University of São Paulo
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Dynamic modeling of native vegetation in the Piracicaba River basin and its effects on ecosystem services

Paulo Guilherme Molin

Thesis presented to obtain the degree of Doctor in Science. Program: Forest Resources. Option in: Forest Ecosystem Preservation

Piracicaba
2014

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versão revisada de acordo com a resolução CoPGr 6018 de 2011

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DEDICATION

To my parents,

José Paulo Molin *and* **Lourdes Rech Molin***, for their eternal love, patience and support.*

To them,

I lovingly dedicate this work.

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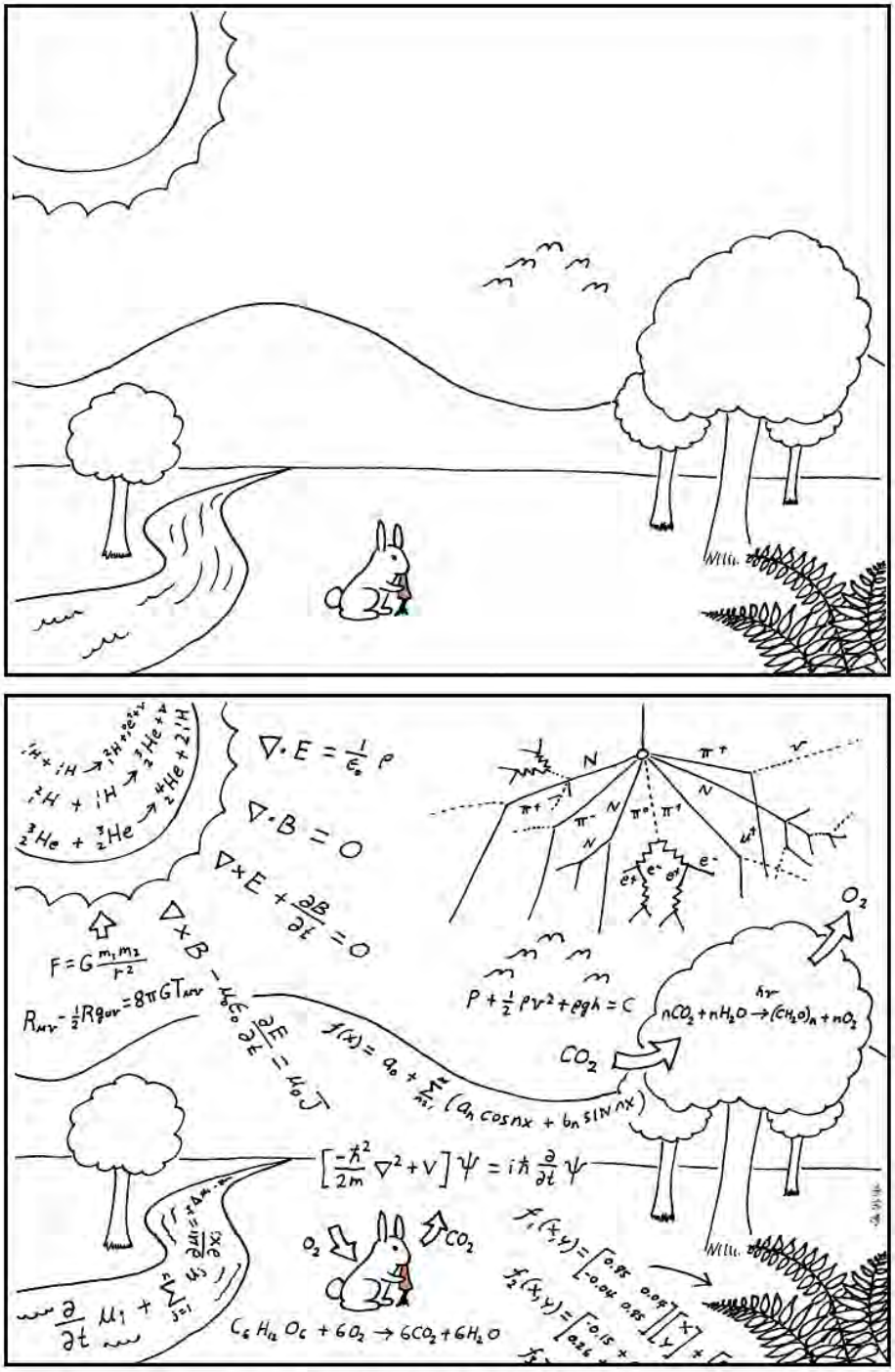
EPIGRAPH



This is how scientists see the world.

– Unknown author

SUMMARY

RESUMO

Modelagem da dinâmica de vegetação nativa na bacia do Rio Piracicaba e seus efeitos na oferta de serviços ecossistêmicos

Levantamentos do Instituto Florestal de São Paulo têm mostrado que no final do século XX a cobertura florestal nativa total do estado atingiu um patamar de perda e que se iniciou então um período de expansão. Rigidez de leis, fiscalização, benefícios econômicos, além de pressão social demonstrados nos últimos anos têm contribuído para essa expansão da vegetação nativa em certos locais. Este estudo propôs modelar a dinâmica da cobertura florestal nativa na bacia do Rio Piracicaba (12.500 km^2), localizada no Estado de São Paulo, para averiguar os possíveis efeitos dessas mudanças nos serviços ecossistêmicos ligados à vazão e regulação de rios, além da própria estrutura da paisagem simulada, interligando-se com biodiversidade e habitat, promovidos pelos remanescentes florestais. Para atingir a proposta estabelecida nesta pesquisa, modelos de dinâmica da vegetação nativa foram desenvolvidos. Foram utilizados mapas temáticos de cobertura e uso do solo dos anos 1990, 2000 e 2010 originados a partir de imagens Landsat 5 TM. Com auxílio do software Dinamica EGO, especializado em modelagem da paisagem, criaram-se três modelos espaciais e temporais da dinâmica florestal, levando em consideração os cenários s*tatus quo (SQ), no deforestation (ND)* e *riparian restoration enforcement (RRE)*. Uma análise usando pesos de evidência foi utilizada para identificar as variáveis de transição florestal. As variáveis foram divididas em dois grupos, (1) físicas e ambientais, consistindo de tipos de solo, rede de drenagem, pluviosidade e presença de fragmentos florestais e (2) antrópicos, consistindo de densidade populacional, produto interno bruto, rede viária, zonas urbanas e predominância de atividade rural. Os cenários resultantes foram analisados por métricas de paisagem para fim de comparação e qualificação dos fragmentos em relação a sua estrutura, interligando-se aos serviços ecossistêmicos de suporte. Por último, foi realizada uma modelagem hidrológica usando o modelo *Soil & Water Assessment Tool* (SWAT) para averiguar a influência da mudança florestal na regulação de vazão de rios e portanto comparar os cenários em relação aos seus efeitos sobre serviços ecossistêmicos de regulação interligados à água. Resultados mostraram que transição florestal ocorreu, passando a cobertura florestal de 24,4% em 1990 para 20,1% em 2000 e então 21,8% em 2010. Cenários resultaram em uma cobertura florestal de 22,4% (SQ), 43,2% (LE) e 28,4% (RRE) para o ano de 2050. A perda de floresta foi identificada como produto de variáveis de natureza antrópica enquanto o ganho florestal foi de variáveis físicas e ambientais. Regiões com melhores condições ambientais resultaram em melhores valores de estrutura da paisagem. SQ foi afetado principalmente pela perda de pequenos fragmentos florestais que funcionam como conectores estruturais da paisagem, potencialmente afetando a biodiversidade e habitat. O deflúvio médio anual foi reduzido em até 10,3% com o incremento florestal observado em ND. Conclui-se que a cobertura florestal na paisagem e os cenários propostos afetam o deflúvio, regulação e a estrutura da paisagem, nos permitindo discutir nas diferenças entre cada cenário e a relação entre dinâmica florestal, estrutura da paisagem, hidrologia e potenciais efeitos nos serviços ecossistêmicos de suporte e regulação.

Palavras-chave: Mata atlântica; Transição florestal; Determinantes espaciais; Modelagem da paisagem; Modelagem hidrológica

ABSTRACT

Dynamic modeling of native vegetation in the Piracicaba River basin and its effects on ecosystem services

Studies from the Forestry Institute of São Paulo State have shown that in the end of the 20th century, the native forest cover of the state of SP reached the maximum level of forest loss. From that point on, a period of forest increase and expansion started. Industrialization, law enforcement, economic benefits, and social pressure experienced in recent years are believed to be contributing to the preservation and regrowth of the native vegetation cover in certain locations. This study proposed to model the dynamics of native vegetation cover in the Piracicaba River basin (12,500 km²) in the state of São Paulo, Brazil, to evaluate possible effects of these changes in ecosystem services related to river flow & regulation and landscape structure, linking to biodiversity & habitat supported by forest patches. To achieve the proposal set out in this research, dynamic models of native vegetation were established. Thematic land cover maps of the years 1990, 2000 and 2010, originated from Landsat 5 TM images, formed the spatiotemporal basis of this study. With the aid of Dinamica EGO (a dynamic modeling software), three future scenarios were created, called *status quo (SQ), no deforestation (ND)* and *riparian restoration enforcement (RRE)*. An analysis using weights of evidence was done to identify forest transition drivers. The drivers are divided into two groups, (1) environmental & physical, consisting of soil types, hydrographic network, rainfall and presence of native forest fragments and (2) anthropic, consisting of population density, gross national product, road network, urban patches and predominant rural activities. Resulting scenarios were analyzed by means of landscape metrics to compare and qualify vegetation patches in relation to structure as proxy for supporting ecosystem services. Finally, Soil & Water Assessment Tool (SWAT), a hydrological model, was used to determine the influence of different forest scenarios in mean annual water yield and regulation processes throughout the basin, and, therefore, compare scenarios as to effects on regulating ecosystem services. Results show that forest transition is indeed occurring, with native vegetation cover parting from 24.4% in 1990, to 20.1% in 2000 and 21.8% in 2010. Scenario results were of 22.4% (SQ), 43.2% (ND) and 28.4% (RRE) for 2050. Forest loss was identified as a product of anthropogenic drivers while regrowth was of physical & environmental drivers. When the area was segmented, regions with greater environmental condition resulted in improved values of landscape structure. SQ scenario was the most affected, losing small patches of forest that could function as structural connectors, and therefore potentially affect biodiversity and habitat. Mean annual water yield was reduced with forest regrowth by as much as 10.3% in ND. We concluded that the dynamics occurring in the landscape and the proposed scenarios affect mean annual water yield, regulation and landscape structure, allowing us to discuss differences between the scenarios and the relation between forest dynamics, landscape structure, hydrology and overtime potential effects over regulating and supporting ecosystem services.

Keywords: Atlantic Forest; Forest transition; Spatial drivers; Landscape model; Hydrological model

1 INTRODUCTION

1.1 Contextualization

Rapid development and historic colonization trends in human society are known for their influence on landscape changes, with even higher indices after the industrial revolution (ELLIS et al., 2010). Current macro changes are more observed in developing countries, focused on agricultural expansion and rural exodus (LAMBIN; MEYFROIDT, 2010, 2011), but can also be seen at smaller scales where infrastructure improvements are made (DEFRIES; KARANTH; PAREETH, 2010) or deteriorating mining occurs (WICKHAM et al., 2007). These changes in land use and land cover are especially harmful to natural ecosystems where impacts on ecosystem services such as forest cover, water regulation, water quality, biodiversity, habitat for wildlife and so many others can be observed (GUO; XIAO; LI, 2000; NELSON et al., 2009; FERRAZ et al., 2014).

Ecosystem services are defined as the benefits people obtain from different ecosystems. They may include a variety of products and benefits such as food, fuel and fiber, and climate regulation, pest control and spiritual or aesthetic benefits (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005). As stated by De Groot; Wilson and Boumans (2002) the first references to the concept of ecosystem functions, services and economic value date back to mid-1960s and early 1970s (ODUM; ODUM, 1972), but until today, despite the increase in research (DE GROOT et al., 2010a; GÓMEZ-BAGGETHUN et al., 2010; POWER, 2010; RAUDSEPP-HEARNE; PETERSON; BENNETT, 2010; BARBIER et al., 2011; ISBELL et al., 2011), frameworks for integrated assessment and valuation remain abstract.

Ecosystem services are “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life” (DAILY, 1997), or, in the words of Kremen (2005), “the set of ecosystem functions that are useful to humans”. In an ecological context, “function” is considered as a “process” and thus “ecosystem function” is a term that considers all stocks of materials, energy and matter that may be processed and exchanged between trophic levels and the environment (GOLDSTEIN, 1999). According to explanations above, at the moment human life benefits from ecosystem functions, there is a “service” being provided.

The concept of ecosystem service emerges also as a way of documenting ecosystem values for mankind by analyzing the benefits provided by natural resources (COSTANZA et al., 1997; DE GROOT; WILSON; BOUMANS, 2002; KREMEN, 2005; MILLENNIUM ECOSYSTEM ASSESSMENT, 2005) but also by valuing processes within natural cycles that operate in endless time and space scales, and generate these benefits (DAILY, 1997). Despite having used these terminologies since the 1960s, it was only in the 1990s that ecosystem services, functions and values were fully accessed (COSTANZA et al., 1997; DAILY, 1997). These discussions led to the creation of the Millennium Ecosystem Assessment (MEA), a requirement made by the United Nations (UN) General Secretary in the year 2000, composed by 1,300 specialists who generated innumerous publications describing conditions and tendencies of the largest ecosystems on Earth, their potential to provision services, options for restoration, conservation and sustainable use (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005; HAINES-YOUNG; POTSCHIN, 2010).

Debates on integrated assessment, along with terminology for practical use, are occurring until this day, but there are agreements to some key questions. First, it is agreed that there is a structure describing relations between ecosystem properties, functions, processes and benefits (DE GROOT et al., 2010a; HAINES-YOUNG; POTSCHIN, 2010). Secondly, ecosystem services are strictly related to the point of view of human benefits, thus ecosystem functions become services once man benefits from them (FISHER; KERRY TURNER, 2008; HAINES-YOUNG; POTSCHIN, 2010). Despite this agreement, the fact that structure and relations can be extensive and have many intermediate steps, they encompass a problem for valuating and strictly separating what is function and what is service (DE GROOT et al., 2010b). In some concepts, a long classification structure or framework connecting processes, functions and structures may provide difficulties for interpreting what part of it is actually a service, after all, one is essential to the other (DAILY, 1997; FISHER; TURNER; MORLING, 2009). Also, if one was to value these processes, functions and structures, there should be strict criteria for classification so that no dual valuation is made (FISHER; KERRY TURNER, 2008; DE GROOT et al., 2010b).

Differences in classification are context specific and usually reflect direct user procedures (FISHER; KERRY TURNER, 2008). In this study, we use a traditional classification system for services that is based on categorical ecosystem functions

(DE GROOT; WILSON; BOUMANS, 2002). Functions are therefore divided into five categories:

Regulation – maintenance of essential ecological processes such as climate and water;

Habitat – places designated for maintenance of biodiversity such as nurseries, refuges and feeding zones;

Support – tangible elements which allow life to exist, such as water and oxygen;

Production – supplies of natural resources such as food and raw material;

Information – ability of an ecosystem to provide cultural and scientific data.

As mentioned before, once these functions serve benefits for man, they become services, that are divided into four classes (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005):

Provisioning – direct suppliers of goods obtained from ecosystems, such as food, water and genetic resources;

Regulating – responsible for regulating ecosystem processes such as climate, water and river flow;

Supporting – required for the making of all other ecosystem services, such as soil, nutrients and primary production;

Cultural – supply of intangible benefits such as recreation and education.

Considering that classification structures can be extensive, some authors suggest that some services can be measured as intermediate, rather than just final. If we consider that a function is essential for the provision of a service, than, this function can be interpreted as an intermediate service (FISHER; TURNER; MORLING, 2009).

Also of importance is the quantification of ecosystem services and the relation between what is actually measured, which can be sometimes a function or a structure (LIU et al., 2010). Quantification or value of ecosystem services can be of economic or non-economic measurements. De Groot, Wilson and Boumans (2002) divide and group these measurements into three categories: economic value, socio-cultural values and ecological values. The first category usually gives monetary importance, and different techniques can be used to apply a price, given market production values of similar services (DE GROOT et al., 2010b). A socio-cultural category implies that people attribute non-economic value to ecosystem services when these influence their well-being, and provide religious, spiritual, and health benefits. Despite being more difficult to measure, some methods of quantification are

already present as it is the case of the Human Development Index (DE GROOT et al., 2010b). The third category implies that services have ecological values and that they can be measured and evaluated trough structure and biophysical metrics, as proxies for these services, with potential further use as alternatives for economic and social measurements as well (DE GROOT et al., 2010b).

One way of measuring ecological values is by quantification of fluxes or qualitative comparison of the increase or decrease of certain natural landscapes. As happens to many ecological processes, direct measurements are sometimes replaced by ecological indicators, which are considered essential for their occurrence. Ecological indicators link biodiversity, properties, functions, services and human well-being and therefore may be used as proxies (DAILY, 1997; DE GROOT et al., 2010a, 2010b).

With this context in mind, we address in this research the effects of a changing landscape, focused on native vegetation (forests), and its contribution to support and regulating ecosystem services related to habitat & biodiversity and water yield & regulation, respectively.

Among the supporting ecosystem services, which are services required for the making of all other ecosystem services, we highlight in this study, native vegetation cover and its importance and need for the existence of other services of regulating and provisioning nature. Supporting services are exemplified here as a structure desirable for biodiversity maintenance and habitat integrity but could also be related to more specific maintenance such as genetic bank protection or soil formation, photosynthesis, primary production and so on (DE GROOT et al., 2010a). An excellent environment, with efficient supporting services, will therefore provide provisioning and regulating services.

Provisioning services, despite not being the focus of this study, are the result of existing supporting services. Examples of these services are usually related to products extracted from native vegetation, such as food and wood (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005).

Regulating services, on the other hand, are here exemplified as the regulation of water yield, but could also be related to other regulations such as carbon stock, climate change and some nutrient cycling (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005).

Since it is difficult to achieve direct measurement and qualification of these services, we used specific proxies throughout the study. Supporting services, exemplified by

the existence of native vegetation, maintenance of biodiversity and habitat integrity are here measured with the aid of landscape structure metrics. Regulating services, in existence due to the support given by native vegetation, are exemplified by water yield and measured using hydrological indicators combined with hydrological modeling.

The examples presented here are clear current topics of great importance in the State of São Paulo, Brazil and especially in the Atlantic Forest region (RIBEIRO et al., 2009; TEIXEIRA et al., 2009; CARAM, 2010; GONZALEZ, 2010; DA SILVA, 2012; FARINACI; BATISTELLA, 2012; FERRAZ et al., 2014).

We selected the Piracicaba River basin, as our study region for its historic landscape changes but also because of its regional economic and environmental importance.

Located mostly in the Brazilian state of São Paulo (90%), and 10% in the State of Minas Gerais, the basin comprehends an area of around 12,500 km^2, and includes 61 municipalities, of which 55 are fully inserted in the basin. Currently, 93% of the 3.1 million inhabitants live in urban centers, making it one of the most populated basins in Brazil (INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA - IBGE, 2010).

As for forest cover, the current situation found in the Atlantic Forest already shows signs of a net gain in some regions, suggesting a forest transition scenario (BAPTISTA; RUDEL, 2006; FUNDAÇÃO SOS MATA ATLÂNTICA; INSTITUTO NACIONAL DE PESQUISAS ESPACIAIS - INPE, 2008; 2009, 2010; TEIXEIRA et al., 2009). Within the state of São Paulo, composed mainly by Atlantic Forest and Cerrado, the tendency of change began in the 90's with stabilization around the year 2000, with forest gain of around 2.82% and later, in 2010, a considerable gain of 95,000 ha (KRONKA; MATSUKUMA; NALON, 1993; KRONKA et al., 2005; INSTITUTO FLORESTAL - IF, 2013). Although differences in estimation and methods may influence these results, most observations suggest a net gain of forests in the State (JOLY et al., 1999; RIBEIRO et al., 2009; TEIXEIRA et al., 2009). The relative impact of environmental versus socioeconomic drivers on these patterns remains unclear.

