Universidade de Brasília
Instituto de Ciências Humanas
Departamento de Geografia
Programa de Pós-Graduação em Geografia

UNIVERSIDADE DE BRASÍLIA

PÓS-GRADUAÇÃO EM GEOGRAFIA

IMPACTOS HIDROLÓGICOS DA RESTAURAÇÃO AMBIENTAL DE UMA ÁREA DE CARSTE NOS CERRADOS

Maria Rita Souza Fonseca

Tese de Doutorado

Brasília-DF, Julho de 2022

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Maria Rita Souza Fonseca

Tese de Doutorado submetida ao Departamento de Geografia da Universidade de Brasília, como parte dos requisitos necessários para a obtenção do Grau de Doutor Geografia, área de concentração Gestão Ambiental e Territorial, opção Acadêmica.

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Universidade de Brasília
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FONSECA, MARIA RITA SOUZA

Impactos Hidrológicos da Restauração Ambiental de uma Área de Carste nos Cerrados, 123 p., (UnB – Geografia, Doutor, Gestão Ambiental e Territorial, 2022).

Tese de Doutorado – Universidade de Brasília. Pós-Graduação em Geografia.

1. Hidrossendimentologia 2. Restauração

3. Carste 4.Cerrados

I. UnB. Geografia II. Título (série)

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Maria Rita Souza Fonseca

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Aos meus pais, Celso e Maria José; aos meus filhos, Maíra e Ravi; e ao meu país, Brasil, dedico esta tese.

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Resumo

Impactos Hidrológicos da Restauração Ambiental de uma Área de Carste nos Cerrados.

Autora: Maria Rita Souza Fonseca
Orientador: Rogério S. E. Uagoda

A erosão hídrica é uma das principais causas de degradação do solo, com vastas áreas cultivadas e naturais sendo perdidas anualmente devido à erosão do solo. Em paisagens cársticas, impactos *on* e *off site* ocorrem devido à vulnerabilidade intrínseca e à mudança da cobertura vegetal natural. Por outro lado, a restauração ecológica é uma alternativa capaz de devolver o equilíbrio ao sistema. Os Cerrados, incluindo as zonas cársticas, vêm sofrendo impactos significativos em suas paisagens naturais, decorrentes da conversão da vegetação natural em áreas antrópicas. Nas áreas cársticas, esses impactos incluem a desertificação rochosa, a erosão do solo, e redução das vazões e piora da qualidade da água. Assim, além da erosão nas vertentes (impacto *on-site*), com a subsequente perda de nutrientes, os efeitos *off-site* da erosão incluem a sedimentação das cavernas e cursos d'água, a jusante. Enquanto a tolerância de perda de solo *on-site* nos Cerrados varia entre 4 e12 Mg ha^{-1} ano^{-1}, a tolerância *off-site*, relativa à sedimentação, é de apenas 1,0 Mg ha^{-1} ano^{-1}. O objetivo geral da pesquisa foi avaliar o comportamento hidrossedimentológico de áreas naturais, degradadas, e restauradas de dois tipos de Neossolos (Litólico e Quartzarênico), presentes numa encosta que converge para a Gruna da Tarimba, uma dolina localizada na APA-Nascentes do Rio Vermelho. Para tanto, o estudo foi dividido em três partes: a) Uma meta-análise de dados de perdas de água e solo de parcelas existentes nos Cerrados; b) Um estudo experimental com parcelas de enxurrada (USLE), instalado numa área degradada de carste de Mambaí-GO; c) Um estudo hidrossedimentológico da vertente convergente para uma dolina, na mesma área. O efeito de diferentes coberturas (solo descoberto, restauração ecológica com espécies nativas, e Cerrado *sensu strictu*) foi analisado durante 3 anos hidrológicos. A vertente no entorno das parcelas foi também restaurada, com espécies nativas e pastagem. Nos três anos analisados, o volume de escoamento superficial nas parcelas diminuiu de 546 mm ano^{-1}, na condição degradada, até 360 mm ano^{-1}, na condição restaurada. A perda de solo média diminuiu de 34 Mg ha^{-1} ano^{-1} para cerca de 5 Mg ha^{-1} ano^{-1}, respectivamente. No caso da parcela sob cerrado natural, a perda de solo foi ainda menor, de 0,7 Mg ha^{-1} ano^{-1}. A tolerância de perda de solo foi ultrapassada na parcela descoberta nos três anos hidrológicos. No último ano analisado, a perda de solo das parcelas restauradas se manteve abaixo da tolerância à erosão *on-site*, com o fator C da USLE passando de 0,44, no primeiro ano, para 0,03 no terceiro ano. Como consequência, houve uma redução de 50% no aporte de sedimentos no exutório da vertente, durante o período de estudo. Apesar da significativa redução da erosão e da sedimentação na vertente restaurada, 29% e 61% da sua área total ainda apresentaram, ao final do 3º ano, perdas de solo acima dos valores toleráveis *on* e *off-site*, respectivamente. Observou-se uma depleção significativa da fração areia no sedimento oriundo das parcelas restauradas, e um enriquecimento dessa fração nas áreas descobertas. Os resultados indicam que a restauração ecológica em áreas de carste dos Cerrados é capaz de gerar significativos serviços hidrossedimentológicos para as cavernas da Gruna da Tarimba e para o rio Vermelho, contribuindo para sua sustentabilidade.

Palavras-chaves: perda de solo, escoamento superficial, área cárstica, Cerrado, restauração ecológica, produção de sedimento, enriquecimento de finos.

Abstract

Hydrological impacts of the ecological restoration of a karst area in the Cerrado

Author: Maria Rita Souza Fonseca
Adivisor: Rogério S. E. Uagoda

Water erosion is one of the main causes of soil degradation, with vast cultivated and natural areas being annually lost due to soil erosion. In carsic landscapes, on and off-site impacts occur due to intrinsic vulnerability and the change in natural vegetation cover. On the other hand, ecological restoration is an alternative capable of returning balance to the system. The Cerrados, including the carstica zones, have suffered significant impacts on their natural landscapes, resulting from the conversion of natural vegetation into anthropic areas. In carstica areas, these impacts include rock desertification, soil erosion, and reduced flow rates and worsening water quality. Thus, in addition to erosion in the strands (on-site impact), with the subsequent loss of nutrients, the off-site effects of erosion include the sedimentation of caves and downstream watercourses. While the on-site soil loss tolerance in the Cerrados varies between 4 and 12 Mg ha^{-1} $year^{-1}$, the off-site tolerance, relative to sedimentation, is only 1.0 Mg ha^{-1} $year^{-1}$. The objective of the research was to evaluate the hydrosedimentological behavior of natural, degraded and restored areas of two types of Neosols (Litholic and Quartzarenic), present on a slope that converges to the Tarimba Group, a sinkhole located in the APA-Springs of the Rio Vermelho. For this, the study was divided into three parts: a) A meta-analysis runoff and soil loss data of existing studies in the Cerrados; b) An experimental study with runoff plots (USLE), installed in a degraded karst area of Mambaí-GO; c) A hydrosedimentologic study of a sinkhole slope, in the same area. The effect of different soil covers (bare soil, ecological restoration with native species, and *sensu strictu* Cerrado) was analyzed for 3 hydrological years. The slope surrounding the plots was also restored, with native species and pasture. In the three years analyzed, the volume of surface runoff in the plots decreased from 546 mm $year^{-1}$, in the degraded condition, to 360 mm $year^{-1}$, in the restored plots. The average soil loss decreased from 34 Mg ha^{-1} $year^{-1}$ to about 5 Mg ha^{-1} $year^{-1}$, respectively. In the case of the plot under natural Cerrado, soil loss was even lower, of 0.7 Mg ha^{-1} $year^{-1}$. Soil loss tolerance was exceeded in the bare plot, in the three hydrologic years. In the last year analyzed, soil loss of the restored plots remained below the on-site tolerance, with the USLE-C varying from 0.44, in the first year, to 0.03, in the third year. As a consequence, there was a 50% reduction in sediment yield to the sinkhole, downstream. Despite the significant reduction of erosion and sedimentation in the restored slope, 29% and 61% of its total area experienced soil loss above the on and off-site tolerances, respectively. A significant depletion of the sand fraction in the sediment from the restored plots and an enrichment of this fraction in the bare plots were observed. The results indicate that the ecological restoration of karst areas of the Cerrados is capable of generating important hydrosedimentologic services for the Caves of the Tarimba Sinkhole and to the Rio Vermelho, contributing to its sustainability.

Keywords: soil and water loss, karst area, Cerrado, ecological restoration, sediment yield.

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Introdução

1. Introdução

A erosão hídrica é uma das principais causas de degradação do solo (Lal, 2001). Estima-se que 100.000 km² de áreas cultivadas sejam perdidos anualmente devido à erosão do solo em todo o mundo, a taxas 10 a 40 vezes superiores à formação do solo (Pimentel, 2006).

Os tipos de erosão do solo incluem a erosão laminar (*interrill*), através do desprendimento de partículas do solo por impacto de gota de chuva e seu transporte por fluxo superficial (Foster et al., 1981; Meyer and Wischmeier, 1969); erosão em sulcos (*rill*), onde partículas são destacadas e transportadas por fluxo canalizado (Foster et al., 1981; Govers et al., 2007); e voçorocas, causada pela combinação de processos de lavagem hidráulica e movimento de massa e o subsequente transporte de sedimentos por fluxo (Bocco, 1991).

No bioma Cerrado estes processos ocorrem naturalmente e quando as áreas naturais são convertidas para outras coberturas. Fonseca et al. (2021) estimou uma perda de solo média anual para a cobertura natural de 0,1 (Mg ha^{-1} ano^{-1}), média anual de pastagem 0,2 (Mg $ha^{-1}ano^{-1}$), e solo exposto 19,4 (Mg ha^{-1} ano^{-1}).

Fica evidente que a vegetação protege o solo contra a erosão, tanto em ambientes agrícolas (Laflen et al., 1978) como naturais (Zuazo and Pleguezuelo, 2008). O dossel das plantas e a serapilheira reduzem o impacto das gotas de chuva (Ma et al., 2014), a serapilheira e as raízes superficiais diminuem o escoamento e o transporte de sedimentos (Hofmann et al., 1983).

Como a vegetação é a única característica natural que protege o solo contra os processos erosivos (Zhang et al., 2016; Zhao and Hou, 2019), quando esta é alterada pelo homem a perda de solo decorrente da erosão hídrica é acompanhada por perdas de nutrientes essenciais do solo (Nadeu et al., 2011; Nie et al., 2015; Wacha et al., 2020).

Em ambientes cársticos, a degradação se dá através de processos de erosão superficial e subterrânea, causadas pela erosão hídrica, por intemperismo químico e pela gravidade, respectivamente (Zeng et al., 2018).

Paisagens cársticas são ambientes altamente frágeis, representando aproximadamente 12% dos continentes terrestres (Febles-González et al., 2012). Dentre as ameaças que afetam as áreas cársticas estão a erosão do solo e degradação progressiva (Hu et al., 2018; Parise et al., 2009).

Na savana brasileira, há várias ocorrências de áreas cársticas, com mais de 11 mil cavernas, formadas por rochas carbonáticas e siliciclásticas (CECAV and ICMBio, 2022). O estado de Goiás apresenta o quinto maior número de cavernas mapeadas no Brasil, abrigando 1.000 cavidades, entre elas a caverna da Tarimba (Caldeira et al., 2021), na região nordeste do Estado. Os Cerrados, incluindo aí as zonas cársticas, vêm sofrendo impactos significativos em suas paisagens naturais devido às pressões antrópicas nos últimos 50 anos, tais como a erosão do solo (Anache et al., 2018; Oliveira et al., 2015; Vanwalleghem et al., 2017).

Como consequência desses impactos, ocorrem a desertificação rochosa (Jiang et al., 2014; Zhao and Hou, 2019), a erosão das partículas finas do solo (Jacinthe et al., 2004; Nie et al., 2015; Wacha et al., 2020), e problemas relacionados à quantidade e qualidade da água (Coxon, 1999; Goldscheider and Drew, 2007; Parise et al., 2009).

Além disso, como os solos de zonas cársticas são geralmente rasos (Zhao & Hou, 2019), sua capacidade de retenção de água é limitada (Jiang et al., 2014; Wang, 2011; Zhao and Hou, 2019). Assim, quando processos erosivos neles ocorrem, solo e vegetação tendem a se degradar conjuntamente (Jiang et al., 2014).

Assim, além dos efeitos da erosão nas vertentes (impacto *on-site*), incluindo a redução da produtividade vegetal e a perda de nutrientes (Lal, 2001), os efeitos *off-site* da erosão resultam na sedimentação dos cursos d'água (Minella et al., 2009). Em áreas cársticas dos Cerrados, esses cursos d'água podem ser tanto rios superficiais como subterrâneos (Merten and Minella, 2006).

Os impactos on site da erosão do solo incluem reduções na produtividade das culturas (Duan et al., 2017) e degradação do solo (Lal, 2001). No entanto, o conceito de erosão tolerável baseado apenas na produtividade do solo (Schertz, 1983), profundidade do solo (Skidmore, 1982) e taxa de formação do solo(Montgomery, 2007) é redutivo (Di Stefano et al., 2016), exigindo que os efeitos off site sejam contabilizados (Bazzoffi, 2009).

O limite máximo de perda de solo tolerável em vertentes nos Cerrados, varia entre 4 e 12 Mg ha^{-1} ano^{-1} (Chaves, 2010). A tolerância *on site* relacionada à perda de produtividade nos Estados Unidos é de 5–12 Mg ha^{-1} ano^{-1} (Schertz, 1983). No Brasil, a tolerância no local varia de 2 a 15 Mg ha^{-1} ano^{-1}, dependendo da profundidade do solo e da relação textural (Bertoni and Lombardi, 1991; Corrêa et al., 2015). Na Europa, Souza et al. (2021) descobriram que a tolerância à perda de solo on site foi inversamente proporcional à taxa de erosão observada e ao tempo, e o usou como indicador de sustentabilidade agronômica

Além da tradicional tolerância à erosão on-site há a tolerância à sedimentação off-site (Tavares et al., 2021), relativa à impactos de assoreamento dos rios e qualidade da água (Fonseca et al., 2021; Lal, 2001). Na falta de outro valor mais adequado, essa tolerância foi tomada como 1,0 Mg ha^{-1} ano^{-1} (Fonseca et al., 2021; Mullan, 2013; Verheijen et al., 2009). Reconhecendo que o conceito de "perda de solo tolerável" pode ser enganosa, Bui et al. (2011) recomendou que limitar a erosão do solo pode ser mais útil do que a tolerância à perda o solo porque é mais amplamente aplicável a uma gama de objetivos ambientais.

Pelo o exposto e reconhecendo que a vertente cárstica da dolina que converge para a Gruna da Tarimba passava por um processo de conversão de cobertura natural para uma pastagem e solo exposto resolveu-se instalar parcelas de enxurrada para o entendimento dos problemas decorrentes desse tipo de conversão.

Devido à sua facilidade de instalação e operação, as parcelas de enxurrada são universalmente utilizadas para avaliar a erosão de sulcos e laminar (Mutchler et al., 1988) al., 1988) em áreas naturais (Renard and Foster, 1985) e agrícolas (Wischmeier, 1960). Portanto, a Equação Universal de Perda de Solo e as estratégias de conservação subsequentes foram formuladas com base nos resultados das parcelas de escoamento superficial (Kinnell, 2019), apesar das limitações preditivas da equação (Wischmeier, 1976) e das incertezas (Risse et al., 1993).

Neste estudo seis parcelas de enxurrada sob diferentes usos do solo, em uma área cárstica degradada do Cerrado brasileiro, foram instaladas na encosta da dolina conectada à Gruna da Tarimba, na Área de Proteção Ambiental Nascentes do Rio Vermelho.

Os resultados do presente estudo poderão ser utilizados na elaboração de políticas públicas em áreas de restauração do Cerrado, como a elaboração do Plano de Manejo da APA das Nascentes do Rio Vermelho (APA-NRV), uma vez que desenvolve-se no escopo do *Projeto Susceptibilidade, Hidrologia e Geomorfologia Cárstica Aplicadas à*

*Conservação do Patrimônio Espeleológico da Área de Proteção Ambiental das Nascentes do Rio Vermelh*o (Termo de Compromisso de Compensação Espeleológica- TCCE nº 01/2018/ICMBIO).

Este projeto vem sendo desenvolvido sob a coordenação técnica e científica do Professor Rogerio S. E. Uagoda (GEA/UnB), que também é orientador do presente projeto de Doutorado, em desenvolvimento no Programa de Pós-graduação em Geografia, da Universidade de Brasília.

1.1. Objetivos

O *objetivo geral* da pesquisa foi avaliar os diferentes estudos de perda de água e solo nos Cerrados, bem como avaliar o comportamento hidrossedimentológico de áreas naturais, degradadas, e restauradas de Neossolos de uma região cárstica do Cerrado, através de parcelas de enxurrada e modelagem hidrossedimentológica.

Os *objetivos específicos* do estudo foram os seguintes:

i. Realizar uma meta-análise de dados publicados sobre escoamento superficial e perda de solo nos Cerrados;
ii. Medir o escoamento superficial e perdas de solo de parcelas de enxurrada sob diferentes coberturas de solo;
iii. Comparar as taxas de erosão medidas com as tolerâncias de perda de solo correspondentes;
iv. Obter o balanço hídrico do solo para os diferentes tratamentos analisados;
v. Calcular o coeficiente de escoamento superficial (CN) e o fator C (USLE) das diferentes coberturas analisadas;
vi. Avaliar a produção de sedimentos e o enriquecimento de finos de uma vertente cárstica a montante de um sumidouro de dolina;
vii. Comparar as taxas de sedimentação com as tolerâncias *off site*; e
viii. Avaliar a eficiência hidrológica da restauração ecológica implementada.

O Projeto buscou responder às seguintes *perguntas*:

- A perda de solo e o escoamento superficial de diferentes áreas do Cerrado brasileiro são sustentáveis sob o ponto de vista dentro (*on-site*) e fora (*off-site*) das propriedades?
- O escoamento superficial e a perda de solo em áreas naturais, degradadas, e restauradas de uma vertente típica que converge para cavernas na APA-Nascentes do Rio Vermelho são ambientalmente sustentáveis?
- A perda de solo que ocorre na encosta estudada é parte de processo de desertificação cárstica e esse poderia ser mitigado por um manejo conservacionista?
- É possível reduzir essas perdas a níveis toleráveis dentro e fora da propriedade, utilizando técnicas de restauração ecológica?

1.2. Área de estudo

A área de estudo do experimento localiza-se na zona rural do município de Mambaí (GO), Brasil, inserida na bacia hidrográfica do Rio Corrente e na Área de Proteção Ambiental Nascentes do Rio Vermelho (APA-NRV).

A geomorfologia da área é composta pelo Chapadão Central (porção superior) remanescente da superfície Sul americana, constituída pelo Grupo Urucuia, formado por arenitos que apresentam sedimentos siliciclásticos não consolidados, sendo o Vale do Paranã (porção inferior) formado de rochas pelíticas intercaladas com os carbonatos da Formação Lagoa do Jacaré, do Grupo Bambuí (Tavares et al., 2021) (Figura 1).

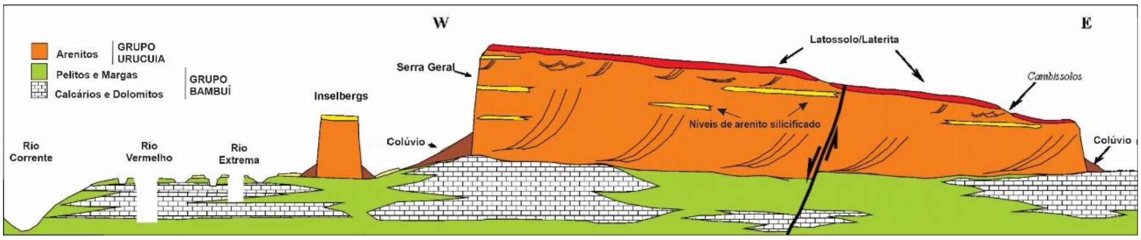

Figura 1 – Contexto geomorfológico (Gaspar e Campos, 2007).

Da base para o topo da sequência estratigráfica encontram-se carbonatos e calcários calcíticos sobrepostos por pelitos da Formação Lagoa do Jacaré, os quais formam Chernossolos e Neossolos Litólicos. Na parte superior da vertente há detritos arenosos advindos do Grupo Urucuia, que formam Neossolos Quartzarênicos (Caldeira et al., 2021; Gaspar & Campos, 2007 adaptado por Uagoda et al., 2019).

A Gruna Tarimba é considerada uma das cavernas mais importantes da região e também uma das maiores do país em comprimento (Hussain et al., 2020). O clima na região é tropical úmido (Aw- Koppen), com subtipo clima de savana, com inverno seco e precipitação dominante nos meses de verão (da Silva et al., 2008). A precipitação média anual é de 1.200 mm e temperatura média é de 25°C.

A área experimental está situada numa vertente cárstica, situada sobre a Gruna da Tarimba, que recebe a enxurrada e o sedimento da dolina através de um sumidouro, situado a jusante das parcelas experimentais (Figura 2).

As bordas da dolina que dá acesso à caverna Gruna da Tarimba foram bastante alteradas pelo desmatamento e pela atividade pecuária extensiva, comuns na região da APA-NRV (Figura 2c).

