approach. With its rich biodiversity and significant forest resources, Bahia has the potential to become a leading example of environmentally responsible production. By embracing a certification focused on deforestation-free areas, the state can attract investments and partnerships from environmentally conscious businesses and consumers worldwide. This certification would not only safeguard the remaining forested areas but also promote responsible land management practices, contributing to the preservation of natural resources, the mitigation of climate change, and the overall well-being of local communities. While forest certification has shown its potential to promote environmental sustainability and enhance the economy, the inclusion of a certification focused on monitoring deforestation-free areas would further strengthen these efforts. By integrating this approach and incorporating the unique environmental context of the southern region of Bahia, Brazil can showcase its commitment to combating deforestation, supporting sustainable land use practices, and serving as a global leader in environmentally responsible production. Conclusively, this review shows that the adoption of forest certification can enormously contribute to the national economy while making the products more competitive on the market. Forest certification is therefore a promising mechanism for environmentally responsible production, a guarantee for workers' rights, and a means of fostering economic growth for nations, especially in developing countries with economies based on commodities.

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Data availability statement

All relevant data are included in the paper or its Supplementary Information.

Conflicts of interest statement

The authors declare there is no conflict.

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Chapter II

Interactions between forest cover and watershed hydrology: A state-of-the-art review

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Interactions between forest cover and watershed hydrology: A state-of-the-art review

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Abstract

The role of trees in watershed hydrology is governed by many environmental factors along with their inherent characteristics and not surprisingly has generated into diverse debates in the literature. Herein, this state-of-the-art review provides an opportunity to propose a conceptual model for understanding the role of trees in watershed hydrology and examine the conditions under which they can be an element that increases or decreases water supply in a watershed hydrology. To achieve this goal, this review addressed the interaction of forest cover with climatic conditions, soil types, infiltration, siltation and erosion, water availability, and the diversity of their ecological features. The novelty of the proposed conceptual model highlights that tree species and densities, climate, precipitation, type of aquifer, and topography are important factors affecting the relationships between trees and water availability. This suggests that forests can be used as a nature-based solution for conserving and managing natural resources, including water, soil and air. To sum up, forests can reduce people's imprint, thanks to their role in improving water and air quality, conserving soil, and other ecosystem services. The outcomes of this study should be valuable for decision-makers when investing in reforestation in a watershed hydrology.

Keywords: Erosion, forest age, forest hydrology, runoff, soil water infiltration, water cycle

1. **Introduction**

Forests cover about one-third of the Earth terrestrial area (FAO, 2016) and play a crucial role in environmental sustainability and human life. They significantly contribute to climate change mitigation by absorbing and storing 30% of carbon emissions (Brack, 2019), reducing greenhouse gas emissions (Arora et al., 2012), providing food to people (Azigwe et al., 2016), and offering several ecosystem services, including water provision, soil conservation, and climate regulation. Forests could positively or negatively affect water storage (i.e., soil water availability) by regulating basic fluxes such as infiltration, and surface runoff, and evapotranspiration (ET). Natural forests have been exploited (Alvarez-Garreton et al., 2019) and destroyed for agricultural activities, one of the main contributors to soil erosion. From 2000 to 2012, approximately 3.2% of forest cover worldwide was converted into agricultural lands (Riitters et al., 2016). This conversion could affect soil water availability (SWA). However, the relationships between trees and basic features of the hydrologic cycle (storage and fluxes of water) are complex and contradictory. For example, a scientific paper has argued that deforestation could increase downstream water availability, whereas others have concluded that afforestation increases downstream water availability and intensifies the water cycle (e.g., Ellison et al., 2012). Other researchers have documented that afforestation decreases water yields, especially trees such as *eucalyptus* and *pinus* (Robinson et al., 1991; Farley et al., 2005; Jackson et al., 2005; Buytaert et al., 2007; Galleguillos et al., 2021).

Trees are fundamentally important in regulating streamflow (Wang et al., 2021). However, some species can reduce groundwater levels because of climate changes and physiological characteristics that may affect ET (Anurag et al., 2021). The transpiration of some trees (e.g., *Phyllostachys edulis*) can be affected by multiple factors such as tree age, size, phenological stages, and soil water content (Delzon and Lustau, 2005; Scott and Prinsloo, 2008; Gu et al., 2019). Thus, tree transpiration can couple with environmental variables to alter the water cycle and water balance on local and regional scales. To meet transpiration needs (Bonan, 2002), trees with deep root systems can extract large volumes of water from depths of 10 meters or more (Le Maitre et al., 1999). On this topic, researchers have argued that there is an interdependence between vegetation and deep groundwater (Yang et al., 2019; Kopeć et al., 2013; Zhou et al., 2013). There is a great need to clarify controversies about the relationship between watershed hydrology, and ultimately the global water cycle. As such, we expect that trees reduce soil erosion, compaction and surface runoff during precipitation. Besides, the change in the hydrological cycle, particularly in extreme precipitation, can intensify negatively with global warming (Zhang, 2019; Tabari, 2020). Likewise, global warming can directly influence precipitation, leading to a greater evaporation rate and thus surface drying (Trenberth, 2011). Similarly, changes in climatological precipitation and evapotranspiration lead to changes in runoff (Arora and Boer, 2001).

A conceptual model should consider how tree communities in forested areas can affect the amount of water in the soil and at a watershed outlet, and their role in controlling erosion and reducing runoff. This model also should consider the impacts of fast-growing forest plantations on the water balance and streamflow compared to those of native forests. Various studies have documented large-scale relationships between hydrological effects and deforestation, forestation (D'Almeida et al., 2007; Vergopolan and Fisher, 2016; Hou et al., 2023), climate change (Zhang et al., 2022). Others have demonstrated relationships between water cycle components (e.g., precipitation and evapotranspiration) and water vapor residence time (Gimeno et al., 2021), and forest maturity (Belmar et al., 2018). For example, to meet their evapotranspiration need, trees use various strategies for searching water in a forested watershed, preferring soil water rather than groundwater (Penna et al., 2013), depending on the period; for example, they could uptake groundwater during dry periods. Regardless of the source, trees affect the partitioning of water between catchment water yield and ET (Knighton et al., 2020;

Lawrence et al., 2011). In the end, water extraction and availability are governed by interactions between macropore flow, matrix storage, and shape of root systems (Brooks et al., 2010) and ultimately these interactions define the ecohydrological functioning of forests (Asbjornsen et al., 2011).

Notably, there is a direct relationship between transpiration and diel fluctuations in streamflow (Asbjornsen et al., 2011), which vary seasonally and spatially (Vose et al., 2016). Nonetheless, there is a need for improving the understanding of the interactions between forest cover and watershed hydrology. Hence, the objective behind this state-of-the-art review is to document the influence of trees on water availability and propose a conceptual model of their role in watershed hydrology and the conditions in under which they can increase or decrease water supply. Therefore, after the introduction section, this paper proceeds as follows: (i) Methodology; (ii) origin of precipitation; (iii) conceptual model of the role of trees in watershed hydrology; (iv) relationships between forests, runoff, and soil erosion control; (v) effects of forests on watershed hydrology at various spatial scales; (vi) relationships between tree species and SWA; and (vii) conclusions and future research.

2. **Methodology**

This state-of-the art review proposed a conceptual model for understanding the role of trees in watershed hydrology and examined the conditions under which they can influence water supply in a watershed hydrology. Scientific documents were searched from literature databases (e.g., Scopus, google scholar and Web of Science) using keywords, including “forest cover”, “planting trees”, “trees”, “water protection”, “water availability”, “infiltration”, “reduce runoff”, “watershed”, “rainforest”, “climate”, “soil compositions”, “evapotranspiration”, “vegetation”, hydrologic cycle”, “topography”, “forest age”, “base flow”, “watershed”. It is noted that Boolean operators “AND” and “OR” were used to associate the aforementioned keywords and thus refine the search results. This review focused on English documents (e.g., papers, reports and books) published from 1933 to 2023. The list of references was also used to search for additional published documents. Information was extracted from the abstracts and conclusions of documents selected after screening their titles and abstracts. It is noted that information from these documents was analyzed by searching for relationships between the keywords used and at least one of the watershed hydrology components (e.g., runoff, infiltration) targeted in this review related to water availability. Finally, data was managed using EndNote to ensure accurate referencing.

3. **Origin of precipitation**

Numerical studies have illustrated that precipitation is recycled over a long distance through trees evapotranspiration that drives winds and moist air transport (Gimeno et al., 2010; Pearce, 2020; Gimeno et al., 2021). Of note, 90% of water evaporated every year precipitates back onto oceans, and the remaining 10% feeds the land branch of the water cycle (Gimeno et al., 2010). The major sources of moisture have their origins in large regions characterized by vertically integrated moisture flux divergence (Trenberth and Guillemot, 1998). The North and South Atlantic sources are globally the first and second largest sources of moisture for precipitation over the continents, respectively (Gimeno et al., 2010). The numerical study of Gimeno et al. (2010) detailed and highlighted how moisture is formed under the effect latent heat fluxes over the ocean and subsequently transport in the atmosphere before reaching the soil surface in the form of precipitation. The effect of orography is a factor that is susceptible to limiting the moisture from the ocean and thus reducing the oceanic contribution in terms of precipitation. Notably, there are other sources (e.g., land evaporation) of precipitation. For example, van der Ent et al. (2010) pointed out that land evaporation provides 40% of terrestrial precipitation, of which 57% is back on land in the form of precipitation. It worth noting that terrestrial precipitation, evaporation recycling, and moisture exportation mainly occur over the continents (Sorí et al., 2023). A decline in precipitation may be linked to deforestation (Sheil, 2018). Of note, moisture recycling is strongly associated with forests expansion. Thus, the larger the expansion, the larger the moisture recycling. Water is precipitated on large regions either by advection from the surrounding areas external to the region and evaporation, or transpiration from the land surface of the region (Brubaker et al., 1993). Notably, precipitation recycling in forests significantly influences the isotopic composition of precipitation in northwestern Amazonia (Ampuero et al., 2020).

4. **Conceptual model of the role of trees in watershed hydrology**

4.1. Soil characteristics and water infiltration

Some trees reduce water on rocky substrates (saprolite, fractured bedrock), particularly when the source is deep below ground, using around 49% for transpiration during dry seasons and 28% during wet seasons (Barbeta and Peñuelas, 2017). Trees that grow in less favorable soil/subsoil conditions consume deepwater reserves due to root adaptation to enhance drought tolerance (Carrière et al., 2020a). The hydrologic response to drought can be either mitigated or exacerbated by forest vegetation, depending mainly on the amount of water used by vegetation and the response of forest population (Vose et al., 2016). In a restoration project, clayey soils recovered infiltration faster than sandy soils (Lozano-Baez et al., 2019; Regelink et al., 2015). This could occur because the aggregating forces in sandy soils are weaker than those in clayey soils. Thus, high soil aggregation is one of the characteristics that can explain and justify high infiltration, which can greatly depend on the historicity of a targeted area.

Reforestation in the tropics and subtropics may improve water infiltration, depending on land use, soil texture, and local climate (Lozano-Baez et al., 2019). It is noted that reforestation should regulate water fluxes (Hock et al., 2009) through infiltration and ET (Harden and Matthews, 2000) depending on soil properties, which are influenced by a set of factors such as slope/topography (Tsui et al., 2004; Begum et al., 2010; Agbeshie and Abugre, 2021), climate, parent material, time, and living organisms (Rhoades, 1996). Reducing soil organic matter content can adversely affect root penetration, thus reducing water infiltration and compromising the role of trees in mitigating erosion. Also, infiltration time would diminish independently of the rainfall's intensity and duration in a mechanically terraced area. The compaction reduces soil infiltration and root penetration. Substrates controlled by regolith and rocks impose drought conditions on oak forest stands (Rodríguez-Robles and Arredondo, 2022). Such soil reduces water infiltration, which drives surface runoff, soil erosion, chemical transfer routes, water quality, and irrigation uniformity (Rashidi et al., 2014). Depending on the size of rock fragments and their aggregation to the soils, they could favor infiltration or enhance soil loss (Descroix et al., 2001).

4.2. Streamflow versus base flow partitioning

Base flow is correlated with forest extension and is crucial to maintain the water yield of a watershed. There is a correlation between changes in forest ET and riparian water table height and riparian area; and in return this can increase ET loss and modulate streamflow (Cadol et al., 2012). Meanwhile, forest cover type and annual temperature affect watershed base flow (Ding et al., 2022). A decrease in total basal area of *pinus* trees can lead to an increase in groundwater recharge, cumulative streamflow, and direct runoff (Bent, 2001; Tarigan et al., 2018). These findings indicate that forest is one of the key factors governing base flow in a catchment. Another study corroborates these findings and outlines that high ET reduces stream flows (Cheng et al., 2002). Besides, changes in forest cover during regeneration modify water flux partitioning (Neill et al., 2021). Other variables, including soil composition and climate conditions, may be among possible factors affecting groundwater recharge and base flow. For example, Zomlot et al. (2015) explained that precipitation, soil texture, and forest cover modulate groundwater recharge, while vegetation cover and groundwater depth affect base flow.

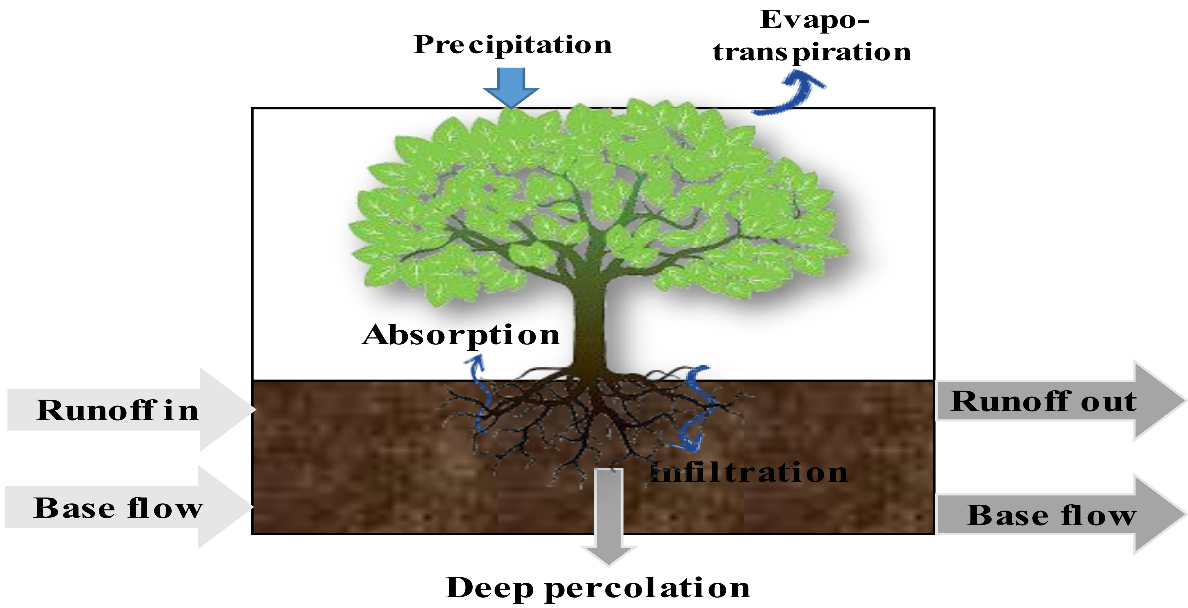
Notably, there is a correlation between rainfall, base flow, and forest area. Khomsiati et al. (2021) argued that the greater the forest area the more stable are flow conditions.

4.3. Evapotranspiration

ET is a key hydrological process, and the only mechanism that supplies water vapor into the atmosphere (Al-Tameemi and Chukin, 2016). It is responsible for the coupling of the land surface energy balance with the terrestrial and atmospheric water balances. The relationships between trees, water availability and water fluxes are linked to hydrologic processes such as groundwater recharge (balance between ET and infiltration) and surface runoff, as shown in **Figs**. 1 & 2. Research conducted in Ghana and Southern Burkina Faso reported that ET consumed 72% of the annual precipitation (Guug et al., 2020). In the Amazon Basin in Brazil and Peru, the forest canopy can induce significant moisture fluxes between land and atmosphere leading to a precipitation-ET loop (Garcia-Chevesich et al., 2017).

4.4. Soil water availability

Soil characteristics such as fractured rock, fracture depth, soil texture, and parental rock interact with vegetation to reduce SWA, which is used here to refer to soil water storage, soil water recharge, rivers, basins, and watershed recharge **(Fig. 1**). Since SWA varies among different substrates and different types of soil (e.g., sand, silt, clay, etc.) and land use land cover, it also influences water quality (Lei et al., 2021). Loamy sand sites could have a SWA greater than sandy clay loam or sandy clay (Dodd and Lauenroth, 1997). Interactions between trees and soil water can be influenced by natural conditions (e.g., topography and slope) and parent material (i.e., geologic material). In such cases, trees can remove more water from the soil if the parent material mostly comprises organic matter.



$$\frac{\Delta\,Treecanopy}{\Delta_t} = P + R_{in} + Abs - ETP - R_{out} - \mathrm{Inf} \quad (1)$$

$$\frac{\Delta SWA}{\Delta_t} = Inf + Gr_{in} - Gr_{out} - D_p - Abs \quad (2)$$

Fig. 1. Conceptual models of water mass balance of a tree canopy delineated by the upper control volume and the soil water of the underlying control volume of the porous media.

In Eqs. (1 & 2), P is precipitation, R_{in} is runoff in, *Abs* is absorption, R_{out} is runoff out, *Inf* is water infiltration, Gr_{in} is the groundwater in, Gr_{out} is the groundwater out, D_p is the deep groundwater, Δ*tree canopy* is the water mass balance of a tree canopy over time interval Δ*t*, Δ*SWA* is water mass balance in the soil at time interval Δ*t*.

SWA is the sum of water in the unsaturated zone (vadose zone) and the saturated zone (water table). **Fig. 1** represents conceptual models of water mass balances of a tree canopy and underlying soil matrix. Castillo and Oyarzun (2021) highlight that forest cover is one of the crucial parameters in forest management that alters the accumulation of water in the vadose and saturated zones of the soil. Therefore, modifying natural forests through deforestation may temporarily increase watershed hydrology, directly impacting the annual hydrograph and thus low and peak flows, streamflow regulation, and flood occurrences. These responses occur quite rapidly. The tree canopy water mass balance is the difference between water inputs and outputs **(Eq. (1)).** When water reaches a tree, a part is lost through ET**,** and another part infiltrates the soil and thus increases SWA. Soil water depletion is the difference between the sum of water inputs (infiltration (*Inf*) and groundwater in (Gr_{in}) and water outputs (groundwater out (Gr_{out}), deep groundwater (D_p), and absorption (*Abs*) (Eq. (2)). Water absorption by roots from the soil depend on tree types, climate, and soil physical properties. Of note, SWA depends on vegetation cover, types, and understory composition. A recent study underscored that groundwater is tightly related to soil water (Costa et al., 2023). As such, exotic or native trees with a higher ET rate deplete SWA and can compete for water with other trees. Thus, we argue that soil moisture is somehow associated with vegetation types. The reduction of SWA and groundwater can also be associated with albedo and latent heat flux as they are among the mechanisms responsible for these changes (Runyan and D'Odorico, 2016).

4.4.1. Relationship between soil characteristics and SWA

There is a link between soil properties and trees (Soong et al., 2020) that can cause a change in SWA (Maxwell et al., 2018). For example, trees can uptake more water from loamy soil, soil with higher organic matter content, or sandy soil than rocky soil (Dodd and Lauenroth, 1997). This is possible because tree roots have more difficulty reaching groundwater beyond the vadose zone (Carrière et al., 2020b). In such a case, soil porosity should be considered because this property can give a false impression that forests retain water. SWA varies from one site to another, depending on soil textures. For example, capillary and hydraulic barriers enable layered soils to hold more water (presence of perched water tables) than nonlayered ones (Li et al., 2014). Similarly, a previous study documented that SWA varied in the following order: loamy sand > sandy loamy and sandy clay sites (Dodd and Lauenroth, 1997).

The reduction of soil particle sizes and tree development lead to organic matter accumulation in the topsoil and thus increase the soil water storage (Hartmann et al., 2021). Notably, biological soil crusts play a paramount role in increasing soil porosity and micro-topography, thus, enhancing infiltration while increasing runoff by the secretion of hydrophobic compounds as well as clogging of soil pores upon wetting (Rodríguez-Caballero et al., 2013). Tree-soil-water availability (TSWA) constitutes a complex system in which trees can increase water yield depending on soil composition. In a landscape with a high elevation, moisture could be more favorable to trees because they do not need to uptake water from the deeper soil. This interaction occurs due to the slope angle's control on SWA. In the TSWA system, water yield can be increased or decreased depending on the characteristics of trees, soil saturation, and infiltration capacity.

4.5. Combined effects of forests and local climate on SWA

There is a synergic effect between forests and local climate on SWA. In this regard, researchers have reported that climate has considerable impacts on water balance components (such as runoff, precipitation, and ET) (Blanco-Gómez et al., 2019; Jiali et al., 2019; Ich et al., 2022) and forest cover (Li and Quiring, 2021; Pecchi et al., 2020). Several scientists have highlighted that climate and trees govern water availability in vegetated areas (Nijland et al., 2010; Dahal et al., 2020), playing an essential role in regulating water security and supply (Daneshi et al., 2021; Zhang et al., 2018), and thus affect drainage and runoff characteristics (Sajikumar and Remya, 2014). Climate change variability is one of the main factors affecting precipitation, hydrological processes, and as the final runoff response. There is an interrelationship between the water cycle and climate change. Notably, evaporation, precipitation, and precipitable water are key components of the water cycle that influence global climate change (Al-Tameemi and Chukin, 2016). Climate change negatively affects the water cycle, freshwater availability, and water security (Kundzewicz, 2008; Yang et al., 2014; Grover, 2015).

In the future, interannual climate variability could be stronger in the Pacific and Indian Oceans and weaker in the Atlantic, while interdecadal climate variability is expected to enhance and reduced warmth in polar and equatorial regions, respectively (Ma et al., 2020). These findings highlight that polar and equatorial region are susceptible to receiving longer precipitation periods than the Pacific and Indian Oceans. These findings may also indicate that different climate change scenarios can lead to different patterns of change in the terrestrial water cycle (Tao et al., 2003). In watershed areas covered by exotic tree plantations in south-central Chile, increasing and decreasing trends in evaporation and percolation rates were registered because of climate change, respectively (Martínez-Retureta et al., 2020). However, a variability of responses may exist, depending on environmental and tree characteristics. For example, large ET is more predominant at high altitudes in the north (Yang et al., 2023). It is noted that any change in forest structure can affect climate and vice-versa (Sheil, 2018)

4.6. Relationship between topographic factors and SWA

Altitude and landscape slopes can determine plants' behavior, modifying SWA and increasing relative humidity through the ET of unused water in (turgescence) and on (intercepted) leaves. A previous study reported that topographic position and slopes interact together to form a thermal gradient and water stress for trees across the landscapes (Gallardo-Cruz et al., 2009). At high altitudes, vegetation uptakes water from the deeper unsaturated soils, developing significant variability in water consumption strategies (Rossatto et al., 2012). Other researchers have underscored that landscape topography influences tree growth (Liu et al., 2020; Adams et al., 2014) and affects mountain forests through its effects on radiation and moisture (Måren et al., 2015). Similarly, another study indicates that soil variation and water loss are important factors of the topographic gradient (Zhang et al., 2014). Thus, in certain cases, the slope gradient can reduce runoff, reduce soil moisture, and enhance ET, which may be associated with biophysical changes (for instance, deeper roots). That probably reduces the soil water stored (Feng et al., 2016; Qiu et al., 2017), which, in return, influences infiltration and ET (Hu et al., 2017), subsurface runoff paths, and erosional processes (Rempe and Dietrich, 2014). The largest average soil moisture values occur on topography with a flat surface configuration (Yu et al., 2018). However, the drainage system could reduce the soil moisture in a determined area. An increase in water table depth may lead to a decrease in the role of the topography of the land surface and the spatial distribution of water when the water table is deep and close to the bedrock surface (Yao and Wang, 2021).

5. **Relationships between forests and runoff and soil erosion control**

Runoff is another important hydrological process and has various responses to forests at different scales (e.g., large, medium, and small scales). Certain vegetation types are more appropriate for reducing erosion than other trees. Leomo et al. (2018) mentioned that forest, pepper, bush, and intercropping are types of vegetation that can minimize erosion. Afforestation reduces runoff and flood peak discharge and controls soil erosion due increased forest cover, canopy structure and density to protect the soil from direct rainstorms. Notably, reforestation could have both positive (decrease wet season runoff) and negative (increase surface runoff) impacts on runoff (Xu et al., 2020). The role of forests in reducing soil loss could vary depending on topography. A study outlined that soil loss varied according to the types of slopes, as soil loss from convex slopes was 1.5 times greater than that from concave and uniform slopes (Ao et al., 2021). In addition, forest cover can reduce soil degradation (Yu et al., 2019), depending on climatic factors and rainfall regime. Along this line of reasoning, another research pointed out that runoff and soil loss were negatively correlated with slope value, organic matter content, tree cover percentage, and soil structural stability (Descroix et al., 2001). Recent research has indicated that fine roots of apple trees reduce SWA (Shen et al., 2022). This reduction may depend on the length and shape of the root system. From a holistic viewpoint, our review corroborates the literature on the relationship between trees, runoff, and soil erosion (Chen et al., 2018) by demonstrating that trees use their canopy and root systems to reduce erosion. However, when doing a careful analysis, we comprehend that this reduction depends on specific conditions (e.g., tree densities, and local climate) and environmental factors such as slope, length of slope, and soil structural characteristics.

5.1. Runoff responses to forest at multiple scales

Runoff response can be influenced by various factors, including forest type, soil properties, and watershed scales. The annual runoff response to land cover change may depends on forest type and the size of a watershed. There is a straight correlation between the runoff coefficient and the watershed scale, where runoff coefficients reduce as watershed areas increase (Zhang et al., 2019). Thus, large-scale watersheds have very few sensitivity factors for runoff, while the impacts of vegetation could be observed in small-and medium-scale watersheds (Zhang et al., 2019). Runoff coefficients depend on both shape and size of a watershed (Wang et al., 2012). Moreover, the type of land cover is a crucial factor affecting the hydrological response of a watershed, and the runoff coefficients to peak flow relationships vary from year to year (Sriwongsitanon and Taesombat, 2011).

5.2. Factors affecting the surface runoff

Surface runoff, or overland flow is generated within a watershed and can be explained by one of two scenarios: (i) the precipitation rate exceeds the infiltration capacity of the soil column, or (ii) the water table reaches the soil surface (Bonan, 2002; Eagleson, 2002). The first process, called "Hortonian", occurs under high rainfall intensities (Horton, 1933), while the second mechanism, called "Dunne", happens under low precipitation intensity with shallow water tables (Dunne and Black, 1970). Surface runoff can be influenced by a set of factors, such as vertical vegetation structure, vegetation distribution pattern, and plant diversity (Liu et al., 2018). Admittedly, vegetation can reduce runoff (Lopes et al., 2021), intercept rainfall (Gardon et al., 2020), and drain stormwater (Nainar et al., 2021).

Plantation type and age can impact runoff and hydrologic processes. For example, Sun et al. (2018) suggested mature plantations rather than young plantations can have a direct impact on soil erosion and runoff. Another study showed that afforestation with *pinus* led to a higher runoff reduction than afforestation with *eucalyptus* in high-rainfall areas (Brown and Nambiar, 2001). Converting natural forests to plantation forests reduces the total amount of runoff (Rahmat et al., 2018). Along the same line of reasoning, Rahmat et al. (2018) did not recommend afforestation in countries with little precipitation because mature forests reduce the amount of runoff. However, old trees may not contribute much to erosion control. Contrary to young trees, unused water in mature forests evaporates and then contributes to air moisture, which can lead to precipitation and, thus, replenish groundwater.

6. **Effect of forests on watershed hydrology at various spatial scales**

The effect of forests on watershed hydrology varies in time and space. For example, at a large spatial scale, forest restoration can enhance precipitation recycling due to atmospheric drawbacks (Van Dijk and Keenan, 2007; Ellison et al., 2012). Large-scale deforestation can have a detrimental effect on watershed hydrology. Wongchuig et al. (2023) documented that the average terrestrial water storage and runoff dynamics in the Amazon Forest are approximately ten times more significant in deforested areas than in forested areas. On the one hand, studies have pointed out that in some regions of the world, large-scale forest restoration can result in higher water yields (Betts et al., 2007; Mao et al., 2015) and thus intensify watershed hydrology (Huntington, 2006). On the other hand, Filoso et al. (2017) underscored that it does not necessarily increase water yields. These findings suggest that the interaction between forests and hydrological processes varies in time and space (Blöschl and Sivapalan, 1995). At smaller scale, little insights were found in the literature about the interaction between reforestation/afforestation and precipitation and thus, it is difficult to postulate that small-scale forest expansions can generate enough moisture recycling to increase rainfall.