This basin, as many others located near this region, has suffered over 200 years of historical landscape changes. Occupation and formation of this landscape is a result of optimum environmental and physical conditions, which allowed agricultural development, and strategic location, which served as a passage for early trade routes between the states of Paraná, São Paulo, Minas Gerais and Goiás. By the

end of the 18th and beginning of the 19th century, the city of Campinas changed from a small village to the condition of the agricultural capital of the state due particularly to coffee plantations. In the early 1930s, even with the coffee economy crises, the Campinas region had already established an industrial structure and was able to partly overcome the disastrous economy through the introduction of other crops and segmentation of existing properties (CENTRO TECNOLÓGICO DA FUNDAÇÃO PAULISTA DE TECNOLOGIA E EDUCAÇÃO - CETEC, 2000). Coffee production brought also infrastructure to the region, especially through the construction of railways and roads for production flow to the capital of São Paulo and later to the port of Santos for exportation (SEMEGHINI, 1991).

Presence of infrastructure, industrialization and the decline of coffee plantations led to other plantation systems, specifically sugarcane and later to orange and *Eucalyptus spp*. Regional agroindustry is today internationally known for sugar, ethanol and concentrated juice production especially in the Piracicaba and Limeira municipalities and fruit, dairy products, poultry and swine in the Atibaia, Jundiaí, Vinhedo and Bragança Paulista municipalities. Other featured raw material processing are pulp and paper, leather and textile (CETEC, 2000).

Agriculture, which has moved from strict coffee plantations to other crops, is nowadays predominantly large scale industry owned sugarcane plantations, mostly for production of sugar and ethanol. This activity is today most common in the West side of the basin, in the Piracicaba, Corumbataí, and some lower parts of the Jaguarí and Camanducaia sub-basins. At the Eastern portion, pasture land is predominant, mostly composed of small, family owned properties, mainly for dairy production. Agricultural zoning is a result of climatic and physical characteristics of the region, with great presence of sugar-cane present in the lower, flat and mechanized lands. Smaller in quantity but still as important, are other land uses such as coffee, orange and *Eucalyptus spp.* plantations and horticulture. These last land uses are scattered throughout the basin, but are locally concentrated (IRRIGART, 2004; EMBRAPA, 2014). Although less representative, they are, in some cases, important international commodities and follow strict environmental certification, commonly recognized as environmentally friendly production systems to native vegetation (ALVES; JACOVINE; SILVA, 2011; PINTO; McDERMOTT, 2013).

Although it has its economical, industrial and commercial importance, the basin also has environmental significance due to the presence of significant Cerrado and

Atlantic Forest remnants and all the ecosystem services that it provides such as the supply of fresh water to the region and especially to the city of São Paulo, through the Cantareira Transposition System (WHATELY; CUNHA, 2007). The construction of this system started in 1966 and, by 1976, the Cantareira System had reached a flow of 22 $m^3 s^{-1}$. Today it reaches around 31 $m^3 s^{-1}$, and provides water to 8.8 million people, mostly in the metropolitan area of São Paulo (COMPANHIA DE SANEAMENTO BÁSICO DO ESTADO DE SÃO PAULO - SABESB , 2014).

The recent economic development, followed by urban development, has created diverse environmental drawbacks, especially related to hydrological resources and demand for water, a tendency that will remain according to future projections (AGÊNCIA DAS BACIAS-PCJ, 2011).

1.2 Questions that guide this study

Given this context, this research focused on a series of questions that follow.

1) Is there forest transition in the Piracicaba River basin?

2) What are the main environmental & physical and anthropogenic drivers for forest suppression and forest regrowth?

3) What are the future projections for forest cover?

4) How will these projections affect landscape structure, measured as proxy for supporting ecosystem services?

5) How will these projections affect mean annual water yield and mean daily water yield, measured as proxies for water regulating ecosystem services?

The hypotheses that guided this research are given bellow.

- Forest transition is occurring in the basin;
- Forest loss is moreover related to anthropogenic characteristics while forest regrowth is furthermore linked to physical conditions;
- Future forest cover should indicate growth patterns in specific regions;
- Landscape structure, mean annual and daily water yield will be affected with forest increase;

1.3 Objectives

We state bellow the main objectives, while detailed objectives are specified in each individual chapter.

1) Analyze native forest cover dynamics in the Piracicaba River basin
2) Analyze native forest cover and how its quantitative and spatial projection influence ecosystem services related to landscape structure and water yield.

1.4 Thesis structure

First of all, this thesis is being written in English due to the internationalization process that this research has achieved during its development. Throughout its development stages, we have gained attention and assistance from four universities around the world. The Federal University of Minas Gerais, through the Center for Remote Sensing, headed by Dr. Britaldo Silveira Soares-Filho, was the first partnership established early in 2011 which resulted in the assistance for the procedures of using the Dinamica – Environment for Geoprocessing Operations (Dinamica EGO) modelling software. Later, early in 2013, a partnership was stablished between us, doctoral student Danielle Bressiani, from the University of São Paulo – São Carlos School of Engineering, a specialist in Soil & Water Assessment Tool (SWAT) modelling software and Dr. Raghavan Srinivasan, professor and director of the Spatial Sciences Laboratory at Texas A&M and also author of SWAT model. Both Bressiani and I visited Texas A&M in 2013, where we established procedures for the use of SWAT in our research, under the supervision of Dr. Srinivasan. Bressiani was therefore responsible for the calibration and validation of the model for the Piracicaba River basin while I was responsible for the production of land cover maps to be used in this procedure and later the combination of the calibrated & validated model along with scenario simulations. Also, later in 2013, a partnership was established with Dr. Sarah Gergel, from the University of British Columbia, at Landscape Ecology Lab. Chapter two of this research was written under her supervision during my visit, with the goal of achieving a peer-reviewed publication.

This first chapter is a contextualization of topics introduced throughout the following chapters, containing also a study area contextualization, key questions, hypotheses and main objectives to which the research was developed around.

Chapters two and three of this thesis were written and structured for the goal of achieving peer-reviewed publications. They both contain individual introductions with elementary theoretical reference and objectives as well as methodological

procedures and results, specific to the themes and are therefore formatted as if submitted for review. As of today, part of chapter two has already been submitted and approved by the journal Scientia Forestalis, comprehending the results acquired from the remote sensing procedures. The subsequent publications should be sent, in the following days, to international journals, comprehending the additional results, separated in the described chapters.

The fourth and last chapter summarizes all results and discussion into answers for the key questions described in chapter one, while meeting the stipulated objectives and hypotheses.

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2 ATLANTIC FOREST TRANSITION IN THE PIRACICABA RIVER BASIN: SYNERGISTIC DRIVERS AND PREDICTIONS

Abstract

Despite persistent suppression, forest cover has shown some increase in specific locations across the world. Atlantic Forest, in Brazil, already shows signs of a net gain, suggesting forest transition. Studies on influence of economic factors and environmental legislations on forest gain are in a spotlight, but some suggest that land abandonment may be a leading cause. Here we examine (i) the role of selected environmental & physical versus anthropogenic factors in driving forest dynamics, (ii) which forest ages are targeted for suppression and (iii) what future forest age mosaic we can expect (more forest cover and full forest maturity?). Three land cover maps dated 1990, 2000 and 2010, elaborated from Landsat 5 TM images, using a method of supervised classification, were used throughout the study. We selected six environmental & physical and six anthropogenic independent drivers to be analyzed by means of weights of evidence, using Dinamica EGO software. Results show (i) that forest regrowth is influenced by multiple drivers, working in synergy. Environmental & physical variables were more related to forest gain while anthropogenic were associated with forest loss. There is a clear pattern of regrowth on pasture and sugarcane plantations when near rivers & forest patches, on steeper slopes and with sufficient rainfall. This observation is linked to land abandonment and resilience potential, due to economic situation of small family-owned pasture land and mechanization of industry-owned sugarcane plantations and to the neighboring conditions; (ii) suppression has targeted both older and newer forests, (iii) future projections reveal that forest gain may come in a slow pace, followed by specific ecosystem services losses or lag in increase, due to continuous trends of older mature forest loss.

Keywords: Atlantic Forest; Forest transition; Driver synergy; Forest regrowth; Forest suppression; Sugarcane; Pasture

2.1 Introduction

Human-driven deforestation and general land use changes, threaten many environmental and social aspects of mankind. Climate regulation, water regulation, erosion control, pollination and recreation are just a few examples of ecosystem services provided and regulated by forests, that can be lost or devalued in cases of high deforestation rates (COSTANZA et al., 1997).

Scientific literature is rich in case studies of causes and processes of forest cover change over different time periods and specific regions. Human activities and actions such as agricultural expansion are frequently identified as proximate causes for deforestation (BAWA; DAYANANDAN, 1997; FERRAZ et al., 2005; RIBEIRO et al., 2009; DEFRIES et al., 2010). On the other hand, the underlying forces can be related

to human dynamics and policies, thus, having an indirect impact on different levels of deforestation. The drivers are indeed of various levels and multiple sources but have also been observed as synergies, acting together in underlying and proximate levels (GEIST; LAMBIN, 2002).

Despite persistent deforestation in some regions, forest cover has also been increasing in specific locations across the world, most commonly reported in first world countries but not limited to them (BAPTISTA; RUDEL, 2006; MEYFROIDT; LAMBIN, 2011). In some cases, abandonment of former agricultural and pasture lands has led to spontaneous regeneration and active planting of new forests. Even though these new forests can increase some ecosystem services on former agricultural lands, they may take more than a life time to match the original forest cover, especially regarding composition, structure, carbon, biodiversity and hydrological values (CHAZDON, 2008; BROWN; ZARIN, 2013). Because plant diversity is important to maintain ecosystem services (ISBELL et al., 2011), it is possible to enforce that primary forests maybe matchless for supporting tropical biodiversity (GIBSON et al., 2011).

Currently, time and effort have been applied to studies and simulations of preservation of old standing forests and also restoration of degraded landscapes. All of which, with the underlying importance of interventions to restore biodiversity, ecological functioning but even more, tangible ecosystem services such as water provision and climate mitigation (LAMB; ERSKINE; PARROTTA, 2005; SOARES-FILHO et al., 2006; TEIXEIRA et al., 2009). Findings for Latin America have shown that emphasis has been put mainly on carbon and water related services, almost always with payments in mind (BALVANERA et al., 2012).

The current situation found in Brazil reveals that the Amazon forest is undergoing steady deforestation (INPE, 2013) while the Atlantic Forest already shows signs of a net gain in some regions, suggesting a forest transition scenario (BAPTISTA; RUDEL, 2006; FUNDAÇÃO SOS MATA ATLÂNTICA; INPE, 2008; 2009, 2010; TEIXEIRA et al., 2009). At a national level, the rates of deforestation are greater than the rates of regrowth (FAO, 2014). Within the state of São Paulo, composed of Atlantic Forest and Cerrado formations, the tendency of change began in the 90's, reaching stabilization around the year 2000, with forest gain of around 2,82% and later in 2010, a considerable gain of 95,000 ha (KRONKA; MATSUKUMA; NALON, 1993; KRONKA et al., 2005; INSTITUTO FLORESTAL, 2013). Although differences

in estimation, and methods may influence these results, most observations suggest a net gain of forests in the state (JOLY et al., 1999; RIBEIRO et al., 2009; TEIXEIRA et al., 2009). The relative impact of environmental versus socioeconomic drivers on these patterns remains unclear.

Here, we address the following research questions: What is the role of environmental & physical versus anthropogenic factors in driving forest loss and gain? Has recent suppression targeted older mature or younger forests? Based on future projections, what impacts are expected for future levels of older versus younger forests?

We used Landsat data to map forest change over a period of 30 years and link these changes to anthropogenic as well as physical & environmental drivers. Our goal is to determine if the causes of change differ for forest loss versus forest regrowth, and determine if forest loss is concentrated over older or newer forests. Lastly, we explore future projections in order to determine possible levels of maturity of future forests.

2.2 Methods

2.2.1 Study region

We used the Piracicaba River basin, composed of 12,500 km^2, of which 90% are located in the Brazilian state of São Paulo, and the other 10% in the State of Minas Gerais (Figure 1). The geographical location is between the parallels 22º 00’ and 23º 00’ S and the meridians 46º 00’ and 48º 30’ W. According to Mortatti et al. (2004), the Piracicaba River presented a mean annual flow of 140,4 m^3 s^{-1}, for the period between 1944 and 1997.

It is also a composition of five sub-basins: Piracicaba (3,700 km^2); Corumbataí (1,700 km^2); Jaguarí (3,300 km^2); Camanducaia (1,000 km^2); and Atibaia (2,800 km^2), in a total of 61 municipalities, of which 55 are fully inserted in the basin. Currently, 3.1 million people live on this land, making it one of the most populated basins in Brazil (IBGE, 2010).

The basin has significant remnants of Atlantic Forest, most of which are found to the East, near Mantiqueira Sierra. This region is known as an interface between Atlantic Forest and Semideciduous Seasonal Forests, more common to the Southwest of the basin. Further to the Northwest, Cerrado formations are more common. The basin is also covered by four main Environmental Protection Areas (EPA). They are Judiaí

EPA, with 43,200 ha, Cabreúva EPA with 26,100 ha, Piracicaba-Juquerí-Mirím EPA with 107,000 ha and the Cantareira System EPA with 249,200 ha. Terrain contrasts from East to West. Mantiqueira Sierra is located over to the East, at around 1.400 m of altitude, becoming lower and less rugged when heading West. Further to the West we find a basalt formation called Cuesta. In between, terrain is moreover formed by small hills with fine sediment, called Peripheral Depression (CETEC, 2000).

This basin, as many others located near this region, has suffered over 200 years of historical landscape changes (CETEC, 2000). Although it has its economical, industrial and commercial importance, it also features environmental significance due to the presence of Cerrado and Atlantic Forest fragments and all the ecosystem services that it provides such as the supply of fresh water to the region and especially to the city of São Paulo, through the Cantareira Transposition System, providing water to 8.8 million habitants, with an average flow of 31 m^3 s^{-1} (WHATELY; CUNHA, 2007).

The recent economic development, followed by urban development, has created diverse environmental drawbacks, especially related to hydrological resources. Agriculture has had a boom in the entire State in the last decades, resulting in intensification, higher productivity and some activity change, particularly between coffee, pasture and sugarcane. Nowadays, there is a predominance of large scale, industry owned, and sugarcane plantation, mostly for sugar and ethanol production. This activity is today most common to the West side of the basin, at Piracicaba, Corumbataí, and some of the lower parts of the Jaguarí and Camanducaia sub-basins. Over to the East, pasture land is predominant, mostly small, family owned properties, mainly for dairy production. Agriculture zoning is a result of climatic and physical characteristics of the region, being sugarcane heavily present in lower, flat and mechanized lands (IRRIGART, 2004; EMBRAPA, 2014).

Lower in quantity but still as important, are other land uses such as coffee, orange and *Eucalyptus spp.* plantations and horticulture. These land uses are scattered in the basin, but locally concentrated. Although they are less representative, they are, in some cases, important international commodities and follow strict environmental certifications, being commonly recognized as environment friendly production systems (PERFECTO et al., 2005; SIRY; CUBBAGE; AHMED, 2005; BROCKERHOFF et al., 2008; PINTO; MCDERMOTT, 2013).

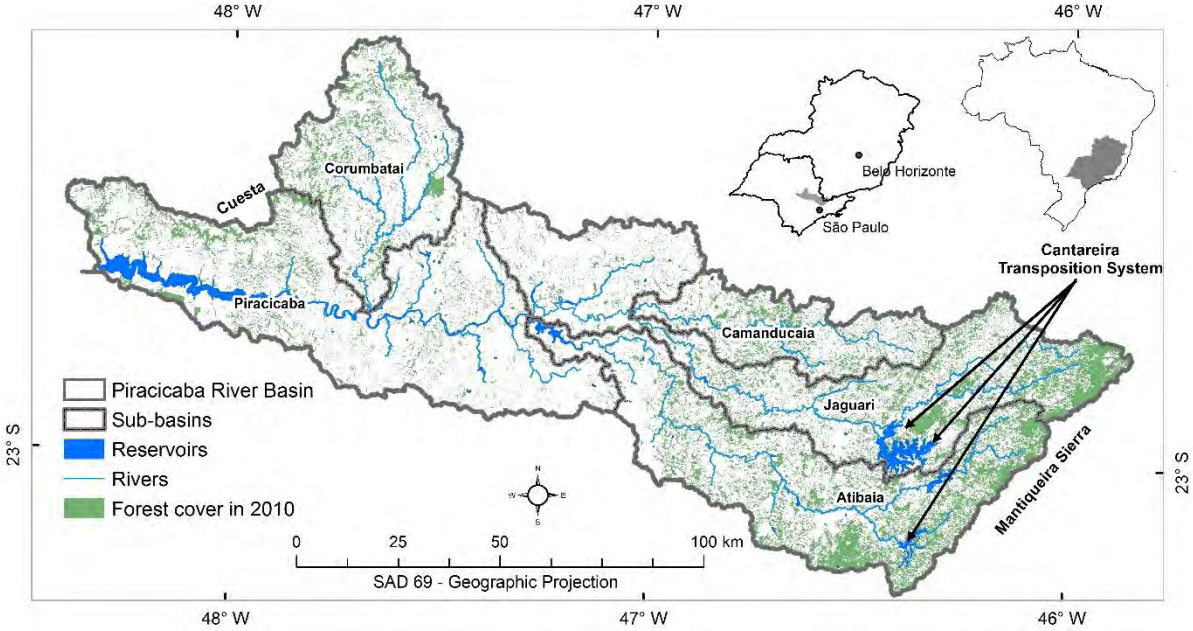

Figure 1 – Location of the Piracicaba River basin and sub-basins in the State of São Paulo, Brazil

2.2.2 Geospatial data

We used a set of three raster land cover maps dated from 1990, 2000 and 2010, elaborated from Landsat 5 TM imagery, using ERDAS imagine 9.1 software. The maps were processed for 1:50,000 scale, with seven classes, including (1) crops, (2) native vegetation, (3) forest plantations, (4) water bodies, (5) pasture land, (6) urban zones and (7) perennial crops, with minimum mapping units of 900 m^2 and pixel size of 30m. Coordinate system was established as *South American Datum 1969*, with *Albers Conical Equal Area* projection. All other geospatial data followed the same coordinate system.

Each of the three maps was a mosaic composition of three individual Landsat 5 TM scenes, with some masking for cloud presence, as seen on Table 1.

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Table 1 – Landsat 5 TM sensor scenes used for the land cover thematic maps elaboration and their respective dates

Scenes	Year	Month
219/76	1990	March
220/75	1990	April
220/76	1990	April
219/76	2000	April
220/75	2000	June
220/76	2000	June
219/76	2010	August
220/75	2010	August
220/76	2010	August
[(a)]219/76	1990	April
[(a)]219/76	2000	August

[(a)] Scenes used for cloud masking

Scene 219/76 from the years 1990 and 2000 presented less than 10% of cloud cover. Clouds were masked and separate scenes from different dates were processed and individually classified to fill the empty spaces.

Image pre-processing

For temporal land cover studies using remotely sensed images, it is necessary to undergo atmospheric corrections in order to remove absorption and reflection effects (CHAVEZ JUNIOR; BERLIN; MITCHELL, 1977; OTTERMAN; ROBINOVE, 1981). We used the Dark Objects Subtraction (DOS) method, which does not take into account the absorption (CHAVEZ JUNIOR, 1988, 1989).

With atmospheric corrections made, the next step was the composition of the six spectral bands. This process was carried out resulting in composite images with the spectral bands 1, 2, 3, 4, 5 and 7.

Geometric correction of all the scenes were performed using the tool AutoSync from ERDAS Imagine 9.1. The images used as reference were of the Global Land Project Survey 2005, which went through plani-altimetric corrections, and therefore, are reliable for this purpose (GUTMAN, 2008). At this stage at least 50 control points were compared between the reference images and the scenes under study, five times higher than recommended by Pouncey, Swanson and Hart (1999), which in total amounted to a residual error of less than or equal to 0.5. lower than recommended by INPE (2005). The model used was a polynomial of third order and resampling method was the nearest neighbor.