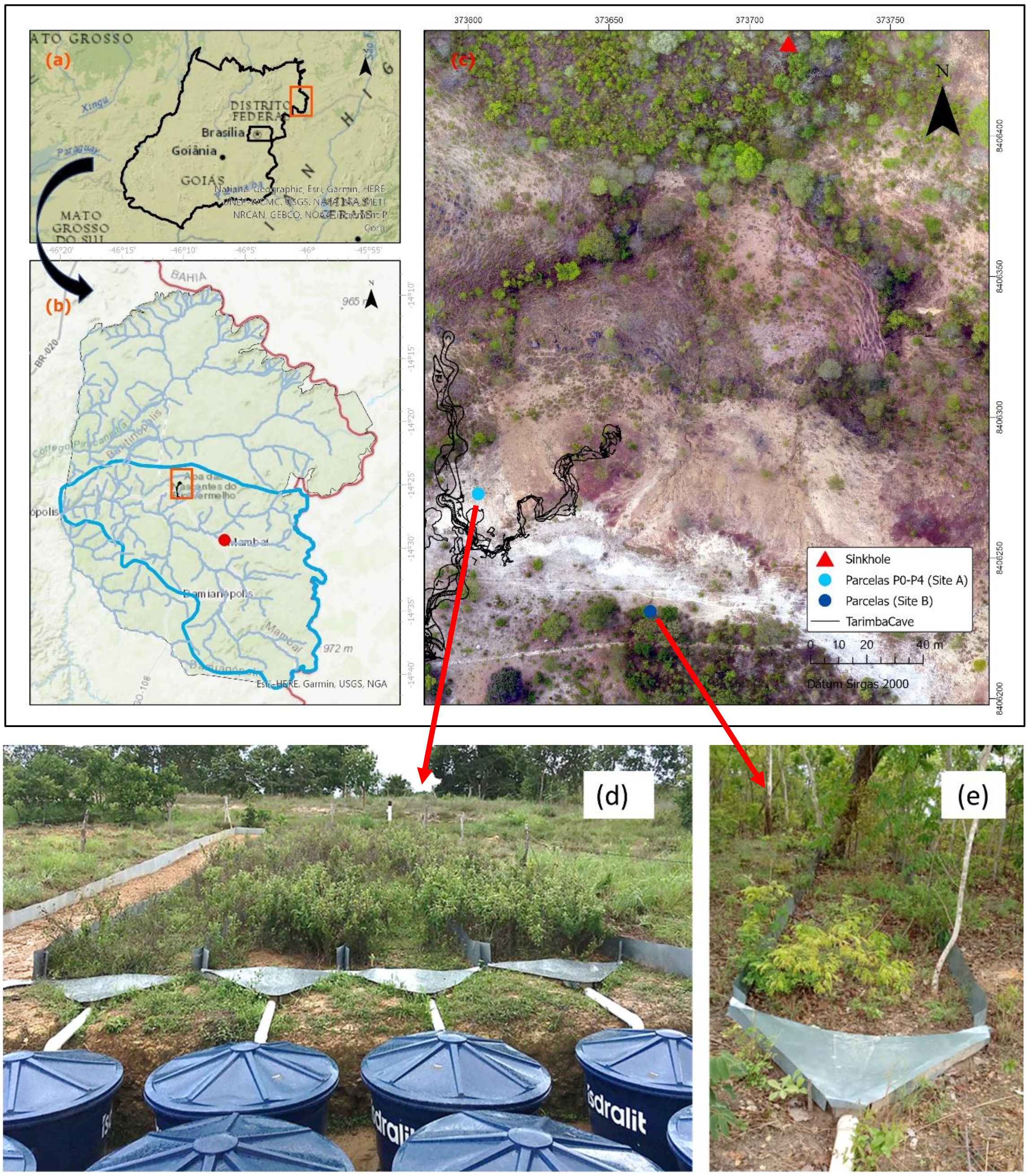

Figura 2 – Localização da área de estudo. (a) estado de Goiás, (b) APA Nascentes do Rio Vermelho e bacia hidrográfica do rio Corrente, (c) vertente cárstica, situada sobre a Gruna da Tarimba, (d) parcelas de enxurradas P0 – P4 (com coberturas solo exposto e em restauração), e (e) P5 (cobertura Cerrado).

As parcelas de enxurrada foram instaladas sobre o Neossolo Litólico (Sítio A), derivado de pelito, e sobre o Neossolo Quartzarênico (Sítio B), derivado de arenito (Figura 2 d, e).

Desenvolvimento

2. Desenvolvimento

2.1. Organização do Trabalho

Esta tese de doutorado foi organizada em três Capítulos:

- Um Capítulo apresentando uma meta-análise de dados de escoamento superficial e perda de solo em parcelas de enxurrada no bioma Cerrado;
- Um Capítulo apresentando os impactos *on-site* (na propriedade), relativo à perda de solo e escoamento superficial e ao balanço hídrico; e
- Um capítulo sobre o impacto *off-site* (a jusante), relativo à sedimentação da caverna e sobre a qualidade do sedimento exportado pela vertente.

A tese foi elaborada em formato de artigos científicos, seguindo as diretrizes da Pós Graduação em Geografia da UnB, permitindo assim sua maior divulgação. Nesse sentido, o estudo resultou em uma publicação internacional Qualis A-1, em 2021 (ESPL-Capítulo 1), e em dois artigos submetidos recentemente a revistas internacionais-A1 (Catena-Capítulo 2, e ESPL-Capítulo 3), ambos em revisão (Figura 3).

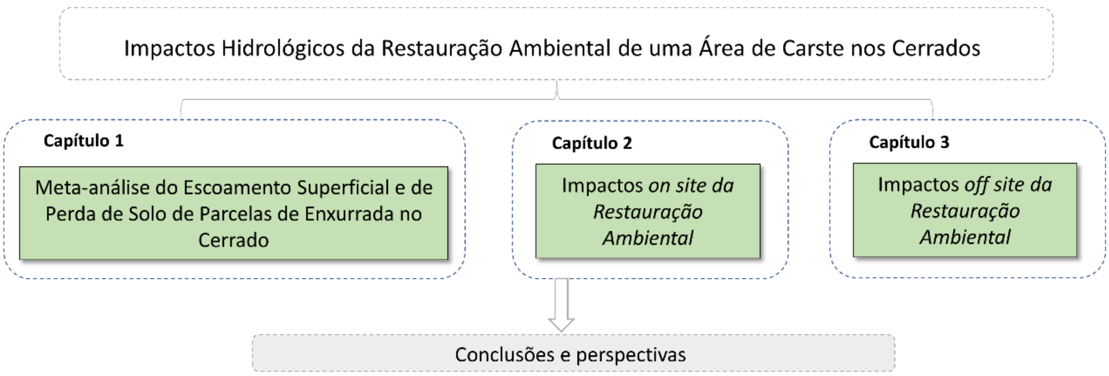

Figura 3 – Organograma da estruturação da Tese.

Capítulo 1

Artigo sobre Meta-análise de Perda de Solo e Escoamento Superficial no Cerrado (Fonseca et al, 2021)

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RESEARCH ARTICLE

ESPL WILEY

Rates, factors, and tolerances of water erosion in the Cerrado biome (Brazil): A meta-analysis of runoff plot data

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Abstract

Due to the high rainfall erosivity and highly erodible soils, water erosion is severe in Brazil. Soil and ecosystem degradation occurs when erosion exceeds on- and off-site soil loss tolerances, with significant socioeconomic and environmental impacts. In the last 50 years, the Brazilian Cerrado had 53% of its original vegetation converted to agriculture and pastureland. Although erosion plot studies exist in the region, the data are fragmented and unexplored, hindering the development of soil conservation policies. The objective of the present research was to compile, systematize, and statistically analyze the existing erosion plot data in the Brazilian Cerrado, correlating the observed results with different environmental and management factors, and with the corresponding soil loss tolerances. Twenty runoff plot datasets of the Brazilian Cerrado, encompassing 5 states, 10 sites, 108 plots, and 360 plot·years were compiled and thoroughly analyzed. Mean annual rainfall, runoff, and soil loss were 1443.5 mm year^{-1}, 83.1 mm year^{-1}, and 8.9 Mg ha^{-1} year^{-1}, respectively. After the data were normalized with respect to plot length, steepness, and climate, runoff and soil loss were found to be significantly higher in soils with impermeable horizons and in land uses without permanent soil cover ($p < 0.05$). Erosion under permanently covered plots was below the on- and off-site soil loss tolerances. A power equation provided the best fit between plot runoff and soil loss ($R^2 = 0.71$; $p < 0.05$), indicating that runoff volume, easier to estimate, could be used as a proxy for upslope erosion. Although erosion plot data cannot be extrapolated to the whole landscape, the research results provide useful elements for the development of sound conservation policies in the Cerrado and in other similar savannas of the world.

KEYWORDS
Cerrado, meta-analysis, runoff/erosion relationship, tolerance

1 | INTRODUCTION

One of the main causes of soil degradation is soil erosion by water (Lal, 2001). It is estimated that 100 000 km^2 of cultivated areas are lost annually due to soil erosion worldwide, at rates 10 to 40 times higher than soil formation (Pimentel, 2006). Erosion on-site processes and impacts, such as those measured in runoff plots, focus on sheet and rill erosion, whereas those that focus on off-site effects, such as watersheds, additionally estimate gully and bank erosion (de Vente & Poesen, 2005).

Vegetation protects the soil against erosion, in both agricultural (Laflen et al., 1978) and natural (Zuazo & Pleguezuelo, 2008) settings. While plant canopy and ground litter reduce raindrop impact (Ma et al., 2014), ground litter and surface roots decrease runoff and sediment transport (Hofmann et al., 1983).

Soil loss and runoff are inseparable processes. Runoff, defined as the excess water not infiltrated in the soil, depends on factors such as rainfall, soil texture and infiltrability (Bazzoffi, 2009). Although runoff plays an important role in soil loss and is easier to assess, their theoretical relationship is not yet fully understood (Ferreira et al., 2012).

However, empirical relationships between runoff and soil loss were obtained at the laboratory (Mamedov & Levy, 2018) and plot scales (Parsons et al., 2006a; Santos et al., 2017), allowing runoff to be used as an estimator of soil loss (Merritt et al., 2003; Yang et al., 1998).

Although equations relating soil loss and runoff exist for specific conditions, the complexity of the detachment and transport processes by rainfall and runoff (Flanagan et al., 2006), including rill side-wall sloughing and head-cutting (Quansah, 1985), hinders the establishment of a universal runoff–soil loss relationship.

On-site impacts of soil erosion include reductions in crop productivity (Duan et al., 2017) and soil degradation (Lal, 2001). However, the concept of tolerable erosion based only on soil productivity (Schertz, 1983), soil depth (Skidmore, 1982) and soil reformation rate (Montgomery, 2007) is reductive (Di Stefano & Ferro, 2016), requiring that off-site effects be accounted for (Bazzoffi, 2009).

On-site tolerance related to productivity loss in the United States is 5–12 Mg ha^{-1} year^{-1} (Schertz, 1983). In Brazil, on-site tolerance varies from 2 to 15 Mg ha^{-1} year^{-1}, depending on soil depth and textural ratio (Bertoni & Lombardi, 1990; Corrêa et al., 2015). In Europe, Souza et al. (2021) found that on-site soil loss tolerance was inversely proportional to the observed erosion rate and time, and used it as an indicator of agronomic sustainability.

Off-site tolerance, the upslope erosion rate beyond which silts up downstream hydraulic structures and aquatic ecosystems (Lal, 2001), is lower than on-site tolerance because of the risk of downstream impairment, and is taken as 1.0 Mg ha^{-1} year^{-1} (Mullan, 2013; Verheijen et al., 2009). Recognizing that the concept of 'tolerable soil loss' can be misleading, Bui et al. (2011) recommended that limiting soil erosion could be more useful than soil loss tolerance because it is more widely applicable to a range of environmental objectives.

Plott experiments are used to assess the processes driving upslope soil erosion (Boardman & Evans, 2019), in natural (Renard & Foster, 1985) and agricultural settings (Hudson, 1993), and are useful to compare the effects of different land treatments (Mutchler et al., 1988). The universal soil loss equation (USLE) and subsequent soil conservation strategies were formulated based on the results of runoff plots (Kinnell, 2019), despite the equation's intrinsic predictive limitations, including the processes of gullying and deposition (Alewell et al., 2019), and the uncertainties of its predictions (Chaves, 2010; Risse et al., 1993).

It is recognized that experimental data from runoff plots, considered upslope source areas, cannot be extrapolated to different landscape scales (Boardman & Evans, 2019), and that caution should be exercised in comparing measurements based on different methodologies (e.g. runoff plots, erosion pins, etc.), temporal or spatial scales (Cantón et al., 2011). If extrapolated across the landscape, erosion rates from plot data would significantly overestimate the process (Evans et al., 2017). Researchers have tackled this issue with the use of appropriate GIS landscape routines (Mitasova et al., 1996) and sediment delivery ratios (Walling, 1983).

However, even distributed watershed models, such as the Water Erosion Prediction Project (WEPP; Nearing et al., 1989), use runoff plot data (Elliot et al., 1989), and route the eroded sediment along the slope with appropriate routines.

Because of its high rainfall erosivity (Silva, 2004) and its erodible soils (Marques et al., 1997), water erosion is severe in Brazil, with significant economic and environmental costs (Hernani et al., 2002). Anache et al. (2017) reported a mean soil loss of 34.5 Mg ha^{-1} year^{-1} in runoff plots in different soils and land uses in the southeastern part of the country, way above the soils' tolerance levels (Corrêa et al., 2015).

The Cerrado, a savanna-type, gently sloping biome covering 200 million hectares in central Brazil, is underlain by weathered (Goedert, 1983) and erodible soils (Klink & Machado, 2005). Since the 1980s, the Cerrado has been a new agricultural frontier in Brazil

TABLE 1 Metadata of the runoff plot studies in the Cerrado biome, used in the present study

Study	Township	State	Years	Plots	Observed data	Source
1	Itirapina	SP	4	9	Runoff and soil loss	Anache et al. (2018)
2	Guanhães	MG	1	6	Runoff and soil loss	Brito et al. (2005)
3	Três Lagoas	MS	1	9	Runoff and soil loss	Cândido et al. (2014)
4	Lavras	MG	1	6	Runoff and soil loss	Carvalho et al. (2007)
5	Planaltina	DF	1	6	Soil loss	Dedecek (1986)
6	Planaltina	DF	5	7	Soil loss	Dedecek (1986)
7	Planaltina	DF	7	7	Soil loss	Dedecek (1986)
8	Lavras	MG	1	7	Soil loss	Dias et al. (2013)
9	Dourados	MS	7	4	Runoff and soil loss	Hernani et al. (1997)
10	Dourados	MS	6	4	Runoff and soil loss	Hernani et al. (1999)
11	Cape Verde	MT	1	5	Runoff and soil loss	Leite et al. (2009)
12	Lavras	MG	1	4	Runoff and soil loss	Lima et al. (2014)
13	Lavras	MG	3	4	Runoff and soil loss	Lima et al. (2018)
14	Itirapina	SP	2	2	Runoff and soil loss	Oliveira et al. (2015)
15	Belo Oriente	MG	2	6	Runoff and soil loss	Pires et al. (2006)
16	Pindorama	SP	12	6	Runoff and soil loss	Prochnow et al. (2005)
17	Sinop	MT	1	6	Runoff and soil loss	Rieger et al. (2015)
18	Lavras	MG	4	2	Runoff and soil loss	Silva et al. (2005)
19	Pindorama	SP	5	2	Runoff and soil loss	Sosa (1987)
20	Pindorama	SP	1	6	Runoff and soil loss	Youlton et al. (2016)

(Borlaug, 2002), with 53% of its original area already converted to agriculture and pastureland (Beuchle et al., 2015).

Although there are runoff plots in the Cerrado, their data are fragmented and underutilized, hindering the development of effective soil conservation policies in the region (Chaves, 2010). Considering the above aspects, the objective of this research was to compile, systematize, and analyse the existing runoff and soil loss data of the Cerrado plots, to statistically correlate the observed results with different environmental and management factors, and to assess their on- and off-site sustainability, with regard to soil loss tolerance. Additionally, a general relationship between soil loss and runoff was sought (Parsons et al., 2006b), facilitating the soil loss prediction process and the establishment of appropriate policy recommendations.

2 | METHODOLOGY

2.1 | Selected runoff plots

Following the comprehensive surveys of Cerdan et al. (2006) and Maetens et al. (2012) in Europe, and of Anache et al. (2017) in southeastern Brazil, 20 runoff plot datasets of the Brazilian Cerrado, encompassing 5 states, 10 sites, 108 plots, and 360 plot years, were compiled and analysed. The selected studies had at least one full year of soil and water loss records. The plot metadata are presented in Table 1, and the location of the plots is shown in Figure 1.

2.2 | Analysis of plot data

The Cerrado plot data include plot dimensions (plot length and area), average slope, annual precipitation volume, soil type and texture, land use and cover type, as well as observed annual runoff and soil loss. The runoff plots studied involved four types of soils, typical of the Brazilian savanna (Goedert, 1983), with different depths and textures (Table 2). Except for the Dystrudept (Inceptisol), a shallow soil with low infiltrability (Silva et al., 2005), the others are deep soils, with relatively high infiltration capacities.

In the present study, the land cover types were classified as *permanent* (natural Cerrado, eucalyptus, pasture, conservation agriculture) and *non-permanent* (conventional agriculture, bare soil), since the former maintains a permanent canopy and/or mulch cover throughout the year (Cerdan et al., 2006).

Conservation agriculture is a set of agricultural practices supported on three pillars: permanent soil cover, minimum or no soil tillage, and crop rotation (Telles et al., 2020). Table 3 describes the land use and land cover characteristics of the runoff plots studied.

TABLE 2 Soils analysed in the present study

Soil type	Texture	Abbreviation
Dystrudept	Clay	DT
Hapludox	Clay; loam	HX
Quartzipsamment	Sand	QZ
Ultisol	Sand; clay	UT

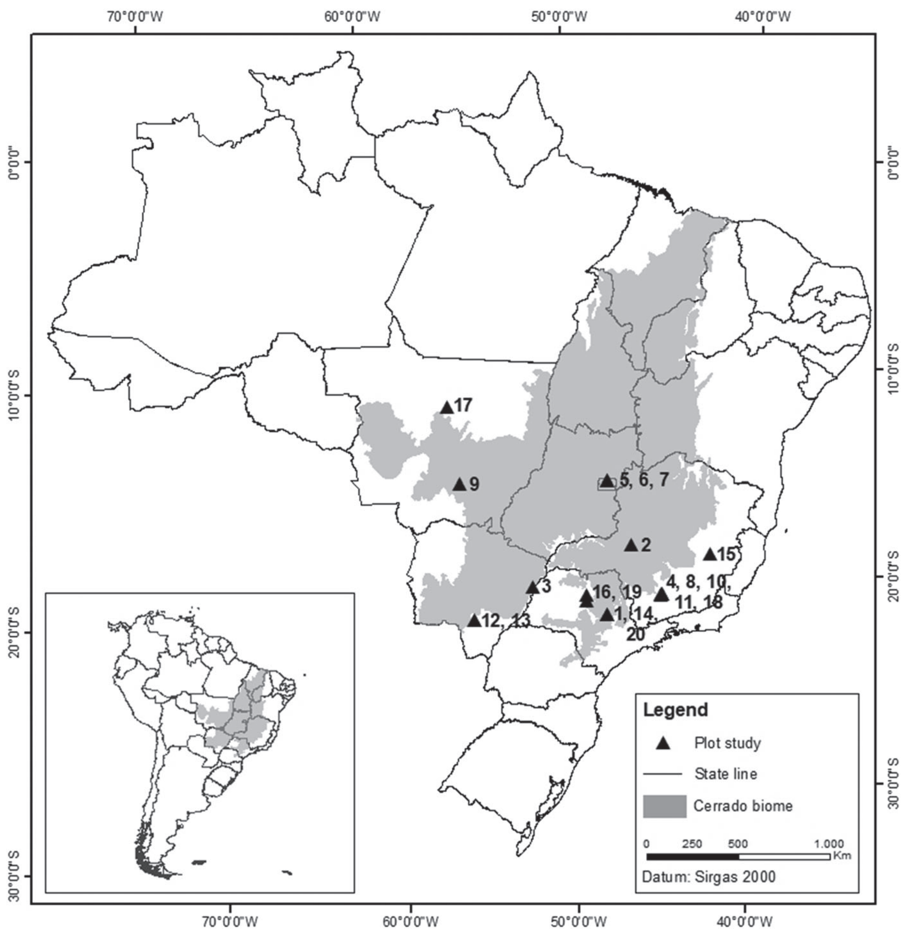

FIGURE 1 Location of the selected runoff plot studies in the Brazilian Cerrado. Some plots fell outside the shaded area because the small map scale does not show some of the biome islands

TABLE 3 Land use and cover characteristics of the runoff plots studied

Land use	Description	Land cover	Abbreviation
Cerrado (savanna)	Mixture of small tress, bushes, grasses	Permanent (P)	CE
Eucalyptus	Regular-spaced eucalyptus trees	Permanent (P)	EU
Pasture	Forage grasses	Permanent (P)	PT
Agriculture (conv.)	Grain crops, under conventional tillage	Non-permanent (NP)	AC
Agriculture (cons.)	No-till crops, sugar cane, coffee	Permanent (P)	AN
Bare soil	Fallow, bare soil	Non-permanent (NP)	BS

In order to directly compare the observed soil loss rates of the different plot studies, and to isolate the effects of soil type, land use, and land cover in the erosion process, soil losses were topographically normalized with respect to the USLE unit plot dimensions (length = 22.1 m, slope gradient = 9%) and mean rainfall erosivity (Bagarello & Ferro, 2010; Maetens et al., 2012; Rieger et al., 2015):

$$A_n = \frac{AR_m}{RLS} \quad (1)$$

where:A_n (Mg ha^{-1} year^{-1}) = normalized annual soil loss with respect to the USLE unit plot and to the mean Cerrado erosivity

A (Mg ha^{-1} year^{-1}) = observed annual soil loss of the individual plot

R_m (MJ mm ha^{-1} h^{-1}) = mean rainfall erosivity of the Cerrado biome

R (MJ mm ha^{-1} h^{-1}) = rainfall erosivity of the plot site

L = plot slope length factor

S = plot slope steepness factor

Since rainfall erosivity (R) reflects the interaction of storm volumes and rainfall intensities of the site (Wischmeier, 1966), and since the monthly rainfall distribution is uniform in the region (Chaves et al., 2004), the normalization with respect to the mean Cerrado erosivity eliminated local climatic effects, an important aspect recognized by Cerdan et al. (2006) and Maetens et al. (2012). Hence, the rainfall erosivity of each plot study was obtained using a Fournier-type equation developed by Silva (2004) for the Cerrado biome:

$$R = 12.592\sum_{i=1}^{12}\left(\frac{M_i^2}{P}\right)^{0.603} \quad (2)$$

where:R (MJ mm ha^{-1} h^{-1}) = site annual rainfall erosivityM_i (mm) = site monthly precipitation$P$ (mm) = site annual precipitation

Mean Cerrado erosivity (R_m) was simply the arithmetic mean of the rainfall erosivities of the 20 sites studied. Although the importance of rainfall intensity on soil loss is recognized (Almeida et al., 2021), the plot data of Table 1 had only monthly rainfall, hindering the analysis of rainfall intensity–soil loss relationships.

