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GUILHERME HENRIQUE EXPEDITO LENSE

MODELAGEM DAS PERDAS DE SOLO POR EROSÃO HÍDRICA EM ÁREAS TROPICAIS BRASILEIRAS

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Tese apresentada como parte dos requisitos para obtenção do título de Doutor em Ciências Ambientais pela Universidade Federal de Alfenas. Área de concentração: Ciências Ambientais.

Orientador: Prof. Dr. Ronaldo Luiz Mincato.

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" Modelagem das Perdas de Solo por Erosão Hídrica em Áreas Tropicais Brasileiras ”

A Banca examinadora abaixo-assinada aprova a Tese apresentada como parte dos requisitos para a obtenção do título de Doutor em Ciências Ambientais pela Universidade Federal de Alfenas. Área de concentração: Ciências Ambientais.

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"Amei a sabedoria e a busquei desde a minha juventude e procurei tomá-la como esposa, pois fiquei enamorado de sua formosura."

(Sabedoria 8:2)

RESUMO

A erosão hídrica é a principal causa de degradação dos solos brasileiros, e este problema deve ser constantemente combatido afim de garantir a conservação do solo. Nesse contexto, a modelagem é uma técnica que pode contribuir para o planejamento de práticas de mitigação da erosão. Dessa forma, o objetivo do trabalho foi modelar a erosão hídrica em cinco diferentes regiões brasileiras. No primeiro caso, foi avaliada a erosão hídrica na sub-bacia hidrográfica do Córrego Coroado, no sudeste brasileiro. Foi aplicado o Método de Erosão Potencial (EPM) para estimar a erosão hídrica entre 1988 e 2018. A conversão de áreas de pastagem e milho em cultivos de café com adoção de práticas conservacionistas e a expansão de áreas de reflorestamento contribuíram para uma redução de 37% nas perdas de solo. No segundo caso, foi estimado o efeito do desmatamento sobre a variação espacial e temporal da erosão hídrica na Bacia Hidrográfica do Rio Xingu, uma das regiões mais afetadas pelo desmatamento na Amazônia Brasileira. Entre 1988-2018, ocorreu um aumento de 12% (52.258 km^2) no desmatamento da floresta amazônica na região, e usando o EPM foi possível verificar que neste período houve um aumento de 312% na taxa de perdas de solo por erosão hídrica, que correspondente a cerca de 180 milhões de toneladas de solo. No terceiro caso, foi estimada a erosão hídrica na Bacia Hidrográfica do Rio Tietê utilizando a Equação Universal de Perda de Solo Revisada (RUSLE). Em 18% desse território as taxas de erosão são superiores aos Limites de Tolerância de Perda de Solo (TPS). No quarto caso, foi estimada a erosão hídrica no Estado de Rondônia, usando a RUSLE. A taxa média de perda de solo foi de 22,50 Mg ha^{-1} ano^{-1} e 19% das áreas do estado devem ser priorizadas na adoção de medidas conservacionistas do solo. No quinto caso, foi estimada as perdas de solo no Sistema Cantareira, um dos maiores sistemas de abastecimento de água do mundo. A RUSLE apontou que em 66% do Sistema Cantareira, as perdas de solo estão abaixo da TPS e, em 34% da região, a erosão hídrica está comprometendo a sustentabilidade dos recursos hídricos e do solo. Em todas as regiões estudadas existem áreas com altas perdas de solo associadas a relevos íngremes, solos com baixa densidade de cobertura vegetal, desmatamento e ausência de práticas conservacionistas. A modelagem permite identificar as áreas prioritárias na adoção de manejos conservacionistas dos solos e é uma forma de destacar o problema da degradação dos solos pela erosão hídrica, conscientizar órgãos públicos e privados sobre a necessidade de mitigar os processos erosivos e de incentivar a elaboração e adoção de políticas ambientais voltadas a conservação do solo.

Palavras-chave: Conservação do Solo; EPM; RUSLE.

ABSTRACT

Water erosion is the main cause of Brazilian soil degradation, and this problem must be constantly combated in order to guarantee soil conservation. In this context, modeling is a technique that can contribute to the planning of erosion mitigation practices. Therefore, the objective of the work was to model water erosion in five different Brazilian regions. In the first case, water erosion was evaluated in the Córrego Coroado sub-basin, in southeastern Brazil. The Erosion Potential Method (EPM) was applied to estimate water erosion between 1988 and 2018. The conversion of pasture and corn areas into coffee crops with the adoption of conservation practices and the expansion of reforestation areas contributed to a 37% reduction in soil losses. In the second case, the effect of deforestation on the spatial and temporal variation of water erosion in the Xingu River Basin, one of the regions most affected by deforestation in the Brazilian Amazon, was estimated. Between 1988-2018, there was an increase of 12% (52,258 km^2) in the deforestation of the Amazon forest in the region, and using the EPM it was possible to verify that in this period there was a 312% increase in the rate of soil losses due to water erosion, which corresponding to around 180 million tons of soil. In the third case, water erosion was estimated in the Tietê River Basin using the Revised Universal Soil Loss Equation (RUSLE). In 18% of this territory, erosion rates are higher than the Soil Loss Tolerance Limits (T). In the fourth case, water erosion was estimated in the State of Rondônia, using RUSLE. The average rate of soil loss was 22.50 Mg ha^{-1} $year^{-1}$ and 19% of the state's areas should be prioritized in the adoption of soil conservation measures. In the fifth case, soil losses were estimated in the Cantareira System, one of the largest water supply systems in the world. RUSLE pointed out that in 66% of the Cantareira System, soil losses are below T and, in 34% of the region, water erosion is compromising the sustainability of water and soil resources. In all regions studied there are areas with high soil losses associated with steep reliefs, soils with low density of vegetation cover, deforestation and lack of conservation practices. Modeling allows the identification of priority areas in the adoption of soil conservation management and is a way of highlighting the problem of soil degradation due to water erosion, raising awareness among public and private bodies about the need to mitigate erosion processes and encouraging the development and adoption environmental policies aimed at soil conservation.

Keywords: Soil Conservation; EPM; RUSLE.

SUMÁRIO

1 INTRODUÇÃO

O solo é um recurso natural essencial para a sobrevivência da vida na terra. Este recurso juntamente com a água, o ar, os nutrientes, a luz e o calor, permite o crescimento e desenvolvimento das plantas terrestres. Além de fornecer vários outros produtos e serviços ecossistêmicos, as plantas desenvolvidas no solo são responsáveis por suprir a maior parte das necessidades alimentares do homem e dos animais. Dessa forma, o solo representa a principal fonte para produção de alimentos na Terra. Além disso, os solos geram diversos serviços ecossistêmicos, sequestram carbono e administram a água, minerais e ciclos biológicos (Cerdà *et al*., 2018; Bertol; De Maria; Souza, 2019).

Infelizmente, a degradação do solo tem avançado aceleradamente principalmente devido à má utilização deste recurso pelo homem. Atividades humanas como o desmatamento, preparo convencional e as mudanças relacionadas ao uso da terra aceleram a erosão hídrica, comprometendo a sustentabilidade do solo. A erosão hídrica consiste no desgaste do solo pelo escoamento superficial da água da enxurrada, ocorrendo a desagregação, transporte e deposição de partículas constituintes do solo, sendo um processo fortemente influenciado pelas atividades antrópicas (Foster, 1982). Em áreas com altas taxas de erosão ocorre a diminuição do potencial produtivo da terra pela perda de nutrientes e de matéria orgânica, a queda do nível de carbono orgânico acumulado no solo o que tem influência no aquecimento global, a perda da microbiota do solo e diversos outros danos como a deposição de sedimentos em cursos hídricos provocando assoreamento e a depreciação da qualidade da água (Borrelli *et al*., 2018; Bertol; De Maria; Souza, 2019).

Muitas regiões brasileiras são acometidas por elevadas perdas de solo devido à erosão hídrica, providas seja por práticas agrícolas inadequadas, por alterações no uso e cobertura da terra oriundas do desmatamento ou pelo desrespeito a aptidão agrícola do solo. Independentemente da causa da intensificação da erosão hídrica, este é um grave problema ambiental que deve ser combatido a fim de garantir a conservação do solo. Além disso, o solo deve ser considerado um recurso natural não renovável já que as taxas de erosão do solo podem superar em muito as taxas de formação / renovação (Medeiros *et al*., 2016; Bertol; De Maria; Souza, 2019).

A fim de reduzir os impactos relacionados a erosão hídrica são necessárias medidas eficazes de conservação do solo buscando diminuir as taxas erosivas e garantir a produtividade da terra. Portanto, uma avaliação quantitativa da erosão hídrica é importante para projetar e direcionar práticas conservacionistas do solo e da água (Panagos; Katsoyiannis, 2019;

Mohammed *et al*., 2020). Buscando suprir essa necessidade foram desenvolvidas técnicas de modelagem para investigação remota da erosão hídrica. A modelagem consiste em equações matemáticas que relacionam fatores naturais como tipos de solo, chuva, vegetação e relevo ao processo erosivo. Esta técnica apresenta baixo custo de implementação e é capaz de fornecer não só uma representação das condições atuais de erosão hídrica, mas também simular as perdas de solo ocorridas no passado e possíveis circunstâncias futuras. Os modelos de erosão do solo são ferramentas úteis para aumentar nossa compreensão dos processos ambientais e apoiar a tomada de decisões no planejamento de gestão do solo e da água (Stefanidis; Stathis, 2018; Wang *et al*., 2019; Guo *et al*., 2020; Luetzenburg *et al*., 2020; Kebede *et al*., 2021).

Apesar da modelagem não causar diretamente a redução da erosão hídrica nos solos brasileiros, ela pode contribuir para o planejamento de práticas de mitigação do processo. Além disso, a modelagem em grandes áreas tem o potencial de introduzir a erosão do solo na agenda de discussões regionais e nacionais, chamar a atenção dos órgãos governamentais no Brasil e incentivar a conscientização da população a respeito da conservação do solo. Vale destacar que os resultados da modelagem podem auxiliar na proposição e adoção de políticas ambientais e agrícolas para redução dos impactos negativos da erosão (Medeiros *et al*., 2016; Alewell *et al*., 2019).

Por fim, o diagnóstico da erosão e a conservação do solo vão de encontro aos Objetivos de Desenvolvimento Sustentável (ODSs), que consistem em propósitos e metas, definidos no ano de 2015 em âmbito global pelas Nações Unidas (ONU), buscando o equilíbrio das dimensões econômicas, sociais e ambientais no desenvolvimento sustentável. A modelagem da erosão hídrica e a conservação do solo, irão atender principalmente aos ODSs relacionados ao combate à fome e incentivo à agricultura sustentável (ODS 2), combate às mudanças climáticas e aos seus impactos (ODS 13) e a recuperação e à promoção do uso consciente dos ecossistemas terrestres (ODS 15) (Sousa; Jesus; Grise, 2022; Nações Unidas, 2024).

Nesse contexto, o objetivo do trabalho foi modelar as perdas de solo por erosão hídrica em diferentes áreas tropicais brasileiras. Foram abordados cinco casos específicos: no primeiro deles foi estimado o efeito das mudanças de uso e cobertura de solo sobre a erosão hídrica na sub-bacia hidrográfica do Córrego Coroado, em Alfenas, Minas Gerais. Nessa sub-bacia ocorreram diversas mudanças no uso da terra nos últimos 30 anos, onde inicialmente a maior parte da área era destinada a atividades agrícolas, porém nos anos recentes ocorreu um processo de reflorestamento da sub-bacia, buscando atender a legislação ambiental brasileira e a demanda de uma produção agrícola de menor dano ambiental, o que é valorizado na comercialização dos produtos agrícolas.

O segundo caso consiste na Bacia Hidrográfica do Rio Xingu que é uma das áreas mais afetadas pelo desmatamento na região amazônica (Villas-Bôas, 2012). Nessa região foi avaliado o efeito do desmatamento sobre a variação espacial e temporal das perdas de solo no período entre 1988 a 2018. No terceiro caso foi feita a modelagem da erosão hídrica na Bacia do Rio Tietê, uma histórica rota de ocupação territorial e exploração dos recursos naturais do interior de São Paulo e do Brasil. No quarto caso, foram estimadas as perdas de solo considerando todo o Estado de Rondônia, uma região que sofreu severamente com o desmatamento nas últimas décadas. Por fim, foi estimado os impactos da erosão hídrica no Sistema Cantareira, uma importante região responsável pelo abastecimento de água da área metropolitana de São Paulo, a maior metrópole da América do Sul.

As áreas estudadas foram selecionadas com base na necessidade do diagnóstico da erosão hídrica e na relação com outras problemáticas ambientais: como a questão do desmatamento da região amazônica na Bacia Hidrográfica do Rio Xingu e no Estado de Rondônia, a relação com a conservação dos recursos hídricos e qualidade da água na Bacia Hidrográfica do Rio Tietê e no Sistema Cantareira e as mudanças de uso e cobertura da terra na sub-bacia hidrográfica do Córrego Coroado.

2 REVISÃO DE LITERATURA

2.1 EROSÃO DO SOLO E SEUS IMPACTOS

A erosão do solo é um processo natural de modificação da paisagem e é reconhecível somente no decorrer de longos períodos de ocorrência. A erosão natural é benéfica já que participa da formação de longas planícies, colinas suaves e vales férteis. Porém, atividades antrópicas intensificam o processo erosivo e dessa forma os agentes atmosféricos (chuva, vento) podem remover, em pouco tempo, solos que levaram séculos para serem formados (Bertoni; Lombardi Neto, 2012).

Nas regiões tropicais ocorre predominantemente erosão hídrica e em regiões áridas e semiáridas, onde a vegetação natural é escassa e sopram ventos fortes, a erosão eólica gera sérias consequências. A erosão provocada pela água pode ser classificada em erosão laminar, sulcos e voçorocas, e essas três formas de erosão podem ocorrer simultaneamente em uma mesma área. A erosão laminar é a remoção da superfície do solo sobre toda uma área, e é o tipo de erosão mais perigoso já que dificilmente é notada em campo. A erosão em sulcos consiste na concentração de fluxos no escoamento da água formando linhas mais ou menos profundas no solo. Essa é a forma de erosão que os produtores agrícolas mais se atentam e é provocada por chuvas de grande intensidade em terrenos declivosos. A ampliação dos sulcos gerados pela erosão ano após ano devido a grandes concentrações de enxurrada formam as voçorocas. A voçoroca é a forma espetacular da erosão devido ao efeito da enxurrada descontrolada sobre o solo (Bertoni; Lombardi Neto, 2012).

A erosão do solo, intensificada pelo atividades humanas, deve ser identificada e mitigada, visto que, independentemente do tipo (hídrica, eólica) e da forma de ocorrência (laminar, sulcos e voçorocas), este processo causa sérios impactos ambientais, sociais e econômicos. Existem exemplos onde a erosão do solo provocou a ruína de algumas civilizações humanas e a queda de antigos impérios. Estas civilizações alcançaram seu desenvolvimento usando recursos naturais, principalmente aqueles que sustentavam a produção agropastoril em terras de alta fertilidade. Por exemplo: os desertos do norte da China, da África, da Pérsia e do norte da Mesopotâmia se desenvolveram esgotando a fertilidade dos solos e seu potencial produtivo, o que intensificou a erosão do solo. O processo erosivo também foi um dos responsáveis pela decadência do Império Romano, devido a remoção dos nutrientes do solo e a consequente degradação de suas características químicas (Bertol; Cassol; Barbosa, 2019).

Nos dias atuais a erosão ainda é um grave problema enfrentado pela sociedade e é a principal causa de degradação dos solos (Bertol; Cassol; Barbosa, 2019). Globalmente, estima-se que a erosão hídrica gere um custo anual de 8 bilhões de dólares do Produto Interno Bruto Global, com impacto direto na segurança alimentar reduzindo a produção de alimentos em 33,7 milhões de toneladas, com aumentos nos preços mundiais variando de 0,4% a 3,5% (Sartori *et al*., 2019). Além disso, de acordo com a Organização das Nações Unidas para Alimentação e Agricultura (FAO) estima-se que 33% das terras do mundo estão degradadas (FAO, 2015).

No Brasil a perda de solo total devido à erosão hídrica foi estimada em 847 milhões de toneladas por ano (Merten; Minella, 2013). Os mesmos autores apontaram que a expansão agrícola sem a adoção de práticas conservacionistas tem o potencial de gerar um aumento na erosão do solo em 800 milhões de toneladas por ano, com pastagens degradadas, cana-de-açúcar e soja, respectivamente, representando os principais cultivos agrícolas contribuintes na geração de sedimentos.

Em estudos mais recentes estima-se que sem a adoção de prática agrícola conservacionista, o potencial anual de perda de solo é de cerca de 3,0 bilhões de toneladas, com 29,5% originário das áreas de cultivo e 61,4% das pastagens. Nesse cenário, o impacto econômico da perda de solo, com base nos custos de reposição de nutrientes com fertilizantes e calcário, foi estimado em 15,7 bilhões de dólares por ano para lavouras e pastagens (Polidoro *et al*., 2021).

Por várias décadas, até por volta de 1960, predominava no Brasil o sistema colonial de produção com pequenas lavouras anuais, situadas em áreas desmatadas. Nessas áreas, o solo era exposto com a retirada da vegetação gerando a intensificação da erosão. Entre 1960 e 1970, o cultivo com preparo do solo mais intenso, a monocultura e o elevado uso de fertilizantes passaram a ser praticados no Brasil. Assim, a intensificação da erosão era causada pela desagregação do solo durante o preparo para o plantio e o impacto das chuvas, além da compactação do solo e da redução dos teores de matéria orgânica (Bertol; Cassol; Barbosa, 2019; Polidoro *et al*., 2021).

A partir de 1970, em algumas regiões do país, os sistemas de manejo do solo com arações e gradagem passaram a ser substituídos por sistemas de manejo que conservavam a vegetação e a fitomassa residual sobre o solo, como a semeadura direta, entre outros. Estes novos sistemas de manejo provocaram a redução da erosão hídrica. Os produtores agrícolas ao verificar a redução da erosão em suas terras com a semeadura direta, passaram a eliminar outras práticas conservacionistas nas lavouras: os terraços e o plantio em contorno, já que dificultavam a mecanização dos cultivos. Com a eliminação dos terraços passou a ser realizada semeadura morro abaixo e, assim, as perdas de água e de solo foram elevadas chegando a ser tão altas quanto as perdas no preparo convencional do solo. Na década de 1990, os agricultores já reconheciam que a redução eficaz da erosão exigia a integração de práticas conservacionistas mecânicas e vegetativas, como os terraços e a manutenção de resíduos culturais (Bertol; Cassol; Barbosa, 2019; Didoné; Minella; Evrard, 2017; Polidoro *et al*., 2021).

Neste cenário, o uso intenso do solo no Brasil e a falta de práticas conservacionistas contribuem para elevar as taxas de erosão hídrica e, consequentemente, o processo de degradação do solo que atinge grande extensão do país com sérios danos, como a redução da fertilidade do solo e do seu potencial produtivo, a perda da capacidade de armazenamento de água, a formação de ravinas e voçorocas, o assoreamento e poluição dos corpos hídricos, com diminuição da qualidade da água (Hernani *et al*., 2002; Beskow *et al*., 2009).

Em muitas regiões do país ainda é verificada a intensificação da erosão hídrica, devido ao desmatamento e às mudanças de uso e cobertura da terra. Além disso, nas pastagens degradadas a situação é preocupante, com o excesso de animais a vegetação é rapidamente consumida deixando o solo exposto e compactado pelo pisoteio dos animais, que diminui a infiltração de água no solo e agrava a erosão. Estas áreas estão entre as mais intensamente erodidas do mundo, configurando um *hot spot* de erosão do solo devido a uma taxa de perda de solo superior a 20 Mg ha^{-1} ano^{-1} (Dias Filho, 2014; Borrelli *et al*., 2017; Bertol; Cassol; Barbosa, 2019; Polidoro *et al*., 2021).

Nos últimos anos também é notório o aumento de áreas com plantio de eucalipto e pinus, que são cultivos que apresentam sérios problemas de erosão, principalmente quando associados ao plantio morro abaixo. Além disso, no processo de colheita destas espécies arbóreas é feito o uso de maquinário pesado, onde a alteração e mobilização da superfície do solo é intensa, o que fragiliza o solo e o torna suscetível a erosão (Bertol; Cassol; Barbosa, 2019; Lense *et al*., 2019; Lense, 2020).

Informações sobre a distribuição espacial da erosão hídrica, em escalas compatíveis com a demanda agrícola, são importantes para o planejamento de uso conservacionista do solo, a fim de alcançar uma agricultura sustentável e reduzir o processo erosivo à medida que o uso da terra e a produção de culturas, pastagens e florestas são intensificados no país. Nesse contexto, a modelagem da erosão hídrica surge como uma ferramenta capaz de produzir dados e de fornecer diagnóstico sobre o estágio de degradação dos solos brasileiros e identificar as áreas que carecem de práticas conservacionistas e de medidas mitigadoras do processo erosivo.

2.2 AVALIAÇÃO DA EROSÃO DO SOLO NO BRASIL

As pesquisas sobre a erosão do solo no Brasil foram iniciadas há mais de sete décadas. Porém, foi apenas em meados de 1970 até os anos 2000, que houve grandes avanços no número de trabalhos, pesquisadores e instituições envolvidos na avaliação da erosão. Apesar destes trabalhos não terem sido realizados de forma sistemática, integrada e continuada, dados valiosos foram obtidos, tanto sobre a avaliação da erosão sob chuvas naturais quando em chuvas simuladas. Embora tenham ocorrido avanços nessa área do conhecimento, ainda há muito a avançar na discussão, sistematização e na padronização de métodos de pesquisa em erosão do solo (Barreto *et al*., 2008; Bertol; Cassol; De Maria, 2019).

No Brasil, as pesquisas em erosão hídrica podem ser divididas em três grupos. O primeiro é voltado principalmente a obter dados para estabelecer os fatores para os modelos preditivos de erosão, como por exemplo trabalhos que determinaram a erodibilidade de diferentes solos brasileiros e a erosividade das chuvas em diferentes regiões brasileiras (Mannigel *et al*., 2002; Sá *et al*., 2004; Saito *et al*., 2009; Silva *et al*., 2009; Martins *et al*., 2011; Aquino *et al*., 2012; Mello *et al*., 2013). O segundo grupo é voltado a aplicação dos modelos de predição da erosão hídrica principalmente em escala de bacia hidrográfica (Beskow *et al*., 2009; Silva *et al*., 2010; Ayer *et al*., 2015; Batista *et al*., 2017; Cunha *et al*., 2017; Mendes Júnior *et al*., 2018; Tavares *et al*., 2019; Sakuno *et al*., 2020; Nachtigall *et al*., 2020). Já o terceiro grupo se dedicou principalmente a alcançar dados para estabelecer a eficácia de práticas conservacionistas e a avaliação da influência de diferentes práticas de manejo do solo nas taxas de erosão hídrica (Engel *et al*., 2009; Merten *et al*., 2015; Bertol *et al*., 2017).

Estudos de avaliação da erosão hídrica em campo, ou por meio de parcelas experimentais, exigem longos períodos de monitoramento para a obtenção de dados representativos, o que envolve custos na manutenção das pesquisas e na mão de obra necessária e, além disso, muitas vezes os resultados não são capazes de representar a heterogeneidade de uma paisagem (Panagos; Katsoyiannis, 2019; Mohammed *et al*., 2020). Dadas as limitações, no Brasil grande parte dos estudos sobre a erosão hídrica consistem na aplicação de modelos preditivos, principalmente em escala de bacia hidrográfica.

A modelagem da erosão do solo consiste em realizar ações buscando representar o fenômeno da erosão por meio de uma equação ou de um conjunto de equações matemáticas. Dessa forma, o objetivo central da modelagem é estimar a resposta do processo erosivo frente aos fatores que o condicionam (Bertol; Cassol; Merten, 2019).

Quando a modelagem é combinada às técnicas de sensoriamento remoto e de Sistema de Informação Geográfica (SIG) é possível representar a distribuição espacial das perdas de solo em determinada região. A aplicação dos modelos preditivos em ambiente SIG permite sobrepor, reagrupar vários mapas temáticos e aplicar equações matemáticas para o cálculo de diferentes fatores erosivos. Por meio do SIG também é possível aplicar a modelagem em extensas áreas, há exemplos de Medeiros *et al*. (2016) e de Teodoro *et al*. (2023) que estimaram a erosão hídrica nos estados de São Paulo e Espirito Santo, respectivamente. Além disso, pela utilização de imagens de satélites, de modelos digitais de elevação do terreno e outros dados, a modelagem pode resultar em estimativas mais precisas. A distribuição espacial das perdas de solo pode servir de base científica para o planejamento e gerenciamento do uso sustentável da terra e para a identificação das áreas com elevadas taxas erosivas, as quais devem ser prioritárias para adoção de práticas conservacionistas (Haidara *et al*., 2019).

Existem diversas modelos de predição da erosão hídrica, dentre os quais, os mais importantes, para o caso do Brasil, são a Equação Universal de Perda de Solo (USLE) e sua versão Revisada (RUSLE) (Wischmeier; Smith, 1978; Renard *et al*., 1997). Podemos destacar a RUSLE por atualmente ser um modelo amplamente utilizado no Brasil e no mundo, devido à flexibilidade e praticidade para áreas com poucas informações disponíveis. A RUSLE foi elaborada a partir de dados de parcelas experimentais realizadas nos Estados Unidos da América (EUA) e estima as perdas de solo em função da erosivividade da chuva (R), erodibilidade do solo (K), fator topográfico (LS), cobertura e manejo do solo (C) e práticas conservacionistas (P) (Renard *et al*., 1997; Prasannakumar *et al*., 2012; Ganasri; Ramesh, 2016; Zerihun *et al*., 2018).

Outro modelo que nos anos recentes tem sido utilizado no território brasileiro é o Método de Erosão Potencial (EPM) (Gavrilovic, 1962). O EPM foi desenvolvido com base em pesquisas de campo onde foram realizadas medições da erosão hídrica durante 40 anos, conduzidas na bacia hidrográfica do Rio Morava, na Sérvia (Dragičević *et al*., 2019). Este modelo foi adaptado as condições edafoclimáticas brasileiras por Sakuno *et al*. (2020). As principais vantagens da utilização do EPM, é que este modelo requer poucos dados de entrada, baseia-se em valores tabelados e permite estimar tanto a fração de sedimento erodido que atinge os cursos hídricos quanto a fração depositada nas depressões do relevo (Lense, 2020).

De acordo com Tavares *et al*. (2019) e Lense *et al*. (2020a), os resultados obtidos com base na aplicação dos modelos RUSLE e EPM se aproximam e quando combinados a SIG, os modelos apresentam as distribuições espaciais de perdas de solo semelhantes, com estimativas de perdas de solos da mesma ordem de grandeza indicando as áreas prioritárias para mitigação da erosão. Portanto, ambos os modelos podem ser usados na estimativa da erosão hídrica, e a escolha de qual usar dependerá das informações disponíveis em cada área e das especificações de cada estudo sobre o processo erosivo.

A predição da erosão é importante pois, cada vez mais, os valores das perdas de solo estimados são utilizados no planejamento de ações de controle da erosão, em diversas escalas no mundo todo (Bertol; Cassol; Merten, 2019). Inclusive, em países da Europa, alguns trabalhos de modelagem da erosão hídrica em grandes áreas são usados para auxiliar na proposição e adoção de políticas ambientais e agrícolas para redução dos impactos negativos da erosão (Alewell *et al*., 2019). No Brasil, embora os estudos de modelagem sejam pouco utilizados pelos órgãos governamentais, a estimativa da erosão hídrica, bem como a compreensão da sua dinâmica temporal, pode auxiliar no planejamento da gestão sustentável de bacias hidrográficas.

Por fim, vale destacar que os modelos devem ser usados corretamente, e acima de tudo de forma consciente de que a modelagem da erosão é, como qualquer outro modelo, uma representação da realidade e não a própria realidade e, portanto, está propensa a erros, que na maioria das vezes são toleráveis (Alewell *et al*., 2019).

2.3 RUSLE

A partir de 1940 foram desenvolvidos os primeiros modelos empíricos de estimativa da erosão nos EUA, culminando nos anos posteriores na elaboração da Equação Universal de Perda de Solo (*Universal Soil Loss Equation - USLE*). A USLE foi formulada baseando-se em observações de perda de solo em mais de 10.000 parcelas experimentais em campo com 3,5 m de largura, 22,1 m de comprimento e 9% de declividade. Embora o foco principal de aplicação da USLE fosse para as condições edafoclimáticas do centro e leste dos EUA, o uso do modelo foi estendido e aplicado em todo o mundo (Wischmeier; Smith, 1978; Prasannakumar *et al*., 2012; Ganasri; Ramesh, 2016). Algumas modificações e revisões foram realizadas na metodologia da USLE e os avanços nas áreas de SIG e sensoriamento remoto, permitiram a incorporação de elementos, variáveis e novos métodos de levantamento de dados, gerando a Equação Universal de Perda de Solo Revisada (*Revised Universal Soil Loss Equation - RUSLE*) (Renard *et al*., 1997).