7. **Relationships between tree species and SWA**

7.1. Fast-growing/commercial trees

Certain fast-growing trees, such as *Eucalyptus globulus* and *E. grandis urophylla*, *Larix principis-rupprechtii* and *Pinus radiata,* reduce water availability in the soil (Ferraz et al., 2019;

Galleguillos et al., 2021; Joshi and Palanisami., 2011; Little et al., 2009; Naghizadeh and Wessels, 2021; Ong et al., 2006; Rahmat et al., 2018; Shem et al., 2009; Sikka et al., 2003; Tian et al., 2021; Zhou et al., 2002; Cassiano et al., 2023) and soil erosion while increasing infiltration and ET (Bonnesoeur et al., 2019), as shown in **Fig. 2b**. This occurs during both their growth stage and their adult stage. Industrial *eucalyptus* overuses stored water when planted in sandy soils (Reichert et al., 2021), and their roots can reach water table depth of 12 meters after only two years (Christina et al., 2016). In such cases, commercial trees function as natural drains, lowering the water table and enhancing local evapotranspiration, a practice known as biodrainage.

A recent study showed that multiple decades of forest operation reduced deep soil moisture reservoirs, illustrating that when *Radiata pinus* trees are replaced by *eucalyptus* subsurface supply to streamflow substantially decreased under dry period conditions (Iroumé et al., 2021). Similarly, *Pinus halepensis* increases water use (Maestre and Cortina, 2004) and, thus, reduces the amount of moisture stored in the soil (Querejeta et al., 2001). Admittedly, these fast-growing tree (monocultures trees) plantations generally transpire more than slow growing forests due to their high interception loss (Cannell, 1999).

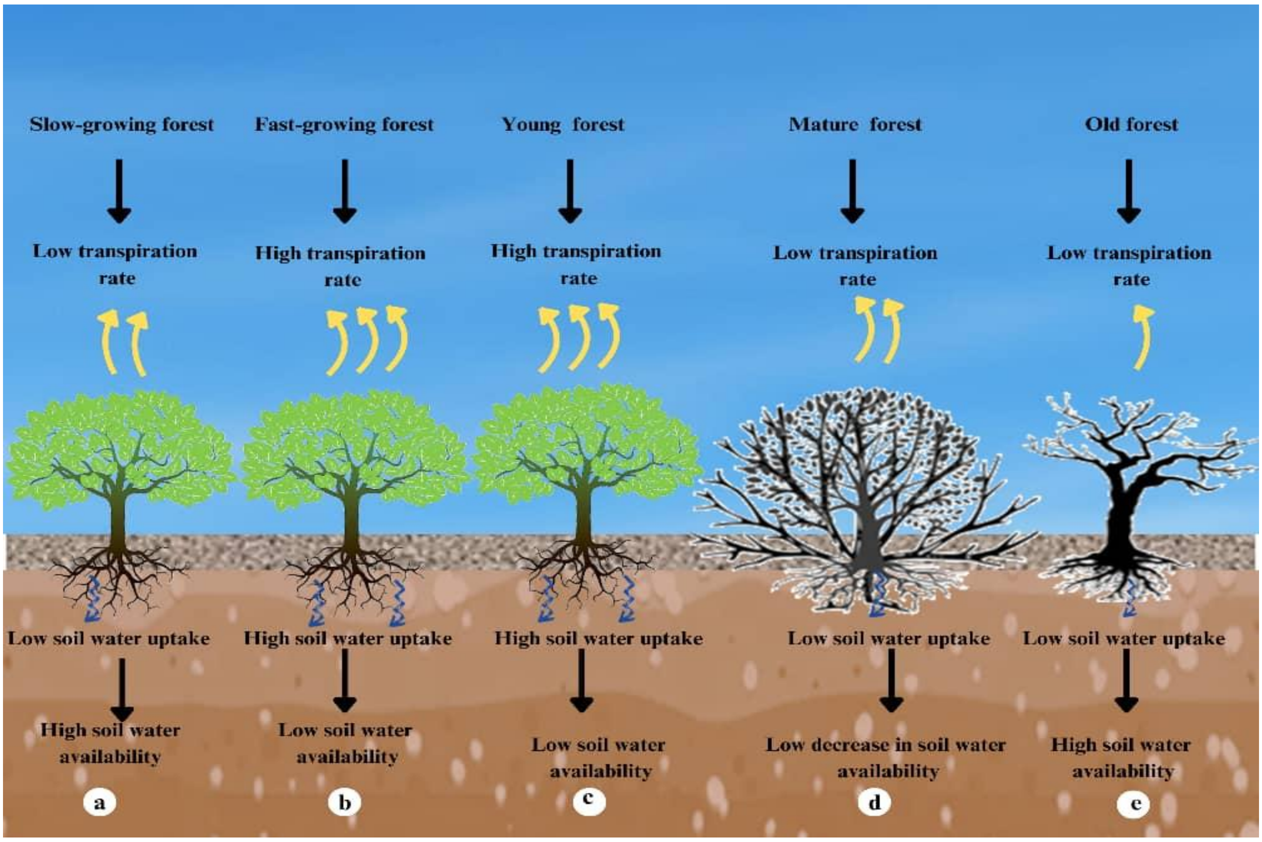


Fig.2. Relationship between trees and a part of vertical water fluxes and soil water availability, illustrating differences between fast and slow-growing forests. Fast-growing forests have a larger impact on soil water availability due to their higher transpiration rates, especially during early ages.

Fast-growing forests have growth and ET rates higher than native forests (**Fig. 2a**). During their growth, they reduce SWA through their root systems, which can reach the groundwater level in a short period. This finding indicates that forest types have a crucial role in water yield because of their different ET magnitudes (Cui et al., 2012). This finding may also show that native species are more adapted to water stress than non-native trees (Filoso et al., 2017). The negative impact of fast-growing forests on water yield is only for a short time because they are generally cut at their youngest age for commercial purposes. Fast-growing forests are unsuitable for afforestation in areas with medium precipitation and brackish groundwater (Sudmeyer and Simons, 2008), and their photosynthetic rates and stomatal conductance are higher than those of slow-growing forests (Liu et al., 2021). This indicates a relationship between the type and age of trees and SWA. Research findings in South Africa indicated that over a 5-year period of afforestation with *pinus* reduced the annual streamflow yield by 44 mm/a for each 10% of catchment planted when trees aged between 10 and 20 years (Scott and Prinsloo, 2008). Similarly, *eucalyptus* plantation reduced over a 3-year period the annual peak flow by 48 mm when 10% of a catchment was afforested (Scott and Prinsloo, 2008). Another study reported that *eucalyptus* and *pinus* reduced on average runoff by 75% and 40%, respectively (Farley et al., 2005).

7.2. Slow-growing forests

In contrast to fast-growing forests, slow-growing forests consume less water and, therefore, have fewer impacts on SWA (**Figs. 2a&b**). This finding indicates that some species are more tolerant to droughts than others. Trees can suppress runoff movement (Iwara et al., 2011) and, thus, affect positively and significantly the water yield (Singh and Mishra, 2012). For example, studies have concluded that commercial forests and trees, and tree densities can enhance infiltration, increase groundwater and are considered the prime regulators within the water cycle (Ellison et al., 2017; Ellison and Speranza, 2020; Iwara et al., 2011; Singh and Mishra, 2012; Ilstedt et al., 2016). As a result, slow-growing forests are suitable for afforestation projects since they have fewer effects on SWA regarding water consumption.

7.3. Effect of Stand density on SWA

Research results in West Africa reported that forest densities maximized groundwater recharge (Ilstedt et al., 2016), which could also be affected by vegetation community types and phenology (Wang et al., 2022). Changes in forest densities can alter the hydrologic processes at the watershed scale (Vose et al., 2016). Simularly, another study reported that SWA increased with an increase in stand density (Chen et al., 2020). However, this may depend on the tree species, stand age, and climate. For example, research finding showed that the plantation of high-density *Pinus sylvestris* significantly reduced the SWA (Nan et al., 2020). This finding corroborates the results of another study that suggests to reduce the density of *Quercus ilex* in semi-arid woodlands to prevent excessive water deficit (Moreno and Cubera, 2008). The reduction in SWA occurred due to tree transpiration (Lie et al., 2018). As such, a reduction in stand density may lead to an increase in SWA in native forest areas (Steckel et al., 2020). Admittedly, competition for resources among trees can also reduce SWA. For example, the results of a study conducted by Giuggiola et al. (2018) underscore that an increase in understory density led to a reduction in SWA.

7.4. Effect of forest age on SWA

Forests/trees play several roles increasing (through infiltration) and decreasing (via evaporation) SWA depending on several factors, including forest age. A relationship between SWA and forest age involves time and space. It was found in the literature that water infiltration increases with forest age (Deuchars et al., 2006). Notably, two temporal scenarios could be presented regarding the influence of tree age on SWA:

7.4.1. Young/juvenile forest Vs SWA

The first scenario is that during their growth, young trees accumulate a large quantity of biomass, grow faster, consume much water, have high ET rates, and reduce the amount of SWA or existing water in a watershed (**Fig**. **2c**). Young *pinus* have ET rates greater than old *pinus* and, thus, may reduce streamflow of a given watershed (Law et al., 2001). As trees pass through multiple phenological phases before reaching maturity, from the juvenile phase to the adult phase, the amount of water they use and associated ET rates may vary across stages of growth **(Fig. 2**). According to Van Dijk and Keenan (2007), ET rates decrease with forest age. Trees can uptake large amounts of soil water and evaporate more water under various hydrogeologic conditions. At earlier ages, they reduce the amount of SWA. Long-term fluctuations in pioneer forest areas and age structure decreased freshwater in riparian forests (Stromberg et al., 2010). Regrowth stands have a higher transpiration rate than old stands (Macfarlane et al., 2010; Roberts et al., 2001) and consume an amount of water approximately twice as much as old-growth stands (Vertessy et al., 2001). This suggests that stand age in plantations is a crucial factor which could be managed to increase water yield since juvenile trees affect water yield more negatively than old trees.

7.4.2. Mature and old forests Vs SWA

The second scenario encompasses mature trees, which consume less water and could evaporate less than younger trees (**Fig. 2d**). This statement is supported by research finding in South Africa, which pointed out that *pinus* and *eucalyptus* plantations 30 and 15 years of age, respectively, appeared to return streamflow to pre-afforestation levels (Scott and Prinsloo, 2008).

A previous study highlighted that annual water use had decreased from 679 to 296 mm for 50-year-old and 230-year-old stands, respectively (Dunn and Connor, 1993). These findings align with a study by Sun et al. (2008) that highlighted that regrowth of hardwood forests might take as long as 8–25 years before recovering that of a mature forest. Mature and old-growth forests have moderate ET and consistent water yield, while managed forest plantations provide low water yield, particularly during the dry season (Jones et al., 2022) and thus affects water flow regulation (Ferraz et al., 2013). This finding corroborates other studies conducted in the Tropical Atlantic Forest region of Brazil such as that of and, in South Africa, in another study conducted by Dye (1996). Mature *eucalyptus* and *pinus* plantation ages positively correlate with water availability (Lesch and Scott, 1997; Scott and Prinsloo, 2008). Undoubtedly, forest age in forested watersheds is correlated with the regional mean annual streamflow (Haydon et al., 1997), which is one of the factors that increases ET partitioning (Wang and Wang, 2017; Wang and Wang, 2018).

The transpiration rate of trees varies in the following order: young forests > mature forests > old forests (**Fig. 2**). Likewise, there is a relationship between the height of a tree and water stress on a watershed scale. For example, tall trees have very high evapotranspiration rates and therefore experience great water stress (Emanuel et al., 2010). The relationship between forest age and SWA follows the previous order, indicating that young forests consume more water than mature and old forests (**Fig. 2e**). In such a case, plants can passively use their roots to enable water redistribution in the soil profile (Bonan, 2002). The effect of forest age on SWA depends on other factors, including the types of trees and climatic conditions. Of note, forest type, species, age, environmental conditions, and forest management practices are among the factors enabling water-use efficiency (Zhang et al., 2023).

7.5. Influence of forests on water quality

Rainfall regimes can combine with forests plantations to modify watershed hydrology. Runoff generation mechanisms can alter water quality, particularly in agricultural lands where runoff may contain pesticides and affect the soil properties of the downstream buffer zones (Syversen and Bechmann, 2004). In such cases, soil properties may be influenced by both tree species and dominant pedogenetic processes (Romanyà et al., 2005). Dense vegetation represents a prominent alternative for reducing colloidal contaminants in surface runoff (Yu et al., 2012) and promotes water conservation (Zongo et al., 2017). In such cases, forests can greatly contribute to ecosystem services and natural resources management, including water. The concentration and total amount of nutrients (e.g., phosphorous and nitrogen) transported in runoff can be affected by soil type (Gilley et al., 2001; Utzig et al., 2023) and thus alter water quality. The movement of nutrients, SWA, and soil production are dependent on and regulated by bedrock weathering (Pedrazas et al., 2021). Some trees have continuous and deep roots to absorb and recover nutrients and thus mitigate the deterioration of water quality. In such cases, trees can be used for phytoremediation techniques to remove trace metals from the soil (François et al., 2023). Several researchers have argued that cacao trees remove trace metals of cadmium from the soil (Oliva et al., 2020; Oliveira et al., 2022; François et al., 2023) and reduce soil degradation problems (Hooke and Sandercock, 2012; Sarvade et al., 2019). Of note, the conversion of forest soils into pastures and row crops may cause deterioration in the quality of water resources (Neary et al., 2009).

8. **Conclusions and future research**

Forests can influence the amount of water available at some stages of the hydrological cycle. In this review, the roles of forests were addressed while considering several factors, such as stand density, forest type, tree species, stand age, and soil composition. This review also analyzed the influence of forest cover on hydrological processes. Overall, it was admitted that afforestation could positively affect soil erosion control on degraded soils. Nevertheless, this impact can change due to tree litter, forming a layer more permeable to infiltration. The findings showed that trees and watershed hydrology have complex interactions, where the effects could be either positive or negative on SWA, watershed yield, and groundwater recharge. The effects of forests on watershed hydrology mainly depend on the type of aquifer and other characteristics such as local or regional climate, canopy type, soil composition, tree density, and landscape topography. The type of trees to be planted should be taken into consideration, as fast-growing trees (e.g., *eucalyptus*, and *pinus*) reduce SWA. The strength of this review lies in the fact that it encompasses a range of evidence about the interaction between trees and watershed hydrology, as well as how different environmental and geological factors can affect this complex relationship. The novelty of this study highlighted that trees' effectiveness in increasing water availability occurs with the use of some specific species used for afforestation on large-scale watersheds, where trees increase groundwater recharge. Afforestation with proper trees can help increase SWA. Admittedly, less dense forests are more likely to increase the different components of the water cycle than denser forests. Also, trees can be used as a phytoremediation technique to reduce transport of chemical elements in surface runoff and thus limit soil degradation and water contamination. Regarding the impacts of trees on runoff, they could reduce it, depending on the type of forest cover (e.g., plantation versus native forests), stand age, density, and species. One of the limitations of this review is that it did not explore the relationship between tree roots and SWA in depth. Further research is necessary to identify other factors (such as shapes and directions of root systems) that may impact the relationship between trees and other components of watershed hydrology. In conclusion, our conceptual model demonstrates that native forests play a crucial role in natural resource management. This review may prove to be helpful to decision-makers in choosing the best alternative for afforestation strategies in some specific areas.

Author contributions

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Chapter III

Impacts of cacao agroforestry systems on climate change, soil conservation, and water resources: a review

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Impacts of cacao agroforestry systems on climate change, soil conservation, and water resources: A review

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Abstract

Agroforestry is crucial for improving water quality deteriorated by anthropogenic activities due to the use of chemical substances, including nitrogen (N), and phosphorous (P) in modern agricultural practices (MAPs). This state-of-the-art review revised the influence of agroforestry systems (AFS) on climate change, soil conservation, and groundwater quality. The novelty of this review is that we found evidences that AFS can improve water quality, reduce nutrient loss, and support biological, chemical, and physical properties of the soil. The surficial geologic controls, slope gradient, soil types, and topographical conditions are factors that alter a watershed dominated by agroforestry areas. In addition, anthropic aspects, including agricultural practices, can also cause loss of water quality in basins dominated by an AFS area. This review is also novel in that it outlines how AFS can be used for the phytoremediation of contaminated soils to reduce aluminum (Al), mercury (Hg), and cadmium (Cd). Therefore, AFS can be used for water decontamination, climate change mitigation, climate adaptation, and soil conservation. Further research is required to investigate the contribution of AFS to soil integrity.
Keywords: agroforestry systems, climate change, ecosystem services, soil conservation, water quality

1. Introduction

Agroforestry is practiced around the world and can deliver several environmental benefits (for instance, climate change mitigation and climate adaptation), soil conservation (e.g., erosion control, enhanced soil fertility) and ways of dealing with the problems posed by MAPs, such as loss of water quality caused by the overuse of chemical fertilizers (e.g., N, and P) and pesticides. MAPs are a type of agricultural practice that is based on maximizing grain yields without respect to quality, the environment or soil conservation. Fertilizers are broadly used in MAPs to improve the soil fertility (Kadigi *et al.*, 2020). Agroforestry is one of the most effective methods to be used to reduce a range of forms of abiotic stress, including drought, metal, and salinity resulting from a combination of climate change and the overuse of pesticides and chemical fertilizers, which have negative impacts on plant yields (Kumari *et al*., 2022). Pesticides are widely used by citizens, agroalimentary enterprises and cooperatives. In 2019, the global consumption of pesticides was estimated at 4.2 million tons (Foong *et al*., 2022). Consequently, the severe impact of pesticide usage in agricultural activities has contributed to more than 50% and approximately 70% of freshwater eco-toxicity and human toxicity, respectively (Foong *et al*., 2022). The chemical elements in soils cause eutrophication and thus the death of some aquatic organisms (Wear & Greis, 2002), as well as compromising the health and life of human beings. It was estimated that 1 out of 8, 1 out of 6, and 1 out of 4 people are at high risk from biochemical oxygen demand, N, and P pollution, respectively (International Food Policy Research Institute & Veolia, 2015). The Institute for Health Metrics and Evaluation (IHME) reported that 3.4% (80 million disability-adjusted life years (DALYs)) of the world's diseases are associated with inadequate water supply and sanitation (Weidema & Fantke, 2018). Drinking contaminated water is responsible for several diseases, including diarrhea, cholera, and hepatitis A. The Global Burden of Disease (GBD) estimated that 1,800,000 people died from diseases associated with water pollution in 2015 (GBD 2015 Risk Factors Collaborators, 2016). Drinking contaminated water causes a total of 485,000 deaths each year as a result of diarrheal diseases (WHO, 2017). Researchers' attention has come to focus on effective ways to deal with these challenges, taking into account the different roles of agroforestry associated with ecosystem services (ES), including groundwater quality.

Differently to MAP_s, agroforestry is a sustainable alternative that protects soils against erosion and enhances water quality (Neto *et al*., 2007). However, there is uncertainty about the impact of cacao (Theobroma cacao L.) trees on water quality. Cacao is generally grown in the shade of a thinned forest (Lobão *et al*., 2007), and is mainly grown in tropical areas of Central and South America, Asia, and Africa (Marita *et al.*, 2001). Cacao is one of the most important perennial crops in the world. Cacao production worldwide was estimated at approximately 3.5 million tons in 2006 (ICCO, 2007). In 1970, the production of cacao represented approximately 0.6% of Brazil's Gross National Product (Fonseca *et al*., 2020). In 2015, the production of cacao in the southeastern region of the state of Bahia, Brazil, represented 87% of the agricultural land (Sanches, 2019). Integrated cacao agroforestry is a sustainable solution, combining forests and agricultural activities to reduce runoff and soil erosion, mitigate climate change through the storage of carbon, provide goods to human beings, and promote conservation and biodiversity. As compared to the preserved forest, CO_2 emissions in cacao AFS are variable and depend on soil characteristics attributed to the type of vegetation cover (Costa *et al*., 2018). Likewise, agri-silvi-based AFS had higher soil moisture than sole crop fields (Bhat *et al*., 2016). Other researchers have argued that AFS have a great influence on the supply of regulating ES and enhancing landscape structure (Kay *et al*., 2018). The adoption of agroforestry can present benefits that significantly differ from other land use. For example, hedgerow based AFS is more efficient than other land uses regarding its role in reducing surface runoff (Bhat *et al*., 2016). AFS soil quality does not differ from that of the forest (Rousseau *et al*., 2012), and can therefore contribute to the regulation of the water cycle and modify soil water availability.

Forests can act as water filters depending on the forest extensions and forest types. For instance, riparian forests contribute to clean water owing to their positive influence on pesticide removal in water-saturated zones (Aguiar *et al*., 2015). The researchers argued that wood>shrubs> grass in terms of forest performance for removal of pesticides in the riparian zones. As such, riparian vegetation has a fundamental role in the ES associated with water quality. In other words, agroforestry can play a crucial role in restoring riparian habitats and solving river problems. In addition, agroforestry can be seen as encouraging environmental protection and biodiversity conservation (Nair, 2011). For example, vegetative filter strips (VFS) can be planted in an area with the purpose of removing sediments, nutrients, and pesticides from both surface and subsurface waters (Dillaha *et al*., 1989; Nair & Graetz, 2004; Nair, 2011). The VFS play an impressive role in controlling erosion rate and maintaining soils in the field (Grismer *et al*., 2006). Also, AFS are suitable strategies for soil conservation and infiltration recovery. Soil water content (SWC) can be increased in the agroforestry areas depending on the type of tree, slow-growing or fast-growing trees, and soil composition. Taking into account the importance of AFS for environmental challenge mitigations, and their importance for ES, this state-of the-art review investigated the influence of AFS on climate change, soil conservation, and groundwater quality. We hypothesize that: (i) agroforestry can control erosion and improve water quality; and (ii) agroforestry can be used for pollution abatement in soils and groundwater. This review proceeds as follows: the introduction, importance of AFS for soil conservation, the role of AFS in enhancing water infiltration, the importance of AFS on soil chemical properties, role of AFS in reducing climate change, influence of AFS on water quality, future challenges and ways forward, and, finally, conclusions.

2. Importance of AFS for soil conservation

Finding a sustainable solution to deal with soil degradation is becoming more and more crucial to maintaining the conservation and integrity of the soil. Soil integrity refers to the capacity of soil to perform its essential functions (Karlen *et al.*, 1997), whereas soil conservation is a way of controlling soil erosion and maintaining soil fertility (Young, 1989), which is crucial for the environment. AFS has an effective role in improving physical properties (**Figure 1**) by adding organic matter, including soil structure, and porosity (Kumar *et al*., 2020). It has been discovered that AFS improve soil structure through the plant material of tree components and the addition of organic material (Muñoz-Rodríguez *et al*., 2019).

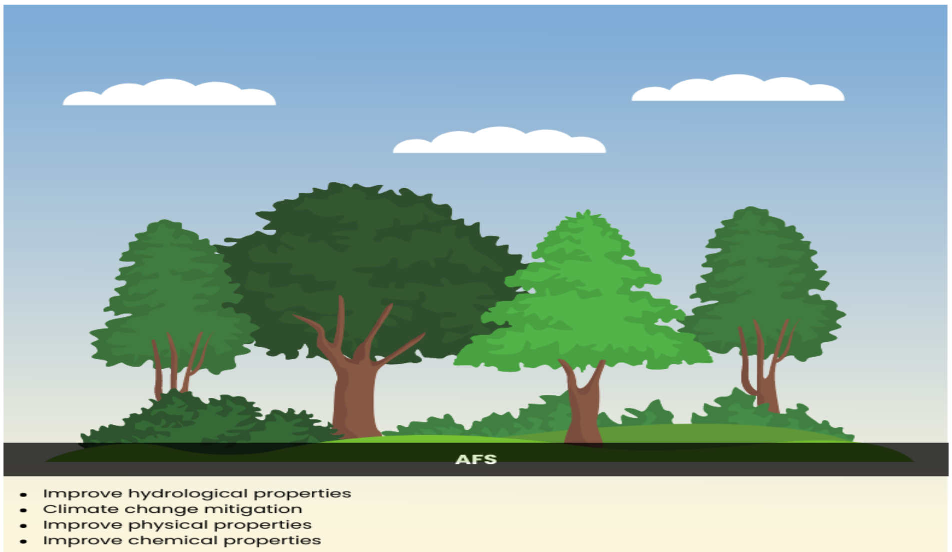

Fig. 1: Benefits of AFS regarding soil conservation and climate change mitigation

Research implemented in Pasto city, Colombia, has shown that the distribution of aggregates changes from 60% to 70.3% before and after agroforestry interventions (Muñoz-Rodríguez *et al.*, 2019). This shows that AFS contribute to strengthening soil structure, and avoid runoff transporting agricultural soil to the sea, lakes, or rivers (**Figures 1 &2**). Noticeably, trees enhance the quantity of water used on-farm for trees and increase the biomass of trees or crops produced per unit of water (Ong, 2006). Another research reported that bulk density, porosity, field capacity, and wilting point varied over the years in agroforestry areas (Arévalo-

Gardini *et al.*, 2015). Along the same line, research on pecan AFS has shown that soil bulk density decreased from 16.13% to 7.10%; however, soil content and total porosity increased (Wan *et al*., 2022). Soil bulk density can vary from one soil type to another, depending on the duration of agroforestry. Soil bulk density, noncapillary porosity, and capillary porosity are often influenced by a set of factors, such as field capacity, saturated water holding capacity, and saturated hydraulic conductivity (Jiang *et al*., 2019). AFS are crucially in rainy regions, and the mountains to slow down heavy rainfall and infiltrate it into the soil, and prevent farmers from losing their crop yields, respectively.

2.1. Importance of AFS for Ecosystem services

Among numerous alternatives that can be used for natural resources management, agroforestry is one of the most sustainable and environmentally friendly. Agroforestry enhances ES via improved soil structure and increases water retention and carbon sequestration (Mukhlis *et al.*, 2022), which can be possible thanks to the decomposition of fine roots of trees (Siegwart *et al*., 2022). As such, the carbon sequestration rate might depend on soil depth. For example, topsoils, upper soil, and lower subsoils can potentially influence carbon sequestration in AFS in the following order: topsoils (0.92 Mg ha^{-1} yr^{-1})> upper subsoils (0.72 Mg ha^{-1} yr^{-1})> lower subsoils (0.52 Mg ha^{-1} yr^{-1}) (Hübner *et al.*, 2021). Moreover, research carried out in the West African Sahel has concluded that agroforestry can promote landscape resilience in the greater region (Ellison & Ifejika Speranza, 2020). For example, AFS have reduced soil loss from 1.22 to 0.17 t/ha over a six-year period (Panwar *et al*., 2018). This reduction might be possible due to the capacity of cacao trees to maintain and improve soil structure. These findings explain that agroforestry has great importance for soil conservation. As such, AFS promote sustainable development (i.e., functional ecosystems, livelihoods, and human security) through the accomplishment of its goals which are providing water, energy, and food security (Elagib & Al-Saidi, 2020). Importantly, AFS also protect land use and maintain good physical, chemical, and biological properties in tropical soils (Arévalo-Gardini *et al.*, 2015; Arévalo-Gardini *et al.*, 2020; Matos *et al.*, 2020), and contribute to the restoration of a local landscape scale (Martin *et al.*, 2021). However, these properties can be improved if cover crops and trees are included in the system (Alegre & Cassel, 1996).

Similarly, scholars underscored that the biological, chemical, and physical characteristics of forest soils contributed to high quality water in streams, and moderated stream hydrology (Neary *et al*., 2009). Research conducted in Ejido, Mexico, has demonstrated that cacao agroforests have lower land change as compared to other types of land tenure (Oporto-Peregrino *et al*., 2020). Cacao agroforestry, it is worth noting, is characterized by a large accumulation of the amount of soil organic carbon (SOC) (Monroe *et al.*, 2016), and it can be used to reduce deforestation and forest degradation (Batsi *et al*., 2020), improve soil fertility, maintain soil ecological functions, and restore soil quality to degraded pasturelands (Suárez *et al.*, 2020).

