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LAND-USE CHANGES AND IMPACTS ON SIMULATING WATER-RELATED COMPONENTS IN A SUBTROPICAL AGRICULTURAL RIVER BASIN

Santa Maria, RS
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A thesis submitted to the PhD Program in Forest Engineering at Universidade Federal de Santa Maria (UFSM, RS) in partial fulfilment of the requirements for the degree of **PhD in Forest Engineering - Silviculture.**

Supervisor: Prof. PhD José Miguel Reichert

Santa Maria, RS
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RESUMO

ALTERAÇÕES NO USO DO SOLO E SEUS IMPACTOS NA SIMULAÇÃO DE COMPONENTES HIDROLÓGICOS NUMA BACIA HIDROGRÁFICA AGRÍCOLA DE CLIMA SUBTROPICAL

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As mudanças de uso e cobertura da terra (LULC) são um dos principais fatores de alteração dos ecossistemas globais, moldando não apenas a paisagem, mas também impactando a contribuição da natureza para as pessoas, incluindo os serviços relacionados à água. Diferentes tipos de uso da terra e de manejo influenciam a hidrologia tanto em escala local quanto em escala de bacias hidrográficas. Portanto, compreender a dinâmica do LULC e como elas afetam os componentes hidrológicos é vital para mitigar seus impactos no ecossistema natural e nos recursos hídricos. Nesse trabalho foi investigada a dinâmica espacial e temporal da bacia do Guaporé (2.490 km²), sul do Brasil, e seu efeito sobre os recursos hídricos. Para um melhor entendimento, o estudo foi dividido em três capítulos. Primeiro, as diferenças espaciais e temporais na distribuição do uso do solo foram computadas aplicando diferentes métricas de paisagem. Em seguida, uma análise de autocorrelação temporal e espacial foi aplicada para identificar tendências na mudança do uso da terra, seguida pela construção de um modelo de autômatos celulares para avaliar os principais fatores de LULC. Finalmente, simulamos a resposta dos processos hidrológicos às mudanças de LULC, utilizando o *Soil & Water Assessment Tool* (SWAT). Os resultados mostraram mudanças no uso da terra ao longo do tempo na bacia do rio Guaporé. A maioria das mudanças é explicada em nível local por fatores sociais e econômicos, onde a intensificação da atividade agrícola promoveu a homogeneização e reduziu a complexidade da paisagem. Estas mudanças observadas destacaram a importância de se considerar a dinâmica da paisagem para avaliar os recursos hídricos. Embora, tanto o cenário estático quanto o dinâmico tenham produzido uma simulação "satisfatória" das vazões históricas, uma análise baseada em um único mapa de base poderia resultar em uma representação irrealista do balanço hídrico, uma vez que diferentes tipos de uso da terra implicam mudanças na infiltração e escoamento de água, interceptação pela copa das plantas, entre outros. Finalmente, nossos resultados destacaram que o controle da mudança de LULC é essencial para quantificar os recursos hídricos e para a gestão da água a longo prazo.

Palavras-chave: Estrutura da paisagem, SWAT, uso e cobertura da terra, dinâmica hídrica, bacia hidrográfica

ABSTRACT

LAND-USE CHANGES AND IMPACTS ON SIMULATING WATER-RELATED COMPONENTS IN A SUBTROPICAL AGRICULTURAL RIVER BASIN

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Land use and land cover (LULC) changes are one of the main driving forces of Global Change, shaping the landscape and impacting nature`s contribution to people, including water-related services. Different types of land use and their management practices influence hydrology at both field and catchment scales. Therefore, understanding the dynamic of LULC and their effect on the hydrological components are vital to mitigate their impacts on the natural ecosystem and water resources. Here, we investigated the spatial and temporal dynamics of the Guaporé watershed (2,490 km²), southern Brazil, and their effect on water resources. For a better understanding, the study was divided into three chapters. First, the spatial and temporal differences in land-use distribution were computed by applying landscape metrics. Then, an autocorrelation analysis was used to identify trends in the land-use changes, followed by building up a cellular automata model to assess the main drivers of land use. Finally, we simulated the response of hydrological processes to LULC changes with the Soil & Water Assessment Tool (SWAT). The overall results showed that the land use has dramatically changed in the Guaporé watershed. Most of the changes were explained at the local level by social and economic factors, where the intensification of agricultural activity has promoted homogenization and reduced landscape complexity. These observed changes highlighted the importance to consider a dynamic land-use to assess water resources. Although both static and dynamic scenarios produced a "satisfactory" simulation of historical discharge, an analysis based on a single baseline map could result in an unrealistic representation of water balance since different land-use types imply changes in water infiltration, runoff, plant canopy, among others. Finally, our results highlighted that controlling LULC change is essential for long-term water management quantifying water resources.

Keywords: Landscape structure, SWAT, land use, water dynamics, watershed.

SUMMARY

OVERALL INTRODUCTION

The process of land use and land cover (LULC) change, especially in intensively cultivated landscapes, is the dominant force behind environmental degradation (FAO, 2015; IPBES, 2018). The impacts of LULC are reported by numerous studies, e.g., anthropogenic pressures on natural ecosystems (Ribeiro et al., 2021; Souza et al., 2020), deforestation (Armenteras et al., 2017; De Espindola et al., 2021), and fragmentation of natural areas (Haddad et al., 2015; Taubert et al., 2018). LULC also strongly affect nature`s contribution to people, including water-related services (IPBES, 2016, 2019). The type of land cover influences the hydrological processes, such as streamflow (Reichert et al., 2017; Valente et al., 2021), evapotranspiration (Van Meerveld et al., 2021), infiltration (Holder et al., 2019), and runoff (Wang et al., 2021). Therefore, understanding how the dynamic of LULC affects the landscape and hydrological components are vital to mitigate their impacts on the natural ecosystem and the water resources.

To date, most hydrological studies do not represent land-use change as a dynamic process. Instead, to evaluate the impacts of LULC, a so-called delta approach is used. For example, Soil & Water Assessment Tool (SWAT) users often apply the delta approach to assess the impacts of LULC by comparing hydrological responses of different land-use scenarios for the same time frame (Wagner et al., 2019). Landscape ecology also uses the delta approach, known as cross-tabulation, to evaluate the effect of LULC in a natural ecosystem (e.g. Talukdar et al., 2021). The cross-tabulation method is generally based on pixel-to-pixel analysis, which quantifies the changes in land use between the initial (T_0) and final (T_1) landscape maps. However, unlike most hydrological studies, a dynamic simulation of land-use change has been implemented frequently to track deforestation, agriculture expansion, and urban growth (Cheng et al., 2020; Oliveira et al., 2019; Rodrigues & Soares-Filho, 2018; Yi et al., 2012).

In landscape ecology, both spatial and temporal LULC changes are modelled dynamically. Most dynamic LULC models are based on cellular models, specifically cellular automata (CA - Oliveira et al., 2019). CA provides an effective way of simulating and predicting the spatial-temporal evolution of complex geographical phenomena (Liu et al., 2007). The model consists of a regular n-dimensional array of cells that interact within a certain vicinity and according to a set of transition rules (Soares-Filho et al., 2002) represented by many forms. For instance, the CA has the capability to represent the LULC changes by quantifying not only the states of conversion between LULC classes but also the rate of conversion among them (Dai, 2010; Gomes et al., 2020).

Among the model families used when considering Geographic Information System (GIS) spatial analysis for land-use change, DINAMICA-EGO (Environment for Geoprocessing Objects) are frequently reported due to its ability to model land-use changes and assess their impact on the environment (Bielecka, 2020). DINAMICA-EGO is based on a cellular automaton model that uses Weights of Evidence and two mechanisms (patcher and expander) to obtain potential change maps and new LULC scenarios (Oliveira et al., 2019; Rodrigues & Soares-Filho, 2018). Initially named DINAMICA, the model was developed in 2002 by Soares-Filho et al. (2002) to simulate deforestation in the Amazonian region during the last decades of the 20th century. Recently, the DINAMICA was embedded in the DINAMICA-EGO (Soares-Filho et al., 2013), a freeware distributed software. Its main strength lies in the possibility of interacting with other tools, such as R and Python programming languages, which allow the development of sophisticated spatially explicit models. The software is also well-recognized by the possibility of a multi-region and multi-scale approach and the inclusion of decision-making processes in the analysis and simulation of land use (Bielecka, 2020).

As already mentioned, most hydrological studies do not incorporate dynamic land-use processes. Usually, a base LULC map is used as input to simulate historical hydrological processes in hydrological models, and the same land-use layer is used for the entire simulation period. Among the various hydrological models employed for watershed assessment studies, the Soil and Water Assessment Tool (SWAT) is one of the few tools that simulate changes in land use and land cover throughout an optional module – the land use update module (SWAT-LUP, Arnold et al., 2012). However, the workload involved during the process of implementing dynamic changes and the lack of a user-friendly graphic interface may have limited the number of studies using the dynamic land-use until now (Moriasi et al., 2019; Pai & Saraswat, 2011). Furthermore, only recently, remotely sensed data and geographic Information Systems (GIS) to reconstruct the historical LULC information became broadly available (Nagaraj et al., 2020; Souza et al., 2020; Wulder et al., 2012).

An example of the impact of GIS information availability is the *Mapbiomas* network initiative. Based on Google Earth Engine Cloud-computing, freely available Landsat data, and a collaborative network of experts, this initiative reconstructed three decades of Brazilian LULC change (Souza et al., 2020). A detailed description of the methodology and a final opensource product, altogether, allowed the development of numerous studies in different fields (e.g. Alencar et al., 2020; Fendrich et al., 2020; Mas et al., 2019; Parente & Ferreira, 2018;

Parente, Mesquita, et al., 2019; Parente, Taquary, et al., 2019; Rosa et al., 2021; Souza et al., 2019). *Mapbiomas* also revealed the LULC spatial and temporal trends in Brazil, characterised by a widespread reduction of natural ecosystems and expansion of anthropogenic activities (Souza et al., 2020).

Despite its novelty, *Mapbiomas* is just another alert of the land-use change process. Several studies have quantified the dynamics of land-use change all over the globe (FAO, 2015, 2020; Winkler et al., 2021). For example, a recent report showed that at a global level, the land-use changes had affected 32% of the worldwide land area in the last six decades (1960-2019) while identifying worldwide trade and agriculture as the main drivers of global land-use change (Winkler et al., 2021). A similar trend is observed in Brazil, where agriculture has increased in all biomes (Souza et al., 2020). Agriculture expansion is commonly associated with environmental degradation because of the suppression of native vegetation to open new crop fields, threatening biodiversity and forest ecosystems (Andrade De Sá et al., 2013; Armenteras et al., 2017; Maranhão et al., 2019; Richards et al., 2012; Vandermeer & Perfecto, 2007). The expansion of arable lands also causes disturbances such as habitat fragmentation/isolation and edge effect, both of which alters the structure and composition of plant communities (Lewis et al., 2015; Lingner et al., 2020) and modifies the spatial patterns of the landscape (Wästfelt, 2021).

Although the main Brazilian environmental legislation act, the Native Vegetation Protection Law (Law n° 12,651/2012; Brasil, 2012), severely restricted deforestation on private properties by establishing areas of permanent protection (APPs) along water streams and on hilltops as well as legal reserves (native vegetation in a section of the property). Most landowners are in non-compliance with legal rules (Soares-Filho et al., 2014; Sparovek et al., 2010). Meanwhile, little is known about the conservation status, distribution, and quality of natural areas within private lands (Vacchiano et al., 2018). Additionally, the level of compliance to Native Vegetation Protection Law can impact nature`s contribution to people by promoting land use degradation and consequently affecting climate, ecosystem stability, water balance, agricultural yield, socioeconomic practices, and biodiversity (Bongaarts, 2019; Ellis & Ramankutty, 2008; Foley et al., 2011; Garibaldi et al., 2020; IPBES, 2018; Resende et al., 2020).

The lack of information on natural areas' quality and conservation status is even more critical for highly modified ecosystems such as the Atlantic Forest biome (Lima et al., 2015). Intensely exploited since the Brazilian colonial period, the forest area today occupies about

12% of its original size (Joly et al., 2014; Souza et al., 2020), distributed in the form of small (< 50 hectares) and numerous fragments throughout the national territory (Ribeiro et al., 2009). Although over the years, continuous efforts by different sectors of society have managed to reduce deforestation rates for the biome (Rezende et al., 2018), the biome still be considered a biodiversity hotspot, and it is heavily threatened by habitat loss and other human activities (Rezende et al., 2018; Rosa et al., 2021). Moreover, most remaining Atlantic Forest (AF) are within private areas, where intervention and deforestation may be allowed by law (Rezende et al., 2018), while the level of protection is well below the 17% recommended by the 10th Convention on Biological Diversity (Herkenrath & Harrison, 2011).

In the Rio Grande do Sul state, the AF originally occupied about 50% of the state's territory; however, the expansion of agriculture over natural areas has reduced this percentage to only 7.9% of forest remnants (Silveira et al., 2017), with a high degree of fragmentation concerning the original coverage. The Rio Grande do Sul AF comprises several forests, including semi-deciduous, deciduous, and mixed temperate. Despite the diversity of phytophysiognomies our knowledge of the *Rio Grande do Sul* Atlantic forest is limited (Lima et al., 2015). Therefore, further reduction on AF biome could increase habitat isolation, loss of endemic species, and decrease agricultural yields by losses in ecosystem services (IPBES, 2016, 2019; Pascual et al., 2017).

In contrast to the conservation status of nature, the impact of anthropogenic activities on water resources has been widely investigated in Southern Brazil. Several works have been conducted to understand the effect of land use on hydrological dynamics, such as sediment yield (Bonuma et al., 2014; de Menezes et al., 2020; Didoné et al., 2014), water budget fluxes and water balance (Ebling et al., 2021; Ferreto et al., 2021; Reichert et al., 2021; Reichert et al., 2017), sediment sources identification (Valente et al., 2021), and water contamination (Bastos et al., 2021; Becker et al., 2009; Kaiser et al., 2010; Kaiser et al., 2015). Most existing studies focus on large watersheds (e.g., Guaporé). Although, few studies consider land use to evaluate the quality and quantity of water resources. There is a need to incorporate dynamic land-use maps to simulate hydrological dynamics.

Here, we quantify the large-scale, long-term LULC changes and their impacts on water-related components in a subtropical agricultural river watershed in the Rio Grande do Sul, Brazil. The Guaporé watershed (GRB) covers ≈2,490 km², and the main river is a tributary of the Jacuí river system, which includes water withdrawals for the Metropolitan Region of Porto Alegre, the capital of the state. Regarding the vegetation, GRB is in an ecotonal region between Deciduous Seasonal Forest, Mixed Ombrophylous Forest, Grassy Woody Steppe, and Native

Fields (IBGE, 2019). Over 60% of the area is anthropized, comprising small-scale farmland, pastures, and scattered urban infrastructure patches. The annual precipitation is around 1768.12 mm $yr^{-1,}$ and the mean annual minimum and maximum temperatures range from 13°C to 23°C. In terms of geology, the study area belongs to the *Serra Geral* Formation, derived from volcanic lava flows (basalt and rhyolite on the top) and characterized by various facies (Caxias, Gramado, and Paranapanema). The watershed elevation ranges from ~50 m to ~900 m above sea level and presents several slope gradients. The soils are predominantly distributed in five orders: Acrisols (*Argissolo*), Regosols (*Neossolo*), Ferralsols (*Latossolo*), Luvisols (*Luvissolo*), and Nitisols (*Nitossolo)*. GRB was selected to conduct this study due to i) its heterogeneity in terms of soil type, relief, land use, and soil management. Altogether, these conditions are representative of a large set of environmental conditions found in Southern Brazil, and ii) research history in the area, GRB has been a subject of many studies (e.g. Bastos et al., 2021; Didoné et al., 2014; Tiecher et al., 2017).

In this study, we do not intend to accurately portray all potential scope of impacts that may occur due to land-use change. However, we will highlight the main implication of considering dynamic land use in assessing water resources. This work is divided into three chapters written in a scientific journal format. The first chapter addresses landscape structure changes over time and identifies its main trends. The second explores the reduction of forest cover, predicting the main centres of deforestation and correlating it with the watershed's social, economic, demographic, and physical factors. Finally, we incorporate dynamic and static LULC maps in SWAT hydrological model to explore the capability of time series land use land cover maps to simulate historical discharge.

CHAPTER I: HOW DO SMALL CHANGES IN LAND-USE AFFECT LANDSCAPE DYNAMICS AND SPATIAL PATTERNS IN BLURRY-BOUNDARY CONDITIONS?

ABSTRACT

Agriculture and pasture have become the main types of land use worldwide. At the same time, natural areas are converted into small patches scattered over the landscape, threatening the provision of ecosystem services and food production. Therefore, the dominance of rural landscapes over natural ecosystems calls for landscape planning to reconcile production with sustainable use of natural resources. Landscapes dynamic can be quantified by adopting metrics to construct different management scenarios and land use to maximize production while conserving natural resources. Based on these premises, this study sought to understand the spatial and temporal dynamics of the Guaporé watershed (2,490 km^2), southern Brazil, during the years 1998 to 2018, using land use and land cover maps, produced by *Mapbiomas*, and to relate the dynamics of different land uses with the expansion or reduction of natural areas. Over time, we observed dynamism in regional agricultural activity, where large pasture areas were converted into arable lands, while the decline of native vegetation for the period analyzed was around 6%. Although this is a small percentage of reduction, the remnants are distributed in the landscape in numerous small fragments (up to 5 hectares). At the same time, they are more susceptible to exploitation and edge effect. The expansion of agricultural land uses contributed to homogenizing and simplifying the landscape, which may affect various ecosystem services and agricultural productivity. The landscape characteristics observed in this study spotted a light on the relationship between land use shape size and pattern distribution. At the same time, it highlighted the necessity of adopting different strategies to manage and conserve natural resources, such as models based on the land-sharing/land-sparing theory, which can help reconcile conservation and agricultural production for the Guaporé watershed. Preservation of small fragments increases the connectivity of the landscape, a strategic element for the preservation of fauna and flora, and consequently the maintenance of ecosystem services.

1 INTRODUCTION

Land use and land cover change and the advance of industrial agriculture contributed significantly to increasing food, timber, and fibre supply. However, current production levels were achieved at the expense of natural resource conservation. Agricultural and pasture areas are the world's primary land use and cover types (Foley et al., 2011), while natural areas have come to occupy less than a quarter of the Earth's land surface (Ellis & Ramankutty, 2008). This dissonance between the extent of productive and natural areas, together with the intensification of agricultural systems through the adoption of monocultures of high-yielding varieties and the intensive use of chemical and mechanical inputs (Tanentzap et al., 2015), has been causing negative environmental impacts on soil, water, air, and biodiversity (IPBES, 2018) while placing agricultural productivity itself at risk. To ensure food security and mitigate the negative impacts of agricultural systems, it is necessary to combine ecosystem services with different land uses through the adoption of sustainable agrosystems and coordinate actions to modify the landscape structure (Fischer et al., 2013; Montoya et al., 2020; van der Esch et al., 2017).

From an ecological perspective, landscapes refer to a mosaic of ecosystems at scales of hectares to many squares' kilometres (Turner & Gardner, 2015). The landscape also represents an interface between social and environmental processes (Turner, 1989), where the geographic space is organized and distributed through a socio-political decision-making process (IPBES, 2018). Nonetheless, most studies adopt distinct perspectives by isolating productive systems from natural areas, even knowing there is an interaction between anthropic and natural environments that influence numerous ecological phenomena, such as hydrological services, e.g., regulation of water flow and quality (Alvarenga et al., 2017; Bastos et al., 2021; Brogna et al., 2017; Ebling et al., 2021; Ferraz et al., 2013; Reichert, Junior, et al., 2021; Valente et al., 2021), carbon storage (Andriollo et al., 2017; Chazdon, 2008; Gibbs et al., 2010; Hanna et al., 2020; Reichert, Gubiani, et al., 2021), climate regulation, biogeochemical cycles, and agricultural productivity (IPBES, 2018; Valente et al., 2020; van der Esch et al., 2017).

Since agriculture is one of the economic sectors that most contributes to environmental degradation, with the conversion of natural areas for agricultural use being one of the main sources of loss of biodiversity and forest ecosystems (Andrade de Sá et al., 2013; Armenteras et al., 2017; Maranhão et al., 2019; Richards et al., 2012; Vandermeer & Perfecto, 2007). In tropical and subtropical regions, for example, agricultural activity is among a major driver of land use and land cover change and suppression of native vegetation (Bongaarts, 2019; Curtis et al., 2018; Lingner et al., 2020; Maranhão et al., 2019; Soterroni et al., 2018). Arable lands also cause disturbances such as habitat fragmentation/isolation and edge effect, alter the structure and composition of plant communities (Lewis et al., 2015; Lingner et al., 2020), and modify the spatial patterns of the landscape (Wästfelt, 2021).

Several studies use landscape metrics to quantify and qualify changes in composition, structure and distribution of spatial patterns in the landscape (Bellón et al., 2020; Ferreira et al., 2019; García-Llamas et al., 2018; Lausch et al., 2015; Lira et al., 2012; Marshall et al., 2020; Seganfredo et al., 2019; Taubert et al., 2018; Turner, 1989). Metrics are also used to describe the impacts of vegetation cover loss and of the fragmentation process on the distribution and behaviour of birds (Barbosa et al., 2017; Dotta et al., 2016), mammals (Bogoni et al., 2020; Delciellos et al., 2018), and arthropods (Gomez-Martinez et al., 2020; van Schalkwyk et al., 2020). Overall, metrics play a crucial role in exploratory and descriptive landscape analysis and monitoring and creating future use scenarios. However, most metrics are computed based on patch area, and distance and are not able *per se* to quantify the functional properties of the landscape, requiring the addition of new components to relate spatial patterns to ecological phenomena at relevant scales (Kupfer, 2012; Nowosad & Stepinski, 2018, 2019).

The concern with functional aspects of the landscape and debates about the relationship of biodiversity with landscape structure has gained importance in recent years, mainly by recognizing and incorporating farmland as a fundamental component in natural ecosystems conservation (Landis, 2017; Rusch et al., 2016). Among different theories, the intermediate landscape complexity hypothesis states that for an anthropized landscape, the effectiveness of biodiversity conservation management depends on landscape structure and landscape complexity, measured through the biodiversity rate (Tscharntke et al., 2012).

This hypothesis goes along with the land-sharing strategy, which advocates integration between agricultural production with biodiversity in optimizing the provision of ecosystem services (Grass et al., 2019; Phalan, 2018). The complementary land-sparing strategy determines that, for species conservation *in situ*, it is necessary to preserve large continuous areas to minimize the adverse effects of fragmentation (Grass et al., 2019). The land-sharing/land-sparing concept perceives the multifunctional aspects of the landscape, where it should be managed considering food production and the generation of environmental services.

Even in areas with a high degree of anthropization, different levels of land-sharing/land-sparing assist the conservation of natural ecosystems since this strategy favours connections between fragments and increases the efficiency of environmental services provided by agricultural areas (Bennett, 2017). However, the land-sharing approach presents some limitations regarding species conservation, especially for those with specialist habits or those demanding large areas (Phalan, 2018). In the context of the debate on 'single large or several small (SLOSS), Fahrig and Storch (2020) highlight that, in highly fragmented landscapes, the preservation of small and numerous areas are fundamental for species conservation. Furthermore, the authors observed that regions with high-level anthropogenic transformation with small and numerous fragments tend to contain more biodiversity than those with large and few fragments. This aspect reinforces the need to adopt different landscape management strategies to preserve natural resources.

The Native Vegetation Protection Law (Law n° 12,651) regulates land-use change on private lands in Brazil by setting limits for deforestation and economic exploitation of vegetation located within rural properties, which shapes the working landscape into a complex mosaic of natural and anthropized areas. The Brazilian regulations protect almost 193 ± 5 Mha of native vegetation (Soares-Filho et al., 2014), but little is known about these natural areas conservation status, distribution, and quality (Vacchiano et al., 2018). In fact, most Brazilian ecosystems are under increasing pressure. Studies have shown that the rate of ecosystem degradation has been growing alarmingly in recent years (Armenteras et al., 2017; Lewis et al., 2015), followed by growth in production and export of agricultural commodities. Since 1950, Brazilian agricultural exports have nearly quadrupled (FAO, 2020), while rural estates occupy approximately 775 million hectares, representing about 75% of the national territory (INCRA, 2018).