A RUSLE é um dos modelos mais amplamente utilizado para estimativa de erosão do solo em pequenas e grandes escalas, a curto e a longo prazo, principalmente devido sua disponibilidade de dados e flexibilidade de aplicação para diversas condições geográficas e edafoclimáticas. Além disso, este modelo pode ser integrado com técnicas de geoprocessamento melhorando a precisão de seus resultados (Ganasri; Ramesh, 2016; Back, 2023; Riquetti *et al*., 2023).

A RUSLE estima as perdas de solo conforme Equação 1 (Renard *et al*., 1997).

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

Em que, A é a perda de solo média anual, em Mg ha^{-1} ano^{-1}; R é o fator erosividade da chuva, em MJ mm ha^{-1} h^{-1} ano^{-1}; K é o fator erodibilidade do solo, em Mg ha^{-1} MJ^{-1} mm^{-1}; LS é o fator topográfico, pela relação entre o comprimento (L) e a declividade da rampa (S), adimensional; C é o fator uso e manejo do solo, adimensional, e P é o fator práticas conservacionistas, adimensional.

A erosividade (R) representa o potencial de uma precipitação pluviométrica e a enxurrada a ela associada em causar erosão em um determinado solo desprotegido. O fator R leva em consideração energia cinética da chuva e sua intensidade máxima em 30 minutos (Wischmeier; Smith, 1978). Utilizando a RUSLE, o fator R pode ser ajustado para várias condições locais e específicas, o que não ocorre quando esse fator e avaliado para a USLE (Bertol; Cassol; Merten, 2019).

Para uma determinação precisa do fator R é necessária a disponibilidade de dados sobre a precipitação em curtos intervalos de tempo e por vários anos, além da seleção minuciosa do melhor método para cálculo do fator, que deve ser baseado em estudos climáticos locais e dados de alta precisão (Panagos *et al*., 2015a). Porém, a falta de registros detalhados sobre as precipitações pluviométricas no território brasileiro é um obstáculo para produzir estimativas de erosividade precisas e confiáveis (Teixeira *et al*., 2022a).

Dessa forma, métodos simplificados usando dados mensais e anuais de pluviômetros são uma das alternativas para prever o fator R no Brasil. Com base na boa confiabilidade e facilidade de uso dessas equações, diversos autores estabeleceram metodologias no país, como o caso do modelo geográfico multivariado, proposto por Mello *et al*. (2013), que permite estimar a erosividade média anual com base no Modelo Digital de Elevação (MDE), usando a latitude, longitude e altitude da área estudada. Este tipo de abordagem possibilita aos pesquisadores e demais profissionais estimar o potencial de erosividade das chuvas para qualquer região do Brasil, construindo um modelo matemático prático para o manejo do solo e da água (Mello *et al*., 2013; Teixeira *et al*., 2022b). Além disso, o fator R pode ser determinado usando dados climáticos de ferramentas de geoprocessamento como o *WorldClim*, que consiste em um conjunto de grades climáticas globais com resolução espacial de cerca de 1 km^2 e o *Global rainfall erosivity* database - *GloREDa*, um banco de dados com valores do fator R calculados a longo prazo com base em 3.625 estações de 63 países ao redor do mundo (Hijmans *et al*., 2005; Panagos *et al*., 2017).

No Brasil, os maiores níveis de erosividade das chuvas são encontrados na região Norte, chegando a valores superiores a 22.000 MJ mm ha^{-1} h^{-1} ano^{-1}, o que segundo Foster, McCool e Renard (1981) corresponde a um potencial de erosividade das chuvas muito alto. Tal fato pode ser explicado pelo elevado volume de chuvas na região, com um valor médio anual de precipitação superior a 3.000 mm. O nordeste brasileiro apresenta o menor potencial de erosividade das chuvas no país, com valores variando de 2.224 a aproximadamente 6.000 MJ mm ha^{-1} h^{-1} ano^{-1}, que segundo Foster, McCool e Renard (1981) consiste em áreas classificadas como de média erosividade. Porém, esta região apresenta alta variabilidade espacial e temporal das precipitações, as quais são instáveis e típicas do clima semiárido (Mello *et al*., 2013; Teixeira *et al*., 2022b). No Sudeste, o fator R também varia muito espacialmente, onde as maiores magnitudes são observadas na região costeira. No Estado de São Paulo, a erosividade anual possui um valor médio de 6.952 MJ mm ha^{-1} h^{-1} ano^{-1}, valor próximo a média nacional que é de 5.620 MJ mm ha^{-1} h^{-1} ano^{-1} (Teixeira *et al*., 2022a; Teixeira *et al*., 2023).

A erodibilidade (K) consiste na suscetibilidade de um solo em sofrer desprendimento de suas partículas pelo impacto da gota de chuva e pelo escoamento superficial da enxurrada (Renard *et al*., 1997). O fator K pode ser determinado de duas formas. A primeira delas é o método direto que consiste na determinação deste fator em parcelas experimentais padronizadas, conforme Wischmeier e Smith (1978), onde os dados de perda de solo são coletados por no mínimo 22 anos em solo permanentemente descoberto, com ocorrência de chuvas naturais ou simuladas (Bertol; Cassol; Merten, 2019). Embora as medições diretas sejam a forma mais confiável de determinar o fator K, esse método é caro e demorado e por essas razões os dados medidos e disponíveis em parcelas experimentais são escassos no Brasil, com estudos dispersos no território e que desconsideram séries de dados de longo prazo (Barbosa *et al*., 2019; Godoi *et al*., 2021).

O fator K também pode ser determinado por métodos indiretos baseados em modelos matemáticos que usam dados dos atributos físicos, químicos e mineralógicos do solo. O nomograma proposto por Wischmeier e Smith (1978), também chamado de nomograma da USLE, é o método indireto mais utilizado para determinar a erodibilidade do solo. Essa equação considera características do solo como textura, estrutura e teor de matéria orgânica (Bertol; Cassol; Merten, 2019; Godoi *et al*., 2021). As estimativas do fator K pelo nomograma da USLE no Brasil variam de 0,0002 a 0,0636 Mg ha^{-1} MJ^{-1} mm^{-1}, com valor médio de 0,0181 Mg ha^{-1} MJ^{-1} mm^{-1}. Os maiores valores do fator K no país ocorrem na Amazônia Ocidental, onde as florestas são a principal proteção do solo contra a ação das chuvas, e assim, áreas com desmatamento podem ser severamente comprometidas pela erosão. Os Latossolos são os solos com menor valor do fator K no Brasil, no entanto podem ser fontes de altas perdas de solo em lavoura com cultivo sem adoção de práticas de conservação do solo (Godoi *et al*., 2021).

Além do nomograma, foram desenvolvidas várias equações alternativas para estimar o fator K, e como resultado, no Brasil os valores de erodibilidade de solo variam muito, inclusive em áreas com características edafoclimáticas semelhantes. Dessa forma, é necessária uma uniformização das metodologias, para que os valores de K possam ser comparados entre si nas mais variadas condições de obtenção (Bertol; Cassol; De Maria, 2019). Além disso, em estudos onde o fator K é determinado por meio de levantamentos bibliográficos, os pesquisadores e demais envolvidos devem realizar uma extensa revisão buscando verificar a metodologia usada e o local onde o fator K foi calculado, a fim de evitar possíveis erros, como por exemplo, quando um mesmo valor do fator K é usado para áreas com características edafoclimáticas totalmente distintas.

O fator LS reflete a influência do relevo na erosão do solo. Esse fator é formado pelo Comprimento do Declive (L), que é a razão de perdas de solo que ocorre em um determinado comprimento de rampa com a perda de solo que ocorre no comprimento de 22,1 m, e pela Declividade do Terreno (S), que representa a razão de perda de solo que ocorre em determinada declividade e aquela correspondente a uma declividade de 9% (Wischmeier; Smith, 1978). Com o auxílio de SIG e usando MDE foram criados métodos que possibilitaram as estimativas automáticas dos fatores L e S para relevos mais complexos. Entre as várias metodologias para o cálculo do fator LS disponíveis na literatura, o modelo proposto por Moore e Burch (1986) tem tido destaque, já que é capaz de representar o fator LS em bacias hidrográficas (Beskow *et al*., 2009; Galdino, 2015). Moore e Burch (1986) baseiam a determinação do fator LS na teoria da Potência Unitária do Escoamento. Essa teoria considera que a água na superfície do solo possui a energia capaz de desagregar e transportar partículas de solo quando estes sedimentos se movem na direção do declive (Yang, 1984).

A resolução espacial do MDE influencia o cálculo do fator LS e, consequentemente, as estimativas de perda de solo. Os MDE com maior resolução representam alterações geomorfológicas com maior precisão, resultando em estimativas mais precisas do fator topográfico. O ideal para o cálculo do fator LS é que os MDE estejam na faixa de resoluções de 3 a 90 m, sendo que os MDE de resoluções mais altas são preferíveis e devem ser usados sempre que possível (Lia; Zhang; Liu, 2021). No Brasil a obtenção de MDE com maior resolução espacial é um problema, visto que na maioria das vezes este tipo de resolução é encontrada apenas em pequenas bacias hidrográficas monitoradas, onde os pesquisadores fazem um levantamento detalhado do relevo devido ao menor gasto envolvido quando comparado a grandes áreas (Beskow *et al*., 2009).

Entre os fatores da RUSLE, o fator C é o mais importante pois representa todas as variáveis de manejo do solo, cobertura vegetal e biomassa residual das plantas que influenciam na erosão. É o fator mais complexo para ser determinado e é o fator pelo qual as ações antrópicas podem interferir mais fortemente na erosão. A cobertura e manejo do solo (C) consiste na razão entre a perda de solo que ocorre em uma parcela experimental com algumas formas de cobertura vegetal e a perda de solo que ocorre em solo descoberto. Esse fator também é obtido em campo por experimentos de longa duração. Além disso, o fator C é um dos mais sensíveis a variações espaço - temporais, pois o parâmetro segue o crescimento da vegetação e a dinâmica das chuvas (Wischmeier; Smith, 1978; Nearing *et al*., 2005; Bertol; Cassol; De Maria, 2019).

Os valores do fator C variam de 0 a 1, onde menores valores são observados em vegetações densas que geram boa proteção do solo e com isso menores taxas de erosão. Valores de C mais próximos a 1 demostram que há redução na vegetação e risco de altos níveis de erosão (Wischmeier; Smith, 1978).

Assim como o fator K, os valores do fator C calculados em parcelas experimentais no Brasil são escassos. Em grande maioria dos trabalhos de modelagem da erosão hídrica realizados em território brasileiro o fator C é determinado a partir de um valor constante encontrado na literatura, o qual foi obtido em parcela experimentais muitas vezes desenvolvidas para regiões diferentes da área de estudo. Essa metodologia não é capaz de representar a heterogeneidade espacial da cobertura vegetal sobre o solo. Buscando contornar estas situações, metodologias para estimar este fator com base em SIG e sensoriamento remoto foram desenvolvidas, como por exemplo, a metodologia de Durigon *et al*. (2014) onde o fator C é calculado por sensoriamento remoto em regiões tropicais utilizando o Índice de Vegetação por Diferença Normalizada (NDVI) (Durigon *et al*., 2014; Almagro *et al*., 2019; Lense *et al*., 2020b).

O uso de metodologias de cálculo do fator C baseadas em índices de vegetação, como o NDVI, é especialmente importante em regiões tropicais, como o Brasil, onde a variação climática anual está diretamente relacionada as mudanças de uso e cobertura do solo, e dessa forma é possível captar as variações espaciais e temporais da vegetação. Além disso, os novos estudos de cálculo do fator C por sensoriamento remoto em território brasileiro indicam um caminho de rotinas que melhoram a precisão das informações obtidas pelas imagens de satélite usando dados meteorológicos (Almagro *et al*., 2019; Lense *et al*., 2020b; Macedo *et al*., 2021).

O fator P representa os efeitos das práticas conservacionistas de suporte na estimativa da erosão hídrica. Esse fator é calculado pela razão de perda de solo ocorrida em uma dada prática conservacionista de suporte em relação a perda de solo ocorrida em parcela padrão. As práticas conservacionistas de suporte são complementares as práticas básicas e na RUSLE são representadas por três tipos: cultivo em contorno, cultivo em faixas com rotação de culturas e terraceamento. O fator P varia de 0 a 1, onde o valor 1 é atribuído a áreas sem a utilização de práticas conservacionistas, desta forma, quanto menor o valor de P melhor a eficácia da prática conservacionista em reduzir as perdas de solo (Renard *et al*., 1997; Bertol; Cassol; De Maria, 2019).

Os valores de P podem ser determinados a partir de classificações de imagens usando dados de sensoriamento remoto e conhecimento específico da área estudada. Essa abordagem requer conjuntos de imagens de alta resolução e levantamentos experimentais na área estudada, que em sua maioria são escassos. Na literatura é possível encontrar várias tabelas e fórmulas que propõem valores do fator P para as diferentes práticas conservacionistas de suporte, como por exemplo Wischmeier e Smith (1978), Renard *et al*. (1997) e Bertoni e Lombardi Neto (2012). Porém, é difícil quantificar e identificar as diferentes práticas conservacionistas de suporte aplicadas em áreas muito grandes (Panagos *et al*., 2015b).

Uma abordagem alternativa para a estimativa do fator P é a utilização de equações empíricas que se baseiam na declividade como a propriedade chave para práticas de conservação do solo (Silva *et al*., 2010; Medeiros *et al*., 2016). Além disso, Panagos *et al*. (2015b) relatam que nos tempos atuais, com a grande disseminação de smartphones, cada agricultor poderia tirar uma fotografia das práticas conservacionistas de suporte adotadas na sua lavoura. Estas fotos com data e coordenadas do Sistema de Posicionamento Global (GPS) seriam registradas em um banco de dados e utilizadas para diversos fins, como o levantamento do fator P em grandes áreas, validação de resultados de modelagem de erosão e diagnóstico da disseminação de práticas conservacionistas de solo.

Como visto, podem ser aplicadas diversas metodologias para determinação dos fatores da RUSLE e, portanto, a utilização deste modelo no Brasil exige um trabalho meticuloso por parte dos pesquisadores da área de ciência do solo no levantamento e cálculo dos parâmetros do modelo a fim de melhorar a precisão dos resultados.

2.4 EPM

O Método de Erosão Potencial (*Erosion Potential Method - EPM*), também conhecido como método Gavrilović, é um modelo de estimativa da erosão hídrica em bacias hidrográficas que foi desenvolvido com base em pesquisas de campo nas décadas de 60 e 70 na Ex-Iugoslávia (Gavrilovic, 1962). O EPM foi amplamente utilizado nas regiões dos Balcãs, na Europa, Oriente Médio e Norte da África (Efthimiou; Lykoudi; Karavitis, 2017; Stefanidis; Stathis, 2018; Tavares *et al*., 2019; Bezak *et al*., 2024). Recentemente, o modelo tem sido aplicado em bacias hidrográficas brasileiras com resultados precisos (Lense *et al*., 2019; Tavares *et al*., 2019; Lense, 2020; Sakuno *et al*., 2020).

A aplicação do EPM é vantajosa em relação a outros modelos devido a forma de obtenção dos parâmetros, já que o método requer poucos dados de entrada e baseia-se em valores tabelados. Além disso, o modelo estima a fração de sedimento erodido que atinge os cursos hídricos e a fração depositada nas depressões do relevo, o que não pode ser determinado usando USLE e RUSLE (Lense *et al*., 2019).

O EPM leva em consideração fatores dependentes do clima, das propriedades do solo, das características topográficas, uso da terra e do grau das feições erosivas da bacia hidrográfica. Originalmente, o modelo é constituído de equações segmentadas e o resultado das estimativas de perda de solo é expresso em m^3 ano^{-1} (Gavrilovic, 1962). Buscando facilitar a compreensão do modelo, Lense *et al*. (2020c) representaram a metodologia do EPM, conforme Equação 2. Além disso, os mesmos autores introduziram na fórmula a variável densidade do solo (Bd) a fim de converter os resultados de m^3 ano^{-1} para Mg ano^{-1}.

$$W_{yr} = \left(\sqrt[2]{\frac{t_0}{10}} + 0{,}1\right) \times H_{yr} \times \pi \times \sqrt[2]{\left[Y \times X_a \times \left(\varphi + \sqrt[2]{I_{sr}}\right)\right]^3} \times Bd \times F \quad (2)$$

Em que: W_{yr} = perda total de solo em Mg ano^{-1}; t_0 = temperatura média do ar, em °C ano^{-1}; H_{yr} = precipitação anual média, em mm ano^{-1}; Y = resistência do solo à erosão hídrica, admensional; X_a = coeficiente de uso e manejo do solo, admensional; φ = coeficiente do grau das feições erosivas, admensional; I_{sr} = declividade média em %, Bd = densidade média dos solos em kg dm^{-3} e F = área (ha).

A temperatura média do ar (t_0) e a precipitação pluviométrica média (H_{yr}) são os fatores climáticos que compõem o EPM. Estes fatores podem ser determinados com base em anos específicos, o que é importante em estudos que avaliam as variações temporais da erosão hídrica, assim como também podem ser consideradas médias históricas de clima. Além disso, o EPM considera a temperatura média do ar em sua metodologia, o que não ocorre na RUSLE. Vale ressaltar que dentre os parâmetros envolvidos nas estimativas do modelo EPM, a t_0 é um dos fatores de menor influência nos resultados (Dragičević; Stepic, 2006).

A resistência do solo à erosão hídrica (Y) varia de acordo com o tipo de solo da bacia hidrográfica. Seus valores são tabulados de 0,20 a 2,00 onde quanto menor o valor maior a resistência do solo à erosão (Tabela 1).

Tabela 1 - Resistência do solo à erosão hídrica (Y).

Sistema Brasileiro de Classificação de Solos (SBCS)	Y
Afloramento Rochoso	0,2
Neossolos Flúvicos, Gleissolos, Organossolos, Planossolos	0,4 - 0,5
Chernossolos, Latossolos, Nitossolos	0,6 - 0,8
Argissolos, Cambissolos, Luvissolos, Plintossolos	0,9 - 1,1
Espodossolos, Vertissolos, Neossolos Regolíticos	1,2 - 1,5
Neossolos Litólicos, Neossolo Quartzarênicos; Neossolos Regolíticos	1,5 - 2,0

Fonte: Adaptado de Sakuno *et al*. (2020).
Nota: SBCS adaptado de Santos *et al.* (2018) e afloramento rochoso não é uma classe de solo.

O coeficiente de uso e manejo do solo (X_a) representa a ação da cobertura vegetal na proteção da bacia hidrográfica contra as precipitações pluviométricas. Os valores variam de 0,05 para áreas com vegetação densa a 1,00 em áreas com solo exposto (Tabela 2). Os parâmetros Y, X_a são determinados com base em valores tabelados que inicialmente foram propostos por Gavrilovic (1962) e posteriormente adaptados as condições de clima e solo do Brasil (Sakuno *et al*., 2020). Os coeficientes X e Y são os fatores cuja variação mais influência nos resultados do EPM e, dessa forma, são os parâmetros que exigem uma maior atenção dos pesquisadores para a sua determinação e uma adequada calibração (Bezak *et al*., 2024).

Tabela 2 - Valores do coeficiente de uso e manejo do solo (X_a).

Uso e manejo	X_a
Mata	0,05 – 0,30
Pastagem	0,30 – 0,50
Cultivos com manejo conservacionista	0,50 – 0,70
Cultivos com manejo convencional	0,70 – 0,90
Solo exposto	0,90 – 1,00

Fonte: Adaptado de Gavrilovic (1962); Sakuno *et al*. (2020).

O coeficiente φ é obtido pela identificação e observação das feições erosivas ocorridas em determinada área, as quais são mensuradas de forma quantitativa com valores variando de 0,1, em áreas com erosão muito fraca, até 1, em áreas com erosão severa (Tabela 3) (Gavrilovic, 1962; Spalevic, 2011). Esse parâmetro pode ser determinado por levantamentos de campo, localizando a forma de erosão mais comum na área, ou com base em imagens de satélite. A influência da topografia no EPM é representada pela declividade (I_{sr}), que tem um efeito menor na variação dos resultados do EPM em relação a Y e X_a, ao contrário do modelo RUSLE, onde o fator topografia é um dos parâmetros de maior influência nos resultados (Bezak *et al.*, 2024).

Tabela 3 – Coeficiente do grau das feições erosivas (φ).

Tipo de erosão	φ
Área com erosão severa (voçoroca)	0,9 - 1,0
Erosão em sulcos	0,8 - 0,9
Erosão laminar	0,2 - 0,7
Áreas cobertas por vegetação nativa	0,10

Fonte: Adaptado de Gavrilovic (1962); Spalevic (2011).

Além da estimativa da erosão hídrica, o EPM inova ao introduzir em sua estrutura o Coeficiente de Retenção (R_u) que permite estimar a fração de solo erodido que atinge os corpos hídricos, caracterizando a entrega de sedimentos. O R_u é calculado conforme Equação 3.

$$R_u = \frac{(O \times D)^{0,5}}{0,25 \times (L+10)} \qquad (3)$$

Em que: O = perímetro da bacia hidrográfica, em km; D = diferença média de elevação, em km; L = comprimento da bacia hidrográfica, em km.

Em grandes bacias hidrográficas, o R_u tende a gerar erros nas estimativa e dessa forma podem ser utilizadas outras equações de determinação da geração de sedimento, como Vanoni (1975), ou abordagens mais sofisticadas que consideram a conectividade de sedimentos (Lense *et al.*, 2020c; Bezak *et al.*, 2024).

O EPM pode ser aplicado em todo o território brasileiro principalmente em áreas com dados escassos. A determinação dos parâmetros do modelo não envolve cálculos complexos como o que acontece na determinação dos parâmetros da RUSLE. Por outro lado, a seleção de dados de alta precisão temporal e espacial, a adequada seleção e estruturação dos valores tabelados do modelo, a calibração dos parâmetros e a validação do resultados a fim de reduzir os erros da modelagem é um processo árduo que exige empenho e dedicação dos pesquisadores.

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3 ARTIGO 1 - EFFECT OF SPATIAL-TEMPORAL VARIATION OF LAND USE AND LAND COVER ON SOIL EROSION

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ABSTRACT - Land use and land cover changes are the main factors of human influence on the erosive process. Thus, this work aimed to evaluate the effect of land use and land cover changes over 30 years on water erosion in a tropical subbasin in southeastern Brazil. The hypothesis was tested that the expansion of coffee and reforestation areas decreased soil losses due to water erosion. The Potential Erosion Method (EPM) was used to estimate water erosion in 1988, 1998, 2008 and 2018. In the first two decades, the predominant land use in the subbasin was corn, while in 2008 and 2018, coffee and reforestation areas became the main land use class in the area. The acquisition of EPM input parameters and data analysis was performed using remote sensing techniques and the Geographic Information System. Between 1988 and 1998, the total soil loss increased by 50.36 Mg year^{-1} due to the conversion of pasturelands to coffee plantations and the increase of deforestation. However, between 1998 and 2018, there was a soil loss reduction of 660.21 Mg year^{-1} (-37.46%), once the conversion of pasture and corn areas to coffee with the adoption of conservation practices, and the expansion of reforestation areas among 1988 - 2018, contributed to the decrease of soil erosion rates.

Keywords: Soil conservation. Modelling. Erosion Potential Method.

EFEITO DA VARIAÇÃO ESPAÇO-TEMPORAL DO USO E COBERTURA DA TERRA SOB A EROSÃO DO SOLO

RESUMO - As mudanças no uso e cobertura da terra são os principais fatores de influência humana no processo erosivo. Dessa forma, o trabalho teve como objetivo avaliar o efeito das mudanças de uso e cobertura da terra ao longo de 30 anos sob a erosão hídrica em uma sub-bacia tropical no sudeste do Brasil. Foi testada a hipótese de que a expansão das áreas de café e reflorestamento diminuíram as perdas de solo por erosão hídrica. Utilizou-se o Método de Erosão Potencial (EPM) para estimar a erosão hídrica em 1988, 1998, 2008 e 2018. Nas duas primeiras décadas, o uso da terra predominante na sub-bacia era o milho, enquanto em 2008 e 2018, o café e as áreas de reflorestamento se tornaram as principais classe de uso da terra da área. A aquisição dos parâmetros de entrada do EPM e a análise dos dados foi realizada por meio de técnicas de sensoriamento remoto e Sistema de Informações Geográficas. Entre 1988 e 1998, a perda total de solo aumentou 50,36 Mg ano^{-1} devido à conversão de pastagens em plantações de café e o aumento do desmatamento. Entre 1998 e 2018, houve uma redução nas perdas de solo de 660,21 Mg ano^{-1} (-37,46%). A conversão de áreas de pastagem e milho em cultivos de café com adoção de práticas conservacionistas e a expansão de áreas de reflorestamento entre 1988-2018 contribuíram para a diminuição das taxas de erosão do solo.

Palavras-chave: Conservação do solo. Modelagem. Método de Erosão Potencial.

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INTRODUCTION

Water erosion is the main form of land degradation worldwide, and it can compromise food, fiber, and energy production, as well as the maintenance of other soil ecosystem services (FAO, 2017). The erosion process affects both the sediment generation area, and deposition site, with physical and socioeconomic impacts, such as reduced agricultural productivity, loss of water storage capacity, nutrient and organic matter removal, and siltation and pollution of watercourses (BESKOW et al., 2009; AVANZI et al., 2013; POSTHUMUS et al., 2015). Land use and land cover are recognized as the main factors of human influence on soil erosion (OUYANG et al., 2010; DEVÁTÝ et al., 2019). The cultivation of new crops without environmental planning decreases soil protection, which increases the runoff process and the erosive intensity.

Brazil have been facing land-use changes with a rapid expansion of some crops since the 1980s, which has also brought changes in the erosion dynamics (MANZATTO; FREITAS JUNIOR; PERES, 2002; DIAS et al., 2016). These changes can be observed in the Coroado Stream subbasin, located in the state of Minas Gerais, southeastern Brazil, which is an ideal place for studies on the relationship between land-use change and water erosion.

In the 1980s, pasture and temporary crops were the predominant land-uses in the subbasin area, as well as most of the southeast region of the country (DIAS et al., 2016). Since 1990, due to government incentives and economic reasons, much of the subbasin was turned into coffee-growing areas (SIMÕES; PELEGRINI, 2010). Moreover, the subbasin experienced a reforestation process to comply with the Brazilian Forest Code (BRASIL, 2012). Thus, significant land-use changes occurred in the area with the replacement of pastures and temporary crops by coffee and reforestation.

Simulation models are an effective way to assess the impacts of land use and land cover changes on soil erosion (VANWALLEGHEM et al., 2017). The use of models with little input data requirements such as Erosion Potential Method (EPM) (GAVRILOVIC, 1962) can support environmental planning in areas exposed to high erosion risks. The EPM is an empirical soil erosion model that takes into account individual subbasin parameters, concerning climate, geological, topographic, and land use and management. Moreover, it is a simple and inexpensive method that can be associated with remote sensing techniques and Geographic Information Systems (GIS) to identify areas of high vulnerability to the erosive process, providing accurate and reliable results (EFTHIMIOU; LYKOUDI; KARAVITIS, 2017).

Although land cultivation generally accelerates the erosion process, understanding the effects of land use and land cover on the spatial and temporal dynamics of erosion, especially in areas under long-term agricultural use, is a critical and necessary issue to identify efficient mitigation measures (DIDONÉ; MINELLA; EVRARD, 2017; OUYANG et al., 2018). Thus, the work aimed to evaluate the effects of the land use and land cover changes over 30 years on water erosion in a tropical subbasin in southeastern Brazil. The hypothesis that the expansion of coffee and reforestation areas decreased the soil losses by water erosion was tested.

MATERIAL AND METHODS

Description of the study area

The study was carried out at the Coronado Stream subbasin that belongs to the Rio Grande basin. The area is located at Capoerinha farm, owned by Ipanema Agrícola SA, in Alfenas, Minas Gerais, Brazil, at coordinates illustrated in Figure 1 (Geographic Coordinate System Datum Sirgas 2000). The Slope Map was elaborated in the ArcMap 10.3 environment, applying the Slope tool (ESRI, 2015), using the Digital Elevation Model (DEM), extracted from the Alfenas Topographic Map, with a spatial resolution of 10 m (IBGE, 1970).