3. Importance of AFS for soil Chemical properties

In recent years, studies have been carried out to searching for sustainable routes to maintain the soil's physicochemical properties, which are essential for its conservation and productivity. Research conducted in Nepal showed that there were significant differences in soil pH, Al content, base saturation, electrical conductivity, organic matter and N content, and cation exchange capacity between AFS and conventional system (CS) (Schwab *et al.*, 2015). This finding was consistent with the finding of research conducted by Tornquist *et al.* (1999) in Costa Rica, which concluded that soil exchangeable bases and pH were lower in agroforestry treatments as compared with pastures. This indicates that AFS have higher soil quality and are more fertile than CS. AFS have more ability to ensure plant and animal health, and to preserve environmental quality, as compared to CS. This finding is supported by another study carried out in the central mid-hills of Nepal by Schwab *et al.* (2015). AFS reduce surface runoff, SOC, and associated nutrient losses by an average of 58%, 65%, 9%, and 50%, respectively (Zhu *et al*., 2020). The reduction of runoff by the AFS is a part of soil conservation and therefore maintains the chemical and physical properties of soil (**Figure 1**). Tree roots in AFS reduce N and P residues in soils by 20% to 100% (Pavlidis & Tsihrintzis, 2018). Findings of other research have also reported that AFS can reduce pesticide leaching and runoff by up to 90% (Pavlidis & Tsihrintzis, 2018). This is possible due to soil and water management in this system (Wang *et al.*, 2017), which can potentially reduce non-essential trace metals in soil. For example, cacao trees enable the reduction of non-essential trace metals such as Cd and thus transfer it from roots to the shoot (Oliveira *et al*., 2022) (**Figure 2**). Another research conducted in the province of Bagua in Amazonas, Peru, indicated that the amount of Cd concentrated in the roots of cacao trees is five times higher than the Cd level in the soils (Oliva *et al.*, 2020). Another study conducted in Brazil highlighted that AFS improved soil chemical indicators based on pH increase, soil nutrient content (Ca, Mg, and K), reduced Al saturation (Silva *et al.*, 2020), and Hg (Béliveau et al., 2017) (Figures 1 & 2). Notably, AFS contribute to the maintenance of soil integrity (Béliveau *et al*., 2017). These findings indicate that cacao AFS is effective at reducing trace metals in soils that are often caused by MAPS. Accordingly, AFS are an effective technique to be used as pollution abatement for soil and water decontamination (**Figure 1**). For instance, trees or woody plants with developed root systems and large biomasses are attractive for vegetation and phytoremediation in metal-polluted sites (Lee *et al.*, 2009; Capuana, 2013).

4. Role of AFS in enhancing water infiltration

AFS play a crucial role in reducing the need to drain, soil evaporation, runoff, erosion, and silting up of rivers (**Figure 2**). The reduction in runoff can be attributed to ground cover plants in AFS, and the slope of the landscape. For example, research carried out in the Philippines reported that the risk of soil erosion on slopes was 8% higher with non-agroforestry use than with agroforestry use (Delgado & Canters, 2012). Other researchers underlined that AFS increased aggregate soil stability, and decreased runoff and soil erosion (Roose & Ndayizigiye, 1997; Marwah, 2012), due to enhanced infiltration (Mwangi *et al*., 2016), which resulted in increasing soil water storage (Zhao *et al.*, 2022). Scholars have pointed out that agroforestry can increase evapotranspiration (ET) in the watershed (Wang *et al*., 2017) and thus reduce water availability owing to water uptake by tree roots (Mwangi *et al*., 2016), which is strongly related to root-length densities and root surface areas (Bayala & Prieto, 2020). Similarly, findings of another study highlight that apple fine roots decrease the SWC (Shen *et al.,* 2022). Importantly, the ability of trees to reduce the SWC depends on tree and soil types. For example, a previous study demonstrated that the SWC decreased rapidly during the summer in Putnam silt loam dominated by agroforestry buffers (Sahin *et al.,* 2016). In contrast, the walnut-wheat alley cropping system (*JTACS*) increased water infiltration in the shallow soil layer during the rainfall season (Wang *et al*., 2015). Water availability through soils and air moisture is one of the main drivers of AFS (Pérez-Girón, 2022). In the case of cacao trees, the density of lateral roots exponentially decreases with depth, and 20% of these roots can uptake water (Mommer, 1999). Research carried out in a coastal area of northern Iran has indicated that the combined scenarios of agroforestry, no-tillage, and rice straw mulch are an effective way to increase groundwater recharge in the region (Mohseni *et al*., 2022). Thus, AFS has among its functions the regulation of rainwater, support of the microclimate, and ecosystem stability (Theresia Sri Budiastuti *et al.,* 2021).

4.1. Factors influencing soil water availability

The effectiveness of agroforestry to increase or reduce the SWC may depend on the type of vegetation cover, and soil properties. Vegetation density and type of vegetation are correlated with soil hydraulic properties (Reddy *et al.*, 2016). Similarly, organic matter produced by AFS enhance the effective rewetting and water retention capacity (Lestari & Mukhlis, 2021).

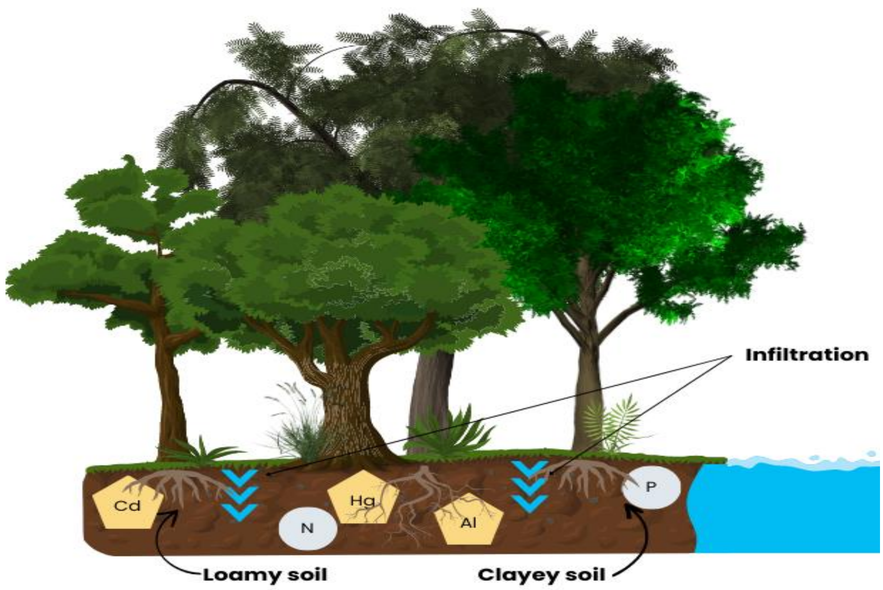

Fig. 2: Role of AFS to enhance soil water infiltration

Agroforestry reduces excessive evaporation and maintains groundwater availability during the dry season (Lestari & Mukhlis, 2021). It also helps improve soil quality by cooperating with water stress and improving hydraulic conductivity and infiltration, owing, perhaps, to grass and cover crops. Therefore, agroforestry is a strategy to be used to control runoff and erosion. Infiltration has a major impact on controlling soil erosion and runoff, soil moisture content, and groundwater recharge (Lozano-Baez *et al*., 2019) (**Figure 2**). Depending on the precipitation regime, runoff carries chemical substances from an agricultural field into the watershed (**Figure 2**). Consequently, syntropic agroforestry systems, which are a type of agroforestry involving very large amounts of organic matter, can be used to improve soil water infiltration (Murta *et al.*, 2021). In addition, there are other solutions such as covering the soil, rather than leaving the bare soil. Recent research has highlighted that AFS reduce the total soil loss and runoff by 37.7% and 19.1%, respectively (Jinger *et al*., 2022).

4.2. Relationship between tree roots and water infiltration

There is a complex relationship between soil water infiltration and tree roots. For example, trees with deep roots increase macropores vertically and modify both micro and mesopores laterally (Wang *et al*., 2017) as well as being more suitable to improve soil moisture content as they are able to lift up or redistribute water to the topsoil via a process known as hydraulic lift (Bayala & Prieto, 2020). Both hydraulic lifting processes and deep roots allow trees to uptake the groundwater from the deeper layer and make it available to understory vegetation with shallow roots (Pérez-Girón, 2022). In addition, deep-root trees improve soil physical conditions and higher soil microbiological activities under AFS (Nair *et al.*, 2008). Thus, the infiltration depends on the land use history prior to the application of agroforestry in an area (Lozano-Baez *et al.*, 2019). For instance, mechanical clearing reduces infiltration rates because the infiltration can depend on the woody vegetation roots. As such, the infiltration rate of soil increase is positively linked with the enhancement of soil mechanical stability (Schnug & Haneklaus, 2002). Furthermore, the greater the infiltration capacity of buffer soils, the more agroforestry species are able to remove pesticides from overland runoff (Dollinger *et al*., 2019). Infiltration may vary with agroforestry practices and plants used and may be affected by plant age, which could reduce baseflow and water yield in the soil or watershed due to a possible increase in ET of some agroforestry species.

4.3. Relationship between water infiltration and soil characteristics

Soil moisture may depend on a range of factors, including the type of land cover and soil characteristics. The uppermost soil layer in the AFS has higher soil moisture as compared to monoculture (Niether *et al*., 2017). On the other hand, Sarto *et al*. (2022) highlight that AFS reduce water availability in the superficial soil layer, particularly during the rainy seasons. These findings indicate that the capacity of trees to recharge the groundwater depends on several factors, including the climate, soil types (e.g., sand, silt, clay, etc.), and soil characteristics, including fracture depth, soil texture, and fractured rock. Water in the soil can be influenced by pedotransfer functions (Reichert *et al*., 2009), which have a great influence on infiltration and water quality. For example, research reported that loamy sites could have a water availability greater than clay loam or sandy (Dodd & Lauenroth, 1997). Similarly, another study indicated that infiltration can be lower in sandy soils than in clayey soils (Lozano-Baez *et al*., 2019). Therefore, these findings indicate that a high infiltration rate depends on soil characteristics, including high soil aggregation. The quality and quantity of coastal waters result from a complex interaction of anthropogenic activities with soil and climate (Daly et al., 2018). In other words, coastal and groundwater quality depend on soil properties and land use. The soil in a constructed wetland is an optimal substrate and has great performance in the adsorption and passivation of pollutants, including N, P, and heavy metals in water (Cheng *et al*., 2021). Another study has outlined that a decrease in soil pH and salinity accompanied by a decrease in Cl and Na concentrations is registered in an area irrigated with treated wastewater combined with the plantation of agroforestry species (Zouari *et al*., 2019).

5. Influence of AFS on water quality

Groundwater quality depends on a complex system (land use, anthropogenic activities, and vegetation cover) in which the physicochemical properties of the soil have a major influence (**Figure 1**). Chemical substances used in MAP_S cause the contamination of water. This challenge draws researchers' attention to focus on AFS, which include as advantages the improvement of water quality, carbon sequestration, biodiversity conservation (Zinkhan & Mercer, 1996; Tomer *et al*., 2008; Jose *et al*., 2009; Kumar *et al*., 2010; Holzmueller & Jose, 2012; Lovell *et al.,* 2017; Alagele *et al*., 2018), nutrient cycling and water retention (Sollen-Norrlin *et al.,* 2020), and reductions in the use of chemicals fertilizers (**Figures 1, 2 & 3**). In addition, AFS reduce water loss and wind velocity, and thus limit wind erosion (Kumar *et al*., 2020). In contrast to AFS, MAP_S can potentially exacerbate soil loss, contaminate groundwater, and cause pollution from other sources (Nair & Graetz, 2004; Schultz *et al.,* 2004). Such practices endanger human health and harm aquatic life because of the deterioration of water quality and eutrophication, respectively. Watersheds may contain suspended solids that are affected by erosional factors in runoff (Rodríguez-Blanco *et al.*, 2019). Certain substances may remain in the waterbody for long periods of time, whereas others remain for only a short time, depending on the substance and the precipitation regime. AFS reduce the losses of herbicides, pesticides, and other pollutants by 55-100% (Zhu *et al*., 2020). The reduction occurs through the extensive lateral root systems that scavenge soil nutrients and thus redistribute them beneath tree canopies (Rhoades, 1996).

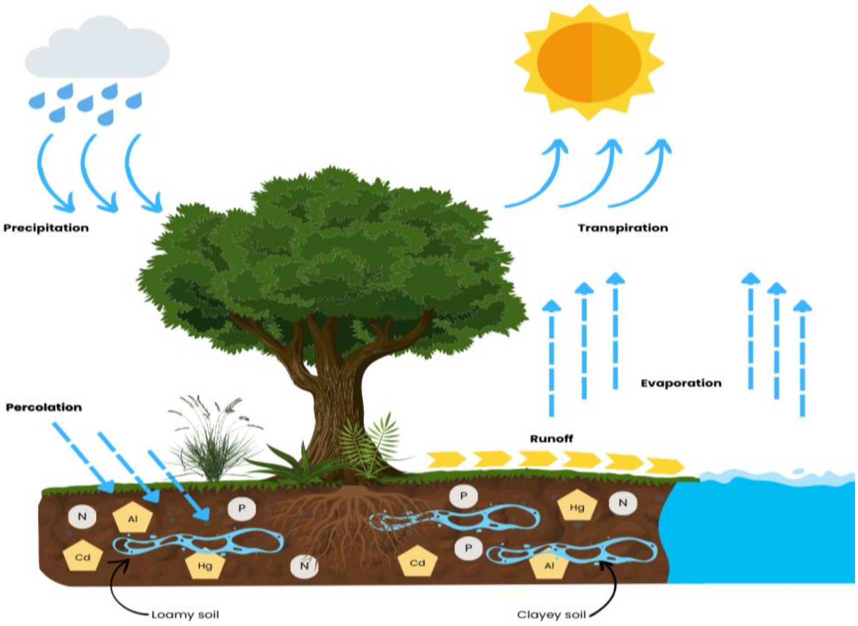

Fig. 3: Role of AFS in improving water infiltration and reducing contaminants in soil

A study conducted in Spain indicated an agroforestry catchment feeds a water reservoir with a sedimentation rate estimated at 3.5 tons/ha per year (Palazón *et al*., 2016). This amount of sediments depends on agricultural practices, particularly the types of chemical substances used in the AFS. Pesticides can threaten groundwater quality if they are used in the root-affected zone (Dollinger *et al*., 2019). Moreover, other factors (for instance, precipitation, slope, soil porosity, and buffer localization) can also play an important role in water pollution. According to Canencia *et al*. (2011), N, P, and potassium levels in watersheds dominated by AFS are moderately high and suitable for crop production. This demonstrates that agroforestry has great potential to remove these chemical elements.

5.1. *Utilization of AFS as a phytoremediation technique*

Watershed contamination is a worldwide problem that can become hypertrophic and thus cause the extinction of some species of aquatic organisms when these chemical elements are in high concentrations. These challenges have compelled researchers' interests to search for facile and sustainable solutions, such as agroforestry. Kim & Cho (2014) found in their study that agroforestry crop fields and open crop fields had similar runoff loadings of total N, total P, and total suspended solids. Notably, AFS can reduce nonpoint source pollution due to tree growth and tree roots taking more space in the soil, reducing the amount of fertilizer in the watershed (Udawatta *et al.*, 2009) and generating cleaner water (Brown *et al.*, 2018) (**Figure 3**). As compared with other existing reviews, this study shows that, in an agroforestry area, trees with deep root systems have a great influence on the effectiveness of removing contaminants in soil and improving groundwater quality through trapping of nutrients, and metals deposited in the surface and topsoil (Kumar *et al*., 2020; Pavlidis *et al*., 2021). Therefore, agroforestry acts as a technique for the remediation of polluted soils owing to tree roots that can uptake the excess agrochemicals that would otherwise pollute groundwater via leaching and surface water via runoff (Pavlidis *et al*., 2018; Pavlidis *et al*., 2020; Pavlidis *et al*., 2021). In other words, AFS can act as a technique to improve water quality, which is an objective of ES (Keeler *et al*., 2012).

6. **Role of AFS in climate change mitigation**

The challenges of managing climate change are increasingly drawing researchers' attention to the need to find sustainable solutions. AFS can be an effective tool for mitigating climate change (De Zoysa & Inoue, 2014; Murthy *et al*., 2016) and carbon and environmental footprints, by reducing greenhouse gas emissions into the atmosphere (Tefera *et al.,* 2019), regulating rainwater (soil and water conservation), and supporting the microclimate (Budiastuti *et al*., 2021), due to the shade provided by this system (Lestari & Mukhlis, 2021). AFS are suitable for climate mitigation, particularly as they have a high density and diversity of shade trees

(Boreux *et al*., 2016). The adoption of AFS with 50% shade cover can reduce the mean temperature (Gomes *et al*., 2020). AFS are climate-resilient agricultural practices (Jhariya *et al*., 2019). Other researchers have pointed out that AFS can contribute to climate adaptation (Verchot *et al.*, 2007). Shade has an important role to play in the reduction of temperature and enhances rainfall (Panozzo *et al.*, 2022). The effect of trees on soil water conservation increases with the intensity of shading. Thus, the density and composition of the species have a great influence on climate mitigation. For example, coffee systems reduce the effect of extreme temperature and precipitation (Lin *et al*., 2008), findings have shown that the shaded coffee AFS improve microclimate conditions and deep-water drainage as compared to unshaded coffee systems (de Carvalho *et al*., 2021).

Of note, AFS have an evident ecological advantage over monoculture, particularly for young apple trees in the semiarid region (Zhao *et al*., 2022). As compared to older cacao trees, shade is more necessary for young cacao tree plantations (Tscharntke *et al*., 2011). In addition, AFS can both contribute to capturing soil carbon and play an important role in mitigating atmospheric CO_2 (Verchot *et al*., 2007; Rita *et al*., 2011). The total carbon (aboveground and root biomass) stored in an AFS, including cacao and shade trees, was 2.5 times higher than in a monoculture (Niether *et al.*, 2020). Shade trees have great importance in cacao AFS because of their role to influence radiation, wind regimes, nutrients, and hydrological cycling (Tiralla *et al.*, 2013). Notably, shade trees within AFS protect the understory cacao against extreme climate (Niether *et al*., 2018). Cacao agroforestry can make an important contribution to mitigating climate change and contribute to the national economy of a country. AFS can also be a promising solution for improving low soil fertility in tropical zones (Mendoza & Parra, 2020) and contribute to the achievement of Sustainable Development Goal 2, which aims to end global hunger.

6.1. Contribution of AFS to natural resources management

People around the world are currently facing food and water insecurity owing to the depletion of natural resources, such as water and crops. This challenge has increasingly spurred scientists to seek out a simple and environmentally friendly solution. Scholars report that AFS maintain high levels of biodiversity and biomass (Sistla *et al*., 2016), which are necessary for the integrity and conservation of the soil. Similarly, other researchers argue that AFS combine production and conservation (Vallejo-Ramos *et al*., 2016) and are the best option to deal with the erosion, and degradation of biodiversity (Saha *et al*., 2010) usually caused by the MAPs. Scientists have reported that farmers are used to felling trees to enlarge their gardens and producing furniture (François *et al*., 2022; Pauleus & Aide, 2020). Importantly, AFS can reduce deforestation and cultivable land extension (François *et al*., 2022), protect environmental areas (Laudares *et al*., 2017), recycle nutrients lost in MPAs (Izac & Sanchez, 2001) via their root systems (Raj, 2020), lift people out of poverty and stem the decline of agricultural productivity (Jama *et al*., 2006). In the other words, AFS is a promising and sustainable option for natural resources management and this can contribute to human well-being and environmental protection.

6.2. Importance of cacao-AFS for water loss reduction

Cacao agroforests are described as being sensitive to climate change and water deficit (Lahive *et al.,* 2019), do not cause undesirable environmental change (Gama-Rodrigues *et al*., 2021), and have a crucial role in drought-tolerant land use (Gateau-Rey *et al.,* 2018), particularly in tropical regions (Schwendenmann *et al*., 2010). Wang *et al.* (2017) have underscored that AFS increase vertical preferential flow and retard the subsurface lateral flow and thus enhance water retention capacity. Another study highlighted that cacao is tolerant to shade, and the maximum photosynthetic rate occurs at an irradiance of around 400 μmol m^{-2} s^{-1} (Arévalo-Gardini *et al*., 2021). Admittedly, unshaded cacao monocultures are vulnerable to climate change (Heming *et al.*, 2022). In contrast, shade trees reduce water loss from top soil via ET of the cocoa (Niether *et al*., 2017). Therefore, canopy structure has a great influence on the ET in AFS (Wang *et al.*, 2021). The age of trees may have a great influence on the capacity of AFS to enhance the advantages and ecological services.

7. Future work and way forward

It is crucial to find a sustainable and environmentally friendly alternative to deal with the problems of soil loss, nutrient loss, soil contamination, climate change, and water contamination. These challenges are prompting scientists to focus more on AFS. Herein, this present review revised the influence of AFS on soil conservation, climate change, and groundwater quality. This study revealed that AFS offer a sustainable approach for dealing with climate change, groundwater contamination, and nutrient loss, among other environmental challenges. The strengths of this study encompass a range of evidence about the relationship between AFS and water quality and how environmental and geologic factors, as well as human activities, can be associated with agroforestry to contaminate the waterbody. Insufficient evidence about how woody plants can influence the intercept and/or reduction of heavy metals in soil is a limitation of this review. Another limitation is that it does not take into account the effects of fast-growing and low-growing trees on soil water availability. This lack of evidence could be filled via an experiment carried out using fast-growing and low- fruit trees planted in arid, semi-arid and/or humid sites where the SWC is measured before and after the observations. Such an observation requires considerations as to the soil type and the different phases of the trees' growth as important factors. Researchers report that fast-growing trees are not suitable for forestation in areas with medium precipitation and brackish groundwater (Sudmeyer & Simons, 2008), and their photosynthetic rates and stomatal conductance are higher than slow-growing trees (Liu *et al*., 2021). Moreover, the interaction between shape of tree roots, the type soil and trees' physiology should be studied to enlarge knowledge concerning trees' behavior towards infiltration or soil water availability. In conclusion, further studies are required to identify other factors that may influence the ability of AFS to improve water quality and to examine how agroforestry age and agroforestry density can influence soil water infiltration.

8. Conclusions

There is an urgent need to find a sustainable option for reducing soil loss, recharging groundwater, and improving water quality altered by industrial activities and MAPs. This review investigated the effects of AFS on soil conservation, climate change, and groundwater quality. It was found that AFS can enable soil water infiltration and ET. We also found that agroforestry is a promising and sustainable alternative for improving water and soil quality and controlling erosion. This review showed that AFS improve soil structure through the vegetal material of tree components. Also, AFS support soil biological, chemical, and physical properties. Therefore, AFS are crucial for soil conservation and the enhancement of water quality. In addition, AFS are a promising alternative to be used to remove trace metals (e.g., Cd, Al, Hg) in contaminated sites. However, several factors, including surficial geologic controls, slope gradient, soil types, and topographical conditions can combine with the AFS to deteriorate water quality in the watersheds. These factors can be accentuated by land use/anthropic activities. The deep-root trees play an important role in reducing pollutants in the soil and enhancing humidity. Our hypotheses were verified because we have found that AFS control erosion and improve water quality, and can be used as a phytoremediation technique to remove trace metals in the soils. The novelties of this review, compared to existing ones, are that AFS can contribute to reducing non-essential trace metals in soils caused by MAPs and improve soil quality by increasing infiltration, soil organic matter, macropores, aggregate stability, and aeration by decreasing erosion, runoff, and soil bulk density, and reducing nutrient loss. Conclusively, the adoption of AFS can be a promising alternative to improve upstream water quality, soil conservation, agricultural sustainability, mitigate climate change and allow carbon sequestration, and reduce environmental footprints.

Author contributions

Mathurin François: Conceptualization, methodology, design, material preparation, data curation, writing-original draft, writing-review & editing, Validation.

Eduardo Mariano-Neto: Supervision, writing-review & editing, Validation

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Data availability statement

All relevant data are included in the paper or its Supplementary Information.

Conflicts of interest statement

The authors declare there is no conflict.

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Chapter IV

Assessing soil erosion and its drivers in agricultural landscapes: the case of southern Bahia, Brazil

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Assessing soil erosion and its drivers in agricultural landscapes: a case of southern Bahia, Brazil

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Abstract

Erosion is a worldwide threat to biodiversity conservation and agricultural yield, and it is linked to deforestation. In this study, we aim to assess soil loss in landscapes of Cachoeira River watershed, in southern Bahia, northeastern Brazil. We estimate the role of forests in diminishing soil erosion using the Revised Universal Soil Loss Equation (RUSLE). We compare real and simulated scenarios in which the forest was replaced by agricultural use, also comparing estimates of erosivity factor (*R* factor) derived from remote sensing and climatological station data. Real and simulated annual soil losses varied from 0 to 15.95 t/year and from 0 to 33.53 t/year along the watershed when climatological station data were used, respectively. However, only 0.04 and 1.67 % of this area is highly and severely exposed to erosion, using data from climatological stations and remote sensing, respectively. Soil loss in the simulated deforested scenario was approximately two times higher than the real annual soil loss, indicating the importance of forest cover to mitigate soil erosion. Moreover, soil loss was 10.5 times greater when using precipitation data from remote sensing compared to climatological stations. Conclusively, the practice of agroforestry can be used as an alternative to avoid erosion.

Keywords: Agricultural, erodibility, QGIS, soil conservation, soil erosion, RUSLE

1. Introduction

Forests are crucial for safeguarding environmental sustainability and, thus, attaining key sustainable development goals. Unfortunately, the current pace at which forests are lost indicates that this process is the leading cause of biodiversity degradation and loss (Magioli & Ferraz 2021). Most importantly, such loss also leads to a decay in the portfolio of services and goods provided by nature – the so-called ecosystem services – that ultimately contribute to human wellbeing. Soils are vital components of the biosphere, providing resources necessary to sustain food production, nutrient cycling, water storage, and habitats that host an amazing diversity of life on earth (Schröder *et al*. 2016; De Deyn & Kooistra 2021). Its presence and quality are closely linked with land use changes, with deforestation being largely responsible for soil erosion, i.e. when soil is carried away and lost. In fact, soil erosion is a global threat (Montanarella *et al*. 2016; Arabameri *et al*. 2020), particularly affecting agricultural productivity and water quality (Benavidez *et al*. 2018; Sidi Almouctar *et al*. 2021). In particular, the transport of nutrients from agricultural lands to river results in water contamination and eutrophication. Nutrients can be transported through river networks from higher to lower elevations (Sarker 2021), often influenced by external factors such as climate change and tectonics (Sarker *et al*. 2019). Soil type plays a crucial role in both soil water retention and nutrient transport via runoff. The interplay of soil type, slope steepness, and slope length significantly affects nutrient runoff from forested areas, influencing the watershed outlet or groundwater recharge. Indeed, water erosion is globally responsible for the loss of 75 billion metric tons of soil every year (Dabral *et al*. 2008), causing a loss of 8 billion dollars in the global economy due to the erosion of fertile land and a reduction in agricultural products (FAO 2019). Although soil erosion can be a natural process, human activities, including non-sustainable agricultural practices, implementation of infrastructure, and other forms of land use changes are exacerbating it. Globally, 84% of the land is degraded by soil erosion (Opeyemi et *al.* 2019), negatively impacting the wellbeing of more than 3.2 billion people (IPBES 2018, Borrelli *et al*. 2021). This major environmental and agricultural threat (Pimentel *et al*. 1995) mainly occurs in agricultural areas, particularly on higher slopes (Salem *et al*. 2014; Ibrahim *et al.* 2019), with the destruction of vegetation cover increasing surface runoff and decreases infiltration (Prosser & Williams 1998).

In Brazil, a country with an economy highly dependent on agricultural production, soil erosion is a major concern. Each year, Brazil experiences an estimated loss of 800 million metric tons of soil (Merten & Minella 2013), placing the country among the global erosion 'hotspots' (Guerra *et al*. 2014). Deforestation is closely linked with runoff and erosion issues. Approximately 70% of the country's population, roughly 140 million people, according to Scarano and Ceotto (2015), reside in the 17 states that were originally part of the Atlantic Forest biome. This intense human presence has significantly diminished this once expansive biome, which was formerly the second-largest tropical forest in the Americas (Tabarelli et al., 2005), now reduced to less than 12% of its original size. While this extensive deforestation likely contributes to increased soil erosion (Zwiener et al., 2017), certain land management practices like agroforestry might alleviate some of these environmental impacts. For instance, in the southern state of Bahia, northeastern Brazil, portions of the native forests have been replaced with cacao (*Theobroma cacao*) agroforests. This type of land use is known for its biodiversity and climate-friendly attributes. However, the effectiveness of such practices in mitigating soil erosion remains a topic that requires further investigation. Herein we aimed to contribute to this discussion by estimating the soil loss associated with erosion in a specific municipality within agroforestry dominated landscapes in southern state of Bahia. The study area lies within the largest forest remnants in Northeastern Brazil (Joly *et al.* 2014), but most of the forest cover in this region comprises cacao agroforests, comprise a forest mosaic hosting most of the rich native biota (Schroth *et al.* 2011). This study also presents an opportunity to compare soil loss assessed using two different data sources, namely remote sensing and climatological stations. Specifically, we raised the following questions: (i) How much soil is lost when non-forested areas are compared to forested areas - comprising the above-mentioned mosaic of native stands and agroforests- and when forest areas are replaced by agricultural areas? (ii) What is the role of rain-induced soil erosion and, (iii) whether such soil loss estimates differ when using remote sensing and climatological station data for the soil erosivity (R factor)?