The USLE L-factor for each plot was (Bagarello et al., 2010; Wischmeier & Smith, 1978):

$$L = (l/22.13)^m \quad (3)$$

where:

L = slope length factor

l (m) = plot slope length

m (0.1–0.5) = exponent dependent on slope steepness

Finally, the USLE S-factor for each plot was simply (Wischmeier & Smith, 1978):

$$S = 0.065 + 0.0456s + 0.00654s^2 \quad (4)$$

where:

S = slope steepness factor

s (%) = plot slope

The normalization Equation (1) permitted the cancellation of plot topographic and climatic effects, allowing the soil loss variability to be a function of pedologic, land use, and land cover effects only. As in the survey of Maetens et al. (2012), prior land use and management effects of the plots were not accounted for.

To verify if plot length and steepness significantly affected runoff (Wischmeier, 1966), a correlation analysis between plot runoff and those two topographic variables was carried out but showed no statistical significance ($p > 0.05$). Hence, for comparison purposes, plot runoff was normalized with respect to annual precipitation only (Bazzoffi, 2009; Maetens et al., 2012):

$$Q_n = 100(Q/P) \quad (5)$$

where:

Q_n (%) = normalized runoff (runoff coefficient)

Q (mm) = mean annual runoff

P (mm) = mean annual precipitation

The effects of soil type, land use, and land cover on normalized runoff and soil loss were evaluated using the Tukey multiple-range test ($\alpha = 0.05$), recommended in cases where confidence intervals are desired and sample sizes are unequal (Dunnett, 2012). Hence, soil type, land use, and land cover type were taken as factors in the Tukey multiple-range test.

Additionally, since runoff plots assess the typical upslope erosion processes of laminar and rill erosion (Hudson, 1993; Mutchler et al., 1988), the normalized soil loss in each combination of soil, land use, and land cover was compared to the corresponding on-site and off-site soil loss tolerances, to evaluate the plot agronomic (Cole & Higgins, 1985) and environmental (Mullan, 2013) sustainability, respectively. On-site soil loss tolerance was obtained in the literature for each soil type, and off-site tolerance was taken as 1.0 Mg ha^{-1} year^{-1} (Mullan, 2013; Verheijen et al., 2009), since downstream impairment occurs with lower soil loss rates (Lal, 2001).

Finally, to obtain the relationship between soil loss and runoff, scatter plots and the corresponding best-fit regressions were obtained (Maetens et al., 2012; Mamedov & Levy, 2018). In the scatterplots, the influence of the cover type and the level of soil loss tolerance was also assessed (Maetens et al., 2012).

TABLE 4 Runoff plot data and metadata of the selected studies

Study	Source	Period	Soil type	Soil texture	Land use	Cover	P (mm)	R (MJ mm ha^{-1} h^{-1})
1	Anache et al. (2018)	2012–2016	Quartzipsamment	Sand	AN	P	1486	7410
					PT	P		
					CE	P		
					BS	NP		
2	Brito et al. (2005)	2003	Hapludox	Clay	CE	P	1180	7127
					EU	P		
					PT	P		
					BS	NP		
3	Cândido et al. (2014)	2012–2013	Hapludox	Clay loam	AN	P	1400	6840
					AN	P		
					CE	P		
					BS	NP		
4	Carvalho et al. (2007)	2003–2004	Hapludox	Clay loam	AN	P	1529	7711
					BS	NP		
5	Dedecek (1986)	1984	Hapludox	Clay	AC	NP	1500	6098
					AN	P		
					PT	P		
					BS	NP		
6	Dedecek (1986)	1978–1985	Hapludox	Clay	AC	NP	1243	6098
					AN	P		
					PT	P		
					BS	NP		
7	Dedecek (1986)	1979–1985	Hapludox	Clay	AC	NP	1243	7503
					AN	P		
					CE	P		
8	Dias et al. (2013)	2010–2011	Ultisol	Clay	AC	NP	1530	7960
					BS	NP		
9	Hernani et al. (1997)	1987–1994	Hapludox	Clay	AC	NP	1350	6424
					AN	P		
					BS	NP		
10	Hernani et al. (1999)	1988–1994	Hapludox	Sand	AC	NP	1350	7113
					AN	P		
					BS	NP		

TABLE 4 (Continued)

Study	Source	Period	Soil type	Soil texture	Land use	Cover	*P* (mm)	*R* (MJ mm ha^{-1} h^{-1})
11	Leite et al. (2009)	2005–2006	Hapludox	Clay	AN	P	1726	8743
					AC	NP		
12	Lima et al. (2014)	2011–2012	Ultisol	Clay	AC	NP	1529	8275
					BS	NP		
13	Lima et al. (2018)	2011–2014	Hapludox	Clay	AC	NP	1530	7761
					AN	P		
					BS	NP		
14	Oliveira et al. (2015)	2012–2014	Quartzipsamment	Sand	CE	P	1108	7725
					BS	NP		
15	Pires et al. (2006)	2002–2004	Hapludox	Clay	EU	P	1500	7545
					EU	P		
					EU	P		
					PT	P		
					CE	P		
					BS	NP		
16	Prochnow et al. (2005)	1960–1972	Ultisol	Sand	AN	P	1444	7324
					BS	NP		
17	Rieger et al. (2015)	2012–2013	Hapludox	Clay	EU	P	1974	7630
					AN	P		
					AC	NP		
					CE	P		
					PT	P		
					BS	NP		
18	Silva et al. (2005)	1998–2002	Hapludox	Clay	BS	NP	1530	7621
			Dystrudept					
19	Sosa (1987)	1980–1985	Ultisol	Sand	AC	NP	1258	7490
20	Youlton et al. (2016)	2011–2012	Quartzipsamment	Sand	AN	P	1459	7854
					PT	P		

TABLE 4 (Continued)

Study	Length (m)	Slope (%)	Area (m^2)	Q (mm)	Q_n (%)	A (mg ha^{-1} $year^{-1}$)	A_n (mg ha^{-1} $year^{-1}$)	T_{on} (mg ha^{-1} $year^{-1}$)	T_{on} source (mg ha^{-1} $year^{-1}$)
1	20.0	9.0	100.0	20.00	1.35	0.80	0.80	8.0	Corrêa et al. (2015)
				56.25	3.79	0.15	0.15		
				2.75	0.19	0.18	0.18		
				172.25	11.59	20.16	20.17		
2	4.0	26.8	96.0	1.52	0.13	0.01	0.02	11.2	Silva et al. (2002)
	14.0	17.6	336.0	11.76	1.00	0.09	0.09		
	4.0	19.4	96.0	8.23	0.70	0.40	0.42		
	4.0	17.6	96.0	22.91	1.94	1.76	1.83		
3	4.0	3.0	96.0	70.42	5.03	0.18	0.20	12.3	Bertoni and Lombardi (1990)
		4.0		1.22	0.09	0.01	0.01		
		3.0		3.12	0.22	0.00	0.01		
		3.0		124.72	8.91	0.51	0.55		
4	24.0	12	288.0	88.48	5.79	0.20	0.19	12.3	Bertoni and Lombardi (1990)
	12.0		48.0	289.69	18.95	67.24	64.64		
5	22.0	5.5	77.0	NA	NA	0.80	0.97	12.3	Bertoni and Lombardi (1990)
				NA	NA	1.10	1.34		
				NA	NA	0.01	0.01		
				NA	NA	3.83	4.66		
6	22.0	5.5	77.0	45.83	3.69	2.53	3.08	12.3	Bertoni and Lombardi (1990)
				33.62	2.70	1.01	1.23		
				3.22	0.26	0.03	0.04		
				58.68	4.72	10.53	12.80		
7	22.0	5.5	77.0	233.60	18.79	15.30	15.12	12.3	Bertoni and Lombardi (1990)
				192.00	15.45	5.50	5.43		
				15.00	1.21	0.10	0.10		
8	12.0	12	48.0	19.97	1.31	0.74	0.69	6.6	Bertoni and Lombardi (1990)
				22.60	1.48	1.91	1.78		
9	22.0	3.0	77.0	96.00	7.11	4.06	4.69	12.0	Bertoni and Lombardi (1990)
				27.00	2.00	0.78	0.90		
				149.00	11.04	7.26	8.38		
10	22.0	3.0	77.0	89.40	6.62	3.99	4.16	12.0	Bertoni and Lombardi (1990)
				19.80	1.47	0.61	0.64		
				146.30	10.84	6.90	7.19		

TABLE 4 (Continued)

Study	Length (m)	Slope (%)	Area (m^2)	Q (mm)	Q_n (%)	A (mg ha^{-1} $year^{-1}$)	A_n (mg ha^{-1} $year^{-1}$)	T_{on} (mg ha^{-1} $year^{-1}$)	T_{on} source (mg ha^{-1} $year^{-1}$)
11	22.1	9.0	77.4	NA	NA	6.18	5.24	12.3	Bertoni and Lombardi (1990)
				NA	NA	13.57	11.51		
12	22.1	12.0	88.4	NA	NA	1.46	1.31	6.6	Bertoni and Lombardi (1990)
				NA	NA	4.20	3.76		
13	12.0	12.0	48.0	93.79	6.13	3.08	2.94	10.9	Bertol and Almeida (2000)
				81.25	5.31	2.31	2.21		
				176.57	11.54	9.93	9.48		
14	20.0	9.0	100.0	2.57	0.23	0.13	0.12	8.0	Corrêa et al. (2015)
				129.60	11.70	12.40	11.90		
15	24.0	34.4	336.0	38.90	2.59	0.13	0.14	7.2	Pires et al. (2006)
		30.6		28.86	1.92	0.19	0.19		
		36.4		53.57	3.57	0.41	0.40		
		24.9	96.0	14.08	0.94	0.10	0.10		
		42.4		12.28	0.82	0.07	0.08		
		32.5		42.80	2.85	6.59	6.47		
16	50.0	10.0	1000.0	NA	NA	9.86	9.98	6.6	Bertoni and Lombardi (1990)
				NA	NA	39.50	39.98		
17	22.0	1.5	132.0	57.40	2.91	0.19	0.18	9.8	Bertoni and Lombardi (1990)
		1.5		34.50	1.75	0.23	0.22		
		1.5		48.10	2.44	0.82	0.80		
		3.0		10.30	0.52	0.02	0.02		
		1.5		47.50	2.41	0.16	0.16		
		1.5		675.30	34.21	16.80	16.32		
18	9.0	12.0	27.0	114.08	7.46	14.90	14.49	12.3	Bertoni and Lombardi (1990)
		15.0		371.04	24.25	205.65	200.04	3.67	Mannigel et al. (2002)
19	40.0	10.8	640.0	NA	NA	33.50	33.16	6.6	Bertoni and Lombardi (1990)
20	20.0	9.0	100.0	56.10	3.85	2.58	2.44	8.0	Corrêa et al. (2015)
				40.50	2.78	0.58	0.55		

R = rainfall erosivity; EU = eucalyptus; PT = pasture; CE = Cerrado; AC = conventional agriculture; AN = no-till agriculture; BS = bare soil; P = permanent cover; NP = non-permanent cover; Q = runoff; A = soil loss; T_{on} = on-site soil loss tolerance.

TABLE 5 Main statistics of the runoff plot data ($N = 50$)

Variable	P (mm)	R (MJ mm ha^{-1} h^{-1})	Length (m)	Slope (%)	Area (m^2)	Q (mm)	Q_n (%)	A (mg ha^{-1} year^{-1})	A_n	T_{on}
Mean	1443.45	7412.64	18.72	12.44	169.35	83.09	5.57	8.92	8.80	9.68
Median	1472.50	7524.00	21.00	9.00	96.00	46.66	2.81	1.01	1.23	10.90
Max	1974.00	8743.00	50.00	42.40	1000.00	675.30	34.21	205.65	200.04	12.30
Min	1108.00	6098.00	4.00	1.50	27.00	1.22	0.09	0.00	0.01	3.67
CV	0.14	0.09	0.58	0.90	1.29	1.39	1.23	3.13	3.09	0.28

P = annual precipitation; R = annual erosivity; Q = annual runoff; Q_n = normalized runoff; A = annual soil loss; A_n = normalized soil loss; T_{on} = on-site soil loss tolerance; CV = coefficient of variation.

3 | RESULTS AND DISCUSSION

3.1 | Cerrado runoff plot data

Table 4 presents the global data and meta-data of the 20 surveyed studies ($N = 50$ plots). It also includes the rainfall erosivity, normalized runoff and soil loss, and the corresponding on-site soil loss tolerances, estimated in the present study.

The statistics of the observed and normalized runoff, and soil loss, as well as the estimated on-site tolerance of the 50 plots, are presented in Table 5. Mean annual precipitation was 1443.5 mm, and mean annual runoff was 83.1 mm, equivalent to 5.8% of mean annual rainfall (runoff coefficient).

In Table 5, observed and normalized soil loss means were practically identical (i.e. 8.9 and 8.8 Mg ha^{-1} year^{-1}, respectively). The coefficients of variation of both runoff and soil loss were above 100%, reflecting the high variability in the environmental and management conditions of the plots.

The runoff and soil loss means in Table 5 were 50 and 74% lower than those obtained by Anache et al. (2017) in runoff plots of southeastern Brazil, respectively. Since the southeastern and Cerrado soils are similar, the soil loss means of the latter were lower because of the reduced rainfall erosivity and because the Cerrado studies had more plots under permanent cover than those of Anache et al. (2017), providing higher infiltrability and greater protection against erosion (Zuazo & Pleguezuelo, 2008).

Compared to the Mediterranean survey of Maetens et al. (2012), the runoff and soil loss means of Table 4 were 45 and 20% lower, respectively, probably because of the higher infiltrability of the Cerrado soils (Bono et al., 2012) compared with the Mediterranean plots.

Considering that the soil loss means of Table 5 were obtained from upslope erosion plots, where laminar and rill processes are dominant (Mutchler et al., 1988), they shall be viewed with caution and should not be extrapolated across the landscape (Boardman & Evans, 2019). However, inter-plot and treatment comparisons are allowed (Hudson, 1993), provided that the boundary conditions are met and appropriate normalization is performed (Maetens et al., 2012).

3.2 | Effects of soil type, land use, and land cover

Tables 6, 7, and 8 present the effects of soil type, land use, and land cover on normalized runoff and soil loss, respectively.

TABLE 6 Normalized runoff (Q_n) and normalized soil loss (A_n) means as a function of soil type, with the corresponding Tukey classes ($\alpha = 0.05$), and the on (T_{on}) and off-site (T_{off}) soil loss tolerances. Means in italics exceeded on and/or off-site tolerances

Soil type	Q_n (%)	A_n (mg ha^{-1} year^{-1})	T_{on}	T_{off}
Dystrudept	24.25a	*141.17a*	3.67	1.0
Hapludox	5.54b	*9.25b*	11.06	1.0
Quartzipsamment	4.43b	*4.77b*	8.00	1.0
Ultisol	1.39b	*7.48b*	6.60	1.0

Note: *a* and *b* denote the statistical levels of the Tukey test means; that is, means followed by different letters are significantly different at 95% probability. Conversely, means followed by the same letter are not statistically different.

TABLE 7 Normalized runoff (Q_n) and normalized soil loss (A_n) means as a function of land use, with the corresponding Tukey classes ($\alpha = 0.05$) and the on (T_{on}) and off-site (T_{off}) soil loss tolerances. Means in italics exceeded on and/or off-site tolerances

Land use	Q_n (%)	A_n (mg ha^{-1} year^{-1})	T_{on}	T_{off}
Bare soil	11.53a	*24.97a*	9.65	1.0
Agriculture (conv.)	6.58ab	*7.13a*	10.34	1.0
Agriculture (cons.)	4.07b	*2.20a*	10.96	1.0
Eucalyptus	2.40b	0.20a	8.52	1.0
Pasture	1.81b	0.20a	9.83	1.0
Cerrado	0.47b	0.08a	9.83	1.0

Note: *a* and *b* denote the statistical levels of the Tukey test means; that is, means followed by different letters are significantly different at 95% probability. Conversely, means followed by the same letter are not statistically different.

TABLE 8 Normalized runoff (Q_n) and normalized soil loss (A_n) means as a function of land cover, with the corresponding Tukey classes ($\alpha = 0.05$) and the on (T_{on}) and off-site (T_{off}) soil loss tolerances. Means in italics exceeded on and/or off-site tolerances

Land cover	Q_n (%)	A_n (mg ha^{-1} year^{-1})	T_{on}	T_{off}
Non-permanent	9.88a	*17.96a*	10.11	1.0
Permanent	2.45b	*1.02b*	9.92	1.0

Note: *a* and *b* denote the statistical levels of the Tukey test means; that is, means followed by different letters are significantly different at 95% probability. Conversely, means followed by the same letter are not statistically different.

In Table 6, runoff and soil loss of the Dystrudept soil (Inceptisol) were statistically higher than those of the other soils. This occurred because the Dystrudept was the only soil in the study with only one cover type (bare plot), resulting in a high erosion rate. Additionally, the shallow depth of the Dystrudept soil reduces water infiltration and increases soil erodibility (Silva et al., 2009), resulting in a high erosion rate.

Because of these pedologic limitations, the Dystrudept is grouped as Class V soil in the US land capability system (NRCS, 1961), making it unsuitable for agriculture and other intensive land uses (Severiano et al., 2009). The other three soils in Table 6 presented statistically similar runoff and soil loss, reflecting the high variability of their different land uses and covers, as shown in Table 3.

Table 7 shows that the highest rates of runoff and soil loss were observed in the bare plots, which decreased as the degree of land cover increased. A similar trend was observed by Anache et al. (2017) and Rieger et al. (2015) in plots of southeastern Brazil.

In Table 7, runoff and soil loss under conservation agriculture were 38 and 70% lower than those under conventional agriculture, respectively, confirming the importance of permanent cover and reduced tillage in lowering runoff and erosion (Castro et al., 1999).

In Table 8, runoff and soil loss means in the permanently covered plots were 75 and 95% lower than in the non-permanent plots, respectively. Similar reductions were found by Anache et al. (2017) and Rieger et al. (2015) in other regions in Brazil. The highly significant differences in the means of Table 8 highlight the importance of maintaining a permanent soil cover, regardless of the land use type (Hofmann et al., 1983; Zuazo & Pleguezuelo, 2008).

Table 8 also indicates that the soil loss mean of non-permanent plots exceeded both on and off-site tolerances, while that of the permanent plots did not, indicating that permanent cover allows for sustainable and stable systems (Hobbs et al., 2008).

3.3 | Runoff–soil loss relationships

Figure 2 shows the relationship between normalized runoff and soil loss in the 20 studies ($N = 50$), and Figure 3 presents the same relationship, with observed runoff and soil loss values. In both figures, the power function provided the best fit between soil loss and runoff, a fact also observed at the laboratory (Mamedov & Levy, 2018) and plot (Maetens et al., 2012) scales. The regressions of both Figures 2 and 3 were statistically significant ($p < 0.001$).

Though the power equation coefficients differ in the Mediterranean (Maetens et al., 2012) and Cerrado (Figure 2) cases, reflecting their specific climate and environmental conditions, the same type of function (power) points to a general runoff/erosion relationship, common in river sediment transport studies (Yang, 1977).

The exponential increase in soil erosion observed in the uncovered plots (triangles in Figures 2 and 3) could be explained by rilling (Mamedov & Levy, 2018), although the plot studies did not provide information about this phenomenon. According to Yao et al. (2008), rilling occurs when threshold values of flow velocity and shear stress are exceeded, generally in non-permanent cover conditions, where the soil surface is unprotected against runoff (Seutloali & Beckendahl, 2015). This includes open patches of farm fields and rangelands, which act as a source of water and sediments (Cantón et al., 2011).

Conversely, permanently covered plots (circles in Figures 2 and 3) were associated with lower runoff and soil loss, about one order of magnitude lower than the rates observed in the uncovered plots (Table 8). Although the data did not describe the dominant types of erosion occurring in the plots, the higher surface roughness and infiltration, and lower runoff volumes observed in permanently covered plots (Cogo et al., 1983), reduce the chances of rilling, a more intensive erosion process than sheet erosion (Cerdan et al., 2006).