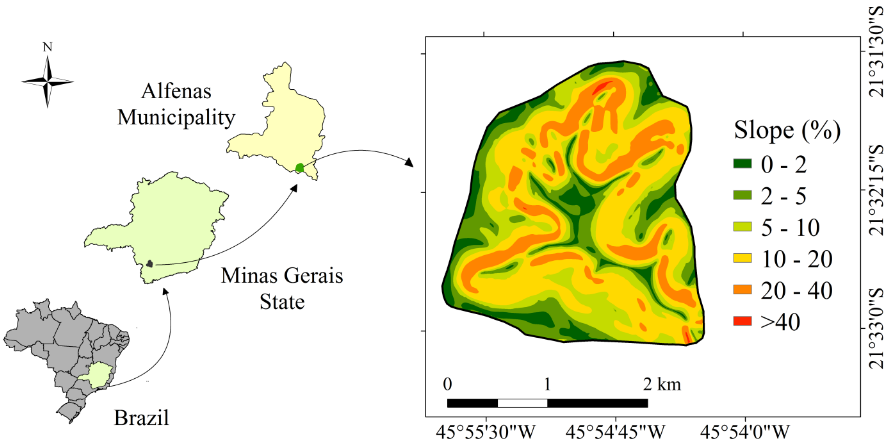

Figure 1. Location and declivity Map of the Coroado Stream Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil.

The climate was classified according to Köppen's classification as Humid subtropical with dry winter and temperate summer (ALVARES et al., 2013), while the soil was sorted as Oxisols (SOIL SURVEY STAFF, 2014). The structure of Oxisols in the subbasin is granular of moderate degree with medium size. The soils in the area have a clayey texture and an average organic matter content of 2.56 dag kg^{-1} (LENSE et al., 2019).

The land use and occupation map (Figure 2) was generated based on the Capoeirinha farm occupation history and images of the Landsat-5 Thematic Mapper (TM) satellites, bands 1, 2 and 3 and Landsat-8 Operational Land Imager (OLI), bands 2, 3 and 4, both corresponding to orbit/point 219/75, obtained from the Imaging Division (INPE, 2019). The following dates were selected: July 05, 1988; July 07, 1998; July 01, 2008 for Landsat-5 TM, and July 13, 2018 for Landsat-8 OLI, once favorable atmospheric conditions were needed. The image processing was performed using ArcGIS 10.3 (ESRI, 2015), and the land-use classification, built from these images, is shown in Table 1.

The soil loss was estimated by water erosion in 1988, 1998, 2008, and 2018. In the first two decades, the predominant land-use in the subbasin was corn, while in 2008 and 2018, coffee became the majority land use class in the area (Table 1).

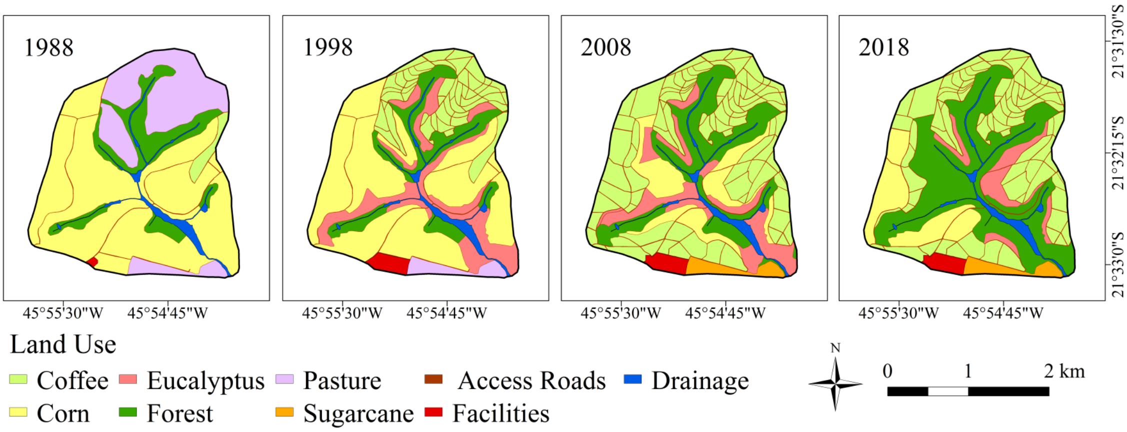

Figure 2. Map of land use classes of the Coroado Stream Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil, in the years 1988, 1998, 2008 e 2018.

Table 1. Land use classes of the Coroado Stream Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil, in the years 1988, 1998, 2008 and 2018.

Land Use	Area (%)			
	1988	1998	2008	2018
Pasture	22.91	2.77	0.00	0.00
Coffee	1.66	16.78	45.56	36.43
Corn	58.21	45.50	12.49	11.70
Forest	12.98	11.90	18.32	34.66
Sugarcane	0.00	0.00	2.70	2.70
Eucalyptus	0.00	15.80	12.37	6.11
Access Roads	0.91	2.41	3.66	3.50
Drainage	3.13	3.13	3.13	3.13
Facilities	0.20	1.71	1.77	1.77

Erosion Potential Method (EPM)

The soil losses by the EPM were estimated using the equations described in Table 2.

Determination of parameters without temporal variation

The area of the subbasin is 5,595 km^2 (F), with an average altitude of 861 m. The perimeter corresponds to 9.28 km (O), and the extension of the area, measured from the watercourses, is 3.32 km (L). The average elevation difference (D) was 66 m, obtained from the difference between the average (861 m) and the minimum altitude (795 m).

The coefficient of retention (R_u) was quantified based on the parameters O, D, and L. The R_u was estimated in 0.118, indicating that 88.20% of the total sediments (W_{yr}) generated from the erosion process is retained in the subbasin. Therefore, only 11.80% represent the real soil loss (G_{yr}), once it is what reaches the water systems. The area has a mean slope (I_{sr}) of 13.54% (Figure 1) and presents a predominance of rolling relief (8 - 20%).

Soil resistance to erosion (Y) changes according to the geology and the soil type. The Y values range from 0.20 (soils with high erosion resistance) to 2.0 (soils with a low erosion resistance). According to Gavrilovic (1962), Oxisols present a Y value of 0.8. The EPM results were converted from m^3 $year^{-1}$ to Mg $year^{-1}$ using subbasin soil average density (Ds) of 1.22 kg dm^{-3} (LENSE et al., 2019).

Table 2. Equations and descriptions of the parameters used to estimate soil losses in the Erosion Potential Method.

Equation	Parameters
(1) $W_{yr} = T \cdot H_{yr} \cdot \pi \cdot \sqrt[2]{Z^3} \cdot F \cdot Ds$	W_{yr} = Annual erosion (Mg $year^{-1}$) T = Coefficient of temperature (dimen.) H_{yr} = Mean annual rainfall (mm $year^{-1}$) Z = Coefficient of erosion (dimen.) F = subbasin area (km^2) Ds* = Soil density (kg dm^{-3})
(2) $G_{yr} = W_{yr} \cdot R_u$	G_{yr} = Soil loss (Mg $year^{-1}$) R_u = Coefficient of retention (dimen.)
(3) $T = \sqrt[2]{\frac{t_0}{10}} + 0.1$	t_0 = Mean air temperature (°C $year^{-1}$)
(4) $Z = Y \cdot X_a \cdot \left(\varphi + \sqrt[2]{I_{sr}}\right)$	Y = Soil resistance to erosion (dimen.) X_a = Coefficient of soil use and management (dimen.) φ = Coefficient of visible erosion features (dimen.) I_{sr} = Mean slope (%)
(5) $R_u = \frac{(O \cdot D)^{0,5}}{0.25 \cdot (L+10)}$	O = Basin perimeter (km) D = Difference in basin elevation (m) L = Length of basin (km)

Notes: dimen. = dimensionless. *Parameter incorporated into the original formula for conversion of m^3 $year^{-1}$ to Mg $year^{-1}$. Source: Gavrilovic (1962).

Determination of parameters with temporal variation

The soil use and management coefficient (X_a) expresses the protection of an area against rainfall action, according to the soil cover. The values range from 0.1 (areas with dense vegetation) to 1.0 (areas with exposed soil). The coefficient of visible erosion features (φ), ranges from 0.1 (sites without signs of erosion) to 1 (sites with severe erosive features-gullies). The coefficients X_a and φ were determined for each year (1988, 1998, 2008, 2018) using the satellite images, and they were classified for each land-use class according to Gavrilovic (1962) (Table 3).

Table 3. Classification of the coefficients: land use and management (X_a) and visible erosion features (φ).

Land Use	X_a
Bare Soil	0.9 - 1.0
Annual cultivation	0.7 - 0.9
Perennial cultivation	0.5 - 0.7
Pasture	0.3 - 0.5
Native Vegetation	0.1 - 0.3
Coefficient of visible erosion features	**φ**
Whole subbasin affected by erosion	0.9 - 1.0
50 - 80% of catchment area affected by surface erosion	0.8 - 0.9
Erosion in rivers, gullies and alluvial deposits	0.5 - 0.8
Erosion in subbasin on 20 - 50% of the catchment area	0.3 - 0.5
Little erosion on subbasin	0.1 - 0.3

Source: Adapted from Gavrilovic (1962) and Efthimiou, Lykoudi and Karavitis (2017).

The erosion coefficient (Z) expresses the soil erosion susceptibility in which values close to 0 indicate lower risks. The Z is determined using Y, X_a, φ, and the mean slope of the area (I_{sr}). The annual precipitation (H_{yr}) and the mean air temperature (t_0) represent the interference of climatic factors in the erosive process, and these parameters were obtained from the "Instituto Nacional de Meteorologia" database (INMET, 2019). The temperature coefficient (T) was determined using t_0. The EPM parameters with temporal variation are shown in Table 4.

Table 4. Parameters values of the Erosion Potential Method in the Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil, in different periods.

Parameters	Year			
	1988	1998	2008	2018
X_a (dimen.)	0.58	0.60	0.54	0.45
φ (dimen.)	0.57	0.57	0.48	0.40
Z (dimen.)	0.44	0.46	0.38	0.30
t_0 (°C year^{-1})	19.93	20.32	20.04	20.62
H_{yr} (mm year^{-1})	1,401.50	1,366.30	1,417.70	1,408.00
T (dimen.)	1.51	1.53	1.52	1.54

Notes: X_a = Average of Coefficient of soil use and management; φ = Average of Coefficient of visible erosion features; Z = Average of Coefficient of erosion; t_0 = Mean air temperature; H_{yr} = Mean annual rainfall; T = Coefficient of temperature; dimen = dimensionless.

The EPM calculations and the spatial distribution of the results were performed at ArcGIS 10.3 (ESRI, 2015) using the Raster Calculator tool.

Model Validation

The soil loss estimates were validated using the total sediment observed (Observed SD), obtained with the water discharge and the daily runoff data, according to Beskow et al. (2009). Data of total solids in the water and the respective discharge from 2008 to 2018, obtained from a weather station operated by the Instituto Mineiro de Gestão das Águas (IGAM), were used for this task. This station is located near the Coroado Stream exutory at coordinates 45°53'35" W and 21°39'55" S. There were no data collected by IGAM before 2007.

Initially, it was built a discharge curve connecting the total transported sediment with the water discharge (Figure 3). Then, the annual transported sediment (Observed SD) was calculated for each year, taking into account the flow x sediment curve, and the daily runoff dataset for 1988, 1998, 2008, and 2018, obtained from the "Agência Nacional de Águas" (ANA, 2019). The Observed SD was then compared with the soil loss estimate provided by the EPM.

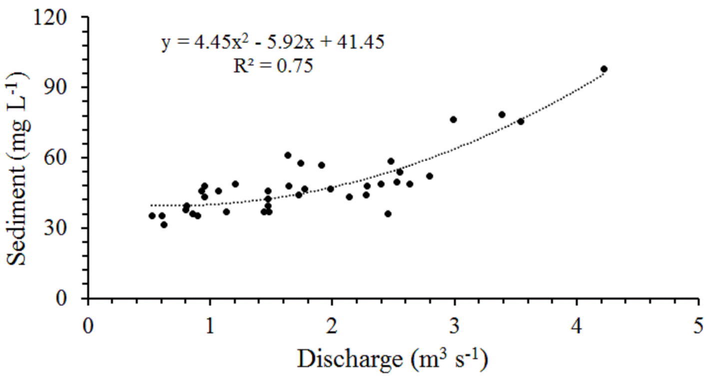

Figure 3. Water discharge curve (sediment transported × water discharge), of the Coroado Stream Subbasin, Alfenas, south of Minas Gerais State, Brazil.

RESULTS AND DISCUSSION

Correlation between the observed and the estimated sediment delivery

The subbasin presented moderate susceptibility to erosion in 1988 and 1998 and low susceptibility in 2008 and 2018. The greatest susceptibility to erosion observed in the years 1988 and 1998, occurred due to the decrease in the vegetation cover of the subbasin caused by the growth of the areas with corn cultivation. Between the analyzed period, the total soil loss (W_{yr}) values ranged from 14,935.81 to 9,340.80 Mg year^{-1} (Table 5), while between 1988 and 1998, the real soil loss (G_{yr}), obtained using the R_u coefficient, increased by 50.36 Mg year^{-1}. The G_{yr} increase in the initial years of the study was due to the conversion of pasture to coffee and portions of the forest to eucalyptus (Figure 2). Between 1998 - 2008 and 2008 - 2018, there was a reduction in G_{yr} values of 352.49 and 307.72 Mg year^{-1}, respectively. The reduction in G_{yr} possibly contributed to the improvement of water quality in the subbasin, since sediment delivery causes the depreciation of water quality and the silting up of watercourses (POSTHUMUS et al., 2015).

Table 5. Annual erosion, real soil loss, estimated sediment delivery (average soil loss), observed sediment delivery and errors of estimate for the Coroado Stream Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil.

Year	W_{yr} (Mg year^{-1})	G_{yr} (Mg year^{-1})	Estimated SD (Mg ha^{-1} year^{-1})	Observed SD (Mg ha^{-1} year^{-1})	Error (Mg ha^{-1} year^{-1} and %)
1988	14509.07	1712.07	3.06	3.53	0.47 (13.31)
1998	14935.81	1762.43	3.15	2.64	-0.51 (19.32)
2008	11948.64	1409.94	2.52	3.14	0.62 (19.75)
2018	9340.80	1102.22	1.97	3.04	1.07 (35.20)

Notes: W_{yr} = Annual erosion; G_{yr} = real soil loss; SD = sediment delivery.

Considering the entire study period, the land use and land cover changes that occurred in the subbasin decreased water erosion. Between 1998, the year with the highest G_{yr} value, and 2018, there was a reduction of 660.21 Mg year^{-1} (-37.46%) in soil losses. The implementation of coffee and the increase of reforested areas contributed to this result. According to Didoné, Minella and Evrard (2017) an effective scenario for reducing water erosion included the implementation of protected forest areas, which was corroborated with the present study. In addition, as noted, coffee cultivation associated with conservation practices has the potential to reduce soil losses compared to annual crops such as corn (LENSE et al., 2019).

The mean soil loss estimate (Estimated SD) ranged from 3.04 to 3.53 Mg ha^{-1} year^{-1} with the lowest average in 2018 (2.64 Mg ha^{-1} year^{-1}). Regarding the Observed SD, it was found a small value in 1988 (Table 5), due to the low precipitation rate in this year (Table 4). Rainfall characteristics are the main factor that affects the Observed SD. According to Mello et al. (2007) and Beskow et al. (2009), rainfall with high erosivity increases Observed SD rates.

The comparison between Observed SD and Estimated SD in 2018 provided an error of 1.07 Mg ha^{-1} year^{-1} (35.20%). For the years of 1988, 1998, and 2008, were found errors of 13.31, 19.32, and 19.75%, respectively. The use of EPM aims to estimate an average annual soil loss (Estimated SD), and on the other hand the Observed SD is obtained from individual rainfall events, for this reason the comparison of these parameters may not coincide and generate high errors (GAVRILOVIC, 1962; EFTHIMIOU; LYKOUDI; KARAVITIS, 2017), as observed in the present study mainly for the year 2018. In addition, the errors may have occurred because the sediment transport monitoring used in the Observed SD calculation did not present data collection before 2007 (due to the high costs involved), resulting in the extrapolation of results to 1988 and 1998, which are outside the collection data period (2007 to 2018). Also, the characterization of indexes from satellite images (X_a, φ), presents some uncertainties associated with land use determination (BESKOW et al., 2009).

Soil erosion modeling is, like any other model, a representation of reality rather than reality itself and is therefore prone to uncertainty. Furthermore, modeling, at least on a large scale, should not only focus on accurate prediction of absolute values but rather function as a tool for testing hypotheses of relative differences between management systems and long-term soil loss trends. Both measured and modeled soil erosion rates are relatively well related to each other. However, if they are much higher than soil formation rates, land management cannot be considered sustainable (ALEWELL et al., 2019). Therefore, despite the errors, modeling is efficient in pointing out sites with unsustainable soil management and should be used as a tool to support the adoption of mitigation practices, seeking to minimize the prejudicial impacts of the erosion process as much as possible (VANWALLEGHEM et al., 2017).

Temporal-spatial distribution of soil erosion in the Coroado Stream Subbasin

The distribution of soil erosion rates revealed the highest soil loss in the steepest areas and in the access roads (Figure 4), due to the exposed soil and heavy machinery traffic. Soil loss values for each year were divided into classes adapted from Beskow et al. (2009) and Avanzi et al. (2013) (Table 6).

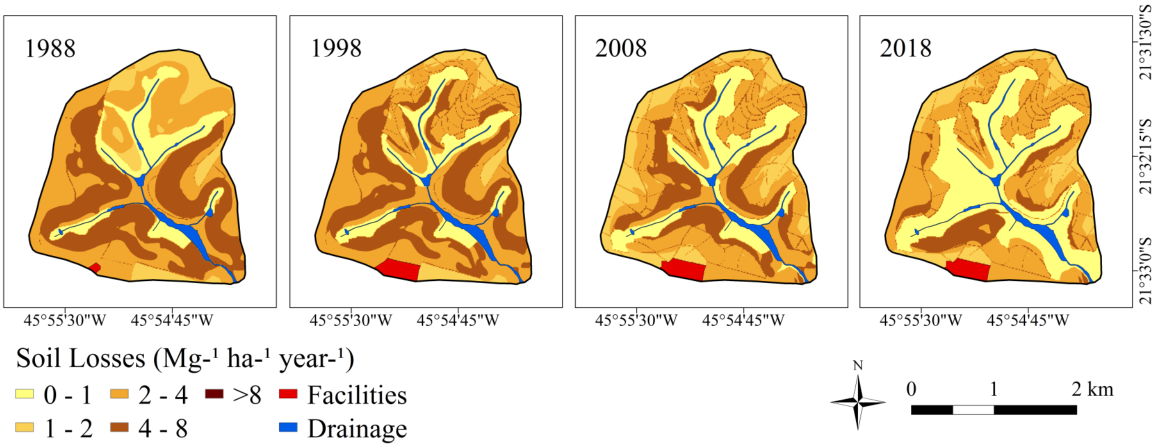

Figure 4. Soil loss classes map of the Coroado Stream Subbasin, Alfenas, south of Minas Gerais State, Brazil, in the years of 1988, 1998, 2008 e 2018.

Table 6. Qualitative classes of soil loss for the Coroado Stream Subbasin, Alfenas Municipality, south of Minas Gerais State, Brazil.

Soil loss ($Mg\ ha^{-1}\ year^{-1}$)	Area (%)				Soil loss class qualitatively
	1988	1998	2008	2018	
0 – 1	16.36	16.8	23.32	39.69	Very Slight
1 – 2	10.30	6.96	15.14	10.38	Slight
2 – 4	40.43	44.16	40.90	37.88	Slight to Moderate
4 – 8	32.91	32.06	20.56	11.96	Moderate
> 8	0.00	0.02	0.08	0.09	High

Notes: Soil loss class qualitatively adapted from Beskow et al. (2009) and Avanzi et al. (2013).

The results indicate that, over the years, there was an increase in areas classified with Very Slight and Slight erosion rates and reduction in areas with soil loss classified as Slight to Moderate and Moderate (Table 6). These results can be explained by the increase of 72.6 ha to 193.9 ha (167.0%) in forested areas from 1988 to 2018. Furthermore, coffee cultivation with the adoption of conservation practices (level planting, weed management between the lines) and the reduction of corn cultivated under conventional tillage, contributed to the positive result (Table 1). Between 1988 and 1998, there was a decrease in areas with soil loss classified as Slight and an increase of areas with Slight to Moderate erosion. This result was affected by the expansion of deforestation and the conversion of the pasture to temporary crops. However, these areas were recovered in the following years. In 1988, no site was identified by the model as presenting a High level of erosion, remaining almost the same in 1998, when less than 1% of the watershed shown this level of degradation. Nevertheless, there was an increase of access roads between 1998 and 2018, which could have caused soil exposition, and combined with the steepest relief, led to a more concerning scenario, where there was also the development of new sites with a high level of erosion.

In general, soil loss decrease over the 30 years of study, mainly due to the increase in reforestation areas, driven by environmental legislation in the Atlantic Forest biome, which requires the allocation of at least 20% of rural property to conservation areas (BRASIL, 2012). Hence, the appropriate use of the public policies could influence the environmental system as a whole, and still contribute to soil preservation, reducing the risk of soil erosion (DEVÁTÝ et al., 2019).

The correct use of land in the steepest areas also contributed to mitigate soil losses, once these areas are the most affected by the erosive process. Between 1988 and 2008, the vast majority of areas with declivity above 20% (200 ha), which were occupied with corn cultivated under conventional tillage were replaced by coffee and reforestation crops (Figure 5), which resulted in a reduction of 146.50 Mg $year^{-1}$ in soil losses. Temporary crops grown in the steepest areas can make the soil more susceptible to runoff, and therefore these areas should be a priority for the adoption of erosion mitigation measures and for planning the appropriate land use in order to reduce the erosion process (BESKOW et al., 2009; CHEN et al., 2019; LENSE et al., 2020).

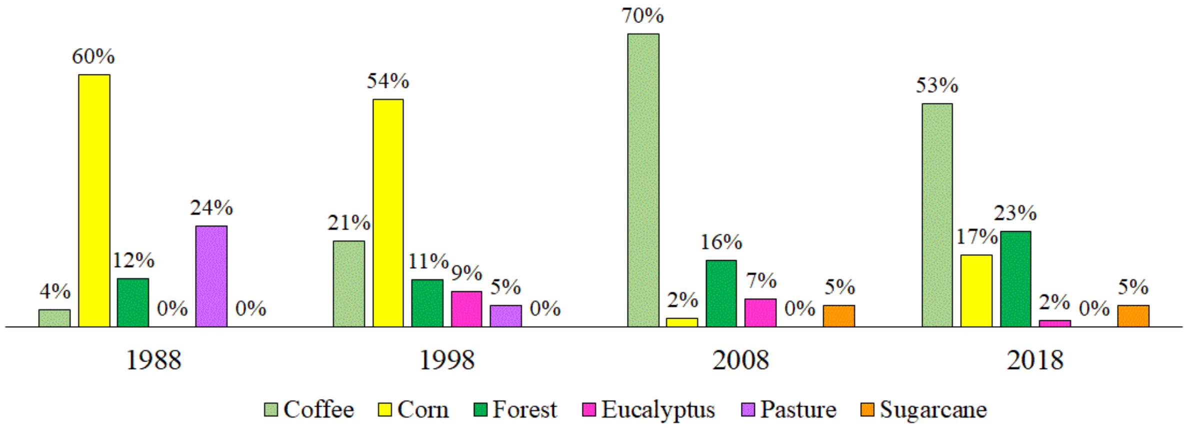

Figure 5. Percentage of land occupied by forest and agricultural crops in areas with declivities above 20% in the Coroado Stream Subbasin, Alfenas, south of Minas Gerais State, Brazil, in the years 1988, 1998, 2008 and 2018.

The study pointed out that, despite the soil loss reduction, high erosion rates still occur in the study area, mainly on access roads and corn cultivation (Figure 4 - 2018). Therefore, to ensure the long-term sustainability of the production system, soil conservation practices should be expanded, even to site with very slight erosion. The reduction of runoff and sediment loss in the access roads can be achieved by the implementation of retention basins. In areas with corn and sugarcane cultivation, the adoption of a continuous no-till system and the building of the terraces to leveling the land can mitigate the erosion problem (DIDONÉ; MINELLA; EVRARD, 2017).

Studies applying the EPM on a temporal scale in tropical regions were not detected. However, our results demonstrated what this model presented efficient results being a useful tool for understanding the temporal and spatial dynamics of the erosive process and for the sustainable land use planning. Moreover, the model is advantageous for areas with little available information, which is common in much of the Brazilian territory.

CONCLUSION

In the Coroado Stream Subbasin the conversion of pasture and corn cultivation in coffee crops with the adoption of conservationist practices and especially the expansion of reforestation contributed to the reduction of the erosive process intensity, supporting the hypothesis tested.

The reforestation in the Coroado Stream Subbasin was driven by the need to comply with the Brazilian Forest Code, thus emphasizing the importance of public policies for environmental conservation.

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The authors thank the Fundação de Amparo à Pesquisa do Estado de Minas Gerais (FAPEMIG) for the scholarship offered to the first author. The group Ipanema Agrícola S. A. for funding the research and conceding the study area.

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4 ARTIGO 2 - EFFECTS OF DEFORESTATION ON WATER EROSION RATES IN THE AMAZON REGION

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ABSTRACT: Deforestation in the Amazon Rainforest is a recurring theme on the national and international environmental agenda. However, little attention has been given to its influence on water erosion and soil degradation. This study aimed to model the effect of deforestation on the spatial and temporal variation of water erosion in a watershed of the Amazon region. Therefore, we hypothesize that the expansion of deforestation, and the consequent changes in land use and cover, contributed to increasing soil losses due to water erosion. The Xingu River watershed was selected as a study area once it is one of the most affected regions by deforestation in the Brazilian Amazon. The estimate of water erosion was performed in the years 1988, 1998, 2008, and 2018 using the Potential Erosion Method (EPM). The application of the model was carried out in a Geographic Information System environment. Between 1988 and 2018, the deforestation in the selected basin increased by 12% (52,258 km^2). In the same period, water erosion increased by 312%, corresponding to about 180 million tons of soil lost per year. The result of the study can help in planning erosion control in the Amazon region.

Key words: cover change; land use; soil conservation; soil loss

Efeito do desmatamento sobre as taxas de erosão hídrica na região amazônica

RESUMO: O desmatamento da Floresta Amazônica é tema recorrente na agenda ambiental nacional e internacional, porém pouca atenção tem sido dada a sua influência na erosão hídrica e na degradação dos solos. Este estudo objetivou modelar o efeito do desmatamento sobre a variação espacial e temporal da erosão hídrica na região Amazônica. Nossa hipótese é que a expansão do desmatamento e as consequentes mudanças no uso e cobertura da terra, contribuíram para aumentar as perdas de solo pela erosão hídrica. A Bacia Hidrográfica do Rio Xingu foi selecionada como área de estudo por ser uma das regiões mais afetadas pelo desmatamento na Amazônia Brasileira. A estimativa da erosão hídrica foi realizada nos anos de 1988, 1998, 2008 e 2018 utilizando o Método de Erosão Potencial (EPM). A aplicação do modelo foi realizada em ambiente de Sistema de Informação Geográfica. Entre 1988-2018, ocorreu um aumento de 12% (52.258 km^2) no desmatamento da floresta amazônica da bacia. No mesmo período houve um crescimento na erosão hídrica em 312% correspondente a cerca de 180 milhões de toneladas de solo perdido por ano. O resultado do estudo pode auxiliar no planejamento de controle da erosão na região amazônica.

Palavras-chave: mudanças no uso; uso da terra; conservação do solo; perda de solo

Introduction

Soils are an essential component of the terrestrial system that produces food, biomass, and raw materials, provides a habitat for flora and fauna, sequester carbon, and manages water, minerals, and biological cycles (Cerdà et al., 2018). In tropical regions, this resource is often degraded due to water erosion, leading to a decline in fertility and its ability to sustain life (Martínez-Mena et al., 2020).

In the Brazilian Amazon region, deforestation intensifies water erosion, mainly during the conversion of deforested areas to agricultural and pasture areas. Although deforestation in the Amazon rainforest is a relevant and recurring topic on the national and international environmental agenda (Tacconi et al., 2019), little is discussed about the influence of this deforestation on erosion rates and soil degradation. Thus, the assessment of the dynamics of the erosion process due to deforestation and the consequent changes in land use and occupation in the region is essential to highlight the importance of soil conservation and assist in proposing effective strategies to reduce erosion (Efthimiou et al., 2017).