The lack of information on natural areas' quality and conservation status is even more critical for highly modified ecosystems, such as the Atlantic Forest biome. Intensely exploited since the Brazilian colonial period, the forest area today occupies about 12%, distributed in small (< 50 hectares) and numerous fragments throughout the national territory (Ribeiro et al., 2009). Although continuous efforts by different sectors of society have managed to reduce deforestation rates for the biome, it remains a biodiversity hotspot. It is heavily threatened by habitat loss and other human activities (Rezende et al., 2018). Furthermore, much of the work on the Atlantic Forest is concentrated in southeastern Brazil (Lima et al., 2015).

This study examines landscape configuration and structure changes in 20 years (1998-2008) in a region of Atlantic Forest, located in the Guaporé watershed, southern Brazil. We specifically aimed to answer: 1) How are spatial-temporal landscape patterns changing in the watershed? 2) What is the relationship between changes in land use and different landscape metrics? 3) How has agricultural activity influenced the change in landscape patterns and its relationship with natural areas?

2 METHODS

2.1 DESCRIPTION OF THE STUDY AREA

The study area is the Guaporé watershed (GRB), covering an area of 2,490 km², located in the state of Rio Grande do Sul, Brazil (Figure 1). The original vegetation of GRB is composed of two biomes highly threatened by anthropogenic activities: Atlantic Forest (≈37%) and Pampa (≈1%). Over 60% of the area is anthropized, comprising small-scale farmland, pastures, and scattered patches of urban infrastructure. The extent of anthropogenic activities poses a significant challenge for sustainable landscape management in the region, threatening natural resources and ecosystem services, including freshwater provisioning (Bastos et al., 2021) and productive agricultural soils (Ambus et al., 2018; Reichert et al., 2021; Tiecher et al., 2017).

The GRB is located in a humid subtropical climate (Cfa) and a Subtropical highland climate with constant rainfall (Cfb) with temperatures ranging from -3°C to 22°C and average annual rainfall between 1600 and 2200 mm, well distributed throughout the year (Alvares et al., 2013). From the geologic point of view, the study area belongs to the Serra Geral Formation, derived from volcanic lava flows (basalt and rhyolite on the top) and characterized by various facies (Caxias, Gramado, and Paranapanema). The watershed elevation ranges from ~50 m to ~900 m above sea level, bringing several slope gradients with it. The upper parts of the watershed are characterized by gentle slopes (6 - 9 %), whereas the lower parts of GRB are characterized by steep slopes (> 45%). The soils are predominantly distributed in five orders: Acrisols (*Argissolo*), Regosols (*Neossolo*), Ferralsols (*Latossolo*), Luvisols (*Luvissolo*), and Nitisols (*Nitossolo*).

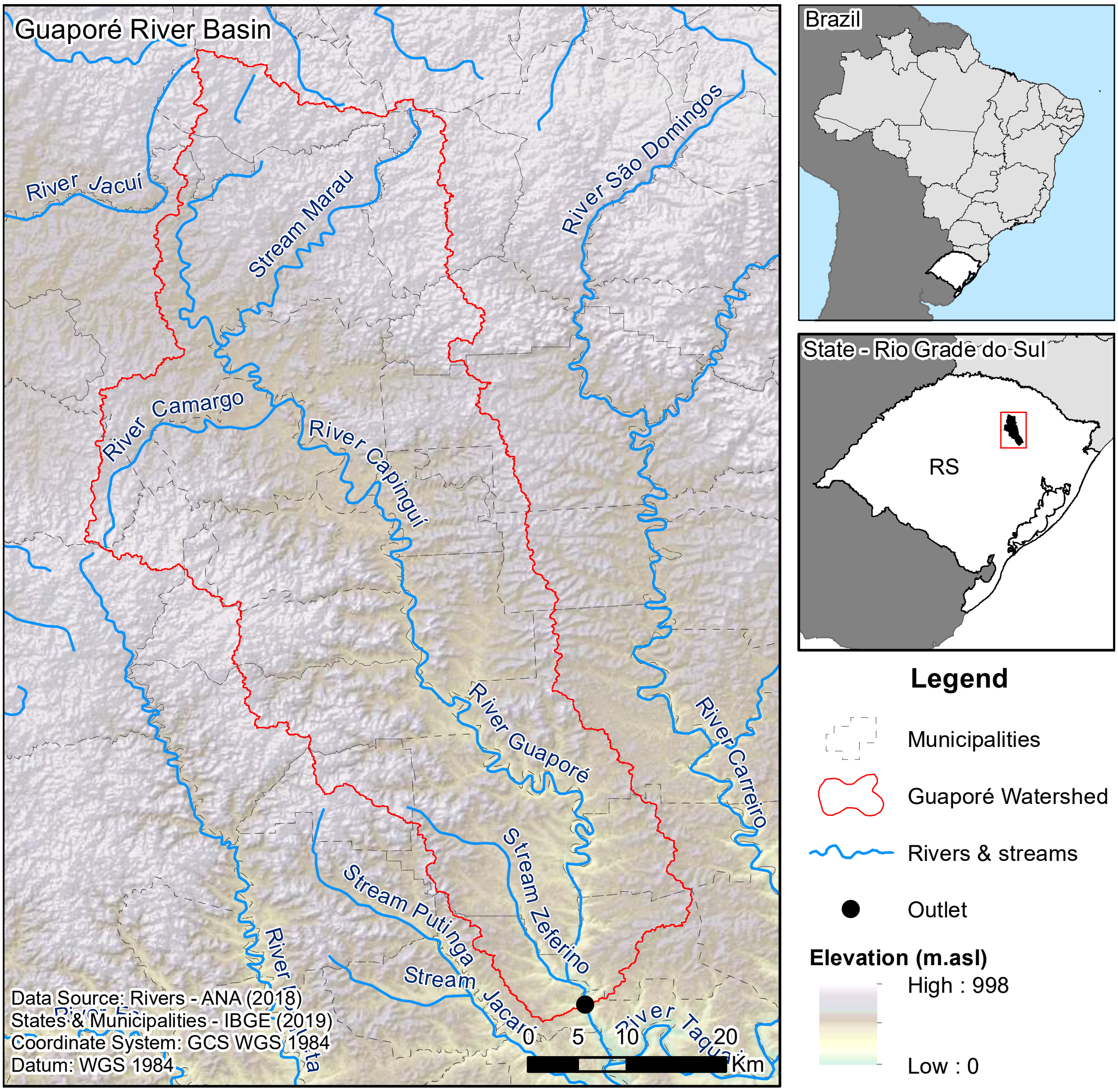

Figure 1: Location of the Guaporé watershed (GRB).
Source: The author.

2.2 DATA AND PROCESSING

To understand the dynamics of the GRB landscape, we used the annual land cover and land use (LULC) maps. The LULC maps covering 1998, 2008 and 2018 were collected from *Mapbiomas* collection 6 (data available at http://plataforma.brasil.mapbiomas.org; MapBiomas, 2021a). The *Mapbiomas* dataset is produced by a collaborative network of specialists in the Brazilian Annual Land Use and Land Cover Mapping Project scope. According to Souza et al. (2020), the *Mapbiomas* collection is derived from a 30 m resolution

Landsat imagery. The Landsat imagery is processed on the Google Earth Engine platform. First, a clean image is created by selecting the cloudless pixels; then, different metrics are extracted for each pixel of the seven satellite spectral bands. At the end of this process, each pixel carries up to 105 layers of information. Next, for each class of land cover/use an automatic classifier called "random forest" is applied to train and to classify samples targets obtained from reference maps. Then the temporal filter is applied at the pixel level to reduce inconsistencies due to changes in coverage and use that are impossible or not permitted. Finally, the maps of each class are integrated into a single map, which represents the coverage and land use for each year. Detailed information regarding the classification methodology is available in the ATBD (Algorithm Theoretical Basis Document; MapBiomas, 2021b).

2.3 LAND-COVER AND LAND-USE TRANSITION MATRIX

A transition matrix was generated to reflect the changes of land-cover and land-use among different time intervals (1998, 2008, and 2018) based on frequencies of occurrence and intersection of other land-use classes. After that, to facilitate a discussion of the results, Sankey diagrams were used to visualize land cover dynamics in the study area (Cuba, 2015). Sankey diagrams can be divided into different segments (lines), representing the transfer areas of land use/cover class, delimited by vertical divisions (stacked bars) representing the land use/cover classes in 1998, 2008 and 2018. The height of each component in the stacked bars is proportional to the amount of area occupied by land use/cover classes, while the thickness of each connecting line represents the magnitude of the changes. The analyses were conducted pixel by pixel-based with the aid of the raster (Hijmans et al., 2020) and networkD3 packages (Allaire et al., 2017) within the R environment (R Core Team, 2020).

2.4 LANDSCAPE COMPOSITION AND STRUCTURE

The interactions between spatial patterns and ecological processes in the Guaporé watershed were assessed by randomly distributing nine points along the drainage network using the Random points along line algorithm. Each point represents the geometric centre of a landscape sample unit (LSU), delimited from 100 km^2 (10,000 hectares) buffers. We then calculated, for each LSU, different landscape metrics (Table 1), chosen to represent size, shape, structure, diversity and clustering of the patches, and to explain the composition and spatial configuration of the landscape (Cushman et al., 2008; Jaeger, 2000; McGarigal & Marks, 1995; Metzger et al., 2009). The metrics and the equations are detailed in the FRAGSTATS software manual (McGarigal, 2014).

In addition to traditional metrics, to quantify, order, and classify landscape patterns, we calculated two landscape complexity metrics (marginal entropy - H(y) and Mutual information - U) derived from information theory (Nowosad & Stepinski, 2019). Together, H(y) and U allow 2D parameterization of landscape patterns, defined as the HYU diagram. First, for the data segmentation process, HYU points were extracted, and Euclidean distances were calculated by Ward's method, according to the methodology proposed by Nowosad and Stepinski (2019). Then, based on the Euclidean distance between points on the HYU diagram, the landscapes were clustered into three distinct groups as a measure of dissimilarity between patterns.

The allocation and delimitation of the LSUs were conducted in QGIS v. 3.120-București software, and the analyses of landscape metrics and landscape complexity were performed in the R environment, using the packages landscapemetrics (HESSELBARTH *et al.*, 2019) and stats (R Core Team, 2020).

Table 1: Metrics used for quantifying and qualifying landscape structure.

Landscape metrics	Abbreviation and range (unit)	Definition	Group
Total (Class) Area	CA > 0 (ha)	A measure of landscape composition. Higher CA values indicate the dominance of the matrix.	Area, density, and edge
Mean Patch Area	AREA_MN > 0 (ha)	Degree of fragmentation as a function of the number of fragments and total area occupied by a given class.	
Number of Patches	NP ≥ 1 (none)	A simple measure of the degree of division or fragmentation. Higher value quantifies greater fragmentation of the landscape, and lower values indicate union or extinction of fragments of the same class.	
Percentage of Landscape	0 < PLAND ≤ 100 (%)	Proportional abundance of each patch type in the landscape. The interpretation of PLAND is the same as described for CA but expressed as a percentage.	

Landscape metrics	Abbreviation and range (unit)	Definition	Group
Largest Patch Index	0 < LPI ≤ 100 (%)	Percentage of total landscape area comprised by the largest patch.	
Total Edge	TE ≥ 0 (ha)	Sum of all the edges of the landscape.	
Edge Density	ED ≥ 0 (m ha^{-1})	TE divided by the total area in hectares.	
Patch Density	PD > 0 (number per 100 ha)	The number of fragments of the class in 100 hectares of the landscape. The interpretation of PD is the same as described for NP.	
Mean Patch Shape Index	SHAPE_MN ≥ 1 (none)	A smaller value indicates a simple fragment shape, which is beneficial for conservation.	Shape
Total Core Area	TCA ≥ 0 (ha)	Sum of the core areas of the entire class in hectares.	Core area
Number of Disjunct Core Areas	NDCA ≥ 0 (none)	A number of disjoint core areas contained in the fragments.	
Core Area Percentage of Landscape	0 < CPLAND ≤ 100 (%)	Proportional abundance of the core areas of the entire class.	
Aggregation Index	AI ≥ 0 (none)	Equals 0 when no fragments of the same class and increases as the as the landscape becomes more aggregated. Equals 100 when the landscape is composed of a single fragment.	Proximity and isolation
Interspersion & Juxtaposition Index	0 < IJI <= 100 (none)	The observed interspersion over the maximum possible interspersion for the given number of patch types.	
Euclidean Nearest-Neighbor Distance	ENN_MN ≥ 0 (m)	Distance to the nearest neighboring patch of the same type. based on shortest edge-to-edge distance	
Cohesion Index	0 < COHESION < 100 (%)	Indicator of physical connectedness of the corresponding patch type.	Contagion and dispersion
Splitting Index	1 ≤ SPLIT ≤ number of cells in the landscape area squared	Number of patches one gets when dividing the total landscape into patches of equal size in such a way that this new configuration leads to the same degree of landscape division as obtained for the observed cumulative area distribution.	

3 RESULTS

The distribution analysis of the different land cover and land use classes over the last 20 years in the Guaporé watershed is illustrated in Figure 2. Low representativeness of the urban infrastructure class (less than 1%) highlights the predominance of rural activity throughout the entire extent of the GRB. Land use classes Agriculture, Pastures, and Mosaic of Agriculture and Pastures represent about 60% of the total area, while the forest formation covers approximately 40% of the landscape. The different uses are distributed differently throughout the watershed: in the upper portion, we observed the predominance of agricultural lands; in the middle third, the landscape is dominated by a mosaic of agriculture and pasture; and in the lower third, the forest component predominates, probably due to the relief - undulated to strongly undulated, which disfavors the implementation of agricultural and ranching activities (Lingner et al., 2020).

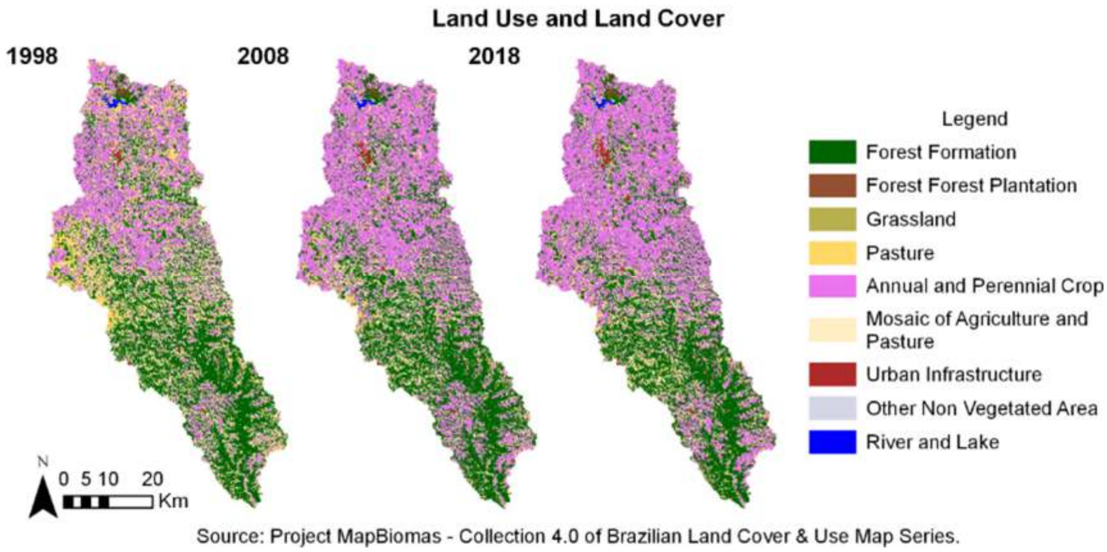

Figure 2: Maps of the historical series (1998, 2008 and 2018) of land cover and land use of the Guaporé watershed, produced from mosaics of Landsat images, with spatial resolution of 30 m, and accessed directly from the *MAPBIOMAS* Project (2019). Source: The author.

Changes in land use and land cover - LULC, computed based on pixel frequency for each land use and land cover class (Figure 3), reinforce the importance of agricultural and livestock activities in the region. In 20 years, the watershed maintained its rural character, with the expansion of agricultural areas on the expenses of forest component reduction. About 10.5 thousand hectares of forest were converted to other uses in the studied period. The loss of total forest cover represents a reduction of 6% in relation to 1998. Although proportionally small, this reduction can affect several ecosystem services and functions besides favouring forest degradation.

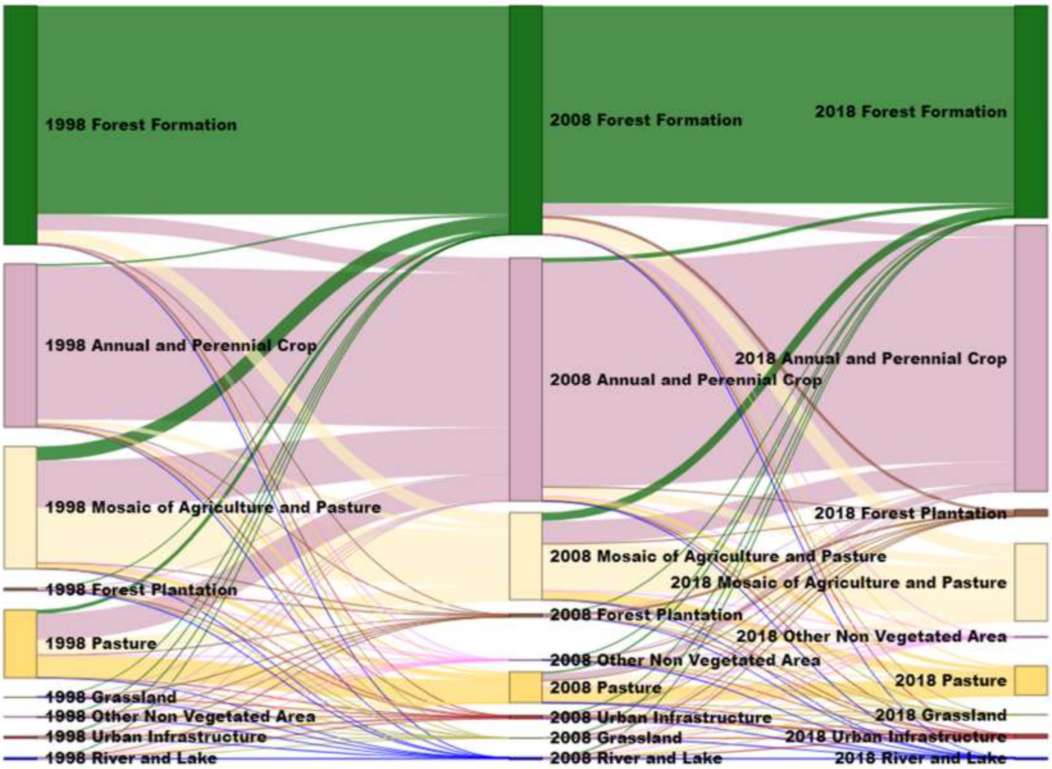

Figure 3: Sankey diagram with the proportion of the different land use and land cover classes - LULC, constructed from transition matrices, based on the frequency of occurrence and crossover of the classes in the different time intervals. Source: The author.

The landscape sample units (LSU, Figure 4) highlight the patterns distribution of the different LULC classes in space and time and the interactions between spatial patterns and ecological processes. The LSUs confirms the trend of forest cover reduction throughout the watershed. In 1998, the minimum area occupied by forest formation was 1,797 hectares (LSU - B), while the maximum was 6,831 hectares (LSU - I), representing about 20% to 70% of the landscape, respectively. In 2018, the values ranged from 1.586 to 6.323 hectares (LSU - B and LSU - H, respectively), representing about 15% to 65% of the landscape (Figure 5 - CA|PLAND metric). Among the different LSUs, the E had the largest vegetation suppression, about 1,000 hectares, while F had the smallest (≈ 60 hectares).

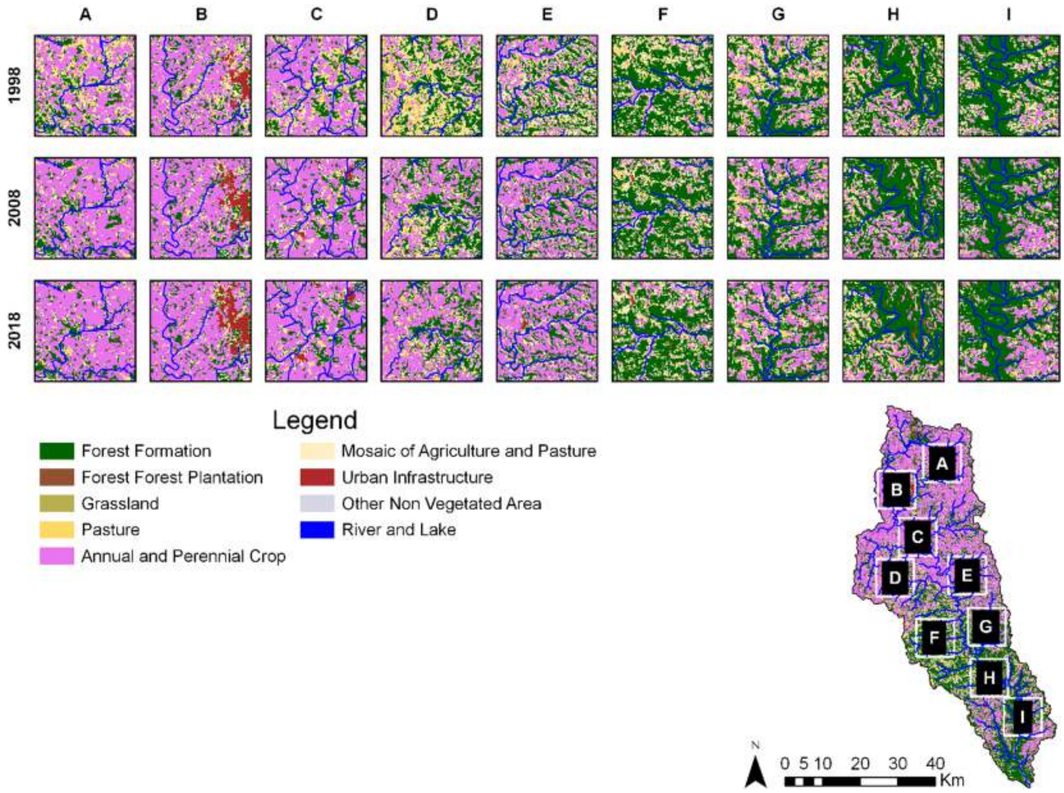

Figure 4: Distribution and land cover and land use (LULC) of Landscape Sample Units (PSU), randomly delineated from 100 km² buffers, along the Guaporé watershed drainage network (GRB). Source: The author.

Although a reduction in forest cover is observed, the number and average area of fragments, and the percentage area of the largest fragment numerically differ little over time (Figure 5 np|area_mn|lpi), an indication that loss of forest cover does not necessarily incur in the creation of new fragments. We also observe that landscapes with lower forest cover (PLAND < 50%), such as LSU - A to LSU - E, which are located in the upper and middle portion of the GRB, have a greater number of fragments (296 to 338), lower average area (5 to 9 hectares) and lower LPI value (1. 2% to 8.6%). By contrast, in landscapes with predominant forest cover (PLAND > 50%), such as LSUs F-I, the number of fragments ranged from 83 to 176, the average area from 25 to 76 hectares, and the LPI from 22% to 60%.

Forest fragments, although numerous, have an average distance between them of 90 to 144 meters, being smaller in areas of low anthropic action, which also influences the connectivity index (80 < AI > 95). Besides, the fragments are relatively well distributed in the landscape (44% < iji > 65%), with the exception of LSU - F, where they are found more isolated (iji = 33%). These results indicate the occurrence of forest pseudo-continuity and structural connectivity of the landscape.

In addition to connectivity, the size of the core area and the shape of the fragments is essential for landscape ecology. The fragmentation process exposes forest remnants to the edge effect, affecting the internal quality of the fragments. Over time we observed a reduction in the total area of edges (TE) and the density of edges (ED) in seven of the nine LSUs (Figure 5). In principle, although the reduction of edges has a positive denotation, this process was followed by a decrease in the total central area (TCA) and the number of disjunct central areas (NDCA), reducing the sizes of preserved habitat within the fragments and indicating the process of fragment extinction. Regarding the shape of the fragments, the constancy of the average shape index metric means no significant changes occurred over time. In general, the forest fragments have complex shapes (1.47 < SHAPE_MN > 1.74), close to a circumference, which minimizes the edge/area ratio, consequently having a smaller edge area.