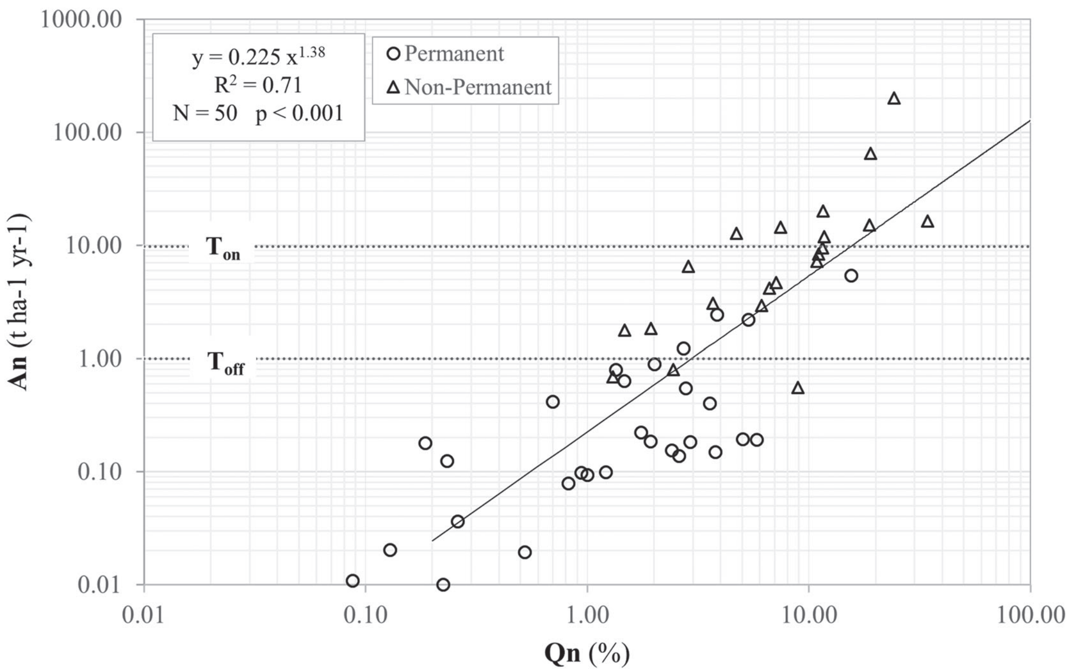

FIGURE 2 Normalized soil loss (A_n) vs normalized runoff (Q_n), showing the permanent and non-permanent land covers. The dotted lines are the mean on and off-site soil loss tolerances

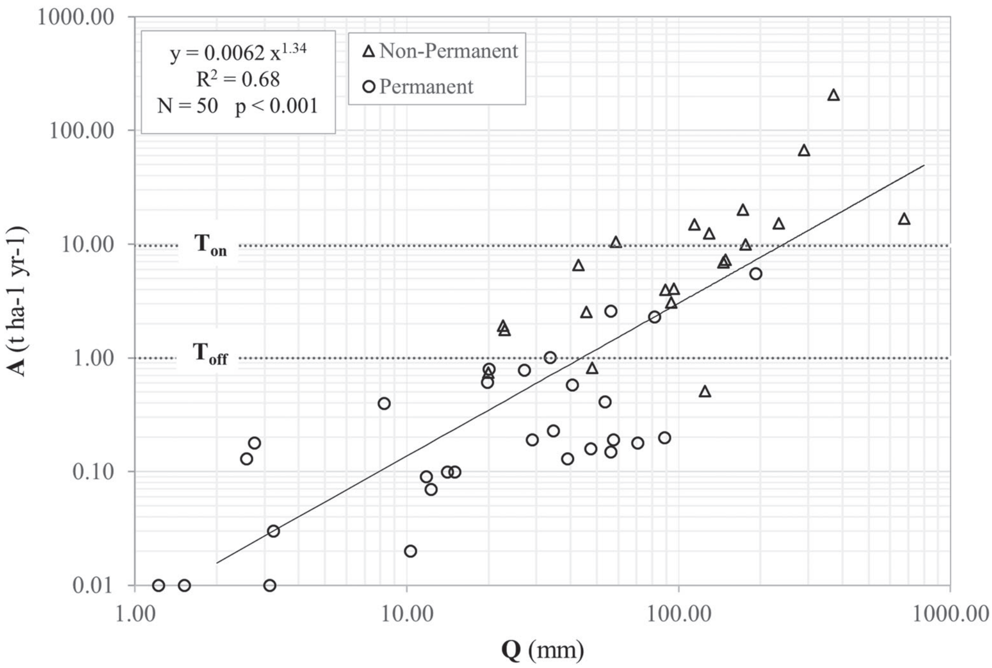

FIGURE 3 Observed soil loss (A) vs observed runoff (Q), showing the permanent and non-permanent land covers. The dotted lines are the mean on and off-site soil loss tolerances

The good fits ($p < 0.001$) obtained in Figures 2 and 3 also allow the utilization of runoff, easily estimated by rainfall–runoff models (e.g. NRCS, 2004) as predictor of annual soil loss, which is difficult to estimate (Gajbhiye et al., 2014). Such relationships facilitate the establishment of soil and water conservation strategies for the Cerrado biome, particularly in data-scarce areas (Chaves, 2010).

3.4 | Implications with respect to soil loss tolerances

In Tables 6, 7, and 8, soil loss means shown in italics exceeded either on or off-site tolerances, indicating situations with potential on (Cole & Higgins, 1985) and off-site (Mullan, 2013) degradation. Table 7 indicates that, on average, erosion in land uses with permanent vegetative cover (pasture, eucalyptus, and Cerrado) did not exceed on and off-site soil loss tolerances (A_n < 1.0 Mg ha^{-1} year^{-1}), a fact previously observed by Li et al. (2009).

The importance of permanent soil cover is also observed in Table 8. Mean normalized soil loss in the permanently covered plots (1.02 Mg ha^{-1} year^{-1}) barely exceeded the off-site tolerance threshold (1.0 Mg ha^{-1} year^{-1}). Furthermore, Figures 2 and 3 indicate that none of the permanently covered plots exceeded mean on-site tolerance (9.7 Mg ha^{-1} year^{-1}), and only four permanently covered plots surpassed off-site tolerance (1.0 t ha^{-1} year^{-1}). This conclusion has important policy implications for the Cerrado biome (i.e. that land uses shall maintain a permanent cover throughout the year, if on and off-site sustainability is sought).

4 | CONCLUSIONS

The results indicate that the highest rates of runoff and soil loss were associated with soils without permanent cover (conventional agriculture and bare soil). These findings are supported by other studies, in Brazil and elsewhere. In permanently covered plots, runoff and soil loss means were 75 and 95% lower than those plots with non-permanent cover, respectively.

A statistically significant power function was obtained between normalized and non-normalized soil loss and runoff. Similar power functions were obtained in the laboratory and in European runoff plots, suggesting that the erosion process in the Cerrado biome follows a universal trend.

On and off-site soil loss tolerances were exceeded in plots with non-permanent covers, but not in permanently covered plots, reinforcing the importance of maintaining a permanent soil cover in the Cerrado biome.

The results and relationships obtained in the present study allow for the establishment of soil conservation policies and strategies in the Cerrado region and in other savanna-type biomes.

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DATA AVAILABILITY STATEMENT

Data are available on request from the authors.

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Capítulo 2

Artigo sobre Impactos O*n-site* da Restauração Ambiental de Área de Carste nos Cerrados

Author's Decision

Thank you for approving "Runoff, Soil Loss, and Water Balance in a Restored Karst Area of the Brazilian Savanna".

Catena

Runoff, Soil Loss, and Water Balance in a Restored Karst Area of the Brazilian Savanna

--Manuscript Draft--

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Abstract:	In karst landscapes, on-site impacts, such as soil erosion, occur due to the removal of the natural vegetation cover. Ecological restoration can slow and even revert that degradation process. Runoff plots with different soils (Lithic entisol and Quartzipsamment) and soil covers (bare soil, restored areas with native species, and undisturbed savanna) were installed in a degraded karst area of the Brazilian savanna, where runoff, erosion, and soil water balance were monitored during 3 hydrologic years. Mean runoff and erosion in the ecologically restored areas (Lithic entisol) decreased from 546 mm yr $^{-1}$ to 360 mm yr $^{-1}$, and from 34 Mg ha $^{-1}$ yr $^{-1}$ to 5 Mg ha $^{-1}$ yr $^{-1}$, respectively. In the case of the undisturbed Cerrado (Quartzipsamment), mean water and soil losses were constant during the 3 years, namely 16 mm yr $^{-1}$ and 0.7 Mg ha year $^{-1}$, respectively. Soil loss tolerance was exceeded the bare plot of the Lithic entisol in all three hydrologic years. In the third year, erosion fell below soil loss tolerance in the restored plots, indicating that soil degradation could be reduced with ecological restoration. The latter improved the soil water balance, increasing infiltration and evapotranspiration, and reducing runoff, indicating that ecological restoration of karst areas could generate significant on- and off-site hydrologic services.
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Title

Runoff, Soil Loss, and Water Balance in a Restored Karst Area of the Brazilian Savanna

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- Karst areas are vulnerable to soil erosion and degradation
- No previous studies existed in degraded karst areas of the Brazilian savanna
- Ecological restoration significantly reduced erosion and runoff
- Erosion was reduced below the soil loss tolerance threshold

Runoff, Soil Loss, and Water Balance in a Restored Karst Area of the Brazilian Savanna

Abstract

In karst landscapes, on-site impacts, such as soil erosion, occur due to the removal of the natural vegetation cover. Ecological restoration can slow and even revert that degradation process. Runoff plots with different soils (Lithic entisol and Quartzipsamment) and soil covers (bare soil, restored areas with native species, and undisturbed savanna) were installed in a degraded karst area of the Brazilian savanna, where runoff, erosion, and soil water balance were monitored during 3 hydrologic years. Mean runoff and erosion in the ecologically restored areas (Lithic entisol) decreased from 546 mm yr^{-1} to 360 mm yr^{-1}, and from 34 Mg $ha^{-1}yr^{-1}$ to 5 Mg $ha^{-1}yr^{-1}$, respectively. In the case of the undisturbed Cerrado (Quartzipsamment), mean water and soil losses were constant during the 3 years, namely 16 mm yr^{-1} and 0.7 Mg ha $year^{-1}$, respectively. Soil loss tolerance was exceeded the bare plot of the Lithic entisol in all three hydrologic years. In the third year, erosion fell below soil loss tolerance in the restored plots, indicating that soil degradation could be reduced with ecological restoration. The latter improved the soil water balance, increasing infiltration and evapotranspiration, and reducing runoff, indicating that ecological restoration of karst areas could generate significant on- and off-site hydrologic services.

Keywords: *soil and water losses, karst areas, cerrado, ecological restoration.*

1. INTRODUCTION

Karst landscapes are highly fragile environments, representing approximately 12% of all continents (Febles-González et al., 2012). Among the threats that affect karst areas are soil erosion and landscape degradation (Hu et al., 2018; Parise et al., 2009).

Among the impacts are selective erosion (Jacinthe et al., 2004; Nie et al., 2015; Wacha et al., 2020), sedimentation and water quality impairment (Coxon, 1999; Goldscheider and Drew, 2007; Parise et al., 2009), which can lead to rocky desertification in semi-arid and arid regions (Jiang et al., 2014; Zhao and Hou, 2019).

In karst environments, degradation occurs through processes of surface and underground erosion, chemical weathering and gravity, respectively (Zeng et al., 2018). Sinkholes absorb surface runoff from upstream areas, becoming the main path for large volumes of water entering cave systems and underground rivers (Febles-González et al., 2012). Sinkholes allow the rapid transmission of pollutants from the surface to underground systems (White, 1988), and are the main route by which sediment is transported to the subsurface (Febles-González et al., 2012).

The slow rate of pedogenesis and the small soil depth are factors that reduce the soil erosion tolerance of karst soils (Wang et al., 1999; Zhao and Hou, 2019). In addition, their low soil permeability and the high erodibility increase the erosion risk (Gan et al., 2002; Zhao and Hou, 2019).

Whenever erosion rates surpass soil loss tolerance, soil degradation occurs (Chaves, 2010b). Assessing the erosion process in karst areas of southeastern China, Peng and Wang (2012) found soil losses of 3.58 Mg ha^{-1} $year^{-1}$ and tolerances ranging from 3.0 to 6.8 Mg $ha^{-1}year^{-1}$, indicating a strong potential for soil degradation. In karst-derived soils in Cuba, Febles-González et al. (2012) found soil losses of 13 Mg $ha^{-1}year^{-1}$, well above the local tolerance levels.

On the other hand, undisturbed vegetation cover is able to prevent erosion of karst soils (Bruijnzeel, 2004; Fu, 2009), reducing surface runoff (Brown et al., 2007; Kosmas et al., 1997; Peng and Wang, 2012), and maintaining the physical and chemical properties of the soil (Kosmas et al., 2000).

It has long been known that vegetation cover protects the soil against erosion, in both agricultural (Laflen et al., 1978) and in natural areas (Zuazo and Pleguezuelo, 2008). While the plant canopy reduces the impact of the raindrops (Ma et al., 2014), ground cover and surface roots decrease runoff velocity and sediment transport (Hofmann et al., 1983). In a recent meta-analysis of the Cerrado biome, Fonseca et al. (2021) reported that erosion rates in areas with permanent vegetation cover remained below on- and off-site soil loss tolerances.

Furthermore, ecological restoration of degraded agricultural areas can improve infiltration and groundwater recharge (Chaves et al., 2012). Although there are studies addressing the relationship between permanent vegetation and hydrological processes in the Brazilian savannah (Fonseca et al., 2021), the effectiveness of ecological restoration against erosion has not been assessed (Honda and Durigan, 2017).

Additionally, there are no studies about soil erosion in karst areas of the Brazilian savanna. In non-karst areas of this biome, Fonseca et al (2021) reported erosion and runoff means of 8.9 Mg ha^{-1} yr^{-1} and 83.1 mm yr^{-1}, respectively. In bare runoff plots of the Cerrado biome, Anache et al. (2019)

reported surface runoff volumes four times greater than that of natural vegetation.

Oliveira et al. (2015) concluded that deforestation in the Cerrado has the potential to increase surface runoff and reduce aquifer recharge, decreasing evapotranspiration at local and regional scales. Furthermore, extensive land-use conversion in the Brazilian savanna can ultimately influence the regional climate. Hoffmann and Jackson (2000) predicted that the conversion of the Brazilian savanna to pastureland would reduce the biome's rainfall and streamflow by 10%. Since karst areas of the Brazilian Cerrado are more vulnerable than non-karst areas, it is expected that the former are being degraded faster than the latter.

Considering the aspects above, the objective of this study was to assess the hydrologic and sedimentologic behavior of degraded and restored plots of a karst area of the Brazilian savanna. The specific objectives of the research were: a) To obtain the water and soil losses of runoff plots under different soil covers; b) To compare the measured erosion rates with the corresponding soil loss tolerances; c) To obtain the soil water balance for the different land-cover treatments; and d) To calculate the surface runoff coefficient (CN) and the C factor (USLE) of the undisturbed and restored vegetation covers.

2 METHODOLOGY

2.1 Study Site

The study site is a karst area located in the municipality of Mambaí (Brazil), within the Corrente River watershed and the Environmental Protection Area of the Vermelho River (APA-NRV). The experiment was located on a karst slope, within the Tarimba sinkhole, which receives the runoff and sediment generated upstream (Figure 1).

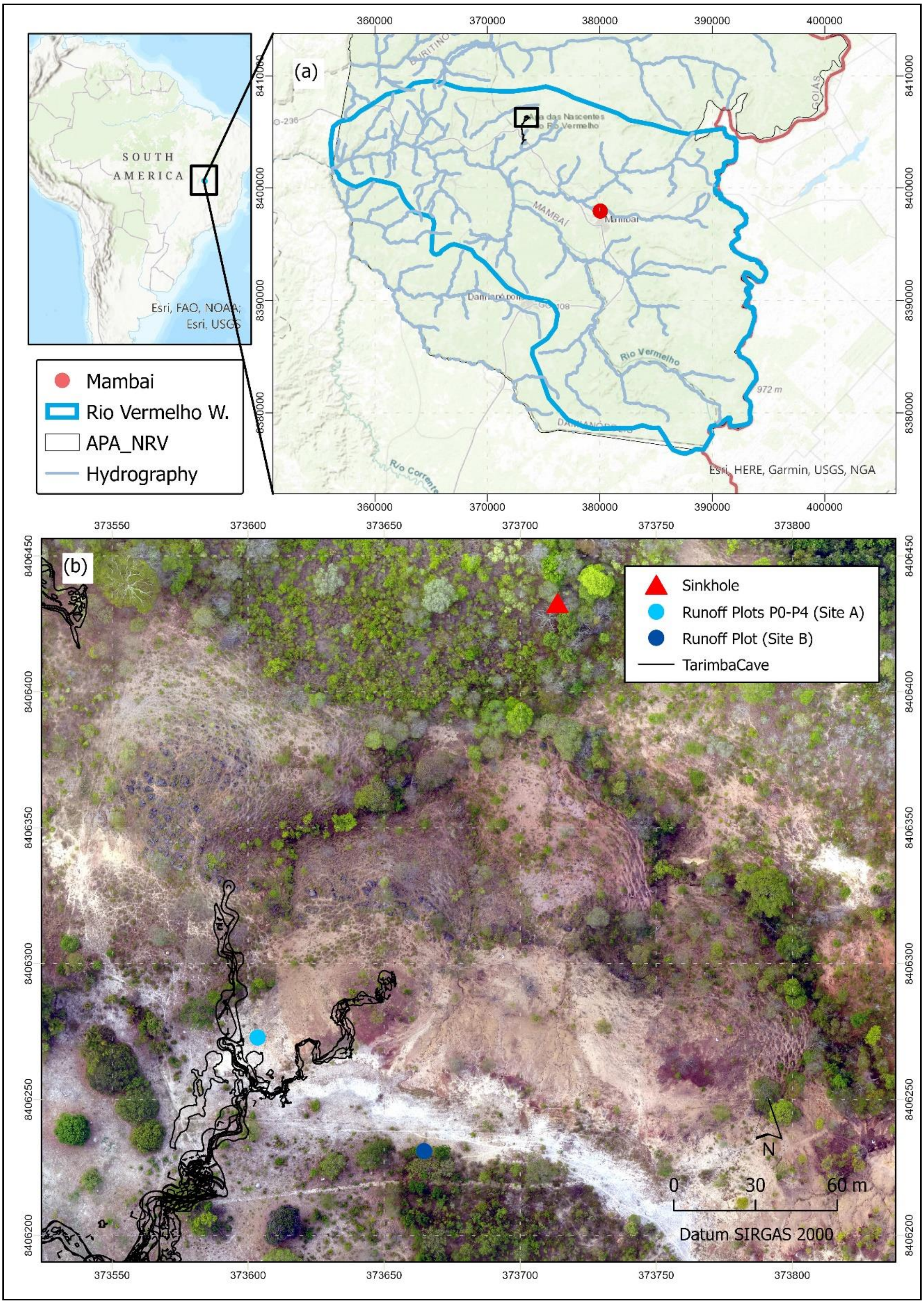

Figure 1 – Location of the study site: (a) Rio Vermelho Watershed, (b) Tarimba cave and sinkhole, and runoff plots.

The Tarimba cave system is considered the most important cave formation in the region and one of the longest in the country (Hussain et al., 2020). The climate in the region is tropical humid (Aw-Koppen), with a savanna climate subtype, with dry winters and wet summers (Silva et al., 2008). Mean annual precipitation is 1,200 mm and mean annual temperature is 25°C.

In its upper portion, the study area is composed by the Urucuia geologic formation (sandstones with unconsolidated siliciclastic sediments), and in the lower portion by the Paranã Valley (pelitic rocks interspersed with carbonates) of the Bambuí geologic group (Tavares et al., 2021).

In the lower part of the study slope, the carbonates and calcitic limestones are overlaid by pelites, and covered by Alfisols and Lithic entisols. In the upper part of the slope there are the sandy debris from the Urucuia Group, covered by Quartzipsamments (Caldeira et al., 2021; Gaspar and Campos, 2007; Uagoda et al., 2019).

In the upper part of the study site, a typical savanna vegetation predominates, associated with the sandstone and with the Quartzipsamment. In the lower portion of the slope, a seasonal forest predominates, situated above the limestone and the Lithic entisol.

The study was carried between October 2018 and September 2021 (36 months), comprising of three full hydrologic years. Two experimental sites, representing the local geologic and soil conditions, were established: i) *Site A* (Lithic entisol), in a degraded pasture area, with five runoff plots; and ii) *Site B* (Quartzipsamment), in an area of undisturbed savannah, with one plot (Figure 1).

2.2 Runoff plots

All six runoff plots had a linear topographic profile, and an average slope of 11%. Plot dimensions were 22.1 m (L) and 1.80 m (W), with an area of 39.8 m². To prevent upstream runoff from entering the plots, they were delimited by galvanized steel boards, inserted into the soil (Youlton et al., 2016). The lower end of each plot had a metal collection trough and a PVC pipe, and the runoff and sediment were collected in two 1,000 L tanks, positioned in tandem (Figure 2).

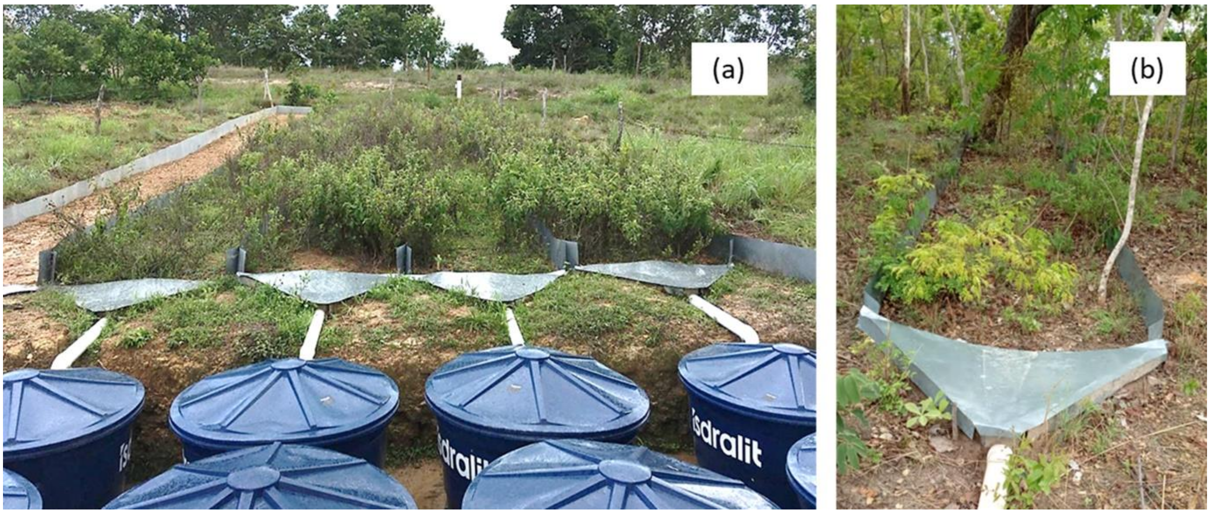

Figure 2 – Runoff plots. (a) Site A – Plots P0, P1, P2, P3, P4 and (b) Site B – Plot P5. December, 2019.