2. **Methodology**

2.1. Study area and land use mapping

The study was carried out in Ilhéus, a municipality located in the southern part of Bahia state, northeastern Brazil (**Figure 1**). This region was originally dominated by a humid tropical climate (Alvares *et al.* 2013), with an average annual rainfall of 1,830 mm, while the annual mean relative humidity and air temperatures are 80% and 23.5 °C, respectively (Medauar *et al*. 2020). The Atlantic Forest biome, which is a hotspot of biodiversity and endemism globally recognized, has been significantly reduced from its original extent (SOS Mata Atlântica & INPE 2017). Most of the forest cover remaining in southern Bahia, including Cachoeira River watershed, is a mosaic of native stands and cocoa agroforestry, mostly cultivated under a traditional mode regionally known as cabrucas (Sambuichi & Haridasan 2007).

Within Ilhéus municipality, our study encompasses the Cachoeira River watershed, covered by different vegetation classes, including pasture, mosaic cultures, and grassland. The Cachoeira River watershed originates in the Ouricana mountain range in the city of Itororó and meets the Salgado River before passing along the coast of the municipality of Ilhéus (Torres *et al.* 2001). It is limited to the north by basins of the Rios de Contas and Almada, to the south by basins of the Pardo and Una Rivers, to the west by the Rio Pardo basin, and to the east by the Atlantic Ocean (Trindade *et al.* 2010). The Cachoeira River watershed lies btween $14°42'S$ and $39°01'W$.

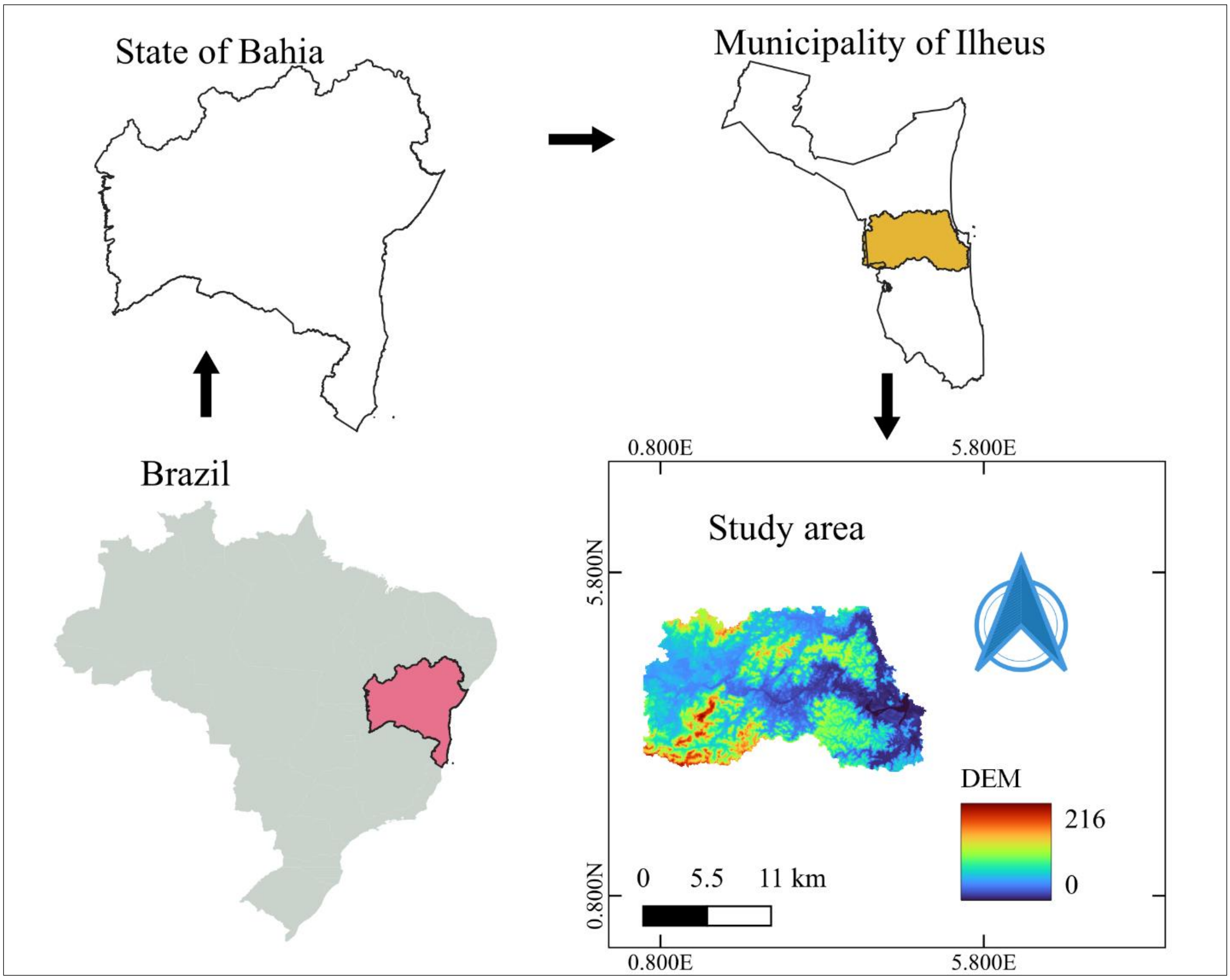


Figure 1. Location map of 272.87 km^2 Cachoeira River watershed

2.2. Modeling

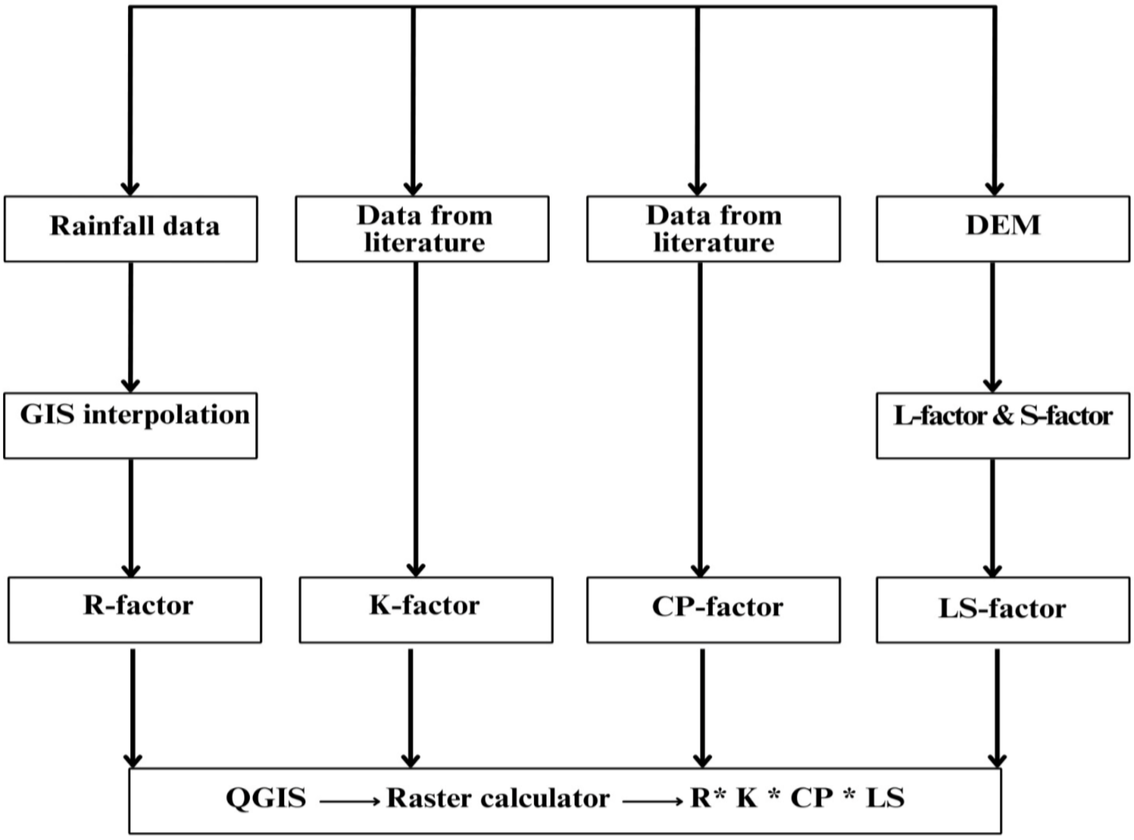
Land use maps from Mapbiomas (https://mapbiomas.org) were used for automatic classifications to identify the spatial distribution of the following classes: forests (including native stands and cacao agroforestry systems), pastures, forestry, open areas, urban centers, exposed soil, and water bodies (e.g., rivers, ponds). Using the QGIS tools we converted vector images to shapefile layers with a 30 m resolution. Two scenarios were considered: real and simulated. In the real scenario, the actual land use land cover (LULC) was used, whereas in the simulated scenario, the forest class was replaced by the agricultural use.

2.3. RUSLE model description

The RUSLE is one of the most popular and widely used to estimate the soil erosion models for agricultural watersheds (Udayakumara *et al.* 2010). This empirical equation considers layers of digital terrain, rainfall erosivity, and interpolated soil erodibility, and the topography for modeling soil erosion (Barbosa *et al*. 2015). In this study, we used the RUSLE model due to its minimal statistics and computation needs (Prasannakumar *et al.* 2012), its flexibility in modeling soil erosion, and its facility to integrate with GIS for spatial analysis (Wischmeier & Smith 1965; Bonilla *et al.* 2010; Prasannakumar *et al*. 2012). According to Renard *et al.* (1997), the RUSLE model can be expressed by multiplying its different factors, as shown in Eq. (1):

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

Where *R* represents the erosivity factor (R factor) and is expressed in $MJ\ mm\ ha^{-1}\ \mathrm{hr}^{-1}\ year^{-1}$. *K* is the erodibility factor expressed in $\mathrm{t.hr.(MJ.mm)^{-1}}$. *LS* is unitless and represents the topographical factor. *C* is the cover management factor, and *P* is the support practice factor. The RUSLE was implemented according to the methodological flowchart shown in **Figure 2**.



Notes: CP: “Crop-management and support practice”; LS: “Topographical factor”; K: “Soil erodibility”; R: “Rainfall erosivity”; GIS: “Geographic Information system”.

Figure 2 Methodological flowchart of the RUSLE model

2.4. R factor

The *R* factor is the power of rain to cause soil erosion by water (Cardoso *et al.* 2020). It is the mathematical expression of erosive power used to determine the average impact of rainfall and runoff on soil erosion in a specific locality and period (Farhan *et al*. 2013). We calculated R factor using two sources of data: 1. Using Google Earth Engine (GEE) to generate the mean precipitation values in our study area between 1992 and 2021 and 2. actual precipitation data provided directly from climatological stations (**Table 1**). Notably, the GEE-based *R* factor was derived using the Kriging interpolation method, whereas we used the Inverse Distance Weighted interpolation for the R factor based on climatological station data.

Table 1. Coordinates of climatological stations used in this study

Climatological station	Code	Latitude	Longitude	Period
Salvador (Ondina)	83229	-13.00583333	-38.50583333	1992-2021
Vitoria da Conquista	83344	-14.88638888	-40.80138888	1992-2021

Salinas	83441	-16.154862	-42.284921	1992-2021
Vitoria	83648	-20.31583333	-40.31694443	1992-2021
Guaratinga	83446	-16.58081	-39.783182	1992-2021
Caravelas	83498	-17.73944444	-39.25861111	1992-2021
Itamarandiba	83488	-17.85972222	-42.85277777	1992-2021

Based on a previous study carried out by Bertoni & Lombardi Neto (1999), the values of the *R* factor of both precipitation data sources were calculated using Eqs. (2 & 3).

$$R = \sum EI \qquad (2)$$

Where R is the erosivity factor.

$$EI = 67.355 \left(\frac{r^2}{P}\right)^{0.85} \qquad (3)$$

EI is the monthly mean erosion index (MJ mm ha^{-1} h year); *r* is the average monthly precipitation (mm); *P* is the average accumulated annual precipitation (mm). Based on previous studies, the calculation of the *R* factor in this study relied on daily to annual data for extrapolation in space and time (Diodato *et al.* 2017; Meusburger *et al.* 2012).

2.5. K factor

The *K* factor determines the natural susceptibility to erosion, which is particularly influenced by soil's properties, such as texture, permeability, shear strength, organic matter, and chemical composition (Efthimiou 2020). The *K* factor is determined by two key factors: the first is the soil's infiltration capacity to withstand detachment and transport by rainfall, and the second is the runoff process (Wischmeier & Mannering, 1969). Obtaining *K* factor values is challenging due to the need for long-term field experiments under natural precipitation conditions (Schick *et al.* 2014). For this reason, we utilized a soil map provided by Mapbiomas to identify soil types prior to gathering *K* factor values from the scientific literature. As these values particularly depend on soil classification (Podhrázská *et al.* 2015) and pedology maps, the *K* value for each soil type was inserted into the attribute table in the soil map layer, thus converting this vector layer into a raster type. The higher the coefficient of soil erodibility, the higher the soil loss.

2.6. LS factor

Slope length (*L*) and slope steepness (*S*) estimate the erosion effect of slope length and slope steepness, respectively (Sidi Almouctar *et al.* 2021). Specifically, slope length is the distance between the point of origin of surface runoff and the point where the slope decreases enough for deposition to begin (Mustefa *et al*. 2020), while steepness is the gradient factor (Ajibade *et al*. 2020). Thus, *LS* is the combination of the *L* factor and *S* factor used to characterize the topographical factor, which has a significant influence on soil erosion. Thus, the *LS* factor represents the influence of topography on soil erosion (Belasri & Lakhouili 2016; Ajibade *et al.* 2020). The greater the slope, the greater the runoff (Kadam *et al*. 2018). Using QGIS, the *LS* factor was calculated through the digital elevation model (DEM), particularly Copernicus Global Digital Surface Model (DSM) of a 30-meter spatial resolution, utilizing the **Eq. (4**) developed by Desmet & Govers (1996):

$$L_{i,j} = \frac{\left((A_{i,j}+D^2)^{m+1}-(A_{ij})^{m+1}\right)}{X_{i,j}^{m} * D^{m+2} * 22.13^{m}} \qquad (4)$$

$L_{i,j}$: Length of slope for cell (i,j), $A_{i,j}$: Flow accumulation area for each cell with coordinates (i,j), D: cell grid resolution of value 30 meters, m: length exponent of the *RUSLE* factor, Xij: flow direction angle (sin i,j + cos i,j). The same software can be used to calculate the Xi,j, which indicates the flow direction with reference to the North. The exponent 'm' can be calculated using the equation developed by Foster *et al*. (1977).

$$m = \frac{\beta}{1+\beta} \qquad (5)$$

β represents the quotient between in-furrow and inter-furrow erosion and be calculated employing the expression developed by Mccool et al. (1987).

$$\beta = \frac{(\frac{sin\theta}{0.0896})}{(3sin\theta^{0.8}+0.56)} \qquad (6)$$

Where θ is the slope in degrees.

The S-factor is calculated from the expression developed by Mccool et al. (1987).

$$S = \begin{cases} 10.8\ sin\theta + 0.03\ (S < 9\%) \\ 16.8\ sin\theta - 0.50\ (S \geq 9\%) \end{cases} \qquad (7)$$

2.7. Vegetation cover, the management factor (C), and Erosion control practice factor (P)

Vegetation cover and the management factor (C) are dimensionless indexes that depend on the extent to which soils are covered by vegetation, as well as the land topography and previous models of land use (Renard *et al.* 1997). Similarly, the *P* factor is a dimensionless index that accounts for control practices that reduce the erosion potential of runoff (Renard *et al.* 1997). It expresses the ratio between expected soil erosion for a given soil conservation practice for up and down tillage (Wischmeier & Smith 1978). Herein the *C* and *P* factors were lumped together (as *CP),* with values ranging from 0 to 1, where 0 corresponds to soil with a high vegetation cover and 1 means the soil is entirely exposed to erosion (Ajibade *et al*. 2020). The LULC map for 2021 were used to extract information on vegetation class, which was then searched for in the scientific literature to find the values of *CP*.

3. **Results and discussion**

3.1. R factor based on remote sensing and climatological station data

The *R* factor varied from 55,541.06 to 67,874.84 $MJ\ mm\ ha^{-1} hr^{-1}\ year^{-1}$ when using remote sensing as the source for precipitation data, whereas such variation using data from climatological stations varied from 5,807.53 to 6,096.82 $MJ\ mm\ ha^{-1} hr^{-1}\ year^{-1}$ (**Figures 3a**

&b). Studies have shown that the *R* factor is one of the most important factors impacting the average soil erosion assessments (Dabral *et al.* 2008; Ganasri & Ramesh 2016), greatly influencing soil concentration loss (Wischmeier & Smith 1978). Low rainfall regimes can lead to an average small amount of soil loss. Clearly, the *R* factor values obtained in our study varied according to the source of precipitation data applied, as shown in **Figures 3**. The *R* factor appears to be a significant parameter in assessing soil loss with RUSLE model. This result is supported by a previous study pointing out that the *R* factor is the most important component in RUSLE and is mainly responsible for soil erosion in an area (Sidi Almouctar *et al*. 2021). One of the reasons that explains the difference between these two values of the R factor in this study is because the precipitation data coming from remote sensing are coarser than those coming from climatological stations.

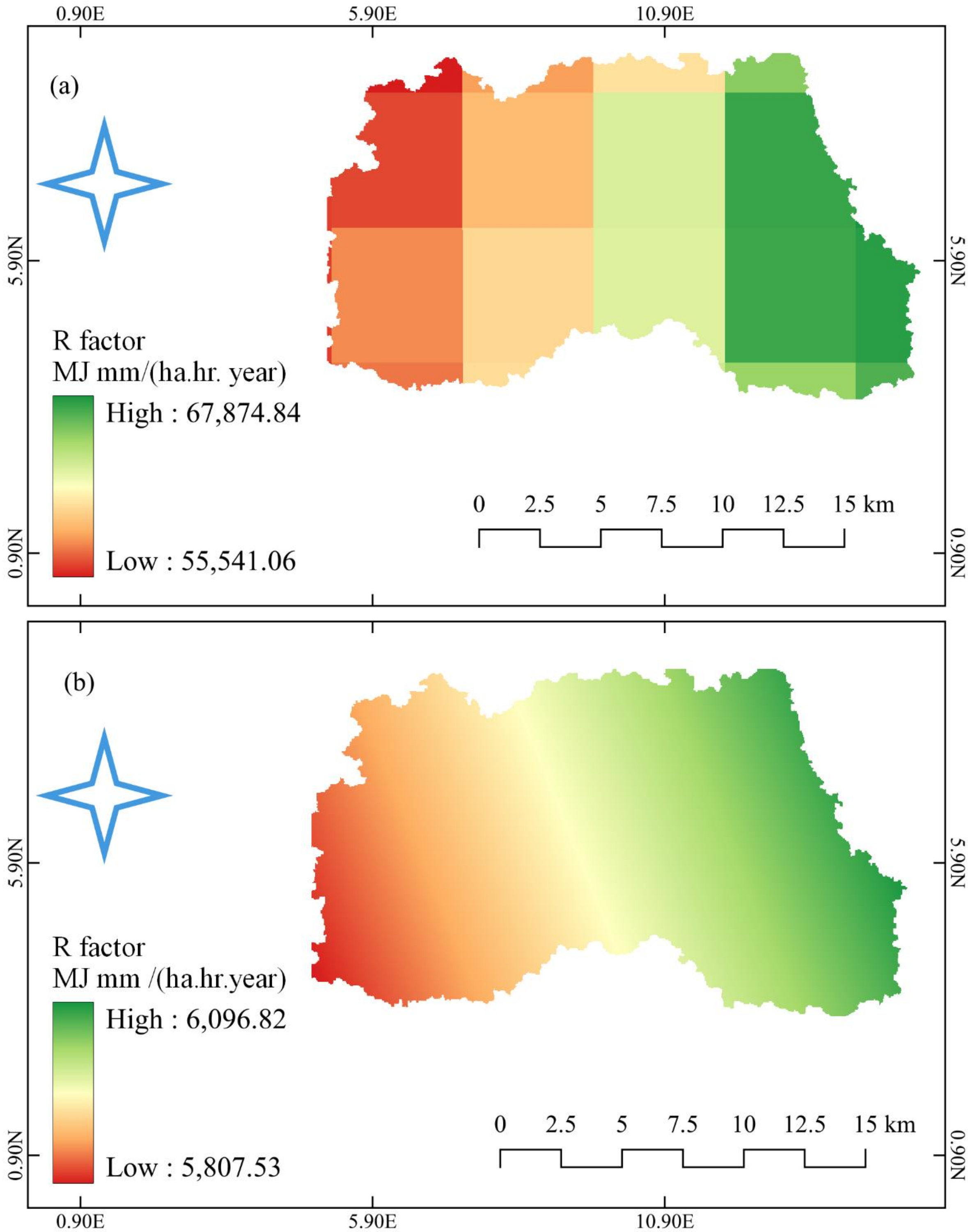


Figure 3: (a) *R* factor map based on precipitation data from remote sensing and (b) *R* factor map based on climatological stations

The results displayed in **Figure 3** indicate that the values of the R factor are influenced by the sources of precipitation. Similarly, findings from a previous study confirmed that soil erosion is driven by the *R* Factor, particularly during the rainy season (Yao *et al.* 2016). Notably, this study corroborates a previous finding that the R factor greatly influences the soil erosion potential in a GEE-based study (Sud *et al*. 2024).

3.2. K factor

The K factor values for the Cachoeira River watershed vary from 0 to 0.03, representing thiormorphic gleyssol and chromic luvisol, respectively (**Figures 4** and **Table 2**). These values are near 0, indicating that the soils in these regions demonstrate resilience to erosion, as noted by Ajibade et al. (2020). Considering these K factor values, and despite the high precipitation levels, the study area exhibits a low susceptibility to erosion.

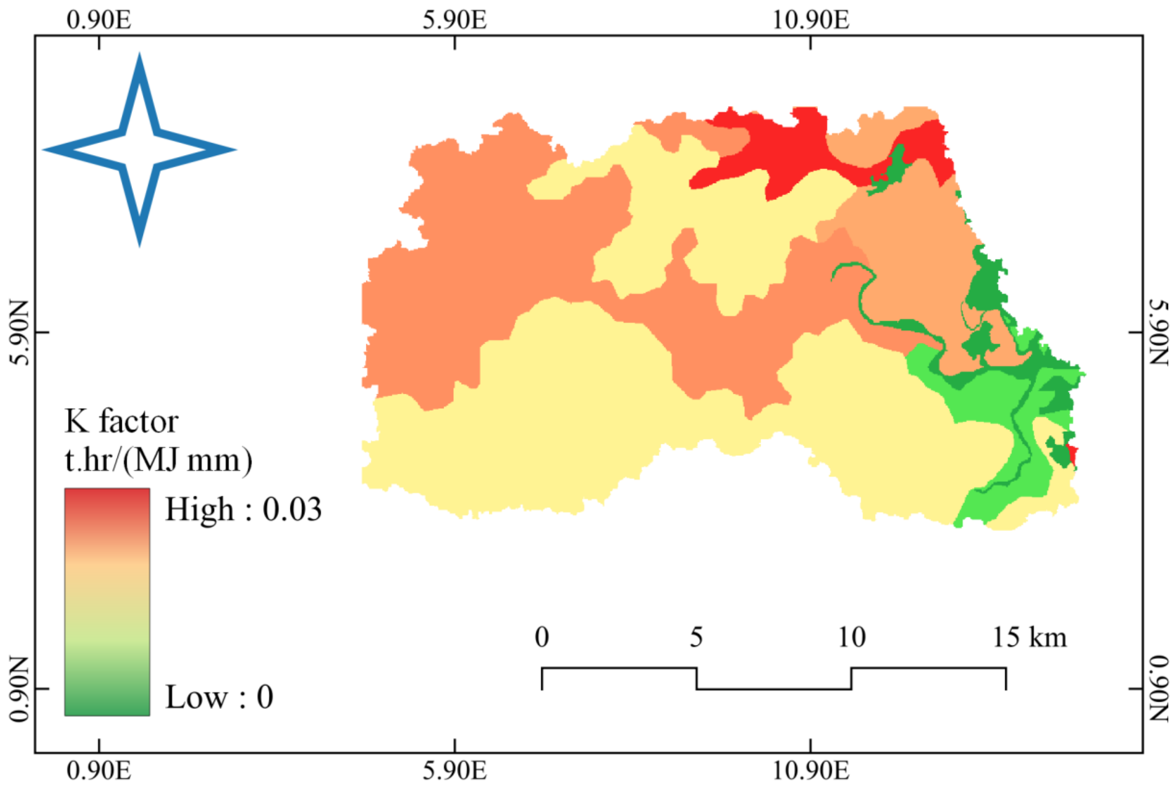


Figure 4: *K* factor map of the Cachoeira River watershed

Table 2 describes the values of the *K* factor along with different soil types in the study areas. The values of the *K* factor varied from 0.004 to 0.03, which explained that the degrees of limitation of *K* factor varied from null to moderate (Giboshi 1999).

Table 2 The values of *K* factors of the Cachoeira River watershed

Soil types	*K* factor	References
Chromic Luvisols	0.024	Embrapa[3]
Thiomorphic Gleysoil	0.004	Embrapa
Humiluvic Spodosol	0.03	Embrapa
Dystrophic Yellow Latosol	0.017	Embrapa
Dystrophic Red Latosol	0.023	Pereira & Cabral 2021

Note: Embrapa is part of the national agricultural research system in Brazil

3.3. LS factor

The LS factor shows how slope length and steepness influence the process of soil erosion (Kadam *et al*. 2018). In this study, the *LS* factor varied from 0.03 to 12.53. It is noted that higher LS values led to higher soil erosion (Fayas *et al.* 2019).