Due to the dimensions of watersheds in the Amazon region and the low availability of information, modeling is an alternative to estimate water erosion and provide a diagnosis of soil losses (Batista et al., 2017). The model application is simple, of low cost, and it can be combined with Geographic Information Systems (GIS) to estimate erosion rates in both spatial and temporal scale. Moreover, it represents the possibility of a remote investigation of the erosion process (Luetzenburg et al., 2020).

In Europe, modeling of large areas often helps in the proposal and adoption of environmental and agricultural policies to decrease the negative impacts of erosion. The European Union (EU) itself has implemented soil protection and conservation in its common agricultural policy to prevent soil degradation (Panagos et al., 2015; Alewell et al., 2019). In Brazil, although modeling studies are not widely used by the government agencies, the estimate of water erosion, as well as the understanding of its temporal dynamics, can be used as a tool in planning the sustainable management of watersheds. Besides, there are no studies that estimate water erosion in large hydrographic basins in the Brazilian Amazon.

In this context, we aimed to model the effect of deforestation on the spatial and temporal variation of water erosion in a sub-basin of the Amazon region. As a hypothesis, it is believed that the expansion of deforestation and consequent changes in land use and cover, contributed significantly to increase soil losses due to water erosion in the region.

Materials and Methods

Study area description

The Xingu River watershed was selected for the study once it is one of the areas most affected by deforestation in the Amazon region (Villas-Bôas, 2012). The watershed has a drainage area of 508,348 km^2, located between the states of Pará and Mato Grosso, Brazil, at coordinates 55º 36’ 14” to 50º 21’ 14” W and 14º 54’ 28” to 01º 38’ 34” S, Datum SIRGAS 2000.

The Xingu River has an average annual flow of about 8,000 m^3 s^{-1}, which makes it the fifth-largest tributary of the Amazon River. The source of the Xingu River is located in the State of Mato Grosso, and it flows north for about 2,000 km before reaching the Amazon River, in the State of Pará (Dias et al., 2015).

The watershed covers an ecological transition area with a wide variety of vegetation, resulting from the environmental dynamics generated by the meeting of the Cerrado and Amazon biomes (Villas-Bôas, 2012). The Köppen climate classification for the region is represented in Figure 1 (Alvares et al., 2013).

The estimate of water erosion due to deforestation and land use and cover changes in the watershed was performed in the years of 1988, 1998, 2008, and 2018, because the year of 1988 the Amazon deforestation started to be monitored with satellite images, adopting a 10 year timescale after the start of monitoring.

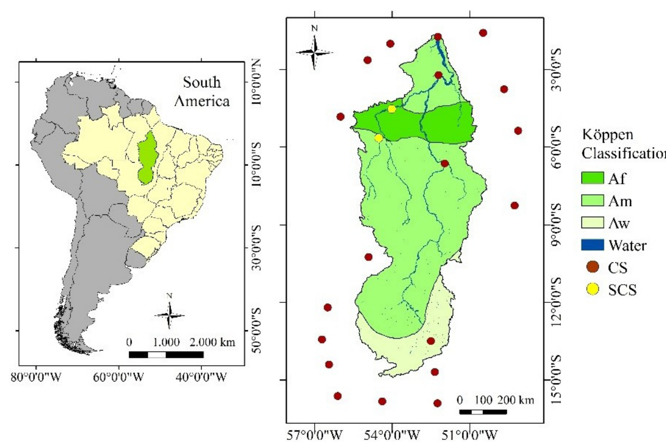

Figure 1. Location and Köppen climate classification of the Xingu River watershed, Brazil. Notes: tropical climate without dry season (Af), tropical monsoon climate (Am), tropical climate with dry winter (Aw), climatological stations (CS), sediment collection stations (SCS). Köppen’s classification adapted from Alvares et al. (2013).

Water erosion modeling

Amongst several models available for soil loss estimate, the Erosion Potential Method (EPM) was selected (Gavrilovic, 1962) because it is a model widely applied worldwide (Efthimiou et al., 2017). The parameter obtaining is not complicated, as well as its application, advantages that make it ideal for a region with low information availability. Also, the model has recently been adapted to Brazilian edaphoclimatic conditions, presenting accurate and reliable results (Sakuno et al., 2020). EPM estimates soil losses according to Equation 1.

$$W_{yr} = \left(\sqrt{\frac{t_0}{10}} + 0.1\right) \cdot H_{yr} \cdot \pi \cdot \sqrt{\left[Y \cdot X_a \cdot \left(\varphi + \sqrt{I_{sr}}\right)\right]^3} \cdot Bd \cdot F \quad (1)$$

where: W_{yr} = total loss of soil, in Mg ha^{-1} year^{-1}; t_0 = average air temperature, in °C year^{-1}; H_{yr} = annual rainfall, in mm year^{-1}; Y = soil resistance to water erosion, dimensionless; X_a = use and management coefficient, dimensionless ; ϕ = coefficient of the degree of erosive features, dimensionless; I_{sr} = average slope of the area, in %, Bd = average soil bulk density, in kg dm^{-3}; and F = area, in ha.

The climatic factors (H_{yr} and t_0) were obtained based on the network of meteorological stations of the National Institute of Meteorology, distributed inside and near the watershed (Figure 1). The climatic data were interpolated by the ordinary kriging method, with adjustment of the spherical model, using the Geostatistical Wizard tool of the ArcGIS 10.3 software (ESRI, 2015). The spatial distribution maps of H_{yr} and t_0 are illustrated in Figures 2 and 3, respectively.

Amongst the studied years, 2008 had the highest average accumulated precipitation (2077 mm year^{-1}), followed by 2018 (2015 mm year^{-1}), 1988 (2012 mm year^{-1}), and 1998 (1672 mm year^{-1}). The average air temperature in the watershed was 25.8, 26.6, 26.1, and 26.5 °C year^{-1} in 1988, 1998, 2008, and 2018, respectively.

The coefficient of land use and management (X_a) represents the soil protection against the impact of raindrops and runoff due to the vegetation cover. The values ranged from 0.05, in areas with dense vegetation, to 1.00, for areas with bare soil (Gavrilovic, 1962). This parameter is particularly important for our work, as it reflects the density of the vegetation cover and thus the deforestation rates at the studied watershed.

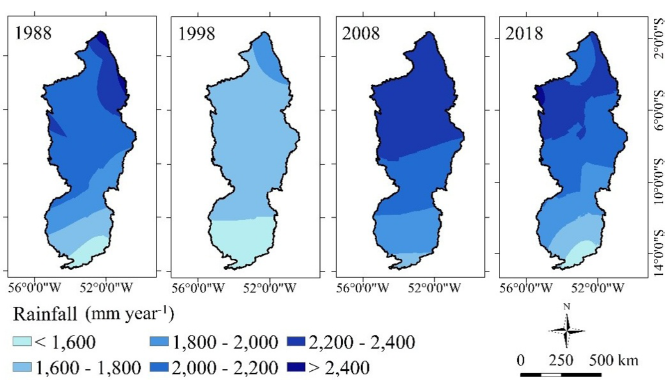

Figure 2. Annual rainfall (H_{yr}) in the Xingu River watershed, Brazil, in the years of 1988, 1998, 2008, 2018.

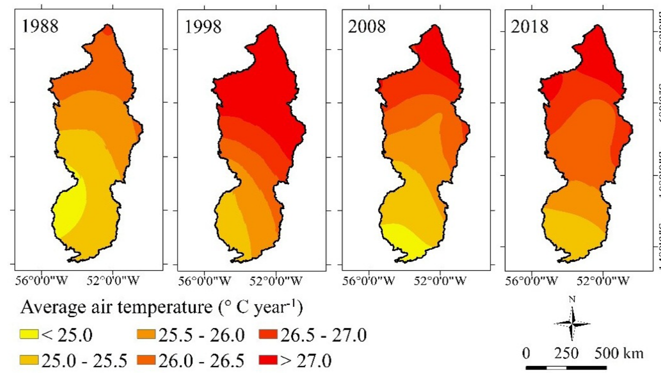

Figure 3. Average air temperature (t_0) in the Xingu River watershed, Brazil, in the years of 1988, 1998, 2008, 2018.

The parameter X_a was determined using tabulated values that were initially proposed by Gavrilovic (1962), and then adapted to Brazilian edaphoclimatic conditions by Sakuno et al. (2020). For the classification of parameter X_a (Table 1), maps of land use and occupation were used in the years of 1988, 1998, 2008, and 2018 (MapBiomas Project, 2018) (Figure 4).

Table 1 shows the percentages of the area occupied by each class of land use.

The ϕ factor indicates the erosion feature that predominates in each sit. For each type, the parameter receives a tabulated value, ranging from 0.10, for areas without any erosion features, to 1.00, for those with severe signs of erosion. Due

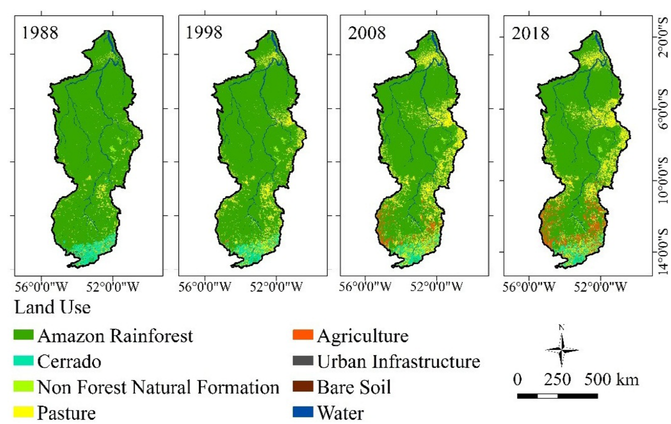

Figure 4. Land use map of the Xingu River watershed, Brazil, in the years of 1988, 1998, 2008, 2018. Adapted from the MapBiomas Project (2018).

Table 1. Classes of land use, and values adopted for the land use and management (X_a) coefficient, and the visible erosion features (ϕ) coefficient at the Xingu River watershed, Brazil.

Land use	1988	1998	2008	2018	X_a**	ϕ***
	Area (%)				(dimen.)	
Amazon Rainforest	89.77	85.25	77.40	74.97	0.05	0.1
Cerrado	4.46	3.78	3.19	3.13	0.30	0.2
Non Forest Natural Formation	1.12	1.20	1.23	1.55	0.40	0.2
Pasture	3.18	8.28	14.6	13.57	0.50	0.5
Agriculture	0.18	0.32	2.43	5.60	0.70	0.6
Bare Soil	0.18	0.06	0.04	0.02	0.90	0.7
Urban Infrastructure*	0.01	0.01	0.01	0.06	-	-
Water*	1.10	1.10	1.10	1.10	-	-

* Areas not considered in the calculation of estimated soil loss. ** Values classified according to Sakuno et al. (2020). *** Values adapted from Lense et al. (2019) and Sakuno et al. (2020). Notes: dimen.= dimensionless.

to the wide extension of the watershed, which makes it hard to identify these features in situ, the ϕ factor was classified according to the land use, using as reference values reported in the specialized literature (Table 1).

The Y parameter varies from 0.20 to 2.00 and represents the resistance of each type of soil to water erosion, on what those with higher Y values are less resistant. Therefore, it was determined according to Sakuno et al. (2020), and classified for each soil type of the watershed: Gleysols (0.5), Latosols (0.8), Nitosols (0.8), Argisols (0.9), Plinthosols (0.9), Cambisols (1.0), Litholic Neosols (1.4), and Quartzarenic Neosols (1.5). The soil classes were defined using the digital soil map of the area (IBGE & Embrapa, 2001), and they are mostly: Argisols (52.3%), Latosols (27.5%), and Litholic Neosols (8.2%) (Figure 5A).

The I_{sr} factor represents the influence of the relief on the erosive process. The slope map of the Xingu River watershed, with a spatial resolution of 30 meters (Figure 5C), was elaborated using the ArcMap 10.3 Slope tool (ESRI, 2015), based on the watershed digital elevation model (Figure 5B), obtained in the digital platform Brazil in Relief of the Empresa Brasileira de Pesquisa Agropecuária. The basin has predominantly smooth wavy relief, with an average slope of 5.3%.

In the basin, the value of Bd was determined using reference values reported in the specialized literature: Latosols - 1.18 kg dm^{-3}, Quartzarenic Neosols - 1.23 kg dm^{-3}, Cambisols - 1.30 kg dm^{-3}, Argisols - 1.31 kg dm^{-3}, Nitosols - 1.32 kg dm^{-3}, Plinthosols - 1.40 kg dm^{-3}, Litholic Neosols - 1.45 kg dm^{-3}, and Gleysols - 1.47 kg dm^{-3} (Baena & Dutra, 1982; Wadt, 2004; Medeiros et al., 2013; Oliveira et al., 2015). Bd is a parameter incorporated in the original EPM formula, to convert volume to mass (m^3 year^{-1} to Mg year^{-1}). The calculations referring to EPM and the spatial distribution of the results were made in GIS using the Raster Calculator tool from ArcGIS 10.3 (ESRI, 2015).

Validation

At Xingu River watershed, the vast majority of hydrosedimentological sample stations are inoperative or

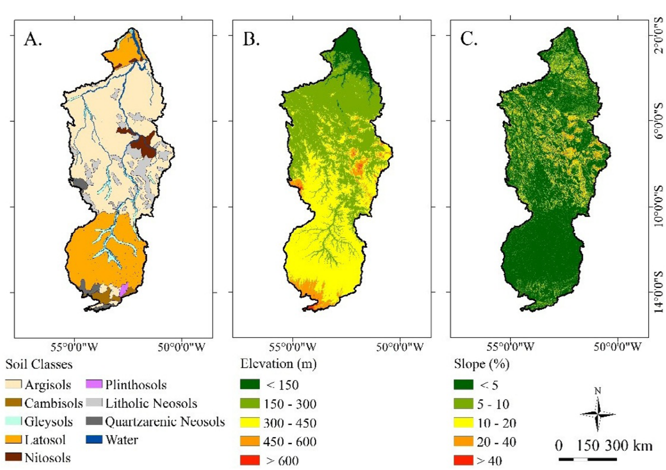

Figure 5. Digital soil map (A), digital elevation (B), and slope (C) model of the Xingu River Basin, Brazil. Digital soil map adapted from IBGE & Embrapa (2001).

with unavailable data. Therefore, the results of this work were validated according to the methodology proposed by Batista et al. (2017). For this purpose, two sediment sample stations, regulated by the National Water Agency (ANA) (Figure 1). The stations are inserted in a drainage confluence area of 122,000 km^2, as informed by ANA.

Data were collected on total sediments transported with the discharge of water and flow, monitored between the years of 1984 and 2019. With this data, a discharge curve was constructed (Figure 6). Then, the annual sediment transported (Observed SD) in the watershed, in each season, was calculated, taking into account the flow versus sediment curve and the set of daily runoff data in the years 1988, 1998, 2008, and 2018, also obtained in ANA.

To compare the Observed SD values with the results provided by the EPM, it is necessary to integrate the model with the sediment delivery rate (SDR). The SDR allows us to estimate the fraction of eroded soil of a given area that reaches the water bodies. The SDR was determined using Equation 2, as proposed by Vanoni (1975).

$$\mathrm{SDR} = 0.472 \cdot \mathrm{A}^{-0.125} \tag{2}$$

where: SDR is the sediment delivery rate, in %, and A is the catchment area in km^2.

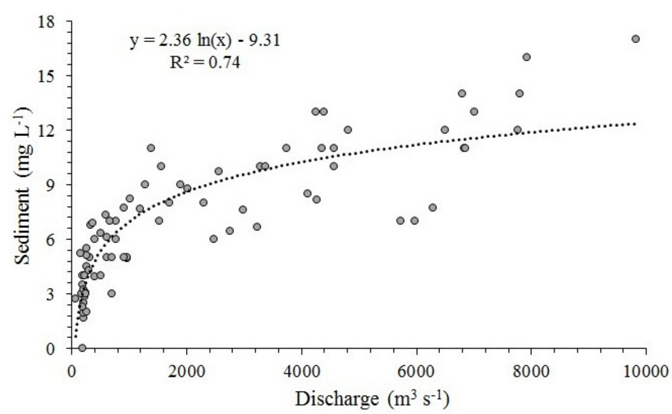

Figure 6. Water discharge curve (sediment transported *versus* water discharge), of the Xingu River watershed, Brazil.

Results and Discussion

Spatial and temporal distribution of soil losses at Xingu River watershed

Between 1988 and 2018, 12% of the Xingu River watershed Amazon Forest area was deforested, which is equivalent to 52,258 km^2. The lowest rate of deforestation was observed in the 1998-2008 period (Table 1). The deforestation reduction in the region, in the current years, is due to the combination of economic factors, such as fluctuations in commodities prices, and government command and control actions, with emphasis on the Deforestation Prevention and Control Plan in the Legal Amazon (Villas-Bôas, 2012; Reydon et al., 2019). However, despite this reduction observed, the Xingu River watershed still has high rates of deforestation: only in 2019, 168,111 ha were deforested. Deforestation in the region remains alarming and in the first four months of 2020, 35,673 ha of Amazon Forest have been deforested (SiradX, 2019, 2020a,b).

Analyzing the deforestation rates, we noticed an increase in pastures and agricultural fields in areas previously occupied by the Amazon rainforest and the Cerrado biome, respectively (Figure 4). The results agree with Dias et al. (2016), whose analysis of the expansion of agriculture in Brazil found a significant increase in agricultural land and pastures in the states of Mato Grosso and Pará between 1985-2012.

As expected, the elevation of deforestation and the land-use changes were followed by an increase in soil losses (Table 2). Between 1988-2018, the estimated erosion rates increased by 312%, corresponding to about 180 million tons of soil lost per year. Due to this rise, the highest average rate was estimated in 2018 (4.7 Mg ha^{-1} year^{-1}).

In the entire period of the study, regardless of the year, there was the predominance of low erosion rates (< 2.5 Mg ha year^{-1}) due to the high presence of the native forest (Amazon Rainforest) in the watershed (Table 3). However, there was an increase of 531% of areas presenting high erosion rates (> 10.0 Mg ha year^{-1}): 2.6% in 1998, 5.7% in 1998, 14.1% in 2008, and 16.7% in 2018. Figure 7 shows the maps with the spatial distribution of soil losses in each season studied.

In both years, average soil loss rates above 10.0 Mg ha year^{-1} occurred in the areas of exposed soil, agriculture, and pastures (Table 3). Land use beyond agricultural potential, fragile soils with inadequate management, and the absence of conservation practices can be the causes of the intensification of erosion in such land uses (Medeiros et al., 2016).

Table 2. Annual soil loss estimated during 1988, 1998, 2008 and 2018, in the Xingu River watershed, Brazil.

Year	Soil loss	
	(Mg ha^{-1} year^{-1})	(Mg year^{-1})
1988	1.13	57,892,886
1998	1.70	86,318,438
2008	4.20	214,366,256
2018	4.70	238,658,936

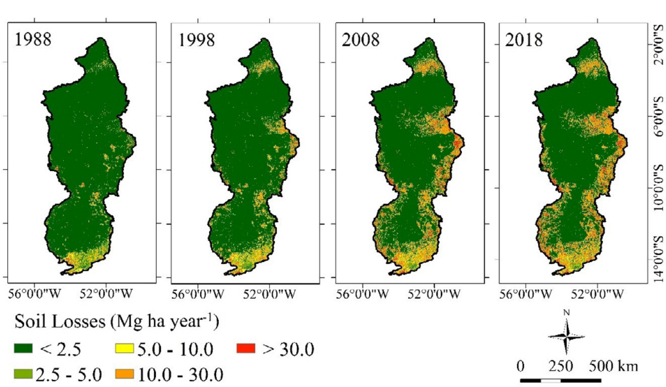

Figure 7. Spatial distribution of soil losses in the Xingu River watershed, Brazil, in the years 1988, 1998, 2008, 2018.

Table 3. Values of water erosion in each land-use class in the years of 1988, 1998, 2008, and 2018 at the Xingu River watershed, Brazil.

Land use	1988	1998	2008	2018
	(Mg ha^{-1} year^{-1})			
Amazon Rainforest	0.5	0.6	1.4	1.5
Cerrado	6.2	6.4	8.5	8.6
Non Forest Natural Formation	6.0	5.9	8.0	8.0
Pasture	8.4	9.5	14.8	15.0
Agriculture	15.4	16.3	19.3	19.3
Bare Soil	20.1	17.0	19.7	17.5

Sediment delivery ratio (SDR)

At the Xingu River watershed, the SDR estimated was 0.091, indicating that 9.1% of the eroded sediments reach water bodies, contributing to the silting and depreciation of water quality. The estimated sediment transport (Estimated SD) ranged from 0.046 to 0.108 Mg ha^{-1} year^{-1} (Table 4).

As for Observed SD, in 1998, the value was smaller than in the other seasons (0.041 Mg ha^{-1} year^{-1}) due to the lower precipitation rate this year (Figure 2). How it was calculated based on the water flow, its results were influenced by the precipitation rates at the watershed.

When compared the values of Observed SD and Estimated SD for the years of 2008 and 2018, EPM overestimated the sediment delivery by 0.022 and 0.033 Mg ha^{-1} year^{-1}, respectively, which corresponds to absolute errors of 33.9 and 44.6%. However, the errors were lower in the years of 1988 and 1998, which may be considered acceptable (Table 4) when taking into account the large extension of the watershed.

The wide variance of the vegetation cover index (X_a) during the period of the study can explain the range of the errors. Due to the high sensitivity of variation, this parameter decisively interferes in the results provided by the EPM (Dragičević et al., 2017). Also, regardless of the period, Xingu River watershed presented a high percentage of native vegetation, which minimizes the rate of sediment delivery for two reasons: i) directly, due to the low rates of water erosion observed in this type of land-use, and ii) indirectly, because the vegetation around the watercourses also acts as a physical barrier, preventing part of the eroded sediments from reaching river beds. Therefore, this increases the SDR estimation error since the model does not simulate this indirect effect.

According to Bagarello et al. (2012), when used for more practical purposes, estimates of soil loss carried out on a large spatial scale are considered accurate if the forecast

Table 4. Sediment transport estimated and observed during 1988, 1998, 2008, and 2018 in the Xingu River watershed, Brazil.

Year	Estimated SD	Observed SD	Error
	(Mg ha^{-1} year^{-1})		(%)
1988	0.046	0.058	20.7
1998	0.050	0.041	21.8
2008	0.087	0.065	33.9
2018	0.108	0.075	44.6

Notes: SD = sediment delivery.

errors do not exceed the erosion observed by a factor of two or three. Therefore, the results of this work are reliable and might assist in the decision making and management of the Xingu River watershed. Moreover, the relative lack of direct measurements of soil losses and sediment transport in the Amazon region highlights the importance of erosion forecasting models to obtain a diagnosis of the process, assisting in its understanding.

Measures to reduce soil erosion

In the current years, the main action to be adopted in the watershed to reduce soil loss rates is the reduction of deforestation, which will only be achieved with the engagement and encouragement of the Brazilian Government. The public power should aim for the correct application, and oversight, of well-directed and effective public policies (Jung & Polasky, 2018; Reydon et al., 2019). According to Stabile et al. (2020), innovations to increase the productivity of agricultural lands in the region, which reduce the need for the expansion of new agricultural frontiers, along with the payment for environmental services provided by the vegetation cover are also actions that can effectively contribute to the reduction of deforestation rates.

Due to the high erosive rates in the areas of exposed soil, agricultural land, and pastures (Table 3), it is necessary to disseminate agronomic practices aimed at preservation in such areas. Therefore, actions like maintaining the vegetation cover, the no-till, and green planting, help to increase soil protection against the rainfalls and the runoff (Martínez-Mena et al., 2020; Wen & Zhen, 2020). Moreover, land use and occupation planning should also be carried out at Xingu River watershed according to its agricultural suitability, especially in more fragile soils that have a high susceptibility to erosion (Litholic Neosols, Quartzarenic Neosols, and Cambisols).

According to Dias et al. (2015), the average total discharge on the agricultural land of the watershed is about 100% higher than those with natural vegetation, which is dangerous since the runoff, associated with erosion, can lead to the transportation of fertilizers residues, causing the contamination of water bodies. Therefore, conservationist practices to minimize the runoff can also contribute to improve the water quality and to reduce the risk of eutrophication. Also, the Belo Monte Hydroelectric Plant, located at the Xingu River watershed, can be beneficiated by the reduction of the erosive processes, since it will lead to lower rates of silting up, increasing its useful life.

It is worth mentioning that deforestation in the Xingu River watershed has a direct effect on the emission of greenhouse gases, and indirect due to the intensification of erosion since gaseous emissions from the soil occur during the erosion stages and in the deposition of sediments. Therefore, reducing deforestation, and consequently, water erosion, minimizes the greenhouse effect and global warming (Lal, 2019).

Finally, this work aimed to introduce the Soil Conservation theme into the agenda of global discussions about the environment. The deforestation of the Amazon region, along with several regional and global negative effects, also leads to soil degradation. However, few references are made until nowadays, and this is scenario can threaten the Amazon rainforest sustainability. The soil is a finite natural resource, which makes the water erosion one of the biggest threats for the environment in the human history, thus, if not properly managed, it can be depleted in a human time scale (Medeiros et al., 2016).

Conclusion

Deforestation and changes in land use between 1988-2018 in the Xingu River watershed, in the Amazon region, intensified the erosion process causing high soil losses. Between 1988-2018, there was an annual increase in erosion rates corresponding to million tons of soil lost per year.

Therefore, the result of this study is a useful tool for erosion control planning in the Amazon region, and for the identification of areas with high rates of soil loss in the Xingu River watershed, that are a priority for the adoption of measures to mitigate erosion.

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5 ARTIGO 3 - MODELING OF SOIL LOSS BY WATER EROSION IN THE TIETÊ RIVER HYDROGRAPHIC BASIN, SÃO PAULO, BRAZIL

Modelagem da perda de solo por erosão hídrica na Bacia Hidrográfica do Rio Tietê, São Paulo, Brasil

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Highlights

About 18% of the Tietê River Basin presents high soil loss.
Places with steep reliefs and low vegetation have high soil losses.
Conservation practices should be encouraged in the region to minimize soil losses.

Abstract

Since the mid-16th century, the Tietê River has been an important route for the territorial occupation and exploitation of natural resources in the interior of São Paulo and Brazil. Currently, the Tietê River is well known for environmental problems related to water pollution and contamination. However, little attention has been focused on water erosion, which is a serious issue that affects the soils and waters of the hydrographic basin. Thus, this work aimed to estimate soil loss caused by water erosion in this basin, which has an area of approximately 72,000 km², using the Revised Universal Soil Loss Equation (RUSLE). The RUSLE parameter survey and soil loss calculation were performed using geoprocessing techniques. The RUSLE estimated an average soil loss of 8.9 Mg ha^{-1} yr^{-1} and revealed that 18% of the basin's territory presents high erosion rates. These are priority zones for conservation practices to reduce water erosion and ensure long-term soil sustainability. The estimated sediment transport was 1.3 Mg ha^{-1} yr^{-1}, whereas the observed sedimentation, which was calculated based on data from the fluviometric station, was 0.8 Mg ha^{-1} yr^{-1}. Thus, the results were equivalent considering the large size of the study area and can be used to assist in managing the basin. Estimating soil losses can help in the planning of sustainable management of

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the Tietê River Hydrographic Basin and highlights the importance of minimizing water erosion, thus helping to prevent additional pollution and contamination with sediments, agrochemicals, and fertilizers.
Key words: RUSLE. Sediment delivery rate. Soil conservation. Soil sustainability.