Land cover classes

In order to perform a supervised classification, seven classes of land cover were selected, similar to what was seen on Valente and Vettorazzi (2003).

(1) Crops (CR): areas covered with sugarcane, or that showed signs of having been planted with the same crop (e.g. post-harvest waste or exposed soil in large areas) and other annually cultivated crops (e.g. maize and beans);

(2) Native vegetation (NV): areas occupied by different native forest vegetation, from Atlantic Forest to Cerrado formations, containing small and large patches with both degraded, regrowth, young and old growth. Also included in this class were older forest patches that may contain *Eucalyptus spp.* and *Pinus spp.*, provided that they do not present features of commercial plantations (e.g. planting in line, as seen on commercial plantations);

(3) Forest plantations (FP): forest land occupied by *Eucalyptus spp.*, *Pinus spp.* or any other commercial species and must be a plantation, providing evidence such as planting in line or of homogeneous look;

(4) Water bodies (WB): lakes, ponds, reservoirs and large rivers that may have water present at the time when the image was taken;

(5) Pasture land (PL): areas used as ranging land for life stock or simply covered by grass with no evidence of a defined use;

(6) Urban zones (UZ): large urban centers or any other agglomeration of constructions that gives evidence of being urban;

(7) Perennial crops (PC): characterized mainly by fruit growing in commercial plantations (e.g. orange and coffee).

Training samples

The training of each individual scene was performed using segmentation method. To implement segmentation method, two parameters are necessary, namely the threshold of similarity (gray levels that can vary to a spectral Euclidean distance) and the threshold size (for maximum pixel region), being used in this study values of 16 and 625, respectively (OLIVEIRA et al., 2003).

The segmentation algorithm used was growth by region and was chosen the option of eight contiguous neighbors to the "seed" pixel, that pixel chosen for the implementation of the segmentation technique. Each class had 25 training samples distributed throughout the image, accounting for a class sample larger than that

proposed by Quartaroli and Batistella (2006). The algorithm used for the supervised classification was the maximum likelihood.

Despite the use of supervised classification, a simple and effective technique, some classes did not differ between themselves. Automatic classification difficulties were observed with UZ, FP and PC classes, and therefore these were visually sketched with the aid of Landsat 5 images and high resolution images provided by Google Earth for the years 2010 through 2013. UZ class was sketched using the Landsat 5 images as background, because it was visually detectable, making it possible to generate the polygons on a scale of 1:50,000. Sketching using Google Earth as a source has been used and approved by Lopes and Nogueira (2011). In the case of Landsat 5, the interpretation was made with the use of visual cues, such as tone, texture, shape, pattern, and its relationship with other objects to identify different classes.

FP and PC classes for 2010 were digitized using high resolution images as a source, provided by Google Earth, using a compatible 1:10,000 scale. For the years 2000 and 1990 we used the polygons generated in 2010 and projected over the Landsat 5 of each year, thus making a visual comparison with manual adjustments, with both addition and subtraction of areas.

Post classification treatment

Filtering step aimed at removing noise or small groups of pixels that do not belong to a certain class and that were misclassified. This filtering was performed using the Majority 3x3 method, automatically filtered with all classes, except for NV. The exception for NV was due to being common the observation of NV patches, sometimes smaller than the pixel used (30m), on highly anthropic environments. To minimize the extinction of smaller fragments of this class in the final maps, the class was masked and "cut", only to be inserted again after the filtering of other classes, preserving the original classification of only this class.

Finally, a mosaic was produced, merging the three scenes of each year to form the land cover thematic map of the Piracicaba River basin.

Thematic map validation

The process of verifying the accuracy of the classifications made was done so by the random distribution of 250 points per map (POUNCEY; SWANSON; HART, 1999).

For validation, we used the Kappa index together with images from Google Earth and Landsat 5 images as a reference. High resolution satellite images provided by Google Earth has been used successfully in the validation of automatic classifications (CHA; PARK, 2007; DORAIS; CARDILLE, 2011; HOLLER, 2013).

The Kappa index is cited as one of the best procedures used to measure the accuracy of thematic classifications which fully represents the error matrix (LANDIS; KOCH, 1977; CONGALTON; GREEN, 2009).

Other geospatial data

Also, vector and spreadsheet data was obtained from official government databases to be used as variables for explaining drivers of forest loss and gain (Table 2). The selection of variables was based on the availability of information, completeness and scale on which they are provided or found. Variables were divided into environmental & physical and anthropic types, based on their natural characteristic. *Soil type* information was gathered from De Oliveira (1999) and concerned the effect that soil type may apply on forest dynamics, especially forest regrowth. *Distance to water* was obtained from Euclidean distance through Geographic Information System (GIS) operations using local topographic data with hydrological networks, provided by Brazilian Institute of Geography and Statistics (IBGE), with a scale of 1:50,000 (IBGE, 2013). *Distance to forest* was also produced using Euclidean distance and was done so using the proprietary land cover map of the year 2000. *Mean annual precipitation* was obtained from a pluviometric map provided by São Paulo State Department of Water and Energy (DAEE) as seen on Woltzenlogel (1990). *Slope* and *altitude* data were obtained from Advanced Spaceborn Thermal Emission and Reflection Radiometer (ASTER) images, provided by the National Aeronautics and Space Administration (NASA) and the Ministry of Economy, Trade and Industry (METI) of Japan, with a scale of 1:50,000 (NATIONAL AERONAUTICS AND SPACE ADMINISTRATION; METI, 2011). *Total population density*, *rural population density* and *Gross Domestic Product (GDP)* were all obtained for the year 2000 from IBGE population census (IBGE, 2010) and applied to the map using individual municipality vector limits (IBGE, 2013). *Distance to transportation* was acquired through Euclidean distance using provided transportation networks in the scale 1:50,000 (IBGE, 2013). *Distance to urban zones* was also obtained from Euclidean distance using extracted urban data from proprietary land cover map of the year 2000. Finally,

predominant land use was obtained from the 2000 proprietary land cover map and applied to individual municipality vector limits (IBGE, 2013). All variables were transformed and resampled to raster data, with matching coordinate systems, scale and resolutions, found on land cover maps. All variables can be found on Appendix A.

Table 2 – Variables used for explaining native vegetation loss and gain

Variable	Data set	Scale	Resolution (m)	Type	Source
Soil type	Soil map	1:500,000	-	Physical	[(a)]Embrapa/IAC
Distance to water (m)	Hidrology network	1:50,000	-	Physical	[(b)]IBGE
Distance to forest (m)	Land cover	1:50,000	30	Physical	Proprietary
Annual mean rainfall (mm)	Pluviometric map	1:500,000	-	Physical	[(c)]DAEE
Slope (%)	[(e)]ASTER DEM	1:50,000	30	Physical	[(d)]NASA/METI
Altitude (m)	[(e)]ASTER DEM	1:50,000	30	Physical	[(d)]NASA/METI
Total population density (inh. km^{-2})	Census 2000	-	-	Anthropic	[(b)]IBGE
Rural population density (inh. km^2)	Census 2000	-	-	Anthropic	[(b)]IBGE
Gross domestic product (R$)	Census 2000	-	-	Anthropic	[(b)]IBGE
Distance to transportation (m)	Transp. Network	1:50,000	-	Anthropic	[(b)]IBGE
Distance to urban zones (m)	Land cover	1:50,000	30	Anthropic	Proprietary
Predominant land use	Land cover	1:50,000	30	Anthropic	Proprietary

[(a)]Embrapa/IAC is *Empresa Brasileira de Pecuária e Agricultura / Instituto Agronômico de Campinas* or Brazilian Enterprise for Agricultural Research / Campinas Agronomic Institute (DE OLIVEIRA, 1999)
[(b)]IBGE is *Instituto Brasileiro de Geografia e Estatística* or Brazilian Institute of Geography and Statistics (IBGE, 2010, 2013)
[(c)]DAEE is *Departamento de Águas e Energia Elétrica do Estado de São Paulo* or São Paulo State Department of Water and Energy (WOLTZENLOGEL, 1990)
[(d)]NASA/METI is National Aeronautical and Space Administration / Japan's Ministry of Economy, Trade and Industry (NASA; METI, 2011)
[(e)]ASTER is Advanced Spaceborn Thermal Emission and Reflection Radiometer

For processing in Dinamica EGO software (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002), all data must be categorized and therefore, all continuous gray-tone maps were classified. For *distance to water* and *distance to forest* maps, we used regular intervals of 30m, the lowest possible buffers for the scale of our information. *Distance to transportation* and *distance to urban zones* maps were categorized in 1000m and 100m intervals, respectively. *Mean annual precipitation*, *Slope* and *altitude* were categorized into intervals of 100 mm, 5% and 100 m, respectively. *Total population density*, *rural population density* and *GDP* were categorized into intervals of 100 inhabitants per km^2, 10 inhabitants per km^2 and R$10 B, respectively. Soil type and predominant land use were already categorical data and needed no further processing.

Variables were tested for spatial independence using the Crammer coefficient (V), which Bonham-Carter (1994) reports that values under 0.5 suggest less association. We found that all our variables, except for the pairs of *GPD* and *total population density*, were lower than the suggested threshold and therefore are spatially independent. For the continuity of the process, we removed *total population density* from our models.

2.2.3 Statistical analysis

We used the set of three land cover images and the Dinamica EGO software (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002), to build transition matrices and identify the main drivers for native forest gain and loss. Drivers were divided into two groups, environmental & physical and anthropic.

Dinamica EGO uses the weight of evidence (WoE), a modified form of Bayes theorem of conditional probability, as an approach for land cover change model (BONHAM-CARTER, 1994). It produces a transition probability map, representing the most favorable areas for change to occur. WoE represents the influence on spatial probability of a transition for each variable and consists of an independent evaluation of descriptive variables with successive aggregation, where empirical probabilities derived from Boolean map layers are used as inputs to a log odds ratio recasting of Bayesian aggregation (SOARES-FILHO; RODRIGUES; COSTA, 2009). We used resulting WoE to analyze which variables best explain the forest changes that occurred in the 2000 to 2010 period and therefore examine which are the current drivers for forest suppression and regrowth (TEIXEIRA et al., 2009).

To quantify forest net loss and gain, simple GIS calculations were applied to the three land cover maps. NV class was overlaid for each year, generating a subtraction layer that identified loss and gain for each period. Gains were furthermore given an age, in years, which was applied to the next period map. If in the next period there was a loss of forest, it was possible to infer if forest loss was on older existing forests or recent regrowth. All analysis was made both at basin and sub-basin levels.

We later created future projections scenarios using the transition matrix from the 2000-2010 period. Three scenarios were created to observe impacts on future levels of older versus younger forests:

(i) Old Growth-Deforestation, where forest suppression is observed only on older forests;

(ii) Equal-Deforestation, where forest suppression is divided into equal parts between different forest ages;

(iii) No-Deforestation, where no forest suppression is observed

2.3 Results

2.3.1 Land cover thematic maps

Processing of Landsat 5 images resulted in three land cover thematic maps dated 1990, 2000 and 2010 (Figure 2). Individual maps can be found in the Appendix B, Appendix C, and Appendix D, respectively. The final maps are in both raster and vector formats, compatible with a 1:50,000 scale and minimum mapping unit of 900 m^2. Kappa coefficients resulted respectively in 76%, 85% and 85%, or even "good", "excellent" and "excellent", according to the classification of Landis e Koch (1977). Although the year 1990 map obtained a relatively smaller kappa coefficient, values are considered acceptable. This lower value can be justified by the difficulties involved in validation due to the lack of high-resolution images for comparison. Error matrix for all three maps can be found on Annex A.

Analyzing the land cover maps and the basin as a whole, it is possible to observe a diverse occupation, but with certain homogeneity according to altitude and terrain.

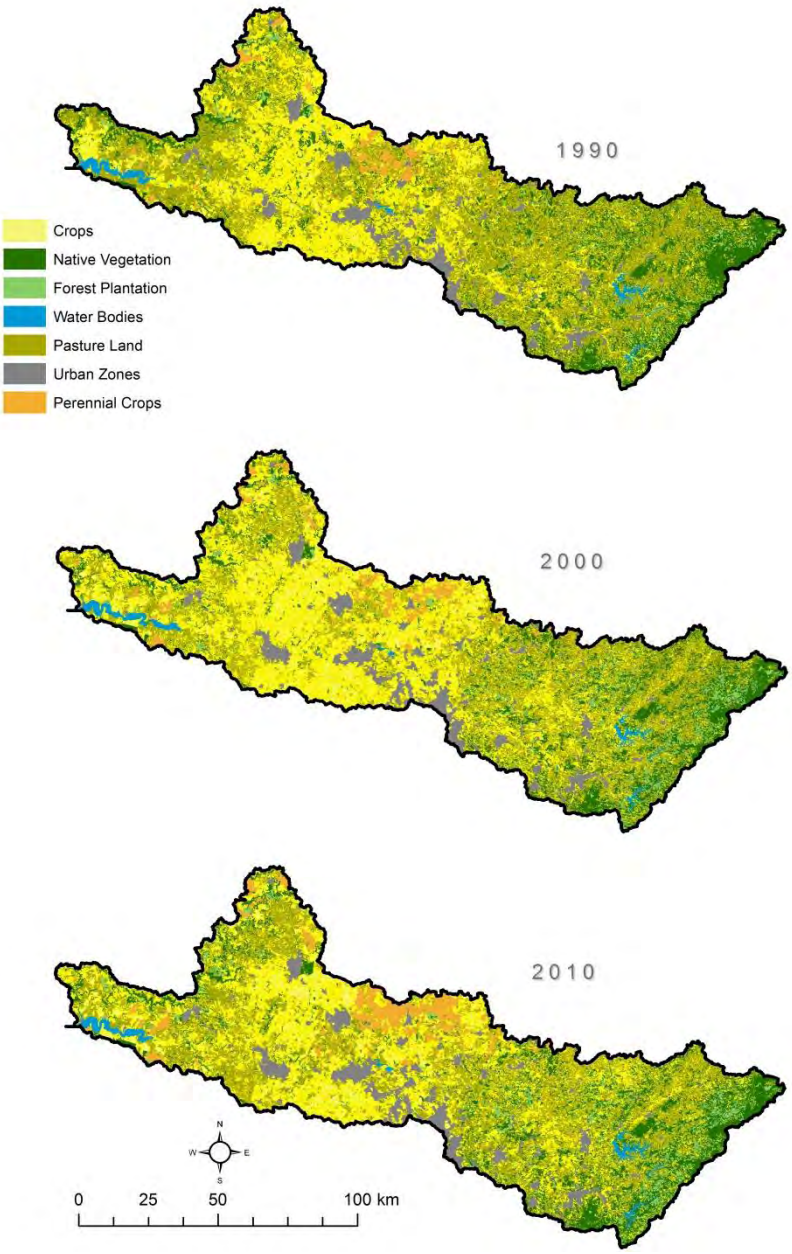


Figure 2 – Land cover thematic map of the Piracicaba River basin for the years 1990, 2000 and 2010

Quantitative analysis of land cover change

The first information gathered from the land cover thematic maps were related to the total area of each class. The distribution for each class in the years 1990, 2000 and 2010 can be seen on Table 3. The same information for each individual sub-basin can be found on Annex B.

Table 3 – Total area and percentage for land use and land cover

Land Cover	1990		2000		2010	
	Area (ha)	%	Area (ha)	%	Area (ha)	%
Pasture land	503,254	40.07	432,195	34.41	422,381	33.63
Native vegetation	307,028	24.44	252,301	20.09	273,217	21.75
Crops	300,836	23.95	392,579	31.26	340,892	27.14
Urban zones	71,904	5.72	83,814	6.67	91,914	7.32
Forest plantations	29,164	2.32	43,597	3.47	53,080	4.23
Perennial crops	24,253	1.93	31,272	2.49	52,144	4.15
Water bodies	18,119	1.44	18,798	1.50	20,854	1.66
TOTAL	1,255,999	100.00	1,255,999	100.00	1,255,901	100.00

PL, here represented mainly by dairy and cattle ranches, but also degraded pastures and abandoned fields, is predominant in the three studied periods, 40.07%, 34.42% and 33.63%, respectively; followed by CR, represented mainly by sugarcane, with 23.95%, 31.26% and 27.14, respectively; and NV lying predominantly at the third position, with 24.44%, 20.09% and 21.75%, respectively.

2.3.2 Overall forest gain and loss

Throughout the Piracicaba basin, we found a clear pattern of forest increase in the period of 2000-2010, with a growth of +1.7 percentage points (pp) as compared to 1990-2000, with -4.4 pp, which also resulted on net loss of forest on all sub-basins. We present in Figure 3, the land cover change observed in the basin along with two distinct examples of observations within sub-basins. Total forest cover for each year and their respective net loss and gain per period are presented on Table 4.

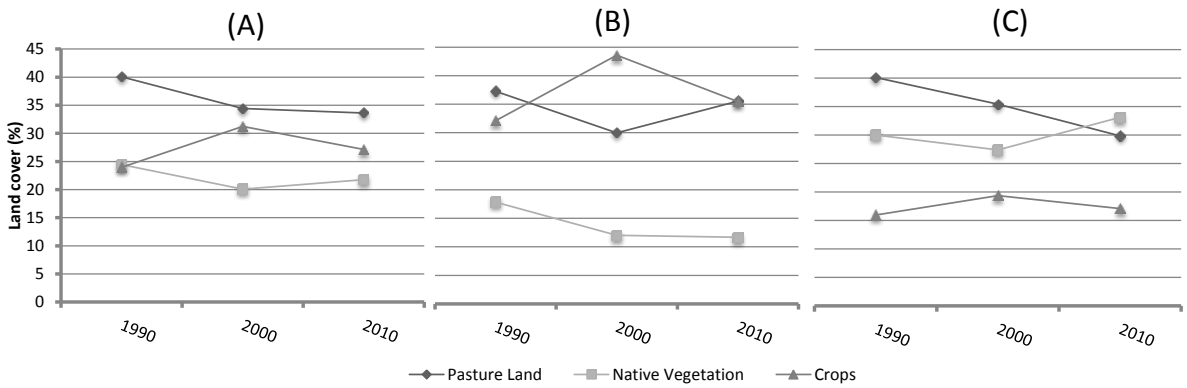


Figure 3 - Piracicaba River basin (A), Piracicaba sub-basin (B) and Atibaia sub-basin (C) land cover change for the years 1990, 2000 and 2010

Table 4 – Forest cover and net forest loss and gain per sub-basin for each studied period

Year/Period	ATIBAIA		CAMANDUCAIA		CORUMBATAÍ		JAGUARÍ		PIRACICABA		TOTAL	
	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%
1990	84213	29.98	26982	25.96	36859	21.60	92792	28.14	66183	17.85	307028	24.44
2000	76882	27.37	24182	23.26	29545	17.32	77173	23.40	44520	12.01	252301	20.09
2010	92900	33.09	24533	23.60	30955	18.14	81525	24.72	43305	11.68	273217	21.75
2000-1990	-7331	[(a)]-2.61	-2800	[(a)]-2.69	-7314	[(a)]-4.29	-15619	[(a)]-4.74	-21663	[(a)]-5.84	-54727	[(a)]-4.36
2010-2000	16018	[(a)]5.71	351	[(a)]0.34	1411	[(a)]0.83	4352	[(a)]1.32	-1216	[(a)]-0.33	20916	[(a)]1.67

[(a)] Values are presented as percentage points, where they equal to the difference of the corresponding dates

2.3.3 Transition matrix & drivers

The model inputs were based on the initial map (2000), the final map (2010), a data file with WoE and a “raster cube” with the spatialized static variable data that could influence change. The first results obtained from the model consisted of landscape transition matrices. This first step was done for both 1990-2000 and 2000-2010 period, for comparison reasons (Appendix E). A summary of the main transitions between NV and other land cover are presented on Figure 4. In general, the Piracicaba River basin (A) presented more NV loss in the first period than in the second. A comparison between the Piracicaba Sub-basin (B) and the Atibaia Sub-basin (C) exemplifies how changes act differently throughout the basin. In B we can observe a force of CR and PL acting on NV, in both periods. While in C, the pressure is of PL, although this weakens in the second period. Also to be noted is the unchanged negative scenario in B of NV becoming CR and PL in both periods. However, in C, we observe a high percentage of FP becoming NV, indicating a possible awareness to legislation by pulp and paper companies. Also, a significant amount of PL also undergoes change to NV in C, followed in the second period with an increase.