3.4. CP Factor

In the Cachoeira River watershed, *CP* values varied from 0.00004 (low) to 0.2 (high) and from 0.0001 (low) to 0.2 (high) in the real and simulated scenarios, respectively (**Figures 5a &b).**

[3] https://www.embrapa.br/busca-de-publicacoes/-/publicacao/todos

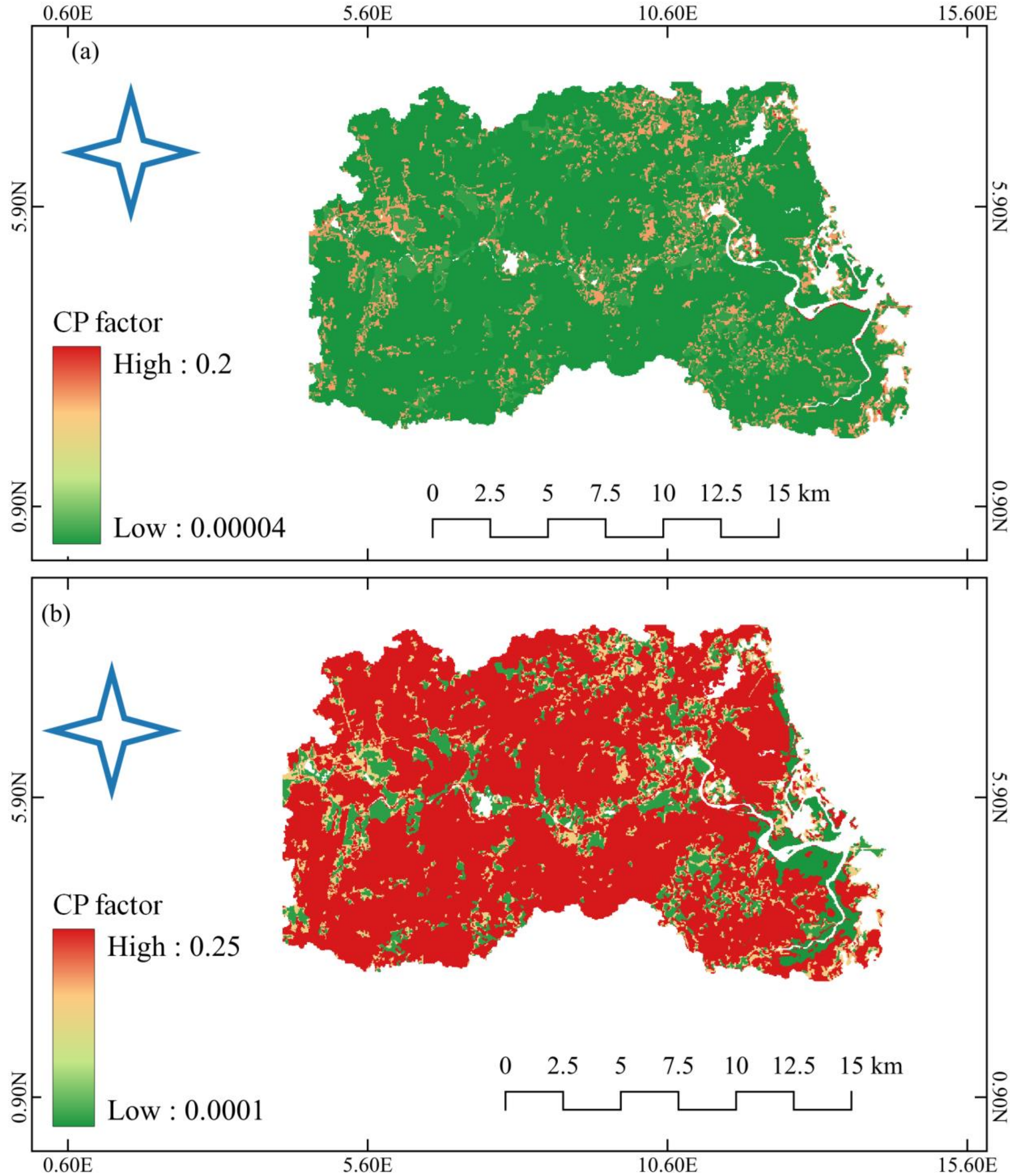


Figure 5: (a) Real *CP* factor map and (b) simulated *CP* factor map of the Cachoeira River watershed

CP values are low as they are close to zero. This means that the soil is covered with vegetation, which is important for reducing the erosion risk. This result is supported by previous studies that have highlighted how agroforestry improves physical, biological, and chemical properties (Fahad *et al*. 2022; François *et al*. 2023). The real and simulated *CP* factors had different effects on soil loss (**Figures 5 a & b**). Therefore, the utilization of cacao agroforestry systems can be a promising alternative for soil erosion reduction and soil conservation in the study areas. It is noted that the LULC has a low contribution to soil erosion in the study area and, therefore, do not represent a potential risk for erosion.

3.5. Annual soil loss

3.5.1. The case of the soil in Cachoeira River watershed using data from sensing remote

The real and simulated annual soil erosions varied from 0 to 1,865.22 t/ha/year and from 0 to 3,909.00 t/ha/year, respectively, along the Cachoeira River watershed when using precipitation data from remote sensing (**Figures 6a & b**). In other words, soil erosion in the real scenario in the Cachoeira River watershed ranged from 0 to 167.87 t/year, with a mean of 0.95 t/year and a standard deviation of 5.76 t/year, whereas in the simulated scenario, soil erosion varied from 0 to 351.81 t/year, with a mean of 16.22 t/year and a standard deviation of 30.60 t/year. Such results show the importance of forests for environmental planning and soil conservation. Our results corroborate a previous study confirming that erosion occurred where the soil had a lack of vegetation protective cover (Pimentel & Kounang 1998). In other words, natural forest, or agroforestry systems reduces soil loss caused by erosion (Lense *et al.* 2022). Accordingly, vegetation cover removal led to a very high risk of erosion (Asadi *et al*. 2017).

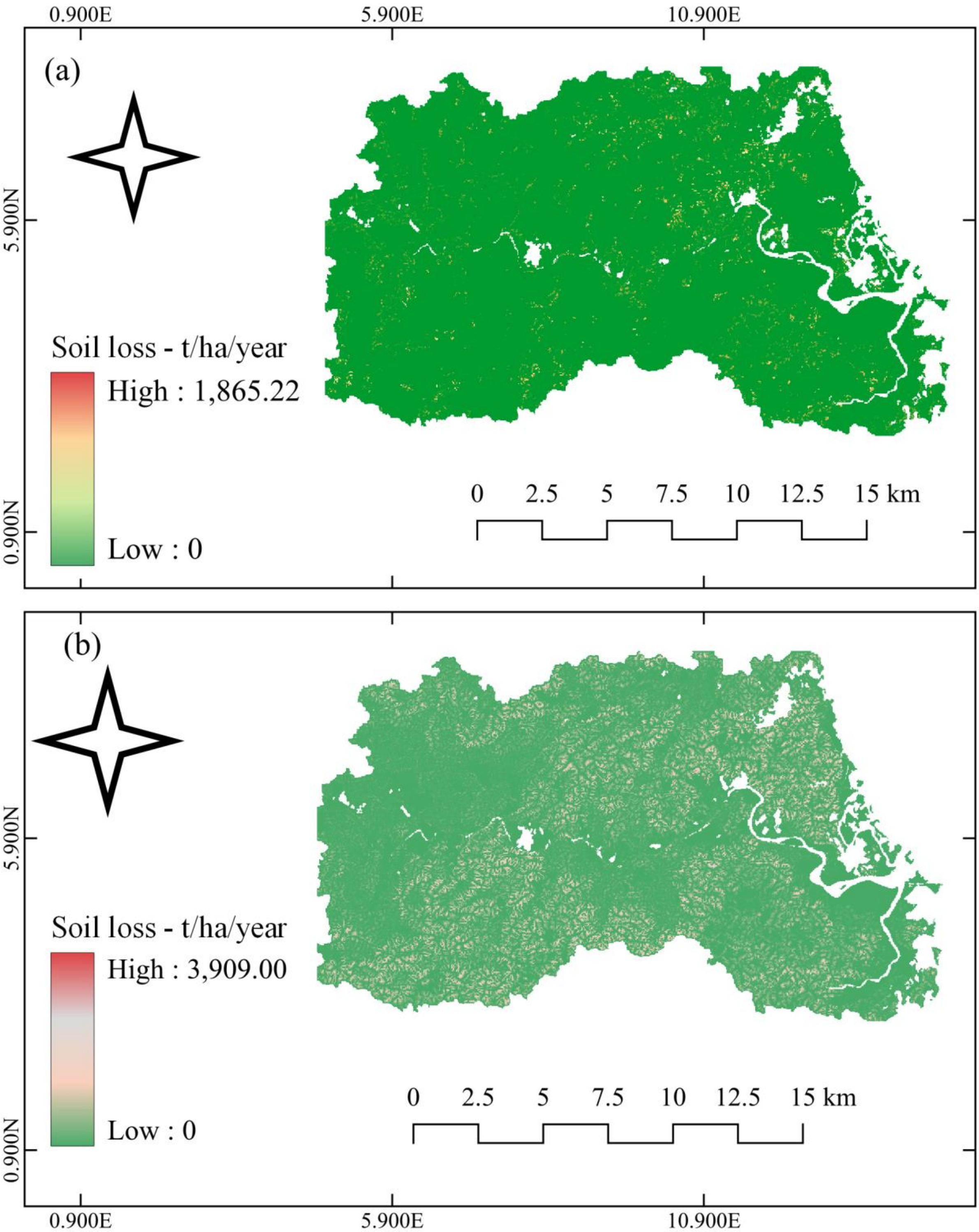


Figure 6: (a) Real soil loss map and (b) simulated soil loss based on precipitation data from remote sensing.

The highest value in t/ha/year of the simulated scenario of soil loss was 2.1 times greater than that of the actual soil loss scenario in the Cachoeira River watershed. This suggests that forests/agroforests can significantly contribute to soil protection against erosion. The results of this study show the sensitivity of *CP* values used in soil loss assessment, in which the soil erosion rate is categorized as very high. Comparably, the soil loss values of this study are greater than those of a study in Nigeria, where soil erosion ranged from 0 to > 756.60 t/ha/year (Opeyemi *et al*. 2019) and in the extreme south of Bahia, Teixeira de Freitas, with Argisol, which reported a total soil loss of 686.85 Mg ha^{-1} (Guimaraes *et al.* 2017). Another study conducted in the municipality of Teixeira de Freitas, southern Bahia, indicated that the area covered by eucalyptus had significantly less soil erosion compared to the uncovered soil area (Ferreira *et al*. 2019). As expected, simulated *CP* factors led to higher soil erosion than the real scenario. In this study, the topographic seems to do not have a great influence on soil erosion. Kadam *et al.* (2018) highlighted that the higher the slope area, the more intense the runoff and flow produced.\

3.5.2. The case of the soil in Cachoeira River watershed using data from climatological stations

When using precipitation data from climatological stations, the actual soil loss assessment varied from 0 to 177.22 t/ha/year, with a mean of 1.01 t/ha/year and a standard deviation of 6.07 t/ha/year. In contrast, the soil loss of the simulated scenario ranged from 0 to 372.52 t/ha/year, with a mean of 17.37 t/ha/year and a standard deviation of 32.69 t/ha/year (**Figures 7a &** b). The results of this study corroborate a previous study in Brazil, which indicated that the expansion of agriculture in forest areas was the major cause of soil erosion in savannah and rainforest biomes (Merten & Minella 2013). As expected, it was found that the soil loss was 2.10 times greater in the simulated scenario compared to the actual soil loss scenario when using climatological station data for *R* factor calculation utilized in the RUSLE model.

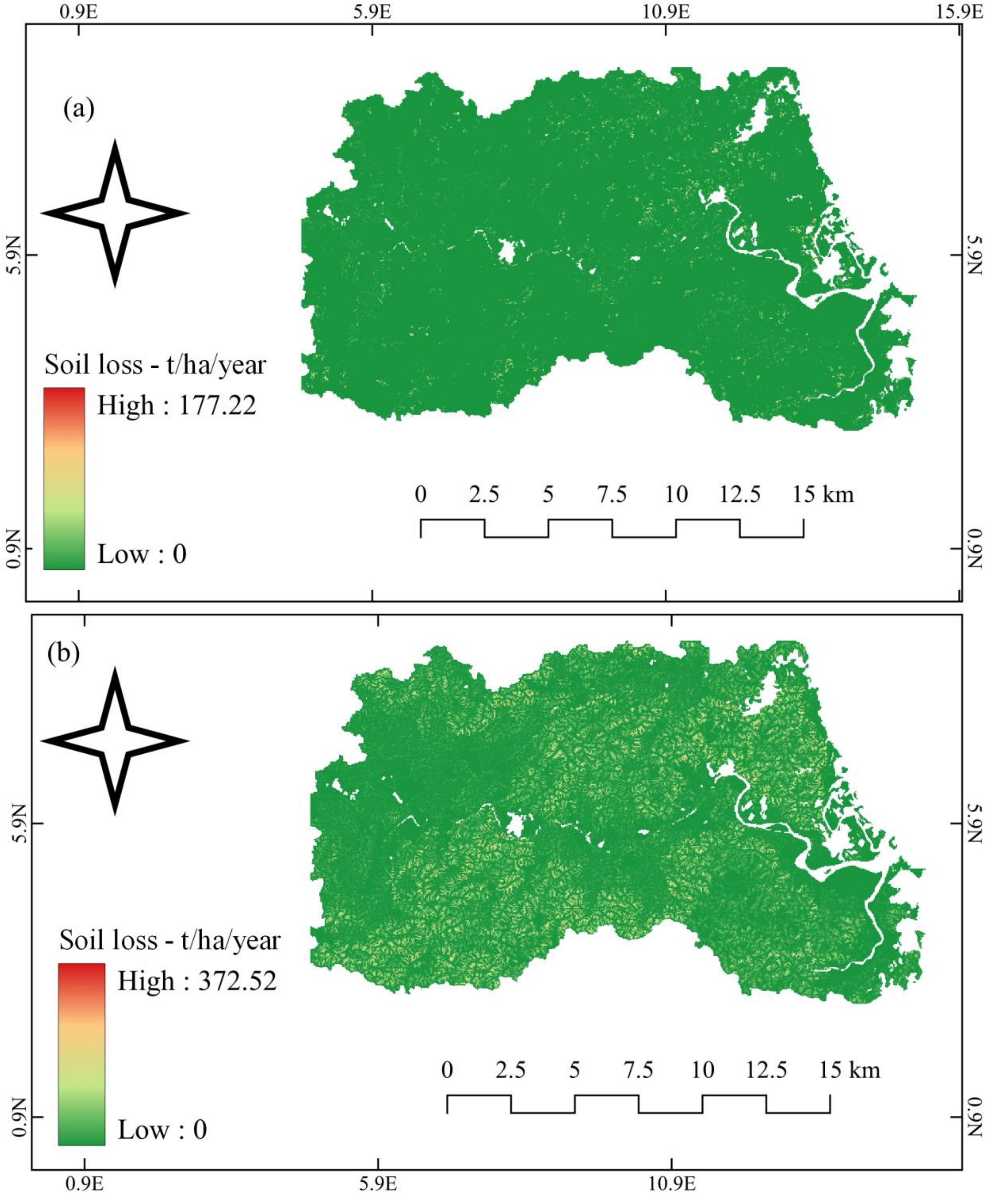


Figure 7: (a) Real soil loss map and (b) simulated soil loss based on precipitation data from climatological stations

Based on the findings of **Figures 6a & 7a**, soil erosion is influenced by the precipitation source used for *R* factor calculation. Notably, higher precipitation values resulted in greater soil erosion, as evidenced by the fact that precipitation data from remote sensing sources led to greater soil erosion compared to data from climatological stations. Importantly, it was found that the mean soil erosion calculated using data from remote sensing sources was 10.5 times greater than that assessed using climatological precipitation data. The findings of this study corroborate the results of another study, indicating that higher *R* values lead to higher soil erosion (Fayas *et al*. 2019). However, at this stage, we cannot assert that one precipitation source is better than the other. This result only indicates that the soil loss estimation is a nominal value that can depend on the source of precipitation data used for the R factor calculation.

3.5.3. *Severity of soil loss in the Cachoeira River watershed when using data from remote sensing and climatological stations*

In terms of classes of soil loss severity, only the real scenarios were considered in this study. It was discovered that the erosion risk ranged from none or slight to very high in the Cachoeira River watershed when precipitation data from remote sensing sources were used. Soil erosion is classified as very low, low, moderate, high, very high, and severe (**Table 3**), according to the classification used by İrvem *et al*. (2007). Most of the areas were classified as having no or slight soil erosion. Notably, more than 85% of this river watershed had low soil erosion. Only 1.67% of the Cachoeira River watershed experienced severe erosion (**Table 3**). Similarly, 96.06% and 0.04% of this river watershed experienced very low and very high soil erosion, respectively **(Table 3).**

Table 3 Severity range and severity class of soil loss in the Cachoeira River watershed when using remote sensing and climatological station data

Severity range	Severity class	Cachoeira River watershed/ R factor data from remote sensing		Cachoeira River watershed/R factor data from climatological stations	
		Area (ha)	Area (%)	Area (ha)	Area (%)
<=5	Very low	21,807.09	88.67	23,671.35	96.06
5−12	Low	1,385.28	5.63	342.36	1.39
12−50	Moderate	462.87	1.88	531.54	2.16
50−100	High	246.87	1.00	85.59	0.35
100 −200	Very high	305.64	1.24	10.35	0.04

> 200	Severe	412.56	1.67	---	---

The areas of high, very high, and severe vulnerability along the Cachoeira River watershed were 246.87, 305.64, and 412.56 hectares, respectively (**Table 3**). These vulnerabilities are likely associated with agricultural practices. Based on the results displayed in **Table 3**, it is noted that the precipitation source used for the *R* factor in the RUSLE model affects the severity class of soil erosion.

4. **Future research and limitations**

Soil erosion assessment is crucial for environmental management and ecosystem services. Various factors, including the *LS* factor and *CP* factor, can be sources of uncertainty in soil erosion rate estimation. A previous study indicated that errors and uncertainties in the DEM influence soil erosion rate estimation (Abd Aziz *et al*. 2012). The *LS* Factor influences soil loss over space and time (Wang *et al*. 2002) due to overlaying in a geospatial environment (Luvai *et al*. 2022). Similarly, Anjitha Krishna *et al*. (2019) underlined that the major uncertainty in soil loss assessment is derived from the *LS* factor. These challenges can be overcome by analyzing the global sensitivity of RUSLE and using different equations for *LS* factor calculation to be used in RUSLE model. Additionally, DEM from different sources, including the Advanced Spaceborne Thermal Emission and Reflection Radiometer, with multiple resolution grids, can be used to partially clarify these issues. A previous study outlined that different DEM sources and resampled grid sizes led to different soil loss estimations (Pandey *et al*. 2021). Moreover, the input model and method used for soil loss estimation can both be sources of uncertainty (Schürz *et al*. 2020). Furthermore, another limitation of this study is that *K* and *CP* values were collected from scientific literature. These factors may influence the soil erosion rate, as it is better to collect data from the study field. Importantly, the utilization of RUSLE model may overestimate soil loss and has some limitations as it does not predict sediment yield or other elements, including gullies (Croke & Nethery 2006). Further research is needed to collect data from the study field for soil erosion assessment while performing global sensitivity and uncertainty analyses of RUSLE model.

5. **Implication of this study for environmental protection**

Anthropogenic activities (e.g., deforestation) are among the drivers causing climate change, erosion, and contamination of natural resources (e.g., water and air quality). The world today is about 0.85 °C higher compared to the average temperatures from the period 1950-1980 (Sarker, 2022). In terms of environmental studies, this study shows that forests can not only be crucial for reducing soil erosion but also for mitigating climate change through carbon sequestration. In other words, the findings of this study demonstrate that forest restoration, afforestation, or tree plantations can serve as techniques for soil conservation and reclaiming damaged lands. Additionally, forests are fundamental for providing a range of services, including preserving biodiversity, wildlife habitat, and freshwater supply (Jackson *et al*. 2008). Indeed, this study demonstrates that anticipatory planning of reforestation is a plausible alternative that can greatly contribute to environmental protection and sustainable soil management.

6. **Conclusions**

Managing soil is crucial for soil conservation, environmental sustainability, sustainable development, and human well-being. This study was carried out in the Cachoeira River watershed in the Atlantic Forest in Bahia State, northeastern Brazil, to assess soil erosion using the RUSLE model. Soil erosion ranged from 0 to 167.87 t/year in the real scenario in the Cachoeira River watershed, whereas it ranged from 0 to 351.81 t/year in the simulated scenario when using remote sensing data. Conversely, when using data from climatological stations, soil losses ranged from 0 to 15.95 t/year in the real scenario and from 0 to 33.53 t/year in the simulated scenario. The novelty of this study is that soil erosion in simulated scenario was two times higher than those in real scenario. Similarly, soil erosion calculated using precipitation data from remote sensing for *R* factor calculation was 10.5 times higher than the soil loss assessed when climatological stations were used for the calculation of the *R* factor. Consequently, the sources of precipitation data used for *R* factor calculation are one of the parameters that can significantly influence soil erosion assessment. These findings highlight the importance of forests for soil conservation and the choice of precipitation source used for *R* factor calculation. In conclusion, the findings of this study may assist decision-makers in adopting a more effective approach to conservation and ecosystem services in the Cachoeira River watershed.

Author contributions

Mathurin François: Conceptualization, methodology, design, material preparation, data curation, writing-original draft, writing-review & editing, Validation.

Eduardo Mariano-Neto: Conceptualization, methodology, supervision, writing-review & editing, Validation

Deborah Faria: Conceptualization, supervision, writing-review & editing, Validation

Maria Carolina Gonçalves Pontes: Writing-review & editing, Validation

Rodrigo Nogueira de Vasconcelos: Data curation, writing-review & editing, Validation

Ulisses Costa de Oliveira: Writing-review & editing, Validation

Heraldo Peixoto da Silva: Writing-review & editing, Validation

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Data availability statement

All relevant data are included in the paper or its Supplementary Information.

Conflicts of interest statement

The authors declare there is no conflict.

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Chapter V

Assessing the global sensitivity of rusle factors: a case study of southern Bahia, Brazil

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Assessing the global sensitivity of RUSLE factors: A case study of southern Bahia, Brazil

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Abstract

Global sensitivity analysis (GSA) of the revised universal soil loss equation (RUSLE) factors is in its infancy but is crucial to rank the importance of each factor in terms of its non-linear impact on soil erosion rate. Hence, the goal of this study was to perform a GSA of each factor of RUSLE for a soil erosion assessment in southern Bahia, Brazil. To meet this goal, three non-linear topographic factor (LS factor) equations alternately implemented in RUSLE, coupled a with geographic information system (GIS) software and a variogram analysis of the response surfaces (VARS) were used. The results showed that the average soil erosion rate in the Pardo River watershed was 25.02 t/ha/yr. Besides, the GSA analysis showed that the slope angle was the most sensitive parameter, followed by the cover management factor (C-factor) and the support practices factor (P-factor) (CP factors), the specific catchment area (SCA), the sheet erosion (*m*), the erodibility factor (K factor), the rill (*n*), and the erosivity factor (R factor). In this study, ranking order mainly depends on the range of parameters *m* and *n* of the LS factor. The novelty of this work is that, contrary to other studies reporting that the uncertainty and sensitivity of LS factors lie in the resolution and Digital Elevation Model (DEM) sources, whereas we illustrated herein that is the values of parameters *m* and *n* that greatly affect this factor and thus, the soil loss estimation. The findings of this study can help hydrologists consider new parameters in the soil erosion assessment.

Keywords: Erodibility, Uncertainty analysis, Soil losss assessment, Variogram Analysis of Response Surface, Sheet and rill erosion

1. Introduction

Soil erosion is a natural process where particles are detached by raindrops splash and ensuing transport through successive deposition and reentrainment downslope (Fernández-Raga et al., 2017; Hajigholizadeh et al., 2018). A set of anthropic factors such as deforestation and timber harvesting with the intent of opening new agricultural lands, or overgrazing, and mismanagement of cropland cause soil erosion (Baskan and Dengiz, 2008). Of note, erosion is one of the global environmental problems leading to reductions in food production (Den Biggelaar et al., 2001), and ultimately economic losses (Telles et al., 2011). A recent study reported that 75 billion tons of soil are globally lost each year and cost approximately $400 billion annually (Gupta, 2019). The problem becomes more and more serious in developing countries and, hence, has been accelerating due to the substitution of forested areas by agricultural areas, which amplify soil erosion and loss of biodiversity. In some countries, deforestation started with the colonization period. For example, the Atlantic Forest biome, which is the second-largest tropical forest on the American continent (Tabarelli et al., 2005), underwent serious deforestation as early as in 1500, particularly when Portuguese settlers arrived in Brazil (Gaspar et al., 2008). However, so far, this biome still has great diversities of plant and animal species, covering 17 Brazilian states (Costa and Faria, 2008), and is home to more than 100 million inhabitants (Scarano and Ceotto, 2015). It is noteworthy that deforestation and agricultural practices are the main causes of soil erosion in the Atlantic Forest.

Previous studies have highlighted that severe soil erosion occurs in agricultural areas, particularly on higher slopes (Salem et al., 2014; Ibrahim et al., 2019), and is considered the main problem in the semi-arid region of Northeast Brazil (Carvalho Junior et al., 2017). Destruction of the vegetation cover increases surface runoff and decreases infiltration (Prosser and Williams, 1998), resulting in soil loss. Northeast Brazil is particularly vulnerable due to high precipitation intensity, land cover alteration, land use and landscape occupation (Santos and Nascimento, 2021).

The revised universal soil loss equation (RUSLE) is one of the most commonly used empirical methods to assess average field-scale soil erosion rate; accounting for rainfall erosivity (*R* factor), soil erodibility (*K* factor), slope length (*L*), and slope steepness (*S*)- often lumped together and commonly called the topographic factor (i.e., *LS* factor), cropland management systems (*C*), and erosion control practice (*P*) (*CP* factor). Each parameter of RUSLE is plagued with uncertainty. For example, various studies have highlighted that the *LS* factor is the most uncertain and sensitive one (Renard and Ferreira, 1993; Krishna et al., 2019; Khanifar and Khademalrasoul, 2020; Das et al., 2022). Like all other factors of RUSLE, any increase in *LS* results in an increase in soil loss (Biesemans et al., 2000). Accordingly, identifying the sources of uncertainty associated with the *LS* factor becomes crucial and needs to be reduced. Studies have highlighted that errors and uncertainty in the digital elevation model (DEM) can affect the estimation of the *LS* factor (Abd Aziz et al., 2012; Ren et al., 2011). Of note, uncertainty in DEMs can originate from multiple sources, including positional inaccuracy, calculation error, and interpolation error (Krishna et al., 2019). Therefore, the spatial effect of the *LS* factor needs to be quantified and modeled (Wang et al., 2002). Studies have pointed out that the slope gradient β and *S* factors decrease when the grid size increases (Lu et al., 2020) decreasing the soil loss estimation (Wu et al., 2005).

Various studies have investigated uncertainties in soil erodibility (Basaran et al., 2006; Ozcan et al., 2008; Wang et al., 2001; Torri et al., 1997). While others have assessed the sensitivity of the *R* factor using various equations (Sakhraoui and Hasbaia, 2023) and the normalized difference vegetation index based *C*-factor (Ayalew et al., 2020). A sensitivity analysis (SA) seeks to determine how uncertainty in the output of a model can be attributed to a set of uncertainty sources in model inputs (Saltelli et al., 2000; Saltelli et al., 2004). Another study documented that SAs can be used to understand the most influential input variable in a targeted equation to an output behavior as the less or non-contributing inputs (Iooss and Lemaître, 2015). Notably, SAs can either be local or global. Global AS (GSA) is the study that examines how uncertainty in a model's output, whether numerical or otherwise, can be attributed to various sources of uncertainty within the model's inputs (Saltelli, 2005). GSA considers simultaneously the values of a set of input parameters for the assessment of the output uncertainty (Homma and Saltelli, 1996), whereas a local sensitivity analysis (LSA) accounts for one-at-a-time variation of input factors. The advantage of GSA over LSA is that GSA considers the entire variation range of the input (Saltelli et al., 2000). To the best of our knowledge, no study has previously evaluated the global sensitivity of RUSLE nor rank the sensitivity of parameters of the equations used to compute the *LS* factor. Therefore, the objective behind this study was to perform a GSA of RUSLE factors as part of a soil erosion assessment in southern Bahia, Brazil.

2. Methodology

2.1. Study area and data

This study was conducted in Canavieiras, Southern Bahia (**Fig. 1**), which has a humid tropical climate (Alvares et al., 2013) with average annual rainfall, relative humidity and air temperature of 1,830 mm, 80%, and 23.5 °C, respectively (Medauar et al., 2020). A mosaic culture and several types of cocoa, such as shaded cocoa, unshaded cocoa, and organic cocoa, which are considered forests, represent the main land use land cover (LULC) of the area.