The ground cover types of the six runoff plots were: a) Bare soil (P0); b) Restored native vegetation cover, without jute mat (P1 and P3); c) Restored native vegetation cover, with jute mat (P2 and P4); and d) Undisturbed Cerrado (savanna) vegetation (P5).

The P0-P4 plots were installed over the Lithic entisol (Site A), and the P5 plot was installed over the Quartzipsamment (Site B) (Table 1). The uncovered plot (P0) was kept bare during all 36 months of the experiment, with periodic application of systemic herbicide (glyphosate) (Brito, 2005).

Table 1 – Soil characteristics of Sites A (P0-P4) and B (P5).

Soil type	**Depht** (cm)	**Texture (%)**			**Class**	**NRCS Hydrologic Group**
		Sand	**Silt**	**Clay**		
Lithic entisol *(Site A)*	15	63.83	31.46	4.71	Sandy-loam	C
	40	41.28	50.25	8.47	Loam	
	70	25.34	66.13	8.53	Silty-loam	
Quartzipsamment *(Site B)*	20	93.04	6.07	0.89	Sand	A
	40	93.87	5.15	0.98	Sand	
	70	92.77	6.11	1.12	Sand	

The P1-P4 plots were restored with native species in September of 2018 (Table 2), receiving the following restoration sequence: i) Micro-drilling (Couto et al., 2010), with 30 holes/m²; ii) Green manure (*Stylosanthes sp*. and *Cajanus cajan*; and iii) Direct planting of seedlings of different savanna species (Farias et al., 2013).

Table 2 - Native species planted in the restored plots (P1-P4).

Trees	**Shrubs**	**Grass**
Pau Santo (*Kielmeyera speciosa*)	Assa Peixe (*Vernonanthura sp.*)	*Andropogon fastigiatus*
Ipê Caraíba Amarelo (*Tabebuia aurea*)	Fedegozinho (*Senna sp.*)	
Jacarandá Bico de Paragaio (*Machaerium acutifolium*)	Copaibinha (*Copaifera sp.*)	
Guatambu do Cerrado (*Aspidosperma macrocarpum*)	Amargoso (*Lepdaploa aurea*)	
Tinguí (*Magonia pubecens*)		
Aroeira Preta (*Myracrodruon urundeuva*)		

Amburana (*Amburana cearenses*)		

The amargoso grass (*Lepdaploa aurea)* and the stylosanthes (*Stylosanthes sp)* were the dominant species in the P1-P4 plots after the 3 experimental years, although individuals of all species introduced in 2018 were observed in the plots at the end of the experiment. The soil cover in the restored plots (P1-P4), measured by the Cline (1944) method, increased from practically 0%, in the beginning of the first year, to 65% at the end of the 3rd year (Figure 3).

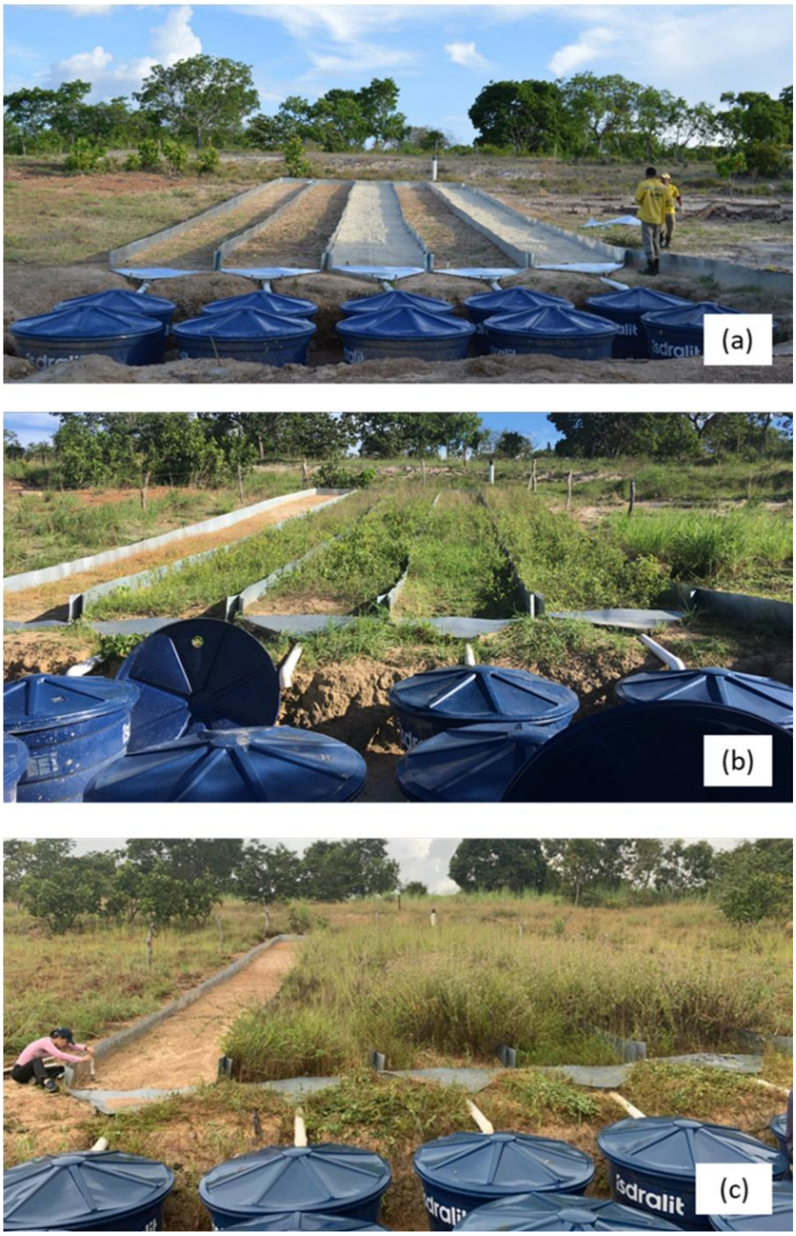

Figure 3 – Runoff plots in different years. (a) 10/30/2018, (b) 12/20/2019, and (c) 04/04/2021.

Plot P5 was installed over an area of undisturbed natural savanna, with 50% tree cover (3 to 8 m high) and the remaining 50% covered with shrubs and grasses. Considering that plots P1-P3 (vegetal restoration, without jute mat), and plots P2-P4 (vegetal restoration, with jute mat) were replicated, their statistical treatments were grouped.

2.3 Precipitation, surface runoff, and soil loss

Precipitation, measured on a local rain gage, runoff, and soil loss were measured biweekly, except in situations of significant rainfall events, when the frequency increased (Carvalho et al., 2007; Cogo, 1978). Thus, 15 hydrological measurements were made in the 1st year, 18 in the 2nd year, and 19 in the 3rd year.

In the case of runoff, the volume was obtained by measuring the water level of the water collection tanks (Bagarello and Ferro, 2004). The unit surface runoff volume in each plot was simply the ratio between the total volume in the two tanks and the total area of the plot:

$$Q = V / A_p \tag{1}$$

where: Q (mm) = unit volume of runoff in the period; V (L) = runoff volume in the period (L); and A_p (m^2) = plot area.

In the case of soil loss, vertically-integrated water samples were obtained in both tanks after agitation with a rod (Bagarello and Ferro, 2004). In the laboratory, the 1 L water + sediment samples, after being decanted with aluminum sulfate and oven-dried at 105°C for 24 h, were weighed. The soil loss in the plot was obtained by the product of the sediment concentration in the bottles and the runoff volume in the collection tanks (Anache et al., 2018; Oliveira et al., 2015):

$$L_s = C\,V / A_p \quad (2)$$

where: L_s (Mg ha^{-1}) = plot soil loss in the period; C (g L^{-1}) = sediment concentration in the sample; V (L) = runoff volume in the collection tanks; and A_p (ha) = plot area.

In order to eliminate the influence of precipitation in the three experimental years and to isolate the restoration treatment effects, annual runoff and soil loss were normalized by the annual precipitation and annual rainfall erosivity, respectively. Thus, the normalized surface runoff in each plot was:

$$Q_n = 100\,Q / P \quad (3)$$

where: Q_n = normalized runoff (%), Q = annual runoff (mm year^{-1}), P = annual precipitation (mm). In the case of soil loss, the annual normalized value was (Fonseca et al., 2021):

$$A_n = 100\,A / R \quad (4)$$

where: A_n = normalized soil loss (t yr^{-1}MJ^{-1}mm^{-1}h), A = soil loss (Mg ha-1yr-1), R = rainfall erosivity (MJ mm ha^{-1} h^{-1}).

The rainfall erosivity of each hydrological year was obtained using a regional Fournier-type equation, based on the monthly and annual rainfall (Silva, 2004):

$$R = 12{,}592\ \sum_{i=1}^{12}\left(M_i^2/P\right)^{0.603} \quad (5)$$

where: R = Annual rainfall erosivity (MJ mm ha^{-1}hr^{-1}), M_i = monthly rainfall (mm), P = annual rainfall (mm).

The effect of vegetation cover on erosion in the restored plots (P1-P4) was assessed by the USLE C factor (Wischmeier and Smith, 1978), namely:

$$C_i = A_i / A_0 \quad (6)$$

where: C_i = USLE C factor (0-1) for cover i; A_i = annual average soil loss in the plot under cover i (Mg ha year^{-1}); A_0 = annual average soil loss in the bare plot (Mg ha year^{-1}).

The annual erosion rate in the runoff plots were compared to the corresponding soil loss tolerance, to assess the degree of permanent degradation by erosion (Chaves, 2010b). Soil loss tolerance values of the Lithic entisol and of the Quartzipsamment were obtained from Bertoni and Lombardi Neto (1991).

2.4 Runoff Coefficient

The CN coefficient for each plot, in the period between two consecutive samplings, was obtained through the iterative solution of the following equation (NRCS, 2004):

$$Q = (P - 0.2\ S)^2/(P + 0.8\ S) \quad (7)$$

where

$$S = (25.400/CN) - 254 \quad (8)$$

where: Q = surface runoff volume (mm), P = precipitation volume (mm), S = abstraction factor (dimensionless), CN = curve-number (0-100, dimensionless). Condition II of antecedent soil moisture (NRCS, 2004) was assumed for the entire period, since this is the condition that most adapts to the Brazilian savanna (Chaves et al., 2012; Chaves and Piau, 2008)

2.5 Soil Water Balance

The soil water balance was carried in each plot, based on measured precipitation, surface runoff, and soil water retention data. The water balance period was bi-weekly, between two successive precipitation and runoff samplings. The soil water balance was obtained through the following equation (Brooks et al., 2003):

$$P - (Q + ETr + Qp) - (ISM - FSM) = 0 \quad (9)$$

where: P= precipitation volume between measurements (mm); Q = surface runoff between measurements (mm); ETr = actual evapotranspiration (mm); ISM = initial soil moisture (mm); FSM = final soil moisture (mm); Qp = deep percolation (mm). The actual evapotranspiration was estimated by the following equation (Brooks et al., 2003):

$$ETr = \min\,(ETo * Kc;\ P + ISM) \quad (10)$$

where: ETr = actual evapotranspiration (mm); ETo = potential evapotranspiration (mm); Kc = crop coefficient; P = precipitation in the period (mm); and ISM = initial soil moisture (mm). Potential evaporation was calculated using the of Hargreaves and Samani (1985) equation for each period between successive measurements:

$$ETo = N * 0.0023 * Rg * (Tmed + 17.8) * (Tmax - Tmin)\text{^}1/2 \quad (11)$$

where: ETo = potential evapotranspiration in the period between collections (mm); N = number of days of the month; R_g = mean solar radiation (mm day^{-1}); T_{med} = mean temperature (°C); T_{max} = maximum temperature (°C); T_{min} = minimum temperature (°C). The climatological data for the ETo calculation were obtained from the Posse INMET station, located 10 km from the study area.

The values of the K_c crop coefficient were taken as 0.5 for the uncovered plot (P0) (Allen et al., 2005), and 1.0 for the vegetated plots (P1-P5) (Wight and Hanson, 1990).

The initial soil moisture (ISM) at the beginning of the hydrological year (October) was obtained from measurements with undisturbed samples of the two soils, at a depth between 0 and 60 cm, corresponding to approximately ¼ of the value of the water retention capacities. The final soil moisture (FSM) was obtained by the following equation (Brooks et al., 2003):

$$FSM = \min\left[(P+SRC-Q-Etr); SRC*Dt\right] \quad (12)$$

where: SRC (mm m^{-1}) = soil water retention capacity; and Dt (m) = mean root depth. The SRC value for the two soils was obtained by water retention curves in the laboratory, from undisturbed samples taken in depths of 0-60 cm (Gardner, 1986):

$$SRC = Fc - Wp \quad (13)$$

where: SRC = soil water retention capacity (mm m^{-1}); Fc = water content at field capacity (0.33 bar); and Wt = water content at wilting point (15 bar), both obtained in a pressure pan, in the laboratory (Bernardo et al., 1987). Finally, deep percolation in the plots was calculated by the following equation (Brooks et al., 2003):

$$Q_p = P + ISM - Q - ETr - FSM \quad (14)$$

3 RESULTS AND DISCUSSION

3.1 Precipitation and surface runoff

The rainfall volumes and rainfall erosivities of the three hydrological years are presented in Table 3. Except for the precipitation of Year 3, which was below the regional mean (Silva et al. 2008; Fonseca et al. 2021), the precipitation volumes of the other years were within the average. Rainfall erosivity in the three hydrologic years followed the same trend observed in precipitation.

Table 3 - Precipitation and rainfall erosivity in the three hydrological years.

Hydrological year	Precipitation (mm)	Erosivity (MJ mm $ha^{-1}hr^{-1}$)
2018-2019	1,223.6	7,440.0
2019-2020	1,295.1	7,746.0
2020-2021	935.5	6,694.1

Figures 4 and 5 show the mean runoff volumes (Q) and mean calibrated runoff coefficients (CN), in the three hydrological years, and Figure 5 shows plot runoff normalized by annual precipitation.

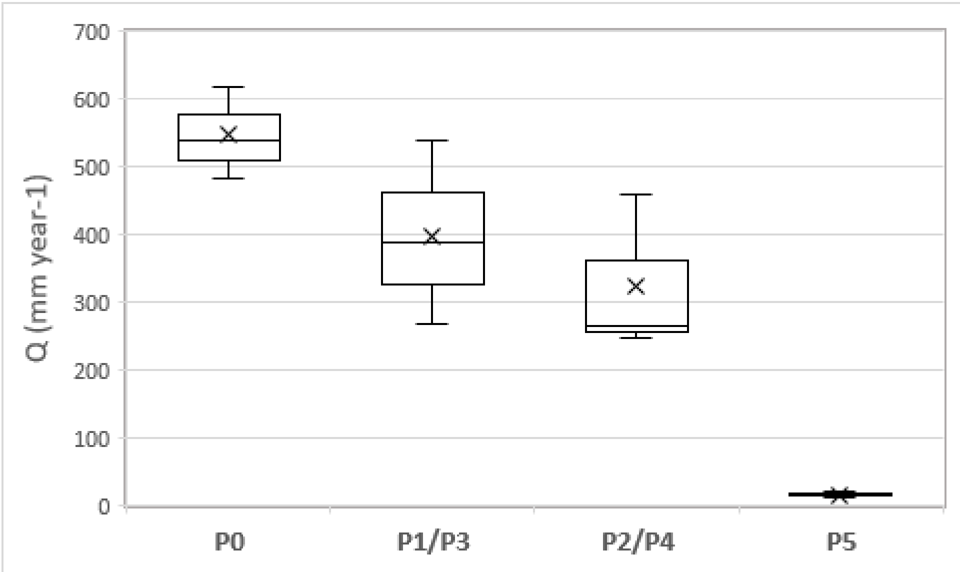

Figure 4 - Mean runoff volumes in the three-year period.

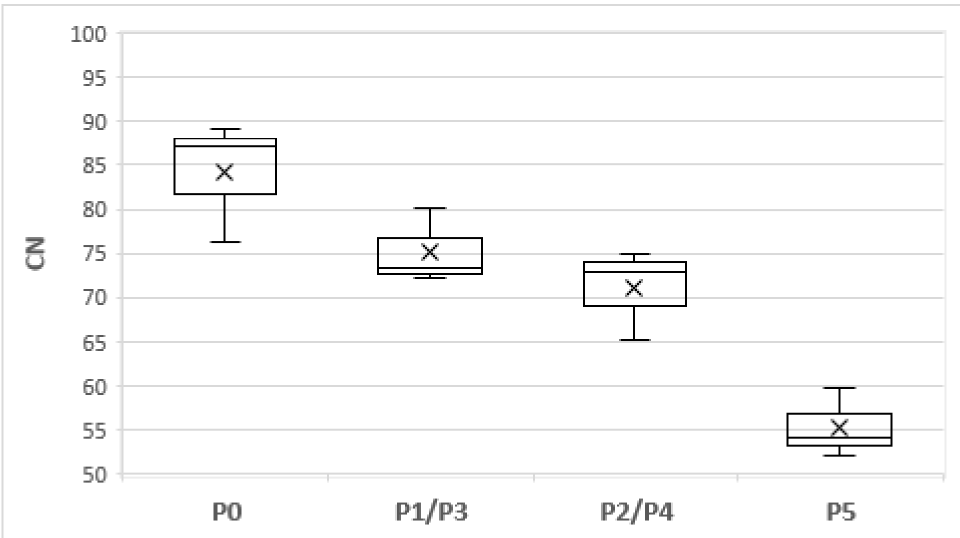

Figure 5 - Calibrated CN coefficient after the 3 hydrological years.

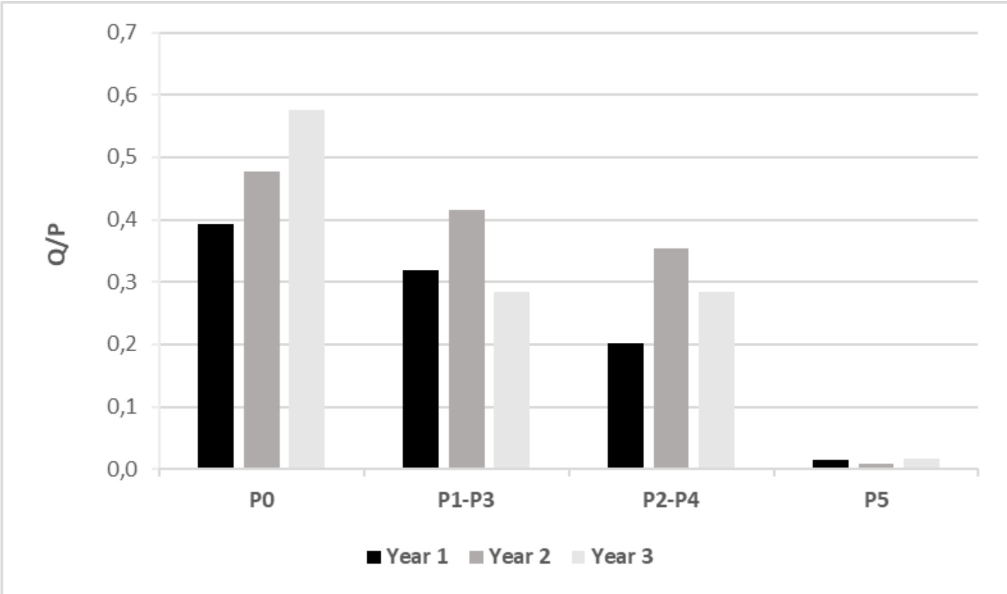

Figure 6 - Runoff normalized by annual P, in the 3 hydrological years.

As shown in Figures 4 and 5, runoff volumes and runoff coefficients in the 3-year period, decreased from P0 (bare soil, Lithic entison) to P5 (undisturbed savanna, Quartzipsamment).

There were no significant differences between the restored plots P1/P3 (without mat) and P2/P4 (with mat). However, significant differences in Q and CN were observed between the bare plot (P0) and the other plots of the Lithic entisol, the same occurring for the P5 plot. Also, the CN coefficients correlated well with those of the literature (NRCS, 2004).

Figure 3 demonstrates that water losses would be significantly reduced in the Lithic entisol with the ecologic restoration, through increased infiltration and reduced runoff volume. This effect was previously found by other authors in karst areas (Peng and Wang, 2012).

In Figure 6, when annual runoff volume was normalized by the precipitation volume, different behaviors occurred between the bare plots (bare, Entisol) and P5 (savanna, Quartzipsamment), and the restored plots (P1/P3 and P2/P4). In the former, the Q/P ratio increased from year 1 to year 3, while in the latter there was an increase in the 2nd year, followed by a reduction in the third year.

Since the Q/P ratio is an unbiased indicator of the runoff process (Chaves et al., 2012), Figure 6 indicates that the ecological restoration in plots P1/P3 and P2/P4 contributed to the gradual increase in infiltration and subsequent reduction in surface runoff (Peng and Wang, 2012).

The jute mat in P2/P4 plots contributed to a small reduction in the runoff volumes and CN, compared to the plot without jute (P1/P3), an effect reported previously (Rice et al., 2001).

3.2 Soil Loss

Figure 7 presents the mean annual soil loss (A), averaged over the three hydrological years, and Figure 8 presents the soil loss normalized by rainfall erosivity (100 A/R), in each of the three years.