The final result, after categorizing all variables, consisted of calculating the WoE for each range, for each variable, allowing us to infer about its contribution over a transition event. Positive values for WoE are associated with transition to forest while negative values imply the opposite (SOARES-FILHO; RODRIGUES; COSTA, 2009). Forest suppression was furthermore observed in specific altitudes, ranging from 400 m to 800 m, near to main roads and urban zones. On the contrary, regrowth was considered a consistent pattern, with a higher probability when on steeper slopes, near rivers and forest patches and higher mean annual rainfall. A selection of the main identified drivers is graphically represented on Figure 5 using the contrast values (C) for comparison. Contrast is the difference between positive and negative

weights, used to correlate drivers for forest suppression and regrowth where positive values indicate ranges more susceptible to change, while negative contrast values indicate less susceptible changes (FORD; CLARKE; RAINES, 2009).

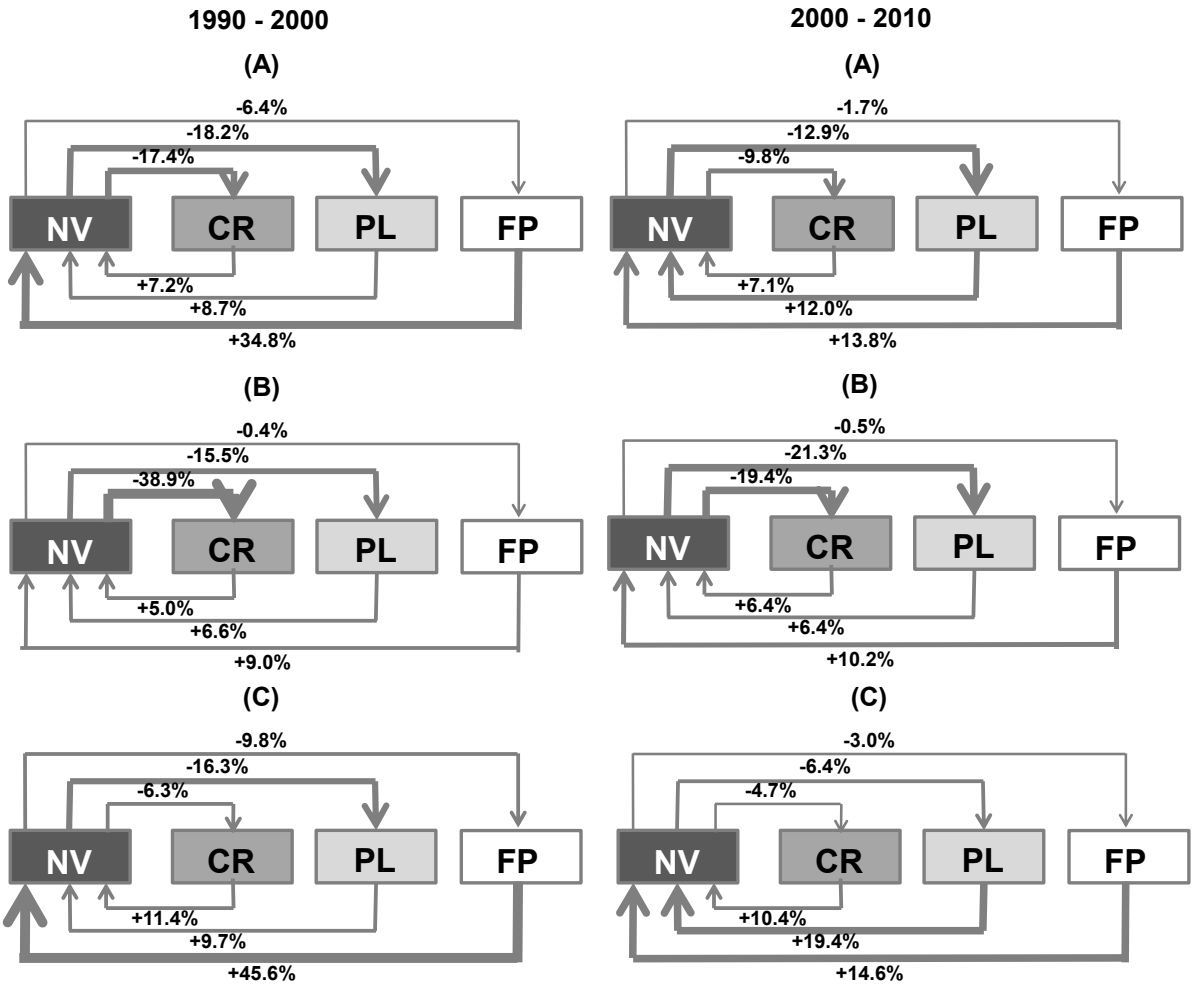


Figure 4 – Net rates (%) of forest suppression and regrowth, for the main transitions, for both 1990-2000 and 2000-2010 periods, where NV is native vegetation, CR is crops, PL is pasture land and FP is forest plantations; (A) is Piracicaba River basin, (B) is Piracicaba sub-basin and (C) is Atibaia sub-basin

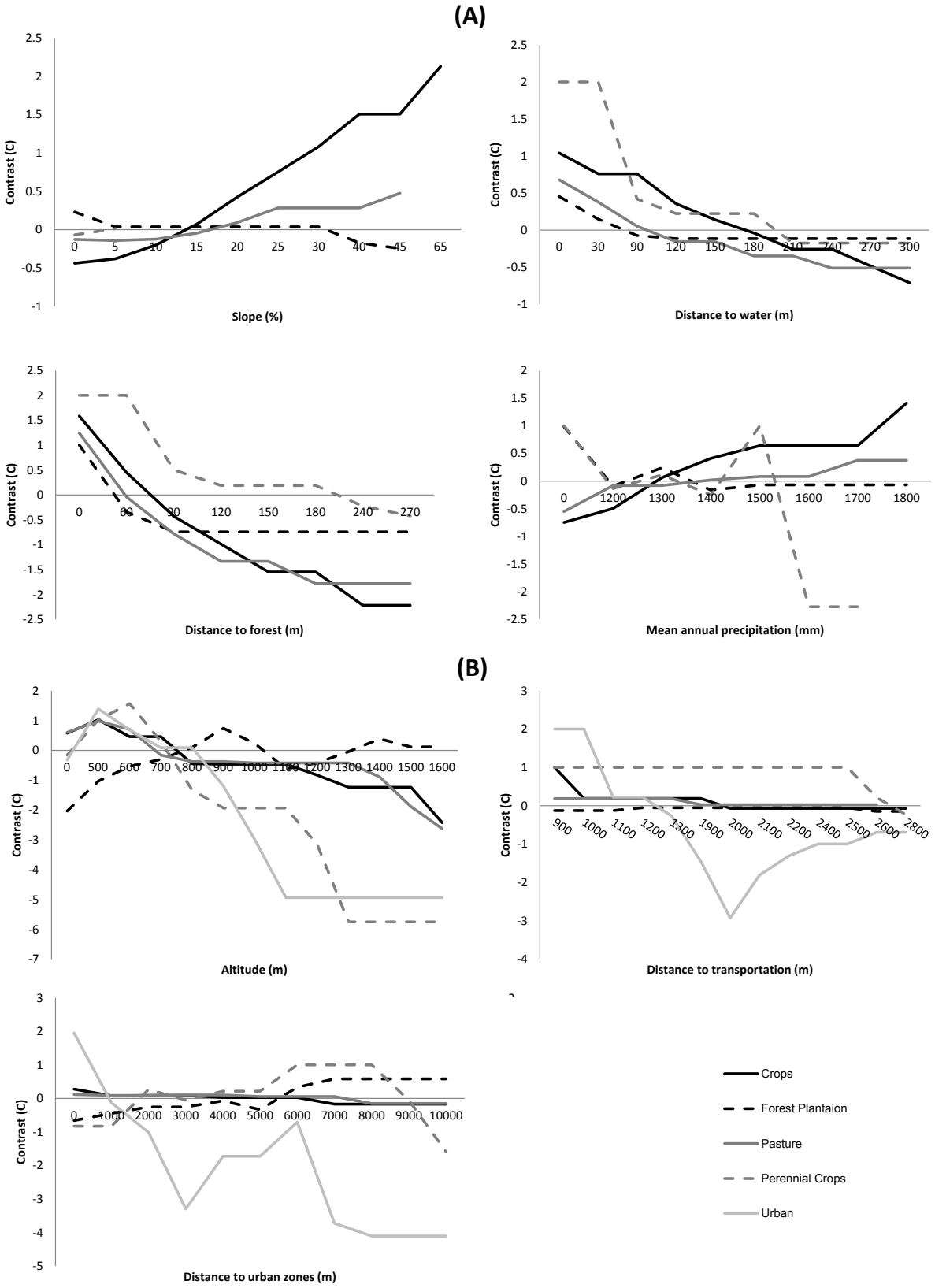


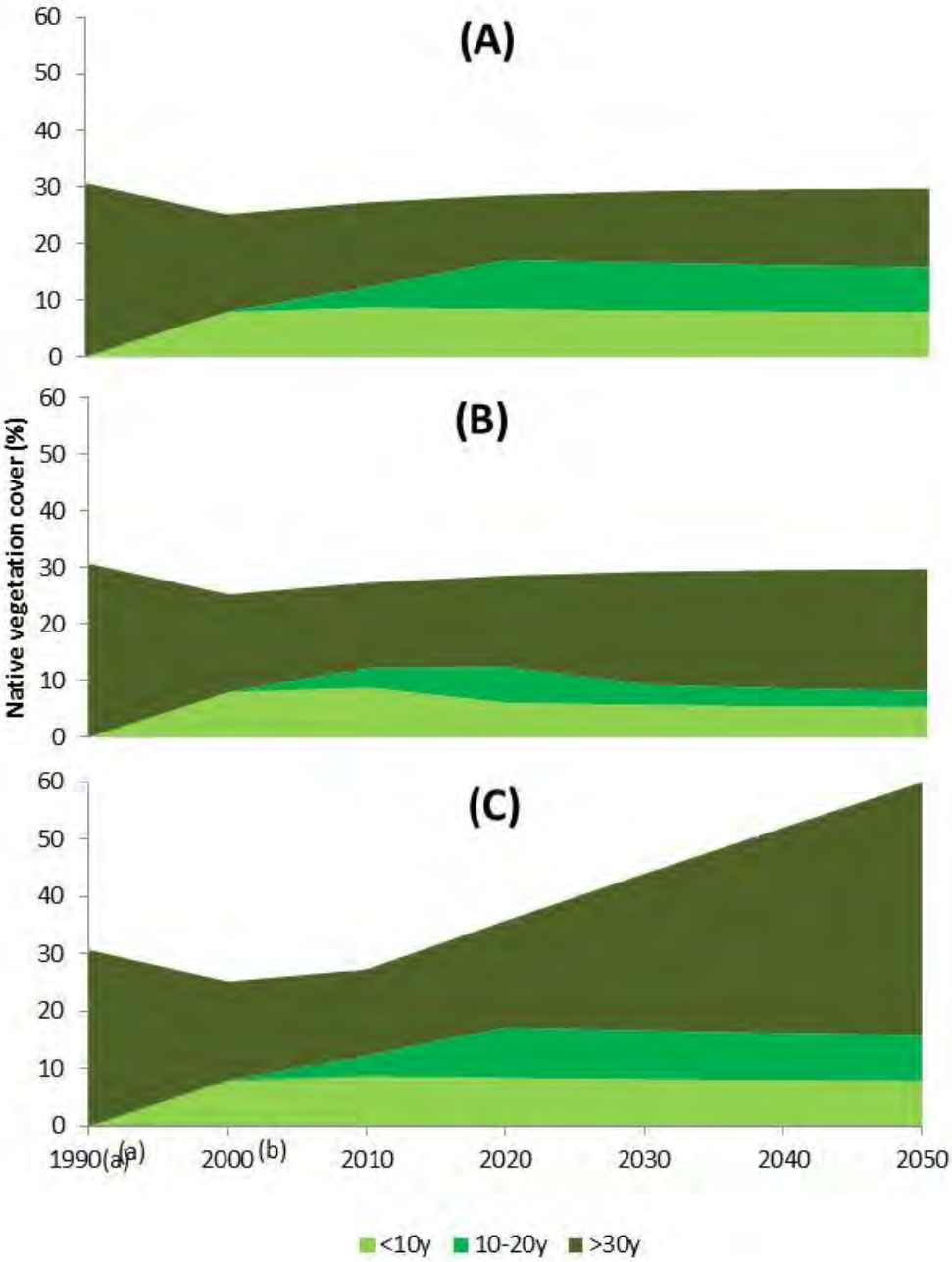
Figure 5 – Contrast values (C) obtained from Weights of Evidence analysis for both regrowth (A) and forest suppression (B) of the main variables, where positive values favor the selected transition while negative values disfavor

2.3.4 Forest age

Recent forest suppression has targeted mostly newer forests (53.5%) while older forests were less deforested (46.5%). From the total forest cover of 2010 (2732 km^2), 52% are old forests, with at least 30 years of age, 16% are old forests with age between 10 and 20 years and 32% are new forests, with less than 10 years of age, of which 31% are regrowth of recent deforested land. From this same analysis we can infer that 10% of all forest in 2010 were the regrowth of deforested land between 1990 and 2000, which was therefore regenerated between 2000 and 2010. Hence, the other 69% of forest gain are transitions from other land uses and were not classified as forests in 1990 or 2000.

2.3.5 Projections

Projection results, elaborated with the aid of the 2000-2010 transition matrix, are presented on Figure 6. In both Old Growth-Deforestation and Equal-Deforestation scenarios, the growth of forest cover maintain the same pattern presented in the 2000-2010 period, differentiating themselves only by the percentage of forest in each age class. For the No-Deforestation scenario, the growth was superior since the forest suppression effect was taken out of the equation, resulting in a greater increase.



(a) 1990 vegetation is not divided into age classes

(b) 2000 is only divided into 10-20 year old and <10 year old forest, due to data availability

Figure 6 – Three scenarios are presented: Old-Deforestation (A), where all forest suppression is done on older forest patches, Equal-Deforestation (B), where forest suppression is divided equally between the age groups and No-Deforestation (C), where there is no forest suppression. Native vegetation ages are set as smaller than 10 years old (<10y), 10 to 20 years old (10-20y) and 30 or more years of age (>30y)

2.4 Discussion

The underlying causes of landscape change allow us to better predict which areas are more vulnerable to change and potentially accelerate the positive processes through different types of incentives, making it an essential knowledge for conservation planning. Here, forest regrowth was explained by environmental & physical drivers while forest loss was explained by anthropogenic drivers. Also, observations suggest that the regrowth of forests were due to land abandonment, derived from multiple drivers working in synergy, that can be tracked back to environmental legislation & surveillance, market needs, economic scenarios, cultural contexts, agricultural mechanization and excellent climate & neighboring conditions, thus implying that resilience potential is an important factor.

Numbers reveal that while new forests have appeared, older, mature and probably better quality forests continue to be lost. Recent forest suppression has targeted both older mature forests as well as new regrowth forests. Together, these observations were projected to the future and resulted in a scenario of higher forest extension, different patch ages and speculated lower environmental services.

Observations

We observed that recent forest suppression (from 2000 to 2010), preferentially targeted younger forests (53.5%) over older, more established forests (46.5%). The small difference in percentage could be due to mapping errors and thus forest suppression rates could be considered equal for both forest classes. Oddly enough, we were able to observe that 31% of all forest gain in 2010 was due to regeneration of forests previously lost in the 1990-2000 period. Hence, areas were clear-cut around the year 2000 but eventually became forests again in a period of around 10 years, supporting once more the suggestion that land abandonment did, in fact, occur, as mentioned by Farinaci and Batistella (2012).

The resulting situation in the 2010 map reveals that although there is a 21.8% forest cover in the Piracicaba River basin, it is a mosaic of different ages of forests with different potentials to provide ecosystem services.

Drivers for forest suppression & forest regrowth

The three main variables associated with forest loss were mostly anthropogenic characteristics. They are: *distance to transportation* and *distance to urban zones*, and *altitude* as the only driver of environmental & physical nature.

Altitude was considered of significant importance for forest cover loss, more often in between 400 and 800 m above sea level. This altitude also represents the optimum condition for sugarcane production and the location of the major agricultural crop production, as well as some perennial crops (EMBRAPA, 2014). Once again we see here the influence of sugarcane production in landscape dynamics, as observed in Rudorff et al. (2010).

Distance to transportation was also identified as a driver for forest suppression. The greatest influence over forest patches was observed within a 3 km buffer and was heavily influenced when around 1km or closer from transportation networks such as highways and railways. The impact of transportation networks on forests may lie on urban pressure due to the economic growth and urban expansion observed in the last 40 years in the region (GALLO, 2001). It is believed that, as a city expands, the logical path of development neighbors large avenues and highways and, as it occurs in the Amazon region, road development may also lead to new forest suppression sites (PFAFF et al., 2007).

Distance to urban zones was also expected to be of influence to forest loss. Forest patches within a buffer of 5 km from urban zones had a tendency of being converted into another land use. This variable also suggests that urban pressure, due to expansion, influences forest dynamics in surrounding areas. This change was expressive in transitions from NV to CR, PL and especially UZ. On the other hand, transition of NV to FP and PC was disfavored when near UZ.

These drivers comply with many other studies of deforestation in tropical regions and are firmly accepted as proximate causes or underlying forces, most often found to work in synergy (GEIST; LAMBIN, 2002). Previous studies have shown that deforestation has been driven by population pressure, wealth, external debts and competition for land, which also partially comply with results shown in this study (BAWA; DAYANANDAN, 1997).

The four main variables associated with forest cover increase were all physical characteristics, namely: *slope*, *distance to water*, *distance to forest* and *mean annual precipitation*.

Slope was identified as an important variable, having forests appearing when terrain slope was higher than 15% where once there was CR and 18% where once there was PL. The explanation for this may lie in the fact that CR, mostly sugarcane plantations in the west of the basin, have become extensively mechanized, which in turn are not manageable in slopes higher than 12% (RUDORFF et al., 2010; SÃO PAULO, 2014). We may be observing here a case of land abandonment due to mechanization procedures in sugarcane plantation. In Brazil, the traditional harvest process consists of manual harvesting following pre-burning (SEGATO et al., 2006). Over the years, this practice has been pointed out as harmful for many reasons, especially to environment and health concerns (CANCADO et al., 2006; MARTINELLI; FILOSO, 2008). Current legislation and the “Green Ethanol” Protocol lead to the extinction of pre-burning and also sugarcane plantations in fields of slope greater than 12% (SÃO PAULO, 2014). Sugarcane potential land has already been categorized by the Sugarcane Agro-Ecological Zoning and legislation is in effect at both federal and state levels (EMBRAPA, 2014).

As for PL, especially family owned, found mostly at the Eastern part of the basin, the reasons for the transition may lie in a more economical and labor related scenario, where regional industrialization and urbanization take labor away from agriculture and pasture, leaving behind less qualified employees and with higher costs (THOMLINSON et al., 1996). Visual interpretation of the maps suggest that, in the early 90’s, there were geometric clear-cuts on old standing forest land in the East Piracicaba River basin, on high slope regions. These lands were classified as PL in the year 2000 and later reclassified as NV, again, in 2010. This observation suggests that land owners increased their PL on slope regions and for some reason abandoned them no longer than 10 to 15 years later. One can speculate that this may be due to the lack of labor, labor costs or even to urban migration, which is more plausible if family situations are taken into account, as seen in Rudel et al. (2005). If one owns both flat and slope PL and needs to abandon one of them, the obvious would be to keep the easier, most cost effective to labor and most productive.