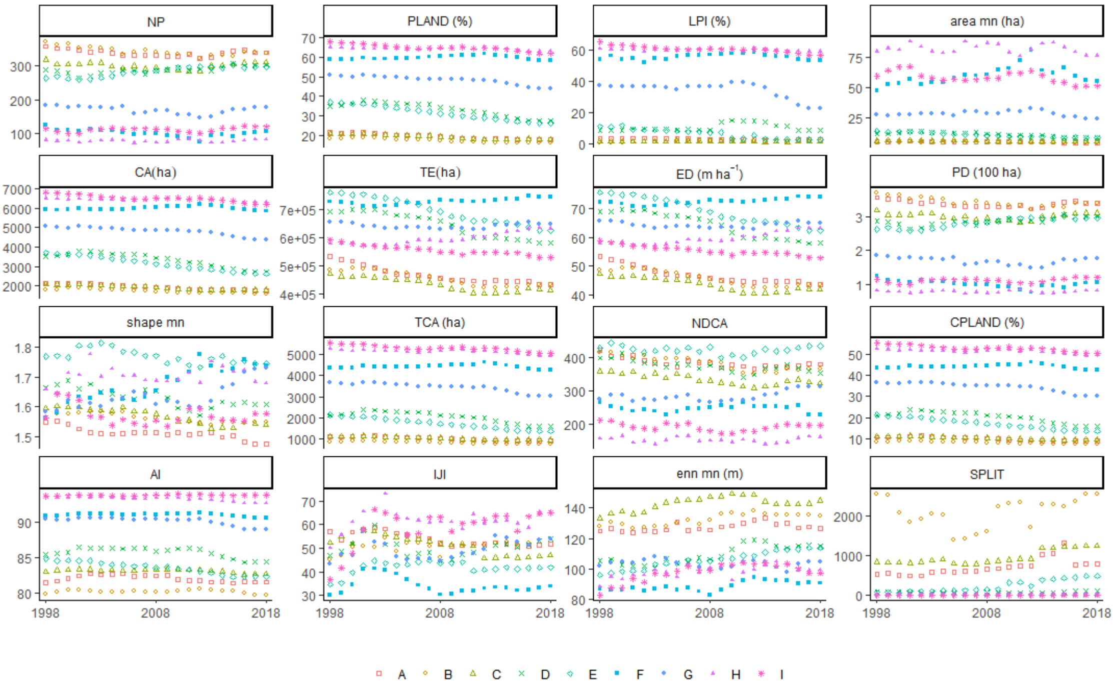

Figure 5: Landscape metrics calculated to represent the size, shape, structure, diversity and clustering of the fragments in the different LSUs. Source: The author.

Changes observed over time in the landscape structure due to changes in size, shape, and distribution of fragments affect the configuration of landscape patterns. Figure 6 shows the dynamism of LULC throughout the Guaporé watershed and illustrates the homogenization process of anthropic environments over time (x-axis - Marginal entropy) and the reduction in their complexity (y-axis - Mutual information). In areas once dominated by forest cover (F, H, and I), an increase in LULC class diversity and landscape complexity occurred, indicative of disturbance processes.

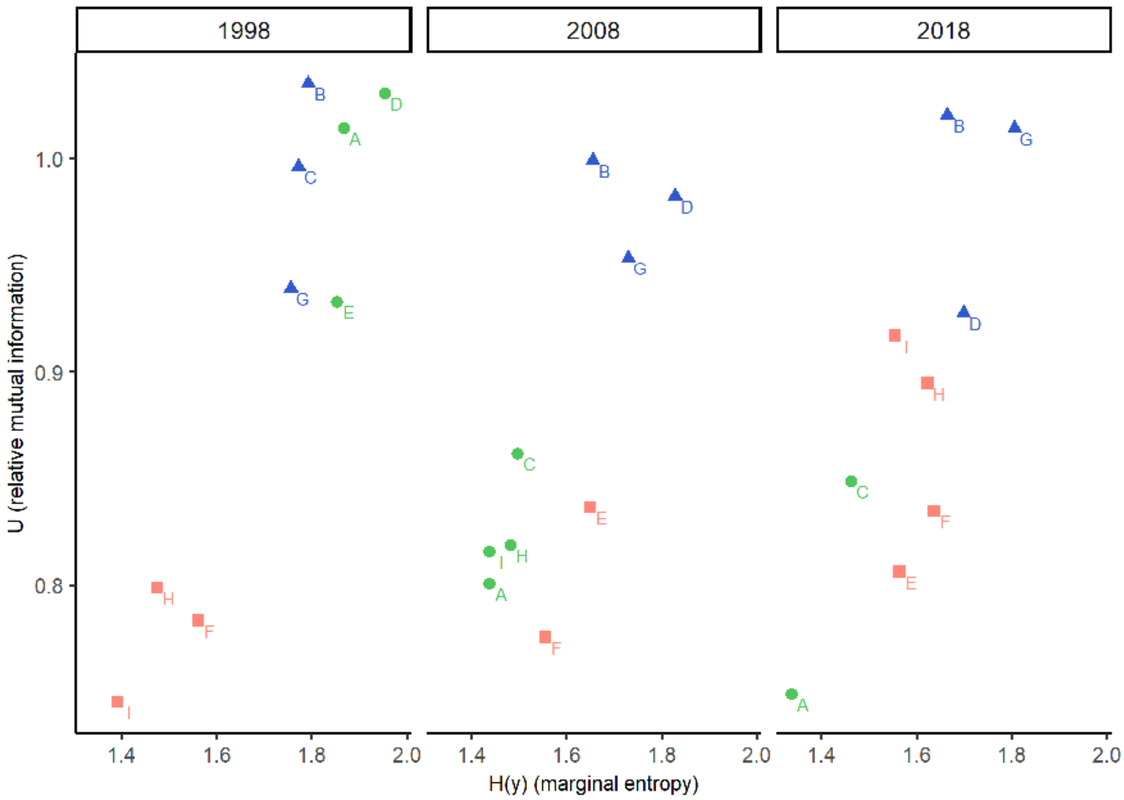

Figure 6: Organization of landscape patterns based on H (y) and U metrics and hierarchical grouping of LSUs into three different classes (represented by the same colour and symbol), calculated from Euclidean distances using Ward's method. Source: The author.

4 DISCUSSION

In rural landscapes, expansion and intensification of the production system are constantly associated with the loss and degradation of forest ecosystems (Andrade de Sá et al., 2013; Armenteras et al., 2017; Maranhão et al., 2019; Richards et al., 2012; Vandermeer & Perfecto, 2007). Changes and dynamics of LULC in space and time are complex processes occurring in response to different social, economic, political, and environmental factors (Barbier et al., 2010; Mather, 1992; Meyfroidt & Lambin, 2011). In the last 20 years, the landscape of the GRB has undergone intense modification, mainly in areas destined for productive activities, with the conversion of the mosaic of agriculture and pasture into permanent crops. Meanwhile, areas with forest cover experienced disturbance cycles, compensated in part by regeneration events (Figure 3). This transition, from pasture and croplands mosaic to permanent crops, can be partially attributed to the increased global demand for agricultural commodities, mainly soybean (Cattelan & Dall'Agnol, 2018; Garrett et al., 2013; Maranhão et al., 2019), at the same time that national policies to protect the Amazon region - Soy Moratorium (Cattelan & Dall'Agnol, 2018; Gibbs et al., 2015; Lima et al., 2019), and expanded use restriction and protection of the Atlantic Forest biome (Rezende et al., 2018; Waroux et al., 2019) were implemented.

Loss of the ability of ecosystems to perform their functions and provide natural services is among the main consequences of changes in LULC, especially when human landscapes replace natural ecosystems. According to Arroyo-Rodríguez et al. (2020), anthropized landscapes should maintain at least 40% forest cover to ensure fauna permanence and provide goods and services. Conversely, areas dominated by an anthropogenic matrix and with less than 40% forest cover, similar to this study, compromise climate regulation, water provision and quality, crop pollination and pest control (Diaz et al., 2006; Grass et al., 2019), and maintenance of wildlife by imposing barriers to the movement of the most diverse groups of dispersers and pollinators (Barbosa et al., 2020; Boesing et al., 2018); and limiting food availability, consequently reducing species richness and diversity (Bogoni et al., 2020).

The transition of uses in the anthropic matrix can intensify natural environments' degradation processes since replacing productive systems also implies changes in soil management. Inappropriate use and management are among the leading causes of accelerated degradation of agricultural land worldwide, often promoting erosion, compaction, and loss of organic matter (FAO, 2015). Recent studies conducted in the GRB demonstrated that intensification of land use and the adoption of inappropriate agricultural practices are responsible for the accelerated process of soil loss and sediment deposition along with the drainage network (Minella et al., 2014; Tiecher et al., 2017). Moreover, the absence of buffer zones due to the reduction of forest cover, such as riparian forests, facilitates the transport of pollutants and sediments through surface runoff, especially during rainfall events, compromising the quality of water in shallow springs (Bastos et al., 2021; Kaiser et al., 2010; Kaiser et al., 2015). Thus, extensive forest cover, especially near the drainage network, and associated with the diversity of uses in the landscape can reduce the connectivity of crops with waterways, mitigating adverse effects of anthropic action on water resources (Alvarenga et al., 2017).

Changes in landscape structure occur distinctly across the watershed, as evidenced by the LSUs. Over the past 20 years, the rates of forest cover loss in the different LSUs were numerically similar, indicating a homogeneous process of forest cover loss throughout the watershed. However, some LSUs are more susceptible to land-use change, as exemplified by units A - E, dominated by an extensive anthropic matrix. A possible explanation for the rates of land-use change in these areas is the landform conformation. Landscape relief conditions are among the main local factors influencing LULC distribution patterns (Lingner et al., 2020; Rezende et al., 2015). LULCMeanwhile, the units F - I contain a continuum of natural vegetation due to natural barriers imposed by the undulating to strongly undulating relief.

Landscape structure metrics also highlighted the relationship between the number and size of fragments with the LULC distribution. In the studied landscapes, there is a predominance of small fragments (< 50 ha), especially in the LSUs dominated by an anthropic matrix, where fragments are smaller (< 5 ha) and numerous (> 300). Similar patterns are observed at the biome scale. In the Atlantic Forest, the biome in which GRB is embedded, landscapes are highly fragmented, with a predominance of small and numerous forest remnants that correspond for about 83.5% of the native vegetation cover (Ribeiro et al., 2009). Forest remnants, even though small, are used by several species to move in the landscape (Barbosa et al., 2017; Ferrante et al., 2017), besides propagating forest species, thus favouring regeneration and restoration (Niemeyer et al., 2020). Therefore, these fragments are fundamental in the conservation and preservation of native vegetation and the maintenance of ecosystem services. However, small fragments are highly susceptible to disturbance and anthropic exploitation and are often extinct from the landscape, as observed in this study. This process reduces the total forest cover area, increases isolation, and reduces connectivity in the landscape (Ferreira et al., 2018).

The LSUs and landscape metrics also highlight the dynamics of different levels of landscape complexity, i.e., the existence of a structural gradient in the watershed landscape that varies over time. Hence, various landscape conservation and management strategies should be used since the effectiveness of local biodiversity conservation management in anthropized landscapes changes with landscape structure (Tscharntke et al., 2012). Another point of emphasis is reducing the complexity of highly anthropized environments in time (LSU A and C), mainly characterized by their low landscape diversity and highly homogeneous structures (Figure 6). Within the logic of the intermediate landscape complexity hypothesis, these areas have low responsiveness to management, compromising the provision and maintenance of ecosystem services and agricultural productivity.

Several studies have shown the need to maintain natural areas within rural landscapes (Arroyo-Rodríguez et al., 2020). Although 53% of Brazilian native vegetation occurs within private lands (Soares-Filho et al., 2014), in this study we observed the existence of areas with little diversity and small forest fragments scattered in the landscape, while others have a high rate of forest cover, demonstrating that native vegetation is unevenly distributed in the landscape.

To reconcile agricultural production with the conservation of natural ecosystems within the GBR landscape and to increase the quality of rural landscapes, different landscape management strategies should be adopted by observing the multifunctional aspects of the landscape. For instance, land-sharing/land-sparing strategies (Brussaard et al., 2010) can be applied to search for more sustainable agriculture in the watershed. The characteristics of the landscapes studied herein show that, although anthropized and fragmented, there is a certain degree of connectivity among the natural areas that favour the adoption of the land-sharing strategy, while in landscapes with a high proportion of natural areas, the land-sparing approach, destining these large areas, probably unsuitable for agriculture or ranching, for conservation.

As highlighted earlier, the organization and configuration of the landscape is part of the socio-political decision-making process (IPBES, 2018; van der Esch et al., 2017). Although the Brazilian legal system determines zones for the protection of native vegetation along the banks of water bodies and the reservation of 20 to 80% of the land within the biome, known as a legal reserve for the preservation of natural areas, which would facilitate the adoption of the mentioned landscape management strategies, the size and allocation of these reserves vary between each Brazilian biome.

5 CONCLUSION

In summary, we observed that, although the Guaporé landscape has preserved its forest remnants in recent years, the expansion and intensification of agriculture has altered the landscape throughout the watershed, simplifying systems and reducing their diversity. These processes are similar to those occurring all over the Atlantic Forest biome and deserve attention, especially because the watershed contains significant forest remnants. The relevance of this forest continuum also stands out for its locality; there are few studies on this biome, considered a global hotspot in the southern region of Brazil and its state of conservation.

Although rural landscapes are constantly associated with the loss and fragmentation of forest ecosystems, the Guaporé watershed has experienced cycles of disturbance, compensated in part by regeneration events of natural vegetation. Even though there was a loss of native vegetation in the period analysed, this did not necessarily imply the creation of new fragments since the number of fragments decreased or remained stable over time. Although the reduction of this index is positive, when associated with the loss of native vegetation, it reveals a process of extinction of small fragments, which can compromise landscape connectivity and the provision of ecosystem services.

The anthropic matrix, represented here by the different agricultural uses, has contributed mainly to landscape dynamics and altered spatial patterns. In general, with an increased degree of anthropization, the fragments of native vegetation are more numerous and smaller in size. Landscape configuration also varies with changes of use within the anthropic matrix. Intensive use and inadequate soil management have negatively affected the water resources of the Guaporé watershed, compromising not only water supply but also food security due to soil degradation, pesticide contamination, nutrient leaching, and reduction of natural areas.

The existence of a spatial-temporal gradient of the landscape, evidenced by the LSU, with a concentration of environments with a higher degree of anthropization in the north of the watershed and those of a lower degree in the south, near the outlet, reinforces the importance of a systemic view of the watershed since modifications in the landscape structure in upstream areas compromise natural ecosystems downstream.

Different management strategies are needed to increase the quality of rural landscapes. Landscape management through land-sharing/land-sparing is a viable strategy to promote more sustainable agriculture in the watershed, reconciling natural areas with agricultural development. Although theories such as 'single large or several small' advocate for the preservation of large areas, the dynamics of landscape structure in the Guaporé watershed, as a whole, makes the adoption of management models based on this premise unfeasible, mainly because of the degree of landscape fragmentation and the dynamics of different agricultural uses.

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CHAPTER II: FUTURE TRAJECTORIES OF NATURAL ECOSYSTEMS: DRIVING FORCES OF LAND-USE CHANGES IN A SUBTROPICAL AGRICULTURAL RIVER BASIN

ABSTRACT

The changes in land use and land cover (LULC) are driven by multiple factors and link directly to ecosystem services and biodiversity. In Brazil, yet the Native Vegetation Protection law regulates the land-use change on private lands, natural ecosystems are still under high conversion pressure. Understanding the dynamics of LULC is critical for maintaining a broad range of ecosystem services, especially in landscapes experiencing intense land transformations. Here, we investigated the relative importance of multiple land-use drivers to predict the recent and future trends of natural ecosystem loss in a region of Atlantic Forest, located in the Guaporé watershed, southern Brazil. To assess trends in the land-use change, we applied time and spatial autocorrelation analysis, while the Dinamica EGO cellular automata model evaluated the importance of multiple land-use drivers. The results suggest a widespread change in natural ecosystems, highlighting extensive deforestation in the North of the watershed and a positive increment of natural areas in the South. Therefore, drivers of land-use changes operate distinctly in the studied watershed. Biophysical characteristics, such as flat reliefs landscape orientation, has a similar scale of influence overall watershed, while socioeconomic factors help explain changes in the local scale. The ongoing process of LULC in the watershed may lead to further deforestation and simplification of a natural ecosystem. If current trends continue in the future (2030 and 2050), we expect a replacement of older, mature forests by younger and less biodiverse forests, which may impact nature`s contribution to people.

Keywords: Atlantic Forest; disturbance; ecosystem services; Anthropogenic actions; Cellular automata; Future land use and land cover change

1 INTRODUCTION

Land use and land cover (LULC) change are one of the main driving forces of Global Change. Over the past 60 years, about a third of the global land surface has experienced an event of land-use change (Winkler et al., 2021). The scale and extent of LULC changes have mainly been affected by biophysical, socio-economic, and demographic factors (Foley, 2017). Although LULC may contribute to the improvement of countries' economies (Kumar et al., 2013), these changes have substantial effects on climate, ecosystem stability, water balance, agricultural yield, socioeconomic practices, and biodiversity (Ellis & Ramankutty, 2008; Foley et al., 2011; Garibaldi et al., 2020; Reichert, Junior, et al., 2021). Thus, analysing and understanding how human activities would change the territory is vital to developing land-use planning strategies and mitigating future impacts of climate change (Bongaarts, 2019).

Deforestation for pasture and agriculture expansion, infrastructure development, cities, and political and financial incentives to land occupation are the main drivers of LULC change in Brazil (Gibbs et al., 2015; Maranhão et al., 2019; Parente et al., 2019; Winkler et al., 2021). Currently, most of all remaining native vegetation in the country (53%) is located in private rural estates rather than Protected Areas (Soares-Filho et al., 2014). Although the Native Vegetation Protection law (Law n° 12,651/2012) regulates conservation on rural private estates, establishing areas of permanent protection (APPs) along water streams and on hilltops as well as legal reserves (native vegetation in a section of the property), farmers are still failing to comply with environmental policies for deforesting APPs or to conserve their minimum legal reserve areas (Rajao et al., 2020; Rezende et al., 2018).

Among the Brazilian biomes, the Atlantic Forest is the most threatened by intense historical and ongoing LULC changes, which has already removed 84-89% of its natural vegetation (Joly et al., 2014; Souza et al., 2020). The remaining Atlantic Forest is mostly comprised of small (<50 ha) isolated, disturbed patches (Ribeiro et al., 2009). However, only 30% of the total vegetation cover is located inside protected areas (IUCN Categories I-VI). In contrast, the remaining 70% of vegetation cover are within private areas, where intervention and deforestation may be allowed by law (Rezende et al., 2018). Due to the importance of private lands, current land cover (LULC) changes are affected mainly by the dynamics of agro-pastoral land uses' expansion and retraction (Rosa et al., 2021).

Recent improvements in geospatial techniques, such as remote sensing (RS) and Geographic Information Systems (GIS), increased our ability to map land use land cover and to analyse the spatiotemporal distribution throughout the world (Amani et al., 2019; Chasmer et al., 2020; De Bem et al., 2020; Souza et al., 2020). Further, the availability of GIS data and spatial statistics has stimulated the development of a myriad of spatially-explicit models, coupling different mathematical and statistical approaches like artificial neural networks, cellular automata, agent-based models, or multiple regressions (Liu et al., 2008; Noszczyk, 2018). Among those approaches, cellular automata (CA) have been one of the most widely used because of their capability to simulate and predict the spatial-temporal evolution of complex geographical phenomena, such as urban growth, forest dynamics, landscape changes, and land-use changes (Leite-Filho et al., 2021).

The CA models consist of a regular n-dimensional array of cells that interact within a certain vicinity and according to a set of transition rules (Soares-Filho et al., 2002) represented by many forms. However, the determination of transition rules poses an additional challenge to derive the CA model structure and related parameters. Furthermore, the definition of transition rules involves experts' knowledge and their individual preferences, which can potentially affect the model structure of CA (Noszczyk, 2018).

Therefore, CA models' calibration is crucial to achieving realistic simulation. Yet, despite the existence of many techniques, the most common one is based on the "trial and error" approach, followed by a visual test to validate the model (Oliveira et al., 2019; Soares-Filho et al., 2002). In turn, in recent years, many techniques have been proposed to derive optimal parameter values according to the best fit between the observed data and various simulated results (Clarke & Gaydos, 1998; Liu et al., 2008). As a result, most CA models consider multiple factors to automatically derive parameter values, such as weighting matrices, logistic regression, and Support Vector Machines.

Our objective was to investigate the relative importance of multiple land-use drivers to explain the recent trends of natural ecosystem loss in a region of Atlantic Forest, located in the Guaporé watershed, southern Brazil. Specifically, i) we assessed the recent trend of deforestation to the watershed, and within municipality boundaries, ii) evaluated the influence of multiple land-use drivers on the current LULC trend, and iii) modelled and projected future scenarios of land use based on the current trends (Business as Usual). Our hypotheses are as follows: a) the historical spatiotemporal changes on native forest cover are directly associated with anthropogenic land uses drivers in the region, b) the current level of compliance with

Brazilian legislation negatively affect the natural ecosystems, and c) the primary vegetation cover it will decrease over time.

2 MATERIALS AND METHODS

2.1 DESCRIPTION OF THE STUDY AREA

See Chapter I, Section 2.1 DESCRIPTION of the study area for a summary of Guaporé watershed characteristics.

2.2 DATA SOURCE AND PROCESSING

2.2.1 Annual land cover/use maps

This study examines trends of deforestation and land-use changes in GRB by using annual land cover and land use (LULC) maps from 1998 to 2019, while changes within the natural ecosystem were represented by the land-use map from 1985. The LULC maps were collected from *Mapbiomas* collection 6 (data available at http://plataforma.brasil.mapbiomas.org; MapBiomas, 2021a). For more information about *Mapbiomes* dataset, see Chapter I, Section 2.2 DATA and processing.

2.2.2 Reclassification of LULC maps

A standard generalised classification scheme based on two classes was defined: Altered and Natural ecosystems. The altered ecosystem corresponds to the anthropic use class of the year. In contrast, the natural ecosystem was subdivided into primary vegetation to represent the natural ecosystem existing before 1985 and secondary vegetation to describe the natural ecosystem existing in the previous years. These classes aim to encompass changes in anthropic areas and within the natural areas, at the same time, to evaluate the susceptibility of primary and secondary vegetation to changes over the years. In addition, the available LULC maps were reclassified based on their cover and use. A detailed table with the correspondence class between LULC and reclassified LULC is provided in Table 1.

Table 1: *Mapbiomas* collections classes reduced (reclassified) in two groups.

Generalised classification	***Mapbiomas* collections**
Natural ecosystem	Forest Formation
	Non-forest Natural Formation
	Grassland
	Rocky Outcrop
	Other Non-forest Formations
Altered ecosystem	Forest Plantation
	Pasture
	Agriculture
	Mosaic of Agriculture and Pasture
	Non-vegetated area
	Urban Infrastructure

2.3 DEFORESTATION SPATIAL PATTERNS ANALYSIS

Changes in the spatial patterns of the natural ecosystem over 21 years timespan (1998 to 2019) within GRB municipalities were assessed through time trend and spatial autocorrelation analysis. First, the trends of cold spot or hot spot for deforestation were identified using the Mann-Kendall test (τ) at the significance of p values ≤ 0.05 and $|\tau| \geq 0.60$. Then, we tested the spatial dependence of the data by using spatial autocorrelation techniques, such as the Global Moran Index and the Getis-Ord Gi* analysis in R software (R Core Team, 2020) using the packages raster (Hijmans et al., 2021) and stats (R Core Team, 2020).

2.3.1 Assessing prediction variables

Both natural and anthropogenic drivers (factors) associated with land-use changes were identified from the literature review (González-González et al., 2021; Molin et al., 2017; Nascimento et al., 2019; Rezende et al., 2015) and selected based on data availability. The selected variables were grouped into three general categories: (i) biophysical, (ii) anthropic, and (iii) demographic factors. First, the data were obtained from official government databases at the municipality level (DEEDADOS, FEPAM, INPE, Sicar and SIDRA; Table 2). Then, all data were converted to spatial raster data based on the political boundaries of the municipality. Additionally, ArcGIS was used to standardize with the exact spatial resolution (30 x 30 meters) and coordinate system (WGS84) to guarantee that all raster files contain the same number of columns and rows.