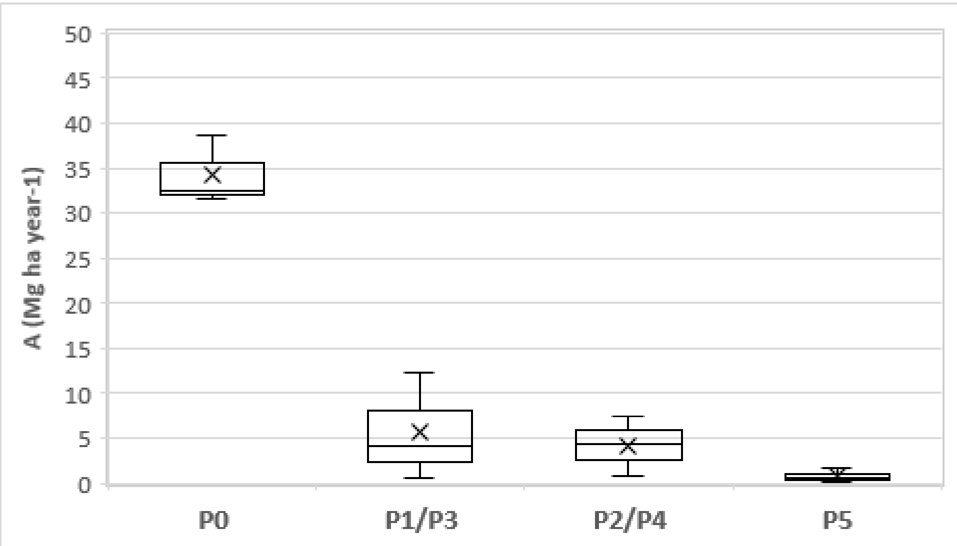

Figure 7 - Mean annual soil loss in the three hydrological years.

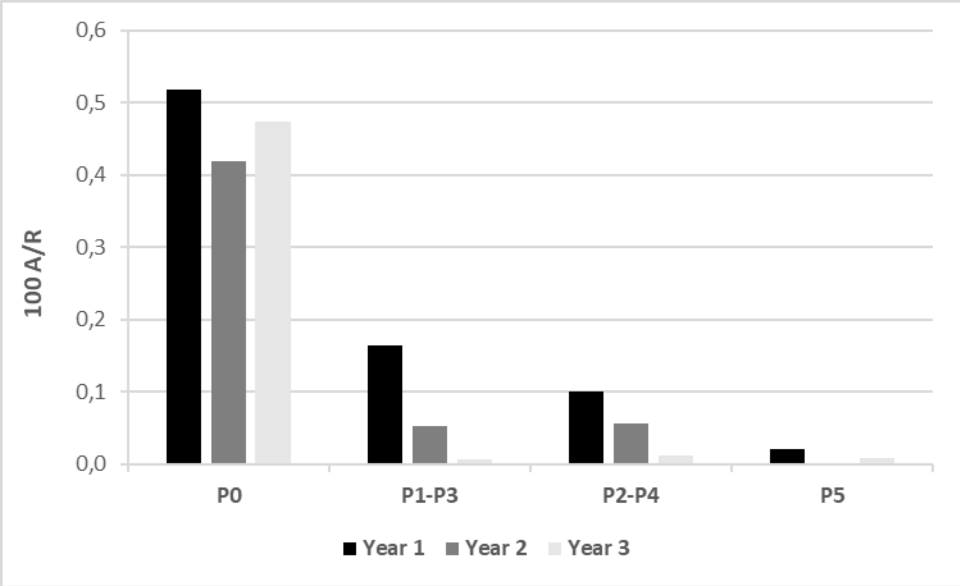

Figure 8 - Normalized soil loss in each hydrological year.

As shown in Figure 7, the mean soil loss over decreased from 34 Mg ha yr^{-1} in the bare plot (P0) to about 5 Mg ha yr^{-1} on the restored plots (P1-P4). In the case of undisturbed savanna (P5), the soil loss was negligible (0.7 Mg ha $year^{-1}$), which was expected for an undisturbed savanna area over a highly permeable soil (Fonseca et al., 2021).

On the other hand, the results normalized soil loss (100 A/R) in each hydrological year (Figure 8) show that, although there was a certain stability in the bare (P0) and undisturbed savanna plots (P5) along the three years, there was a gradual reduction in the restored plots (P1-P4) in the same period. This was due to the increase in vegetation cover, on and above the ground, over the three years (Figure 3), protecting the soil against rainfall and runoff. Similar results, indicating the importance of vegetation cover in erosion control, were reported by Peng and Wang (2012) in a karst area in China.

The exponential relationship between the amount of vegetation cover and soil loss, previously identified in the literature (Stocking, 1988; Zuazo and Pleguezuelo, 2008), can be observed in the restored plots (P0-P4) (Figure 9).

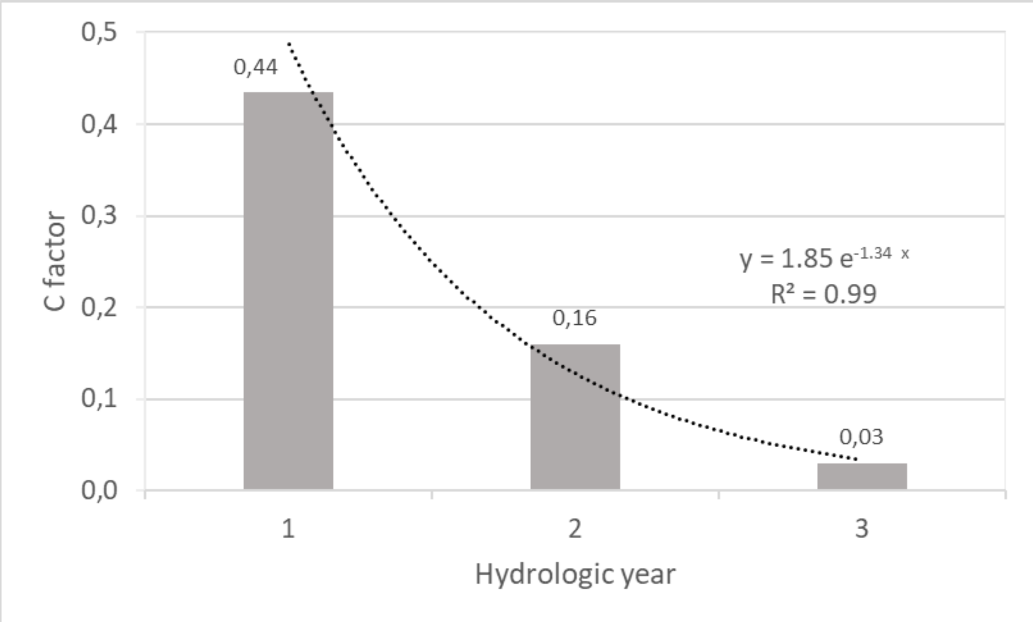

Figure 9 - Evolution of the USLE C-factor of the restored plots (P1-P4).

According to Figure 9, the USLE C-factor of the plots under restoration would change from 0.44 (Year 1) to 0.03 (Year 3), indicating that soil loss was reduced by 93% after three years, and by 97%, with respect to the bare soil (P0).

In the bare plot (P0), pedestal-type erosion, resulting from the protection provided by small stones on top of the soil surface, was observed (Figure 10). This armoring effect (Megahan, 1974) may have contributed to the reduction of normalized soil loss in the bare plot in the 2nd year (Figure 8), followed by rilling in the 3rd year (Fonseca et al. 2021) (Figure 11).

Figure 10 - Pedestal erosion (armoring) in plot P0 (bare), in 05/23/2020.

Figure 11 - Rills in plot P0 (bare) in 2/7/2021.

3.3 Soil Loss Tolerance

The soil tolerance (T) for both the Lithic entisol (plots P0-P4) and Quartzipsamment (plot P5) was 4.2 Mg ha year^{-1} (Bertoni and Lombardi Neto, 1991). Figure 12 indicates that soil loss tolerance was exceeded in the bare plot (P0) in all three hydrological years. The same occurred for the first two years in the restored plots over the Lithic entisol (P1-P4). However, in the 3rd hydrological year, erosion was below soil loss tolerance. In the case of plot P5 (undisturbed savanna under Quartzipsamment), soil loss was below the erosion tolerance during the three years of the study.

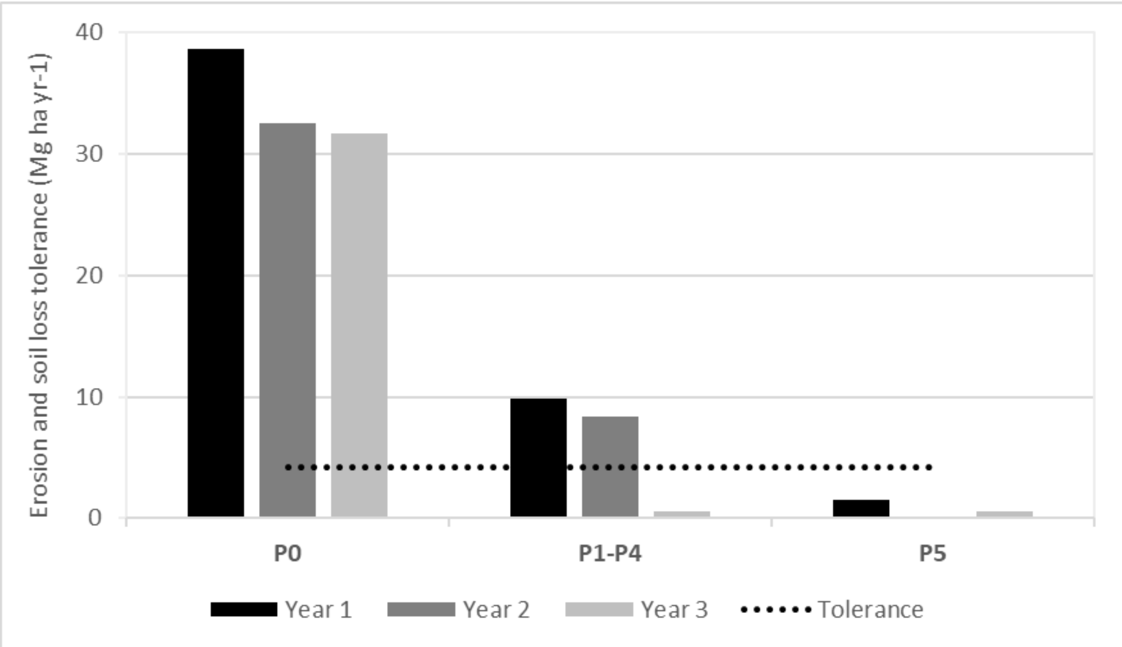

Figure 12 - Soil loss and erosion tolerance in plots P0-P5, in the three hydrological years.

Figure 12 indicates that the ecological restoration in the Lithic entisol plots (P1-P4) would, in the long run, provide a sustainable condition, i.e., A<T (Chaves, 2010a; Wischmeier, 1976), eventually improving soil loss quality (Chaves et al., 2017), with corresponding off-site environmental services, such as reduction in sedimentation (Chaves et al., 2004).

4.6 Soil Water Balance

Figure 13 shows the normalized variables of the soil water balance. In the Lithic entisol, the decreased runoff observed between the bare (P0) and the restored (P1-P4) plots was compensated by a corresponding increase in evapotranspiration. If this restoration were to be implemented at the regional (savanna) scale, they could counterbalance the decreasing trends in regional rainfall (Hoffmann and Jackson, 2000), observed in the last 50 years (Campos and Chaves, 2020).

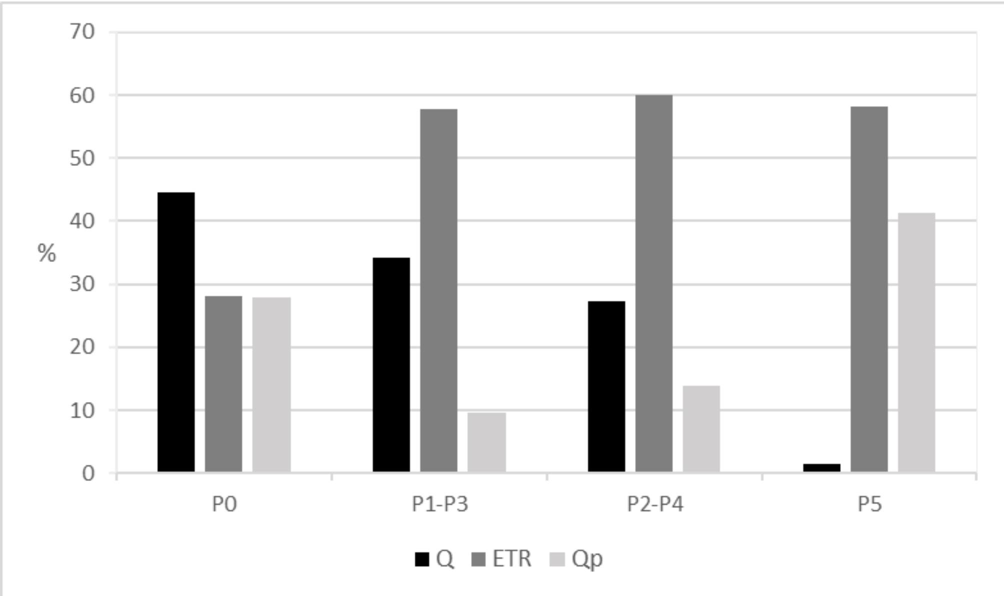

Figure 13 - Mean annual soil water balance variables, normalized by precipitation.

In the case of plot P5, its low runoff rate (1.5% of P) was due to the high natural permeability of the Quartzipsamment and to the undisturbed natural savanna cover. Although the deep percolation/recharge was higher in that plot, the actual evapotranspiration was similar to the restored plots.

5 CONCLUSIONS

In this study, carried over three hydrological years, the effects of the ecological restoration of a degraded karst area were assessed. The results indicate that, in the case of the Lithic entisol, surface runoff and soil loss were significantly reduced as vegetation cover increased.

At the end of the 3rd year, surface runoff was reduced by 52%, compared to the unrestored condition. In the case of soil loss, the reduction was 97%. Mean erosion rate in the bare entisol plot (34.3 Mg ha $year^{-1}$) was 8 times higher than the soil loss tolerance (4.2 Mg ha $year^{-1}$), contributing to its permanent degradation. In the case restored plots, the erosion rate at the end of the third year (0.6 Mg ha $year^{-1}$) was 7 times lower than the tolerance, showing the hydrologic effectiveness of the restoration.

The ecological restoration also improved the soil water balance, with a reduction in surface runoff and an increase in evapotranspiration, which could counterbalance the decreasing trends in regional precipitation and recharge. Despite the natural vulnerability of the karst areas of the Brazilian savanna, the results indicate that ecological restoration of degraded lands could generate important hydrologic services to the Vermelho river watershed, and to other similar river basins in the Brazilian savanna.

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Title Page

Title

Runoff, Soil Loss, and Water Balance in a Restored Karst Area of the Brazilian Savanna

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Declaration of interests

☐The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Maria Rita S Fonseca reports equipment, drugs, or supplies and travel were provided by University of Brasilia.

Capítulo 3

Artigo sobre Impactos *Off-site* da Restauração Ambiental em Área de Carste nos Cerrados

Earth Surface Processes and Landforms

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Sediment Yield and Composition of a Restored Karst Basin in the Brazilian Savanna

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Sediment Yield and Composition of a Restored Karst Basin in the Brazilian Savanna

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Keywords:	sediment yield, karst area, savanna, ecological restoration, sediment composition

Abstract:	Karst areas are vulnerable landscapes due to their environmental fragility and existing human activities. Permanent vegetal cover is a protective factor that, when absent, contributes to the degradation of karst areas. The objective of this study was to assess the sedimentologic (off-site) effects of the ecologic restoration carried in a 3.5 ha karst basin of the Brazilian savanna (Cerrado). Precipitation, surface runoff, and soil loss were measured in representative runoff plots (22.1 m x 1.84 m) installed inside the basin, used to assess the yearly sediment yield to a downstream sinkhole, using an appropriate sediment delivery ratio. The study was carried during three hydrologic years, between 2018/19 and 2020/21. The predominant soils in the basin were Lithic entisol and Quartzsamment. The original land covers were natural savanna, seasonal forest, and degraded pastureland (Andropogon gayanus), the latter being restored with native species and with a reformed pasture, respectively. The USLE factors R, K and C, as well as the sediment enrichment of the soil textural classes were obtained from the runoff plots, to assess the sediment yield and the sediment enrichment/depletion arriving at the sinkhole, situated downstream. Precipitation and rainfall erosivity ranged from 936 to 1,223 mm, and from 6,694 to 7,746 MJ mm ha-1hr-1, respectively. Although the ecological restoration of the basin reduced annual sediment yield from 29.2 Mg yr-1 (1st year) to 14.6 Mg yr-1 (3rd year), off-site soil loss tolerance was still exceeded in 61% of the basin area. Ecological restoration also reduced the sand content of the sediment entering the underground caves and rivers downstream, contributing to their hydrologic sustainability.

Earth Surface Processes and Landforms

Sediment Yield and Composition of a Restored Karst Basin in the Brazilian Savanna

Abstract

Karst areas are vulnerable landscapes due to their environmental fragility and existing human activities. Permanent vegetal cover is a protective factor that, when absent, contributes to the degradation of karst areas. The objective of this study was to assess the sedimentologic (off-site) effects of the ecologic restoration carried in a 3.5 ha karst basin of the Brazilian savanna (Cerrado). Precipitation, surface runoff, and soil loss were measured in representative runoff plots (22.1 m x 1.84 m) installed inside the basin, used to assess the yearly sediment yield to a downstream sinkhole, using an appropriate sediment delivery ratio. The study was carried during three hydrologic years, between 2018/19 and 2020/21. The predominant soils in the basin were Lithic entisol and Quartzsamment. The original land covers were natural savanna, seasonal forest, and degraded pastureland (*Andropogon gayanus*), the latter being restored with native species and with a reformed pasture, respectively. The USLE factors R, K and C, as well as the sediment enrichment of the soil textural classes were obtained from the runoff plots, to assess the sediment yield and the sediment enrichment/depletion arriving at the sinkhole, situated downstream. Precipitation and rainfall erosivity ranged from 936 to 1,223 mm, and from 6,694 to 7,746 MJ mm $ha^{-1}hr^{-1}$, respectively. Although the ecological restoration of the basin reduced annual sediment yield from 29.2 Mg yr^{-1} (1^{st} year) to 14.6 Mg yr^{-1} (3^{rd} year), off-site soil loss tolerance was still exceeded in 61% of the basin area. Ecological restoration also reduced the sand content of the sediment entering the underground caves and rivers downstream, contributing to their hydrologic sustainability.

Keywords: *sediment yield, sediment enrichment, restored karst areas.*

1. INTRODUCTION

Due to their environmental fragility and anthropic use, karst areas are susceptible to desertification (Jiang *et al*., 2014; Zhao and Hou, 2019). Karst desertification has been reported in the European Mediterranean (Jiang *et al*., 2014), in the Dinaric region (Gams and Gabrovec, 1999), and in Southwest of China (Jiang et al., 2014; Zhang et al., 2016). One of the processes that contributes to karst desertification is water erosion (Jacinthe et al., 2004; Nie et al., 2015; Wacha et al., 2020), with subsequent on- and off-site impacts.

Since permanent vegetal cover is the only natural feature that protects the soil against erosive processes (Zhang *et al*., 2016; Zhao and Hou, 2019), when it is removed soil erosion occurs, followed by the loss of essential soil nutrients (Nadeu et al., 2011; Nie et al., 2015; Wacha et al., 2020). As soils in karst zones are generally shallow (Zhao and Hou, 2019) and its water holding capacity is limited (Jiang *et al*., 2014; Wang, 2011; Zhao and Hou, 2019). Thus, when erosive processes occur, soil and vegetation of karst areas degrade together (Jiang *et al*., 2014).

In the Brazilian savanna, there are several karst areas, with 11,000+ caves, formed by carbonate and siliciclastic rocks (CECAV & ICMBio, 2022). The mid-western state of Goiás has the fifth largest number of caves in Brazil, including the Tarimba cave/sinkhole system (Caldeira et al., 2021). In these landscapes, there are surface and underground rivers, permanent and ephemeral (Merten and Minella, 2006).

Karst landscapes in the Brazilian savanna have experienced significant impacts due to human pressures in the last 50 years, such as accelerated soil erosion (Oliveira *et al.*, 2015; Vanwalleghem *et al.*, 2017; Anache *et al.*, 2018) and sedimentation (Minella et al., 2009). In addition to the soil degradation that occurs when erosion surpasses on-site soil loss tolerance (Chaves, 2010), off-site impacts, such as erosion rates above 1.0 Mg ha^{-1} yr^{-1}, silt rivers and deteriorate water quality (Lal, 2001; Fonseca *et al*., 2021).

In this context, soil loss, sediment yield, and sediment texture need to be assessed in the Brazilian savanna, in both karst and non-karst basins, serving as indicators of the effectiveness of land use and management (Minella *et al*., 2009; Walling and Collins, 2008), since the latter is important for the protection of caves and underground rivers against sedimentation (Kurecic et al., 2021).

Although experiments with runoff plots are useful to assess the degree of erosion in basin slopes, under natural (Renard and Foster, 1985) and agricultural areas (Hudson, 1993), if plot estimates are extrapolated to the whole landscape, sedimentation can be overestimated (Evans et al., 2017). Therefore, results from runoff plots or erosion models must be properly adjusted by appropriate sediment delivery ratios to assess sediment yield in basin outlets (Chaves, 2010; Renfro, 1975).

Since soil erosion is a selective process (Ellison, 1950; Hashin *et al*., 1998; Tesfahunegn and Vlek, 2014), enrichment or depletion of fines can occur (Nie et al., 2015; Wacha et al., 2020; Walker et al., 1978), relative to the original soil texture. In this process, fine particles, which have a higher potential for nutrient adsorption (Nie et al., 2015), are preferentially transported by thin flow (Hashin *et al*., 1998), causing the eutrophication of downstream watercourses (Kinnell, 2012). Conversely, rill erosion enriches the sediment with coarser particles and aggregates (Jiang et al., 2018), because of the higher competence of concentrated flow.

The recent conversion of the natural vegetation in the Tarimba watershed, situated immediately above the cave system, contributed to its silting with sand-sized sediments during the late Holocene (Caldeira et al., 2021). Considering the aspects above, the present study aimed to assess the sediment yield and the sediment enrichment/depletion of a restored karst basin in the Brazilian savanna, to compare the sedimentation rates to the off-site soil loss tolerances, and to evaluate the hydrologic effectiveness of the ecological restoration.