Distance to water, although it first reminds us of ecological situations and vector of seed dispersal (HOWE; SMALLWOOD, 1982), the main reason for its importance as a driver is related to legislation. According to the Brazilian forest code, all hydrological networks are subject to a protection buffer of 30 to 500m, depending on river width (BRASIL, 1965; 2012; ZAKIA; PINTO, 2013). On these buffers, although they should

undergo active reforestation, the former legislation, from 1965, only implied that no anthropic activity should be done; leaving the area untouched and subject to spontaneous regeneration, if environmental conditions are appropriate. Sugarcane plantations once again come in mind because, until early 2000, there were hardly any observations of forest patches on riparian buffers on sugarcane plantations (MARTINELLI; FILOSO, 2007; SILVA et al., 2007). It was not only until a few years later that sugarcane plantations started respecting the riparian buffer, most probably because of commercial obligation regarding exportation and certifications but also law enforcement (SOARES-FILHO et al., 2014). In other words, what is observed in this study is land abandonment in riparian buffer zones, near rivers and creeks, due to compliance to legislation, because of economical and commercial needs. In some cases, land abandonment led to forest regeneration, most spontaneous, in riparian buffer zones of up to 200 m from water bodies, when conditions were appropriate, thus implying that there would be potential resilience.

Distance to forest observations are similar to those of *Distance to water* as new forests were found within buffers up to 100 m from other forest patches. This suggests that growth of new forests on abandoned areas is favored by the neighboring older forests. The older forests function as vectors for seed dispersal and biodiversity and may provide suitable micro-climatic conditions for the spontaneous regeneration of new forest patches (HOWE; SMALLWOOD, 1982).

Mean annual precipitation was also an important driver for forest regrowth. New forests regenerated where mean annual precipitation exceeded 1300mm, especially where there once was CR or PL. This suggests that this variable is enhancing previous drivers, functioning as a supporter for spontaneous new patches. What is likely occurring is a synergy between this variable and previously mentioned. For instance, we may have scenarios in either PL or sugarcane plantation where land abandonment is occurring and new forests are only appearing due to excellent climate and neighboring conditions, explaining also why different sub-basins with different land uses and previous forest cover, had different forest regrowth results.

Forest transition at the sub-regional level

At the sub-basin level, we observed distinct forest growth indices, indicating the influence and pressure of regional individualities. These regional characteristics drove and most probably drive the situation of forest transition differently in each

location. For example, a very distinct characteristic is the prevailing agricultural land use. In the Piracicaba and Corumbataí sub-basins, the large industry-owned sugarcane plantations predominate and are rarely mixed or rotated with any other agricultural land use. Due to its historical farming characteristics, they are also not forest friendly since, until few years back, harvesting was still done only after complete burning of leaf biomass, resulting in a cloud of fire that could burn any nearby forest patch. Coincidently, in the last period, Corumbataí presented a very low forest cover gain/loss ratio while Piracicaba practically showed a draw. On the other hand, Atibaia and Jaguari sub-basins presented the highest forest cover growth. As for their prevailing agricultural land use, we highlight the small family-owned dairy pasture land and in some cases a significant amount of industry owned *Eucalyptus* spp. plantations. Differently from the sugarcane plantations, these land uses naturally are a more forest friendly environment. In fact, the pulp and paper industry follow strict commercial certifications that make them respect environmental legislations regarding native forest boundaries and restoration programs, which, in turn, they make use of for marketing potential (SIRY; CUBBAGE; AHMED, 2005; MEYFROIDT; LAMBIN, 2011; PINTO; MCDERMOTT, 2013). Also an important factor, the elevated presence of large and older native forest fragments in neighboring regions potentiate spontaneous forest regrowth, especially through seed dispersion (HORN; CODY; DIAMOND, 1975; HOWE; SMALLWOOD, 1982; RUDEL et al., 2005).

Future projections

Projections based on observed rates of forest regrowth and forest suppression suggest that the same quantity of forests present in 1990 will only occur again in a time span of over 40 years. Even with recovery of forest cover, the forest patches with different ages would most probably provide different ecosystem services, when compared to fully mature forests (CHAZDON, 2008; BROWN; ZARIN, 2013). With that in mind, it could take even more time for forests to reach maturity and provide the same benefits provided by primary vegetation, creating a time lag in observed forest cover versus specific ecosystem services. The No-Deforestation scenario is the only one that showed potential to achieve higher quantities of forest of higher age class and at faster rate. The only reason for its differentiation is the absence of forest suppression in the equation, something highly doubtful unless rigorous surveillance, inspections and control are in effect.

Implications

Given the rate of loss, forest regrowth and the drivers found, we may expect that some ecosystem services will perhaps increase only when new forests reach maturity, leading to a scene of progressive forest growth and a decline or lag in specific ecosystem services. To better understand this hypothesis, we used future scenarios and respective forest ages to predict ecosystem services according to forest maturity. We established that 30 year-old (or older) forest patches could have potential for providing 100% ecosystem services, 20 year-old forests could have a 70% potential and 10 year-old forest could have 40%. Based on findings from Banks-Leite et al. (2014), we also established that a 30% forest cover, with 30 year-old patches, or older, is the maximum potential of the landscape, therefore with 100% ecosystem services support (Figure 7).

We can observe from this supposition, that only the No-Deforestation scenario has a potential for returning 100% of ecosystem services in the near future, due mostly to its potential to achieve 30% of area cover. We can also observe that although Old Growth-Deforestation and Equal-Deforestation scenarios provide the same amount of forest cover over the years, their potential for ecosystem services return is different. If more old growth forests are lost, rather than the new regrowth, the potential ecosystem services return is lower. Also interesting, is that even though the No-Deforestation scenario achieves a 30% area cover between 2020 and 2030, the provision of full ecosystem services would only be achieved later, after 2040. This lag is a result of aging forests which would only promote certain environmental benefits after maturity.

Results found in this study can be applied to conservation planning, but are not only limited to that. Legislation and surveillance should be stricter if increases in ecosystem services are a goal to be achieved in the future. Efforts should be given to protect older mature forests as they tend to be more beneficial than the newer patches. Environmental marketing promoted by exported commodities certification needs should be expected in the future, especially in sugarcane and *Eucalyptus* spp. plantation sub-products and thus regrowth potential may still increase. Economic incentives, either public or private, and investments on restoration should consider the spatial location when destining their funds. It seems obvious that some regions have more potential for having rapid restoration projects by spontaneous

regeneration while other regions lack climate and neighboring conditions, therefore in need of active planting investments.

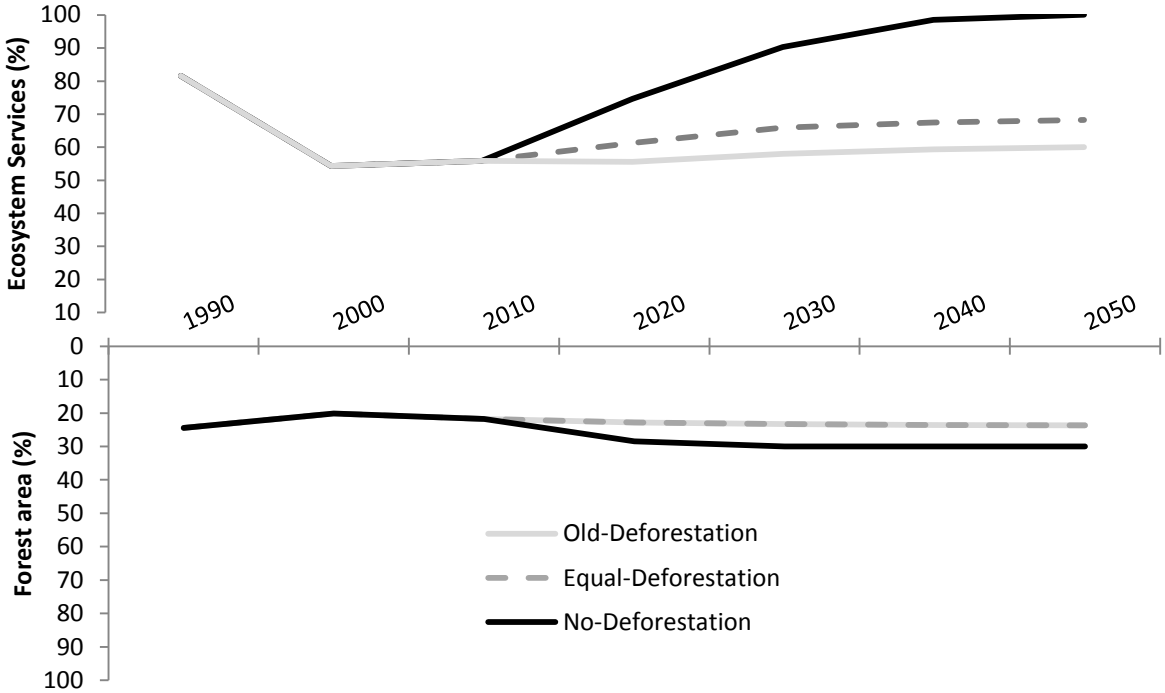


Figure 7 – Suppositions of ecosystem services potentials according to different forest cover and age scenarios

2.5 Concluding remarks

Our results have shown that, as well as forest suppression, forest regrowth is influenced by multiple drivers working in synergy. Environmental & physical characteristics are linked to forest gain while anthropogenic characteristics are related to forest loss. Land abandonment may be a leading cause for forest regrowth, ahead of other factors such as legislation or social causes, but forest regrowth is limited to a synergy of factors that potentiate resilience. Understanding the drivers of native forest increase is essential in order to comprehend the potential for landscape restoration and potential reestablishment of ecosystem services.

Forest suppression has targeted both older mature and newer less biodiverse forests. Future projections reveal that forest gain may come in a slow pace, followed by specific ecosystem losses or lag for increasing, due to continuous trends of older mature forest loss. It is important to question now which mechanisms will guarantee that regrowth forests are not once again deforested in the future.

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3 FOREST SCENARIOS AND POTENTIAL EFFECTS ON SUPPORTING AND REGULATING ECOSYSTEM SERVICES

Abstract

The presence of forest patches in a landscape promotes several benefits that can be associated with ecosystem services. Among the supporting ecosystem services, services required for the making of all other ecosystem services, we highlight in this study native vegetation cover and its importance and need for the maintenance of biodiversity and habitat integrity but also for the existence of other services of regulating nature. Regulation of river flow and water yield are clear examples of this support given by forests and native vegetation, also current topics of great importance in the Atlantic Forest region located in the State of São Paulo, Brazil. In this study, we selected the Piracicaba River basin as a study region. Here we analyze (i) projections of future forest cover scenarios and (ii) its effects on supporting ecosystem services related to landscape structure and (iii) regulating ecosystem services linked to water yield & regulation. Historic land cover maps dated from 2000 and 2010 were used to project scenarios of forest increase into the next four decades, using the Dinamica EGO software, the weights of evidence method and collections of anthropogenic and environmental & physical drivers. Scenarios were set to be *status quo* (SQ), *no deforestation* (ND) and *riparian restoration enforcement* (RRE), where in the first, observed tendencies were replicated, in the second, the same was done but without a forest suppression effect and in the third, all riparian buffer zones were applied to the last observed land cover map. Resulting maps were compared using selected landscape metrics, calculated by the V-late software, as a proxy for biodiversity and habitat. We then calibrated a hydrological model (SWAT) for the year 2010, later exchanging the observed land cover with the proposed scenarios, thus comparing mean annual water yield and regulating hydrological indicators for each scenario and decade. Results show that (i) overall the best scenario for forest cover is LE, followed by RRE and later SQ; (ii) SQ is losing small forest patches and gaining area in larger patches, resulting in a final cover of 22.4% in 2050; ND is by far the most promising forest cover with a final cover of 43.2%; and RRE resulted in stretched patches covering 28.4% of the basin, all compared to an observed 21.8% cover in 2010. Segmentation of the basin resulted in different landscape structure values, suggesting that regional characteristics play a fundamental role in forest dynamics; (iii) mean annual water yield was lower when forest gain was observed although variance was also lower, thus implying more regulation while total water yield tends to be lower.

Keywords: Atlantic Forest; Forest projection; Landscape structure; Hydrological modeling; Ecosystem services

3.1 Introduction

The occupation and colonization of Brazil was done in the traditional way, replacing natural ecosystems mainly by agricultural and urban landscapes. The loss of natural environments can be seen in all biomes from the Amazon to the Atlantic Forest, Cerrado, Caatinga, Pantanal, Mangroves and the Pampa (BACHA, 2004). Much is

said about the deforestation of the Amazon region, its consequences for biodiversity (VIEIRA et al, 2008.), main causes (DINIZ et al., 2009; RIVERO et al., 2009), and the potential that the region has for carbon sequestration (CARVALHO et al., 2010). Despite being historic, studies show that the degradation of these ecosystems still occur in the present, sometimes under strong influence of public policies, socioeconomic pressures (VAN DE STEEG et al., 2006) and even on account of impacts of surrounding land use, such as the Cerrado in the State of São Paulo (DURIGAN; SIQUEIRA; FRANCO, 2007).

Originally composed of more than 80% of forest cover, a mixture of Atlantic Forest and Cerrado, today in São Paulo, there is only 17.5% of the original vegetation cover; a number that was four percent points lower in 1992 (KRONKA; MATSUKUMA; NALON, 1993; SÃO PAULO, 2002; KRONKA et al., 2003, 2005; SÃO PAULO, 2010). The original biomes gave way to large urban centers and mainly farmland, today characterized mostly by pasture and sugarcane (RUDORFF et al., 2010). The loss of forest cover observed in the state can lead to distinct side effects on biodiversity, habitat, water quality and quantity, carbon storage, among others (WILCOX; MURPHY, 1985; MORAES et al., 1998; DEFRIES et al., 2010).

Change indicated by recent surveys show that there is a tendency for net gain of native vegetation cover in the state. Nevertheless, there are still no concrete indications of why these changes occur and where they occur, at what rate and whether this is a trend for other regions as well. One hypothesis is that it is a result of more strict environmental legislation, adoption of measures against deforestation and higher pressure from society for the preservation of the environment, but it also results from land abandonment and optimum environmental conditions (FARINACI; BATISTELLA, 2012).

The presence of forest patches in an heterogeneous landscape has great importance, either for biodiversity, ecology, wellness, quality and quantity of water or even for aesthetic reasons (GENTRY, 1992; OSBORNE; KOVACIC, 1993; TAHVANAINEN; TYRVÄINEN; NOUSIAINEN, 1996; LOWRANCE et al., 1997). Moreover, forests influence processes considered basic for the survival of species of flora and fauna in a context that translates into ecosystem ecology where there is a flow of energy and matter through organisms and their environment. This is, the capture of light energy by plants, their conversion into organic matter and transfer through the food chain (GOLLEY, 1993; CARPENTER, 1998; TROPPMAIR, 2000).

In another context, forests are part of the landscape, contributing to a network of inter-relationships of all biota and the spatial environment, improving relationships between ecological processes in the environment and specific ecosystems, a science defined as landscape ecology (TROLL, 1971). In this context, there is a concern with the interrelationship between man and his habitat and potential environmental problems that may arise considering the spatial distribution of ecosystems (WIENS, 1976; TURNER, 1989; BARRETT; BOHLEN, 1991; NAVEH; LIEBERMAN, 1994; METZGER, 2001).

Current landscapes are the result of several modifications, including variability in abiotic conditions such as climate, topography and soils; biotic interactions; current and historical patterns of settlement and land use; and natural succession dynamics and disturbance (TURNER; GARDNER; O´NEILL, 2001). Economic trends are mentioned as direct influences on land use and land cover change in the state, as it was the case of coffee expansion and how it is today the case of sugarcane expansion throughout Brazil (RUDORFF et al., 2010).

Landscape ecology has emphasis on interaction between spatial and ecological processes and structures, therefore, methods are needed to describe and quantify the spatial structure of the landscape. Numerous metrics can be performed for quantification, some not necessarily of spatial characteristics, measuring proportions or relative quantity while others focus on spatial characteristics such as connectivity and core area (GUSTAFSON; PARKER, 1994).

Technological development has contributed greatly to advances in studies of landscape ecology. Progresses in computing, remote sensing and GIS have facilitated and optimized the processing of various data. Many studies of landscape patterns are conducted on data of land use and land cover that have been digitized and stored in a GIS environment (TURNER; GARDNER; O´NEILL, 2001).

Despite the diffusion of GIS, these are limited in the investigation of landscape structure providing space for the development of complementary tools, targeting specific metric analyzes. This is the case of the Vector-based Landscape Analysis Tools Extension (V-LATE) (LANG; BLASCHKE, 2007), developed at the University of Salzburg and available for free in the web, similar to FRAGSTATS, a spatial pattern analysis program for categorical maps (McGARIGAL; MARKS, 1995), developed by the Oregon State University, and also free.

There is large interest in studies of landscape dynamics, especially when it comes to spatial and temporal landscape simulations (TURNER, 1987; MULLER; MIDDLETON, 1994), urbanization (ALMEIDA, 2003), socioeconomic influences on landscape changes (WANG; ZHANG, 2001) and deforestation (FERRAZ et al., 2005; FEARNSIDE et al., 2009).

Also of interest are studies that model historical and future landscape scenarios and their influences and effects on other systems, as it is the case of Coe,Costa and Soares-Filho (2009) who modeled the influence of Amazon deforestation on evapotranspiration (ET) and flow of the Amazon River. Similarly, Teixeira et al. (2009) modeled landscape dynamics in the Atlantic Forest to investigate influences on issues of forest species conservation and management.

A model can be defined as an abstract representation of a system or process and is dynamic when it has a temporal dimension, where the inputs and outputs vary over time (TURNER; GARDNER; O´NEILL, 2001; ALMEIDA, 2003), overcoming current limitations of GIS that are based on static views (BURROUGH; McDONNEL, 1998). An example of a dynamic modeling software is the Dinamica EGO (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002), created by the Center for Remote Sensing (CSR) at the Federal University of Minas Gerais, and specialized in the study of modeling and investigation of dynamics of environmental systems, which uses a WoE method (BONHAM-CARTER, 1994; SOARES-FILHO; RODRIGUES; COSTA, 2009).

One of the main purposes and objectives of dynamic models is to simulate and investigate dynamic spatiotemporal changes occurring on structure and pattern of a landscape and its consequences on natural and ecological resources (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002). According to Burrough and Mcdonnel (1998), dynamic spatial modeling can be defined as a "mathematical representation of a real-world process in which a location on the Earth's surface changes in response to variations in driving forces", making it a valuable tool for planning and studying transition events in the landscape.

A change occurring in the structure or landscape pattern can interfere with population dynamics and habitat availability, promoting greater risks of extinction or displacement of these to other landscapes, creating a threat to the conservation of biodiversity (HARRIS, 1984). Changes are generally caused by human occupation, mainly by processes of deforestation and forest fragmentation that directly influence

ecosystem services promoted by nature (TURNER, 1989; COLLINGE, 1996; JORGE; GARCIA, 1997; MILLENNIUM ECOSYSTEM ASSESSMENT, 2005).

Ecosystem services may be briefly defined as the ability that nature has to provide benefits for quality of life and human health. Ecosystem services may include a variety of products and benefits such as food, fuel and fiber, and climate regulation, pest control and spiritual or aesthetic benefits (MILLENNIUM ECOSYSTEM ASSESSMENT, 2005). The MEA describes these benefits and breaks them into specific services called *provisioning*, *regulating*, *supporting* and *cultural services*.

Among the supporting ecosystem services, services required for the making of all other ecosystem services, we highlight in this study native vegetation cover and its importance and need for the maintenance of biodiversity and habitat integrity but also for the existence of other services of regulating and provisioning nature. Regulation of river flow and water yield are clear examples of this support given by forests and native vegetation, and are also current topics of great importance in São Paulo, especially in the Atlantic Forest region (RIBEIRO et al., 2009; TEIXEIRA et al., 2009; CARAM, 2010; GONZALEZ, 2010; DA SILVA, 2012; FARINACI; BATISTELLA, 2012; FERRAZ et al., 2014).

These topics on ecosystem services are of great interest to ecologists, economists, and especially to landowners and legislators. The first group, ecologists, have long recognized the benefits promoted by ecosystems. Economists on the other hand, seek to quantify and derive economic benefits from these services to support strategies for landowners and legislators (BROWN; BERGSTROM; LOOMIS, 2007).