Table 2: Drivers of land-use change.

Category	Variable retained	Source	Scale/ Resolution	Year
Demographic factor	Population density	Demography/SIDRA	municipality	2010
Anthropic factors	Agricultural practices	Agricultural Census/SIDRA	municipality	2019
	Vegetation extraction (ton)	Production of Vegetable Extraction and Forestry/SIDRA	municipality	2019
	Silviculture (ton)	Production of Vegetable Extraction and Forestry/SIDRA	municipality	2019
	Agricultural Production (Area planted/ harvested yield ton)	Municipal Agricultural Production/SIDRA	municipality	2019
	Conservation Practice	Municipal Agricultural Production/SIDRA	municipality	2019
	Livestock (livestock units)	Municipal Livestock Survey/SIDRA	municipality	2019
	Socioeconomic development Index (IDESE)	DEEDADOS	municipality	2019
	Farmland Structure	Sicar	1:50.000	2018
	Soil uses	Sicar	1:50.000	2018
Biophysical factors	Hydrography	FEPAM	1:25.000	2013
	Soil*	FEPAM	1:25.000	2018
	Digital elevation model (DEM)	INPE/TOPO DATA	30,30 m	2011
	Slope[1]	INPE/TOPO DATA	30,30 m	2011
	Aspect[2]	INPE/TOPO DATA	30,30 m	2011

[1,2] Slope and Aspect were derived from the Digital Elevation Model (DEM) obtained from the INPE/TOPO DATA database.

The anthropic and demographic factors were weighted by the area of the municipalities and grouped into different classes. The socioeconomic factors: vegetation extraction, silviculture, agricultural production and livestock were divided into six categories of productivity, based on the distance of each value in relation to mean ($\bar{x}$) (High $x > 1.1\bar{x}$, Medium High $x > 1.05\bar{x}$, Medium $x > 0.95\bar{x}$, Medium Low $x > 0.9\bar{x}$ and Low $x < 0.9\bar{x}$). Where the means were the sum of each socioeconomic factor divided by the number of municipalities within the GRB. The agricultural practice was classified according to the percentage of farmlands adopting conservative practices (up to 95% - High, from 95% to 75% - Medium, and lesser 75% - Low). While farmland structure was described based on the number of fiscal modules following the Brazilian regulations (<1 smallholding, 1>4 small, 4>15 medium, and > 15 large properties), and the soil use class were named based on Sicar's vector files: Production area - AP, Legal Reserve - LR and Areas of Permanent Protection – APP. To the socioeconomic development Index - IDESE, the values were divided into three classes (Low, Medium and Hight), while Demographic density was divided into five categories (<1, 1>10, 10>25, 25>100, >100).

Biophysical were used either as continuous variables, such as the Digital Elevation Model (DEM) and the hydrography vector, which this last one had the perennial watercourses filtered and its density calculated from the ArcGIS line density function; or as categorical variables represented by slope classes, none (0-3%), Gentle (3-8%), Moderate (8-20%), Steep (20-45%), Extremely steep (45-75%), Excessively steep (>75%), Aspect (N, NE, E, SE, S, SW, W, NW), and Soil groups: Acrisols (*Argissolo*), Regosols (*Neossolo*), Ferralsols (*Latossolo*), Luvisols (*Luvissolo*) and Nitisols (*Nitossolo*).

2.4 MODELING CHANGES ON NATURAL ECOSYSTEMS

Spatial modelling of the landscape was made based on historical changes in land cover/use assessed between 1998 and 2019 by using the software Dinamica EGO (Oliveira et al., 2019. Then, we assembled all predict variables into a single multilayer raster file using the cube algorithm. Moreover, due to the role of the *spatially explicit variables* and to improve the model's efficiency, the watershed was divided into two sub-regions (North and South). Then, each sub-region was analyzed following different steps represented as a separate model to keep simplicity (Figure 7).

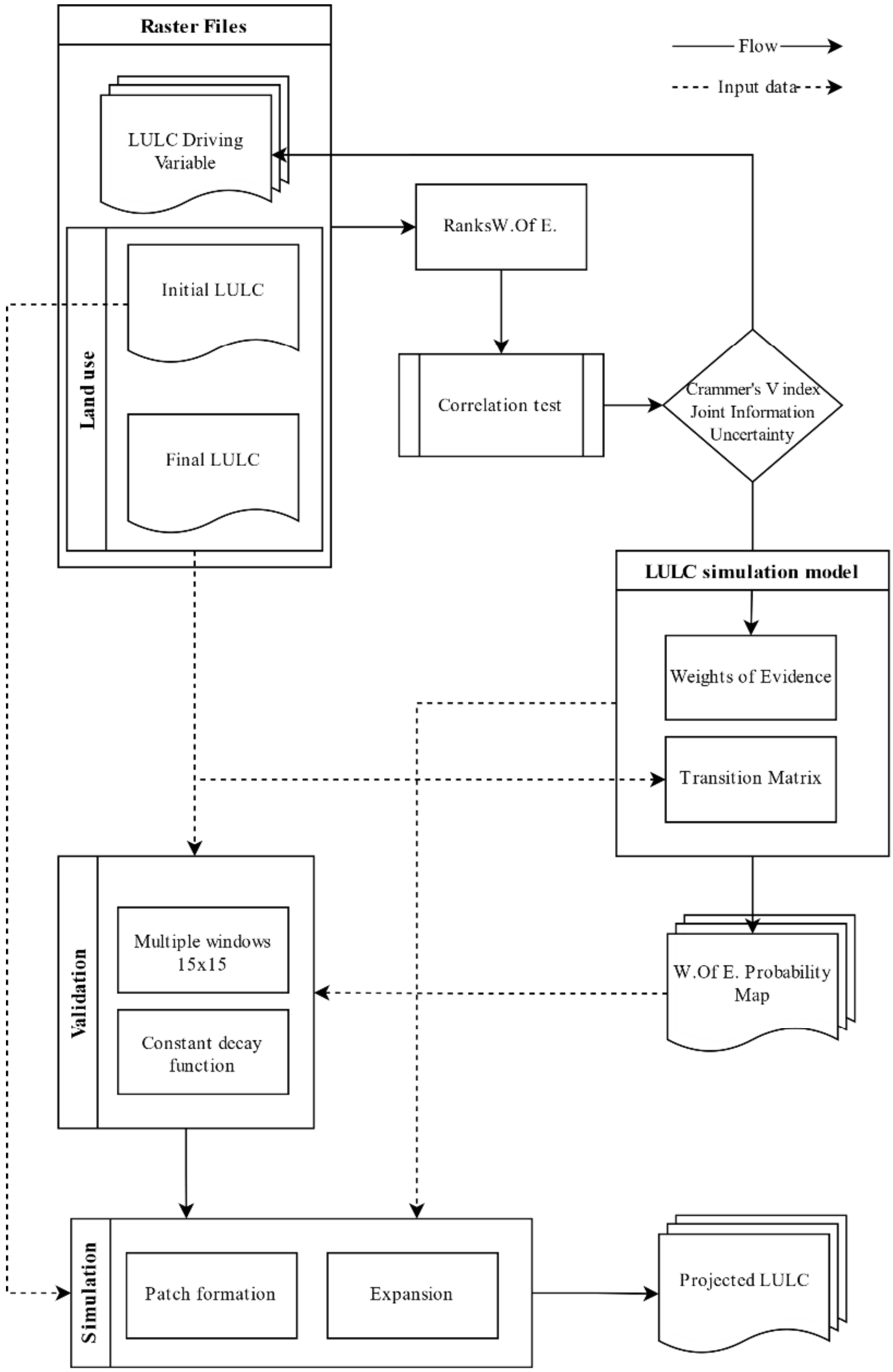

Figure 7: Modelling procedures workflow. LULC - Land cover and land use; W. of E. - Weights of Evidence. Source: The author.

2.5 TRANSITIONS AND EXPLORATORY ANALYSIS OF THE DATA

The annual transition rates from altered ecosystems to primary and secondary vegetation, and vice-versa, between 1998 ($T_{initial}$) and 2019 (T_{final}) were quantified through the Markovian method by using the Calc Net Transition Rates tool. To estimate the influence of each driver on the spatial probability of an i-j transition, we used weights of evidence (Agterberg et al., 1990; Goodacre et al., 1993) and logistic regressions. The weights coefficients were estimated for the categorical predictors, while the logistic regression coefficients were estimated for the continuous ones. The weights and logistic regression coefficients indicate the effect of drivers on changes in primary and secondary vegetation. Positive weights/coefficients promote a loss on natural ecosystems, negative ones reduce it, and close to zero values do not affect. To evaluate the assumption that variables are spatially independent, correlation analysis between the predictor variables were performed. The variables were tested using the Crammer's V index and Joint Information Uncertainty (JIU). According to Bonham-Carter (1994), values of V and JIU when greater than 0.5 indicate a greater probability of spatial dependence. Therefore, one of the variables must be eliminated or combined, ensuring that no redundancy is made in the model.

2.6 PROJECTION AND VALIDATION OF CHANGES ON NATURAL ECOSYSTEMS

Based on the current trend of the drivers (Business as usual), the projection of land-use changes was carried out to estimate the areas of the natural and altered ecosystems in the years 2030 and 2050. The validation of the models was conducted by applying an exponential decay function and employing multiple window similarity analysis, with windows sizes ranging from 1x1 to 15x15 pixels (Almeida et al., 2008). While the simulation was conducted using the trial-and-error method by varying the parameters of the transition functions (Mean Patch Size, Patch Size Variance, and Patch isometry) until the model approximated more closely to the structure of the final historical landscape (Soares-Filho et al., 2002). Spatial analyses, modelling and simulations were performed with Dinamica EGO freeware (Oliveira et al., 2019; Rodrigues & Soares-Filho, 2018) and R.

3 RESULTS

3.1 AN OVERALL PORTRAYER OF THE NATURAL ECOSYSTEM IN 2019

We estimated that over 105 thousand hectares (Kha), or 37% of the Guaporé Watershed, was covered by the natural ecosystem in 2019. Despite the large extent of native vegetation, most of these areas occur on private lands and are not under the protection of Brazilian regulation. Based on the landowner self-declaration, available in the Brazilian Rural Environmental Registry (CAR), we mapped approximately 46 Kha of natural cover within legal protection lands (LRs and APPs) in GRB.

Natural ecosystems under legal protection and legally available for conversion have different rates of change. Overall, mostly ≈ 82% of the landscape available for conversion was comprised of old-growth forests (pre-1985). In contrast, nearly 40% of the legally protected ecosystems in the Guaporé watershed have been altered by anthropogenic activities, even though the APP and LR have long been recognised as essential for biodiversity conservation and their legal status limit the use (Brancalion et al., 2016; Brock et al., 2021). Furthermore, most of the disturbed areas are within the legal reserves and in APPs of small streams (≤10 meters), which may endanger water quantity and quality (Bastos et al., 2021; Tiecher et al., 2017).

3.2 CHANGES IN THE SPATIAL PATTERNS OF LAND COVER/USE

The spatial patterns on the transition rates of primary and secondary vegetation to altered ecosystems between 1998 to 2019 are spatially correlated and present two distinct trends, which allowed to group the municipalities and divide the watershed into two regions – North and South (Figure 8). In the North, primary and secondary vegetation have been converted in altered ecosystems at 1.77% and 2.22% per year, respectively. In the South, although changes from natural areas to altered ecosystems had occurred, we also noted a process of agricultural land abandonment, highlighted by the changes rates of the altered ecosystem into secondary vegetation (1.18% per year). The regeneration process in abandoned areas reduced the overall trend of natural vegetation loss. It suggests differences in the factors, at least at the regional scale, that cause changes in nature and anthropogenic assets along the watershed.

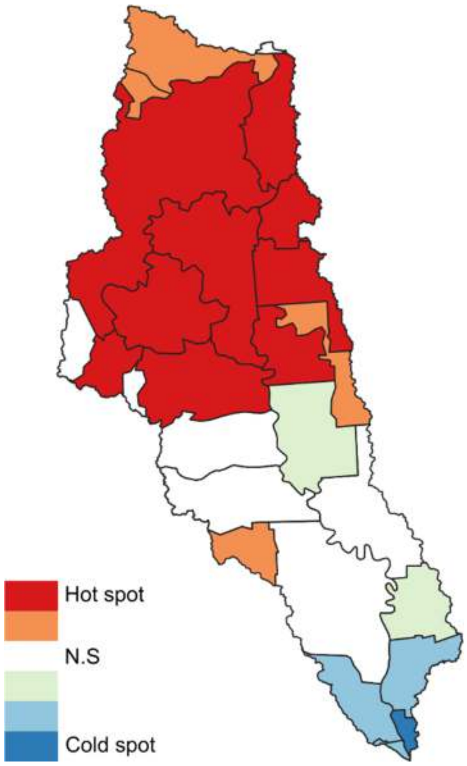

Figure 8: Heat map of spatial patterns on the transition rates of primary and secondary vegetation to altered ecosystems by GRB municipality using the hot spot analysis (Getis-Ord Gi*). Red colours represent negative changes in natural areas cover, while blue colours represent positive changes. N.S: not significant. Source: The author.

Changes in land use are driven by natural and anthropogenic factors (IPBES, 2018). The biophysical, socioeconomic, and demographic drivers vary along the watershed. Consequently, they play different roles in each region, as evidenced by the cells' values of transition probabilities (primary and secondary vegetation to altered ecosystem), obtained through the weight’s weight coefficients (Appendix B).

Overall, the biophysical factors have similar trends in both regions: areas near the previously altered ecosystem, above 400 m a.s.l, with low density of streams, gentle slopes (0-8%), and north-facing are potential targets for anthropic disturbance. Such landscapes features can influence the accessibility to a natural ecosystem and consequently the vulnerability to conversion into different land use. In addition, drivers of land-use change such as conservation practice, vegetation extraction and livestock have significance on land-use change at the local level. In the South, the lack of conservation practice and low productivity of the vegetation extraction increase the probability of change in land use over the studied years. In the North, despite the significance of the livestock density on observed land-use change, the values of weights of evidence do not show a clear trend for the analysed period.

From the magnitude of the weights of evidence, it is evident that farmers defy deforestation because lands managed to exploit the native vegetable resources such as wood, latex, seeds, fibres, fruits, and roots. Likewise, APP and LR show a negative association with primary and secondary vegetation deforestation.

3.3 MODELLING FUTURE TRAJECTORIES OF NATURAL ECOSYSTEMS

Observed and predicted maps between 1998 and 2019 were similar, ranging from 34% to 44% at the pixel level (0.09 ha) and increases to 85% - 96% at 15 x 15-pixel window (1.35 ha). Altogether, the validation process made it possible to project the primary and secondary forest cover to the future using the historical trend as part of the simulation model. The similarity values are satisfactory for model validation. However, the difference between the real and the simulated map shows a slight overestimation of the land uses for primary and secondary vegetation.

The Business as Usual (BAU) scenario was simulated from the 2019 land use map corresponding to 2030, 2040, and 2050 land use maps. The futures scenarios predict a loss of nearly 10% of the natural ecosystem between 1998 and 2050 due to the land cover/use trend. In the North, around 159 km^2 natural areas will be lost by 2050, while the South will experience a reduction of 5% (40 km^2) of their extent in 1998.

Overall, the anthropogenic land use will increase in the whole watershed as the primary forest, and the secondary forest is substituted by anthropic areas (Figure 9). However, part of the primary vegetation is expected to be converted into secondary forest. Most of the land-use changes will occur in the North region, increasing human disturbance and impacting the natural ecosystem.

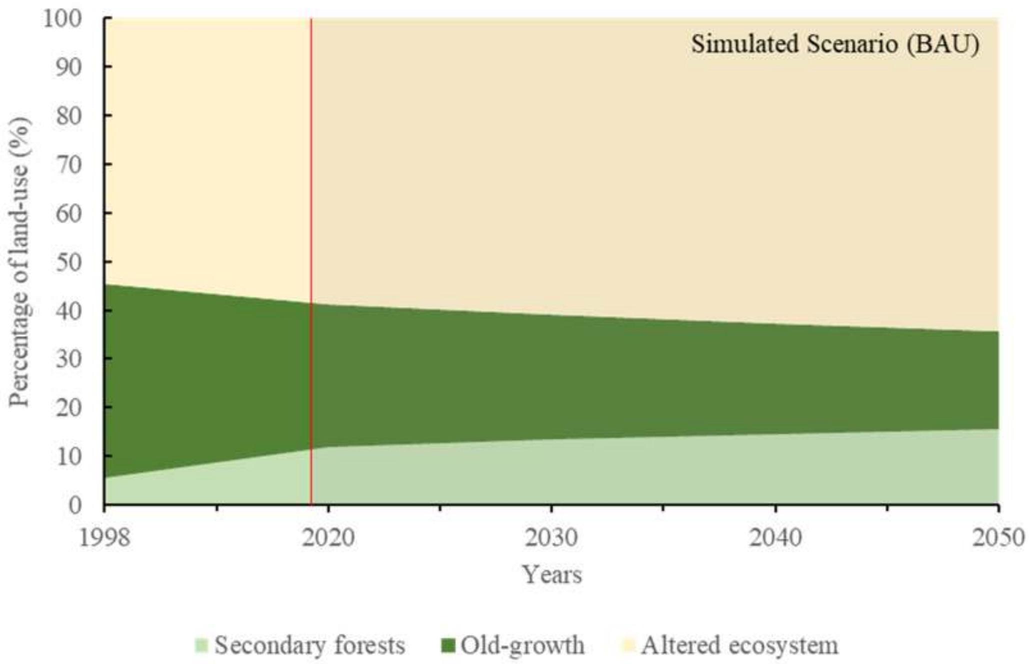

Figure 9: Percentage of land-use composition from 1998 to 2019 based on *Mapbiomas* land use maps (left side of red line). Right side of red line the percentage of land-use composition for simulated Business as Usual (BAU) scenarios (from 2020 to 2050). Source: The author.

4 DISCUSSION

Guaporé land used 2019 map reveals that most of the natural ecosystems in private properties are legally available for conversion to other uses. Similar results are observed throughout the Brazilian territory. Although the Native Vegetation Protection Law restrict deforestation by protecting 193 ± 5 Mha of natural areas within LRs and APPs, Soares-Filho et al. (2014) estimated that almost 88 ± 6 Mha of Brazil's native vegetation could be legally converted to anthropogenic use. Nonetheless, most of the landowners are in non-compliance with legal rules (Soares-Filho et al., 2014; Sparovek et al., 2010). Even traditional agriculture areas fail to comply with environmental regulations. Similar to our study, Sparovek et al. (2010) reported a widespread illegal land use of LRs and APPs throughout the country, including the southeast regions. This fact can be partially explained by partial enforcement of the environmental regulation, such as the Native Vegetation Protection Law and Atlantic Forest Law (Brock et al., 2021; Soares-Filho et al., 2014; Soterroni et al., 2018); and Brazil’s inability to tackle illegal deforestation (Rajao et al., 2020).

The level of compliance to Native Vegetation Protection Law can impact nature`s contribution to people, such as food provision, water flow regulation, and water quality. The APPs were created to protect environmentally fragile areas such as steep slopes, water springs, riparian zones, and mountaintops. Therefore, changes in land use in these areas jeopardize the ability of nature to support human well-being. In our study, even considering the landowner self-declaration, we identified an environmental debt in APPs areas, mainly in those related to small streams. Riparian zones are important to control soil erosion, to protect streams against the input of sediments and nutrients loading (Guidotti et al., 2020; Hanna et al., 2020; Monteiro et al., 2016; Valera et al., 2019; Vieira et al., 2018).

While anthropogenic land use, such as agriculture, are commonly reported as a pervasive threat to water quality and biodiversity (Johnson et al., 2021). Furthermore, several authors had already spotlighted the relationship between water quality and agricultural land use in watersheds of the southern Atlantic biome (Bastos et al., 2021; Bonuma et al., 2014; Didoné et al., 2014; Kaiser et al., 2010; Kaiser et al., 2015; Tiecher et al., 2017). The latest work in the watershed associated the diminishment of water quality with the precarious rural sanitation conditions and agricultural activity (Bastos et al., 2021). The conversion of the natural ecosystem, mainly in RLs and APPs areas, and water pollution faced by GRB altogether increase the concerns about the capability of nature to continue to provide water-related services, mainly due to the importance of the watershed to supply water for the Porto Alegre city, capital of Rio Grande do Sul.

4.1 DRIVERS OF LAND-USE CHANGE

According to our results, the North area of the watershed was a hots pot for negative change in primary and secondary vegetation, while a cold spot had been identified in the South due to the expansion of secondary forest. Although the differences in trends of vegetation cover justify the division into two regions, other local drivers must be considered to understand the spatial land use pattern. This study observed that land-use changes induced by biophysical factors operate at a similar scale of influence in both regions, even though the watershed has complex geography.

Biophysical characteristics, such as relief, distance to human settlement, altitude, aspect correlate with land-use change and natural ecosystem degradation (Calaboni et al., 2018; Molin et al., 2017). In general, natural ecosystem conservation and secondary forests' expansion were concentrated in lands with low suitability for agriculture (Lingner et al., 2020; Silva et al., 2016), such as areas with steep slopes or poor soils. In contrast, altered ecosystem expansion occurred mainly in flat to smooth-wavy terrains. These areas are more stable environments, resulting in favourable areas to the occupation and agricultural production due to better soil physical proprieties and water retention when compared to concave areas (Lemos et al., 2021; Reichert et al., 2016). Additionally, the transport and mechanization of crop fields are facilitated by flat terrains (Rezende et al., 2015), increasing the pressure to convert the natural ecosystem into an altered ecosystem. Likewise, slope, distance to an altered ecosystem, altitude, and aspect were stronger predictors of land-use change. Areas near pre-existing settlements and in low altitudes are preferred to further agricultural expansion due to the accessibility condition (Mendoza-Ponce et al., 2021). In contrast, areas considered less productive, such as the southern, southeast, and southwest slopes (an indicator of solar radiation), are generally left to conserve natural vegetation (Lemos et al., 2021).

Although all biophysical factors described in this study showed correlation with land-use changes, they are not able to fully explain land conversion without considering multiple factors interaction (Geist & Lambin, 2002; van Vliet et al., 2016). Thus, prior studies adopted a conceptual model that combines a variety of factors to address the complexity and identify trade-offs of land-use change processes while providing a better understanding of human-environmental systems (van Vliet et al., 2016). Based on Geist and Lambin's (2002) framework, followed by a literature review, we observed that the biophysical factors were enhanced by the effects of proximate causes and underlying driving forces. In a broader context, the increase of international trade triggered unprecedented changes in the Brazilian agricultural sector (Cattelan & Dall'agnol, 2018; Winkler et al., 2021), including consolidated agriculture areas, such as the southern regions. While at local scales, the expansion of mechanized agriculture, agriculture intensification, land tenure, and smallholder farming increase the pressure on natural ecosystems (Maranhão et al., 2019; Song et al., 2021).

In contrast to biophysical factors, which explain the depletion of a natural ecosystem at a regional scale, the pathways of land-use change are linked to different socioeconomic factors. As previously mentioned, the altered ecosystem in the South is predicated based on the farmers' decisions to adopt conservation practices, whereas vegetation extraction promotes the conservation of natural areas. However, such simplifications of the causes of land-use change led to a miss concept of cause–consequence (Lambin et al., 2001). The real driving forces behind socioeconomic factors are different responses to economic opportunities and environmental awareness (Prokopy et al., 2019).

In the last decades, the world soybean demand and profitability has increased (Cattelan & Dall'agnol, 2018; Song et al., 2021). In response to this economic opportunity, the Brazilian agricultural sector moved towards export-oriented commodity farms (Garrett et al., 2013). Even in regions with the scarcity of land for grain crop expansion, such as southern Brazil, soybean planted areas are increasing due to a large-scale replacement of cropland or pasture by soybean fields (Flexor & Leite, 2017; Lapola et al., 2013). The expansion of soybean also increased farmers' positive environmental attitudes. Compared to other crops, soybean is commonly cultivated during the summer under a no-till system followed by winter crops (Wade et al., 2015). The no-till system increases soil organic carbon and available water capacity, reduces soil compaction, improves soil aggregate size and stability, and increases porosity (Ambus et al., 2018; Lozano et al., 2016; Peres da Rosa et al., 2021; Reichert et al., 2018). By improving soil health, conservation practices, such as no-till, reduce land degradation, resulting in land being spared for nature (Villoria et al., 2014).