2. METHODOLOGY AND METHODS

2.1 The Tarimba Basin

The study area is comprised of a 3.5 ha basin located in the slopes of the Tarimba sinkhole, in the municipality of Mambaí, in central Brazil

(Figure 1). The Tarimba sinkhole is part of the Corrente river basin, protected by the Vermelho River Protection Area. Table 1 presents the physiographic characteristics of studied area.

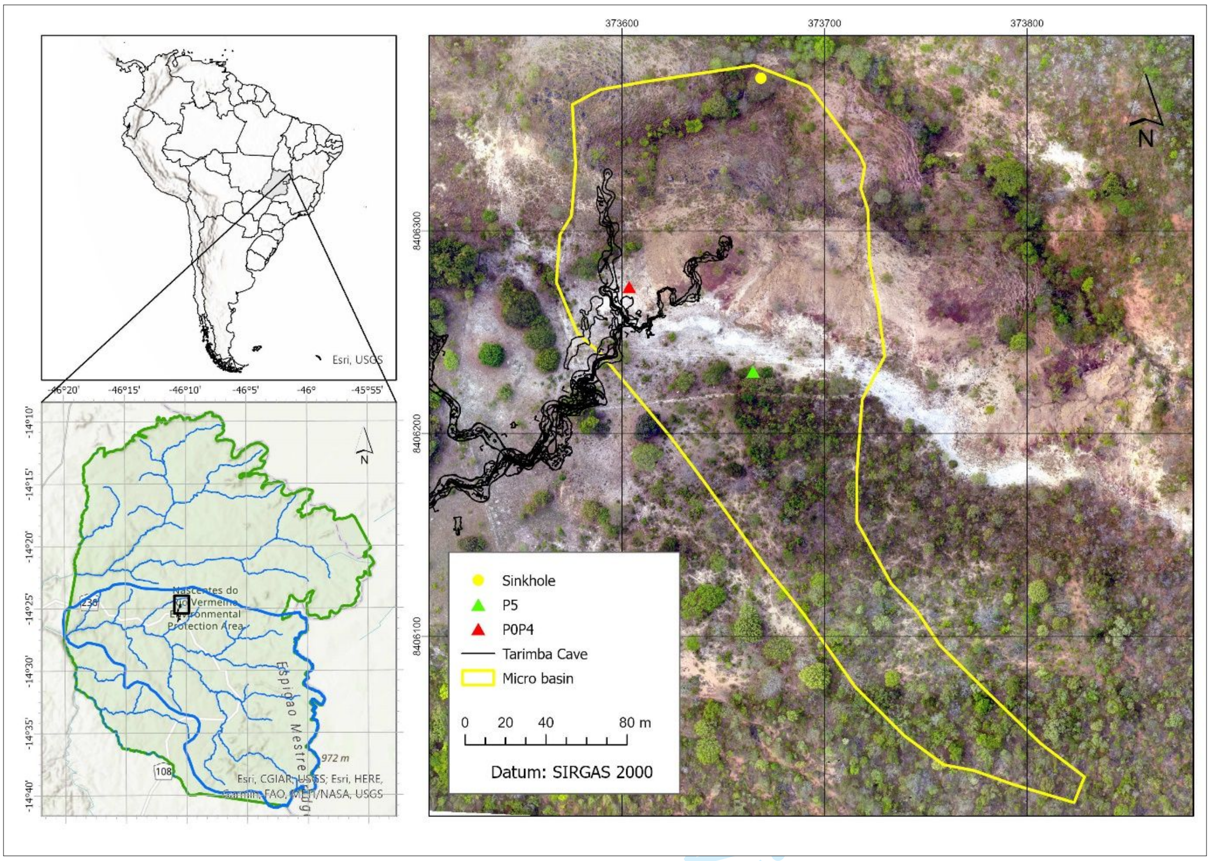

Figure 1. The Tarimba sinkhole and the study basin, in Central Brazil.

Table 1. Location and physiographic characteristics of the studied basin.

Basin Centroid	**Area (ha)**	**Perimeter (m)**	**Mean Slope (%)**	**Mean altitude (m)**
14.413º S; 46.172º W	3.5	991.0	14.2	790

The Tarimba sinkhole sits on top of the Tarimba cave system, one of the longest in Brazil (Hussain et al., 2020). The regional climate is tropical-humid (Aw, Koppen), with dry winters and rainy summers (Silva *et al.*, 2008). Mean annual precipitation is 1,200 mm and mean annual temperature is 25°C.

The upper portion of the Tarimba basin is comprised of the Urucuia Group, formed by sandstones with unconsolidated siliciclastic sediments. In the lower basin, pelitic rocks dominate, interspersed with carbonates of the Bambuí Group (Tavares et al., 2021). According to Gaspar and Campos (2007), the basin soils are Quartzsamment (upper slope), covered by undisturbed savanna and degraded pastureland, and Lithic entisol (lower slope), covered by seasonal forest and degraded pasture. Figure 2 presents the altimetry, slope, soils, and land cover of the Tarimba basin.

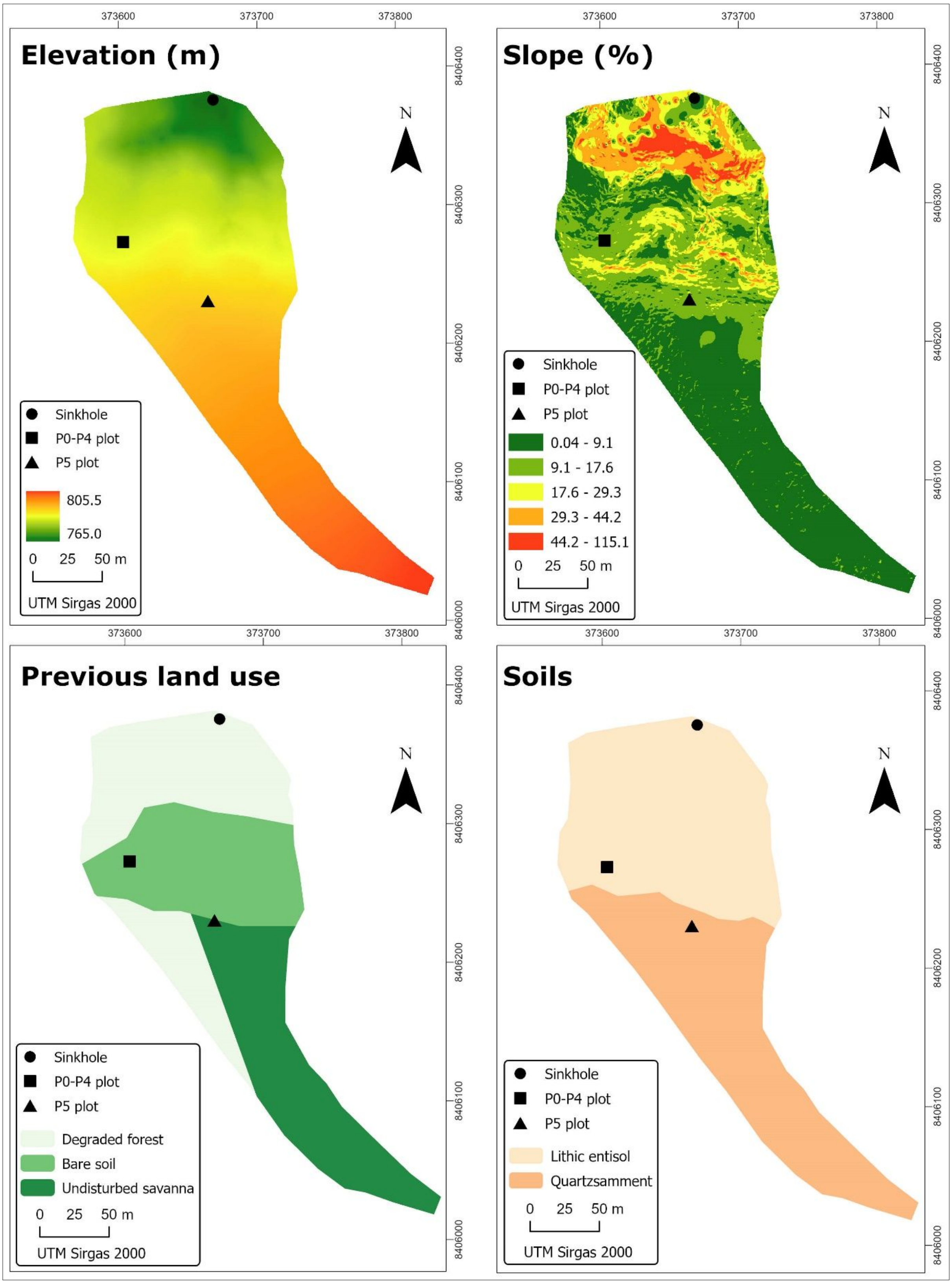

Figure 2. Elevation, slope, soils, and land cover of the Tarimba basin.

3.2 Runoff plots

Six USLE-type runoff plots (Mutchler *et al*., 1988) were installed inside the Tarimba basin, where water and soil losses generated in rainfall events were measured during three consecutive years (2018-2021). The plots had dimensions of 22.1 m x 1.80 m (39.8 m^2) and a mean slope of 11% (Figure 3).

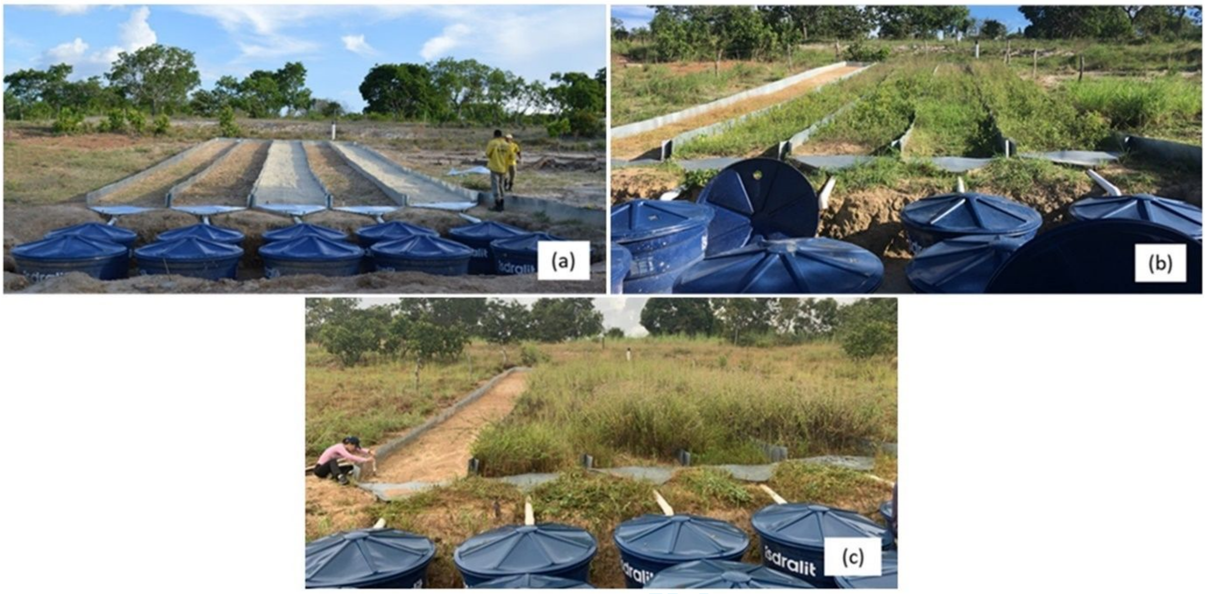

Figure 3. (a) Runoff plots in 10/30/2018, (b) 12/20/2019, and (c) 04/04/2021.

The runoff plots were installed in two different sites, representing the geology and pedology of the karst basin (Table 2). In Site A, located in an area of degraded pasture under Lithic entisol, five plots were installed: i) One bare plot (P0), kept bare with the periodic application of glyphosate herbicide; and ii) Four plots (P1-P4), restored with native savanna species in October of 2018.

To improve the soil properties, plots P1-P4 were initially planted with green manure (*Stylozanthes* sp. & *Cajanus cajan*), followed by direct planting of savanna native species, including: a) Trees: Pau Santo

(*Kielmeyera speciosa*), Ipê Caraíba Amarelo (*Tabebuia aurea*), Jacarandá Bico de Paragaio (*Machaerium acutifolium*), Guatambu do Cerrado (*Aspidosperma macrocarpum*), Tinguí (*Magonia pubecens*), Aroeira Preta (*Myracrodruon urundeuva*), Amburana (*Amburana cearenses*); b) Shrubs: Assa Peixe (*Vernonanthura sp.*), Fedegozinho (*Senna sp.*), Copaibinha (*Copaifera sp.*) Amargoso (*Lepdaploa aurea*); and c) Grass: Andropogon (*Andropogon fastigiatus*).

Plots P2 and P4 received a jute mat over the soil surface, and plots P1 and P3 were installed without the mat. In the degraded areas of the Lithic entisol, the ecological restoration was similar to plots P1-P4, with jute mats installed in the eroded areas of the slope.

In Site B, plot P5 was installed over undisturbed savanna, with 50% of tree cover and 50% of shrubs and grasses. The dominant species were: *Tabebuia aurea, Maioria pubecens, Copaifera martii, Lepdalloa aurea, Aspidosperma macrocapom,* and *Stilozanthes sp.* The degraded pasture on the Quartzsamment soil was restored with *Andropogon gayanus* grass.

3.3 Precipitation and Runoff

A standard WMO rain gage was installed in the center of the basin, where daily rainfall volume was measured. Runoff volume was obtained bi-weekly from 2-1,000 L collection tanks, installed in the lower end of each plot, or when a major rainfall event occurred. The plots were monitored during the rainy summers from October 2018 to September 2019 (total of 16 runoff collections), from October 2019 to September 2020 (20 collections), and from October 2020 to September 2021 (19 collections).

3.4 Plot Soil Loss

Soil loss in the collection tanks of the runoff plots was obtained by submerging 1 L bottles after agitating the water in the tanks (Bagarello and Ferro, 2004). In the laboratory, the samples were decanted with aluminum sulfate, oven-dried at 105°C for 24 h, and weighed. The plot soil loss in the sampled period was obtained by the following equation (Oliveira *et al*., 2015; Anache *et al.*, 2018):

$$P_s = C\,V / A_p \qquad (1)$$

where: P_s (Mg ha^{-1}) = soil loss in the analyzed period; C (g L^{-1}) = sediment concentration in the sample; V (L) = runoff volume from the plot's tanks in the period; e A_p (ha) = plot area.

3.5 Sediment Enrichment

Sediment enrichment was calculated by comparing the texture of the sand, silt, and clay fractions of the sediment with those of the original soil, obtained by composite soil samples (0-20 cm) around the plots (Figure 4).

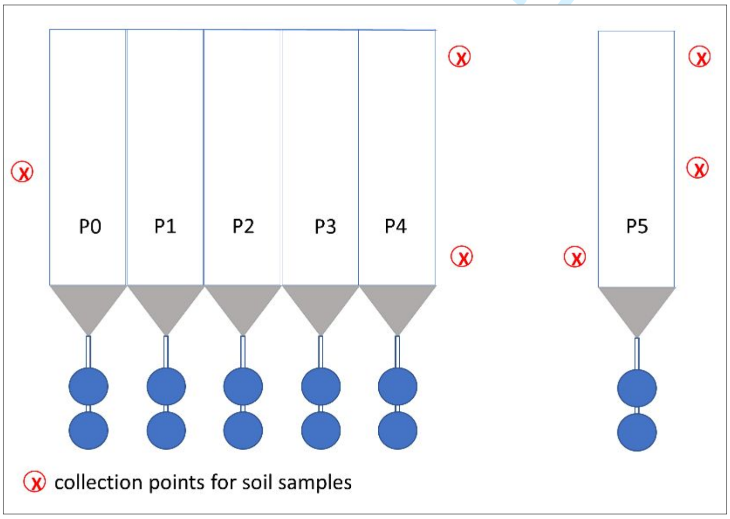

Figure 4. Runoff plots and soil sampling sites for the enrichment ratio.

Five random sediment samples were selected for each plot, in each year. The texture of the original soil and of the sediment samples was obtained with a BetterSize ST® granulometer, using the three USDA (1987) textural class sizes (sand, silt, clay).

After the sediment fractions of the 3 textural classes were obtained, enrichment/depletion ratio was obtained for each textural class using the following equation (Ni et al., 2022):

$$ED_i = P_s/P_o \quad (2)$$

where: ED_i= sediment enrichment/depletion of textural class *i*; P_s = % of the textural fraction *i* in the sediment; P_o = % of textural fraction *i* in the original soil. $ED_i > 1$ indicates fraction enrichment in the sediment, while $ED_i < 1$ indicates fraction depletion (Flanagan and Nearing, 2000). It was assumed that the sediment enrichment/depletion of the runoff plots was the same of the basin outlet, located 0.1 km downstream, because of the small basin area (Chaves, 2010).

3.6 Basin sediment yield

The annual sediment yield in the basin outlet was obtained by the following equation (Renfro, 1975; Chaves, 2010):

$$Y = SDR \cdot A_t \quad (3)$$

where: Y (Mg year^{-1}) = annual sediment yield to the basin outlet; SDR (0-1) = sediment delivery ratio; and A_t (Mg year^{-1}) = total basin soil loss. The sediment delivery ratio was estimated by Vanoni's (1975) equation, used by Chaves (2010) in the Brazilian savanna:

$$SDR = 0.42\ A_b^{-0.125} \quad (4)$$

where: SDR (0-1) = sediment delivery ratio; A_b (km^2) = basin area. The soil loss (A) in each basin cell (1 m^2) was obtained by spatial analysis in the GIS, using the USLE (Wischmeier and Smith, 1978):

$$A = R\ K\ L\ S\ C\ P \quad (5)$$

where: A (Mg ha^{-1} $year^{-1}$) = mean annual soil loss; R (MJ mm ha^{-1} h^{-1}) = rainfall erosivity; K (Mg MJ^{-1} mm^{-1}) = soil erodibility; L (dimensionless) = slope length factor; S (dimensionless) = slope steepness factor; C (dimensionless) = land cover and management factor; P (dimensionless) = conservation practices factor.

The rainfall erosivity of each hydrologic year was obtained with a regional Fournier-type equation, based on the monthly and annual rainfall of the site (Silva, 2004):

$$R = 12{,}592 \sum_{i=1}^{12} \left(M_i^2/P\right)^{0.603} \quad (6)$$

where: R (MJ mm $ha^{-1}hr^{-1}$) = annual rainfall erosivity, M_i (mm) = monthly precipitation, P (mm) = annual precipitation.

Soil erodibility was calculated using the mean annual soil loss of the bare plot during the 3 hydrologic years, using the following equation (Wischmeier and Smith, 1978):

$$K = A\ /\ (R\ LS) \quad (7)$$

where: K (Mg ha ha^{-1} MJ^{-1} mm^{-1}) = soil erodibility, with the other variables previously defined.

The LS factor was obtained for each basin pixel with the GIS, using the equation of Mitasova *et al*. (1996):

$$LS\ (x,y) = (m+1)\ [A\ (x,y)\ /\ 22.1]^m\ [\sin s\ (x,y)\ /\ 0.09]^n \quad [8]$$

where: LS (x,y) (dimensionless) = USLE LS factor for cell (x,y); A (x,y) (m^2/m) = unit upstream contribution area of cell (x,y); s (x,y) (degrees) = slope of cell (x,y); m (0.4-0.6) = exponent depending on the predominant type of erosion (rill or inter-rill); n (1.0-1.4) = exponent that depends on the predominant type of erosion (rill or inter-rill).

The C factor of the degraded and restored areas was obtained from the runoff plots in each hydrologic year by the following equation (Wischmeier and Smith, 1978):

$$C = A_i / A_0 \quad (9)$$

where: C (0-1) = USLE C factor for plot i; A_i (Mg ha^{-1} $year^{-1}$) = annual soil loss of plot with cover i; A_0 (Mg ha^{-1} $year^{-1}$) = annual mean soil loss of the bare plot. The C factor for the undisturbed savanna was obtained from Gomes *et al*. (2017). In the case of the USLE P factor, since there were no conservation practices in the basin, it was taken as 1.0 (Chaves and Piau, 2008; Wischmeier and Smith, 1978).

4. RESULTS

4.1 USLE Factors

Table 2 presents the texture of the basin soils, as well as the value of their erodibility and off-site soil tolerance.

Table 2. Texture, soil erodibility (K) and off-site tolerance (T_{off}) of the basin soils.

Soil	**Sand** (%)	**Silt** (%)	**Clay** (%)	**K factor** (t ha ha^{-1} MJ^{-1} mm^{-1})	**T_{off}** (Mg ha^{-1} yr^{-1})
Quartzsamment	91.2	6.9	1.9	0.0014	1.0
Lithic entisol	28.2	57.7	14.1	0.0035	1.0

Table 3 presents the precipitation volumes, annual erosivity, and annual C factor of the restored plots, in each of the three hydrologic years. Figure 4 shows the basin maps of USLE K and LS factors.

Table 3. Annual rainfall and erosivity, soil cover, and annual C factor of the restored plots, in the three hydrologic years.