Resumo

Desde meados do século 16, o rio Tietê tem sido uma importante rota de ocupação territorial e exploração dos recursos naturais do interior de São Paulo e do Brasil. Atualmente, o Rio Tietê é bastante conhecido pelos problemas ambientais relacionados à poluição e contaminação das águas. No entanto, pouca atenção tem sido dada à erosão hídrica, que é um problema sério que afeta os solos e as águas da bacia hidrográfica. Assim, este trabalho teve como objetivo estimar as perdas de solo causadas pela erosão hídrica nesta bacia, que tem uma área de aproximadamente 72.000 km², utilizando a Equação Universal de Perdas de Solo Revisada (RUSLE). O levantamento dos parâmetros RUSLE e o cálculo das perdas de solo foram realizados utilizando técnicas de geoprocessamento. O RUSLE estimou uma perda média de solo de 8,9 Mg ha^{-1} ano^{-1} e revelou que 18% do território da bacia apresenta altas taxas de erosão. Estas são zonas prioritárias para práticas de conservação para reduzir a erosão hídrica e garantir a sustentabilidade do solo a longo prazo. O transporte de sedimentos estimado foi de 1,3 Mg ha^{-1} ano^{-1}, enquanto a sedimentação observada, calculada com base nos dados da estação fluviométrica, foi de 0,8 Mg ha^{-1} ano^{-1}. Assim, os resultados foram equivalentes considerando a grande extensão da área de estudo e podem ser utilizados para auxiliar na gestão da bacia. A estimativa das perdas de solo pode auxiliar no planejamento do manejo sustentável da Bacia Hidrográfica do Rio Tietê e destacar a importância de minimizar a erosão hídrica, auxiliando na prevenção de poluição adicional e contaminação por sedimentos, agroquímicos e fertilizantes.
Palavras-chave: RUSLE. Taxa de entrega de sedimentos. Conservação do solo. Sustentabilidade do solo.

Introduction

The Tietê River is the largest river in the state of São Paulo at 1,136 km long, and it is used for navigation, water supply, and power generation. The river arises in the municipality of Salesópolis in São Paulo and flows in an E-W direction into the Paraná River along the border with the state of Mato Grosso do Sul (Biblioteca Virtual, 2018). At the beginning of Brazil's colonization, this river was a source of transportation, water, and food through fishing and hunting. Therefore, it was fundamental for the emergence of villages in the second half of the 16th century that would become important towns, such as São Miguel, Carapicuíba, Santana de Parnaíba, Barueri, Guarulhos, Itaquaquecetuba, Mogi das Cruzes, and the city of São Paulo, which has been stated capital since 1681 and was established based on the installation of Jesuits near the Tietê in 1554 (Nóbrega, 1978; Bruno, 1991; Zanirato, 2011).

However, despite the historical and economic importance, it plays for the state of São Paulo and for Brazil, the Tietê River has become better known for its environmental problems, which are mainly related to the pollution and contamination of its waters due to the accelerated and disorderly urbanization process in the region (Seabra, 2018). The

Tietê River has become a receptacle for waste, sewage, and industrial waste produced by urban areas built around it (Zanirato, 2011). Along 163 km of the river, between the municipalities of Mogi das Cruzes and Cabreúva, the water quality is classified as poor and unsuitable for use and aquatic life (SOS Mata Atlântica, 2020).

Although pollution is a widely discussed problem, water erosion in the Tietê River Hydrographic Basin is an issue that has received little attention. In addition, the few studies on water erosion found in the region are carried out in sub-basins of the Tietê River Hydrographic Basin, not considering the entire área (Fernandes et al., 2012; F. M. Santos et al., 2020). Water erosion is a serious environmental problem for Brazil because it leads to soil degradation through the loss of fertility, reduction in microbiota, and decreases of soil carbon stocks; thus, it has direct impacts on global warming. In addition, erosion causes siltation and pollution of water bodies through the deposition of sediments, fertilizers, and agrochemicals (Panagos et al., 2018).

Changes in land use and land cover in the Tietê River Basin over time have caused deforestation, intensive land use, and overgrazing, which have caused high erosion rates that have compromised agricultural production in vast areas. Thus, evaluating and monitoring water erosion and understanding its extension and magnitude can be implemented in conjunction with more efficient management practices aimed at soil conservation (Prasannakumar et al., 2012; Borrelli et al., 2018).

Therefore, modeling is a relatively simple method of assessing the magnitude of water erosion because it is uncomplicated to implement and interpret. Water erosion models require minimal resources compared to field experiments and can be applied using data available in scientific repositories (Ganasri & Ramesh, 2016; Alewell et al., 2019). The Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997), which is adapted from the USLE (Wischmeier & Smith, 1978), is the most commonly used model for estimating soil losses (Alewell et al., 2019). The RUSLE is a flexible and practical model for areas with low data availability, and its results can assist in the management and conservation of natural resources at the scale of hydrographic basins (Zerihun et al., 2018; Alewell et al., 2019).

Under this scenario, this work aimed to estimate the soil losses due to water erosion in the Tietê River Hydrographic Basin using the RUSLE.

Materials and Methods

Study area

The study was carried out in the Tietê River Hydrographic Basin, which has an area of approximately 72,000 km² and is located in Southeast Brazil at coordinates from 51° 34' 5" to 45° 38' 20" West and 20° 31' 55" to 24° 0' 32” South, Datum SIRGAS 2000 (Figure 1). According to the Köppen classification, a tropical (Aw) climate is predominant in the West, a humid subtropical (Cfa) climate is predominant in the central portion of the basin, and a temperate oceanic (Cfb) climate is found in the East closer to the coast (Alvares et al., 2013). The average annual rainfall varies between 1,202 and 3,085 mm (Marcuzzo, 2020).

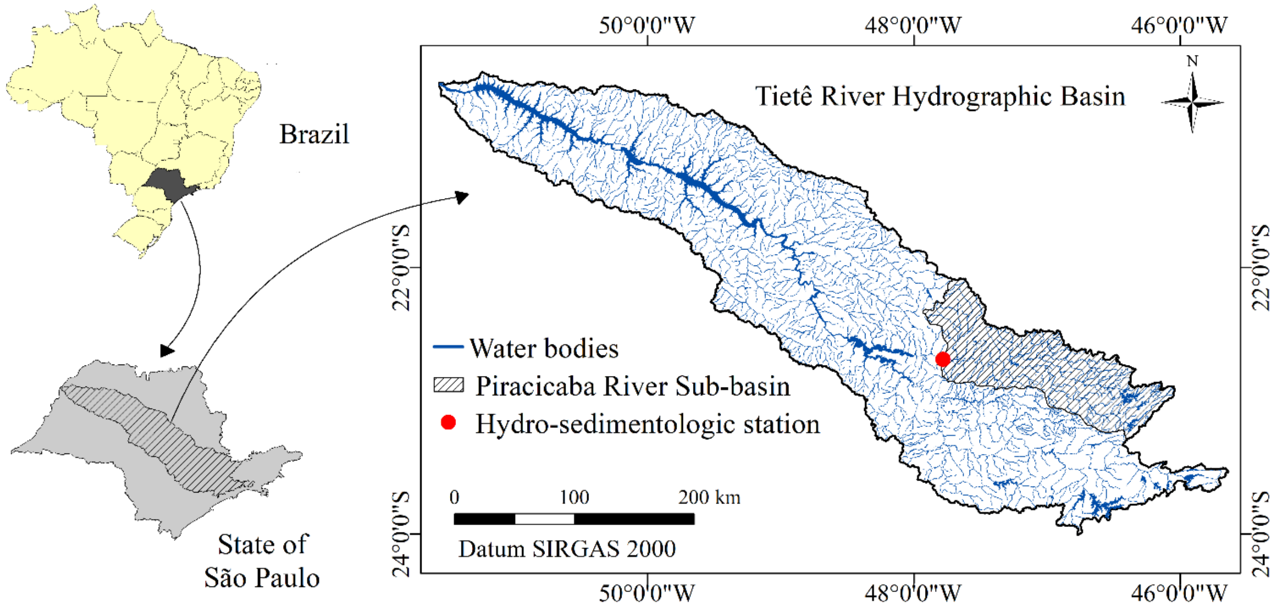

Figure 1. Location of the Tietê River Hydrographic Basin, Brazil.

Regarding the soil types, mostly Argisols (43.96%), Latosols (35.06%), and Neosols (4.78%) are found in the basin. Other less frequent soil classes are Cambisols (3.64%), Gleissoles (1.42%), Nitosoils (0.9%), Organosols (0.37%), Planosols (0.32%), Luvisols (0.07%) and Chernosols (0.02%). The map of Figure 2 was obtained from the soil map of the State of São Paulo, which was revised and enlarged (scale 1:250,000) (Rossi, 2017).

The territory is mainly occupied by pastures (37.90%), sugarcane (30.10%), forest formations (14.40%), urban areas (6.65%), planted forests (3.41%), water bodies (3.20%), temporary crops (2.58%), cerrado (1.20%), perennial crops (0.40%) and non vegetated areas (0.16%). The class "planted forests" consists mainly of eucalyptus plantations. The land use (Figure 3) from 2019 was obtained from the digital platform Mapbiomas Project (2020). The Mapbiomas Project consists of an open access platform that provides data on land use and land cover in the Brazilian territory with a high level of precision and which are prepared involving a collaborative network with experts in biomes, land uses, remote sensing and Geographic Information Systems (Mapbiomas Project, 2020).

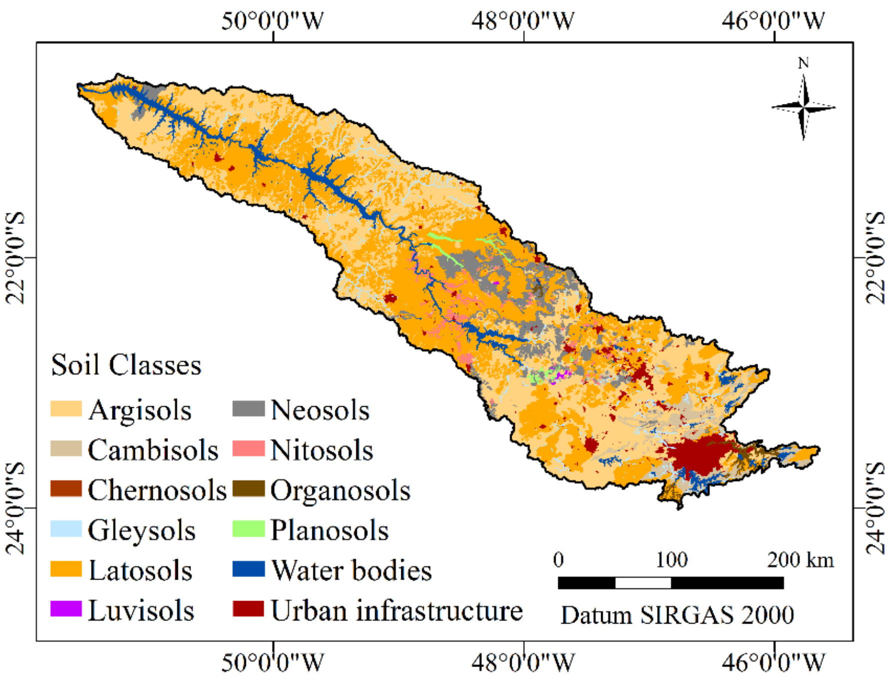

Figure 2. Soil map of the Tietê River Hydrographic Basin, Brazil, based on Rossi (2017).

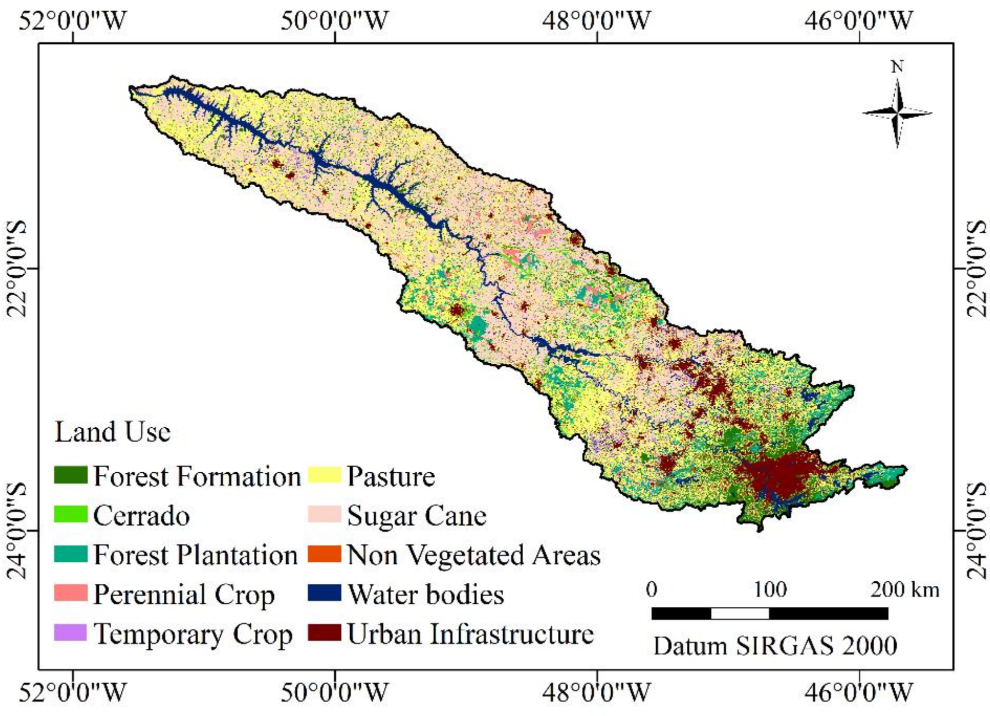

Figure 3. Land use and land cover map (2019) of the Tietê River Hydrographic Basin, Brazil, obtained from the Mapbiomas Project (2020).

The basin has altitudes between 248 and 2,030 m, with an average of 580 m (Figure 4A). The Digital Elevation Model (DEM), with a spatial resolution of 30 m, was extracted from the digital platform "Brasil em Relevo" (Miranda, 2005). From the DEM, the slope map was generated (Figure 4b) with the Slope tool on ArcMap 10.5 (Environmental Systems Research Institute [ESRI], 2016). The relief predominantly presents slopes between 3 and 8%.

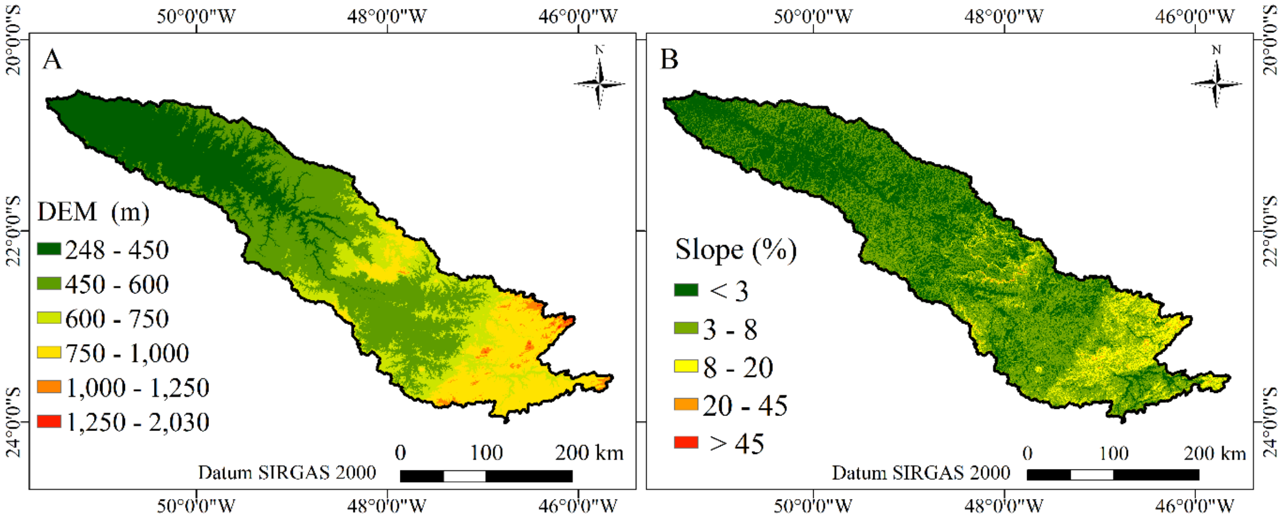

Figure 4. Digital elevation model (DEM) (A) and slope map (B) of the Tietê River Hydrographic Basin, Brazil. DEM extracted from Miranda (2005).

Revised Universal Soil Loss Equation – RUSLE

The RUSLE model estimates soil losses in a given area according to Equation 1:

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

where A is the mean annual soil loss, Mg ha^{-1} yr^{-1}; R is the rainfall erosivity factor, MJ mm ha^{-1} h^{-1} yr^{-1}; K is the soil erodibility factor, Mg ha^{-1} MJ^{-1} mm^{-1}; LS is the dimensionless topographic factor; C is the dimensionless land use and management factor; and P is the dimensionless conservation practices factor.

The R factor represents the ability of rainfall and associated runoff to cause soil erosion, and it is given by the product of the kinetic energy of rain by its maximum intensity in 30 minutes (Wischmeier & Smith, 1978; Bertol et al., 2019). In the Tietê River Basin, detailed rainfall records are lacking; thus, the R factor was generated according to the multivariate geographic model for the southeast region of Brazil, as proposed by Mello et al. (2013). With this model, it is possible to estimate the R factor using latitude, longitude, and altitude data, as applied in Equation 2 in each pixel of the DEM, which was performed using the Raster Calculator tool (ESRI, 2016).

$$R = -399433 + 420.49 \cdot A - 78296 \cdot LA - 0.01784 \cdot A^2 - 1594.04 \cdot LA^2 + 195.84 \cdot LO^2 + 17.77 \cdot LA \cdot LO - 1716.27 \cdot LA \cdot LO + 0.1851 \cdot LO^2 \cdot A + 0.00001002 \cdot LO^2 \cdot A^2 + 1.389 \cdot LA^2 \cdot LO^2 + 0.01364 \cdot LA^2 \cdot LO^3 \quad (2)$$

where R is the rainfall erosivity factor in MJ mm ha^{-1} h^{-1} yr^{-1}, A is the altitude in meters, and LA and LO are the latitude and longitude, respectively, both in negative decimal degrees.

The K factor represents the soil's natural susceptibility to erosion, which in turn is influenced by its physical, chemical, and mineralogical attributes (Wischmeier & Smith, 1978). Determining this factor in situ is an onerous process that requires at least 22 years of data collection in an experimental plot under natural rain conditions (Bertol et al., 2019). Due to these limitations, its determination was based on the soil erodibility values for the state of São Paulo reported in the specialized literature (Table 1).

Table 1
Soil erodibility factor (K)

Soil Classes *	K** Mg ha^{-1} MJ^{-1} mm^{-1}	Soil Classes *	K** Mg ha^{-1} MJ^{-1} mm^{-1}
Argisols	0.0425	Luvisols	0.0312
Cambisols	0.0508	Neosols	0.0510
Chernosols	0.0309	Nitosols	0.0237
Gleysols	0.0361	Organosols	0.0610
Latosols	0.0162	Planosols	0.0097

*Brazilian Soil Classification System (H. G. Santos et al., 2018). ** Values from Mannigel et al. (2002), A. M. Silva and Alvares (2005).

To assign the K factor values according to the existing soil classes, the soil map (Figure 2), which was clipped for the study area, was converted into a 30 m spatial resolution raster, where the values of each grid cell represented the soil erodibility.

The RUSLE includes the single topographic factor LS, which represents both the influence of slope length (L) and terrain slope (S) on soil erosion once they occur together within a terrain area because they are interdependent (Bertol et al., 2019). The LS factor was calculated using a Geographic Information System according to Mitasova et al. (1999) methodology, which is represented by Equation 3.

$$LS = (m+1) \cdot \left(\frac{FA \cdot 30}{22.13}\right)^m \cdot \left(\frac{\sin(S)}{0.0896}\right)^n \quad (3)$$

where LS is the topography, dimensionless; FA is the flux accumulation expressed as the number of cells in the DEM grid; S is the slope of the basin in degrees; m and n are empirical parameters ranging from 0.4 - 0.6 and 1.0 - 1.4, respectively, according to the predominant type of erosion in the area (laminar or furrows); and 30 is the spatial resolution of the DEM, m.

The FA parameter was calculated from the DEM (Figure 4A) using the Flow Accumulation tool in ArcMap 10.5 (ESRI, 2016). The parameters m and n were defined as 0.4 and 1.0, respectively, assuming the prevalence of laminar erosion in the basin because sugarcane, pastures, and forest formations are the main land use classes in the region.

Among the RUSLE factors, the C factor is the most important because it represents the effect of all variables of soil management, vegetation cover, and residual plant biomass on soil erosion. As the K factor, this parameter can be used in the field in long-term experiments (Bertol et al., 2019). Considering the application of erosion modeling at a basin scale, the C factor values can be extracted from the scientific literature considering the land use classes (Panagos et al., 2015; Batista et al., 2017). Thus, the values of C for the Tiete River Hydrographic Basin are represented in Table 2.

Table 2
Land use and management factor (C)

Land use	C* dimensionless	Land use	C* dimensionless
Forest formation	0.0004	Pasture	0.0500
Cerrado	0.0020	Sugarcane	0.1124
Forest plantation	0.0470	Not vegetated areas	1.0000
Perennial crop	0.1350	Water bodies**	-
Temporary crop	0.2060	Urbanization**	-

*Values of F. G. B. Silva et al. (2010); Cunha et al. (2017); Nachtigall et al. (2020). **Areas not considered in the calculation of soil loss.

The land use map, a 30 m spatial resolution raster, was employed to determine the C factor, in which each grid cell value represents its values.

The P factor expresses the ratio of soil losses under a given supportive conservation practice compared to the losses in a standard plot condition, that is, without such practice (Wischmeier & Smith, 1978). The RUSLE considers three types of supportive conservation practices: contour cultivation, strip cultivation, and terracing. These practices generate subfactors that when multiplied together, result in P values ranging from 0 to 1, with values closer to 0 indicating a higher efficiency of soil erosion reduction (Renard et al., 1997).

In the state of São Paulo, although agriculture is assumed to be largely mechanized at all stages of production, conservation practices cannot be spatially determined (Medeiros et al., 2016). Thus, based on the work of F. G. B. Silva et al. (2010) and Medeiros et al. (2016), the slope (α) was used as a key property for the adoption of conservation practices. For this purpose, the slope map was used (Figure 4b) as follows: for gradients of inclination lower than 5%, a value of P = 0.6 was assumed; and for gradients higher than 20%, a value of P = 1 was assumed. For gradients between 5 and 20%, the P factor can be defined using Equation 4.

$$\mathrm{P} = 0.69947 - 0.08911 \cdot \alpha + 0.01184 \cdot \alpha^2 - 0.000335 \cdot \alpha^3 \quad (4)$$

where P is a dimensionless conservation practice factor and α is the slope in %.

All parameters and the spatialization of the soil loss results were calculated in ArcMap 10.5 software using the Raster Calculator tool (ESRI, 2016).

Validation

Since the RUSLE does not specify the sediment delivery rate (SDR), it is necessary to separately calculate this parameter, which represents the ratio between gross erosion in a given area and the amount of sediment that reached water bodies; thus, sediment deposited on the relief is excluded. The SDR was determined using Equation 5, as proposed by Vanoni (1975).

$$SDR = 0.472 \cdot \mathrm{A}^{-0.125} \quad (5)$$

where SDR is the sediment delivery rate in % and A is the hydrographic basin area in km².

Sediment production can be measured in situ (usually at fluviometric stations) and used to validate the estimates of soil losses caused by water erosion. Therefore, after combining the RUSLE and SDR outcomes, it was possible to validate the model using data on the total sediments transported with water discharge and daily runoff, as proposed by Beskow et al. (2009) and Batista et al. (2017).

Initially, a curve relating the total sediments transported in the hydrographic basin and the water discharge was built (Figure 5). Data monitored between 2007 and 2020 at a hydrosedimentological station located in the Piracicaba River Hydrographic Sub-basin were used (Figure 1). These data are provided by the “Agência Nacional de Águas e Saneamento Básico” (ANA), which has several stations in the Tietê River Hydrographic Basin area. However, certain stations present discontinuous and sparse measurements, while others are inoperative.

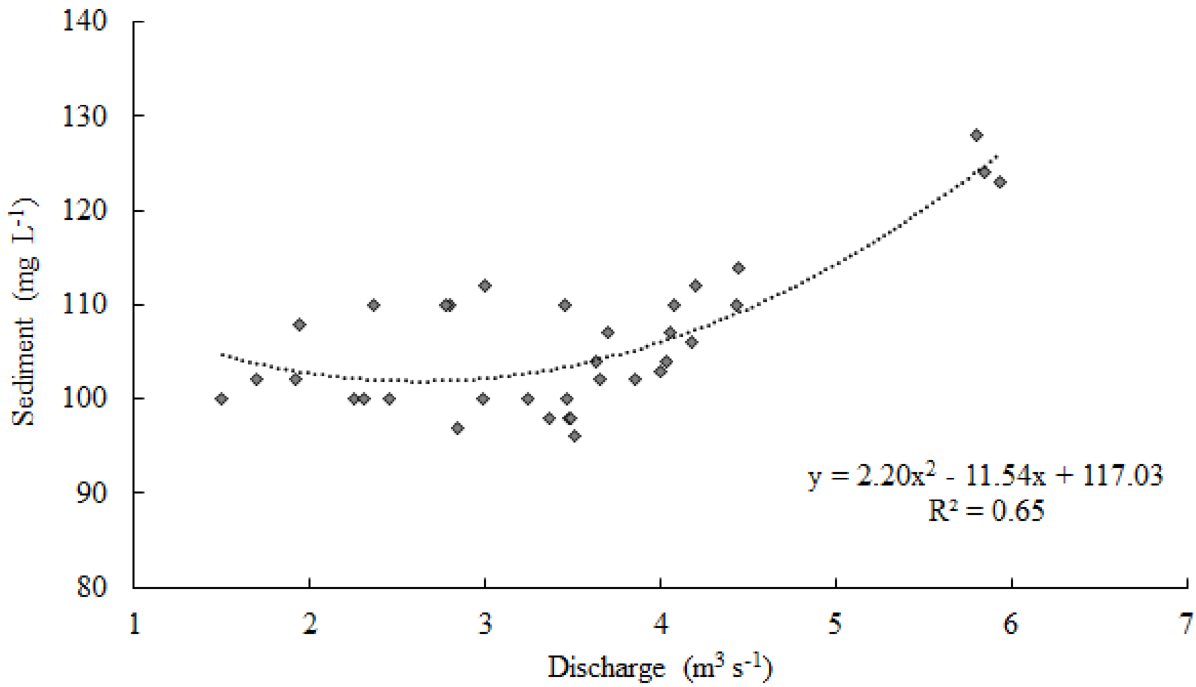

Figure 5. Water discharge curve (sediment transported × water discharge) in the Piracicaba River Hydrographic Sub-basin, Brazil.

The absence of these data hinders the sustainable planning of the entire basin; moreover, identifying sites of sediment deposition into water bodies is difficult. Therefore, environmental problems, such as siltation and water quality reductions, are caused by erosion processes. The validation was based on data from a sub-basin of the study area, as performed by Beskow et al. (2009) and Batista et al. (2017).

The observed sediment was determined based on the sediment x flow curve of the daily runoff of the Piracicaba River sub-basin (9,769 km²), which was obtained from the ANA database. Then, it was compared to the sediment estimated by the model (RUSLE/SDR).

Results and Discussion

The R factor ranged from 7,090 to 12,464 MJ mm ha^{-1} h^{-1} yr^{-1}, with an average of 7,734 MJ mm ha^{-1} h^{-1} yr^{-1} (Figure 6A). These findings corroborate Mello et al. (2013), who classified the region as being of “strong erosivity” based on the high rainfall rate.

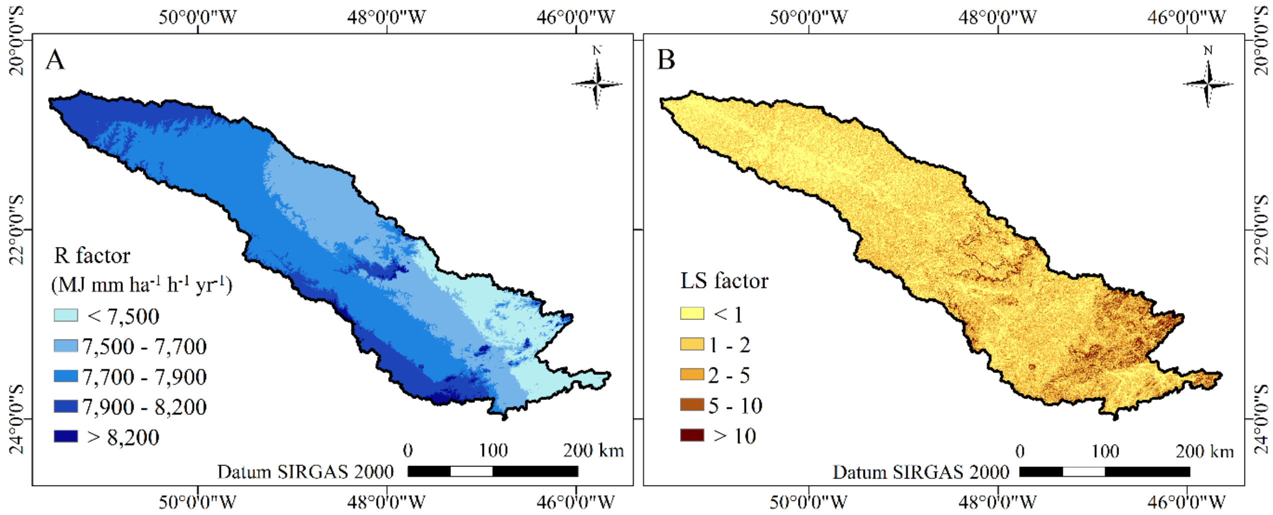

Figure 6. Spatial distribution of rainfall erosivity (A) and topographic (B) factors.