In contrast to the soybean fields, farmers tend to have negative environmental attitudes in areas where the economy is based on tobacco crops. Tobacco cultivation is considered the most environmentally destructive agricultural practice, mainly in areas with few rural developments and low socioeconomic (Leppan et al., 2014; WHO, 2017). Evidence suggests that tobacco growing contributes to the reduction of native forest cover, depletion of soil and soil fertility, soil erosion, and water contamination (Bastos et al., 2021; Becker et al., 2009; Bonuma et al., 2014; Kaiser et al., 2015; Reichert, Gubiani, et al., 2021; Thomaz & Antoneli, 2021). By harming the environment, tobacco growing also contribute to increasing poverty. Tobacco companies have encouraged families to grow tobacco by providing all the necessary inputs, financial support, and technical assistance (Reichert, Gubiani, et al., 2021). However, smallholder tobacco farmers generally earn barely enough to cover their costs. The result is that many small-scale farmers fall into debt to the tobacco companies due to the unfair contract system (Leppan et al., 2014). The impoverishment of these families could facilitate environmental depletion since poor farmers tend to overuse the available natural resources (Masron & Subramaniam, 2018).

An alternative to improve environmental quality while reducing the rural community's dependency on cash crops is to encourage the development of sustainable production systems. In our study area, many small family farms generate their income from integrated production systems with various food crops, such as corn, oats, wheat, and dairy cattle. Forests are also an important source of income, mainly in occupancy areas of *Ilex paraguariensis* St. Hil. (Erva-mate), a native species used as a tea, found in the interior Brazilian Atlantic forest (Eibl et al., 2017). Traditional erva-mate systems have a wide diversity of forest species that ensure nutrient cycling (Ilany et al., 2010) and is often cultivated without chemical inputs (Chaimsohn & Radomski, 2016). Altogether, the cultural importance of erva-mate associated with the sustainable use of natural resources by farmers provides important information about the capacity of the community to recognize the value of conserving the forest remnants (Christo et al., 2012).

4.2 A FUTURE TRAJECTORY OF LAND USE

The future simulations of land-use change suggest that anthropogenic activities such as agriculture, livestock, and pasture, will drive most of the changes in natural ecosystems. The model's predictions proved to be efficient when compared with observed data yet distributing the anthropic cells over the years in a more realistic scenario is a challenging task since the influential factors oscillate over time (De Brito et al., 2021), while the spatial resolution affects the accuracy of land use simulation (Rodrigues & Soares-Filho, 2018). In general, coarse resolution cannot correctly depict the landscape patterns at local scales, but it can still capture the overall possibilities of the future (Cheng et al., 2020). Further, it is essential to note that the process to assess the impacts of direct and indirect drivers on natural ecosystems poses several challenges due to the limited capability of the model to represent complex realities (Resende et al., 2020).

Considering the BAU scenario, the rates of natural ecosystem change, even though still higher for the Atlantic Forest biome, are within a similar range observed by other authors (e.g., Molin et al., 2017). Further, it is worthy of highlighting that the forest dynamics presented in this study comply with the literature. Different studies showed a widespread stable condition or regrowth of natural areas in the Brazilian Atlantic Forest (Brancalion et al., 2019; Ferreira et al., 2019; Rezende et al., 2018). Despite the current depicted trends indicating a pathway towards a sustainable future scenario, careful analysis showed a hidden process of replacing older, mature forests with younger and less biodiverse forests. Similar results were mentioned by Rosa et al. (2021), which identified an ongoing reduction of older native forest cover and the continuous increase of younger native forest cover (<10 years old).

Furthermore, even though Atlantic Forest has specific legislation to protect all forest patches of intermediate and late-successional stages (>10 years old) from deforestation (Silva et al., 2018), similar to the observed in our study, other authors reported conflicts between use, mainly areas under permanent preservation, and Legal Reserve (e.g., Patrício et al., 2019). These results indicate the GRB might go through the forest transition process, where the decrease of deforestation rates over the last years was not followed by a similar increment of newly vegetated areas. At a local scale, change in the landscape could enhance in the long term the effects of climate change and the capability of the land to generate promising environmental and socioeconomic benefits in the future.

5 CONCLUSION

The main objective of this research was to quantify and qualify the driver of land-use change and create a predictive model enable to assess LULC in future scenarios. The results showed a widespread change in the natural ecosystem in the Guaporé watershed, whereas the maps reveal the importance of private land for the conservation of natural resources. Furthermore, most natural ecosystems were within private properties, which increased the concern about the environmental quality due to partial enforcement of the environmental regulation and ongoing ecosystem degradation.

Drivers of land-use change operate distinctly in the studied watershed, leading to more extensive deforestation in the North and a positive increment of natural areas in the South. Despite the complex geography of the watershed, the biophysical characteristics operate at a similar scale of influence in both regions. In contrast, pathways of land-use change in each area will depend on different socioeconomic factors.

Although the biophysical and socioeconomic factors helped to predict land-use change in this study, a simplification based on the cause-consequence should be avoided. Land transformation is a complex process with multiple factors interaction. Thus, to fully understand the process of land-use change, we must also consider the effects of proximate causes and underlying driving forces.

Finally, our models show how the ongoing land-use change in the watershed may lead to further deforestation and simplification of a natural ecosystem. Based on the results, it is expected a replacement of older, mature forests with younger and less biodiverse forests, which may impact nature`s contribution to people.

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APPENDIX - A

Table 3: The result of Mann–Kendall trend test for change in natural ecosystem per municipality in Guaporé watershed, Rio Grande do Sul from 1998 to 2019.

Municipality	Z-Value	Sen's slope	S	Var(S)	P-value	Tau	Result
Água Santa	-3.15	-2	-91	815	0.002	-0.53	No trend
Anta Gorda	-0.28	-9.17	-9	817	0.78	-0.05	No trend
Arvorezinha	2.73	168.25	79	817	0.006	0.46	No trend
Camargo	-5.46	-728.75	-157	817	<0.0001	-0.92	Trend exists
Casca	-5.95	-449.4	-171	817	<0.0001	-1	Trend exists
Dois Lajeados	-5.46	-374	-157	817	<0.0001	-0.92	Trend exists
Doutor Ricardo	3.99	169.5	115	817	<0.0001	0.67	Trend exists
Encantado	4.2	18.1	121	817	<0.0001	0.71	Trend exists
Gentil	-5.32	-174.6	-153	817	<0.0001	-0.89	Trend exists
Guaporé	-4.2	-222	-121	817	<0.0001	-0.71	Trend exists
Ibirapuitã	-3.08	-27.5	-89	817	0.002	-0.52	No trend
Ilópolis	-3.57	-52.67	-103	817	<0.0001	-0.6	Trend exists
Itapuca	-1.75	-207.2	-51	817	0.08	-0.3	No trend
Marau	-5.81	-955.5	-167	817	<0.0001	-0.98	Trend exists
Mato Castelhano	-4.34	-175.5	-125	817	<0.0001	-0.73	Trend exists
Montauri	-5.88	-481.67	-169	817	<0.0001	-0.99	Trend exists
Muçum	-5.46	-33.71	-157	817	<0.0001	-0.92	Trend exists
Nova Alvorada	-5.67	-910.44	-163	817	<0.0001	-0.95	Trend exists
Passo Fundo	-4.38	-21.64	-126	816	<0.0001	-0.74	Trend exists
Santo Antônio do	-5.88	-270.67	-169	817	<0.0001	-0.99	Trend exists
Serafina Corrêa	-5.88	-280.47	-169	817	<0.0001	-0.99	Trend exists
Soledade	-5.74	-251.22	-165	817	<0.0001	-0.96	Trend exists
União da Serra	-5.53	-382.67	-159	817	<0.0001	-0.93	Trend exists
Vespasiano Corre	-5.95	-318.5	-171	817	<0.0001	-1	Trend exists
Vila Maria	-5.67	-556.93	-163	817	<0.0001	-0.95	Trend exists

*p value < 0.05 indicates rejection of the null hypothesis of no trend and thus, revealing the existence of the trend

APPENDIX - B

Table 4: Weights of evidence ranges and coefficients for the North region.

Transition	Variable	Range Lower Limit	Weight Coefficient
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	0	0.769
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	30	-0.111
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	60	-0.562
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	90	-0.853
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	300	-1.851
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	330	-2.684
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	360	-3.956
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	420	-16.892
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	0	0.489
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	30	0.149
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	60	-0.049
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	120	-0.147
Primary vegetation to Altered ecosystem	Aspect	N	0.131
Primary vegetation to Altered ecosystem	Aspect	NE	0.091
Primary vegetation to Altered ecosystem	Aspect	E	-0.046
Primary vegetation to Altered ecosystem	Aspect	SE	-0.135
Primary vegetation to Altered ecosystem	Aspect	S	-0.169
Primary vegetation to Altered ecosystem	Aspect	SW	-0.059
Primary vegetation to Altered ecosystem	Aspect	W	0.118
Primary vegetation to Altered ecosystem	Aspect	NW	0.165
Primary vegetation to Altered ecosystem	DEM	0	-0.452
Primary vegetation to Altered ecosystem	DEM	400	-0.059
Primary vegetation to Altered ecosystem	DEM	500	-0.247
Primary vegetation to Altered ecosystem	DEM	600	0.063

Transition	Variable	Range Lower Limit	Weight Coefficient
Primary vegetation to Altered ecosystem	DEM	700	0.366
Primary vegetation to Altered ecosystem	DEM	800	-0.008
Primary vegetation to Altered ecosystem	Hydrography	0	0.541
Primary vegetation to Altered ecosystem	Hydrography	300	0.171
Primary vegetation to Altered ecosystem	Hydrography	400	-0.032
Primary vegetation to Altered ecosystem	Hydrography	500	-0.003
Primary vegetation to Altered ecosystem	Farmland Structure	Smallholding	0.035
Primary vegetation to Altered ecosystem	Farmland Structure	Small properties	0.06
Primary vegetation to Altered ecosystem	Farmland Structure	Medium properties	-0.248
Primary vegetation to Altered ecosystem	Farmland Structure	Large properties	-1.105
Primary vegetation to Altered ecosystem	Livestock	High	-0.087
Primary vegetation to Altered ecosystem	Livestock	Medium-High	0.372
Primary vegetation to Altered ecosystem	Livestock	Low	0.061
Primary vegetation to Altered ecosystem	Agricultural Production	High	-0.02
Primary vegetation to Altered ecosystem	Agricultural Production	Medium-High	1.093
Primary vegetation to Altered ecosystem	Agricultural Production	Low	0.017
Primary vegetation to Altered ecosystem	Slope	none	0.299
Primary vegetation to Altered ecosystem	Slope	Gentle	0.302
Primary vegetation to Altered ecosystem	Slope	Moderate	0.35
Primary vegetation to Altered ecosystem	Soil	Ferralsols	-0.035
Primary vegetation to Altered ecosystem	Soil	Acrisols (PVAd)	0.296
Primary vegetation to Altered ecosystem	Soil	Nitisols	-0.208
Primary vegetation to Altered ecosystem	Soil	Luvisols	-0.037
Primary vegetation to Altered ecosystem	Soil uses	Other	-0.59
Primary vegetation to Altered ecosystem	Soil uses	PA	0.672
Primary vegetation to Altered ecosystem	Soil uses	APP	-0.531
Primary vegetation to Altered ecosystem	Soil uses	RL	-0.947
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	0	0.178
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	30	-0.212

Transition	Variable	Range Lower Limit	Weight Coefficient
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	60	-0.353
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	90	-0.488
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	120	-0.689
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	150	-0.979
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	180	-0.863
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	210	-0.733
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	240	-0.062
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	270	-0.354
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	330	-2.503
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	390	-1.656
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	480	-1.482
Secondary vegetation to Altered ecosystem	Aspect	N	-0.034
Secondary vegetation to Altered ecosystem	Aspect	NE	-0.183
Secondary vegetation to Altered ecosystem	Aspect	E	-0.368
Secondary vegetation to Altered ecosystem	Aspect	SE	-0.21
Secondary vegetation to Altered ecosystem	Aspect	S	0.109
Secondary vegetation to Altered ecosystem	Aspect	SW	0.258
Secondary vegetation to Altered ecosystem	Aspect	W	0.387
Secondary vegetation to Altered ecosystem	Aspect	NW	0.243
Secondary vegetation to Altered ecosystem	DEM	0	-0.263
Secondary vegetation to Altered ecosystem	DEM	400	0.008
Secondary vegetation to Altered ecosystem	DEM	500	-0.284
Secondary vegetation to Altered ecosystem	DEM	600	0.014
Secondary vegetation to Altered ecosystem	DEM	700	0.403
Secondary vegetation to Altered ecosystem	DEM	800	0.627
Secondary vegetation to Altered ecosystem	Hydrography	0	-0.436
Secondary vegetation to Altered ecosystem	Hydrography	300	0.138
Secondary vegetation to Altered ecosystem	Hydrography	400	-0.093
Secondary vegetation to Altered ecosystem	Hydrography	500	0.005

Transition	Variable	Range Lower Limit	Weight Coefficient
Secondary vegetation to Altered ecosystem	Farmland Structure	Smallholding	0.063
Secondary vegetation to Altered ecosystem	Farmland Structure	Small properties	0.014
Secondary vegetation to Altered ecosystem	Farmland Structure	Medium properties	-0.208
Secondary vegetation to Altered ecosystem	Farmland Structure	Large properties	-0.73
Secondary vegetation to Altered ecosystem	Livestock	High	-0.097
Secondary vegetation to Altered ecosystem	Livestock	Medium-High	0.397
Secondary vegetation to Altered ecosystem	Livestock	Low	0.106
Secondary vegetation to Altered ecosystem	Agricultural Production	High	-0.046
Secondary vegetation to Altered ecosystem	Agricultural Production	Medium-High	0
Secondary vegetation to Altered ecosystem	Agricultural Production	Low	0.036
Secondary vegetation to Altered ecosystem	Slope	none	0.115
Secondary vegetation to Altered ecosystem	Slope	Gentle	0.218
Secondary vegetation to Altered ecosystem	Slope	Moderate	0.355
Secondary vegetation to Altered ecosystem	Slope	Steep	-0.241
Secondary vegetation to Altered ecosystem	Slope	Extremely steep	-1.737
Secondary vegetation to Altered ecosystem	Slope	Excessively steep	-4.445
Secondary vegetation to Altered ecosystem	Soil	Ferralsols	-0.017
Secondary vegetation to Altered ecosystem	Soil	Acrisols (PVAd)	0.456
Secondary vegetation to Altered ecosystem	Soil	Nitisols	-0.147
Secondary vegetation to Altered ecosystem	Soil	Luvisols	-0.156
Secondary vegetation to Altered ecosystem	Soil uses	Other	-0.451
Secondary vegetation to Altered ecosystem	Soil uses	PA	0.396
Secondary vegetation to Altered ecosystem	Soil uses	APP	-0.395
Secondary vegetation to Altered ecosystem	Soil uses	RL	-0.884

Table 5: Weights of evidence ranges and coefficients for the South region

Transition	Variable	Range Limit	Weight Coefficient
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	0	1.125
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	30	0.324
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	60	-0.151
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	90	-0.447
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	120	-0.698
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	150	-0.998
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	240	-1.388
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	270	-1.799
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	630	-3.066
Primary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	660	-5.543
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	0	0.801
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	30	0.45
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	60	0.197
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	90	-0.028
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	150	-0.213
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	180	-0.429
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	270	-0.835
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	300	-1.137
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	360	-1.509
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	570	-3.274
Primary vegetation to Secondary vegetation	Distance to Existing Secondary vegetation	600	-4.282
Primary vegetation to Altered ecosystem	Aspect	N	0.067
Primary vegetation to Altered ecosystem	Aspect	NE	0.046
Primary vegetation to Altered ecosystem	Aspect	E	-0.006
Primary vegetation to Altered ecosystem	Aspect	SE	-0.103
Primary vegetation to Altered ecosystem	Aspect	S	-0.227
Primary vegetation to Altered ecosystem	Aspect	SW	-0.053
Primary vegetation to Altered ecosystem	Aspect	W	0.139

Transition	Variable	Range Limit	Weight Coefficient
Primary vegetation to Altered ecosystem	Aspect	NW	0.175
Primary vegetation to Altered ecosystem	DEM	0	-0.915
Primary vegetation to Altered ecosystem	DEM	100	-2.098
Primary vegetation to Altered ecosystem	DEM	200	-1.148
Primary vegetation to Altered ecosystem	DEM	400	0.507
Primary vegetation to Altered ecosystem	DEM	500	0.11
Primary vegetation to Altered ecosystem	DEM	700	0.468
Primary vegetation to Altered ecosystem	DEM	800	0.538
Primary vegetation to Altered ecosystem	Hydrography	0	1.206
Primary vegetation to Altered ecosystem	Hydrography	100	0.545
Primary vegetation to Altered ecosystem	Hydrography	200	0.228
Primary vegetation to Altered ecosystem	Hydrography	600	-0.107
Primary vegetation to Altered ecosystem	Hydrography	700	-0.161
Primary vegetation to Altered ecosystem	Farmland Structure	Smallholding	0.132
Primary vegetation to Altered ecosystem	Farmland Structure	Small properties	0.137
Primary vegetation to Altered ecosystem	Farmland Structure	Medium properties	-0.02
Primary vegetation to Altered ecosystem	Farmland Structure	Large properties	-0.016
Primary vegetation to Altered ecosystem	Vegetation extraction	High	-0.127
Primary vegetation to Altered ecosystem	Vegetation extraction	Medium-High	-0.188
Primary vegetation to Altered ecosystem	Vegetation extraction	Medium	0.051
Primary vegetation to Altered ecosystem	Vegetation extraction	Low	0.132
Primary vegetation to Altered ecosystem	Agricultural Production	High	0.143
Primary vegetation to Altered ecosystem	Agricultural Production	Medium-High	-0.078
Primary vegetation to Altered ecosystem	Agricultural Production	Low	-0.535
Primary vegetation to Altered ecosystem	Agricultural practices	Up to 95%	-11.495
Primary vegetation to Altered ecosystem	Agricultural practices	From 95% to 75%	-0.06
Primary vegetation to Altered ecosystem	Agricultural practices	Under 75%	0.118
Primary vegetation to Altered ecosystem	Slope	none	0.991
Primary vegetation to Altered ecosystem	Slope	Gentle	0.941

Transition	Variable	Range Limit	Weight Coefficient
Primary vegetation to Altered ecosystem	Slope	Moderate	0.787
Primary vegetation to Altered ecosystem	Slope	Steep	-0.185
Primary vegetation to Altered ecosystem	Slope	Extremely steep	-1.962
Primary vegetation to Altered ecosystem	Slope	Excessively steep	-3.364
Primary vegetation to Altered ecosystem	Soil	Acrisols (PVAd)	0.523
Primary vegetation to Altered ecosystem	Soil	Nitisols	0.562
Primary vegetation to Altered ecosystem	Soil	Luvisols	0.173
Primary vegetation to Altered ecosystem	Soil	Regosols	-0.617
Primary vegetation to Altered ecosystem	Soil	Acrisols (PVa)	0.266
Primary vegetation to Altered ecosystem	Soil uses	Other	-0.675
Primary vegetation to Altered ecosystem	Soil uses	PA	0.534
Primary vegetation to Altered ecosystem	Soil uses	APP	-0.77
Primary vegetation to Altered ecosystem	Soil uses	RL	-1.121
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	0	0.22
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	30	-0.121
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	60	-0.31
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	150	-0.695
Secondary vegetation to Altered ecosystem	Distance to Existing Altered ecosystem	180	-0.724
Secondary vegetation to Altered ecosystem	Aspect	N	-0.084
Secondary vegetation to Altered ecosystem	Aspect	NE	-0.177
Secondary vegetation to Altered ecosystem	Aspect	E	-0.284
Secondary vegetation to Altered ecosystem	Aspect	SE	-0.263
Secondary vegetation to Altered ecosystem	Aspect	S	0.124
Secondary vegetation to Altered ecosystem	Aspect	SW	0.731
Secondary vegetation to Altered ecosystem	Aspect	W	0.889
Secondary vegetation to Altered ecosystem	Aspect	NW	0.316
Secondary vegetation to Altered ecosystem	DEM	0	-0.332
Secondary vegetation to Altered ecosystem	DEM	100	-1.312
Secondary vegetation to Altered ecosystem	DEM	200	-1.965

Transition	Variable	Range Limit	Weight Coefficient
Secondary vegetation to Altered ecosystem	DEM	300	-0.587
Secondary vegetation to Altered ecosystem	DEM	400	0.27
Secondary vegetation to Altered ecosystem	DEM	500	-0.014
Secondary vegetation to Altered ecosystem	DEM	700	0.417
Secondary vegetation to Altered ecosystem	DEM	800	0.46
Secondary vegetation to Altered ecosystem	Hydrography	0	1.262
Secondary vegetation to Altered ecosystem	Hydrography	100	0.446
Secondary vegetation to Altered ecosystem	Hydrography	200	0.061
Secondary vegetation to Altered ecosystem	Hydrography	500	-0.085
Secondary vegetation to Altered ecosystem	Farmland Structure	Smallholding	0.029
Secondary vegetation to Altered ecosystem	Farmland Structure	Small properties	-0.026
Secondary vegetation to Altered ecosystem	Farmland Structure	Medium properties	-0.314
Secondary vegetation to Altered ecosystem	Farmland Structure	Large properties	0.484
Secondary vegetation to Altered ecosystem	Vegetation extraction	High	-0.238
Secondary vegetation to Altered ecosystem	Vegetation extraction	Medium-High	-0.081
Secondary vegetation to Altered ecosystem	Vegetation extraction	Medium	-0.149
Secondary vegetation to Altered ecosystem	Vegetation extraction	Low	0.318
Secondary vegetation to Altered ecosystem	Agricultural Production	High	0.107
Secondary vegetation to Altered ecosystem	Agricultural Production	Medium-High	0.015
Secondary vegetation to Altered ecosystem	Agricultural Production	Low	-0.372
Secondary vegetation to Altered ecosystem	Slope	none	0.826
Secondary vegetation to Altered ecosystem	Slope	Gentle	0.742
Secondary vegetation to Altered ecosystem	Slope	Moderate	0.563
Secondary vegetation to Altered ecosystem	Slope	Steep	-0.305
Secondary vegetation to Altered ecosystem	Slope	Extremely steep	-1.331
Secondary vegetation to Altered ecosystem	Slope	Excessively steep	-3.385
Secondary vegetation to Altered ecosystem	Soil	Acrisols (PVAd)	0.295
Secondary vegetation to Altered ecosystem	Soil	Nitisols	0.485
Secondary vegetation to Altered ecosystem	Soil	Luvisols	0.066

Transition	Variable	Range Limit	Weight Coefficient
Secondary vegetation to Altered ecosystem	Soil	Regosols	-0.393
Secondary vegetation to Altered ecosystem	Soil	Acrisols (PVa)	0.069
Secondary vegetation to Altered ecosystem	Soil uses	Other	-0.527
Secondary vegetation to Altered ecosystem	Soil uses	PA	0.265
Secondary vegetation to Altered ecosystem	Soil uses	APP	-0.39
Secondary vegetation to Altered ecosystem	Soil uses	RL	-0.975

CHAPTER III: COMPARING DIFFERENT APPROACHES TO SIMULATING HISTORIC DISCHARGE BASED ON DYNAMIC LAND USE AND COVER MAPS AND THEIR IMPLICATIONS

ABSTRACT

Land use types and management practices significantly affect hydrological processes. Several studies have evaluated the effects of land-use and land cover (LULC) change on water resources. However, the simulation has often been done without considering land-use dynamics. In this study, we assessed to what extent the Soil and Water Assessment Tool (SWAT) model can be improved by incorporating time series land use and land cover maps. To demonstrate the difference between static and dynamic land-use approaches, the SWAT model was used in the Guaporé watershed (2,490 km^2), southern Brazil, to simulate the response of hydrological processes to LULC from 2008 to 2020. Mapbiomas produced the LULC map*s*, and the model performance was assessed based on Nash–Sutcliffe efficiency coefficient, Kling–Gupta efficiency criterion, and Percent bias. Overall, we observed that both simulations are classified as “satisfactory”. However, over time, spatial changes in land use and cover affect the range of fitted values to Alpha_BF, GW_REVAP, RCHRG_DP, CH_K2, SOL_AWC, and SOL_BD parameters. Land-use changes also affect hydrological processes. In general, the static land-use sub estimates evapotranspiration (ET), soil water content (SW), and surface runoff (SURQ) when compared to the dynamic land-use approach. Therefore, an analysis based on a single baseline map could result in an unrealistic representation of water balance since different land-use types imply changes in water infiltration, runoff, plant canopy, among others. Finally, our results highlighted that controlling LULC change is essential for long-term water management quantifying water resources.