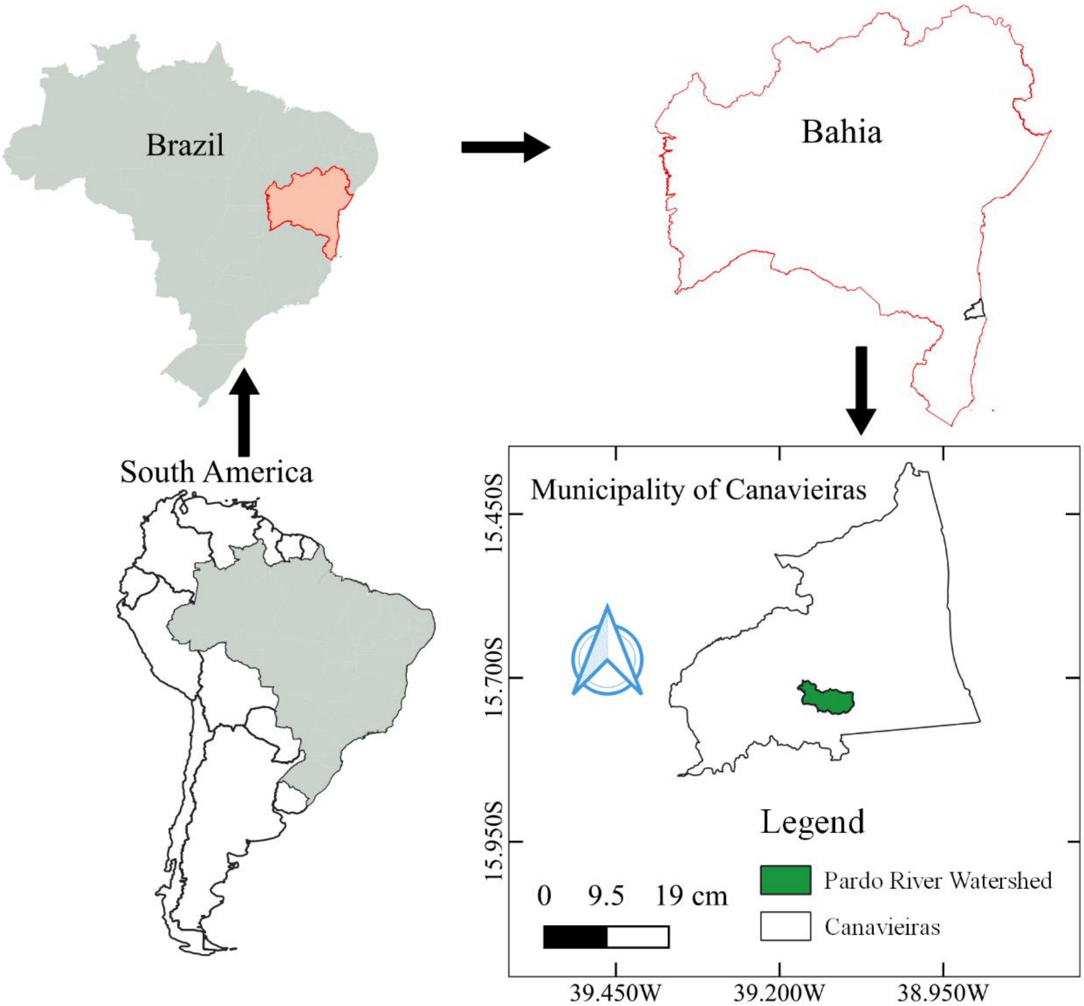


Fig. 1. Localization of 30.07 km^2 Pardo River watershed

The Pardo River crosses two Brazilian states. It rises in the state of Minas Gerais, mainly in the municipality of Rio Pardo, and completes its course in that of Canavieiras (Silva et al., 2017). The Pardo River watershed used in this study is the most degraded part of the Pardo River.

2.2. Modeling

MapBiomas was used for identifying the forest LULC. Freely available information (e.g., Mapbiomas initiative: https://mapbiomas.org/; and Landsat images of 30-m resolution) was used to assist automatic LULC classifications, including the following classes: forests (native stands and grouped agroforestry systems), pastures, forestry, open areas, urban centers, exposed soil and water bodies (rivers, ponds, ponds). Agroforestry was classified according to the existing MapBiomas methodology.

2.3. Calculation of soil loss

Soil erosion was calculated using RUSLE (Renard et al., 1997) with the support of QGIS tools. Layers of DEM, *R*, and interpolated *K* were used to model soil loss (Barbosa et al., 2015). Each raster file had the same resolution (30 m). Soil loss was estimated using **Eq. (1**). After conversion of vector files into raster ones, a raster calculator was used to calculate soil loss as follows.

$$A = R \times K \times LS \times C \times P \tag{1}$$

Where *R* represents the erosivity factor in MJ.mm.(ha.hr.year)$^{-1}$, *K* the erodibility factor expressed in t.hr.(MJ.mm)$^{-1}$, *LS* dimensionless slope length and steepness factors, *C* the crop-management factor, and *P* the support practice factor. In this study, *CP* was lumped together as the aim was to assess the soil erosion.

2.4. Rainfall erosivity (R factor)

Farhan et al. (2013) defined the *R* factor as a mathematical expression of the erosive power produced by the average rainfall and runoff causing soil erosion at a specific location. In this study, average monthly precipitations were collected from seven climatological stations (**Table 1**). The data were processed in a spreadsheet software and presented geographically before conversion in the comma-separated value (.csv) format and imported into QGIS for conversion in shapefile format. Inverse Distance Weighting (IDW) was used for the spatial interpolation of the erosivity factor.

Table 1. Location of the climatological stations used in this study (François et al., 2024)

Rainfall stations	Codes	Latitude	Longitude	Period
Salvador (Ondina)	83229	-13.00583333	-38.50583333	1992-2021
Vitoria da Conquista	83344	-14.88638888	-40.80138888	1992-2021
Salinas	83441	-16.154862	-42.284921	1992-2021
Vitoria	83648	-20.31583333	-40.31694443	1992-2021
Guaratinga	83446	-16.58081	-39.783182	1992-2021
Caravelas	83498	-17.73944444	-39.25861111	1992-2021
Itamarandiba	83488	-17.85972222	-42.85277777	1992-2021

To obtain reliable values, Cassol et al. (2008) recommended a minimum series of 23 years of precipitations. In this study, the mean precipitation was calculated for years 1992 to 2021, using Eqs. 3 & 4 as recommended by Bertoni and Lombardi Neto (1999):

$$R = \sum EI \qquad (3)$$

$$EI = 67.355 \left(\frac{r^2}{P}\right)^{0.85} \qquad (4)$$

Where *EI* is the monthly mean erosion index (MJ.Mm.ha^{-1}.hr^{-1}.yr^{-1}), the *R*-factor (MJ.mm ha^{-1}.hr^{-1}.year^{-1}), *r* the average monthly precipitation (mm), and *P* the average annual precipitation (mm).

2.5. Soil erodibility (K factor)

K factor represents the soil vulnerability to erosional agents; hence, it is limited to a set of sub-factors, including detachment and transport of soil particles (Borselli et al., 2012). Of note, soil erodibility spatially changes with LU and land form (Baskan and Dengiz, 2008). In this study, soil physical and textural properties were used to estimate erodibility, while the soil map from MapBiomas was used to identify soil types. The value of each type of soil was taken from the literature and registered in the attribute table before the conversion of this vector file into a raster file.

2.6. Topographical factor (LS Factor)

The *LS* factor is the combination of slope length (*L* factor) and slope steepness (*S* factor) and thus reflects the effect of topography on erosion (Belasri and Lakhouili, 2016). The *LS* factor is the ratio of soil loss per unit area of a field 22.1-m long and 9% slope (Wischmeier and Smith, 1978; Morgan, 1995). In this study, it was calculated through the 30-m Copernicus DEM previously filled using *fill sinks*. QGIS was used to calculate the *LS* factor according to Eqs. 5, 6 & 7 developed by Mitasova et al. (2013), Griffin et al. (1988), and Moore and Wilson (1992), respectively**.** The *LS* factor flowchart is presented in **Fig. 2**. Various studies have highlighted that the 30-m Copernicus DEM is a clearly more accurate representation of the

Earth's surface when compared to other sources, including TanDEM-X, SRTM, and NASA DEMs (Marešová et al., 2021; Ghannadi et al., 2023; González-Moradas et al., 2023).

$$LS = (m+1)(\frac{SCA}{22.13})^m \, (\frac{Sin\beta}{0.0896})^n \quad \text{where } m = (0.4-0.6) \text{ and } n = (1.0-1.3) \quad (5)$$

$$LS = (m+1)(\frac{SCA}{22.13})^m \, (\frac{Sin\beta}{0.0896})^n \quad \text{where } m = (0.2-0.6) \text{ and } n = (1.0-1.3) \quad (6)$$

$$LS = (\frac{SCA}{22.13})^m \, (\frac{Sin\beta}{0.0896})^n \quad \text{where } m = (0.4-0.56) \text{ and } n = (1.2-1.3) \quad (7)$$

Where β stands for the slope angle in degrees; m and n for sheet and rill erosion; and SCA the specific catchment area (m^2/m), that is the ratio of upstream catchment area of a contour line to its length (Yang et al., 2011). Of note, in Eq. 5 m and n vary from 0.4 to 0.6, and from 1.0 to 1.3, respectively (Mitasova et al., 1996; Mitasova et al., 2013). Whereas Griffin et al. (1988) used Eq. (6) and documented that m ranged from 0.2 to 0.6 and n between 1.0 and 1.3. While in Eq. (7), m varies from 0.4 to 0.56 and n between 1.2 and 1.3 (Moore and Wilson, 1992).

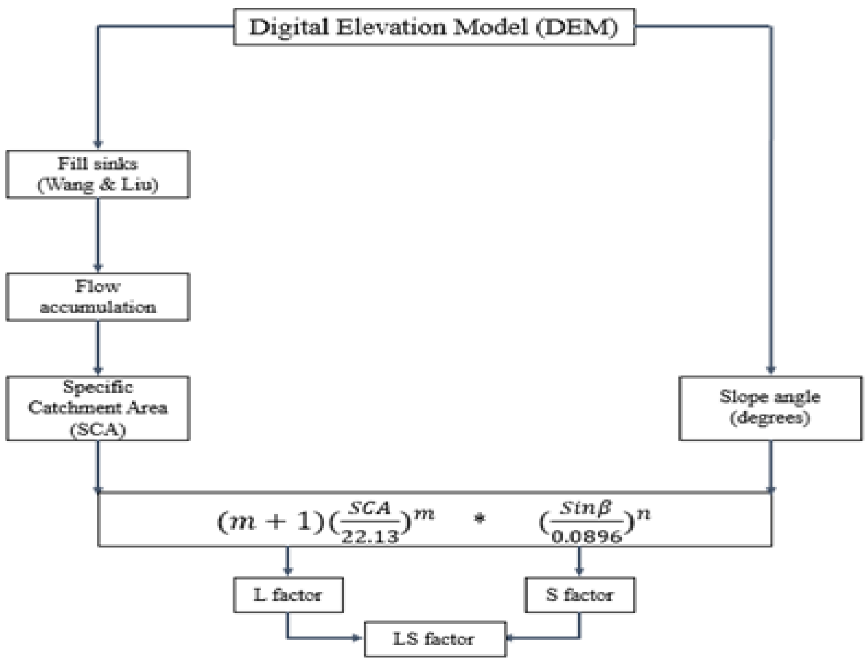


Figure 2. Flowchart of LS factor calculation via SAGA QGIS

In this study, we used the mean values of *m* and *n* (m = 0.5 and n = 1.15), and (m = 0.4 and n = 1.15) in Eqs. (5) & (6), respectively. In addition, we also used the mean of *m* (0.48) and *n* (1.25) in Eq. (7). According to Cerdan et al. (2006), *m* and *n* are linked to the removal of soil layer caused by raindrop splash. We used the filled DEM as input to calculate the flow accumulation matrix using the "*Flow accumulation*" algorithm, which was then used as input for the total catchment area (*TCA*) to extract the *SCA* using the algorithm "*Flow with and specific catchment area*" available in QGIS through hydrology tools in Saga. The multiple flow direction (MFD) method was used to execute this algorithm.

2.7. Crop-management and support practice (CP factor)

The combination of the vegetation cover and management factor (*C* factor) and the support practice (*P* factor) is often treated as a unique factor when there are not any protective management practices considered. The *C*-factor is dimensionless and depends on the degree of vegetation cover, the canopy, the topography of the terrain, and the antecedent LU practice (Renard et al., 1997) and ranges from 0 (total land cover) to 1 (no land cover) (Wischmeier and Smith, 1978). The *P*-factor is also dimensionless resulting from the ratio between expected soil losses for a given soil conservation practice and up and down tillage (Wischmeier and Smith, 1978). The CP value ranges from 0 to 1 and were taken from the literature given the 2021 LULC map. They were inserted in the attribute table before the conversion of the vector layer in a raster layer using QGIS.

3. Sensitivity associated with the RUSLE equation

The variogram analysis of the response surfaces (VARS) was used for the global sensitivity analysis, using as inputs the minimum and maximum values of each RUSLE factor. The inputs used in VARS were the parameter values of the chosen *LS* equation along those of the *CP*, *R*, and *K* factors (**Table 2**).

Table 2. The inputs used in VARS for global sensitivity analysis

Input factors	(F_1)		(F_2)		(F_3)	
	Min	Max	Min	Max	Min	Max
CP	0.0	1.0	0.0	1.0	0.0	1.0
R	5606.545	5775.109	5606.545	5775.109	5606.545	5775.109
K	0.028	0.0592	0.028	0.0592	0.028	0.0592
m	0.4	0.6	0.2	0.6	0.4	0.56
n	1.0	1.3	1.0	1.3	1.2	1.3
β	0	35.436	0	35.436	0	35.436
SCA	8.786	15535.153	8.786	15535.153	8.786	15535.153

Notes: F_1, F_2, and F_3 are inputs of RUSLE parameters used with the *LS* equations (5), (6) and (7) of Mitasova et al. (2013), Griffin et al. (1988), and Moore and Wilson (1992), respectively.

The values of the input factors *m, n,* β, and *SCA* were used to calculate the *LS* factor, where *m* and *n* were obtained from sampling from VARS, and the other two parameters were kept constant given the use of one DEM only.

3.1. Soil loss calculations based on VARS sampling

VARS has been acknowledged as a computationally efficient and reliable when compared to other GSA approaches (Bajracharya et al., 2020). This tool can generate a set of sensitivity indices, such as those based on derivative, variance, and variogram concepts (Razavi et al., 2019) and offers various advantages, including easy integration, low computational cost, and the ability to compare the indicators of Sobol (Sobol, 2001), Morris (Morris, 1991), and Integrated Variogram Across a Range of Scales (IVARS), which is a perturbation scale in the factor space, measures the change in the model response, and provides a set of sensitivities for the VARS model (Gordon et al., 2023). Admittedly, some parameters, including factor space (up to a user to define the sampling resolution) and "star-based" sampling strategy, known as Sensitivity Test Analysis Routines (STAR), should be well defined first, which provides the degree of importance of each factor, along 90% confidence intervals of the factor space. Meles et al. (2021) outlined that STAR sampling consists of vertices (STAR centers) defining subsets of the factor space from which parameters are randomly sampled across the full set of factor space. The function of STAR is to assess how variations in the input parameters can modify the output. The ranking of the input factor samples does not determine the ranking of the output factor. Indeed, we used 100 stars (resulting in a total of 4600 simulations) and a sampling resolution of 0.1 (corresponding to 10% of the parameter ranges) for the parameter space to generate the outputs. It is worth noting that an increase in the number STAR results in an increase of the number of simulations. The sampling resolution represents the distance between the locations of pairs of points in the parameter space and also ensures that all the parameter ranges are taken into account in the sampling process (Korgaonkar et al., 2020). Similarly, other studies recommended the utilization of a sampling resolution of 0.1 to avoid degradation of the "constant mean assumption" that can occur at larger scales (Razavi and Gupta, 2016a, 2016b; Razavi et al., 2019). Of note, the minimum and maximum values of each should be inserted in the factor space. Razavi et al. (2019) recommended running VARS with an extensive amount of time to deal with the problem of factor placement in terms of importance. Thus, there were 4600 maps of the *LS* factor, as illustrated in **Fig. 3**.

The soil loss calculation was carried out using two different methods. Method 1 involved applying the soil loss equation pixel by pixel. Using the *LS* results and the values of *R*, *K*, and *CP*. Method 1 was thus applied to each pixel included using the 4600 matrices already calculated for *LS* factor, resulting into the same number of maps where each one was made up of 27,799 pixels, resulting in 127,875,400 possible outcomes. Meanwhile, Method 2 added an extra step by taking into account the average value of the previous calculations, as presented in Eq. (8).

$$Pixel\ (\hat{A}) = \frac{1}{4600}\sum_{i=1}^{4600} A_i \qquad (8)$$

In other words, it aggregated the average of all the values of soil erosion in a pixel. Each pixel thus contained 4600 values for soil erosion, and in this method, the average was calculated to produce a representative map of the value of soil erosion per pixel. Examples of calculations for Methods 1 and 2 are illustrated in **Figs. 4 & 5**. The average value map is composed of pixels where the value of each pixel is the average of the 4600 possibilities, as shown **Fig. 6**. It is noted that the map of average values is a uniform map consisting of the average of all pixels.

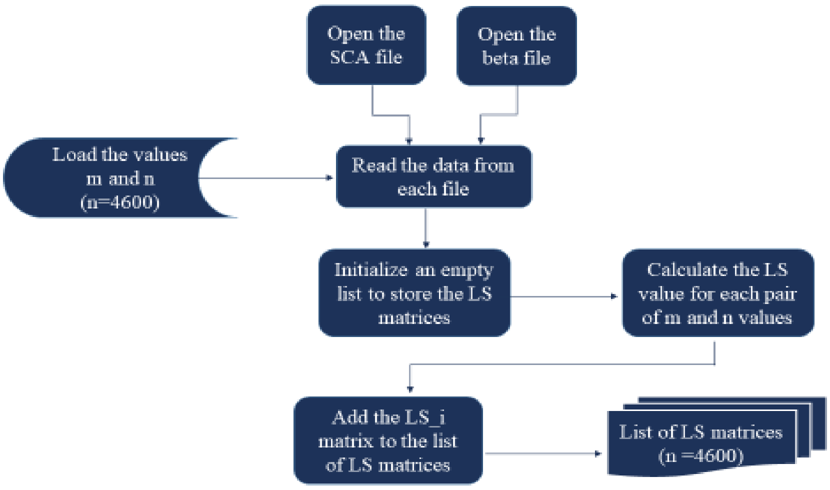


Figure 3. Example of calculation of the *LS* factor for each each pixel

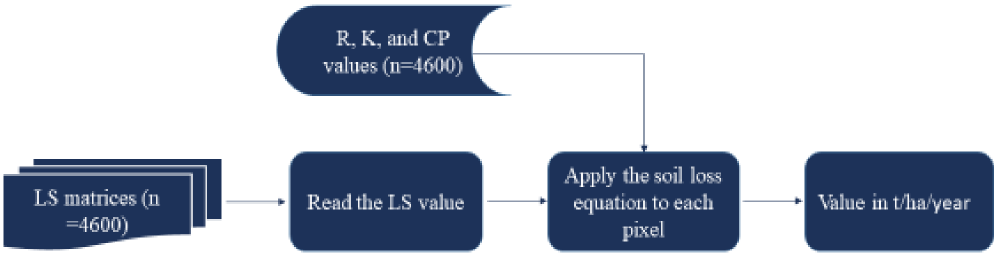


Figure 4. Method 1-calculation of the soil loss for each pixel

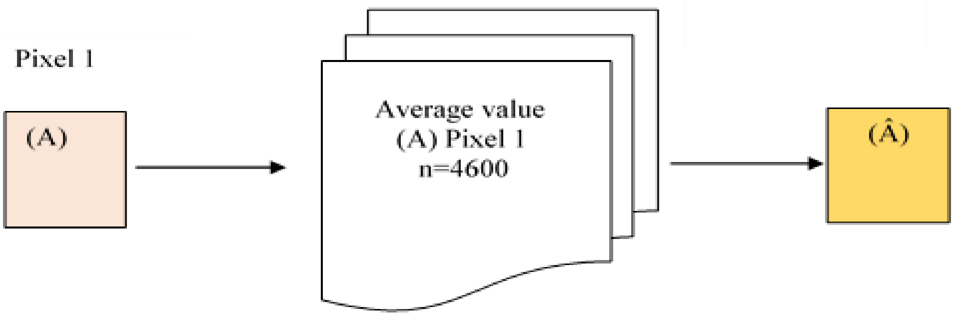


Figure 5. Method 2- calculation of 4600 soil loss maps

$\hat{A}_{(1,1)}$	$\hat{A}_{(2,1)}$
$\hat{A}_{(1,2)}$	$\hat{A}_{(2,2)}$

Figure 6. Average of 4600 possibilities of the pixels

4. Results and discussion

This section first introduces the nominal soil losses using each *LS* equation and then those resulting from the uncertainty analysis.

4.1. Nominal soil erosion losses

4.1.1. R-Factor

The *R* factor ranged between 5606.55 and 5775.11 MJ mm ha^{-1} hr^{-1} $year^{-1}$ with a mean of 5693.78 and a standard deviation of 43.66 MJ mm ha^{-1} hr^{-1} $year^{-1}$ (**Fig. 7**). In this study, the erosivity is considered average/high because the *R* value is between 5,905 and 7,357 MJ.mm $ha^{-1}.hr^{-1}.year^{-1}$ (Carvalho, 1994).

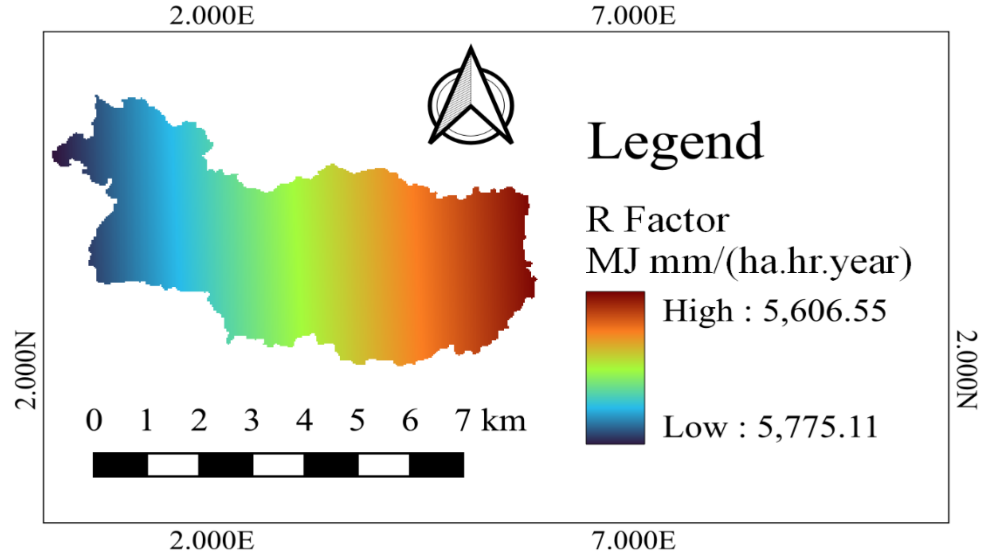


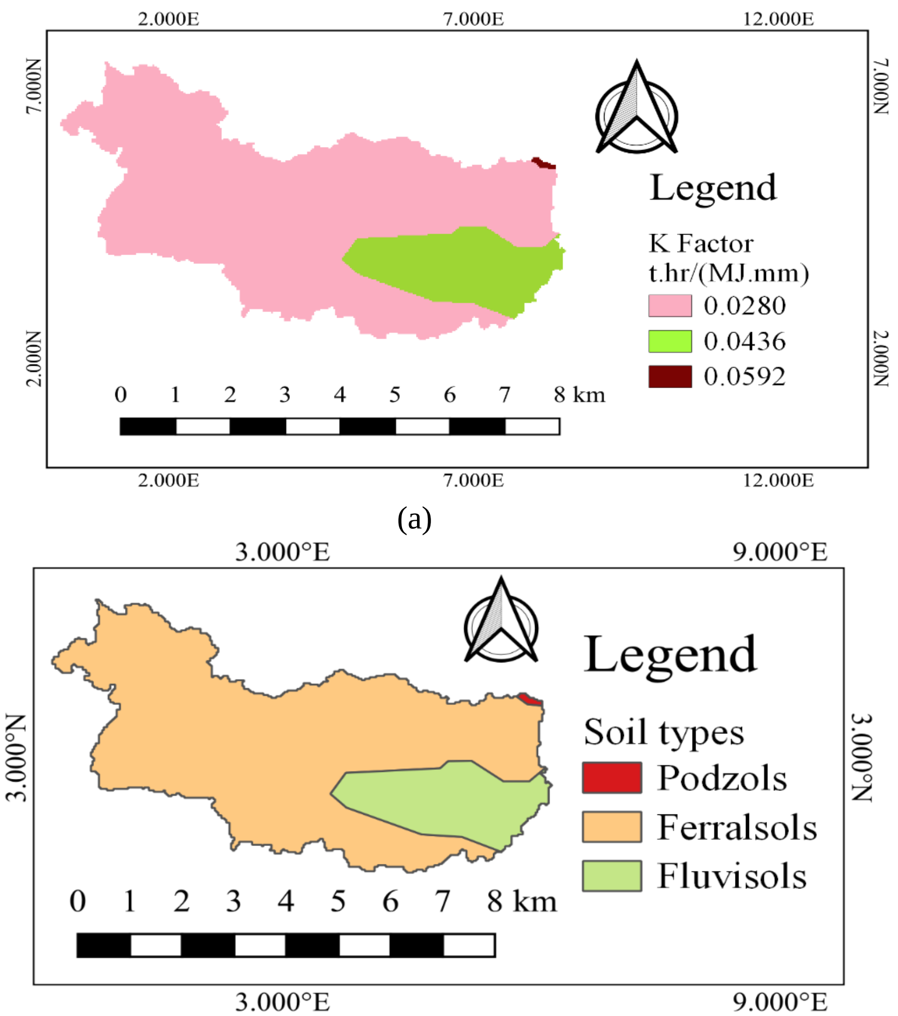
Figure 7. Erosivity map for the watershed of Pardo River

The *R* value is slightly lower to those of other studies conducted in the extreme west of the state of Bahia, particularly in the watershed of the Ondas River, where it varied from 5,870 to 6,866 MJ mm ha^{-1} hr^{-1} $year^{-1}$ (Fistarol et al., 2016) and from 6,177.36 to 9,968.21 MJ mm ha^{-1} hr^{-1} $year^{-1}$ (Nascimento and Santos, 2019)

4.1.2. K factor

There are three soil types in the study watershed: Ferralsols, Podsol, and Fluvisols with values of *K* values of 0.028, 0.046, and 0.0592 t.hr. $(MJ.mm)^{-1}$, respectively (**Figs. 8a & b**).

Based on **Table 1** and soil classification, soils in the study area were considered fairly resistent to erosion (*K* values range from 0.0132 to 0.0329), moderately sensitive to erosion (*K* values between 0.0329 and 0.0461), and fairly sensitive to erosion (*K* values between 0.0461 and 0.0593) (Bollinne and Rosseau, 1978). Not surprisingly, Ajibade et al. (2020) highlighted that soil is resilient to erosion have *K* values close to zero. Of note, the *K* value varied from one type of soil to another not only because of basic soil properties but also because of human activities (Benito et al., 2023; Huang et al., 2022).



(a)

(b)

Figure 8. (a) Soil types and (b) *K*-factor map

Another study reported that other factors, including the slope position, can influence the *K* values (Chen et al., 2023). Based on the K values, the study area has a low vulnerability to soil erosion.

4.1.3. LS Factor

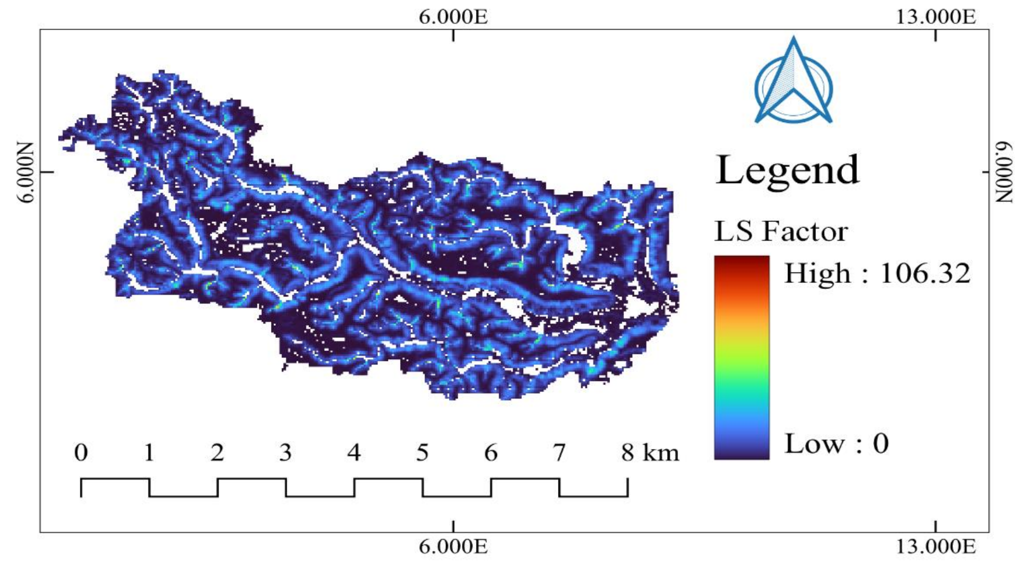
The *LS* factor is primarily a function of the *SCA* and the slope. The results showed the *LS* factor lied between 0 (latter and lower part) to 106.32 (steeper and upper part) **(Fig. 9a**), from 0 to 55.50 (**Fig. 9c**), and from 0 to 71.16 **(Fig. 9d),** with average values of 7.11, 5.09, and 5.03, respectively (**Table 3**). The largest *LS* values are found at the highest elevations, which have the highest soil erosion.