The LS factor (Figure 6B) had an average value of 1.98, indicating that much of the relief of the Tietê River Basin has low vulnerability to erosion. However, approximately 4% of the region presented LS values greater than 10, which can be considered zones of high vulnerability due to the higher speed of runoff (Beskow et al., 2009). These locations are concentrated mainly in the eastern portion of the basin, near the Serra do Mar. This result highlights the need to align agriculture practices with the implementation of conservation measures to reduce the speed of surface runoff and, consequently, water erosion in these regions.

Regarding the soil types, the Tietê River Basin presents an average K factor of 0.029 Mg ha^{-1} MJ^{-1} mm^{-1}. According to Mannigel et al. (2002), it can be classified as having medium erodibility (between 0.015 and 0.030 Mg ha^{-1} MJ^{-1} mm^{-1}). This outcome can be explained by the high occurrence of Latosols (35.06%), which are soils with low susceptibility to erosion, mainly due to their well-developed structure and good natural permeability (Bertol & Almeida, 2000).

In turn, Argisols are predominant in the area (43.96%), and this soil type has a high K score (0.0425 Mg ha^{-1} MJ^{-1} mm^{-1}). These soils have an eluvial A horizon with a coarse and often sandy texture and are prone to water erosion due to their fragile structure and poor aggregation. In addition, during longer rainfall events, runoff could reach the textural B horizon, which has a high clay content and is less permeable, thus leading to greater soil losses (Medeiros et al., 2016; H. G. Santos et al., 2018). It is noteworthy that when soils with low and high K scores are subjected to management that does not include conservation practices, high soil losses can occur (Lense et al., 2021).

The RUSLE estimated an average soil loss for the area of 8.9 Mg ha^{-1} yr^{-1}, which corresponds to a total annual loss of approximately 63 million tons of soil. The spatial distribution of these losses was qualitatively classified according to Avanzi et al. (2013) and is illustrated in Figure 7.

In 18% of the Tietê River Basin, soil losses were classified as high and very high (above 15.0 Mg ha^{-1} yr^{-1}). These areas are mainly concentrated in places with steep reliefs and low vegetation cover and therefore should be a priority for the adoption of conservation practices aimed at mitigating the harmful impacts caused by water erosion. Mainly in these areas with high soil losses, the land use planning for the Tietê River Hydrographic Basin must respect agricultural suitability and soil type vulnerability. As an example of the application of this practice, we can highlight the technical guidelines for licensing the sugar-alcohol sector in São Paulo, which are based on the State Agro-environmental Zoning of the Sugar-Alcohol Sector, which classifies the regions of São Paulo into suitability categories for sugarcane cultivation. Thus, since 2008, in areas classified as unsuitable, environmental licenses have not been granted for the installation or expansion of activities in the sugar-energy sector (São Paulo, 2008; Medeiros et al., 2016). According to Medeiros et al. (2016), respecting land use capacity is a primary factor for the sustainable use of natural resources and should be considered an essential condition for defining public policies aimed at soil conservation.

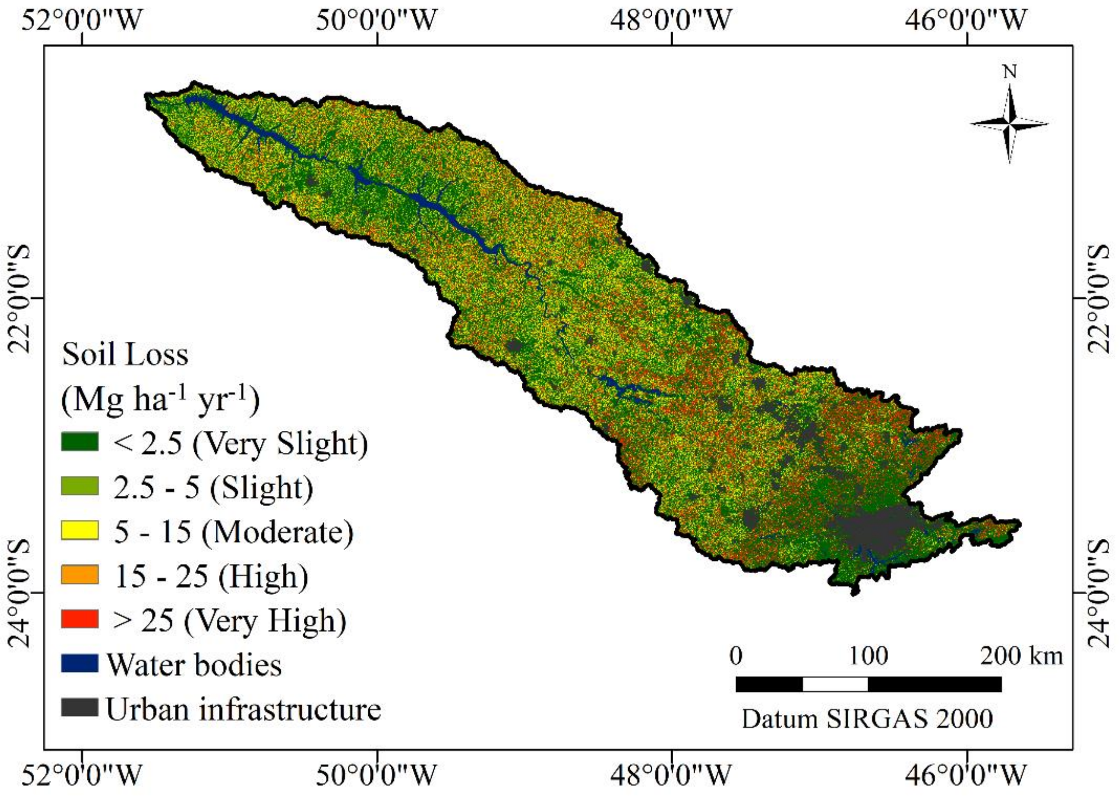

Figure 7. Spatial distribution of soil losses. Qualitative soil loss classes adapted from Avanzi et al. (2013).

The Brazilian Forest Code (Lei nº 12.651 de 25 de maio de 2012, 2012) itself offers some interesting resources to combat soil loss in areas with a high rate of water erosion. The protection of riparian vegetation and the implementation of legal reserve areas, requirements provided for in the Forest Code, when applied to rural properties, contribute to reducing the problem of water erosion. For example, it is planned to implement a legal reserve in areas with steeper relief (slopes or parts thereof, with a slope greater than 45°). In this way, the Forest Code is an important instrument for erosion control in Brazil, but there is a need for greater inspection in order to ensure that the requirements laid down by law are met. In addition, this legislation is not enough to combat erosion, farmers need technical guidance to plan and implement conservation practices and also financial assistance to encourage them to adopt these practices (Merten, 2013).

Considering the land use classes, the highest erosion rates were observed in non-vegetated areas (exposed soils) (32.4 Mg ha^{-1} yr^{-1}), followed by temporary crops (17.4 Mg ha^{-1} yr^{-1}), planted forest (13.6 Mg ha^{-1} yr^{-1}), pasture (11.5 Mg ha^{-1} yr^{-1}), sugarcane (10.5 Mg ha^{-1} yr^{-1}) and perennial crop (10.3 Mg ha^{-1} yr^{-1}) areas. In areas with forest formation and cerrado, soil losses were 2.5 and 1.6 Mg ha^{-1} yr^{-1}, respectively.

In areas with perennial and temporary crops, soil conservation techniques can be used, such as direct planting, land leveling through terrace construction and level planting, and reconstitution and maintenance of permanent preservation areas and legal reserves, to reduce soil losses and ensure the sustainability of agricultural production systems. In addition, due to the high presence of sugarcane crops in many regions, in the off-season period, when the soil is highly devoid of vegetation cover, coincides with periods of higher rainfall (Corrêa et al., 2018).

In this scenario, water erosion mitigation practices can be done from the implementation of policies to finance agricultural production and agricultural insurance conditioned to the adoption and implementation of soil conservation practices, which promote the sustainability of agricultural systems in the long term. The effectiveness of such practices can also be promoted by permanent rural extension programs, regulatory policies for the certification of sustainable agricultural production and, in addition to strengthening regulatory and inspection agencies, by rigorous inspection of the implementation of these conservationist practices.

Thus, management practices aimed at maintaining soil cover and reducing soil exposure time are essential, which is also valid for areas of planted forest (eucalyptus) and perennial and temporary crops, where soil losses are concentrated in the period of soil exposure during planting/seeding. In this context, adopting crop rotation and intercropping and reducing residue incorporation into the soil by plowing, direct planting, green manure, and spontaneous vegetation management would be good alternatives for maintaining soil cover/protection. Conservation practices aimed at increasing soil protection through vegetation directly interfere with the C factor of RUSLE, contributing to the reduction of estimates of soil losses, since among the parameters of RUSLE, the C factor is the main influencing factor human about the erosive process (Ouyang et al., 2010; Devátý et al., 2019). In addition, the C factor is one of the most sensitive to variations in space and time, since this parameter follows the growth of vegetation that varies according to the dynamics of rainfall (Nearing et al., 2005).

The sustainable use of soils allows for the maintenance of organic matter, improves the structure and water infiltration, and reduces surface runoff. Indirectly, it reduces the need to add fertilizers and agrochemicals and mitigates impacts related to contamination, eutrophication, and siltation of water bodies. Moreover, the reduction of water erosion has the potential to decrease agricultural production expenses. According to Dechen et al. (2015), the losses of P, K^+, Ca^{2+}, and Mg^{2+} in Brazilian temporary crops can represent additional expenses on the order of US$ 1.3 billion each year.

The SDR in the Piracicaba River sub-basin, which is calculated at 0.149, indicates that 14.9% of soil losses reach water bodies. This fraction of soil can cause siltation and contains nutrients and contaminants, thus leading to the depreciation of water quality and aggravating the water pollution of the Tietê River. The estimated sediment transport, which was calculated based on integrating the RUSLE/SDR, was 1.3 Mg ha^{-1} yr^{-1}, whereas the observed sedimentation, which was calculated based on data from the fluviometric station, was 0.8 Mg ha^{-1} yr^{-1}. Thus, the results were equivalent considering the large size of the study area and can be used to assist in making planning decisions and managing the basin.

It is noteworthy that water erosion modeling performed over large areas, especially when used for practical purposes, is considered accurate if the estimation errors do not exceed the observed erosion by a factor of two or three times (Bagarello et al., 2012). Furthermore, as observed in this study, the RUSLE tends to overestimate soil losses, as indicated by Amorim et al. (2010), but generates tolerable errors and shows greater precision in areas with high rates of water erosion. According to Beskow et al. (2009), modeling errors are associated with the difficulty of accurately determining the RUSLE factors, especially the LS parameter, which requires a DEM with better spatial resolution (such DEMs are often found only in small subbasins), as well as the C factor, because of the uncertainties associated with determining land use over large areas.

Regardless of the errors, the estimation of soil losses in large areas, such as the Tietê River Basin, should be interpreted as a tool to assess the dimensions of the erosive process and assist in the proposition and adoption of environmental and agricultural policies to minimize adverse impacts (Alewell et al., 2019).

Finally, modeling large-scale soil losses is a method of highlighting the importance of soil conservation and drawing attention to this topic, which is often not considered in national and international discussions. Soil is a resource considered nonrenewable within the human time scale; therefore, stimulating and ensuring its conservation is essential for the sustainability and survival of current and future generations (Medeiros et al., 2016; Bertol et al., 2019; Lense et al., 2020).

Conclusions

In the Tietê River Hydrographic Basin, 18% of the area has soil losses above 15.0 Mg ha^{-1} yr^{-1}, which were classified as areas with high rates of water erosion. These areas are mainly concentrated in places with steep reliefs and low vegetation cover. These areas should be a priority in the mitigation of water erosion, and conservation practices should be encouraged across the region to minimize soil losses as much as possible and ensure long-term soil sustainability.

Estimating soil losses can help in the planning of sustainable management of the Tietê River Hydrographic Basin and highlights the importance of minimizing water erosion, thus helping to prevent additional pollution and contamination with sediments, agrochemicals, and fertilizers.

Acknowledgments

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SOIL SCIENCE

6 ARTIGO 4 - SOIL LOSSES IN THE STATE OF RONDÔNIA, BRAZIL

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ABSTRACT: *In the state of Rondônia, deforestation, and inadequate soil use and management have intensified the water erosion process, causing degradation of agricultural land. Modeling is a tool that can assist in the adoption of targeted and effective measures for soil and water conservation in the region. In this context, the objective of the research was to model soil losses due to water erosion in the state of Rondônia using the Revised Universal Soil Loss Equation (RUSLE). The parameters related to rain erosivity, relief, erodibility, and soil cover, as well as the conservation practices of the state of Rondônia, were considered. The modeling steps were performed with the aid of the Geographic Information System. Results were validated with data of total sediments transported with water discharge. The estimated total soil loss was about 605 million tons per year, corresponding to an average loss of 22.50 Mg ha^{-1} year^{-1}. In 19% of the state, the erosion rate was higher than the soil loss tolerance(T), and these areas should be prioritized for adopting measures to mitigate the erosion process. The RUSLE underestimated the generation of sediments at 0.56 Mg ha^{-1} year^{-1}, which corresponds to an error of 18.60%. Results obtained can assist in the development of different soil use and management scenarios and provide options for policymakers to encourage soil conservation in the state of Rondônia.*
Key words: *water erosion, soil conservation, RUSLE, amazon region.*

Perdas de solo no Estado de Rondônia, Brasil

RESUMO: *No Estado de Rondônia, o desmatamento, o uso e o manejo inadequado dos solos têm intensificado o processo de erosão hídrica, gerando a degradação de terras agrícolas. Nesse cenário, a modelagem é uma ferramenta que pode auxiliar na adoção de medidas direcionadas e eficazes de conservação do solo e da água na região. Assim, o objetivo do trabalho foi modelar as perdas de solo por erosão hídrica no Estado de Rondônia utilizando a Equação Universal de Perda de Solo Revisada (RUSLE). Foram considerados os parâmetros referentes a erosividade da chuva, relevo, erodibilidade e cobertura do solo e as práticas conservacionistas do Estado de Rondônia. As etapas da modelagem foram realizadas com auxílio de Sistema de Informações Geográficas. Os resultados foram validados com dados de coleta de sedimentos totais transportados com a descarga d'água. A perda de solo total estimada foi cerca de 605 milhões de toneladas ao ano, correspondente a uma perda média de 22,50 Mg ha^{-1} ano^{-1}. Em 19% do Estado a taxa erosiva foi superior aos limites de tolerância de perda de solo (TPS), sendo que essas áreas devem ser priorizadas para adoção de medidas de mitigação do processo erosivo. A RUSLE subestimou a geração de sedimentos em 0,56 Mg ha^{-1} ano^{-1}, o que corresponde a um erro de 18,60%. Os resultados obtidos podem contribuir para elaborar distintos cenários de manejo e uso do solo e fornecer alternativas aos formuladores de políticas agrícolas e ambientais, com o intuito de incentivo a conservação do solo no Estado de Rondônia.*
Palavras-chave: *erosão hídrica, conservação do solo, RUSLE, região amazônica.*

INTRODUCTION

The state of Rondônia was the scene of several changes in land use and occupation, marked by developmental public policies and territorial occupation based on the removal of native vegetation (PIONTEKOWSKI et al., 2014). Extensive areas of Amazonian forest have been converted to crops and pastures with inadequate agricultural practices. Currently, the state of Rondôniais still one of the most affected by deforestation in the Brazilian Amazon (INPE, 2020). According to SCHLINDEWEIN et al. (2012), deforestation, combined with inadequate soil management and the high rate of precipitation in the northern region of Brazil, intensified the water erosion process, generating a loss of nutrients and organic matter, and the degradation of Rondônia agricultural lands.

The modeling of water erosion is a tool that can assist in the adoption of appropriate and

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efficient measures for soil and water conservation in Rondônia. Erosion models increase our understanding of environmental processes and facilitate land use and occupation planning, as well as decision-making in watershed management (PANAGOS & KATSOYIANNIS, 2019). Also, this type of approach positively influences the proposition and adoption of agricultural and environmental policies to control water erosion (ALEWELL et al., 2019, MEDEIROS et al., 2016).

The modeling is based on mathematical equations that express the relationships between natural factors (rain, soil cover, soil properties, and topography) and the erosion process. Among the modeling techniques, the Revised Universal Soil Loss Equation - RUSLE (RENARD et al., 1997) deserves to be highlighted because it is a model widely used in Brazil and worldwide. The RUSLE is a flexible model, applicable in different regions, with different edaphoclimatic conditions, and the determination of its parameters is easy. Moreover, extensive scientific literature compare and assess the efficiency of RUSLE results (ALEWELL et al., 2019).

The RUSLE application can be easily integrated into the Geographic Information System, allowing estimation of soil losses on a large scale and the spatialization of the results (GANASRI & RAMESH, 2016; BARROS et al., 2018). Therefore, given the above, the objective of the research was to model soil losses due to water erosion in the state of Rondônia, using RUSLE.

MATERIALS AND METHODS

Study area description

The State of Rondônia covers an area of 237,574 km^2, located in the northern region of Brazil, between the coordinates 66°36'49" to 60°43'17" W, and 13°41'57" to 7° 58'33" S, Datum SIRGAS 2000 (Figure 1). The climate of the region, according to the Köppen classification, is the Aw type (rainy tropical), with average annual temperatures around 25.5 °C and an annual rainfall regime over 2,000 mm (ALVARES et al., 2013; PIONTEKOWSKI et al., 2014).

The state is mainly occupied by the Amazon Forest (62.55%), pastures (32.92%), other natural formations (2.45%), agriculture (1.02%), water bodies (0.90%), urbanization (0.15%) and other non-vegetated areas (0.01%). The land use and occupation map (Figure 1) was adapted from the Map Biomas Project (2018).

The soil classes of the region are mostly Latosols (38.5%), Argisols (30.7%), and Neosols (14.4%), and other soil classes are illustrated in Figure 2A, which was prepared using the digital soil map of Brazil on scale 1:5,000,000 (IBGE; EMBRAPA, 2001). Although, the region presents an average altitude of 206 m, the maximum exceeds 1,126 m, at the Pico do Tracuá, the mountain range of Pacaás Novos (Figure 2B). The digital elevation model (DEM), with a spatial resolution of 30 meters, was extracted from the digital platform named Brazil in Relief (MIRANDA, 2005),

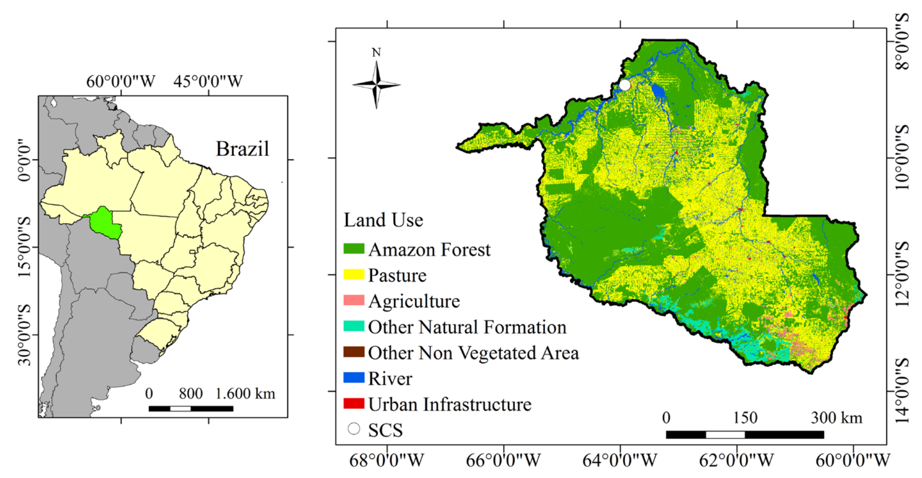

Figure 1 - Location and Land Use Map of the State of Rondônia, Brazil. Notes: sediment collection station (SCS). Adapted from the MapBiomas Project (2018).

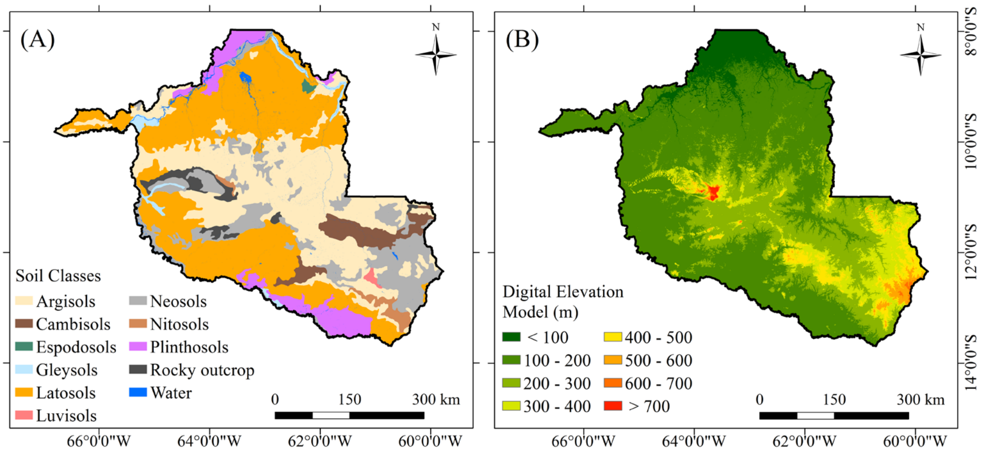

Figure 2 - Digital soil map (A) and digital elevation model (B) of the State of Rondônia, Brazil. The digital soil map was adapted according to IBGE-Embrapa (2001). Total area of each soil class: Argisols = 72,994 km^2; Cambisols = 9,511 km^2; Espodosols = 476 km^2; Gleysols = 3,804 km^2; Latosols = 91,540 km^2; Luvisols = 476 km^2; Neosols = 34,238 km^2; Nitosols = 3,091 km^2 and Plinthosols = 14,741 km^2.

regulated by the "Empresa Brasileira de Pesquisa Agropecuária - Embrapa."

Revised Universal Soil Loss Equation - RUSLE

The RUSLE model is represented according to Equation 1.

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

Where: A is the average annual soil loss, in Mg ha^{-1} $year^{-1}$; R is the rainfall erosivity factor, in MJ mm ha^{-1} h^{-1} $year^{-1}$; K is the soil erodibility factor, in Mg h MJ^{-1} mm^{-1}; LS is the topographic factor, dimensionless; C is the cover and management factor, dimensionless; and P is the support practices, dimensionless.

The R factor reflects the effect of the intensity of rainfall on soil erosion, that is, its rainfall erosivity. Due to the lack of detailed data on the duration and intensity of rainfall in Rondônia, the R factor was determined according to MELLO et al. (2013) (Equation 2). The calculation was performed from each DEM cell using the Raster Calculator tool from the ArcMap 10.3 software (ESRI, 2015).

$$\begin{gathered} R = 69{,}908 + 2{,}713.076 \cdot LA + 1{,}940.569 \cdot LO + \\ 0.0008671 \cdot A^2 - 141.233 \cdot LA^2 + 16.5387 \cdot LO^2 + \\ 46.014 \cdot LA \cdot LO + 0.0004417 \cdot A \cdot LO^2 - \\ 3.39 \cdot 10^{-7} \cdot A^2 \cdot LO^2 + 0.1905 \cdot LO^2 \cdot LA^2 + 0.00262 \cdot LA^2 \cdot LO^3 \end{gathered} \quad (2)$$

Where: R is the rainfall erosivity factor, in MJ mm ha^{-1} h^{-1} $year^{-1}$; A is the altitude, in meters; LA is latitude, and LO is the longitude, both in negative decimal degree.

The K factor shows each soil class susceptibility to the erosion process, and the higher its value, the greater is the risk of erosion occurrence. The K factor is determined according to the soil properties, which are obtained from the sampling points or soil loss plots. However, the large size of the state of Rondônia makes impossible to detail the soil properties for calculating the K factor by indirect methods. Thus, the parameter it was adopted from values reported in the literature, considering the entire Brazilian territory (Table 1), since no studies indicate the K factor value specifically for the Rondônia soils.

The LS factor represents the influence of the relief on soil losses. This parameter was calculated according to the methodology of Moore and Burch (1986), which is based on the DEM (Equation 3).

$$LS = \left(\frac{FA \cdot 30}{22.13}\right)^{0.4} \cdot \left(\frac{\sin(S)}{0.0896}\right)^{1.3} \quad (3)$$

Where: LS is the topographic factor, dimensionless; FA is the flow accumulation expressed as the number of cells in the DEM grid; S is the watershed declivity, in degree; and 30 is the spatial resolution of the DEM, in meters.

The S parameter was calculated using the ArcMap 10.3 slope tool (ESRI, 2015). In the region, the average slope is 4.5%, indicating a predominance of smooth wavy relief (3%-8%).

The C factor ranges from zero to one representing the effects of vegetation cover on water erosion rates. In exposed soil areas, C factor is one, and the higher the levels of vegetation cover on the

Table 1 - Soil erodibility factor (K).

Soil Classes*	K**	Soil classes*	K**
	Mg h MJ^{-1} mm^{-1}		Mg h MJ^{-1} mm^{-1}
Argisols	0.0425	Luvisols	0.0312
Cambisols	0.0508	Neosols	0.0510
Espodosols	0.0590	Nitosols	0.0237
Gleysols	0.0010	Plinthosols	0.0335
Latosols	0.0162		

*Brazilian Soil Classification System (SANTOS et al., 2018). ** Values adapted from MANNIGEL et al. (2002); SILVA & ALVARES (2005); GOMES et al. (2017).

soil, the lower the value of C. This parameter was determined based on the Normalized Difference Vegetation Index (NDVI) according to the methodology proposed by DURIGON et al. (2014) (Equation 4).

$$C = \frac{-NDVI+1}{2} \quad (4)$$

NDVI ranges from −1 to +1 and it is an indicator of vegetation vigor, with higher values attributed to areas of higher plant density. This index is calculated according to TUCKER (1979) (Equation 5):

$$NDVI = \frac{NIR-RED}{NIR+RED} \quad (5)$$

Where: NIR and RED are the spectral bands of the near-infrared (851 - 879 nm), and red (636 - 673 nm), respectively.

The NDVI was calculated using images from the Landsat-8 Operational Land Imager (OLI) satellite obtained in the image catalog of the "Instituto Nacional de Pesquisas Espaciais" (INPE). Due to the large extension of the study area, 16 images, dated from July to October 2019, with orbits and points, including the entire state, were selected. Image processing (mosaic composition and image treatments), as well as the NDVI and the C factor calculations were performed in the ArcMap 10.3 (ESRI, 2015).

The P factor, conversely varies according to the presence or absence of conservationist management practices of the soil. Once again, the large dimension of the state makes it difficult to determine this parameter *in situ*. Thus, for each land-use class, values available in the specialized literature were used.

In the other non-vegetated areas, the assigned P factor value was one, while in agriculture and pasture areas, 0.5, and for the Amazon Forest and other natural vegetation formations, 0.01 (BERTONI & LOMBARDI NETO, 2014). All parameters were converted into the raster data format and multiplied among themselves to execute the RUSLE equation, using the Raster Calculator tool of the ArcMap 10.3 (ESRI, 2015).

The estimated soil losses were compared with the soil loss tolerance (T) limits. T is a parameter reflecting the maximum rate of water erosion that will still allow a level of sustainable crop productivity (WISCHMEIER; SMITH, 1978). The limits adopted were obtained based on the values presented in the literature for Brazilian soils and are represented in table 2.

The RUSLE estimates the total water erosion, including both the eroded soil that stays retained in the relief depressions, as well as those that reach the water bodies in the defluvium area. The estimate of this fraction of soil that reaches the water bodies is possible from the sediment delivery rate coefficient (SDR). Therefore, the SDR was calculated using Equation 6 (VANONI, 1975).

$$SDR = 0.472 \cdot A^{-0.125} \quad (6)$$

Where: SDR is the delivery rate of sediments, in %; and A is the watershed area, in km^2.