Year	**Annual Precipitation** (mm)	**R Factor** (MJ mm $ha^{-1}hr^{-1}$)	**% Soil Cover** (P1-P4)	**C Factor** (P1-P4)
1	1,223,6	7,440.0	10.0	0.44
2	1,295.1	7,746.0	45.0	0.16
3	935.5	6,694.1	91.0	0.03

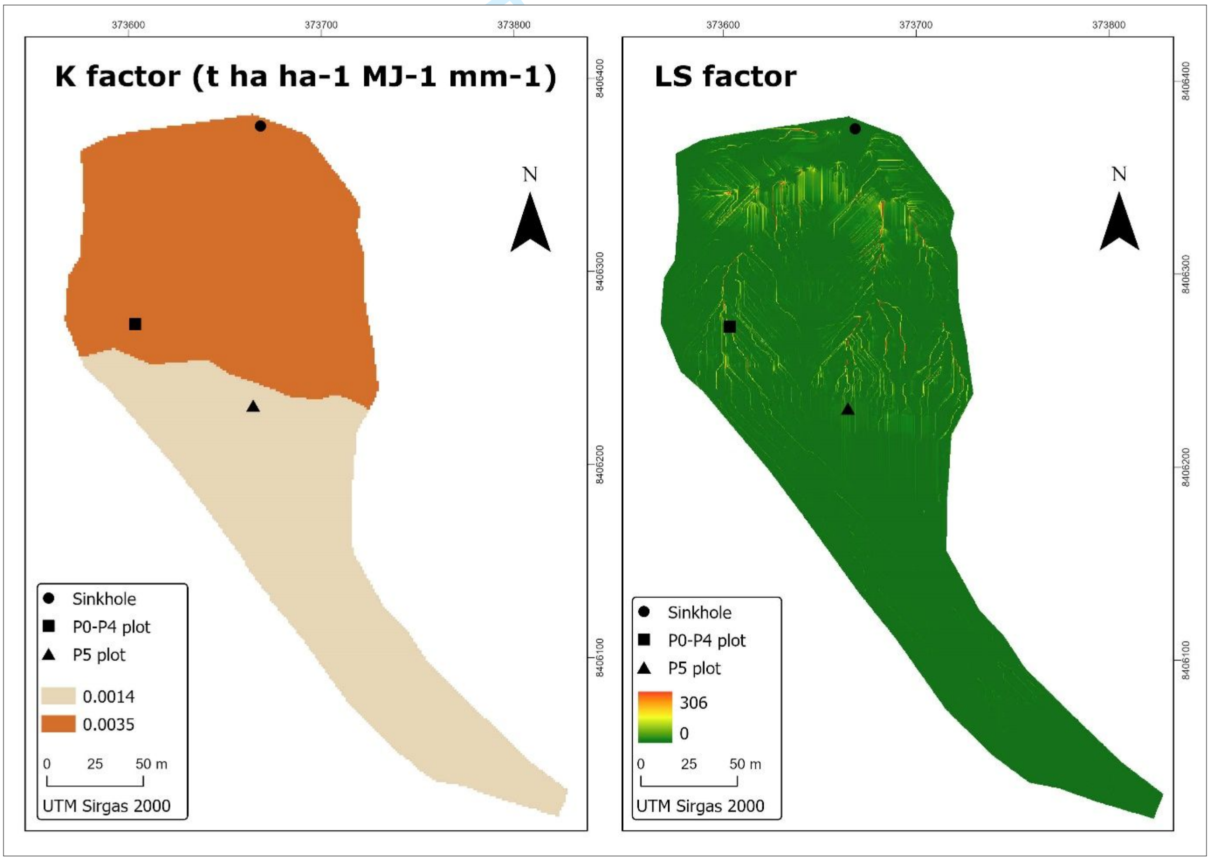

Figure 5. Basin soil erodibility (K) and LS factor.

According to Table 2 and Figure 5, soil erodibilities were 0.0014 (Quartzsamment) and 0.0035 (Lithic entisol). According to Table 3, annual rainfall erosivity ranged from 6,694 to 7,746 MJ mm $ha^{-1}hr^{-1}$, depending on the volume of annual precipitation (935 to 1,295 mm, respectively).

Conversely, the mean C factor of the four restored plots decreased considerably from the first (0.44) to the third year (0.03) of the experiment (Figure 6). There was a small difference between the 3-year C-factor of the plots with jute mat (C=0.20) and without mat (C=0.21), indicating a negligible effect of that type of surface protection.

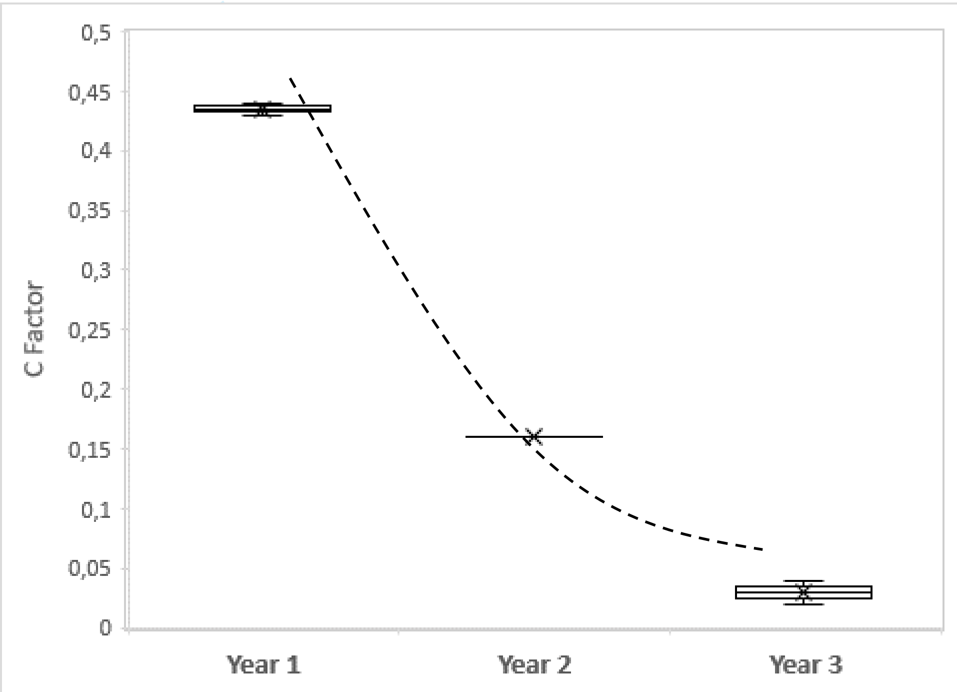

Figure 6. C-factor of the restored plots (P1-P4) during the study.

Since the degraded pastures adjacent to the plots received a similar ecological restoration, the basin C-factor also decreased gradually with time (Figure 7).

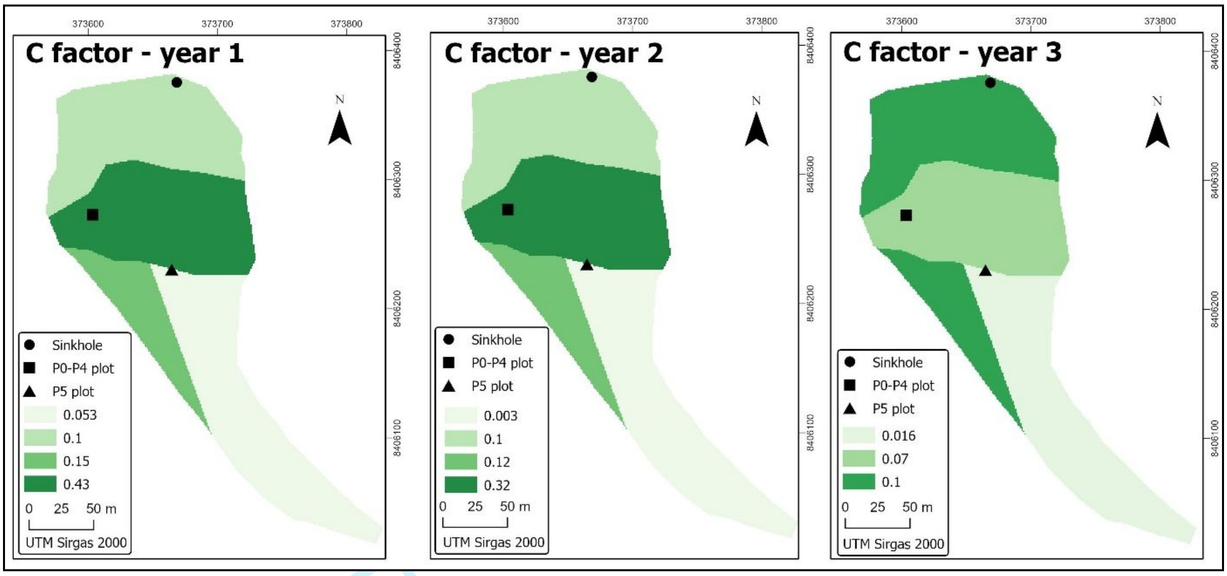

Figure 7. Basin C-factor in the 3 years of the study.

4.2 Soil Loss and Sediment Yield

Mean yearly soil loss in the basin is shown in Figure 8, for each hydrologic year. According to that Figure, there was a gradual reduction in on-site erosion, particularly in the restored areas. Soil loss was higher in the Lithic entisol (shown in yellow and red in Figure 8), due to its higher erodibility and slope grade.

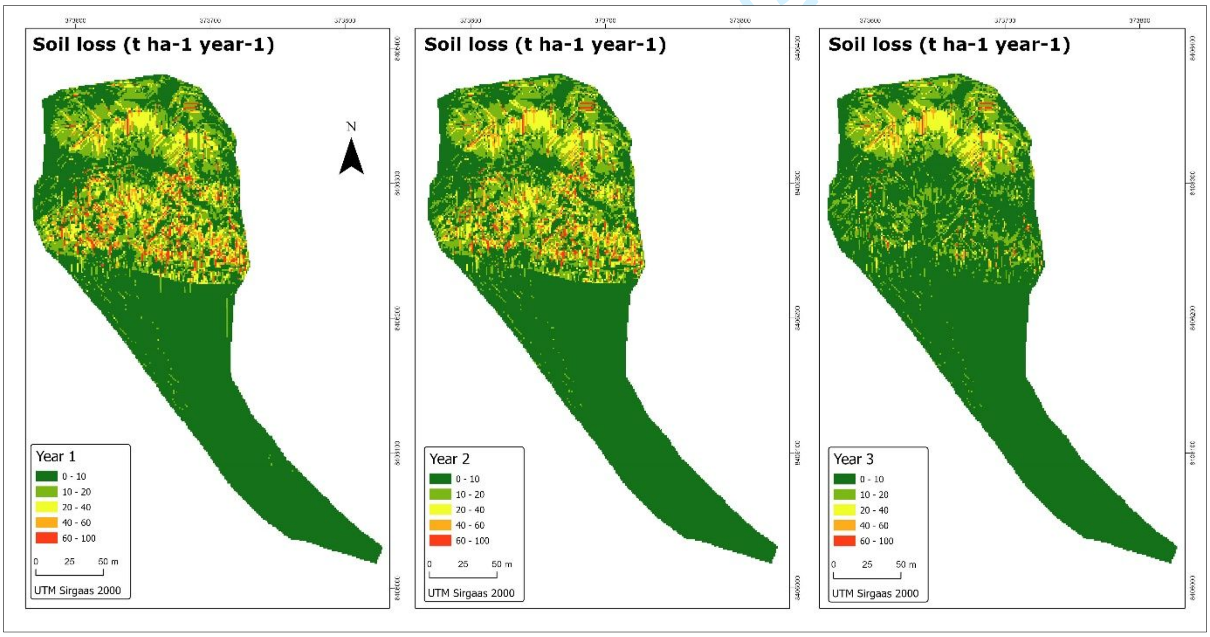

Figure 8. Mean annual soil loss in the basin, in the three hydrologic years.

Table 4 presents the basin annual soil loss and sediment yield, during the three hydrologic years. The basin sediment delivery ratio was 0.72, indicating that 72% of the eroded sediment generated inside the basin reached its outlet, every year.

Table 4. Mean and total soil loss, and annual sediment yield in the basin, in the three hydrologic years.

Year	**Mean Soil Loss** (Mg ha^{-1} yr^{-1})	**Total Soil Loss** (Mg yr^{-1})	**Annual Sediment Yield** (Mg yr^{-1})
1	11.6	40.6	29.2
2	9.3	32.6	23.4
3	5.8	20.3	14.6

According to Table 4, there was a significant reduction in the basin soil loss along the three years analyzed, as well as in the sediment yield reaching the basin outlet, indicating that the restoration generated important on- and off-site hydrologic services.

Figure 9 shows the basin areas where erosion exceeded on-site (T_{on}) and off-site (T_{off}) soil loss tolerances, three years after the ecological restoration. Erosion rates in the basin exceeded the on-site and off-site tolerances in 29% and in 61% of the basin area, respectively.

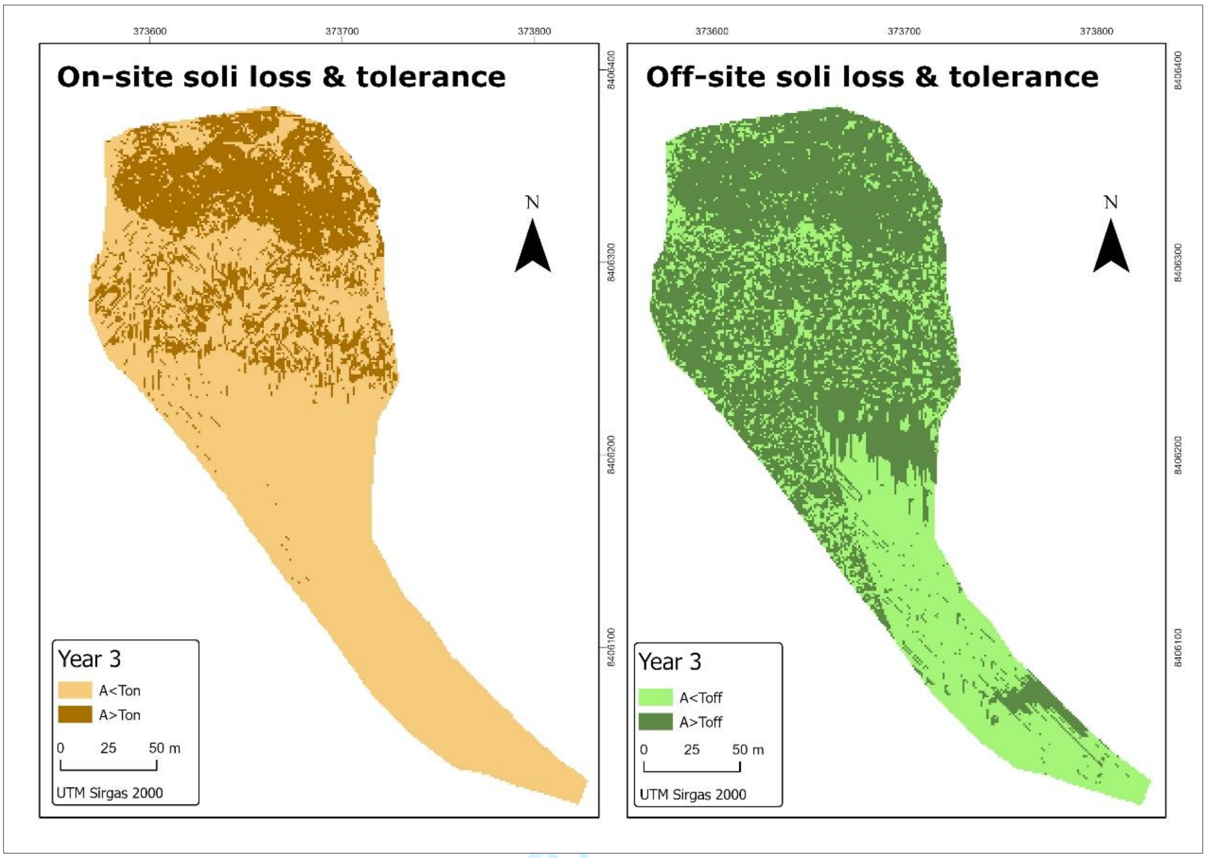

Figure 9. Basin areas where soil loss was surpassed on-site (left) and off-site (right) soil loss tolerance.

4.3 Sediment Enrichment and Depletion

Table 5 presents the sediment enrichment or depletion in the runoff plots. According to Table 5, there was an enrichment of the sediment sand fraction in the bare plot (P0), while in the restored plots (P1-P4) showed a slight depletion of sand, compared with the original soil. In the case of the silt fraction, the opposite occurred. Table 5 also indicates that all plots showed a depletion in the clay fraction with respect to the original soil.

Table 5. Sediment enrichment/depletion ratio in the Lithic entisol plots.

Plot	**Sediment enrichment/depletion**		
	Sand	**Silt**	**Clay**
P0 (bare)	1.48 b	0.88 a	0.52 a
P1-P3 (no mat)	0.84 a	1.15 b	0.70 b
P2-P4 (mat)	0.76 a	1.20 b	0.67 ab

4. DISCUSSION

The erodibility of the basin soils (Table 2) was about 10 times lower than those obtained by Castro *et al*. (2011) in similar soils of Brazil. As opposed to conventional fallow plots, where the soil is tilled downhill every year (Wischmeier and Mannering, 1969), the bare plot in the present study was not tilled, and therefore naturally consolidated and more resistant to erosion (Knapen *et al*., 2008).

The exponential reduction observed in the C-factor of the restored plots (Figure 6) was expected, due to the increase in the permanent soil cover over time (Laflen *et al*.,1985; Stocking, 1988). Fonseca *et al*. (2021) showed a significant correlation between the degree of permanent soil cover and runoff volume in different areas of the Brazilian savanna, reducing the occurrence of rill erosion.

With respect to the 50% sediment yield reduction observed after the ecological restoration, Chaves *et al*. (2004) found a similar sedimentation abatement (73%) in a restored basin in Brazilian savanna. The ecological restoration allowed for the retention of coarser particles, depleting the sand content of the sediment. As suggested by Shi *et al*. (2012), the lower runoff energy in the restored plots reduced the sediment transport capacity, retaining the heavier sand and enriching the smaller sediment fractions.

The significant reduction in basin sediment yield provided by the ecological restoration (Table 6) and the depletion of sand in the sediment reaching the sinkhole reduces the silting potential of the underground Tarimba cave system. Kurecic *et al*. (2021) found similar sedimentological benefits after the restoration of karst areas of Croatia.

Although the sedimentological benefits of three years of basin restoration are significant, there is still room for improvement. Chaves *et al.* (2004) reported that the sedimentological benefits of ecological restoration in the Brazilian savanna stabilize after 10 years.

5. CONCLUSIONS

This study demonstrated the sedimentologic benefits of the ecologic restoration of a degraded karst basin in the Brazilian savanna. Three years after the restoration of natural and pasture areas, there was a 50% reduction in soil loss and in sediment yield, compared to the previous condition. The reduction was associated with the increase of permanent soil cover, provided by the native plants and pasture, decreasing runoff and erosion. Additionally, as the restoration progressed, there was a depletion in the sand content of the sediment, reducing the silting potential of the underground cave system.

In spite of the benefits observed after three years of ecological restoration, erosion rates still exceeded on- and off-site soil loss tolerances in 29% and in 61% of the basin area, respectively, particularly in the lower slope, because of the high soil erodibility of the Entisol and the high slope steepness. The results indicate that there is still room for erosion and sediment yield abatement, which would improve with the maturity of the ecological restoration process.

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6. Conclusões

Na meta-análise de perda de solo e escoamento superficial (Capítulo 1), foi concluído que a perda de solo nos Cerrados está exponencialmente relacionada ao volume de escoamento superficial, sendo ambos inversamente proporcionais ao grau de cobertura permanente do solo.

Além disso, nos estudos com cobertura permanente do solo, tais como savana natural, reflorestamento, e plantio direto, as perdas de solo estiveram abaixo das tolerâncias *on- e off-site*, indicando que ela é fundamental para a manutenção da sustentabilidade agronômica e hidrológica da paisagem. Uma função exponencial foi ajustada entre o volume de escoamento superficial e a perda de solo, permitindo que a última seja estimada a partir da primeira, em áreas com dados sedimentológicos escassos.

No experimento com parcelas de enxurrada em área de carste do Cerrado (Capítulo 2), os resultados indicam que, no caso do Neossolo Litólico, o escoamento superficial e a perda de solo foram significativamente reduzidos à medida que a cobertura do solo aumentou, em função da restauração ambiental com espécies nativas.

No final do 3º ano, o escoamento superficial foi reduzido em 52%, em comparação com a condição não restaurada. No caso da perda de solo, a redução foi de 97%. A taxa média de erosão na parcela de Neossolio Litólico descoberto (34,3 Mg ha^{-1} ano^{-1}) foi 8 vezes superior a tolerância à perda de solo (4,2 Mg ha^{-1} ano^{-1}), contribuindo para sua degradação permanente. No caso das parcelas restauradas, a taxa de erosão ao final do terceiro ano (0,6 Mg ha^{-1} ano^{-1}) foi 7 vezes menor que a tolerância, mostrando a eficácia hidrológica da restauração.

A restauração ecológica também melhorou o balanço hídrico do solo, com redução do escoamento superficial e aumento da evapotranspiração, o que poderia contrabalançar as tendências decrescentes da precipitação e da recarga da água subterrânea regional. Apesar da vulnerabilidade natural das áreas cársticas do cerrado brasileiro, os resultados indicam que a restauração ecológica de áreas degradadas pode gerar importantes serviços hidrológicos para a Gruna da Tarimba e para a bacia do rio Vermelho.

Esta pesquisa também demonstrou os benefícios sedimentológicos da restauração ecológica de uma bacia cárstica degradada no cerrado brasileiro (Capítulo 3). Três anos após a restauração das áreas naturais e de pastagens, observou-se uma redução de 50% na perda de solo e no aporte de sedimentos, relativamente à condição degradada. A redução foi associada à cobertura permanente do solo, proporcionada pelas plantas nativas e pastagens, diminuindo o escoamento superficial e a erosão. Além disso, à medida que a restauração avançava, houve uma depleção da fração areia do sedimento, reduzindo o potencial de assoreamento do sistema de cavernas, a jusante.

Apesar dos benefícios hidrossedimentológicos observados após três anos de restauração ecológica na vertente, as taxas de erosão ainda excederam as tolerâncias de perda de solo *on* e *off site* em 29% e em 61% da área total da vertente, respectivamente, principalmente na sua porção inferior, em consequência da alta erodibilidade do Neossolo e da alta declividade. Os resultados indicam que ainda há espaço para redução da erosão e da produção de sedimentos, na medida em que a restauração hidrológica da vertente atinja uma maior maturidade.

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