Table 3. Statistics of *LS* factors based on various equations

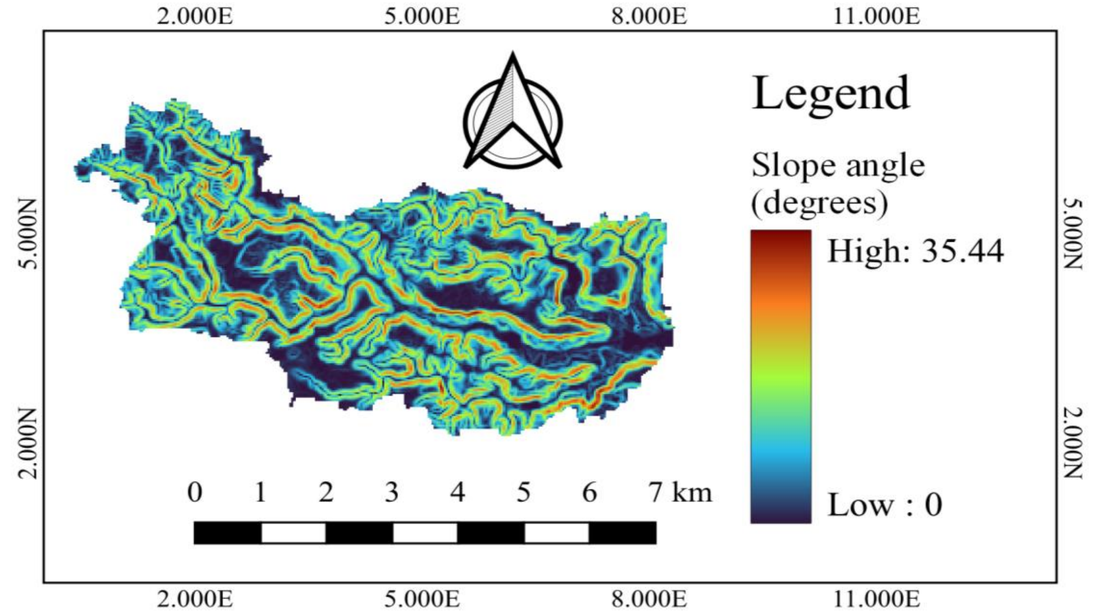
Eq. N°	LS equations	[*m*, *n*]	Min	Max	STD-Dev
(5)	$(m+1)(\frac{SCA}{22.13})^{m}(\frac{Sin\beta}{0.0896})^{n}$	[0.5, 1.15]	0.0	106.32	7.07
(6)	$(m+1)(\frac{SCA}{22.13})^{m}(\frac{Sin\beta}{0.0896})^{n}$	[0.4, 1.15]	0.0	55.50	5.03
(7)	$(\frac{SCA}{22.13})^{m}(\frac{Sin\beta}{0.0896})^{n}$	[0.48, 1.25]	0.0	71.16	5.05

Notes: Min: "minimum"; Max: "maximum", STD-Dev: "standard deviation"

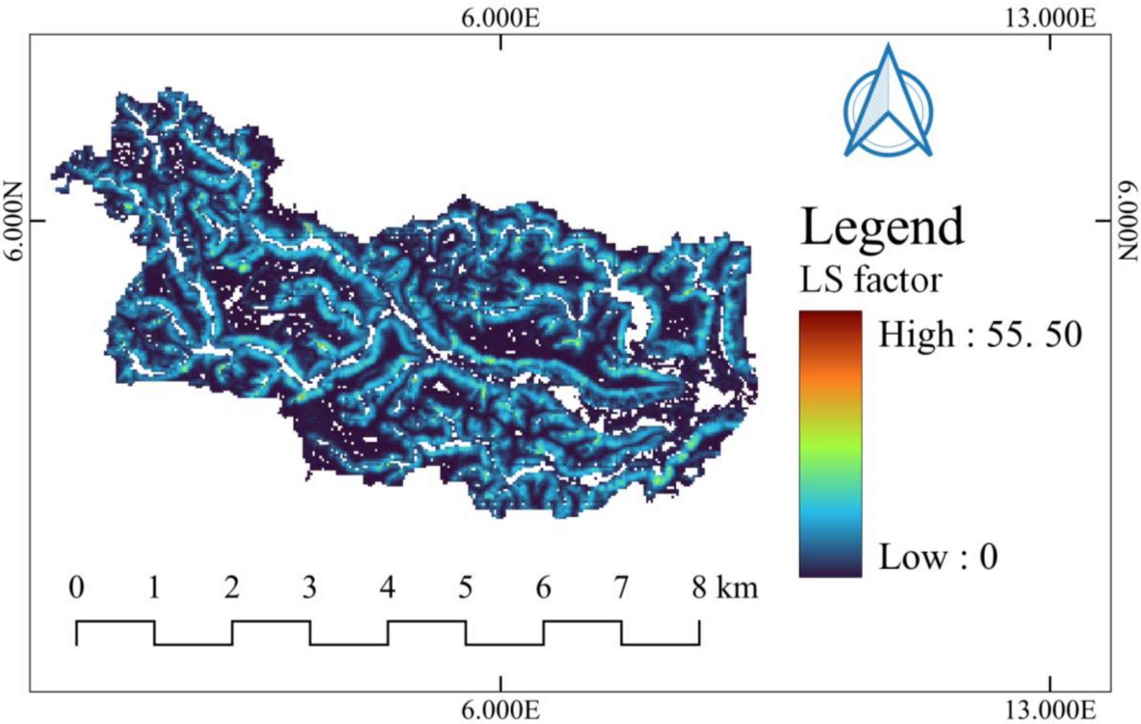
The slope angle varied from negligible to 35.44 degrees (**Fig. 9b**). Importantly, there is a direct relationship between the *LS* factor and the equations used, as well as the soil erosion.



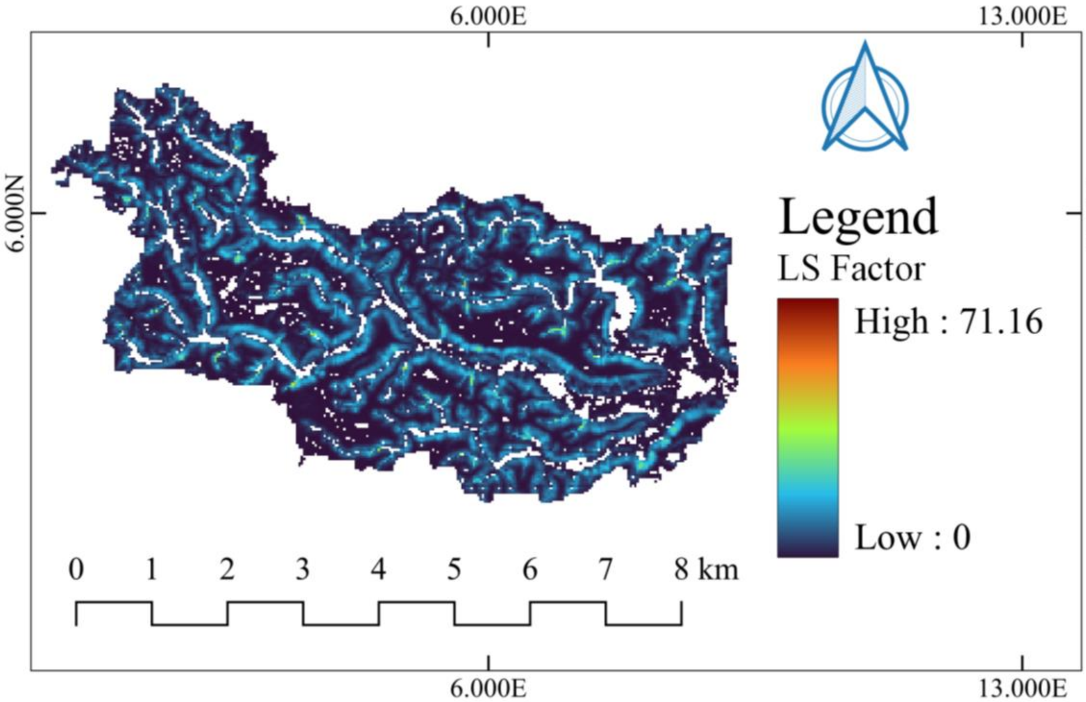
(a)


2.000E
5.000E
8.000E
11.000E
Legend
Slope angle
(degrees)
High: 35.44
Low : 0
5.000N
2.000N
0 1 2 3 4 5 6 7 km

(b)


6.000E
13.000E
6.000N
Legend
LS factor
High : 55. 50
Low : 0
0 1 2 3 4 5 6 7 8 km

(c)



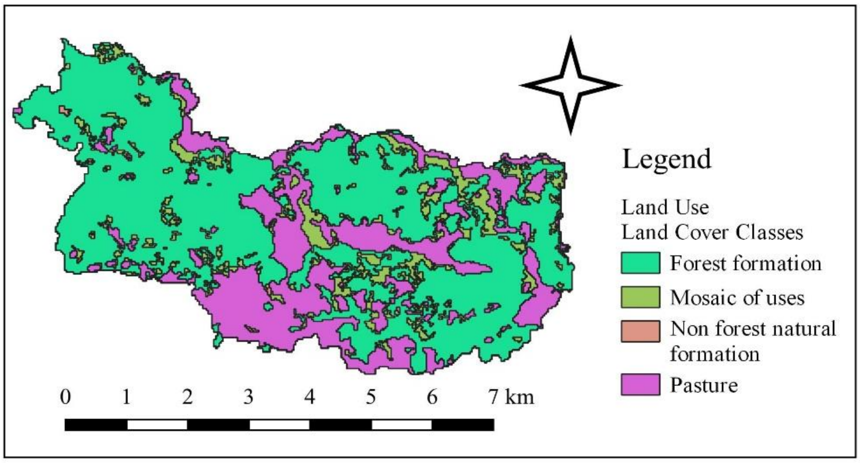
(d)

Figure 9. (a, c & d) *LS* factor maps and (b) slope angle for the Pardo river watershed

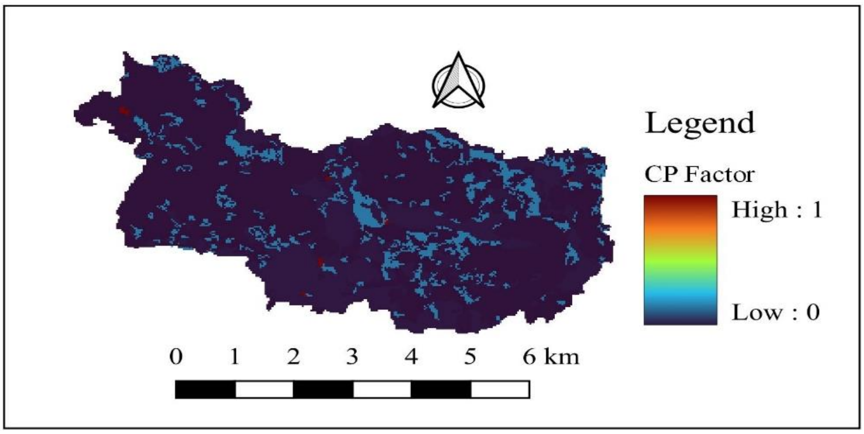
LS values shown in **Figs. 9a, c, & d** were calculated via the utilization of **Eqs. ((5), (6), & (7))**, employing mean values of and *m* and *n* of 0.5 and 1.15, 0.4 and 1.15, and 0.4 and 1.25, respectively. As mentioned before, a larger *LS*-factor value increases potential soil erosion as an increase in slope length accelerates flow velocity (Rahman and Shakir, 2023; Khan et al., 2023). Thus, Das et al. (2022) argued that *LS* is the most important factor for assessing soil erosion when using RUSLE.

4.1.4. CP Factor

The *CP* values varied from 0 to 1 (**Fig. 10b**). The highest value (1) corresponded to non-natural forest formation (e.g., degraded areas), and the lowest value (0) was water (**Figs. 10a &b**). The *CP* values close to zero indicate that the cropping-management and support practice factors provide strong resilience to soil erosion compared to values close to one, which represent exposed or degraded areas.



(a)



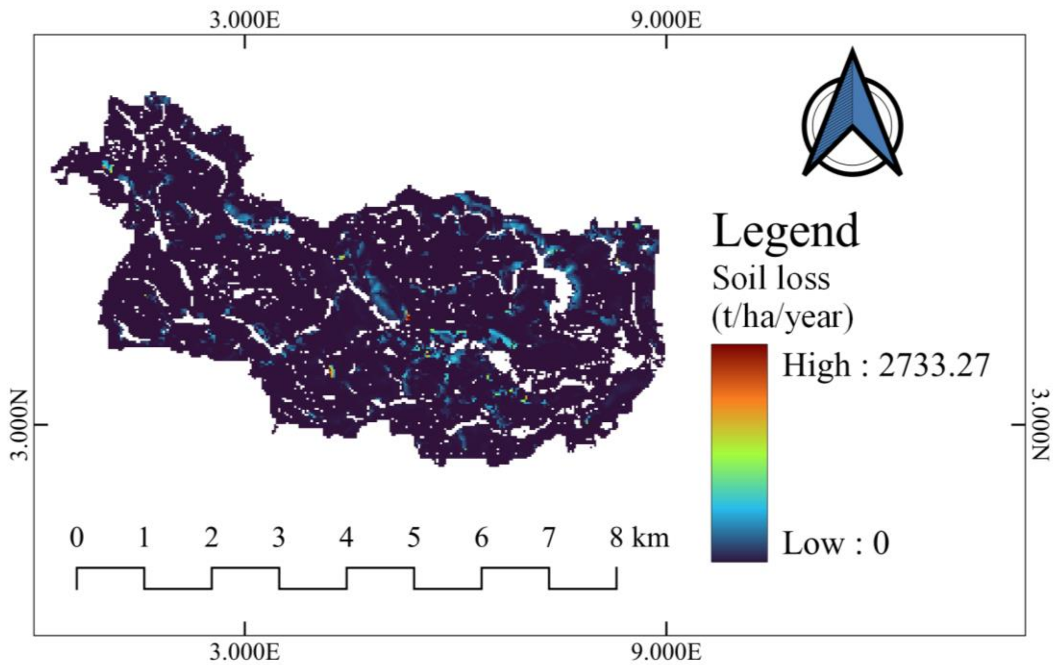
(b)

Figure 10. (a) LULC map and (b) *CP* factor map

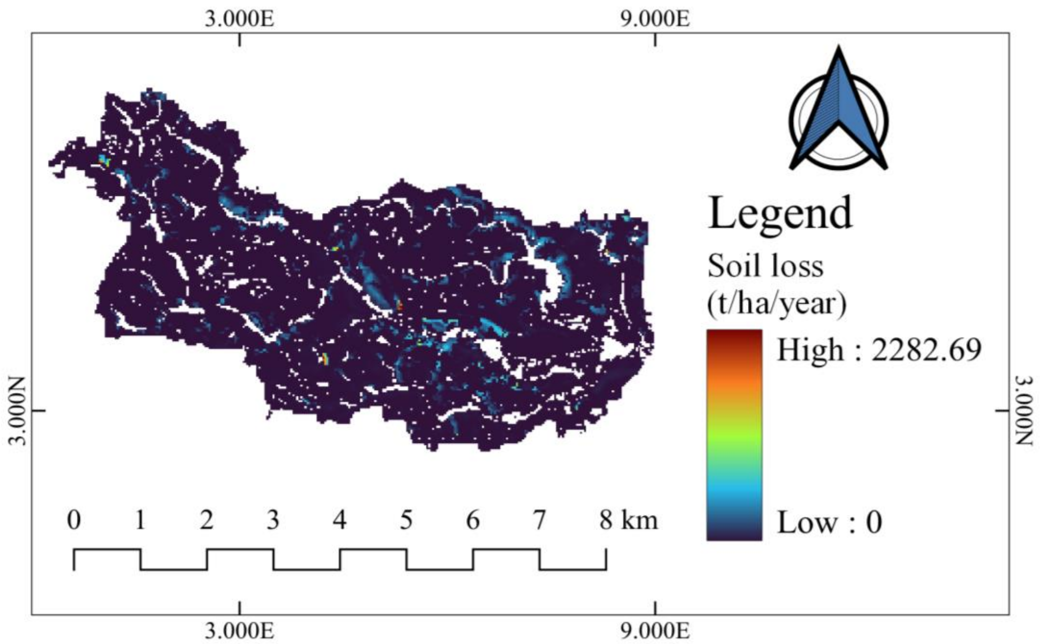
4.1.5. Potential Soil loss

The potential nominal soil loss was assessed via the raster calculator of QGIS. The soil erosion predicted when using the *LS* equation based on Eq. (5) ranged from 0 to 2733.27 t/ha/year (**Fig. 11**), with an average rate of 29.77 t/ha/year and a standard deviation of 117.52 t/ha/yr, whereas those based on Eq. (6) and Eq. (7) varied from 0 to 2282.69 and between 0 and 2103.27 t/ha/year, with averages of 23.76 and 21.52 t/ha/year, respectively (**Figs. 11a, b &c, Table 4**). These results indicate that soil losses vary along with the *LS* equation used. The soil erosion in this watershed was lower compared with that of another study carried out in the same region, the extreme western region of Bahia, which varied from 0 to 1846.39 t/year (Ferreira et al., 2019). The average soil loss derived from these three equations was estimated at 25.02 t/ha/year. The mean soil loss in this watershed was higher than the average of that of the Rio

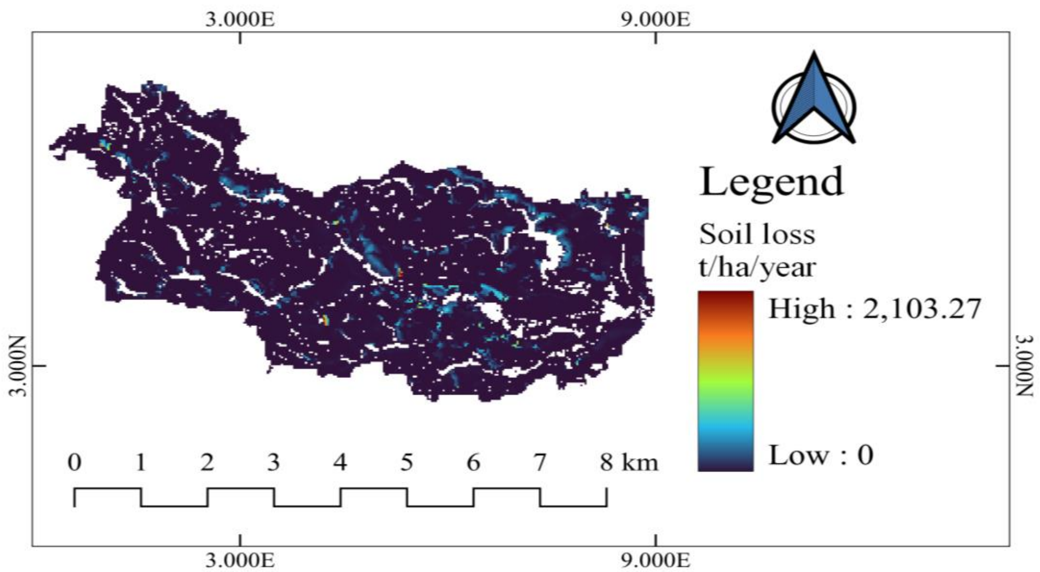
Ondas watershed, estimated at 13.36 t/ha/year, which is located in the extreme western part of the state of Bahia (Fistarol and Santos, 2020) that of another study conducted in southern Bahia, the Rio da Dona watershed of, which had an average of 21.19 t/ha/year (Trindade, 2018). These findings characterize watersheds with different degradation levels. Another aspect that may cause the difference is the equations that were used for the calculation of the *LS* factor in those studies. Indeed, a previous study highlighted that the *LS* equations used may have led to overestimated soil losses compared with others (Michalopoulou et al., 2022). In terms of severity, and according to the classification proposed by İrvem et al. (2007), soil erosion can be classified as none or slight, moderate, high, and very high (**Table 4**).



(a)



(b)



(c)

Figure 11. Soil erosion maps calculated using LS equations: (a) (5), (b) (6) et (c) (7).

Table 4. Estimation of soil erosion based on several equations

LS	RUSLE	$[m, n]$	Min	Max	Mean	STD-Dev
(5)	$R * K * CP * (m+1)(\frac{SCA}{22.13})^m (\frac{Sin\beta}{0.0896})^n$	[0.5, 1.15]	0.0	2733.27	29.77	117.52
(6)	$R * K * CP * (m+1)(\frac{SCA}{22.13})^m (\frac{Sin\beta}{0.0896})^n$	[0.4, 1.15]	0.0	2282.69	23.76	91.94
(7)	$R * K * CP * (\frac{SCA}{22.13})^m (\frac{Sin\beta}{0.0896})^n$	[0.48, 1.25]	0.0	2103.27	21.52	85.67

Notes: Min: “minimum”; Max: “maximum”, STD-Dev: “standard deviation”

The results displayed in **Table 4** showed that the soil erosion calculated using LS equation (5) led to the highest value (2733.27 t/ha/year) compared with the other two LS equations, whose maximum annual soil erosion rates were 2282.69 and 2103.27 t/ha/year, respectively. Importantly, most of the study area is considered protected, as most of the area had a soil loss considered low (**Table** 5).

Table 5. Severity range and severity class of soil loss in Pardo River watershed

Severity range t.(ha.year)$^{-1}$	Severity classes	using LS (5)		Using LS (6)		Using LS(7)	
		Area (km^2)	Area (%)	Area (km^2)	Area (%)	Area (km^2)	Area (%)
<5	Vey low	19.18	76.69	19.36	77.44	19.70	78.80
5−12	Low	1.36	5.44	1.59	6.36	1.58	6.32
12−50	Moderate	2.22	8.88	1.90	7.6	1.67	6.68
50−100	High	0.37	1.48	0.35	1.4	0.41	1.64
100−200	Very high	0.56	2.24	0.66	2,64	0.69	2.76
>200	Severe	1.32	5.27	1.14	4.56	0.96	3.84

Remarkably, depending on the equation used, the severe erosion class in this area varied from 3.84 to 5.27 % (**Table 5**). Similarly, between 76.69 and 78.80% of this watershed area has a soil loss considered very low, as compared with 68.58% of the Rio de Janeiro watershed, located in the extreme west of Bahia (Nascimento and Santos, 2019). Based on Tables 4 & 5 and **Figs. 11 (a, c &d)**, it is obvious that the soil erosion is only a nominal value that depends exclusively on each factor of RUSLE. These findings may also explain the uncertainties existing in the soil erosion calculation.

4.2. Uncertainty and global sensitivity of RUSLE parameters

The global sensitivity of the RUSLE factors using VARS showed the importance of each one of them and intrinsically of the parameters of the *LS* factor (β, m, n, SCA) (**Figs. 12 & 13**) and all the possible soil erosion equations used ((**Figs. 13(a), (b)&(c)**). The GSA showed that the slope angle (β) was the most sensitive parameter. This result is in line with other studies in the literature indicating that slope steepness is more sensitive than slope length to the *LS* factor used in the assessment of soil erosion (Wang et al., 2002). This study demonstrated that *m* and *n* affect the value of *LS*. Evidently, the sensitivity ranking of *LS* parameters is as follows: slope angle (β also referred as *b* in the figures), *SCA*, *n*, and *m*. In terms of GSA, the parameters of RUSLE had the same order of importance when using equations (5) and (7) (**Figs. 12 & 13**). One of the reasons that may explain that is because the value of *m* in both equations is approximately equal (0.5, and 0.48). Conversely, the GSA of RUSLE parameters using (6) had a different ranking (β, *m*, *CP*, *SCA*, *K*, *n*, and *R*) than when the other two. It is noted that, independently of the equation used, we found that the *R* factor is the least influential parameter of RUSLE.

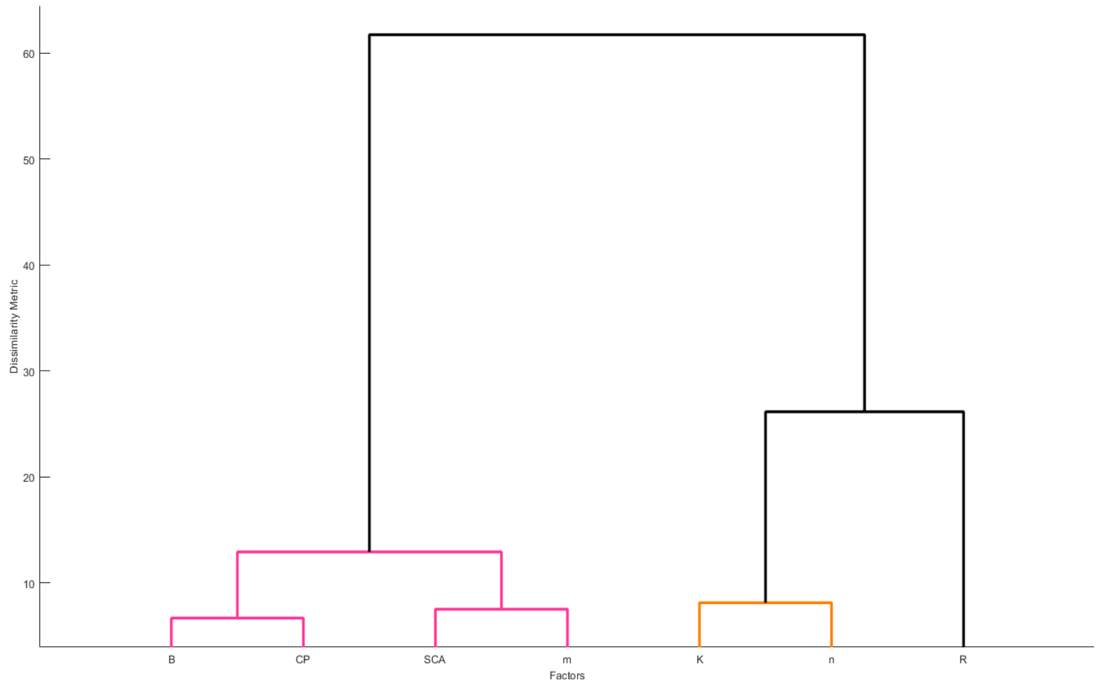


Figure 12. Dendogram of factors generated by VARS

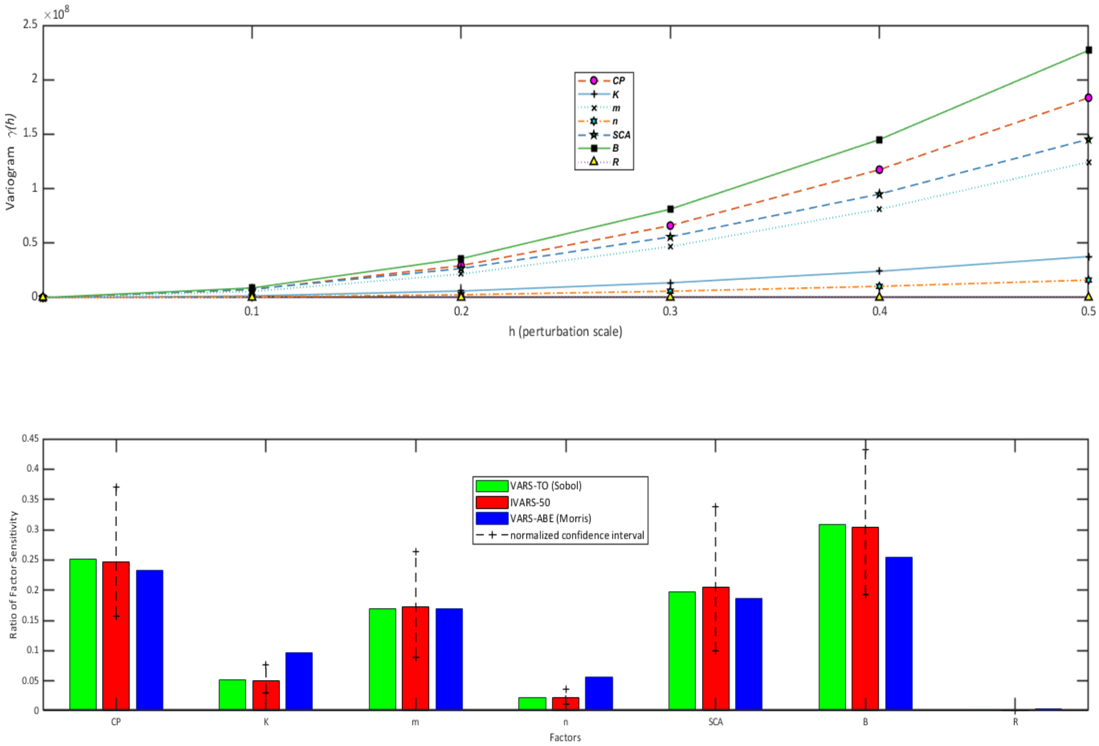


Figure 13. Visualization of directional variograms

Moreover, the GSA showed the importance order of the other parameters as follows: *CP*, *K*, and *R* (**Figs. 12 & 13).** Importantly, the discrepancies between the minimum and maximum values of each factor can be one of the reasons for the importance of the parameter order. Notably, **Fig. 13** shows the sensitivity through a set of perturbations of scales and bar charts, indicating the importance of the factors based on the derivative, variance, and covariogram-based approaches. These values vary along with *m* and *n*, as the equations used for *LS* calculations were all non linear. These results provide an uncertainty range for the potential soil loss.