Validation

The RUSLE results were validated according to the methodology of BESKOW et al. (2009). For this purpose, data from a hydrosedimentological station (Figure 1) regulated by the "Agência Nacional de Águas" (ANA), located in a drainage confluence area of 122,000 km^2, were used.

The monitoring conducted by ANA did not show enough frequency to monitor the entire hydrological year (only four or five collections per year due to high costs). Thus, to increase the number of samples and the accuracy of the validation, a regression test was made between the data of total

Table 2 - Soil Loss Tolerance (T).

Soil Classes*	T**	Soil classes*	T**
	Mg ha^{-1} year^{-1}		Mg ha^{-1}year^{-1}
Luvisols	3.25	Nitosols	11.10
Espodosols	7.79	Latosols	12.73
Argisols	8.61	Cambisols	13.65
Neosols	10.48	Gleysols	14.14
Plinthosols	11.00		

Notes: Soil loss Tolerance (T) *Brazilian Soil Classification System (SANTOS et al., 2018). ** Values adapted from MANNIGEL et al. (2002), MARTINS et al. (2010) and NUNES et al. (2012).

sediments transported with the discharge of water and flow, monitored between the years 1984 and 2019 (Figure 3). Then, the annual sediment transported in 2019 was calculated considering the linear regression ($y = 0.018x + 11.619$) and the daily runoff data set for the area.

RESULTS AND DISCUSSION

The rainfall erosivity (R factor) of the state of Rondônia ranged between 8,962 to 12,409 MJ mm ha^{-1} h^{-1} year^{-1} (Figure 4A), which is in agreement with was observed by MELLO et al. (2013). These authors classified the erosivity on the State as compelling, due to the high rainfall rate in the Amazon region. The LS factor had an average of 1.3 (Figure 4B), indicating that 97% of the state has relief with low vulnerability to erosion. Conversely, in 3% of the region area, the LS factor was greater than 10, and these areas are classified as highly vulnerable to water erosion (BESKOW et al., 2009).

A large part of these steep relief areas is part of the Pacaás Novos National Park (PNPN), a Conservation Unit (UC) of full protection, created by Decree nº 84019 of 1979. The Park has an area of 708,669.90 ha, and its creation protected the ecosystems by controlling the activities developed in these areas, such as scientific research, educational activities, and ecological tourism (BRASIL, 2000).

Thus, as these are areas destined to the integral conservation of natural resources, they have natural protection against the acceleration of the erosive process caused by human activities. However, in the state of Rondônia, there are still steep areas that are not part of any UC, although, according to the Forest Code, Law 12,651 of 2012 (BRASIL, 2012), any steep slope higher than 45º constitutes a Permanent Preservation Area.

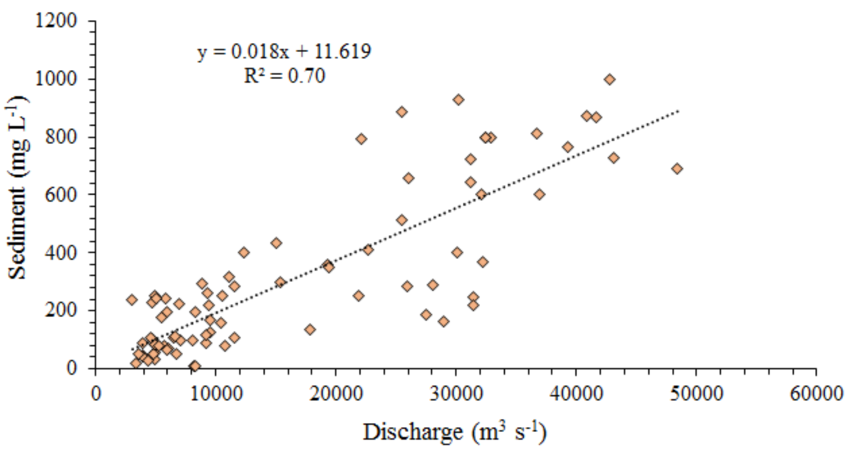

Figure 3 - Regression between the sediment transported and water discharge.

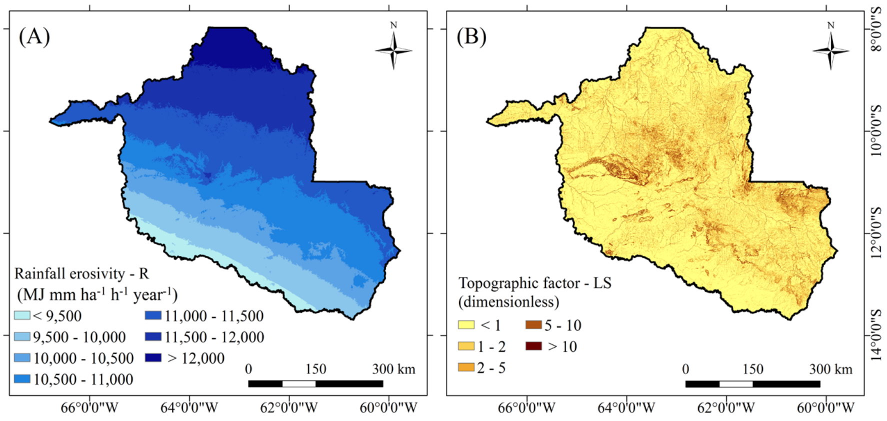

Figure 4 - Spatial distribution of Rainfall erosivity - R (3A), and Topographic factor - LS (3B).

Therefore, in these locations, practices aiming soil conservation must be respected and encouraged, since they are not areas controlled as the UC, but are vulnerable to the energy gain of the runoff due to the topography, intensifying the erosion process (STEINMETZ et al., 2018).

The NDVI of the state of Rondônia shows higher values in the forest areas, showing high plant density (Figure 5A). Low NDVI values were observed in the areas of exposed soil and pastures, indicating the degradation of these sites. According to DIAS-FILHO (2014), most pastures of the region are degraded, which was also confirmed by the vegetation index.

As for the C factor, it is normalized between 0 and 1 and inversely proportional to NDVI, so the lowest values of the parameter were observed in the Amazon Forest areas (Figure 5B), indicating good soil protection by the vegetation cover.

It is worth mentioning that most studies that apply RUSLE to Brazilian soils use the methodology for determining the C factor based on values reported in the literature (BESKOW et al., 2009; BATISTA et al., 2017). However, the adoption of a static value, especially at large scales, cannot represent the heterogeneity of vegetation density. The calculation of C factor using NDVI, allows estimating

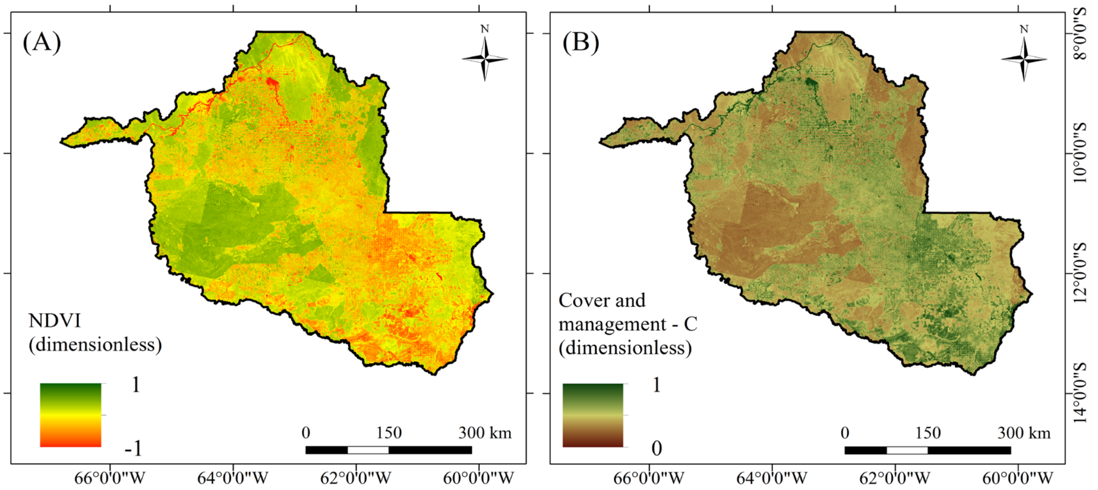

Figure 5 - Normalized Difference Vegetation Index – NDVI (5A), and Cover Management – C factor (5B).

the vegetation cover from each pixel of a satellite image, and thus, it is a more accurate representation of the vegetation cover compared to values present in the literature, which were often calculated from researches developed in different regions (LENSE et al., 2020). Also, the methodology of DURIGON et al. (2014), was developed for Brazilian tropical conditions, showing accurate results to determine the spatial and temporal dynamics of C factor in large areas (ALMAGRO et al., 2019; LENSE et al., 2020). Thus, the C factor was effective in representing variations in vegetation cover in Rondônia.

The total soil loss estimated by RUSLE was about 605 million tons per year, corresponding to an average loss of 22.50 Mg ha^{-1} year^{-1}. The spatialization of soil losses, obtained by the RUSLE, is represented in Figure 6. Comparing the results of soil loss with T, it was observed that in 19% of the State of Rondônia the erosive rate was higher than the tolerable limits, thus generating intense soil degradation. These areas are located mainly in places with high LS values and low rates of vegetation cover.

In the state of Rondônia, high soil loss was observed in pasture areas (62.85 Mg ha^{-1} year^{-1}), which occurred mainly due to the low rate of vegetation cover in this class of use, which was verified through NDVI (Figure 5A), and incorporated into RUSLE through C factor (Figure 5B). The soil losses were higher in non-vegetated areas (77.30 Mg ha^{-1} year^{-1}) and agriculture (32.70 Mg ha^{-1} year^{-1}), and minors in forests (2.24 Mg ha^{-1} year^{-1}) and other natural formations (1.50 Mg ha^{-1} year^{-1}). As for soil classes, water erosion was greater for those with higher K factor values (Table 1), especially when combined with land-use types with lower vegetation density (Table 3).

Together Argisols and Neosols make up most of the territory of the State of Rondônia (45.1%). These soils have low resistance to water erosion and when they were occupied by classes of use with little or no vegetation cover, they reached soil losses above their T limits (Table 2), reaching critical levels of water erosion (Table 3). Even Latosols, which are soils with higher resistance to the erosion process and with a low erodibility, presented high losses (> 12.73 Mg ha^{-1} year^{-1}) in areas with pasture and agriculture (Table 3). In the state of Rondônia there is compelling erosivity, so all the soils in the area are subject to high erosion rates; and therefore, management practices, changes in land use and variations in vegetation cover play an important role in reducing erosion rates, especially in soils most vulnerable to water erosion.

Results point to the need for planning, implementation, and dissemination of more effective

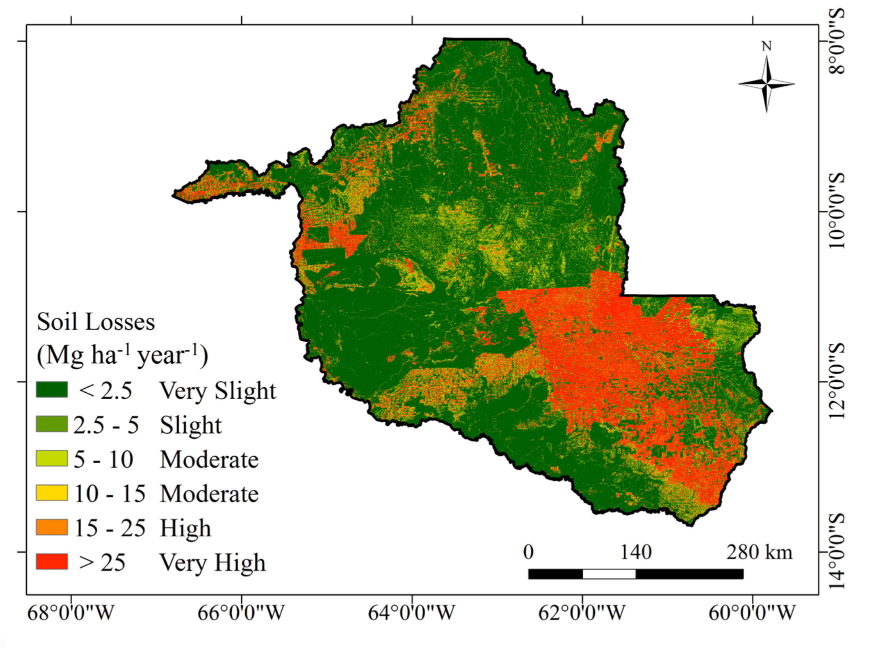

Figure 6 - Spatial distribution of soil losses in the State of Rondônia, Brazil.

Table 3 - Estimation of soil loss by soil class and respective land-use in the state of Rondônia, Brazil.

Soil Classes	Amazon forest	Pastures	Agriculture	Other natural formations	Other non-vegetated areas
	-----Soil losses (Mg ha^{-1} year^{-1})-----				
Argisols	2.70	79.52	48.75	8.12	181.85
Cambisols	4.30	79.71	50.49	2.90	125.40
Espodosols	1.60	40.25	39.33	3.40	-
Gleysols	0.04	1.90	8.20	0.09	-
Latosols	0.65	24.50	16.75	0.81	69.38
Luvisols	4.96	68.39	37.86	22.55	-
Neosols	4.10	109.60	62.14	3.92	70.68
Nitosols	3.67	70.68	27.70	7.15	-
Plinthosols	0.80	39.62	25.22	0.40	42.62

soil management techniques and conservation practices for agricultural and pasture areas, as well as the reduction of exposed soil areas, to reduce areas with an erosion rate higher than the T limits. Also, participation and encouragement by State and Municipal Governments, in partnership with the Federal Government, is essential to popularize conservationist practices and to elaborate a broad erosion mitigation plan. According to MONTANARELLA (2015) and ALEWELL et al. (2019), high rates of soil losses occur, not because of a lack of knowledge on how to protect soils, but because of a lack of public policies for their conservation.

It is worth mentioning that, to achieve the reduction of soil losses, it is also necessary to reduce deforestation since the State of Rondônia has high rates of this process (INPE, 2020), and the conversion of native vegetation to agricultural land or pasture can intensify water erosion.

Also, in the long term, adequate land-use should be sought according to its agricultural potential and suitability, due to the high presence of areas with soils vulnerable to water erosion occupied by degraded pastures (47.80; 43.40 and 26.60% of Argisols, Cambisols, and Neosols, respectively). According to MEDEIROS et al. (2016), public policies must be developed and implemented to the potential and capacity of land-use to be considered primary factors to determine the sustainable agricultural use of natural resources.

The SDR obtained using Equation 6 was 0.109, indicating that approximately 11% of all eroded soil in the region reaches watercourses causing silting and depreciation of water quality. Thus, the estimated sediment transport was 2.45 Mg ha^{-1} year^{-1}. Based on results of total solids and an average flow, of 22,37 m^3 s^{-1}, we calculated the observed sediment as 3.01 Mg ha^{-1} year^{-1}.

Comparing the values, the RUSLE underestimated the generation of sediments in 0.56 Mg ha^{-1} year^{-1}, which corresponds to an error of 18.60%. According to PANDEY et al. (2007), errors under 20% can be considered acceptable. Therefore, the results obtained are reliable and can assist in the planning of water erosion mitigation measures in Rondônia.

Regardless of the errors, large-scale modeling should be interpreted as a tool to assess the magnitude of the erosion process, as well as trends over time, system responses to determining factors, soil use and management practices. Moreover, soil erosion modeling is crucially necessary for the planning of public policies mitigated this process (ALEWELL et al., 2019).

CONCLUSION

The state of Rondônia presents high erosivity, soil classes with different erodibilities, and predominantly, reliefs with low vulnerability to erosion. Therefore, vegetation cover and management of support practices were the main factors responsible for the soil loss variations.

The RUSLE estimated water erosion with acceptable precision, indicating that, in 19% of the State of Rondônia, soil losses were greater than the tolerable limits (T), and these areas should be prioritized for adopting measures to mitigate the process.

Results of this study can contribute to the elaboration of different soil management scenarios, as well as to provide alternatives to agricultural and environmental policymakers, encouraging soil conservation in the state of Rondônia.

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DECLARATION OF CONFLICTS OF INTERESTS

The authors declare no conflict of interest. The founding sponsors had no role in the design of the study; in the collection, analysis, or interpretation of data; in the writing of the manuscript, and in the decision to publish the results.

AUTHORS' CONTRIBUTIONS

All authors contributed equally for the conception and writing of the manuscript. All authors critically revised the manuscript and approved of the final version.

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7 ARTIGO 5 - MODELING OF SOIL LOSS BY WATER EROSION AND ITS IMPACTS ON THE CANTAREIRA SYSTEM, BRAZIL

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Abstract: The Cantareira System is one of the largest water supply systems in the world, supplying about half of the water consumed by 22 million inhabitants in the Metropolitan Region of São Paulo, in southeastern Brazil. In this scenario, in view of climate change, silting is a serious environmental threat and a major challenge to the sustainability of water reservoirs. Therefore, identifying the provenance of sediments is an essential tool to support soil conservation policies, slowing erosion processes and mitigating the deposition of sediments in water reservoirs. Thus, this study aimed to model soil losses—sediment production, by water erosion in the Cantareira System, based on the RUSLE model—Revised Universal Soil Loss Equation, GIS—Geographic Information System and SR—Remote Sensing. The work was conducted on data obtained from online platforms of Brazilian public institutions. The results indicate an average rate of soil loss of 13 Mg ha^{-1} yr^{-1}, which corresponds to an annual loss of 3 million tons, of which 22% reaches water bodies. The data also show that: (1) in 66 % of the Cantareira System, soil losses are below the soil loss tolerance limits, and, in 34% of the region, water erosion is compromising the sustainability of water and soil resources; (2) the areas with the greatest soil losses are predominantly located in planted forests, agricultural crops and non-vegetated areas; and (3) sectors with high rates of soil loss require the adoption of conservationist practices aimed at reducing sediment production rates and thereby increasing supply and improving water quality.

Keywords: soil losses; sediment delivery; soil conservation; RUSLE

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1. Introduction

Life on earth depends on topsoil, a resource that is rapidly degraded but takes hundreds to thousands of years to regenerate [1]. Water erosion, the main cause of soil degradation, is a natural process that is intensified by human activities and causes losses of soil and organic matter and reduced fertility [2–4]. In addition, water erosion deteriorates the ecological environment, as sediments also carry nutrients such as nitrogen, phosphorus, potassium and pollutants to water bodies [5,6]. Sediments deposited in watercourses lead to siltation and, consequently, a reduction in the useful volume of water resource reservoirs [7–12].

The Cantareira System is one of the largest water supply systems in the world; it has six reservoirs interconnected by a channel and 48 km of tunnels. It has the capacity to supply 33 thousand liters of water per second, crossing the natural barrier of the Serra da Mantiqueira and supplying almost half of all the water consumed by the 22 million inhabitants of the Metropolitan Region of São Paulo in southeastern Brazil [13,14]. The Cantareira System is a work of engineering from the 1970s that was planned to provide clean and abundant water in a scenario of severe environmental and land use changes, water stress and acute pollution [14–16].

The eroded sediments that enter the water supply system cause the degradation of its quality due to increased levels of suspended solids, which increases the need for water treatment and the costs involved in this process [17,18]. In addition, sediment pollution contributes to the wear and tear of the water treatment and distribution infrastructure, requiring more frequent maintenance and replacement of equipment utilized in the turbidity removal process [14,19,20]. In this scenario, it is relevant to carry out innovative studies to advance knowledge and assist in the management of environments of this nature, whether in the Cantareira System or in other water reservoirs worldwide.

Models for estimating water erosion are tools capable of quantitatively estimating the rates of soil loss and sediment deposition and thus help propose effective erosion control practices [21,22]. Modeling is a simple, easy-to-interpret technique that requires minimal resources and can be implemented with readily available information in areas exposed to high erosion risks. Additionally, this tool allows the assessment of large-scale erosion rates, overcoming the main limitation of experimental plots in the field [23,24].

Considering the lack of studies on soil losses and their impacts throughout the Cantareira System, the objective of the present study was to model soil losses due to water erosion in the Cantareira System and to identify priority areas for the implementation of mitigation measures. It also aims, from the spatial distribution of the data, to identify the areas with the highest rates of soil loss and thereby define those that are priorities for intervention and impact mitigation. Furthermore, it also aims to encourage scientific debate within the scope of water resource reservoirs, contributing to their more effective management.

2. Materials and Methods

2.1. Study Area

The Cantareira System is a hydraulic set of structures that ensures the transfer of part of the flow from the Piracicaba, Capivari and Jundiaí watersheds to the largest metropolitan region in Brazil, and one of the largest in the world. The system is located in southeastern Brazil between coordinates 45°51′47″ to 46°42′40″ W and 22°36′40″ to 23°25′52″ S (Figure 1). The Cantareira System occupies an area of 228,278 ha, with 55% in the state of São Paulo and 45% in the state of Minas Gerais. According to the Köppen classification, the climate is predominantly temperate oceanic (Cfb), with cold, dry winters and hot, humid summers. The average annual rainfall is 1570 mm, and the average annual temperatures range from 18° to 20 °C [25,26].

The altitude ranges from 734 to 2026 m (Figure 1B). The lowest areas (<900 m) are in the state of São Paulo, in the municipalities of Mairiporã and Nazaré Paulista, while the highest areas (>1700 m) are in the state of Minas Gerais, in the municipalities of Camanducaia and Sapucaí-Mirim. The region has an average slope of 24%, with a predominance of strongly undulating relief (20–45%). The areas with flat and gently undulating reliefs (<8%) are in the valleys of the main rivers, whereas the steepest reliefs (>20%) are distributed throughout the region on the faces of the hills and mountains of the Cantareira System (Figure 1C).

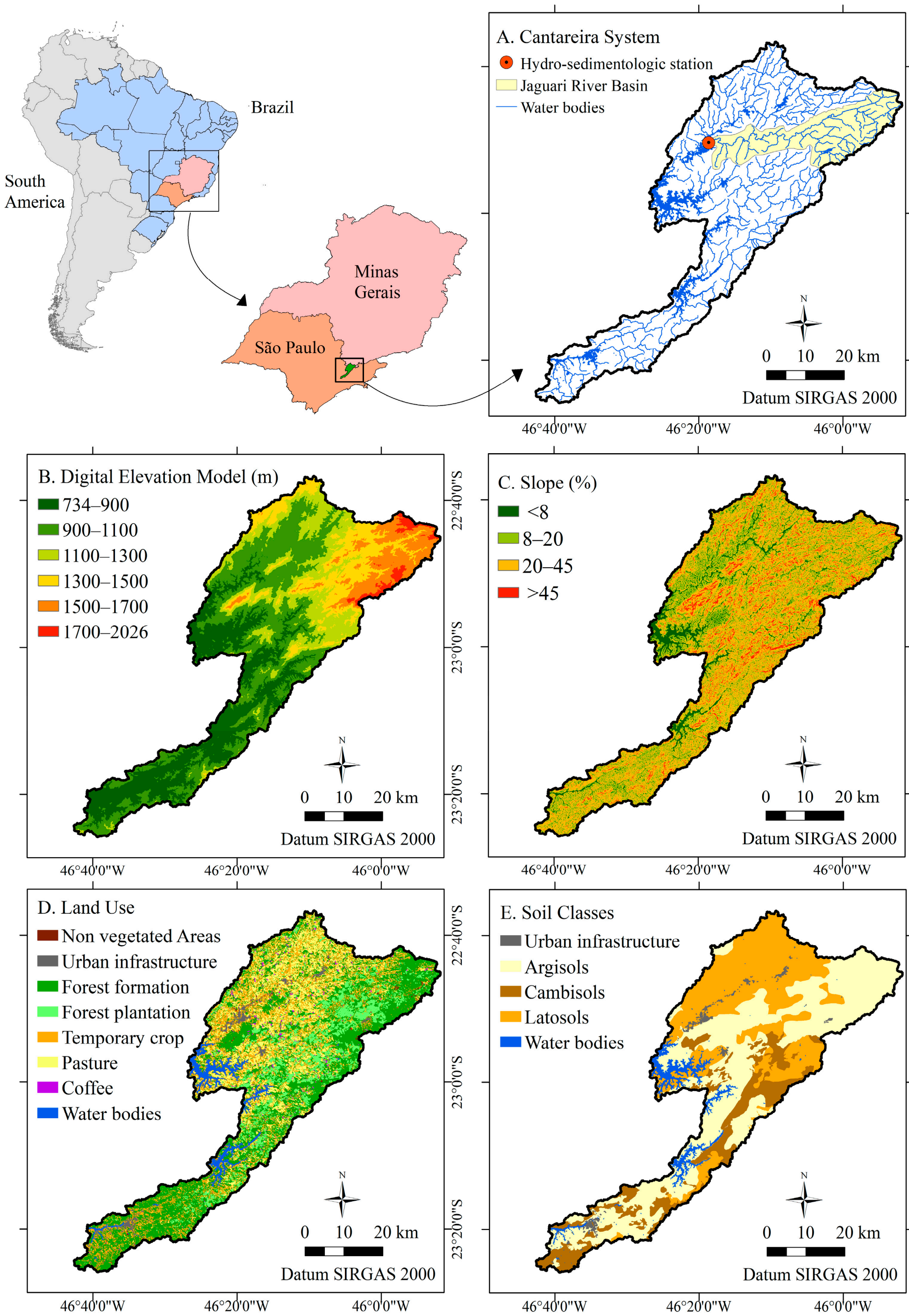

Figure 1. Study area settings. (**A**) Location of the study area; (**B**) digital elevation model (m a.s.l.); (**C**) slope map; (**D**) land use map from Mapbiomas, Collection 7 [27] and (**E**) soil class map of the Cantareira System adapted from soil maps of the states of Minas Gerais [28] and São Paulo [29]. The geographic coordinate system is UTM Sirgas 2000. The basemap is available on ArcMap v. 10.5.

The Cantareira System is occupied by forest formations (45.1%), planted forest (8.46%), pastures (18.80%), temporary crops (22.47%), coffee cultivation (0.27%), water bodies (3.18%), urbanization (1.47%) and non-vegetated areas (0.25%) (Figure 1D) [27]. The pastures present in the Cantareira System are mostly degraded with low productivity and high negative environmental impact [26,30,31]. The planted forest consists mainly of eucalyptus plantations, an important economic activity in the region, with two types of demands: the wood is mainly utilized as fuel, and part of the production is allocated for paper manufacturing [26]. The soils of the Cantareira System consist of Argisols (45.1%), Latosols (34.1%) and Cambisols (16.15%) [30,31] (Figure 1E).

2.2. Revised Universal Soil Loss Equation (RUSLE)

The Universal Soil Loss Equation (USLE) [32] and its revised version (RUSLE) [33] are the most popular and applied empirical models to predict soil loss by erosion worldwide [34,35]. The RUSLE is a practical, easy-to-interpret and flexible model for different edaphoclimatic conditions. In addition, the application can be integrated with geoprocessing techniques, which improve the accuracy of its results, requiring minimal resources. The RUSLE calculates an area's average annual soil loss by multiplying its factors according to Equation (1).

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

where A is the average annual soil loss in Mg ha^{-1} yr^{-1}; R is the rainfall erosivity factor in MJ mm ha^{-1} h^{-1} yr^{-1}; K is the soil erodibility factor in Mg ha^{-1} MJ^{-1} mm^{-1}; LS is the topographic factor, dimensionless; C is the soil use and management factor, dimensionless; and P is the conservation practices factor, dimensionless.

The general flowchart of the methodology demonstrates the procedures adopted to obtain soil loss rates (Figure 2).

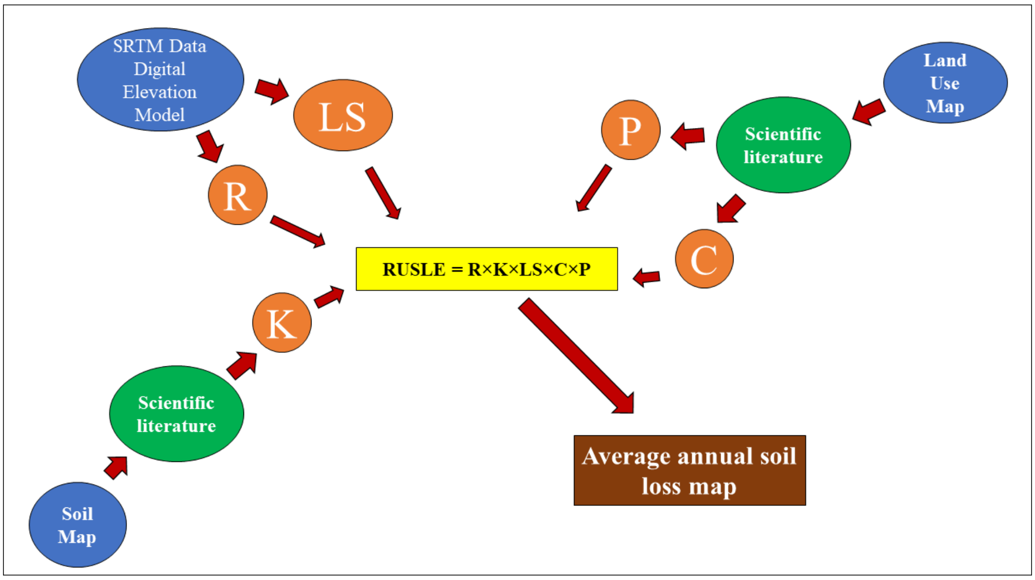

Figure 2. Flow chart of the methods used to estimate soil erosion.