Keywords: Watershed hydrological modelling; Dynamic land use; Static land use; Streamflow; watershed management; Atlantic Forest

1 INTRODUCTION

Different types of land use and their management practices influence hydrology at both the field and catchment scales. For example, under normal rainfall conditions, trees can intercept and transpire more water than other vegetation types, contributing to the reduction of runoff, soil erosion, and local flood risk while also ensuring reasonable recharge to maintain baseflow (Van Meerveld et al., 2021). A shift in the land use to cropping systems with the associated soil management practices can induce changes in water balance. Evidence from several hydrological modelling studies has demonstrated that the type of land cover and management practices affect streamflow, evapotranspiration, infiltration, and annual runoff in different parts of the world; e.g., in Brazil (Ferreto et al., 2021; Valente et al., 2021), Germany (Mehdi et al., 2015), Austria (Schürz et al., 2019), Ethiopia (Dibaba et al., 2020), Indonesia (Marhaento et al., 2017), and in Southeast Asia (Shrestha et al., 2018). Therefore, hydrological models that always assess water balance components must consider land use as an essential factor.

Impacts of land use and cover changes (LULC) and the associated land management on water resources have been widely investigated (Koch et al., 2012; Marhaento et al., 2017; Serrão et al., 2022; Zhang et al., 2021). Usually, a base LULC map is used as input to simulate historical hydrological processes in hydrological models, and the same land-use layer is used for the entire simulation period. However, models represented by a single LULC map lack the capability to describe the history and intensity of land use (Moriasi et al., 2019; Pai & Saraswat, 2011). Therefore, to provide a more realistic approach for scenario simulations and to represent temporal land-use changes, researchers are making attempts to incorporate time-series land use and cover maps into the model based on processes (Lee et al., 2019; Moriasi et al., 2019; Pai & Saraswat, 2011).

Among various hydrological models employed for watershed assessment studies, the Soil and Water Assessment Tool (SWAT) simulates land use and land cover changes through an optional module – landuse update module (Arnold et al., 2012). Pai and Saraswat (2011) provided a scheme explaining the mechanism by which the landuse update module (LUP) works. According to the authors, the module allows the user to modify land use distribution by updating the fractional areas of every hydrological response unit during the model run. To activate the LUP module, two files are needed: (i) "lup.dat" containing the information about when the land use has changed, and (ii) "fnam_file.dat" with the updated fraction values for any HRU for a particular period (Arnold et al., 2012). Even though it has been almost a decade since implementing the LUP module, dynamic integration of land-use change with hydrologic models is rarely found in the literature. The workload involved during the process of implementing dynamic changes and the lack of a user-friendly graphic interface may have contributed to the scarcity of studies using dynamic land-use change in the past, such as applying the LUP module (Pai & Saraswat, 2011) for the SWAT model. However, this changes with the increasing implementation of programming tools, such as "R" and freely available global land use map information.

The use of remotely sensed data and Geographic Information Systems (GIS) to map and reconstruct the historical LULC information was relatively limited until the early 2000s. Only in late 2008, when all Landsat data became freely available on a United States Geological Survey (USGS) online archive, generating increasingly refined products, such as historical land cover, was possible (Wulder et al., 2012). Later, cloud computing Google Earth Engine (GEE) platform with machine learning algorithms allowed to process of a large geospatial dataset and classify land covers over large areas without the need for high-performance computing resources or large-scale commodity cloud computing resources (Gorelick et al., 2017). But it was only very recently, with the improvement of machine learning techniques applied to image processing, that new potential methodologies started to be studied to detect land use and cover changes, as an example of the *Mapbiomas* network initiative (Souza et al., 2020).

The *Mapbiomas* network reconstructed LULC time-series information over three decades in Brazil by combining 30-m-resolution Landsat data, GEE, machine learning, and a network of local experts (Souza et al., 2020). Due to the open access policy, all products, methods, and tools of the *Mapbiomas* Project are freely available on the internet (https://mapbiomas.org/) for non-commercial use. With *Mapbiomas,* it has become more feasible to distinguish different types of land use and cover in more recent years, whereas the LULC map accuracy has increased.

An evaluation of the historical effects of land-use change can be used to understand current hydrological processes and predict the consequences of future LULC changes on water resources. According to Winkler et al. (2021), the main drivers of global land-use change over the last six decades are international trade, expansion in agriculture, socio-economic development, and climate change. The temporal dynamics of global land-use change analysis *per* LULC category show large annual variability in agricultural land-use change, which may justify integrating time series land use and cover maps. Therefore, by implementing different approaches (e.g., static and dynamic land use implementations) to model the hydrologic process, it would be of particular interest to assess which model setup shows a significant difference in the water balance components.

We hypothesize that the hydrological model setup with dynamic land use and cover can more accurately reproduce historic discharge during the calibration/validation phase. The setup of a multiple land use map provides a more realistic approach for scenario simulations and thereby improves the relevance of future assessments on water-related ecosystem services. This study assesses water balance components of the Guaporé watershed. To activate the land use update module, we used the SWAT-LUT Graphical User Interface (Moriasi et al., 2019). The SWAT-LUT implements the use of multiple LULC maps by automatically processing the temporally distributed land use, geoprocessing, and creating new HRUs. The results of this analysis will illustrate the relationship between long-term LULCs and the hydrologic process in a watershed. Furthermore, the results demonstrate the usefulness of integrating dynamic land-use changes during the calibration/validation phase to simulate LULC impacts on hydrology components.

2 MATERIALS AND METHODS

2.1 DESCRIPTION OF THE STUDY SITE

The study area is the Guaporé watershed (GRB), covering an area of ≈2,490 km², located in the Rio Grande do Sul, Brazil. The Guaporé is a tributary of the Jacuí river system, which includes water withdrawals for the Metropolitan Region of Porto Alegre, the state's capital. The original vegetation of GRB is composed of two biomes highly threatened by anthropogenic activities: Atlantic Forest (≈37%) and Pampa (≈1%). In addition, over 60% of the area is anthropized, comprising small-scale farmland, pastures, and scattered urban infrastructure patches. The extent of anthropogenic activities poses a major challenge for sustainable landscape management in the region, threatening the natural resources and ecosystem services, including freshwater provisioning (Bastos et al., 2021; Becker et al., 2009; Ferreto et al., 2021; Kaiser et al., 2010; Kaiser et al., 2015; Reichert, Junior, et al., 2021; Valente et al., 2021) and productive agricultural soils (Ambus et al., 2018; Reichert, Gubiani, et al., 2021; Tiecher et al., 2017).

The agricultural uses in the Guaporé landscape are heterogeneous. In the upper part of the watershed, the main crops are soybean (*Glycine max*) and maize (*Zea mays*) in summer and wheat (*Triticum aestivum*) in winter. In the lower part, the fields are cultivated with tobacco (*Nicotiana tabacum L.*) in summer, maize in spring, and oats (*Avena strigosa* Schreb.) or ryegrass (*Lolium multiflorum* Lam.) in winter. Other land-uses in the catchment include areas afforested with Eucalyptus (*Eucalyptus* spp.) and pastures. In a few cases, natural vegetation within the riparian zone has been converted to perennial grassland to allow the livestock to access the river or to arable farmland (Tiecher et al., 2017). Although the crop systems differ along the catchment, most farmers use similar soil management practices. The grain production is generally cultivated under no-till (Fuentes-Llanillo et al., 2021). It uses identical management practices, while small farmers who produce tobacco are prone to adopt a conventional tillage system (Reichert, Gubiani, et al., 2021; Thomaz & Antoneli, 2021).

The GRB is located in a humid subtropical climate (Figure 1-A; Cfa) and a subtropical highland climate with uniform rainfall (Cfb; Alvares et al., 2013). Based on measured data (2008-2020), the mean annual precipitation in GRB is around 1768.12 mm yr^{-1}, and the mean annual minimum and maximum temperatures range from 13°C to 23°C. Mean annual potential evapotranspiration (PET), estimated by the Hargreaves method (Hargreaves et al., 1985) from measured minimum and maximum temperature, is 1292 mm yr^{-1}, and the mean AET obtained from SWAT output (2008-2020) for this study area is 910 mm yr^{-1}. In terms of geology, the study area belongs to the *Serra Geral* Formation, derived from volcanic lava flows (basalt and rhyolite on the top) and characterized by various facies (Figure 1-B, Caxias, Gramado, and Paranapanema). The watershed elevation ranges from ~50 m to ~900 m above sea level and presents several slope gradients Figure 1-C. The soils are predominantly distributed in five orders: Acrisols (*Argissolo*), Regosols (*Neossolo*), Ferralsols (*Latossolo*), Luvisols (*Luvissolo*), and Nitisols (*Nitossolo*) Figure 1-D (Santos et al., 2018; Wrb, 2015).

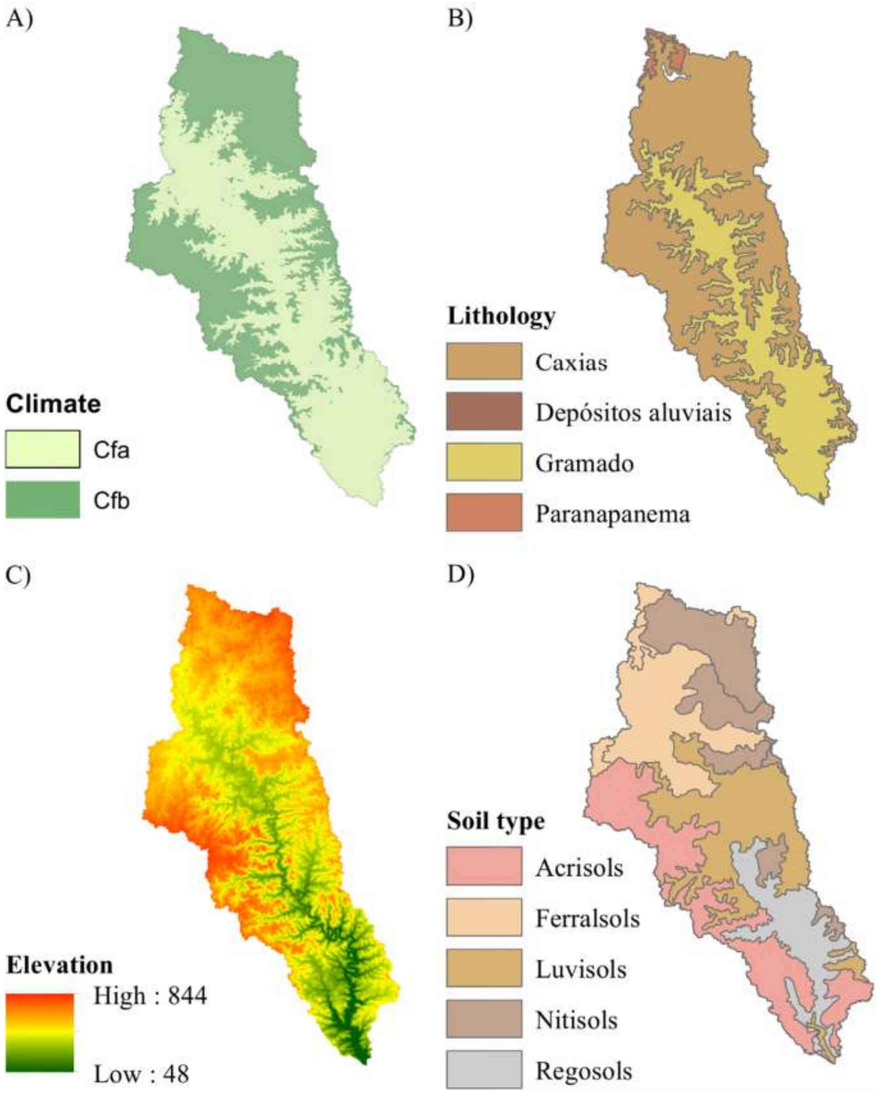

Figure 1: Climatic, Geological, Topographical and Pedological features of the Guaporé watershed.
Source: The author.

2.2 SWAT MODEL DESCRIPTION

In this study, we used the eco-hydrological model SWAT (Arnold et al., 2012; Neitsch et al., 2011) to simulate the hydrological processes of the catchment at a daily time step. First, the SWAT model delineates the catchment into multiple sub-watersheds topologically connected by stream networks (Strauch et al., 2012). Then, each sub-watershed is subdivided into lumped hydrologic response units (HRUs) consisting of homogeneous land-use, soil types, slope, and management characteristics (Neitsch et al., 2011).

In each HRU, SWAT estimates the components of the hydrological cycle by separating the process into two major divisions: (I) land phase, which controls runoff and erosion processes, soil water movement, evapotranspiration, canopy storage, crop growth and yield, soil nutrient and carbon cycling, and (II) routing phase, which is defined as the movement of water, sediments, etc, through the channel network to the watershed outlet (Arnold et al., 2012). The land phase of the hydrological cycle is based on the following equation:

$$SW_t = SW_o + \sum_{i=1}^{t}(R_{days} - Q_{surf} - E_a - W_{seep} - Q_{gw}) \quad \text{(Eq. 1)}$$

where SW_t is the soil water content (mm d^{-1}), t is time (days), Sw_0 is the initial soil water content (mm d^{-1}), and R_{days}, Q_{surf}, E_a, W_{seep}, and Q_{gw} are the amount of precipitation (mm d^{-1}), the surface runoff (mm d^{-1}), evapotranspiration (mm d^{-1}), the amount of water entering the vadose zone from the soil profile (mm d^{-1}), and the amount of return flow (mm d^{-1}), respectively.

The meteorological input data required by SWAT includes daily precipitation, maximum/minimum air temperature, solar radiation, wind speed, and relative humidity. Each hydrological component is estimated through SWAT sub-models related to climate, hydrology, erosion/sediment, land cover and plant growth, nutrients, pesticides, and land management (Neitsch et al., 2011). The Soil Conservation Service (SCS) Curve Number (CN) method is used to estimate surface runoff from daily precipitation (Mishra & Singh, 2003).

The SWAT model provides three options for modelling potential evapotranspiration: (i) Penman-Monteith (Howell & Evett, 2004), (ii) Priestley–Taylor (Priestley & Taylor, 1972), and (iii) Hargreaves (Hargreaves et al., 1985). In this study, we opted to select the Hargreaves since the other methods require variables not available in sufficient resolution or quality.

$$\lambda E_0 = 0.0023 \cdot H_0 \cdot (T_{max} - T_{min})^2 \cdot (T_{mean} - 17.8) \quad \text{(Eq. 2)}$$

where λ is the latent heat of vaporization (MJ kg^{-1}), E_0 is the potential evapotranspiration (mm d^{-1}), H_0 is the extraterrestrial radiation (MJ m^{-2} d^{-1}), T_{max} is the maximum air temperature for a given day (°C), T_{min} is the minimum air temperature for a given day (°C) and T_{mean} is the mean air temperature for a given day (°C).

2.2.1 Model set-up and Inputs

In this study, we implemented SWAT model version SWAT 2012 (Revision 681) and the ArcSWAT Interface version 2012.10.24 to simulate daily discharge time series at the catchment outlet. The initial model set-up includes topography data, which consist of a 30 m spatial resolution digital elevation model (DEM - USGS, 2012) with a minimum, maximum and mean values of 49, 836 and 563.5 m, respectively, slope layer derived from DEM and classified into five slope classes (0 – 3 %, 3 – 8 %, 8 – 20 %, 20 – 45 %, > 45%) based on Embrapa classification, soil layer from Brazilian soil survey maps datasets (Samuel-Rosa et al., 2020; Santos et al., 2011), land-use maps retrieved from *Mapbiomas* (Souza et al., 2020), crop production and land management practices data obtained from a municipality survey (available at https://sidra.ibge.gov.br/), and meteorological data collected from the National Hydro-meteorological Network (available at https://snirh.gov.br and https://bdmep.inmet.gov.br/).

The watershed was delimited into 56 subbasins based on 2500 hectares of the minimum drainage area to form the origin of a stream. The main outlet is in *Doutor Ricardo* (stream-station Santa Lucia - ref. nr. 86580000, Figure 2). Daily precipitation data (2008–2020), maximum/minimum air temperature (2008–2020), wind speed (2008–2020), and relative humidity (2008–2020) were obtained from the HIDROWEB database maintained by *Agência Nacional de Águas* (ANA) and from *Instituto Nacional de Meteorologia* (INMET). The weather stations were assigned automatically to a subbasin by a simple spatial interpolation method. In total, six rainfall gauging stations were used as precipitation input data for SWAT, and three weather stations with air temperature, wind speed, and relative humidity data. The gaps in the measured records of daily precipitation, maximum/minimum air temperature, wind speed and relative humidity were filled by the Weather Generator (WXGEN) algorithm included in SWAT (Neitsch et al., 2011). In contrast, the daily solar radiation was derived from the Global Weather Database.

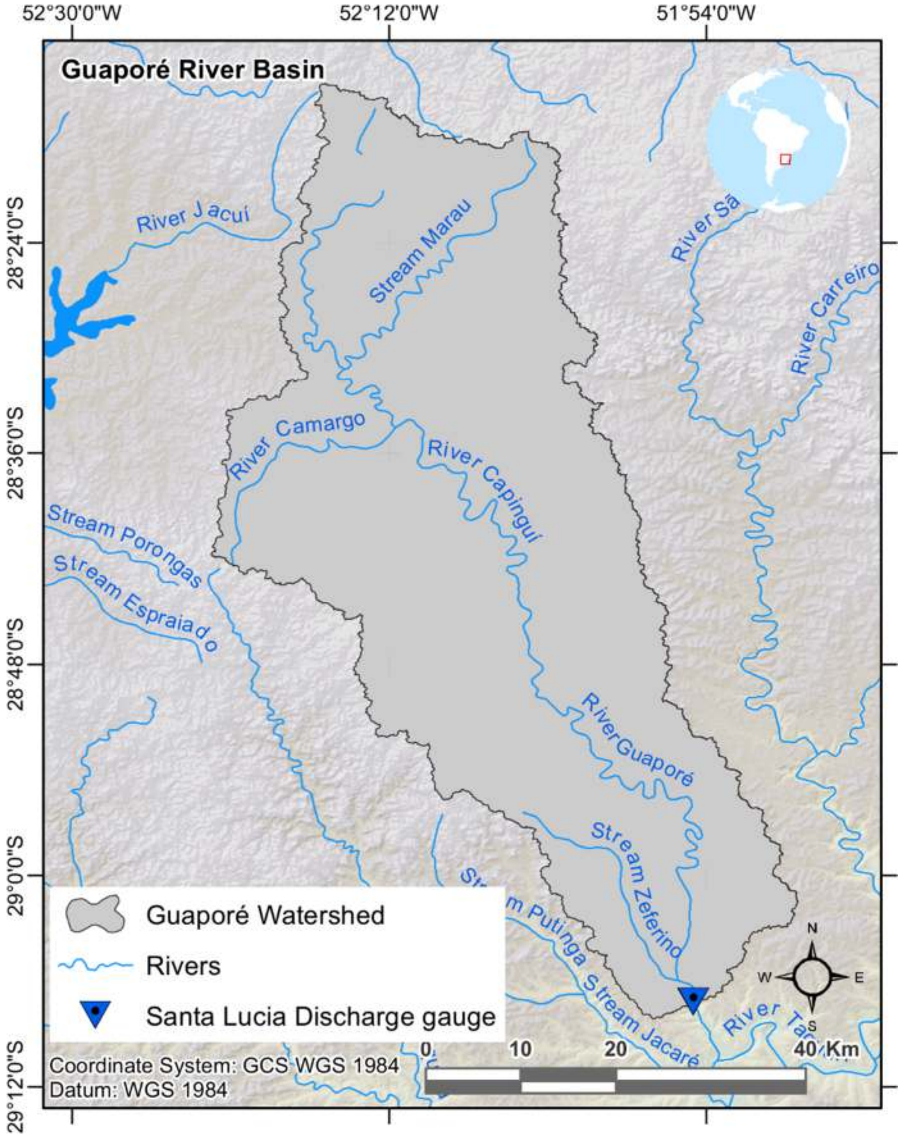

Figure 2: Study site Guaporé watershed (GRB). Source: The author.

The HRUs were created using the multiple HRU option in SWAT by applying a threshold of 0%, 10%, and 10% for land use, soil, and slope classes, respectively. Applying a 10% threshold means that soils and slope ranges whose areas are less than 10% of the subbasin area are ignored during the HRU delineation within each subbasin. The area below the threshold is reassigned (merged) with neighbouring HRUs. A threshold of 0% for land use was selected to avoid eliminating small land-use types from HRU creation and maintain diversity in the land use types. Altogether, 1,497 HRUs were created during the HRU definition process.

Land management information (typical planting, fertilizer application, tillage and harvest dates) for crops and soil management practices were implemented based on historical series of temporary and permanent crops, available at the Municipal Agricultural Production report and on the Brazilian agricultural calendar (Conab, 2021). Incorporating land management practices and changes in land use in the SWAT model provides a more precise representation of field conditions and improves hydrological budget simulations. In a 12-years time span, land-use changes have been dynamic in GRB, with a decrease in pasture and forest areas and an increase in permanent crop areas. Therefore, in this study, the initial 2008 land use and the distribution of HRUs were updated every four years through SWAT-Landuse Update Tool (Moriasi et al., 2019) based on geospatial datasets from 2008 to 2020 available at *Mapbiomas* (Souza et al., 2020).

The SWAT-LUT provides a graphical interface to process multiple land use maps needed to activate the LUP module in SWAT. New HRU areas are calculated by overlaying the spatial information of the existing SWAT project (e.g., soil and sub-watershed map) with the land use maps without setting thresholds. The final product is a set of files, one for each year of the simulation period, with the fractional area of each HRU. These files are then copied into the project folder and used for SWAT model simulations (Moriasi et al., 2019; Pai & Saraswat, 2011).

2.3 DATA SOURCE AND PROCESSING

2.3.1 Annual land use and land cover maps

In ArcSWAT, land use maps perform a grid-based overlay operation, which identifies a unique combination of land use, soil, and slope. This process enables the model to reflect differences in evapotranspiration and other hydrologic conditions for different land use. In this study, to build the baseline model, we used a land use map from the year 2008, while three other land use maps with timestamps of 2012, 2016, and 2020 were utilized to simulate changes in land use distribution for the 2008–2020 period.

The annual land use/cover maps were retrieved from the *Mapbiomas* platform (data available at http://plataforma.brasil.mapbiomas.org). According to Souza et al. (2020), each map is derived from a 30 m resolution Landsat imagery. The Landsat imagery is processed on the Google Earth Engine platform. First, a clean image is created by selecting the cloudless pixels. Then, different metrics are extracted for each pixel of the seven satellite spectral bands. At the end of this process, each pixel carries 105 layers of information. Next, for each land use/cover class, an automatic classifier called "random forest" is applied to train and classify samples targets obtained from reference maps. Next, a temporal filter is applied. Finally, the maps of each class are integrated into a single map, which represents the coverage and land use for each year. Detailed information regarding the classification methodology is available in the ATBD (Algorithm Theoretical Basis Document).

Furthermore, to present the LULCs maps in a form acceptable in the ArcSWAT, we defined a common generalised classification scheme, where each LULC map was reclassified based on their cover and use. A detailed table with the correspondence class between LULC and reclassified LULC is provided in Table 1.

Table 1: *Mapbiomas* collections classes reclassified in SWAT land-use classification.

***Mapbiomas* collections**	**ArcSWAT classification**
Forest Formation	FRST
Forest Plantation	EUCA
Wetlands	RNGE
Grassland	RNGE
Pasture	PAST
Mosaic Agriculture and Pasture	AGRL
Urban Area	URHD
Other non Vegetaded Areas	URHD
River, Lake and Ocean	WATR
Soybean	SOYB
Other temporary Crops	AGRL

FRS: Forest, EUCA: Eucalyptus, RNGE: Grasslands, PAST: Pasture, AGRL: Crops, URHD: Urban Areas, WATR: Water, SOYB: Soybean

2.3.2 *Dominant crop rotation patterns*

Cropping system characteristics have direct and indirect effects on hydrologic. Therefore, SWAT simulates plant growth based on 35 crop-growth parameters stored in a SWAT crop database. Here, twelve years (2008-2020) of LULC data, prepared as described earlier, were used for the analysis of land-use trends, whereas the annual crop information from 2008 to 2020 for the GRB was extracted from the "*Produção Agrícola Municipal* – PAM" database (available at https://sidra.ibge.gov.br/pesquisa/pam/tabelas). Furthermore, to identify the dominant crop rotation patterns in each municipality, a threshold of 10% was set based on the relative cultivated area (Figure 3). At the same time, a generic AGRL type was assigned for those crops that did not reach the threshold value.