The calculations of *LS* values based on VARS sampling are described in **Fig. 14**. The sample size of each boxplot corresponds to the total number of pixels available for the calculation.

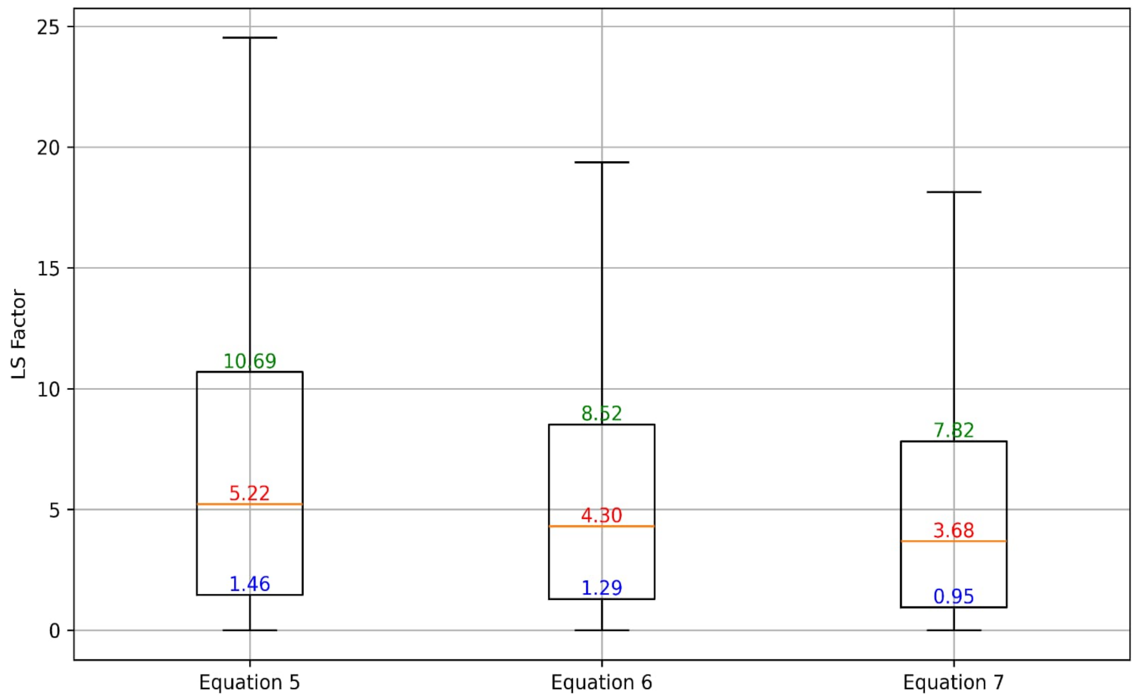


Figure 14. Comparison of *LS* factors of the **Eqs. 5, 6 & 7** based on VARS sampling. The data sample includes *n* = 127,875,400. Blue, red, and green values represent the second quartile, median, and third quartile in the boxplot.

Each box represents the interquartile range (IQR), with the median indicated by a horizontal line. It is noted that the whiskers extend up to 1.5 times the IQR above and below the box, and individual values, or 'outliers,' have been eliminated for easier visualization. Taking into account all possible combinations of *LS*, *CP*, *K*, and *R* values specified in **Table 2**. The value of the soil loss was individually calculated for each pixel. The boxplot illustrates the distribution of all the results obtained for the 27,799 pixels contained in the 4,600 maps (matrixes). The results shown in **Fig. 15** correspond to those obtained using LS Eq. (5). The results of Method 2 showed that the average value of all pixels within the watershed boundaries ranged from 0.04 to 3997.54 t/ha/year (**Fig. 16**). This result is slightly lower compared to that of a study conducted in the Indaiá watershed stream in Mato Grosso do Sul, Brazil, where soil erosion ranged from 0 to 4082.16 t/ha/year (Da Cunha et al., 2017). Notably, the results of Method 2 were calculated using LS **Eq. (5)** (see **Table 3**). To better illustrate the distribution of soil erosion values, **Fig. 17a** presents the boxplot of the average values without considering outliers, whereas **Fig. 17b** shows the second and third quartiles of the total distribution of mean values of (Â).

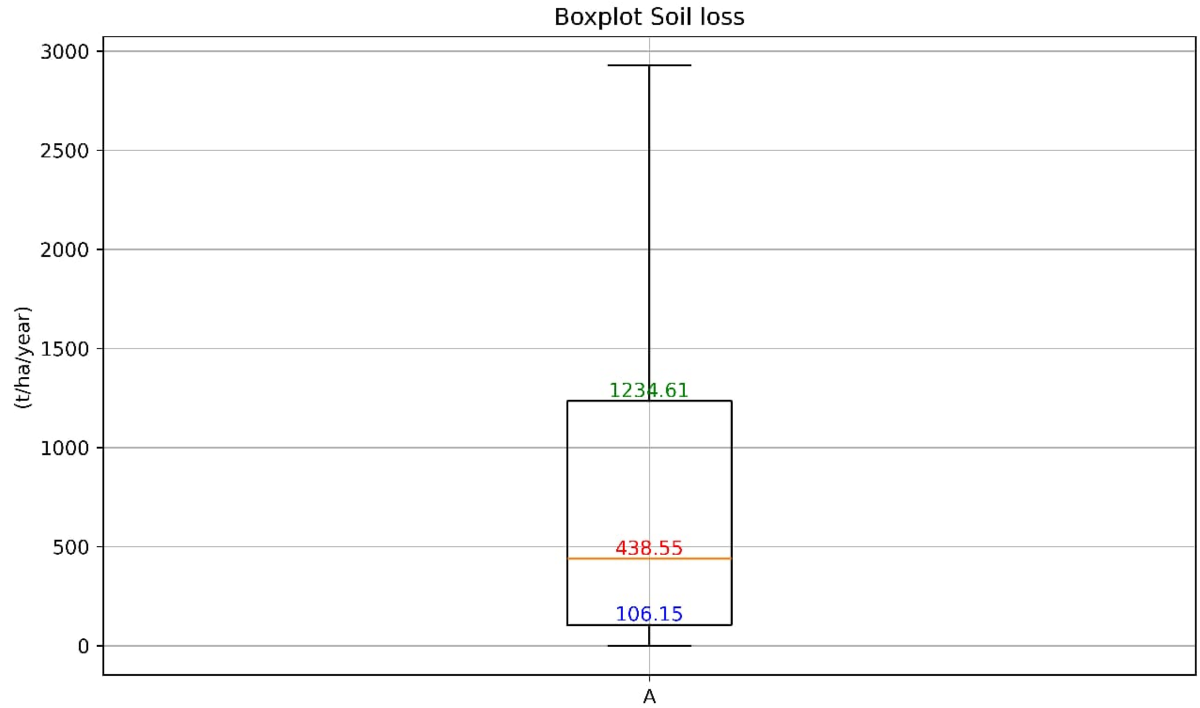


Figure 15. Method 1 - Potential soil erosion calculation results for all 4600 *LS* maps using Eq. (5) (n = 127,875,400). Blue, red, and green values indicate the second quartile, median, and third quartile in the boxplot.

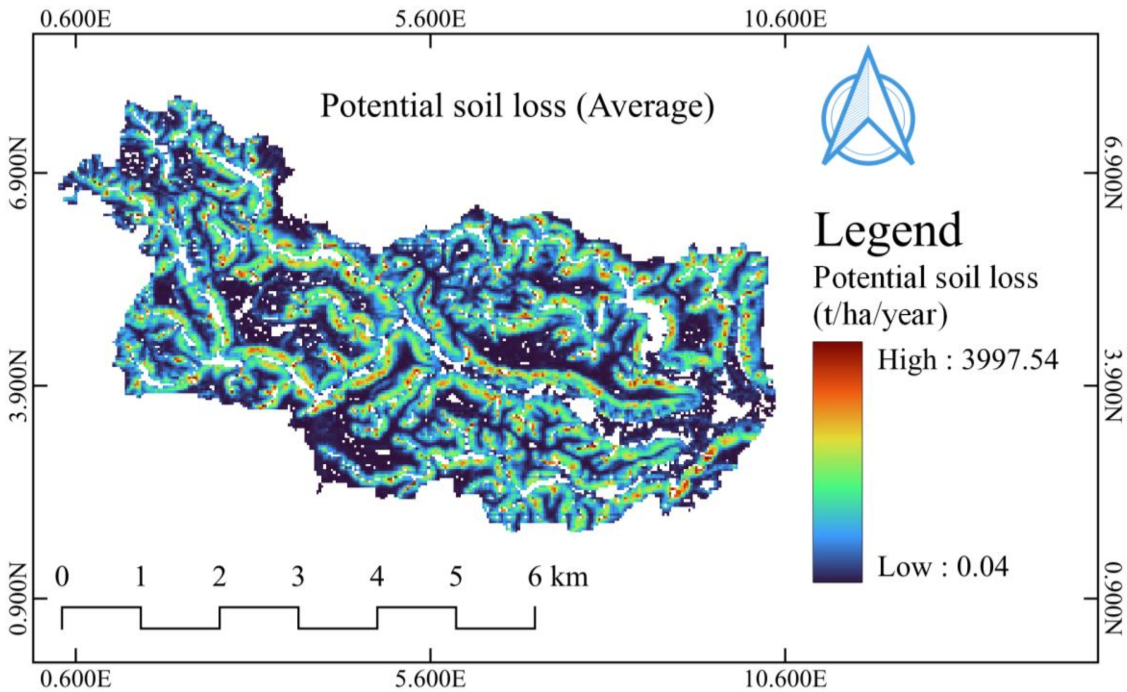
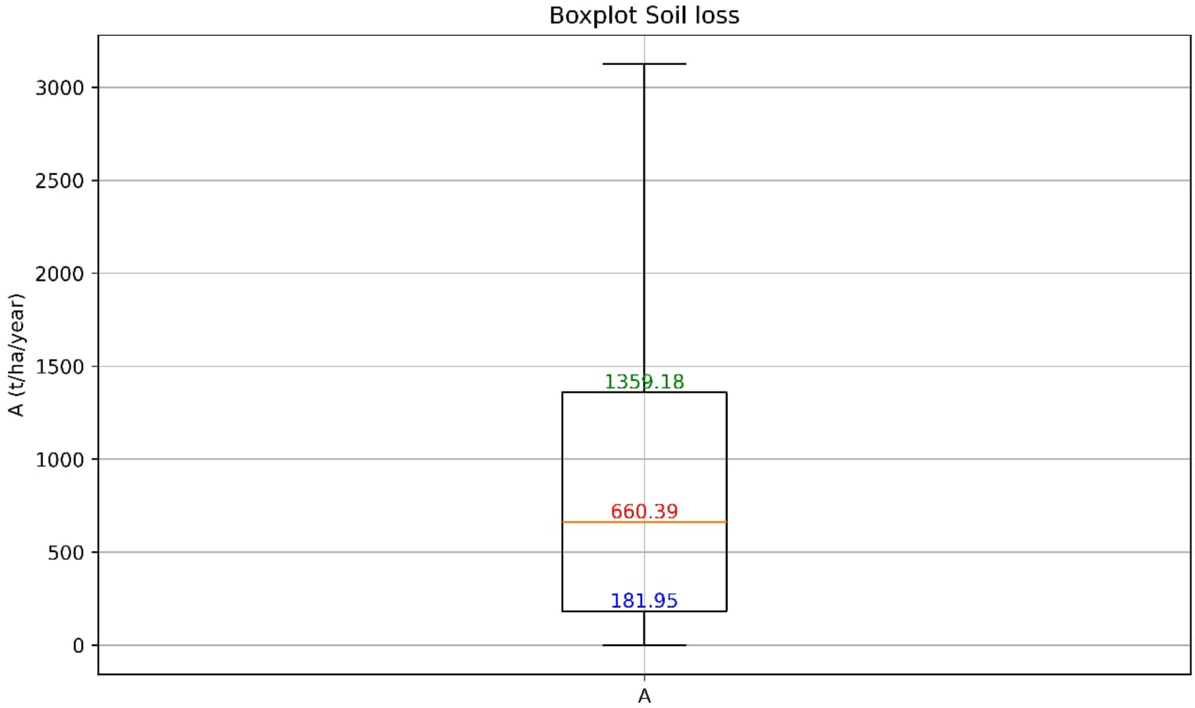
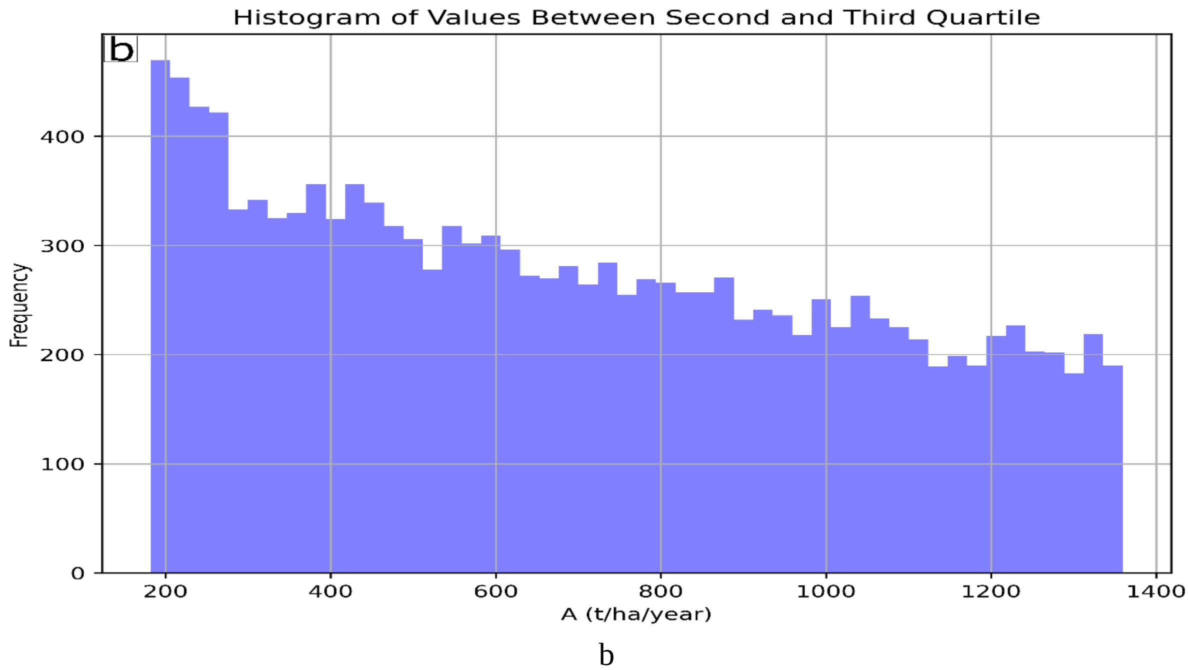


Figure 16. Method 2- map of average values per pixel of the soil erosion



a



b

Figure 17 Soil loss rate: (a) boxplot using method 2 - Box plot of average values of soil erosion (Â) and (b) histogram of values between second and third quartiles of the distribution of average values of (Â).

Notably, the average value (Â) is presented in **Fig.18**, considering the values of the second and third quartiles.

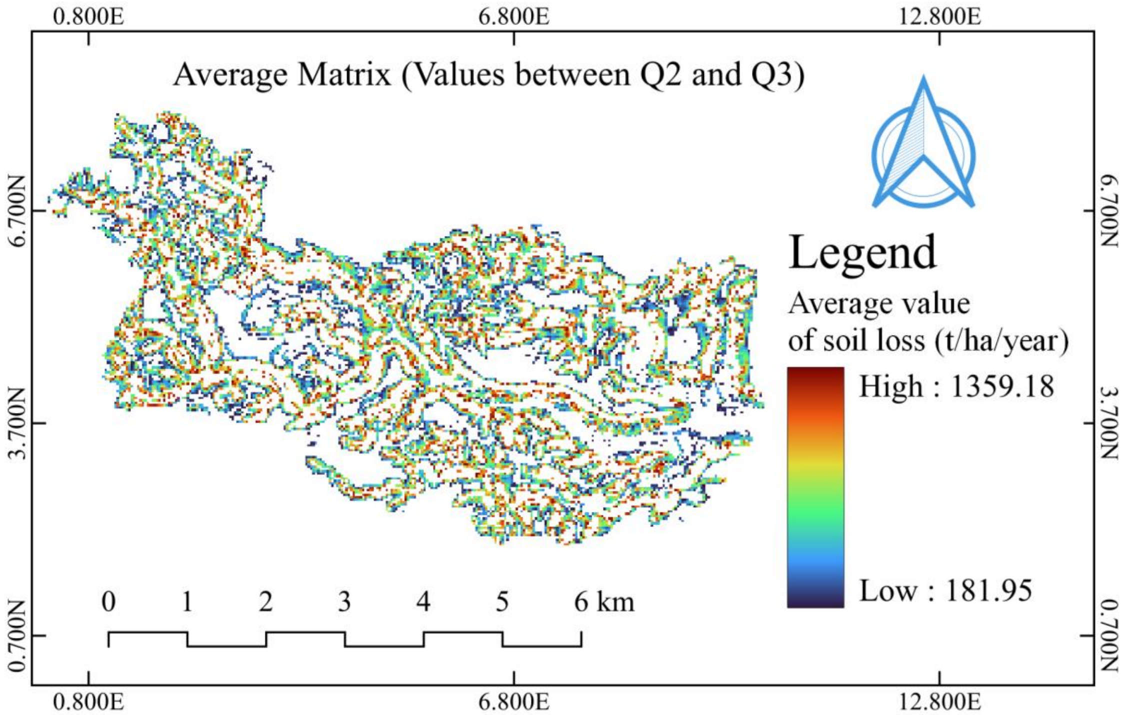


Figure 18. Method 2-average values of (Â) between the second and third quartile.

Based on results of Method 2 and **Fig. 18**, it was found that the average of the soil loss between the second and third quartile values varied from 181.95 and 1359.1 t/ha/year. This result is lower compared to finding of a study carried out in the western region of Bahia, which ranged from 0 to 1846.39 t/year (Ferreira et al., 2019). Besides, comparing the distribution of soil erosion values, taking into account all the results obtained using Method 1 with that of Method 2 using LS **Eq. (5)**, we found that the latter led to a higher soil loss average, as shown **Fig. 19**. Finally, using Method 1, we compared the results of the three LS equations (**Eqs. 5, 6 & 7**) (**Fig. 20**).

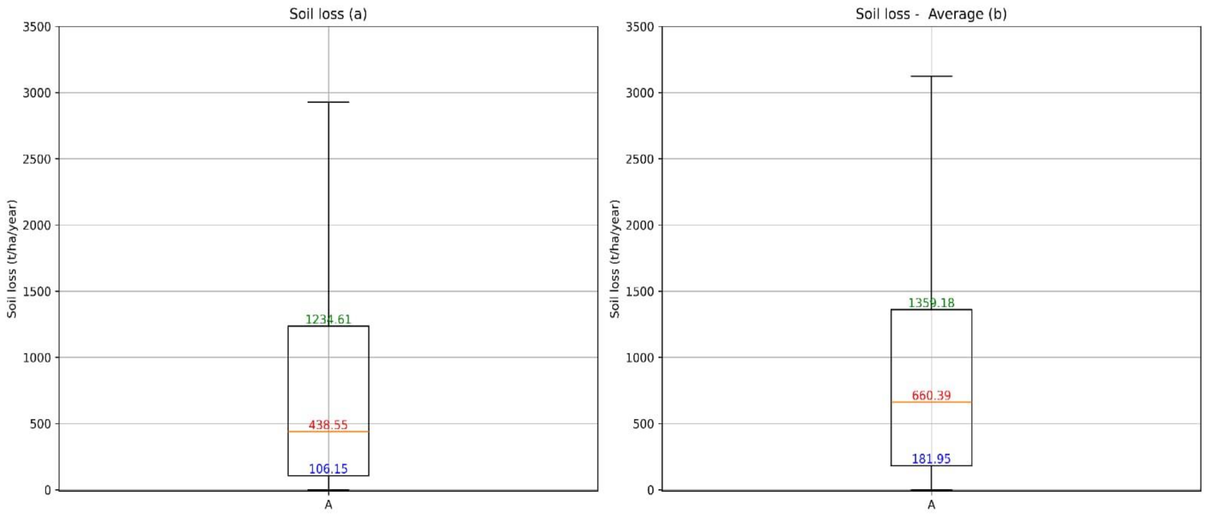


Figure 19 Comparison of two calculation methods of soil erosion- (a) Method 1 and (b) Method 2.

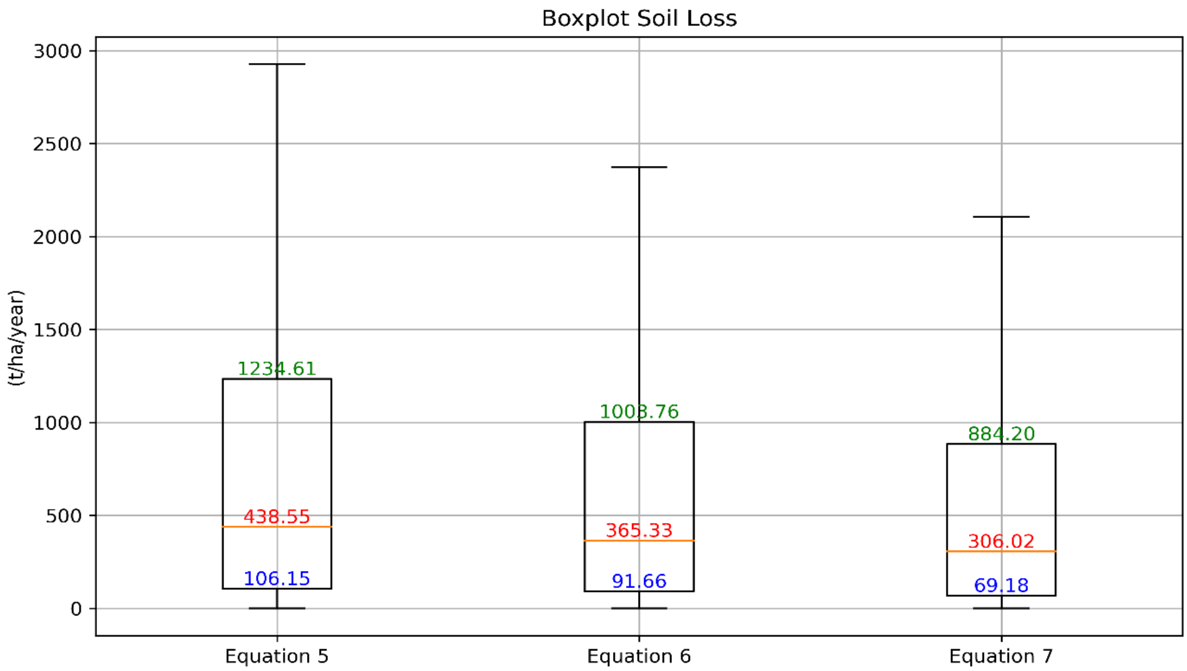


Figure 20. Comparison of results for the three LS equations. Each boxplot consists of 127,875,400 values

In these box plots, blue, red, and green values correspond to the second quartile, median, and third quartile in the distribution (**Fig. 20**). The median given each *LS* calculation is as follows: Eq. 5> Eq. 6> E. 7 **(Fig. 20**). Such a finding may indicate that soil loss value estimated using **Eq. (7)** has lower uncertainty level compared with that of **Eqs.** 5 & 6. In other words, data dispersions of soil loss displayed by Eq. (5) are less regrouped around the median compared to those of **Eqs. 6 & 7**, as shown **Fig. 20**. The severity classes of the soil erosion calculated via

VARS was categorized as weak, moderate, medium, strong, and very strong, as classified by Alves et al. (2022), as shown **Table 6**.

Table 6. Severity range and severity class of soil loss in Pardo River watershed using Method 1 using on LS equations (5), (6) and (7).

Severity range t.(ha.year)$^{-1}$	Severity classes	LS Eq. (5)		LS Eq. (6)		Eq. (7)	
		Area (km^2)	Area (%)	Area (km^2)	Area (%)	Area (km^2)	Area (%)
0−400	Weak	9.44	40.71	10.18	40.71	11.60	46.36
400−600	Moderate	2.40	10.97	2.74	10.97	2.86	11.45
600−1600	Medium	8.55	37.31	9.33	37.31	8.56	34.24
1600−2400	Strong	3.03	8.82	2.20	8.82	1.59	6.37
>2400	Very strong	1.56	6.23	0.54	2.16	0.38	1.55

The results introduced in Table 6 show that only 6.23%, 2.16%, and 1.55% of the Pardo River watershed have soil loss severity considered very high when using LS Eqs. (5), (6), and (7), respectively (**Table 6**). These results indicate that the percentage of soil loss in a watershed varies depending on the equation used for *LS* factor calculation, as well as the severity range used to categorize the soil loss. In terms of soil loss severity, the percentages of soil loss obtained using LS Eqs. (6) (2.16%) and (7) (1.55%) in this study are lower than those obtained in the Verdinho River Basin, Southwestern Goiás, which was 3.17% (Alves et al., 2022).

5. Conclusions

The average soil erosion calculated via QGIS lied between 21.52 and 29.77 t/ha/year, whereas the median of the soil estimated via VARS ranged from 306.02 and 438.55 t/ha/year, depending on the equation used for the *LS* factor. A global sensitivity analysis indicated that the slope angle (β) is the most sensitive parameter, followed by the *CP* factor, SCA, *m*, *K* factor, *n*, and *R* factor. It is noteworthy that β, *m*, *n*, and *SCA* are all parameters of the *LS* factor. The novelty of this study, as compared with others, is the sensitivity analysis of *m* and *n* is often neglected while it was highlighted that they can greatly influence in the end the potential soil erosion estimation. As far as the sensitivity of the parameters of three formulations of LS factor go, they are in the following order of importance: β, *SCA*, *m*, and *n*. Moreover, the results of this study showed that soil erosion is only a nominal value that depends on each factor of RUSLE, particularly the *LS* factor. In conclusion, our findings may help assess the uncertainty of any soil erosion assessment, taking into account the variability of *m* and *n*, particularly.

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Data availability statement

All relevant data are included in the paper or its Supplementary Information.

Conflicts of interest statement

The authors declare there is no conflict.

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Geral conclusions

This thesis assessed ecosystem services in the Atlantic Forest of Southern Bahia, addressing various aspects. These include experiences of forest certification in the Atlantic Forest of Southern Bahia, the role of the forest in watershed hydrology, the role of cacao agroforestry systems in soil conservation and improvement of water quality, and soil loss calculation using RUSLE, among others. Through this thesis, it was revealed that forests have several benefits such as soil and water decontamination, and climate change mitigation. Moreover, this thesis outlined forests can increase the SWA, depending on factors including climate, tree type, tree density, and tree age. Regarding soil loss estimation, it was found that soil loss is not only a nominal value, but also a rich philosophy and a range of uncertainties and sensitivities of the factors composing the RUSLE model. For example, the soil loss value depends on the origin of precipitation data used for the *R* factor calculation, as well as the equation used for *LS* calculation. Due to the uncertainty associated with the data and equations used to calculate the factors of the RUSLE model, it is necessary to use each factor of the RUSLE model with great caution and maximum precision to avoid overestimation or underestimation of soil loss values from a watershed. Therefore, there is a need to conduct experimental studies to better adapt the results of each factor of the RUSLE model to the field reality. In conclusion, the results of the different chapters of this thesis can help decision-makers when investing in ecosystem services in the Atlantic Forest region of southern Bahia.