2.2.1. Rain Erosivity (R)

Erosivity (R) represents the potential of rainfall and its associated runoff to cause erosion in unprotected soil. The R factor considers the kinetic energy of the rain and its maximum intensity within 30 min [32].

Due to the lack of detailed rainfall records in the Cantareira System, the R factor was generated according to the multivariate geographic model for Brazil [36] (Equation (2)), which allows estimating the R factor from the latitude, longitude and altitude of the studied area. Such methodology [36] has provided accurate results in several studies in other Brazilian regions [34,37,38].

$$\begin{gathered} R = -399433 + 420.49 \times A - 78296 \times LA - 0.01784 \times A^2 - 1594.04 \times LA^2 \\ +195.84 \cdot LO^2 + 17.77 \times A \times LO - 1716.27 \times LA \times LO + 0.1851 \times LO^2 \cdot A \\ +0.00001002 \times LO^2 \times A^2 + 1.389 \times LA^2 \times LO^2 + 0.01364 \times LA^2 \times LO^3 \end{gathered} \tag{2}$$

where R is the rainfall erosivity factor in MJ mm ha^{-1} h^{-1} yr^{-1}; A is the altitude in m; and LA and LO are the latitude and longitude, respectively, both in negative decimal degrees.

2.2.2. Soil Erodibility (K)

The erodibility (K) expresses the susceptibility of soil to suffer detachment of its particles by the impact of raindrops and by the surface runoff [33]. This factor is calculated in experimental plots based on the physical, chemical and mineralogical properties of the soil. However, determining factor K in the field is a process that requires several years of high-cost experiments. Therefore, factor K values were obtained from the erodibility of soils of the State of São Paulo [39,40], which are 0.0228, 0.0254 and 0.0162 Mg ha^{-1} MJ^{-1} mm^{-1} for Argisols, Cambisols and Latosols, respectively.

2.2.3. Topographic Factor (LS)

The LS factor represents the influence of length (L) and the inclination (S) of an area on soil water erosion. The LS factor was calculated by Equation (3) [41].

$$LS = (m+1) \times \left(\frac{FA \times 30}{22.13}\right)^m \times \left(\frac{\sin(S)}{0.0896}\right)^n \quad (3)$$

where LS is the dimensionless topographic factor; FA is the flux accumulation expressed as the number of grid cells in the digital elevation model; S is the slope of the area in degrees; m and n are empirical parameters that vary from 0.4 to 0.6 and from 1.0 to 1.4, respectively, according to the predominant type of erosion (sheet or rill); and 30 is the spatial resolution of the digital model elevation in meters.

The parameters m and n were defined as 0.4 and 1.0, respectively, assuming the prevalence of sheet erosion in the area, due to forest formations and pastures representing the highest proportion of land use.

2.2.4. Soil Use and Management Factor (C) and Factor of Conservation Practices (P)

The soil use and management factor represents the effect of all soil management and vegetation cover variables on water erosion. The C factor is obtained in the field in long-term experiments. This factor can also be determined from the scientific literature, considering the land use classes with characteristics like those in the study area. Therefore, the C values for the Cantareira System were defined from the modeling of soil losses in an area of the state of São Paulo [42]. From the land use map, C values were defined for each use class: 0.0004 for native forests and 0.047 for planted forests. For the other land use classes, the C factor was 0.05 for pastures, 0.02 for temporary crops, 0.0135 for coffee-growing areas and 1 for non-vegetated areas. Urbanization and water bodies were not considered in the calculation of soil losses.

Due to the large dimensions of the Cantareira System area, factor P in situ was not determined. Thus, the parameter was defined for each land use class based on specialized literature. In the class of use "non-vegetated areas", the value of P was 1; in temporary cultivation, coffee and pasture cultivation was 0.35; and in forest and planted forest was 0.2 and 0.56, respectively [35].

2.3. Geoprocessing and Spatial Analysis

To obtain the parameters and modeling, all the data processing steps were developed using ArcMap 10.5 [43]. The digital elevation model (DEM), with a spatial resolution of 30 m, was obtained based on elevation data extracted from the digital platform "United States Geological Survey" [44]. The slope map was prepared based on the digital elevation model with the Slope tool [43]. The land use map was extracted from Collection 7 of 2021, on the MapBiomas platform [27]. MapBiomas is a collection of annual land use and land cover maps in Brazil. They are made from several observations and a large database, which generate high-precision maps [34,45,46]. The soil map was prepared using the

Union tool [43], based on the revised and enlarged soil map of the state of São Paulo (Scale 1:250,000) [29], and the soil map of the state of Minas Gerais (Scale 1:650,000) [28].

The calculation of the R factor (Equation (2)) was performed using the Raster Calculator tool [43] for each cell, with a spatial resolution of 30 m in the digital elevation model. From the soil map, the values of the K factor were assigned according to the soil classes of the Cantareira System using the Editor tool [43]. The soil map was converted to a 30 m spatial resolution raster file by the Polygon to Raster tool [43].

Regarding Equation (3), the FA parameter was calculated from the digital elevation model (Figure 1B) using the Flow Accumulation tool [43]. The slope of the Cantareira System (S) was determined from the slope map. The LS factor was calculated using the Raster Calculator tool [43] based on each cell of the digital elevation model. Using the Editor tool [32], the values of the C and P factors were assigned to each land use class of the Cantareira System. The C and P factors were converted to a raster file with a spatial resolution of 30 m by Polygon to Raster tools [43].

The RUSLE factors were multiplied by the Raster Calculator tool [43], giving the spatial distribution of soil losses. The resulting maps were represented with a resolution of 30 m, standardized by the cell size of the digital elevation model (Figure 1A). The soil losses in the Cantareira System were classified as low (<10 Mg ha^{-1} yr^{-1}), moderate (10–20Mg ha^{-1} yr^{-1}), high (20–50 Mg ha^{-1} yr^{-1}) and very high (>50 Mg ha^{-1} yr^{-1}) [35] by the Reclassify Raster tool [43]. In addition to determining priority areas for the adoption of conservation management, the results were compared with the Soil Loss Tolerance (T) [47] values by the Raster Calculator tool [43]. In the Cantareira System, the T values were determined to be 10.5, 11.6 and 13.9 Mg ha^{-1} yr^{-1} for the Argisols, Cambisols and Latosols, respectively [39,48].

2.4. Data Validation

The RUSLE estimates all water erosion that occurs in each area, both the soil that is eroded and retained in the relief and the fraction of soil that reaches water bodies. The estimation of sediment production, that is, the fraction of soil that reaches water bodies, is possible by integrating the sediment delivery rate coefficient (SDR) with the RUSLE model. The SDR was obtained by Equation (4) [49].

$$\mathrm{SDR} = 0.472 \times \mathrm{A}^{-0.125} \quad (4)$$

where SDR is the sediment delivery rate in %, and A is the catchment area in km^2.

Sediment production can be directly observed and measured in the field, and the data are generally obtained from hydro-sedimentological monitoring stations of national monitoring institutions. The observed sediment delivery rate can be used to validate soil loss estimates. Therefore, after integrating the RUSLE results with the SDR, it was possible to validate the model results using data from total sediments transported with water discharge and daily runoff [50].

First, a curve was constructed relating the total sediments transported in the watershed and the water discharge (Figure 3). Data monitored between 2016 and 2021 at a hydro-sedimentological station located in the Jaguari River watershed were utilized (Figure 1A). This station is maintained by the Minas Gerais Institute for Water Resources Management—IGAM, and the data were obtained from the Hidroweb platform of the National Water and Basic Sanitation Agency. Although there are several sediment monitoring stations in the Cantareira System, there are stations with discontinuous and sparse measurements, and some are inoperative [51]. Due to this, the validation was based on a sub-catchment [50] of the study area.

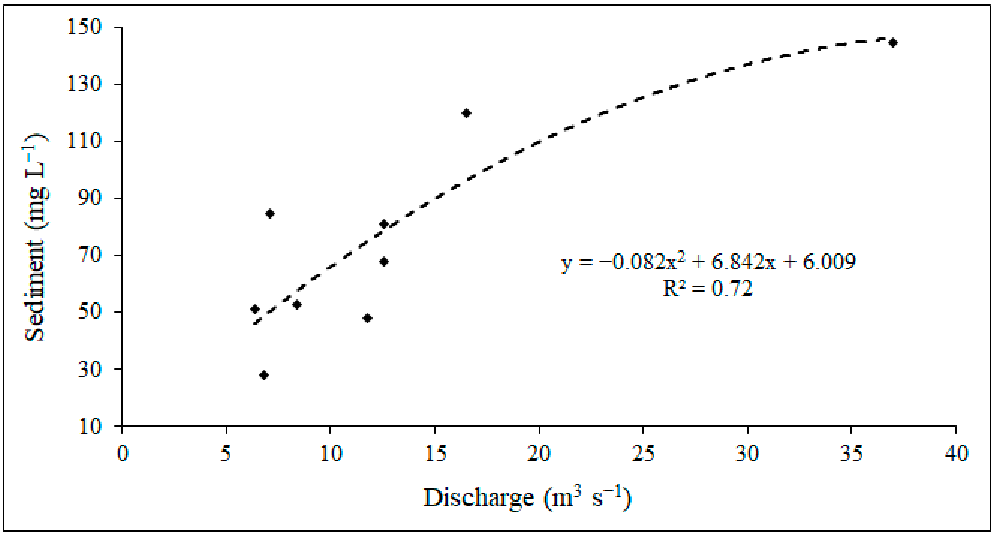

Figure 3. Water discharge curve (sediment transported × water discharge) in the Jaguari River Hydrographic Subbasin, Brazil.

The sediment production observed at the hydro-sedimentological station was calculated considering the sediment x flow curve and the daily flow dataset, referring to the Jaguari River Subbasin (400.5 km^2), obtained from the National Water and Basic Sanitation Agency. The observed sediment was compared with the sediment estimated by the RUSLE/SDR.

3. Results and Discussions

3.1. RUSLE Factors

In the Cantareira System, the R factor ranged from 7203 to 12,448 MJ mm ha^{-1} h^{-1} yr^{-1}, with an average value of 7843 mm ha^{-1} h^{-1} yr^{-1} (Figure 4A). The values show the region as of "strong erosivity" [36], due to heavy rains in the Cantareira System. The R values obtained are also consistent with those of the Tietê River watershed [34], adjacent to the studied area. In the Cantareira System, the highest values of R are associated with the highest altitudes in the northeast of the area. The highest R values occur predominantly in regions with a predominance of Argisols.

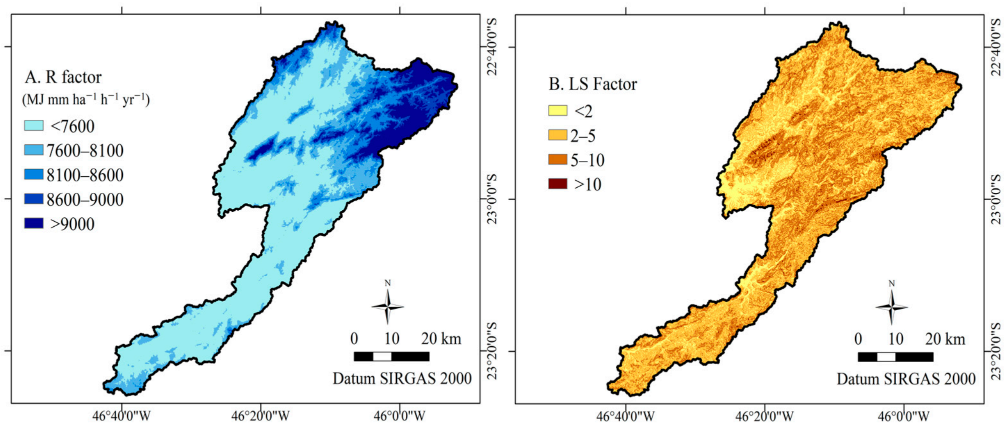

Figure 4. Map of the spatial distribution of the rainfall erosivity factor—R (**A**) and the topographic factor—LS (**B**) in the Cantareira System, Brazil. Note: factor LS = dimensionless. The geographic coordinate system is UTM Sirgas 2000. The basemap is available on ArcMap v. 10.5.

The average value of the LS factor is 4.2, and the maximum is 34 (Figure 4B). In this range, the sites with LS greater than 10, which represent only 1% of the area, may be highly susceptible to water erosion [50]. Despite the low percentage, such areas are distributed throughout the area (Figure 4B), which reveals the importance of sustainable soil and water management in tropical watersheds in southeastern Brazil [37]. The steepest areas play an important role in the hydrological cycle. Such areas concentrate the surface runoff of rainwater, which makes the soils more vulnerable to erosion and mass movements, causing damage to the quality of water and soils [26].

In the Cantareira System, in addition to the high values of R and LS (Figure 4), there is a predominance of Argisols and Cambisols with high erodibility (Figure 1E) and, consequently, high susceptibility to erosion associated with lower tolerance limits for soil loss. Therefore, sustainable management practices for C and P factors are essential to protect soils from erosion. In addition, in the Cantareira System, land use based on agricultural aptitude and the ability to use each soil is essential [52]. The sustainable use of the soil promotes the maintenance of organic matter in the soil, the improvement of its structure and water infiltration, the reduction of surface runoff and the need to add fertilizers and pesticides, and the mitigation of damages related to contamination, eutrophication and silting up of water bodies [34].

3.2. Soil Losses and Priority Areas for Conservation Management

In the Cantareira System, in 62% of the area, low soil loss rates predominate, and, in 13.7%, they are moderate. However, in 17.3% (39,492 ha), soil losses are high and, in another 7% (15,979 ha), very high (Figure 5).

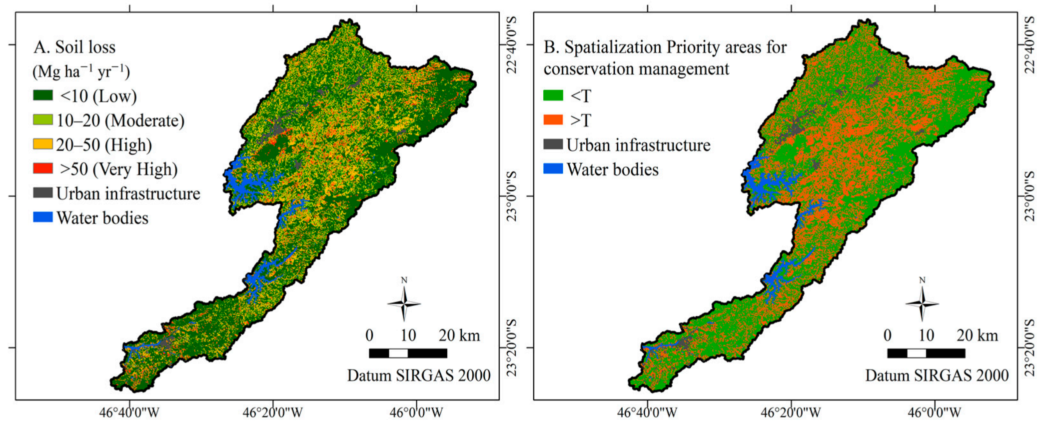

Figure 5. Map of the spatial distribution of soil loss rates (**A**) and priority areas for conservation management (**B**) in the Cantareira System, São Paulo, Brazil. The geographic coordinate system is UTM Sirgas 2000. The basemap is available on ArcMap v. 10.5.

The soil loss estimates in the Cantareira System were compared with the Soil Loss Tolerance (T), which consists of an indicator index of the maximum intensity of the erosion process that still allows an economically sustainable production of agricultural crops [32,53]. Thus, the T values allow identifying priority areas for the adoption of erosion mitigation measures to promote soil sustainability [49,54]. In the Cantareira System, 34% of the area has losses above the T limits (Figure 5B).

The modeling by RUSLE estimated an average rate of soil loss for the region at 13 Mg ha^{-1} yr^{-1}, which corresponds to an annual loss of about 3 million tons. On average, the results were well below the average of 30 Mg ha^{-1} yr^{-1} obtained by RUSLE for the entire state of São Paulo [52]. On the other hand, the results are similar to those of the Tietê River watershed, also in the state of São Paulo, with an average rate of soil loss of 8.9 Mg ha^{-1} yr^{-1} [34]. The variation in RUSLE results in areas that make up the same region could be attributed to methodological differences in obtaining its factors. However, the application of different methods for obtaining RUSLE factors in the same area resulted in similar magnitudes of soil loss rates [50,55]. The selection of the methodology for calculating the RUSLE factors depends on the data available for the area and the possibility of comparing the results with the erosion values observed in the field. However, in this case, the adopted methodology was validated by the data of total sediments observed in a hydro-sedimentological station.

The predominance of areas with low soil loss in the Cantareira System (66%) is mainly due to the high percentage of forest formations (45.1%). The average rate of soil loss in this use was very low, 0.1 Mg ha^{-1} year^{-1}, which points to the importance of natural and reforested vegetation cover in reducing soil loss and water conservation [34,35]. Notably, in the Cantareira System, important initiatives have recognized the role of forest cover in water management and its benefits, such as the reduction of erosion. These initiatives generate measures to implement forest restoration and conservation as a central strategy to protect the Cantareira System from the water crisis [14].

In the Cantareira System, soil and water conservation programs are worth mentioning, including the Nascentes Program of the state government of São Paulo, whose objective is to optimize and direct public and private resources for the restoration of degraded areas, the river basin committees that are responsible for water management planning, mediating conflicts over water resources and defining mechanisms for charging for water use, and the Water Producer Program of the Municipality of Extrema in Minas Gerais, created in 2005, which is a payment program for environmental services, whose main objective is to increase forest cover in sub-basins that drain into the Cantareira System, aiming to control the impacts of soil erosion on water quality to increase water infiltration and to promote aquifer recharge. The municipality allocates approximately 3% of its budget to the program, demonstrating the support of the local community with soil and water conservation [14,51].

It is worth highlighting the role of the Basic Sanitation Company of the state of São Paulo (SABESP), the public institution responsible for operating the Cantareira System, in managing the region's soils. SABESP has been committed to reforesting and restoring its private areas and conserves approximately 35 thousand hectares of the Cantareira System [14]. In addition, federal and state laws, such as the Brazilian Forest Code and the São Paulo watershed law, are important legal instruments to encourage the conservation of natural resources in the Cantareira System.

Even in the face of water management and soil erosion reduction initiatives, 34% of the Cantareira System soil losses are above the limits of T. These areas are distributed throughout the region (Figure 5B). Considering that planted forests, agricultural crops and areas without vegetation presented average soil losses higher than T values (Figure 6), they are a priority for the adoption of measures to mitigate water erosion.

To reduce the impacts on the hydro-sedimentological cycle, mechanical, edaphic and vegetative conservationist practices must be adopted. In steep areas, techniques to slow surface runoff should be used, such as vegetation cords, border strips and windbreaks in watershed dividers. Channels and stairs can be used to direct flows from the slopes and relief breaks to the infiltration basins. In low areas and floodplains, the maintenance of riparian forests is essential. In general, level planting and terracing are efficient and relatively low-cost practices and should be widely used [34,56].

In temporary crops, the alternatives are direct planting, cultivation with minimal soil disturbance and alternating planting with legumes. In coffee and silviculture, practices that favor the maintenance of living or dead vegetation cover, such as alternating weeding and joint planting, combined with dense planting, are options. Integrated cropping, livestock and forestry systems, pasture rotation and edaphic practices can also be adopted [34,56].

The conversion of non-vegetated areas and degraded pastures into areas of sustainable agricultural production will promote the growth of agricultural production and the supply of ecosystem services and, at the same time, will prevent soil erosion [56]. Furthermore, areas with very high soil losses should be priority targets for conservation programs for the region's natural resources. Identifying these priority areas for mitigating water erosion is essential for managing the Cantareira System, as random reforestation of 2% of the area would reduce sediment production by 8% [14]. However, by directing the restoration towards areas of greater susceptibility to erosion, the carrying of sediments into water bodies can be reduced by 36% [14].

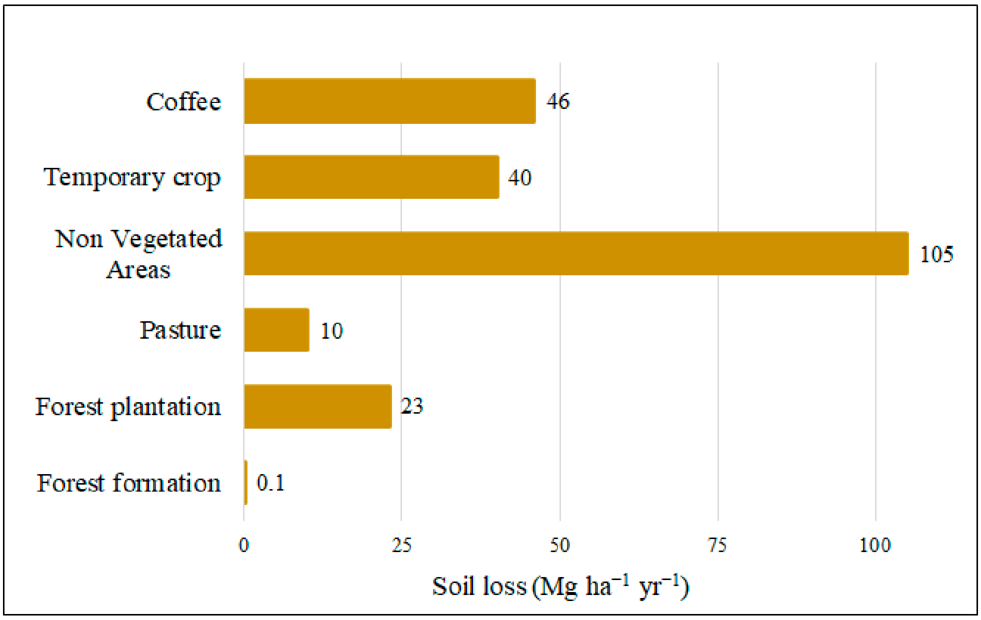

Figure 6. Land use classes and soil loss rates in the Cantareira System, São Paulo, Brazil.

3.3. Validation of Results

The sediment delivery rate (SDR) in the Jaguari River watershed was calculated at 0.22, indicating that 22% of soil losses in the region reach water bodies. By RUSLE/SDR integration, it was estimated that the region has an average sediment generation of 2.8 Mg ha^{-1} yr^{-1}, whereas the observed sediment transport, which was calculated based on data from the hydro-sedimentological station, was 0.8 Mg ha^{-1} yr^{-1}. Therefore, the RUSLE model overestimated soil losses in the Cantareira System region, like the overestimation also observed in modeling soil losses in the Rio Grande Basin in southeastern Brazil [50]. These results are supported by studies comparing RUSLE with other models for estimating soil loss and sediment generation, which showed that RUSLE tends to overestimate the results [57].

RUSLE overestimates soil losses and presents tolerable errors, but greater accuracy of the results is observed in areas with greater soil losses [55]. The spatial distribution of soil losses generated by RUSLE has good accuracy, and sites with high estimates are most susceptible to the erosion process [58].

Water erosion modeling is a representation of reality, and not reality itself, and is therefore subject to errors, which in most cases are acceptable [59]. In general, measured and modeled soil loss rates show the same order of magnitude [60,61]. Therefore, modeling is an efficient way to identify areas with different soil management [59].

Thus, the estimation of the RUSLE model for the Cantareira System should be interpreted as a tool capable of identifying areas with greater susceptibility to erosion and thereby directing resources and actions to priority locations in the adoption of conservationist measures for soil and water. In addition, the evaluation of erosion in large areas such as the Cantareira System is only economically viable by modeling. Estimating the magnitude of erosion and its spatial trends is essential for planning conservationist soil management and water conservation.

4. Conclusions

In the Cantareira System, 66% of the area has low soil losses, mainly attributed to areas with forest formations, which are the result of soil and water conservation programs implemented in the region. There are still areas with large soil losses (34%), demonstrating that advances are needed in the adoption of conservationist practices in crops, aiming to reduce the generation of sediments and increase the availability and quality of water in the Cantareira System.

The adopted methodology identified areas susceptible to soil loss and defined priority areas for the implementation of mitigation measures. The results contribute to the scientific debate on future works, bringing more information and knowledge to society and sector managers.

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8 CONSIDERAÇÕES FINAIS

Nas regiões brasileiras existem áreas com altas taxas de erosão hídrica o que tem agravado a degradação dos solos. Em todas áreas estudadas, as maiores perdas de solo estão associadas aos relevos íngremes, com cultivos agrícolas de baixa densidade de cobertura vegetal e ausência de práticas conservacionistas. As mudanças de uso e cobertura da terra são o principal agravante da erosão hídrica nos solos brasileiros, principalmente quando ocorre o desmatamento, como visto na Região da Bacia Hidrográfica do Rio Xingu. Além disso, as mudanças no uso da terra e as variações na cobertura vegetal sem respeitar a aptidão agrícola de cada solo comprometem severamente os sistemas de produção agrícola, principalmente nas regiões com presença de solos de maior suscetibilidade a erosão hídrica, como os Argissolos, Cambissolos e Neossolos.

A identificação das áreas com erosão severa por meio da distribuição espacial das perdas de solo permite que os gerenciadores e os tomadores de decisões direcionem práticas conservacionistas especificas a fim de minimizar as perdas de solo. Após a identificação dos locais prioritários para mitigação da erosão, o ideal é que as práticas conservacionistas de solo e água sejam adotadas e maximizadas em todas as regiões brasileiras, visto os vários benefícios que essas técnicas podem trazer ao ambiente, como por exemplo a manutenção da fertilidade e da microbiota do solo, a redução das perdas de carbono orgânico e, consequentemente, a redução das emissões de gases do efeito estufa.

Os modelos RUSLE e EPM podem ser aplicados mesmo em regiões brasileiras com poucas informações fornecendo resultados precisos. A RUSLE apresenta várias metodologias para cálculo de seus parâmetros e várias informações de aplicações prévias em solos brasileiros, o que demanda dos pesquisadores a estruturação de uma metodologia de modelagem consistente e capaz de minimizar os erros da aplicação do modelo. Para a utilização do EPM os pesquisadores precisam ser cautelosos na seleção de seus valores tabelados, tentando conciliar estes valores da melhor forma possível com as características edafoclimáticas de cada região brasileira. Com a aplicação dos modelos com auxílio de SIG e sensoriamento remoto, durante o levantamento dos parâmetros deve se priorizar a utilização de dados de maior precisão e maior resolução espaço-temporal, quando disponíveis.

A maior dificuldade de aplicação da modelagem em solos brasileiros é a validação dos resultados. Validações feitas com base em trabalhos de campo são onerosas e exigem um tempo considerável de avaliação. Além disso, os dados de estações hidrossedimentológicas usados para validações são escassos e descontínuos e muitas estações, atualmente, estão inoperantes.

Uma solução recomendada seria que órgãos públicos ampliassem o número de estações hidrossedimentológicas operantes e garantissem a manutenção daquelas que já estão ativas. As universidades e outros centros de pesquisa também poderiam estruturar projetos de avaliação da erosão em campo a longo tempo, a fim de auxiliar na validação dos resultados e calibração de parâmetros dos modelos.

A modelagem é uma técnica valiosa para estimar quantitativamente a erosão hídrica e avaliar suas interações com outros fatores, assim como também para fornecer um diagnóstico da variação do processo erosivo no tempo e no espaço. Essa ferramenta, quando bem utilizada, pode auxiliar no manejo conservacionista dos solos brasileiros. A modelagem, quando aplicada em grandes áreas, como nos casos do Estado de Rondônia e da Bacia Hidrográfica do Rio Tietê, também deve ser entendida como uma forma de destacar a problemática da erosão hídrica, conscientizar a população a respeito da necessidade de minimização desse processo e incentivar a elaboração e adoção de políticas ambientais voltadas a conservação do solo.

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