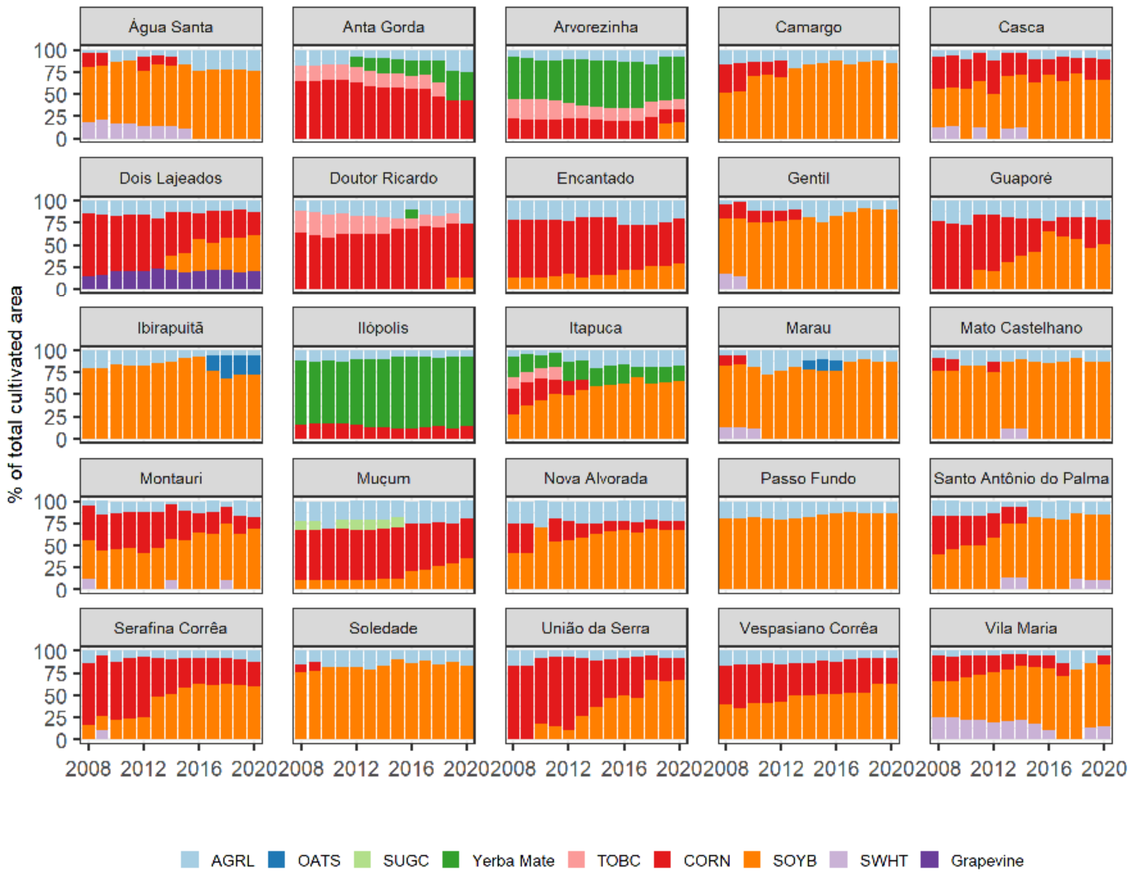

Figure 3: Percentage of total cultivate area per crop. Source: Table 5457: Municipal Agricultural Production. AGRL: Agricultural Land-Generic; OATS: Oats; SUGC: Sugarcane; TOBC: Tobacco, CORN: Corn; SOYB: Soybean; SWHT: Wheat. Source: The author.

We applied crop rotations uniformly across all agricultural areas (AGRL) in the watershed based on dominant crop rotation patterns and slope characteristics. Five rotation systems were established OATS-CORN-SWHT, TOBC-CORN, SOY-CORN, Yerba Mate and Grapevine. Then, to properly enter crop rotations in the SWAT model, the management operations were scheduled based on dates instead of the default of accumulated heat units. The rotation includes tillage operations and nitrogen and phosphorous fertiliser applications after harvest and before planting dates. The planting and harvest were based on the Brazilian Calendar of Grain Planting and Harvesting (Conab, 2021), while the fertilizer applications followed Tedesco et al. (2004) recommendations.

2.3.3 Soil database

A dataset of 218 soil profiles (*pedons* collection) was used in this study (available at http://pedometria.org) to create a "conceptual soil profile" containing the information required by SWAT (per cent sand, silt, and clay, bulk density, cation exchange capacity, saturated hydraulic conductivity, soil depths, water storage, organic carbon, and soil erodibility factor). According to the current Brazilian Soil Classification System (Santos et al., 2018), the soil classification was reviewed and later correlated with the international system World Reference Base for Soil Resources (Wrb, 2015). The quality of the dataset was assessed based on the presence of the values of physical properties. For each observation, we verified if the clay, silt, sand data were available. In the case of missing values, the profile was removed from the analysis.

The collection of *pedons* was converted into a standardized soil horizon structure based on morphological description and types of horizons. A new set of labels (A, B, Bi, Bt, Bw e C) were assigned to each horizon through the generalised horizon labels routine. Next, soil properties were aggregated through soil depth functions using the slice-wise algorithm from AQP package (https://ncss-tech.github.io/AQP/). This algorithm is based on the premise that *a representative depth function for some soil property (e.g., clay content) can be generated from a collection of soil profiles by summarizing this property along with depth slides* (Beaudette et al., 2013). The algorithm distributes the soil attributes by one-centimetre layers and posterior aggregation by horizons/layers.

For the profile collection, summary statistics are computed along with the slices. An estimate of central tendency and spread around that tendency for each depth slice is reconstituted as a single "representative depth function" (Beaudette et al., 2013), to which we refer as the conceptual soil profile. Bellow follows the syntax used to reassemble soil attributes by horizon:

$$slab(soilProfile, \sim genhz + \text{clay} + \text{silt} + \text{sand} + \text{bulk density} + \text{AWC} + \text{carbon}, \text{ slab.fun} = \text{mean/median}, \text{na.rm} = \text{TRUE}) \quad \text{(Eq. 3)}$$

where $genhz$ is the generalized horizon labels, AWC is the available water capacity of the soil layer, slab.fun is a native argument of the function to process each 'slab' of data, and na.rm refers to the logical parameter that tells the function whether or not to remove NA values from the calculation (Beaudette et al., 2013).

Soil hydraulic parameters were estimated using pedotransfer functions due to the lack of available data. Measured values of sand, silt, clay, and bulk density at each soil layer are used as inputs to the ROSETTA API (Zhang & Schaap, 2017) to assess the values of saturated hydraulic conductivity and water storage, while the *solum* depth (A+B horizon) was used as a *proxy* for the maximum rooting depth of soil profile.

2.3.4 Definition of hydrological modelling scenarios

To assess the added value of hydrological discharge simulation using dynamical LULC, monitoring data (2008–2020) was used for model calibration and validation, under two scenarios: (I) static land use cover scenario (SLUC), using a static LULC map of the watershed in 2008; and (II) dynamic land-use cover scenario (DLUC), integrating historical LULC maps from *Mapbiomas*. The performance was then evaluated at the daily, monthly, and annual time scales in both cases.

2.3.5 SWAT calibration, validation and uncertainty analysis

With a 5-year warm-up period, the SWAT model was calibrated and validated using a multi-criteria sensitivity analysis at the Santa Lucia stream-station for discharge at the daily time step from January 1, 2008, to December 31, 2015. Initially, 18 sensitive model parameters for discharge (Table 2) were selected based on a review of the existing literature (Andrade et al., 2019; Bonuma et al., 2014; A. d. N. Ferreira et al., 2021; Fukunaga et al., 2015; Strauch et al., 2012; Strauch & Volk, 2013). Then, the initial simulation was performed using default parameter values obtained from the SWAT databases (either standard values or user-defined values). Thereafter, the values were changed over a certain range by increment, by a fraction or by replacing to obtain a maximum fit of simulated to measured variables. Regarding sensitivity analysis and calibration/validation process, we followed a framework proposed by Schürz et al. (2019). This framework is based on the STAR-VARS approach (Razavi & Gupta, 2016a, 2016b) whereby the most sensitive model parameters are screened, ranked, and then selected. After that, the values are adjusted (within predefined ranges) to obtain a maximum fit of simulated to measured variables. Thereby, three rounds of model simulations with a sample of 5000 parameter combinations selected by using Latin Hypercube Sampling were conducted. In each round, SWAT model performance was evaluated through by visual assessment of the plotted hydrographs, and statistical analysis, i.e., the coefficient values of Nash–Sutcliffe Efficiency (*NSE (Nash & Sutcliffe, 1970)*, Kling–Gupta efficiency criterion (KGE (Gupta et al., 2009), and percentage bias (*PBIAS (Gupta et al., 1999)*.

Table 2: Selected SWAT model parameters for the GRB watershed's sensitivity analysis and calibration procedure.

Parameter	Description	Type of change	Initial range
Groundwater			
ALPHA_BF.gw	Baseflow alpha factor (1/days)	absval	0.01 to 1
GW_REVAP.gw	Groundwater "revap" coefficient	absval	0.01 to 1
REVAPMN.gw	Percolation to the deep aquifer to occur (mm H2O)	absval	0 to 500
RCHRG_DP.gw	Deep aquifer percolation fraction	absval	0.01 to 1
GW_DELAY.gw	Groundwater delay time (days)	absval	0 to 250
GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur (mm H2O)	absval	0 to 5000
Surface Runoff			
CN2.mgt	Initial SCS runoff curve number for moisture condition II	pctchg	-20 to 20
SURLAG.hru	Surface runoff lag coefficient	absval	0.1 to 12
Potential and Actual Evapotranspiration			
ESCO.hru	Soil evaporation compensation factor	absval	0.01 to 1
EPCO.hru	Plant uptake compensation factor	absval	0.01 to 1
CANMX.hru	Maximum canopy storage (mm H2O)	absval	0.1 to 6
Time of Concentration			
OV_N.hru	Manning's "n" value for overland flow	absval	0 to 1
Soil Water			
SOL_K.sol	Saturated hydraulic conductivity (mm/hr)	relchg	-0.5 to 2
SOL_BD.sol	Moist bulk density (Mg/m3 or g/cm3)	relchg	-0.5 to 2
SOL_AWC.sol	Available water capacity of the soil layer (mm H2O/mm soil)	relchg	-0.5 to 2
Lateral Flow			
LAT_TTIME.hru	Lateral flow travel time (days)	absval	0.5 to 50
Channel Water Routing			
CH_N2.rte	Manning's "n" value for the main channel.	relchg	-0.5 to 1
CH_K2.rte	Effective hydraulic conductivity in main channel alluvium (mm/hr)	absval	0 to 300

'relchg' and *'pctchg'* alters a parameter by a fraction or a percentage of the initial parameter value, *"abschg"* adds an absolute value to the initial parameter value; and *"absval"* replaces a parameter by an absolute value.

The NSE describes the deviation from the unity of the ratio of the square of the difference between the observed and simulated values and the variance of the observations (Eq. 4):

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs} - \bar{Y}\right)^2}\right] \quad \text{(Eq. 4)}$$

where Y_i^{sim} is the simulated variable, Y_i^{obs} is the measured variable at time t, and $\bar{Y}$ is the mean measured variable. The NSE can range from -∞ to 1. Values between 0.0 and 1.0 are generally viewed as acceptable levels of performance; however, the model performance only can be evaluated as "satisfactory" if NSE > 0.50 monthly flow, while values from 0.65 to 0.75 are classified as "good" and values above 0.75 are "very good" (Boskidis et al., 2012; Moriasi et al., 2007).

The KGE is based on the decomposition of the NSE and the mean square error (MSE) into three components: correlation, variability error, and bias error. A three-dimensional criteria space is created, where the KGE values are calculated in terms of the Euclidian distance from the ideal point and the Pareto three-dimensional surface (Eq. 5):

$$KGE = 1 - \sqrt{(r-1)^2 + (\alpha - 1)^2 + (\beta - 1)^2} \quad \text{(Eq. 5)}$$

where r is the linear regression coefficient between the measured and simulated variable (correlation term), $\alpha = \sigma_s/\sigma_m$ is the variability depicted as the ratio between the standard deviation of the simulated and measured variable, $\beta = \mu_s/\mu_m$ is the bias ratio (i.e. ratio between the mean simulated and mean measured variable), with σ_s and σ_m the standard deviations and μ_s and μ_m the means of simulated and measured variables, respectively. KGE values range from – ∞ to 1, where 1 represents the ideal value. An interesting feature of the KGE is that it reflects the lower limit of r, β and α, i.e., for a given value KGE, it implies that the worst of the three components is higher than or equal to KGE (Piniewski et al., 2017).

Per cent bias (PBIAS) measures the average tendency of the simulated data to be larger or smaller than their observed counterparts. The optimal value of PBIAS is 0.0, with low-magnitude values indicating accurate model simulation. Positive values indicate model underestimation bias, and negative values indicate model overestimation bias. PBIAS is, generally, expressed in percentage and is calculated using (Eq. 6):

$$PBIAS\% = 100 \cdot \frac{\sum_{i=1}^{n}(S_i - O_i)}{\sum_{i=1}^{n} O_i} \quad \text{(Eq. 6)}$$

where S_i and O_i are respectively simulated and the observed variable for a day i. Similar to NSE, PBIAS ≤ ±25% are classified as acceptable, and PBIAS ≤ ±10% are "very good" (Boskidis et al., 2012; Moriasi et al., 2007).

2.3.6 Evaluation effects of LULC in water balance components

Besides model performance evaluation, mean annual values of surface run-off (SURQ), baseflow (GWQ), soil water content (SW), evapotranspiration (ET) and water yield (WY) were used to assess the impact of the spatial and temporal variation of land uses during the calibration phase. Then the data was submitted to the Kolmogorov-Smirnov ($p < 0.05$) normality test. Considering that the data presented normal distribution, the parametric t-paired test with Bonferroni correction was used to verify the hypothesis that the water balance components simulated by static land use and dynamic land-use approaches present a significant difference between them ($p < 0.05$).

3 RESULTS

3.1 MODEL PERFORMANCE AND PARAMETER VALUE IDENTIFICATION ON DISCHARGE SIMULATION

The overall performance of the model in terms of Nash–Sutcliffe efficiency coefficient, Kling–Gupta efficiency criterion, and Percent bias showed quite satisfactory results for both scenarios: static land and dynamic land cover. Figure 4 presents the distribution of the model performance metrics obtained for calibration and validation for SLUC and DLUC setups. The NSE, KGE and PBIAS values of the DLUC setup were 0.65, 0.67 and 2.8% for the calibration period and 0.73, 0.62 and -3.3% for the validation period. However, the model performance was good in the case of the SLUC setup with NSE, KGE and PBIAS values of 0.66, 0.68 and 0.4 for calibration; and 0.74, 0.61 and -6.4% for validation.

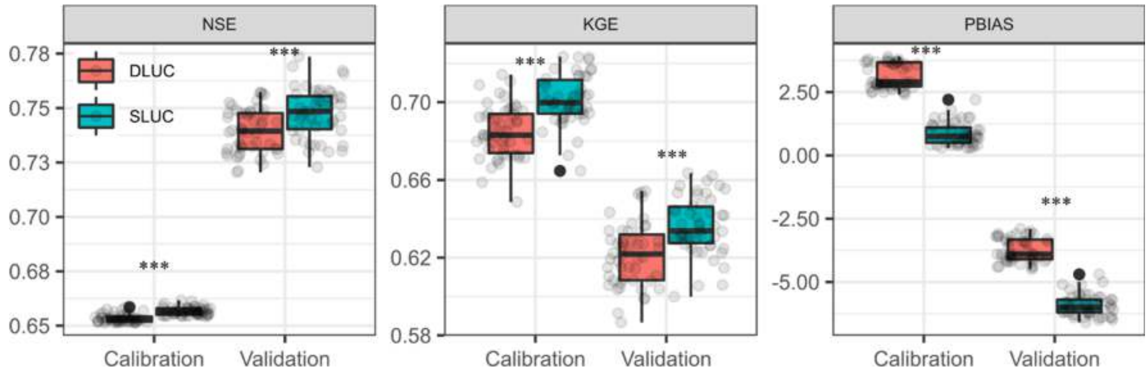

Figure 4: Variation of SWAT performance for calibration (2008–2015) and validation (2016–2020) at daily time-step. ***p value < 0.01 indicate significant differences between comparisons of DLUC and SLUC. Source: The author.

The comparison of measured daily values of streamflow and simulated values during the calibration and validation period are shown in Figure 5. Visually, it is possible to observe that the variability and timing of streamflow were quite compatible with observed values in both land use models' scenarios, except for a few peak flows. In addition, despite simulated values exhibiting good performance in both scenarios, the PBIAS coefficient indicates a slight deviation in the SWAT results in relation to the measured flow in either land-use setup.

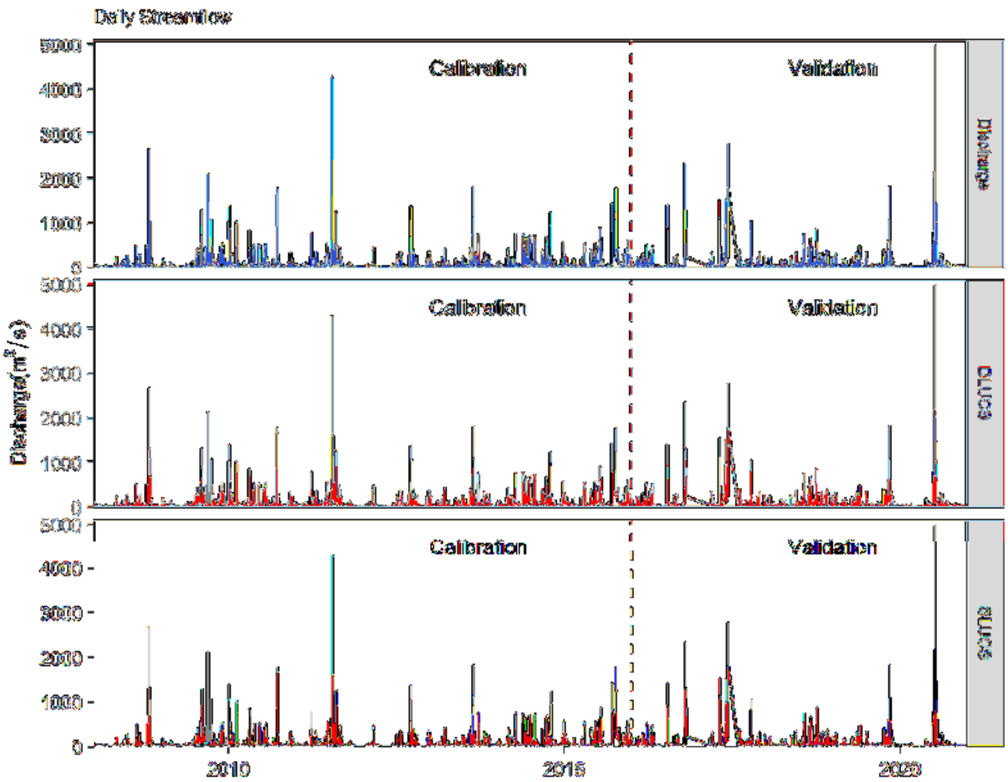

Figure 5: Daily simulation results of streamflow during calibration (2008–2015) and validation (2016–2020). Source: The author.

The sensitive model parameters for discharge were calibrated and constrained within three rounds of model simulations for both scenarios: static land use cover (SLUC) and dynamic land use cover (DLUC). These parameters are related to evapotranspiration, soil moisture, groundwater, surface runoff, and streamflow processes (defined in Table 2). As shown in Table 3, the final range values assessed through parameter identifiability differs between the two model setups. In addition, the Alpha_BF, GW_REVAP, RCHRG_DP, CH_K2, SOL_AWC, and SOL_BD parameters were impacted by spatial changes in land use and cover over time (Table 3).

Table 3: Calibrated discharge-related parameters to Static land use cover (SLUC) and Dynamic land use cover (DLUSC) and their calibration ranges and optimal values.

Parameters	**Static land use cover**		**Dynamic land use cover**	
	Range	**Fitted value**	**Range**	**Fitted value**
ESCO_North	[0.011, 0.141]	0.048	[0.901, 0.931]	0.929
ESCO_South	[0.901, 0.947]	0.947	[0.800, 0.865]	0.826
ESCO_FRST_North	[0.084, 0.631]	0.232	[0.077, 0.631]	0.631
ESCO_FRST_South	[0.023, 0.842]	0.521	[0.023, 0.730]	0.023
EPCO_North	[0.419, 0.495]	0.465	[0.633, 0.969]	0.871
EPCO_South	[0.372, 0.995]	0.995	[0.301, 0.997]	0.439
EPCO_FRST_North	[0.748, 0.932]	0.758	[0.648, 0.919]	0.919
EPCO_FRST_South	[0.783, 0.998]	0.932	[0.176, 0.249]	0.226
CANMX_North	[0.610, 2.452]	0.744	[0.759, 2.446]	2.446
CANMX_South	[1.090, 2.995]	2.340	[1.090, 2.995]	1.090
CANMX_FRST_North	[0.017, 3.795]	3.795	[0.017, 3.891]	0.622
CANMX_FRST_South	[0.952, 3.735]	3.671	[0.952, 3.735]	3.624
GW_REVAP	[0.016, 0.025]	0.019	[0.025, 0.035]	0.030
REVAPMN	[55.495, 499.567]	499.567	[55.495, 422.043]	391.125
SOL_K_North	[1.052, 1.862]	1.304	[1.138, 1.954]	1.229
SOL_K_South	[1.525, 1.924]	1.630	[1.527, 1.897]	1.541
SOL_BD_North	[0.847, 1.979]	1.351	[0.258, 0.299]	0.284
SOL_BD_South	[1.021, 1.841]	1.021	[0.353, 0.443]	0.367
SOL_AWC_North	[-0.411, -0.306]	-0.357	[0.268, 0.343]	0.309
SOL_AWC_South	[-0.092, 0.199]	0.199	[0.553, 0.849]	0.791
ALPHA_BF	[0.602, 0.958]	0.690	[0.277, 0.816]	0.419
RCHRG_DP	[0.032, 0.246]	0.174	[0.012, 0.147]	0.105
GW_DELAY	[0.006, 0.989]	0.328	[0.009, 1.329]	0.492
GWQMN	[29.599, 438.342]	271.111	[23.52, 492.743]	271.111
SURLAG_hru_North	[1.168, 11.509]	2.191	[1.572, 11.244]	2.637
SURLAG_hru_South	[0.115, 11.270]	0.552	[0.364, 9.229]	0.552
SURLAG_hru_FRST_North	[0.764, 11.597]	10.698	[0.491, 10.698]	10.698
SURLAG_hru_FRST_South	[0.253, 10.514]	0.336	[0.206, 7.448]	0.336
CN2_North	[5.037, 6.954]	5.762	[0.329, 2.338]	0.952
CN2_South	[-19.705, -10.248]	-17.127	[-19.953, -1.873]	-14.253
CN2_FRST_North	[8.204, 19.29]	8.309	[8.309, 19.966]	8.309
CN2_FRST_South	[-19.087, 6.564]	-6.712	[8.419, 19.287]	13.315
LAT_TTIME_North	[1.107, 7.267]	5.369	[10.028, 19.712]	19.154
LAT_TTIME_South	[1.087, 1.986]	1.717	[1.036, 1.986]	1.648
CH_N2_North	[-0.496, -0.099]	-0.463	[-0.479, -0.207]	-0.302

CH_N2_South	[-0.442, 0.210]	-0.366	[-0.496, 0.248]	-0.049
CH_K2_North	[4.440, 99.967]	64.910	[1.098, 78.773]	46.398
CH_K2_South	[30.349, 99.644]	91.902	[61.544, 79.779]	75.781
OV_N_North	[0.005, 0.992]	0.906	[0.003, 0.995]	0.720
OV_N_South	[0.061, 0.951]	0.753	[0.021, 0.962]	0.195
OV_N_FRST_North	[0.047, 0.993]	0.230	[0.019, 0.968]	0.903
OV_N_FRST_South	[0.013, 0.981]	0.193	[0.013, 0.999]	0.464

3.2 HISTORICAL LAND-USE CHANGE CHARACTERISTICS

Land-use in the past 12-years has significantly evolved in the Guaporé watershed (Lima, in press). In this study, three LULCs maps (2012, 2016 and 2020) were used to update the initial 2008 land use during the 12-year period of the LUC module. As a result, the number of HRUs increased from 1,497 to 3,312 due to the introduction of new land-use types. Figure 6 shows the area coverage distribution of the major land-use types over the years, while the changes of each LULC class at different periods (2008-2012, 2012-2016, and 2016-2020) is presented in Table 4.