Among the supporting services provided by native vegetation, habitat and biodiversity are emphasized in this study. Habitat fragmentation can be defined as changes in habitat configurations resulting from segmentation which in turn also result in biodiversity loss (FAHRIG, 2003). Usually, fragmentation is assessed through local and intensive studies but can also be made through remote sensing, giving faster access to information for decision making (CORRETT, 1995). Landscape structure is also recognized as biodiversity substitutes and have been used in Atlantic Forest studies (RIBEIRO et al., 2009). They do, however, have limitations since functional connectivity and species distribution is not considered, but can be useful to establish general guidelines when data is limited (FAIRBANKS; REYERS; VAN JAARSVELD, 2001).

It is believed that among the many benefits of native vegetation patches, there are those aimed at river flow regulation and water production, sometimes even referred to as myths (CALDER, 2004). The myths usually apply to public knowledge, which sometimes mistaken the effects of forest cover increase and the output hydrological indicators such as water yield, infiltration, runoff of rainwater, soil protection and sedimentation in a watershed (COOPER et al., 1987; SHERIDAN; LOWRANCE; BOSCH, 1999; MOMOLI; COOPER; CASTILHO, 2007). To achieve values for hydrological indicators, large-scale hydrological studies are most often impossible to be conducted due to high costs and long lead-time. To overcome this situation, hydrological models are often developed and applied to predict impacts of various activities on water quality and quantity (PESSOA et al., 1997).

An example of hydrologic model is the Soil and Water Assessment Tool (SWAT), which promotes the simulation of physical processes in a watershed, with the aim to predict the impacts of land cover change on surface and groundwater flow, sediment yield and water quality (KING et al., 1996). The model is spatially semi-distributed with the watershed being subdivided into numerous Hydrologic Response Units (HRU), thus reflecting the different topography, soil types and land cover, assumed to be homogeneous in hydrologic response to land cover change. Output flow is calculated for each HRU and propagated to obtain total flow for each sub-basin and later for the entire watershed. The main components of the model include hydrology, climate, sediment, soil temperature, plant growth, nutrients, pesticides and agricultural management, each more or less necessary according to the study being conducted (ARNOLD et al., 1998).

Watersheds have singularities and particular behaviors given their characteristics, thus, calibration is the process of adjusting the parameters of the model to increase efficiency in the simulation of the basin (GUPTA; SOROOSHIAN; YAPO, 1999). Calibration of hydrological models is complex and time consuming. To assist in this process there are several methods of automatic calibration, such as the calibrator SWAT-CUP, a calibrator for SWAT models (ABBASPOUR, 2011).

Here, we address the following research questions. What are the future projections for native vegetation cover? How will these projections affect supporting ecosystem services related to landscape structure? How will these projections affect water regulating ecosystem services linked to mean annual water yield and regulation?

We used previous thematic land cover maps and collections of anthropogenic and environmental & physical drivers along with their WoE to project NV scenarios for the next four decades. We then evaluated the scenarios for selected landscape metrics and selected hydrological indicators to obtain results linking NV cover change to ecosystem services of supporting and regulating nature.

Our goal was to determine if the simulated change in forest cover could affect potential services related to habitat, biodiversity and water regulation.

3.2 Methods

3.2.1 Study region

The study region is the same as presented on *2.2.1 Study region*.

3.2.2 Dynamic landscape modeling

The landscape modeling process described here uses the WoE method (BONHAM-CARTER, 1994; SOARES-FILHO; RODRIGUES; COSTA, 2009) to select the variable most related to observed land cover changes, quantify their influences and later simulate transitions. The procedures here described are a continuation of Chapter 2, regarding the use of the resulting WoE and transition matrices. Spatial data was the same as described on *2.2.2 Geospatial data* and resulting land cover maps from *2.3.1 Land cover thematic maps*.

In the described chapter, a transition probability map was produced, representing the most favorable areas for change to occur. Resulting WoE was used to analyze which variables best explain forest changes that occurred in the 2000 to 2010 period and therefore examined which are the current drivers for forest suppression and regrowth (TEIXEIRA et al., 2009).

For optimum calibration results, we divided our study region into six sub-basin segments (Figure 8). The segments were selected by visual interpretation of land cover, terrain and soil and are therefore modified sub-basin regions. This was done to better assist calibration parameters and result in a more significant final simulation. Thus, simulation was individually processed for each unit. Resulting six scenario segments are later mosaicked into one individual thematic map.

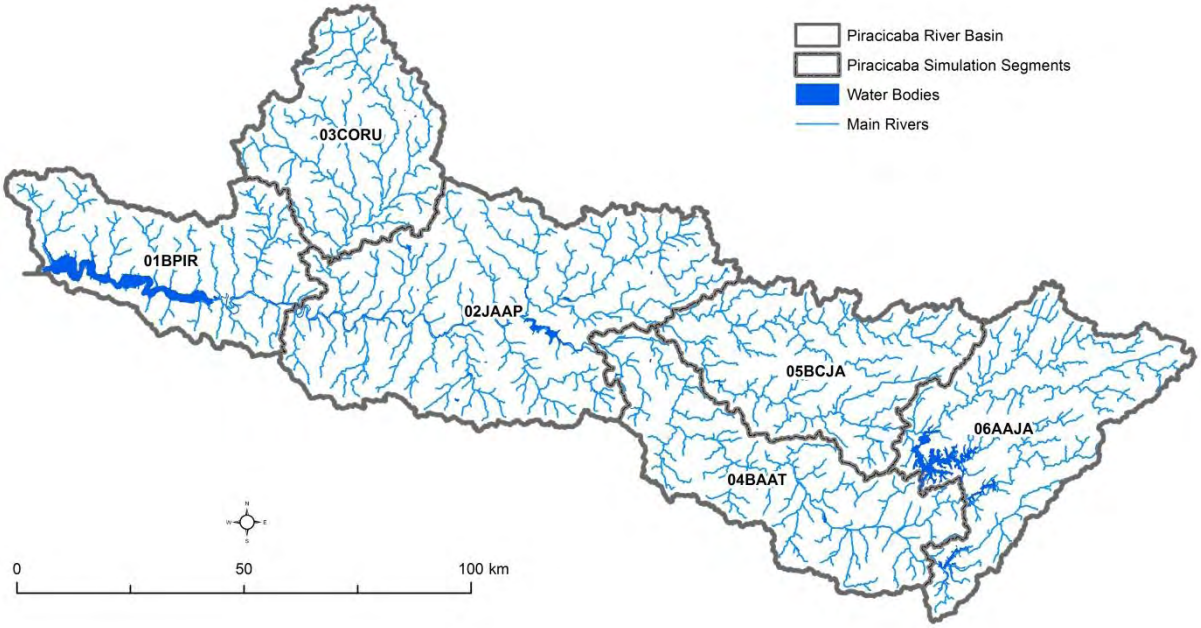

Figure 8 – Segmentation of the Piracicaba River basin into six units, which were individually simulated on Dinamica EGO, where 01BPIR represents a lower region of the Piracicaba sub basin, 02JAAP is a composition of a the higher section of the Piracicaba sub-basin and lower Jaguarí sub-basin, 03CORU is the Corumbataí sub-basin, 04BAAT is the lower Atibaia sub-basin, 05BCJA is the middle Jaguarí and Camaducaia sub-basins, and 06AAJA is a composition of both higher Atibaia and higher Jaguarí sub-basins

Land cover change simulation model

Dinamica EGO was used to run the landscape change simulations. The model was calibrated for the 2000-2010 period, using the respective transition matrices and WoE coefficients obtained by cross-tabulating the 2000 and 2010 maps. Annual time-step simulation maps were derived from this procedure.

In order to achieve a similarity between simulated and observed landscapes, different models were tested by changing the parameters of the transition functions called *Expander* and *Patcher*. The first function is dedicated to the expansion or contraction of present patches found in the landscape while the second is dedicated to generation of new patches, through a seeding mechanism (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002).

Calibration and Validation of the simulation

The selection of variables for the modeling analysis considered the independence between pairs of selected variables to explain the same transition of land cover. For

procedures of model calibration, we verified the association or spatial dependence between the selected variables that explain a same transition. For this, we used Cramer (V) and Joint Information Uncertainty (JIU). According to Bonham-Carter (1994), if V and JIU values are higher than 0.5 than there is higher probability of spatial dependence and thus one of the variable must be eliminated or both combined, ensuring that no redundancy is made in the model.

Calibration also occurs through changes in the algorithms expander and patcher, which defines the spatial characteristics and percentages of transitions in question. These functions integrate cellular automata rules that are applied to mimic and repeat neighborhood influence on a transition of a certain cell state, as seen on Teixeira et al. (2009). In this phase, mean patch size, variance and isometry values are inserted.

Calibration is completed by obtaining the correlation maps, where a validation can be calculated. Validation test is done in order to compare the simulation to the observed maps (difference map). In Dinamica EGO, this process occurs by applying algorithms of multiple windows with constant and exponential decay functions on the difference maps. The output parameters refer to tabular values of maximum and minimum similarity between the maps, using windows of 3x3, 5x5, 7x7, 9x9 and 11x11, from a fuzziness of location method (HAGEN, 2003). For this validation process, we accepted the models with greater similarity between the maps. The use of indexes with multiple window resolutions is justified by the impossibility of reproducing spatial situations with complete accuracy, mainly due to the randomness of some transitions (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002).

Basin segmentation into six parts, due to the regional characteristics and properties, allowed better parameterization, thus ensuring a more satisfactory result and contemplating regional individualities.

Simulation of future scenarios is valid when using a calibrated and validated model, thus representing the transitional processes in an elapsed time interval. In this study, the temporal resolution used is 10 years, allowing a reliable projection until the year 2020, but which can be extrapolated with appropriate remarks (SOARES-FILHO; COUTINHO CERQUEIRA; LOPES PENNACHIN, 2002). Here, we used the observed map dated 2010 as a starting point, transition parameters from the 2000-2010 period, and 2010 variables, projecting the future scenarios using parameters validated for each basin subdivision. Projections were extrapolated for every 10 years until the year 2050, with a total of four maps per scenario.

Scenario modeling

The scenarios were established as *Status Quo (SQ)*, *No Deforestation (ND)* and *Riparian Restoration Enforcement (RRE)*. All scenarios, except RRE, were produced using the landscape trajectories. In SQ, we assumed the historical trend that occurred between the year 2000 and 2010 would continue to occur in the following decades. Therefore, we projected the observed changes onto the future, using the transition matrices rates in a Markovian approach and with the aid of WoE. For LE, the same was done, except that the forest suppression affect was taken out of the equation (through transition extraction), as if law enforcement would generalize the banning of any forest suppression. Thus, modeled scenario would only take into account the regrowth of vegetation, maintaining NV present in the year 2010 in future projections along with new patches and the expansion of present patches. This scenario is a projection of what the landscape would appear to be in the future if law enforcement prohibits any kind of forest suppression. At last, in RRE there was no automated trajectory. We used as a source the 2010 land cover map and overlaid a modified riparian protection vegetation (*Permanent Preservation Area*), as stated by local law (BRASIL, 1965, 2012; MARTINELLI, 2011; SPAROVEK et al., 2011; SOARES-FILHO et al., 2014). To achieve this product, we used available hydrology maps at 1:50,000 scale (IBGE, 2013), containing the main rivers and lakes in the region. Rivers that were represented as polygons at this scale were automatically buffered for 100m, considering that they had a width between 50m and 200m. Features identified as major rivers were buffered for 50m, assuming they fall into the category of rivers with width between 10m and 50m. All other rivers were buffered for 30m, assuming therefore that they fall into the category of rivers with width between 1m and 10m (ZAKIA; PINTO, 2013). Water bodies, such as lakes and reservoirs, with more than 20 ha, where also buffered for 100m, as were the remaining water bodies, with 30m buffer. No effort was made to buffer other features, such as water springs, hill tops and slope areas, as stated by law, due to the complexity that they encompass. Thus a modified concept of riparian protection vegetation scenario was created for the year 2050, assuming that by this year all vegetation would be at least 30 years old. A summary of individual characteristics for each of the scenarios can be found on Table 5.

Table 5 – Modelled scenarios for the Piracicaba River basin and respective characteristics

Status Quo	No Deforestation	Riparian Restoration Enforcement
Landscape dynamic trajectories	Landscape dynamic trajectories	Riparian protection as stated by law
Starting from 2010 map	Starting from 2010 map	Starting from 2010 map
Heterogeneous vegetation age	No suppression effect heterogeneous vegetation age	30 year old riparian vegetation

Running simulations

Scenarios SQ and ND were run using a simulation with patch formation and expansion. Scenario RRE was produced by means of an overlay of the 2010 land cover map with a 1:50,000 riparian buffer zone. In all three scenarios, the regrowth of NV was considered in both new patches (patcher function) and as a complementation of existing patches (expansion function).

Concerning modeling simplification, not all land cover transitions were implemented in the model. We selected only transitions between CR, PL and NV, as these were the most important and most present transitions occurring in the study area.

As was done for the observed land covers of 1990, 2000 and 2010, these simulations were also dated as to how old the forest patches are at the time of simulation. Ages were set as up to 10 year old, 10 to 20 year old and 30 or more years of age.

3.2.3 Hydrological modeling

At this stage, we modeled each of the hydrological responses for the proposed land cover scenarios. SWAT model was used specifically to model mean annual water yield (mm), rainy and drought season daily water yield (mm) and water regulation (variance in daily water yield for total, rainy and drought periods). These hydrological indicators were obtained as a proxy for regulating ecosystem services.

The process of calibration and validation of the hydrological model used in this study follows methodology described and elaborated by (BRESSIANI; SRINIVASAN; MEDIONDO, 2014).

Model setup & data sets

Piracicaba SWAT model was constructed using freely available information on the web and provided data from water management agencies after meetings or e-mail contact and phone calls. Digital Elevation Map (DEM) was built using ASTER Global Digital Elevation Model Version 2 (GDEM V2) dated 2011, with 30-meter postings

and 1 x 1 degree tiles (NASA; METI, 2011). The ASTER DEM was hydrologically corrected to be used for the hydrological modelling, where “flow direction”, ”sink” and “fill” procedures were made using ESRI ArcGIS 10.1.

One of the first steps in hydrological simulations using SWAT consists of data gathering, managing and normalization. SWAT data base requires hydro-physical information (number of layers, depth of the lower boundary of each surface layer, bulk density, available water capacity, saturated hydraulic conductivity and percentage of soil particles). Necessary soil parameters such as number and depth of profiles, texture and organic matter, were collected from the expanded caption of the provided soil maps from the Agronomic Institute of Campinas (IAC) (DE OLIVEIRA, 1999) or generated through a pedotransfer functions (PTF), similar to what has been seen in Tomasella; Hodnett and Rossato (2000), and developed by Saxton and Rawls (2006). These data are used to estimate saturated hydraulic conductivity and the available water capacity, components of SWAT. Soil map and a list of soil types and respective percentage of the watershed can be found in Appendix F.

Land cover map was provided from our own thematic map dated from the year 2010 with the original 1:50,000 scale. The map contained information on land cover but was also linked to a data base with specific parameters on plantation data, management, harvest and operations which were adjusted or established during calibration procedures. The last spatial database necessary for the completion of the HRUs was slope information, derived from the ASTER DEM and can be found on Appendix G.

Daily weather information was collected and processed from official records, provided by the National Meteorological Institute (INMET) and Luiz de Queiroz College of Agriculture at University of São Paulo (ESALQ/USP). Gathered or derived data range from maximum and minimum air temperatures, relative humidity, wind speed, atmospheric pressure to solar radiation. Solar radiation, wind speed and relative humidity were simulated by SWAT, using climate generator, based on supplied data base.

For calibration and validation purposes, there is also a need for observed values of river flow and precipitation, provided by stream flow and rain gauges. Data availability is limited in both time interval and provided date. Data used was similar to what was used by Caram (2010), who selected 61 precipitation gauge stations and performed

a graphical analysis of the period to identify gaps in the data of each station. Precipitation data series used were selected from the National Water Agency (ANA) and DAEE stations. Here, a total of 187 precipitation gauge stations located in and around the basin were used, with log histories as late as 1960. Data was interpolated using the PCP_SWAT program developed by Zhang and Srinivasan (2009). Using an interpolation method of Inverse Distance Weighted (IDW), the result is an individual simulated station for each sub-basin, as seen in Figure 9. Also selected, were 16 stream flow gauges, presented on Figure 9 and Table 8. All data used for simulation procedures were in a daily interval, ranging from the year 1995 to 2011, while the first 5 years (1995 to 1999) are used as a warm-up period.

Calibration and validation

For this study, sensitivity analysis and calibration were performed by means of optimization methods available in SWAT-CUP software, where the algorithm Sequential Uncertainty Fitting (SUFI-2) was used (ABBASPOUR, 2011). SUFI-2 contemplates all sources of uncertainty and the degree to which all uncertainties are considered is performed by calculating the *P-factor*, which is the percentage between measured data and by 95% prediction uncertainty, which is calculated in percentage levels of 2.5% and 97.5% of the cumulative distribution of results dependents of a change in a variable obtained from Latin Hypercube Sampling, or simply by the withdrawing of 5% of bad simulations (ABBASPOUR, 2011).

Piracicaba basin was calibrated for the period of 2000-2010 at three individual flow gauge stations (numbers 7, 12 and 13), thus, segmenting the basin into three separate regions as seen on Figure 9. Due to the presence of the Cantareira Transposition System, hydrological modeling was done only for the regions downriver from the reservoirs. Upriver regions are masked out and considered as *inlands*. Water regulation provided by the system would not allow an integration of the sub-basins and would also input difficulties in calibration of upper land water flow due to the lack of gauge stations for the proposed time lapse. Considering that the system regulates water flow at this point and that its output flow is provided, calibration of the remaining of the basin was done so only for its lower portion, resulting in a total modelled area of 10,416 km^2 (Figure 9).

Thus, the basin was subdivided into three regions, each defined to conduct the calibration individually. First, a region upland of Piracicaba was calibrated and

identified as A1 on Figure 9. Calibration parameters from region A1 were transported to region A2 and A3, parallel tributaries with similar regional characteristics. Secondly, the Corumbataí (region B) was calibrated separately, starting out from the parameters set in in A1, A2 and A3. Finally, region C was calibrated using the last flow gauge station, nearest to the Piracicaba River mouth, located in the municipality of Piracicaba, and later extrapolated for the continuity of the basin, region D (Figure 9).

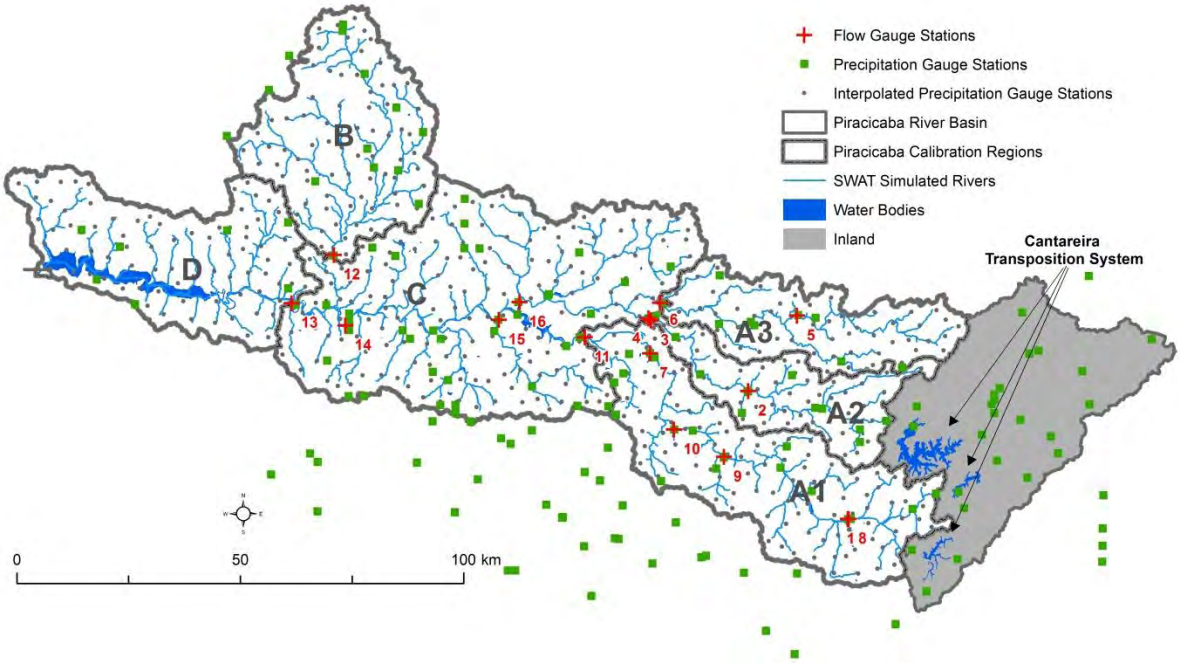

Figure 9 – Piracicaba River basin with calibration sub divisions, upstream masked out regions (inland) and flow gauge station

Parameterization was performed for the three individual regional segments (A, B and C), in all three possible levels; basin, sub-basin and HRU levels. At basin level (BSN), the following parameters were changed for all three segments. To calculate potential ET, the model offers three methods: Hargreaves, Priestley-Taylor and Penman-Monteith, where the Priestley-Taylor option was chosen after testing comparisons. Soil ET compensation factor (ESCO) was set to 0.75. Daily curve number was set to be calculated as function of plant ET, with a plant ET curve number coefficient (CNCOEF) of 0.5. Surface runoff lag coefficient (SURLAG) was set to 0.0855.

The following groundwater (GW) and HRU parameters were set individually for each region. Initial depth of water in the shallow aquifer (SHALLST); Initial depth of water

in the deep aquifer (DEEPST); Groundwater delay time (GW_DELAY); Groundwater "revap" – the movement of water from shallow aquifer to the root zone – (GW_REVAP); Threshold depth of water in the shallow aquifer for revap (REVAPMN); Deep aquifer percolation fraction (RCHRG_DP); Threshold depth of water in the shallow aquifer required for return flow to occur (GWQMN); Baseflow alpha factor – is a direct index of groundwater flow response to changes in recharge (ALPHA_BF); and Lateral flow travel time (LAT_TTIME), as seen on Table 6.

Table 6 – Groundwater (GW) and hydrological response units (HRU) parameters used for calibration of each individual region of the Piracicaba River basin

GW and HRU parameters	Regions				
	A1	A2	A3	B	C
[a]**SHALLST (mm)**	4000	4000	4000	4000	4000
[b]**DEEPST (mm)**	10000	10000	10000	10000	10000
[c]**GW_DELAY (days)**	95	95	95	95	93.23337
[d]**GW_REVAP**	0.02	0.02	0.02	0.0308	0.0734
[e]**REVAPMN (mm)**	3001	3001	3001	3001	3425
[f]**RCHRG_DP**	0.05	0.05	0.05	0.0875	0.1
[g]**GWQMN (mm)**	3000	3000	3000	3883.33	3123.333
[h]**ALPHA_BF (1 days^{-1})**	0.0587	0.055	0.034	0.0349	0.04726
[i]**LAT_TTIME (days)**	15	15	15	0.4	2.333333

[a] SHALLST is Initial depth of water in the shallow aquifer
[b] DEEPST is Initial depth of water in the deep aquifer
[c] GW_DELAY is groundwater delay time
[d] GW_REVAP is groundwater "revap" – the movement of water from shallow aquifer to the root zone – varies from 0 to 1
[e] REVAPMN is the threshold depth of water in the shallow aquifer for revap
[f] RCHRG_DP is deep aquifer percolation fraction – varies from 0 to 1
[g] GWQMN is the threshold depth of water in the shallow aquifer required for return flow to occur
[h] ALPHA_BF is baseflow alpha factor – is a direct index of groundwater flow response to changes in recharge
[i] LAT_TTIME is lateral flow travel time

Some parameters suffered changes afterwards, being updated by multiplication of a constant. In HRU levels, the average slope length (SLSUBBSN) was corrected by -4.15% in regions A1, A2, A3, B and C; and Manning's "n" value for overland flow (OV_N) was corrected by -7.66% for region B. In management practices (MGT), initial runoff curve numbers for moisture condition "II" (CN2) were corrected by -4.50% in regions A1, A2 and A3, while region B was corrected by -3.6167% and C by +3.9995%.

Other MGT parameters were tempered with, set individually for each land cover class, they are: Initial leaf area index (LAI); Initial dry weight biomass (BIO_INIT), measured in kg ha^{-1}; and total number of heat units or growing degree days needed to bring plant to maturity (PHU_PLT). Also, the maximum canopy water storage in

mm (CANMX) was set for each individual land cover class. Values are seen on Table 7.

As for management practices, parameters were set using tabular data provided by SWAT interface, where PC was considered as orange plantations, PL as range land and CR as sugarcane plantation.

Table 7 – Management (MGT) and hydrological response units (HRU) parameters used for calibration of individual land cover class

MGT and HRU Parameters	Land cover classes					
	NV10	NV20	NV30	FP	PC	PL
[(a)]LAI	2.5	3.5	5	4	4	3
[(b)]BIO_INIT (kg ha^{-1})	150	300	600	400	400	400
[(c)]PHU_PLT	3800	3800	3800	3500	3000	3000
[(d)]CANMAX (mm)	7	12	20	15	15	0

[(a)] LAI is Initial leaf area index (HARJUNIEMI, 2014)
[(b)] BIO_INIT is Initial dry weight biomass, measured in kg ha^{-1} (HARJUNIEMI, 2014)
[(c)] PHU_PLT is total number of heat units or growing degree days needed to bring plant to maturity
[(d)] CANMX is maximum canopy water storage in mm (GUIRAO; TEIXEIRA FILHO, 2013)

The performance of the models to assess calibration and validation was done through measures such as coefficient of Nash-Sutcliffe efficiency (NSE), coefficient of determination (R^2) and percentage of bias (PBIAS) (GUPTA; WAGENER; LIU, 2008; SEXTON et al., 2010; LOOPER; VIEUX, 2012).

Hydrological model divided the Piracicaba River basin into 522 sub-basins. These watersheds were further divided into numerous HRUs according to land use, soil properties and slope classes.

Calibrations results are presented on Table 8 where the 16 gauge stations used for calibration and validation are described, and on Figure 10 we present a hydrograph comparison of an observed versus calibrated gauge station. Calibration was run for a 11 year period, from 2000 to 2011, using three calibrated gauges (numbers 7, 12 and 13) while validation was done using remaining 13 gauges (Figure 9).

Table 8 – Flow gauge stations used for calibration and validation of the model and their performance: coefficient of Nash-Sutcliffe efficiency (NSE), coefficient of determination (R^2), and percentage of bias (PBIAS)

Number	Sub-basin	Station Code	Municipality	River	NSE	R^2	PBIAS
01	79	62670000	Atibaia	Atibaia	0.76	0.88	-0.88
02	69	3D-009T, P51	Morungaba	Jaguarí	0.78	0.88	4.82
03	34	62615000	Jaguariúna	Jaguarí	0.71	0.88	-7.94
04	18	3d-008_P49	Jaguariúna	Jaguarí	0.86	0.73	-2.34
05	13	3d-002	Monte Alegre do Sul	Camanducaia	0.67	0.85	3.56
06	9	3d-001	Jaguariúna	Camanducaia	0.77	0.90	-1.58
07	6	3d-003T, p59	Campinas	Atibaia	0.76	0.87	3.95
08	80	3E-63	Atibaia	Atibaia	0.65	0.84	15.81
09	40	3D-006T, P55	Itatiba	Atibaia	0.73	0.87	13.58
10	23	3D-007T, P56	Valinhos	Atibaia	0.75	0.88	9.58
11	3	4D-009RT, P57	Morungaba	Jaguarí	0.80	0.91	13.40
12	73	4D-021	Piracicaba	Corumbataí	0.66	0.84	-4.36
13	139	4D-007	Piracicaba	Piracicaba	0.83	0.92	8.73
14	175	4D-015T, P46	Piracicaba	Piracicaba	0.83	0.92	14.61
15	165	4D-010T, P47	Americana	Piracicaba	0.83	0.91	8.61
16	135	4D-013T, P58	Limeira	Jaguarí	0.79	0.91	1.36

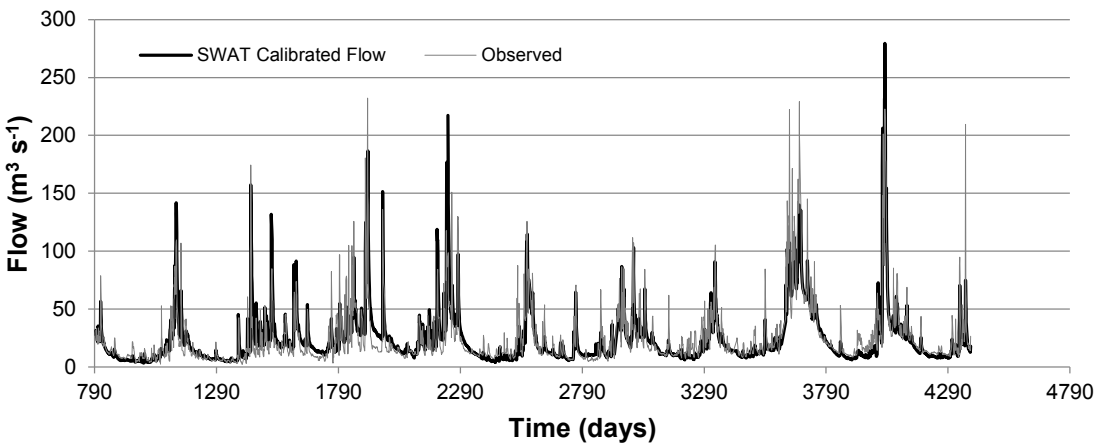

Figure 10 – Observed and calibrated flows for the gauge station 3D-003T

Observed performance indicates that all R^2 had values superior to 0.73, with a maximum of 0.92. NSE values were all superior to 0.66, where a “1” indicates a perfect model whereas zero or negative value indicates that the mean is equal or superior estimator when compared to the model. Values in the interval of 0.7 and 0.8 indicate a good adjustment, thus the model is efficient in simulating the data (WHITE; CHAUBEY, 2005; SEXTON et al., 2010).

PBIAS resulted in both negative and positive values, where negative indicates a super estimation of simulated data and positive, an under estimation, while zero is a perfect simulation. The complexity of this model, concerning size and spatial distribution, can be an explanation for these results, although still acceptable (WHITE; CHAUBEY, 2005; GUPTA; WAGENER; LIU, 2008; SEXTON et al., 2010; LOOPER; VIEUX, 2012).

Specific details on calibration and validation of the SWAT model can be found on (BRESSIANI; SRINIVASAN; MEDIONDO, 2014).

Hydrological simulation using land cover scenarios

Simulations were made through the substitution of the land cover map used during calibration stages. We used the set of thematic maps produced in the landscape modeling stage in place of the original observed 2010 land cover map, in individual model runs, using exact same parameters. Since the only change was land cover, more specifically NV, PL and CR, the SWAT outputs were comparable between themselves, allowing an evaluation between scenarios: SQ and LE, for the modelled years of 2020, 2030, 2040 and 2050, RRE for 2050 and the 2010 observed. Thus, a total of nine simulations were produced, along with another 2010 calibrated scenario. All parameters used for calibration purposes were replicated for each individual simulation.

In a later stage, we examined the simulated water processes, comparing certain hydrological indicators differences such as mean annual water yield (mm), rainy and drought season daily water yield (mm) and water regulation (variance in total, rainy and drought season daily yields). Using this method, we were able to compare the same rain events using only different land cover scenarios, thus observing potential changes in water-regulating ecosystem services.

3.2.4 Data analysis

To better understand the effects of land use practices, management, social and environmental characteristics over a regions landscape dynamics, we evaluated three scenarios consisting of different alternatives of forest dynamics. We modeled these three scenarios to evaluate the effects that these changes on forest cover could have on specific supporting and regulating ecosystem services. Supporting services are here represented as biodiversity and habitat. We measured these

through landscape structure using landscape metrics such as forest cover (m^2), patch density (number of fragments per 100ha), patch size, edge density (m/ha), core area index (%), mean shape index, proximity (dimensionless) and mean distance to nearest neighbor (m), all of which work as a proxy for biodiversity and habitat. Regulating services in the other hand are here represented by mean annual water yield (mm), rainy and drought season daily water yield (mm) and water regulation (variance and amplitude in daily water yield for total period and selected rainy and drought season), all of which work as proxy for water production and regulation, these measured through hydrological indicators. We later compare each scenario using these particular indicators, thus obtaining considerable comparison values for forest cover and potential ecosystem services.

Here we specify how simulated data from both the landscape dynamics and hydrological model were analyzed and compared to achieve information for interpretation and discussion on supporting and regulating ecosystem services.

Landscape structure of simulated scenarios

Landscape structure was quantified as to total forest cover and therefore, only NV was selected for this analysis. To analyze more specific landscape metrics we selected 70 random landscape samples of the Piracicaba River basin, distributed across the region, as seen on Figure 11. The basin was also divided into three segments called Higher Piracicaba, Middle Piracicaba and Lower Piracicaba. Segmentation aids in assessing regional differences that may be occurring. Each sample is composed of a 25 km^2 (5 x 5 km) unit, which scattered across the landscape provide an optimum pattern of possible landscape observations.

The landscape samples were intersected with the three observed land cover maps dated from 1990, 2000 and 2010 but also with each of the three land cover scenario maps, dated from 2020, 2030, 2040 and 2050. The only exception was for RRE which did not have a timed simulation and therefore is presented only for the year 2050. Thus, a total of 12 landscape metric analyses were made, only for NV class.

Landscape metrics were calculated using the V-LATE extension for ArcGIS (LANG; BLASCHKE, 2007). We selected a total of eight indexes, which aim to assess the NV cover (%), patch density (number of fragments per 100ha), patch size (ha), edge density (m ha^{-1}), core area index (%), mean shape index (dimensionless), proximity (dimensionless) and mean distance to nearest neighbor (m). Indexes provide

information and work as proxy for habitat and biodiversity, elements of supporting ecosystem services. Details on indexes can be found on Table 9.

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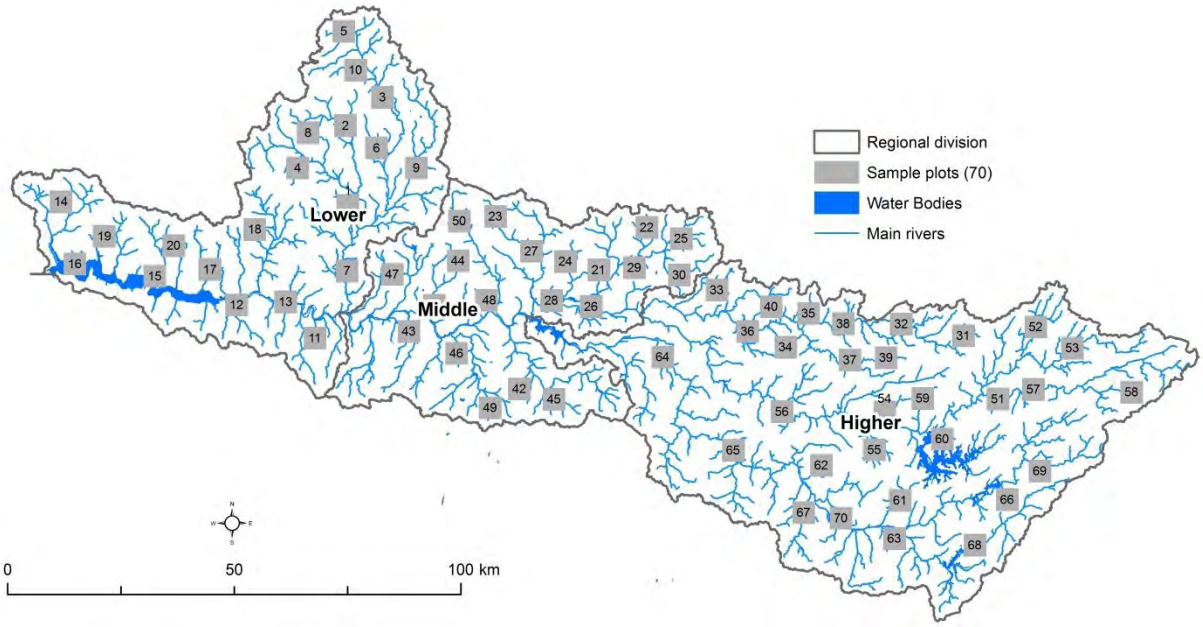


Figure 11 – Distribution of the landscape samples across the Piracicaba River basin divided into regions of Higher Piracicaba, Middle Piracicaba and Lower Piracicaba

Table 9 – Landscape structure metrics used for proposed native vegetation scenarios of the Piracicaba River basin

Name	Abbreviation	Unit	Characteristics
Native Vegetation Cover	PLAND	%	Proportion of landscape occupied by native vegetation
Patch Density	PD	n $100ha^{-1}$	Patch density normalized for 100 ha
Mean Patch Size	MPS	ha	Mean patch size
Edge Density	ED	m ha^{-1}	Edge density
Core Area Index	CAI	%	Core area proportion of landscape after a 50 m edge removal
Mean Shape Index	MSI	-	Values are greater or equal to 1. When closer to 1, represent squared patches, when higher, represent irregular shapes
Proximity	PROXIM	-	Patch proximity index, calculated for a radius of 5 km. Measures both degree of patch isolation and degree of fragmentation. The larger the corresponding value, more equal sized, contiguous and closer are the forest patches [(a)]
Mean Distance to Nearest Neighbor	ENN_MN	m	Edge to edge mean distance to nearest neighbor [(b)]

Source: [(a)] (GUSTAFSON; PARKER, 1994); and [(b)] (McGARIGAL; MARKS, 1995)

Metrics were analyzed at both landscape level (Piracicaba River basin) and at each individual segment (higher, middle and lower Piracicaba), all as mean values from their respective landscape samples.

Hydrological indicators

Considering that precipitation was the same throughout the hydrological modeling stage and that only land cover maps were exchanged, the hydrological indicators selected for comparison in each scenario were mean annual water yield (mm), rainy and drought season daily water yield (mm) and water regulation (variance in daily water yield for total, rainy and drought season daily water yield). These hydrological indicators were obtained as a proxy for water production and regulation, examples of regulating ecosystem services. Other complementary mean annual outputs were also obtained. They are surface runoff (mm), lateral soil flow (mm), groundwater (mm), deep aquifer recharge (mm), ET (mm) and potential evapotranspiration (mm).

Selected rainy season was identified as a period of 100 contiguous days with the most rainfall. The same was done for the selection of drought season, where 100 contiguous days with the least amount of rain were selected. Specific dates for rainy

and drought season comprehend precipitation records from November 18, 2009 to February 2, 2010 and July 29, 2007 and November 5, 2007, respectively.
Statistical analysis was made using software Past, where variance was obtained for the totality of 4018 logs and the respective 100 logs of each selected season.

3.3 Results

3.3.1 Landscape scenarios

Calibration and Validation of simulations

In general, transitions were found to be 90% from expansion while only 10% from new patch formation. All three scenarios were run using final selected parameters of patch formation and expansion, for each region, and can be found in Annex C.
Calibration was completed by obtaining validation through correlation maps, applying algorithms of multiple windows with constant and exponential decay function. For this validation process, we accepted the models with greater similarity between the maps, with maximum values ranging between 0.46 and 0.51 for exponential decay and 0.70 and 0.82 for constant decay. Complete results can be found in Annex D.

Scenario results

Examples of the 2050 final NV scenarios are presented in Figure 12 and total land cover is seen on Table 10. NV cover maps for each of the years and respective scenarios are presented in Appendix H, Appendix I and Appendix J. Total basin values for NV cover and age classes are presented on Figure 13.
LE presented the highest increase in total vegetation cover, with up to 43.2% of the total landscape. This was mostly due to the inexistence of forest suppression in the proposed scenario. RRE was the second highest figure, with a total of 28.4% of forest cover, meaning that at least 6.6% of the landscape today should be vegetation under the current laws. At last, SQ reached a maximum of 22.5% of NV cover, only 0.7 pp higher than the observed in 2010.
SQ and ND scenarios resulted in a concentration of forest patches in previously forested regions. Concentrations of forest patches was more observed to the West and East ends, where terrain is also more irregular. Forest aging was also observed in all scenarios.