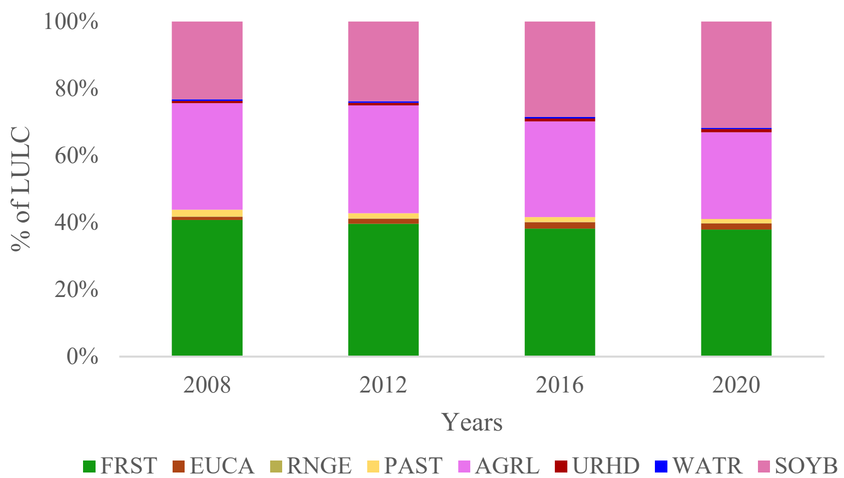

Figure 6: Distribution of major land uses in Guaporé watershed. FRS: Forest, EUCA: *Eucalyptus*, RNGE: Grasslands, PAST: Pasture, AGRL: Crops, URHD: Urban Areas, WATR: Water, SOYB: Soybean. Source: The author.

Overall, the land use dynamic at the watershed scale showed an increase in soybean fields and decreased forest and pasture areas from 2008 to 2020. In a 12-years lifespan, the cultivated soybean area increased by 22,049 hectares. The primary source of land for new soybean fields in Guaporé was existing cropland, pasture, grassland, and Forest. In 2020, agricultural land and soybean fields made up the most significant percentage of the Guaporé watershed, with 60%. The second largest LULC class was forest cover, with 38%. Built-up area, water feature, Eucalyptus, Grasslands, and Pastureland covered only a small portion of the watershed, with these five LULC types together representing just below 5%.

Table 4: Total area and percentage of LULC change between 2008-2012, 2012-2016, and 2016-2020.

LULC Class	**Total Change in the Area (2008-2012)**		**Total Change in the Area (2012-2016)**		**Total Change in the Area (2016-2020)**	
	km²	**%**	**km²**	**%**	**km²**	**%**
FRST	-30.36	-2.8	-37.94	-3.7	-6.48	-0.6
EUCA	13.39	52.0	10.96	28.0	-2.47	-4.9
RNGE	0.23	10.6	0.09	3.8	0.24	9.7
PAST	-10.33	-20.1	-3.19	-7.8	-6.86	-18.1
AGRL	10.87	1.3	-94.70	-11.2	-69.43	-9.3
SOYB	14.50	2.4	120.99	19.4	84.99	11.4
URHD	1.94	10.7	2.50	12.4	1.90	8.4

FRS: Forest, EUCA: *Eucalyptus*, RNGE: Grasslands, PAST: Pasture, AGRL: Crops, URHD: Urban Areas, WATR: Water, SOYB: Soybean

3.3 HYDROLOGICAL IMPACTS OF LAND-USE CHANGE

The effect of land-use changes on the hydrological process is presented in Figure 7, as average annual water yield, evapotranspiration, surface runoff, soil water content, and Groundwater in the watershed between 2008 and 2015. Under the influences of DLUC, the simulated evapotranspiration (ET), soil water content (SW), and surface runoff (SURQ) have increased when compared to SLUC. In contrast, water yield (WYLD) and baseflow (GWQ) showed minor changes with a general increasing trend under the DLUC. The difference among the water balance components within SLUC and DLUC setups may indicate the spatial and temporal variation of land uses in the Guaporé watershed.

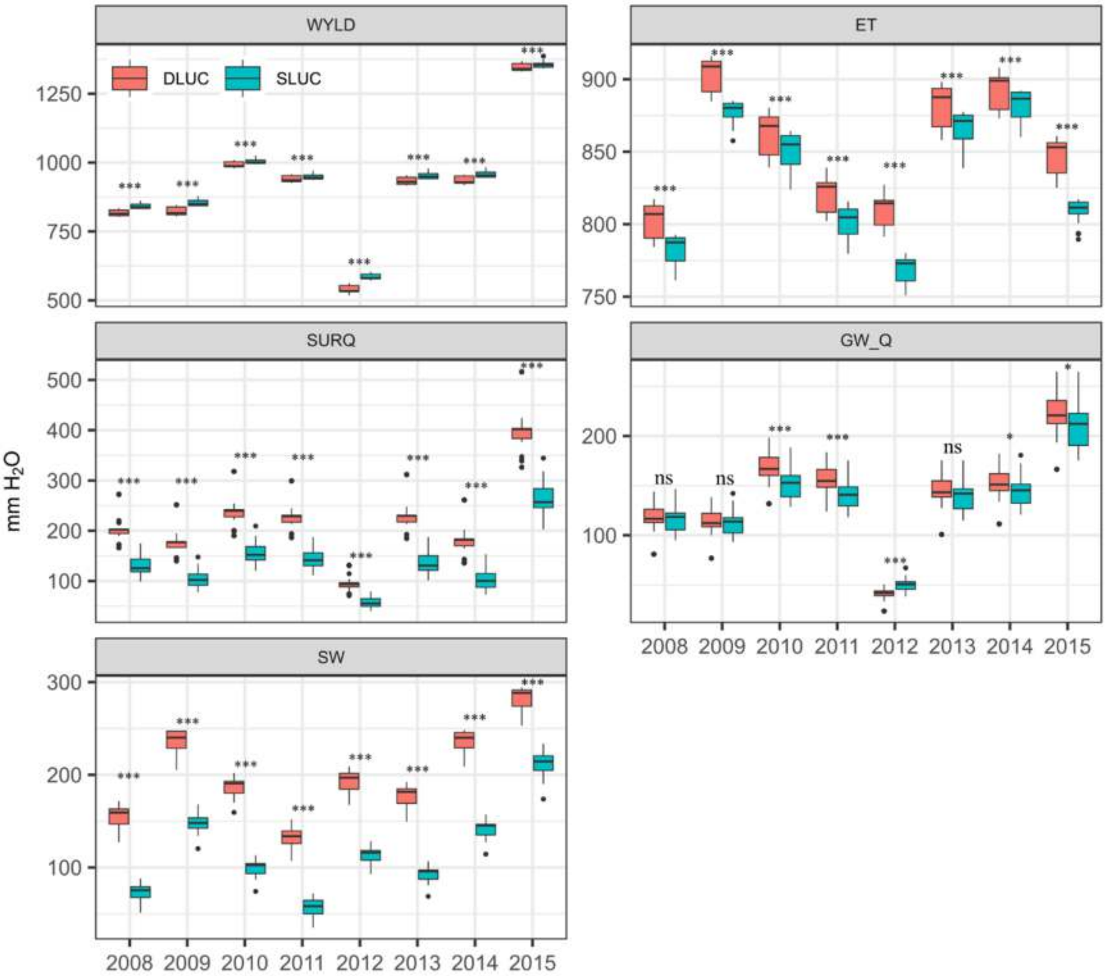

Figure 7: Variation of mean annual surface run-off (SURQ), baseflow (GWQ), soil water content (SW), evapotranspiration (ET) and water yield (WY) in mm for each of the scenarios. *p value < 0.05 and p *** value < 0.001 indicates rejection of the null hypothesis of no difference and thus, revealing the existence of the difference between SLUC and DLUC values (APPENDIX – A). Source: The author.

4 DISCUSSION

4.1 PARAMETER IDENTIFIABILITY AND CALIBRATION

Aiming to minimize the uncertainty in the model due to land-use distribution, our study implemented a temporally dynamic representation of crops planted and their related management practices in the watershed by altering the LULC layer every four years. While a static scenario with a land-use map from the year 2008 was also considered. Concerning model performance, although both dynamic and static scenarios differ between them, their results were in accordance with other research, which emphasizes the capability of SWAT in modelling a subtropical watershed (Bonuma et al., 2014; Ferreira et al., 2021; Fukunaga et al., 2015; Hernandes et al., 2018; Monteiro et al., 2016; Oliveira et al., 2020; Pereira et al., 2014). Further, good calibration and validation performance is evident in the static scenario; however, considering the criteria established by Moriasi et al. (2007), both simulations are classified as "satisfactory". In fact, the adoption of subjective criteria such as Nash-Sutcliffe, KGE, and PBIAS *per se* to identify a unique best fit that is represented by a single set of model parameters could lead to misleading results. Alternatively, all solutions within a satisfactory range should be considered in hydrology modelling (Abbaspour, 2021). The results also agree with the finds of Wang et al. (2018), who described the effect of static and dynamic land-use inputs to simulate flow. According to the authors, both the static and dynamic land-use inputs provide acceptable calibration and validation results.

Similar to model performance, the simulated hydrograph for both scenarios could produce reasonably good results. Although the static and dynamic land-use scenarios capture most of the hydrograph information, some differences exist between simulated data and observed streamflow. Overall, the variability and timing of streamflow were compatible with observed values, except for some cases of streamflow peaks. Furthermore, in both scenarios, the streamflow simulation roughly coincides with no major differences between the two streamflow results, which could be associated to other factors out of the scope of this study, such as spatially heterogeneous distribution of rainfall data, errors in the input data, limitation of model structure, and limitation in the governing equations solved by SWAT. The difference between simulated and observed streamflow due SWAT limitations are widely reported in the literature and expected to occur within an acceptable range.

The temporal variability of land use patterns has a significant impact on the identifiability and calibration of SWAT model for the Guaporé watershed. Considering the results with and without the LUC module, differences in parameter range were greater among those related to groundwater processes such as Alpha_BF, GW_REVAP, RCHRG_DP, along with Channel Water Routing as an example of CH_K2, and Soil water (SOL_AWC and SOL_BD). Recent studies have shown the hydrological responses to different land-use types (Ferreto et al., 2021; Reichert et al., 2017; Valente et al., 2021). Lee et al. (2019) described the importance of considering multiple land use maps and temporal changes in parameters related to LUCs, such as Alpha_BF (alpha-factor). The alpha factor is a direct index of groundwater flow response to changes in recharge (Neitsch et al., 2011). Although estimating the alpha factor based on measured flow data is possible, most calibrations are done without considering it.

Therefore, as demonstrated in our study, a dynamic update of land-use associated with time-series parameters, as an example of alpha-factor showed by Lee et al. (2019), can help the SWAT users simulate more accurate hydrologic responses.

Despite the relationship of parameters with land use and land cover, most SWAT studies focus on calibration process and model efficiency without considering the impact of temporal changes in the landscape. This is the case for Gw_Revap and RCHRG_DP, which controls water movement from a shallow aquifer to unsaturated soil layers and deep aquifer contributions on groundwater flow, respectively. Those parameters have been identified on sensitivity analysis as the most important for the SWAT calibration phase by many other researchers worldwide (Bauwe et al., 2019; Fukunaga et al., 2015; Nasiri et al., 2020; Nyeko, 2014). Both parameters play an important role in the hydrological process: the first control the effects of evapotranspiration losses, while the second contributes to the river baseflow.

Similar to groundwater parameters, the literature lacks discussion about the effect of landscape dynamics on Channel Water Routing and Soil Water, yet they have been reported as sensitive parameters in many studies (Beharry et al., 2021; Narsimlu et al., 2015; Tudose et al., 2021; Yousuf & Singh, 2019). The CH_K2 parameter controls the exchanges between river and groundwater and, therefore, affects watercourse permeability. A high value of CH_K2 indicates the presence of losing streams within a watershed. Since most tributaries are perennial rivers in GRB, we do not expect channel transmission losses. Therefore, in a realistic scenario, we should get low CH_K2 values. In general, the adjustment of CH_K2 is an easy pathway to lower and smoother the simulated discharge peak flow while keeping the water cycle balance intact.

The amount of water available for plant uptake when the soil is at field capacity is represented by SOL_AWC. However, temporal changes in land cover types could affect the capability to transfer water from the lower soil level to the root zone (Sun et al., 2020). In addition, the soil physical (e.g., bulk density) are sensitive to management practices and land use (Haghighi et al., 2010; Reichert et al., 2016; Reichert, Gubiani, et al., 2021). Therefore, changes in land use also led to different constrained range values of SOL_BD, as observed in our study.

4.2 HISTORICAL LAND-USE CHANGES

As the results showed, the land-use in the past 12-years has significantly evolved in the Guaporé watershed. Thus, the variation in the spatial pattern of the land use classes reshape the landscape and can affect the hydrological process, consequently water assessment. This spatiotemporal variability became more evident when the LUC module was activated, which resulted in an increment of HRUs. As already mentioned, the HRUs have unique combinations of land cover and soil types for each subbasin. Therefore, changes in land cover, such as agriculture expansion, may lead to a different landscape configuration. Here, the new HRUs accounted for either the land use classes not present in the baseline map or due to SWAT-LUT not incorporating the previously HRU thresholds (Moriasi et al., 2019). The HRU thresholds are alternatives to increase computational time efficiency while maintaining model accuracy since a large number of HRUs can either reduce computational time efficiency or even exceed the computation limits (Jiang et al., 2021; Zamani et al., 2021).

Our study observed a predominant shift from Forest, Pasture and agricultural land to soybean fields. This transition can be partially attributed to the expansion and modernization of the transport system, the development of agricultural machinery, and the increase in international trade in the last decades (Cattelan & Dall'agnol, 2018; Richards et al., 2012). Additionally, the shift from agricultural land to soybean fields revealed a pathway towards the expansion of export-oriented commodity farms.

Although soybean crops occupy most Brazilian cultivated areas (Maranhão et al., 2019), the spatial difference in soybean yield and planted area are affected by local variations in supply chain and environmental conditions (Garrett et al., 2013; Waroux et al., 2019). For example, in southern Brazil, soybean production already occupies the most productive land. In contrast, competition from other crops, dense population, and massive urbanization further minimize the land availability for grain crop expansion (Flexor & Leite, 2017). Alternatively, Brazilian agriculture has increased its efficiency by adopting multi-cropping strategies, which allows the farms to cultivate two or more crops in the same area throughout the year (Xu et al., 2021). The most common double-cropping system is the soybean-maize, and wheat, oats, and other crops can replace maize offseason. All together, agriculture intensification and favourable weather conditions, such as well-distributed rainfall throughout the year, high solar radiation availability, an optimum temperature range, contribute to maximizing yields without further converting natural ecosystems.

The increase of production without any increase in the agricultural frontier can reduce the negative impact on ecosystems and agrosystems. However, double-cropping systems are highly vulnerable to climate change and El Niño Southern Oscillation (Brumatti et al., 2020), which can put agriculture itself at risk. Furthermore, climate variability reduces the period of the best sowing window of the crops, also their cultivation in succession, which decide which soybean sowing date to be considered a dilemma faced by the farmers as highlighted by Nóia Júnior and Sentelhas (2019). Finally, double cropping also affects water balance components. Spera et al. (2016) demonstrated water-recycling potentials are higher through agricultural intensification than by a single cropping system, while Nkwasa et al. (2020) showed the effect of management practices representation on the variation of evapotranspiration.

Due to their extent and agriculture intensification, most of the negative impacts in the landscape are commonly associated with soybean fields. However, several studies from catchments in southern Brazil have also shown the negative effect of long-term tobacco monoculture (Bastos et al., 2021; Bonuma et al., 2014; Didoné et al., 2014; Reichert, Gubiani, et al., 2021; Tiecher et al., 2017), an important cash crop for the local economy of some municipalities. Unlike soybean fields, mostly cultivated under a no-till system (Cattelan & Dall'agnol, 2018), tobacco is mostly cultivated under conventional tillage (Thomaz & Antoneli, 2021). In addition, farmers usually adopt an industrial-technological package provided by tobacco companies (Leppan et al., 2014), which includes seeds, fertilizers, and technical support, and guarantees the marketing of the produce. Although this integrated production system allowed the expansion and provided support to tobacco processing, it has been equally applied to all producers without considering social and environmental aspects, as reported by Leppan et al. (2014). As a result of these practices, tobacco cultivation is reported as the primary driver of soil degradation (Reichert, Gubiani, et al., 2021; Reichert, Pellegrini, & Rodrigues, 2019; Thomaz & Antoneli, 2021), water contamination (Bastos et al., 2021; Becker et al., 2009; Kaiser et al., 2010), surface runoff and soil erosion (Bonuma et al., 2014; Reichert, Pellegrini, Rodrigues, et al., 2019). Therefore, the temporal changes observed in our study can help to improve the impact assessment of this crop in soil and water.

4.3 HYDROLOGICAL IMPACTS OF LAND-USE CHANGE

The presented analysis suggests that the representation of temporal land-use changes significantly affects hydrological components. Therefore, an analysis based on a single baseline map could result in an unrealistic representation of water balance. In this study, we observed that the yearly evapotranspiration (ET), soil water content (SW), and surface runoff (SURQ) are greatly influenced by the temporal changes in land use distribution. Despite the scarce number of research applying the multitemporal maps approach, land-use changes are commonly reported as an important variable to simulate water balance trends (Dogan & Karpuzcu, 2021; Kordrostami et al., 2021; Tayebzadeh Moghadam et al., 2021).

Anthropogenic changes over the years, such as agricultural intensification, urbanization and deforestation, are usually related to increased evapotranspiration and surface runoff (Pan et al., 2020; Serrão et al., 2022). Further, seasonality also strongly influences evapotranspiration (Nkwasa et al., 2020), yet our study only evaluated the hydrological response due to agricultural land use representation. In southern Brazil, a study by Reichert et al. (2017) demonstrated the impact of different land use on water balance components, where watershed vegetation cover had a strong influence on runoff and volume of water stored into the soil. Therefore, other management practices and land cover could help to improve soil quality, consequently soil infiltration and water retention and availability.

Temporal evolution of land use can impact water yield assessment. Although total water yield changes were relatively small, this variation could be explained by the accountability of different land-use changes over the years. According to Cui et al. (2021), the impact on ecosystem services, including water yield, can be explained to a certain extent by land land-use change. Different land-use types imply changes in water infiltration, runoff, plant canopy, among others, leading to variation in water balance components. Therefore, controlling LULC change is essential to quantify water resources for long-term water management.

5 CONCLUSION

This study aimed to examine the calibration process and the responses of water balance components to historical land use and land cover changes in the GRB utilizing the SWAT hydrological models and land-use update module. The results showed that (1) the land use in the past 12-years has significantly evolved in the Guaporé watershed, with the expansion of agricultural areas, mainly soybean crop fields, and a decrease in forest and pasture areas. Therefore, the variation in the spatial pattern of the land use classes reshape the landscape and can affect the hydrological process, consequently water assessment. (2) The dynamic land-use also increases the landscape's complexity over time since changes in spatial pattern result in a new configuration of land use, soil, and slope classes. Consequently, new HRUs were incorporated into the model by activating the LUP module. (3) The satisfactory performance during the calibration and validation process showed the capability of both scenarios, SLUC and DLUC, to simulate historical discharge. (4) Although, the difference in parameters constrained range values between SLUC and DLUC to indicate the influence of land-use distribution overtime on the calibration process. Thus, to a more realistic representation of field condition model performance analysis should be carried out with a plausible set of parameters ranges; and (5) water balance components are affected by the land-use representation since different land-use types imply changes in water infiltration, evapotranspiration, runoff, and plant canopy.

Overall, the results of this study showed that the DLUC scenario has the ability to simulate the impacts of LULC on historical discharge. However, although both methods produced a satisfactory model performance, parameters constrained varies in SLUC and DLUC. Further, water balance components were affected by spatial and temporal changes in land use. In this regard, future research is needed to investigate the extent of the impact of LULC for a more accurate assessment of water services.

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APPENDIX – A

Table 1: Results of t-tests between comparisons of DLUC and SLUC scenarios for performance variables NSE, KGE and PBIAS during the calibration and validation procedures.

Stage	Variable	Scenario	Scenario	statistic	df	P-value	P-value adj
Calibration	NSE	DLUC	SLUC	-61.293	49	5.57E-48	5.57E-48
Validation	NSE	DLUC	SLUC	-4.203	49	0.000111	0.000111
Calibration	KGE	DLUC	SLUC	-6.441	49	4.87E-08	4.87E-08
Validation	KGE	DLUC	SLUC	-4.67	49	2.37E-05	2.37E-05
Calibration	PBIAS	DLUC	SLUC	23.2	49	4.43E-28	4.43E-28
Validation	PBIAS	DLUC	SLUC	25.197	49	1.05E-29	1.05E-29

p value < 0.05 indicate significant differences between comparisons of DLUC and SLUC.

Table 2: Results of t-tests between comparisons of DLUC and SLUC scenarios for mean annual surface run-off (SURQ), baseflow (GWQ), soil water content (SW), evapotranspiration (ET) and water yield (WY).

Variable	Scenario	Scenario	Statistic	df	p	p adj
WYLD	DLUC	SLUC	-23.577	399	1.30E-77	1.30E-77
ET	DLUC	SLUC	25.883	399	2.10E-87	2.10E-87
SURQ	DLUC	SLUC	41.136	399	1.32E-145	1.32E-145
GW_Q	DLUC	SLUC	5.141	399	4.28E-07	4.28E-07
SW	DLUC	SLUC	113.506	399	8.06E-306	8.06E-306

p value < 0.05 indicate significant differences between comparisons of DLUC and SLUC.

OVERALL CONCLUSION

Although not all potential impacts were considered to portrayal the land-use changes and their effects on landscape and water-related service, some key messages can be extracted from this study:

Land-use changes in the past two decades have been dynamic in Guaporé Watershed. The anthropogenic activities in GRB promoted a widespread reduction of natural areas. Despite the forest loss, the number of fragments decreased, which indicate the extinction of small fragments. We also identified the anthropic matrix as the main driver of landscape dynamics. Therefore, the configuration of the landscape changes with changes of use within the anthropic matrix.

There are pieces of evidence that drivers of land-use change operate distinctly in the watershed. In the North, the extent and dynamic of natural areas differ from those in the South, reinforcing the importance of a systemic view of the watershed. This also highlights the importance of different management strategies to increase the quality of rural landscapes. Due to the configuration and spatial pattern of the watershed, landscape management through land-sharing/land-sparing is a viable strategy to promote more sustainable agriculture in the watershed, reconciling natural areas with agricultural development.

The rural areas play an essential role in conserving natural resources since most natural ecosystems are within private properties. Therefore, the current LULC changes in GRB are affected mainly by the local dynamics of agro-pastoral land uses' expansion and retraction. However, land transformation is a complex process with multiple factors interaction. Thus, to fully understand the process of land-use change, we must also consider the effects of proximate causes and underlying driving forces.

Based on the drivers of land-use change, our models show how the ongoing land-use change process in the watershed may lead to further deforestation and simplification of a natural ecosystem. In addition, the results show replacing older, mature forests with younger and less biodiverse forests may impact nature`s contribution to people.

Additionality, the variation in the spatial pattern of the land use classes reshape the landscape and affect the hydrological process. The simulation of historical discharge based on static and dynamic land-use inputs highlighted the importance of considering temporal changes in land use. Yet both SLUC and DLUC scenario has the ability to simulate the impacts o LULC on historical discharge, water balance components, such as evapotranspiration (ET), soil water content (SW), and surface runoff (SURQ) are greatly influenced by the temporal changes in land use distribution. In this regard, future research is needed to investigate the extent of the impact of LULC, for a more accurate assessment of water services.

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