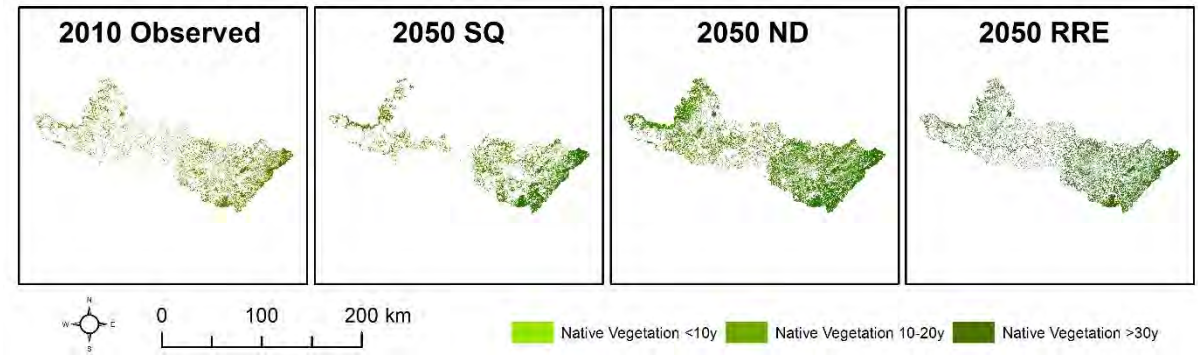
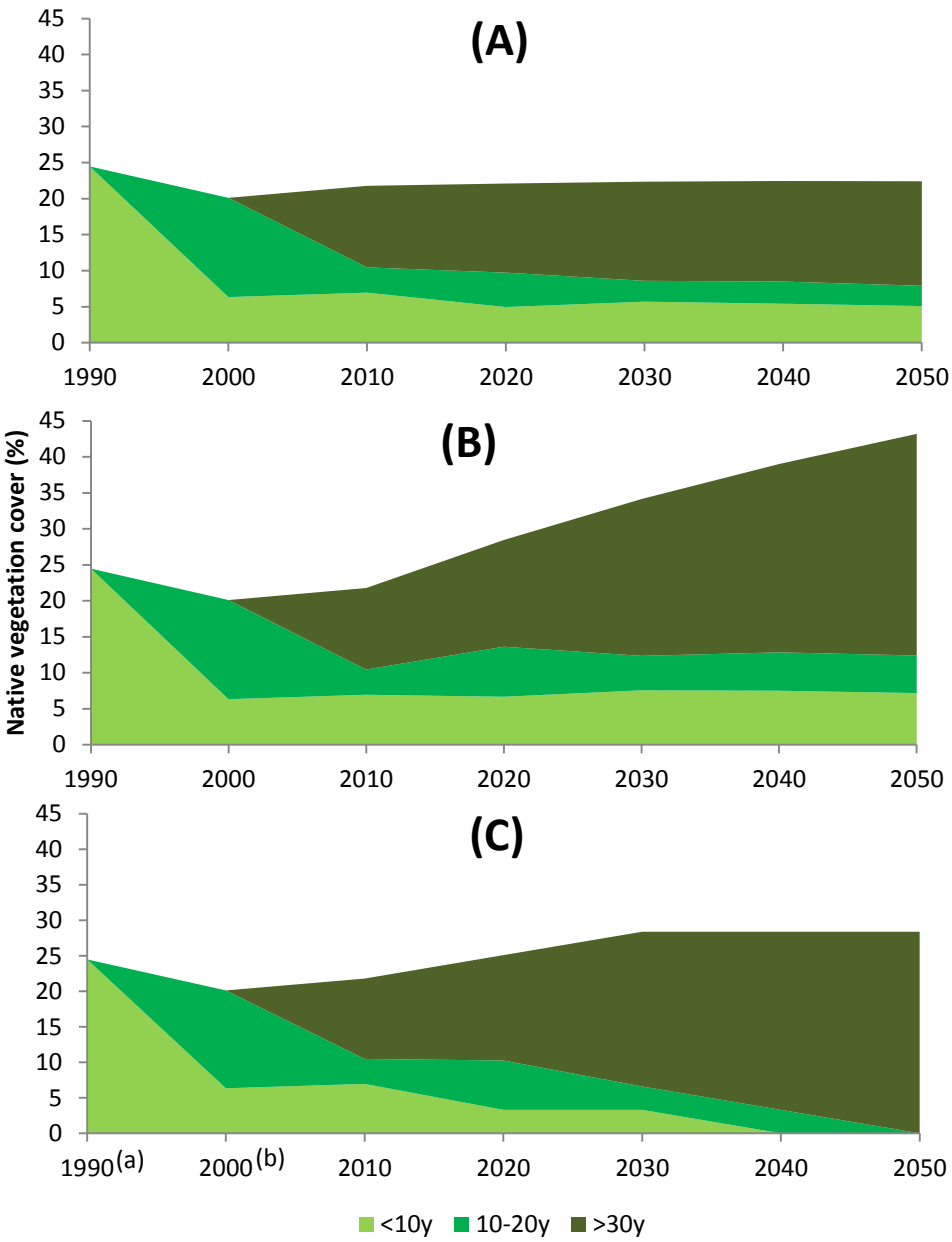


Figure 12 – Native vegetation cover maps of the 2010 observed scenario and the 2050 simulated scenarios of Status Quo (SQ), No deforestation (ND) and Riparian restoration enforcement (RRE). Native vegetation ages are set as smaller than 10 years old (<10y), 10 to 20 years old (10-20y) and 30 or more years of age (>30y)

Table 10 – Land cover (%) of observed and resulting simulated scenarios of Status Quo (SQ), No deforestation (ND) and Riparian restoration enforcement (RRE)

Land Cover	**Observed**			**SQ**				**ND**				**RRE**
(%)	**1990**	**2000**	**2010**	**2020**	**2030**	**2040**	**2050**	**2020**	**2030**	**2040**	**2050**	**2050**
[(a)]**CR**	24.0	31.3	27.2	28.4	29.6	30.7	31.7	25.4	23.8	22.4	21.1	24.9
[(b)]**FP**	2.3	3.5	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	3.9
[(c)]**WB**	1.4	1.5	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.6
[(d)]**PL**	40.1	34.5	33.7	32.1	30.7	29.5	28.5	28.7	24.6	21.2	18.3	30.4
[(e)]**UZ**	5.7	6.7	7.3	7.3	7.3	7.3	7.3	7.3	7.3	7.3	7.3	6.8
[(f)]**PC**	1.9	2.5	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.0
[(g)]**NV**	24.5	20.1	21.8	22.1	22.4	22.5	22.4	28.5	34.1	39.0	43.2	28.4

[(a)] CR is crops
[(b)] FP is forest plantations
[(c)] WB is water bodies
[(d)] PL is pasture land
[(e)] UZ is urban zones
[(f)] PC is perennial crops
[(g)] NV is native vegetation



(a) 1990 vegetation is not divided into age classes

(b) 2000 is only divided into 10 year old and 20 year old forest, due to data availability

Figure 13 – Three native vegetation scenarios are presented: Status quo (A), No deforestation (B) and Riparian restoration enforcement (C). Each scenario provides their respective native vegetation cover (%) and age proportion, where <10y is native vegetation with less than 10 years of age, 10-20y is native vegetation with 10 to 20 years of age and >30y is native vegetation with 30 or more years of age. The first three dates are of observed data, while following are simulated

3.3.2 Landscape structure

We present below our results for the eight chosen landscape metrics, with analysis made both at landscape level and regional levels for NV patches found in the 70

samples. Results are individually represented and aimed to assess NV cover (A), patch density (B), mean patch size (C), edge density (D), core area index (E), mean shape index (F), proximity (G) and mean distance to nearest neighbor (H) on Figure 14. Individual graphs can be found from Appendix K through Appendix Z. A Kruskal-Wallis statistical test with Mann-Whitney pairwise comparisons to identify significant differences between landscape metrics are presented on Annex E.

Results show significant difference in landscape structure between observed 2010 samples and 2050 scenarios. The only exceptions was for SQ scenario which did not result in a statistical difference, for most metrics, when compared to observed landscape.

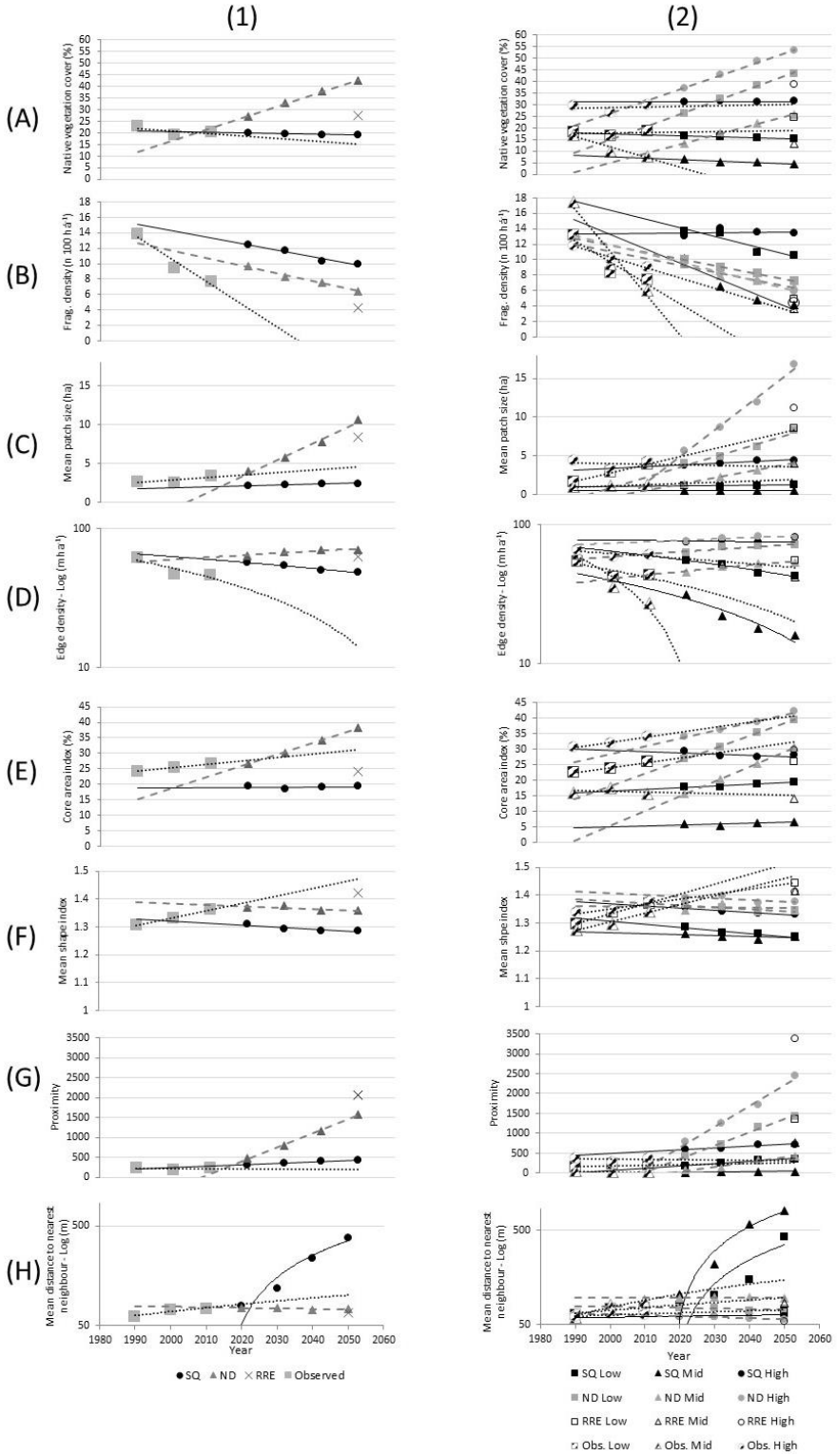


Figure 14 – Selected landscape metrics, of which column (1) represent mean values for landscape level of Piracicaba River basin and column (2) represent mean values for regional level: lower (low), middle (mid) and higher (high) Piracicaba River basin regions. Results are given in each line: native vegetation cover (A), patch density (B), mean patch size (C), edge density (D), core area index (E), mean shape index (F), proximity (G) and mean distance to nearest neighbor (H), where SQ is Status Quo, ND is No Deforestation and RRE is Riparian Restoration Enforcement

3.3.3 Hydrological simulations & indicators

SWAT was run at both annual and daily resolutions. With the extraction of the inland, land cover proportion for each scenario suffered a change due to the extraction of this portion of land, which can be seen along with mean annual water yield on Figure 15. SQ scenario revealed a small decrease while ND and RRE continued to achieve forest gain.

Annual averages for water balance components for each correspondent scenario and time step are presented on Table 11, in relation to mean annual precipitation (P) which was of 1376.9mm. Surface runoff corresponds to runoff generated in the watershed for the simulation; Lateral soil is lateral flow contribution to streamflow; Groundwater is groundwater contribution to stream; Total water yield is water yield to streamflow from HRUs in watershed; and Evapotranspiration is actual evapotranspiration in watershed for simulation.

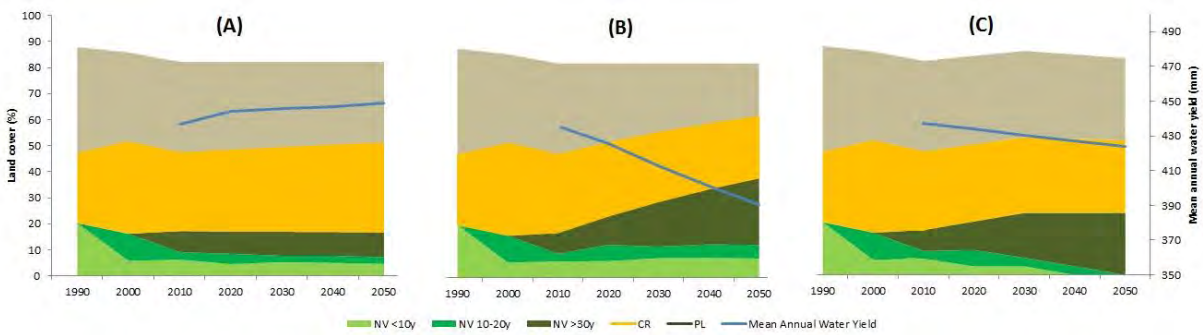


Figure 15 – Land cover (%) for the Piracicaba River basin without inland and respective mean annual water yield (mm), where A is status quo, B is no deforestation, C is riparian restoration enforcement, CR is crops, PL is pasture land, NV <10y is native vegetation with less than 10 years of age, NV 10-20y is native vegetation with 10 to 20 years of age and NV >30y is native vegetation with 30 or more years of age

Table 11 – SWAT model mean annual outputs for simulated landscape scenarios of the Piracicaba River basin in relation to mean annual precipitation (P) of 1376.9mm, where SQ is status quo, ND is no deforestation and RRE is riparian restoration enforcement

MEAN ANNUAL BASIN VALUES (% of P)	Calibrated	SQ				ND				RRE
	2010	2020	2030	2040	2050	2020	2030	2040	2050	2050
Surface Runoff	10.34	10.42	10.45	10.44	10.44	9.76	9.20	8.70	8.27	9.42
Lateral Soil	5.42	5.50	5.46	5.45	5.47	5.35	5.24	5.12	5.03	5.26
Groundwater	16.02	16.39	16.53	16.66	16.75	15.93	15.66	15.44	15.21	16.14
Total Water Yield	31.75	32.28	32.41	32.51	32.63	31.03	30.08	29.24	28.49	30.79
Evapotranspiration	67.15	66.64	66.51	66.42	66.30	67.88	68.83	69.66	70.41	68.09

Every output suffered changes with the exception of precipitation, which was the same used for all models. In an overall analysis for annual data, SQ scenario had a tendency to provide more water yield after each decade whereas ND and RRE provided less, when compared to 2010 calibrated. Surface runoff had a small increase in the first cycles of SQ, later stabilizing, while other scenarios only decreased. Lateral soil had a small increase in the first cycle of SQ and later decreased but remaining higher than 2010, while other scenarios only decreased.
Groundwater had a continuous increase in SQ and also in RRE while ND presented a continuous decrease. ET, on the other hand, had a continuous decrease in SQ while other scenarios increased when compared to 2010.
Correlations were created for comparison between forest cover, ET, runoff and water yield, concerning annual averages for all scenarios (Figure 16).

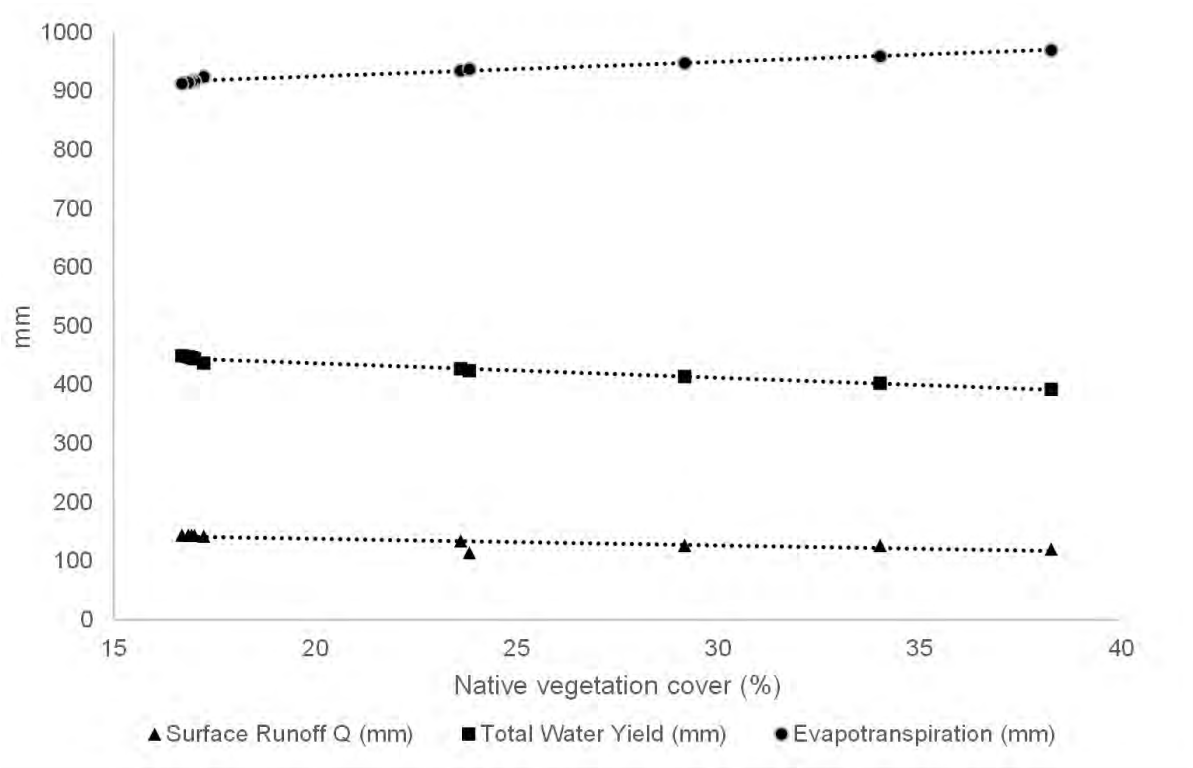


Figure 16 – Correlations for comparisons between total forest cover, evapotranspiration, runoff and water yield

Daily water yield (mm) was processed for the creation of a daily flow duration curve (m^3 s^{-1}), containing a total of 4018 data logs prevenient of 11 years of simulation (Figure 17). Results show how flow is reduced with forest cover increase.

We also created a rainy and drought season hydrographs, which exemplify 100 days of the most extreme seasons (Figure 18 and Figure 19). Results show an attenuation on peaks when forest cover is increased but also a lower water yield during drought season.

On Figure 20 we show a spread for total daily water yield and for the respective rainy and drought daily water yields, where a visible amplitude in variance difference is noticeable between ND and other scenarios, as well as observed 2010.

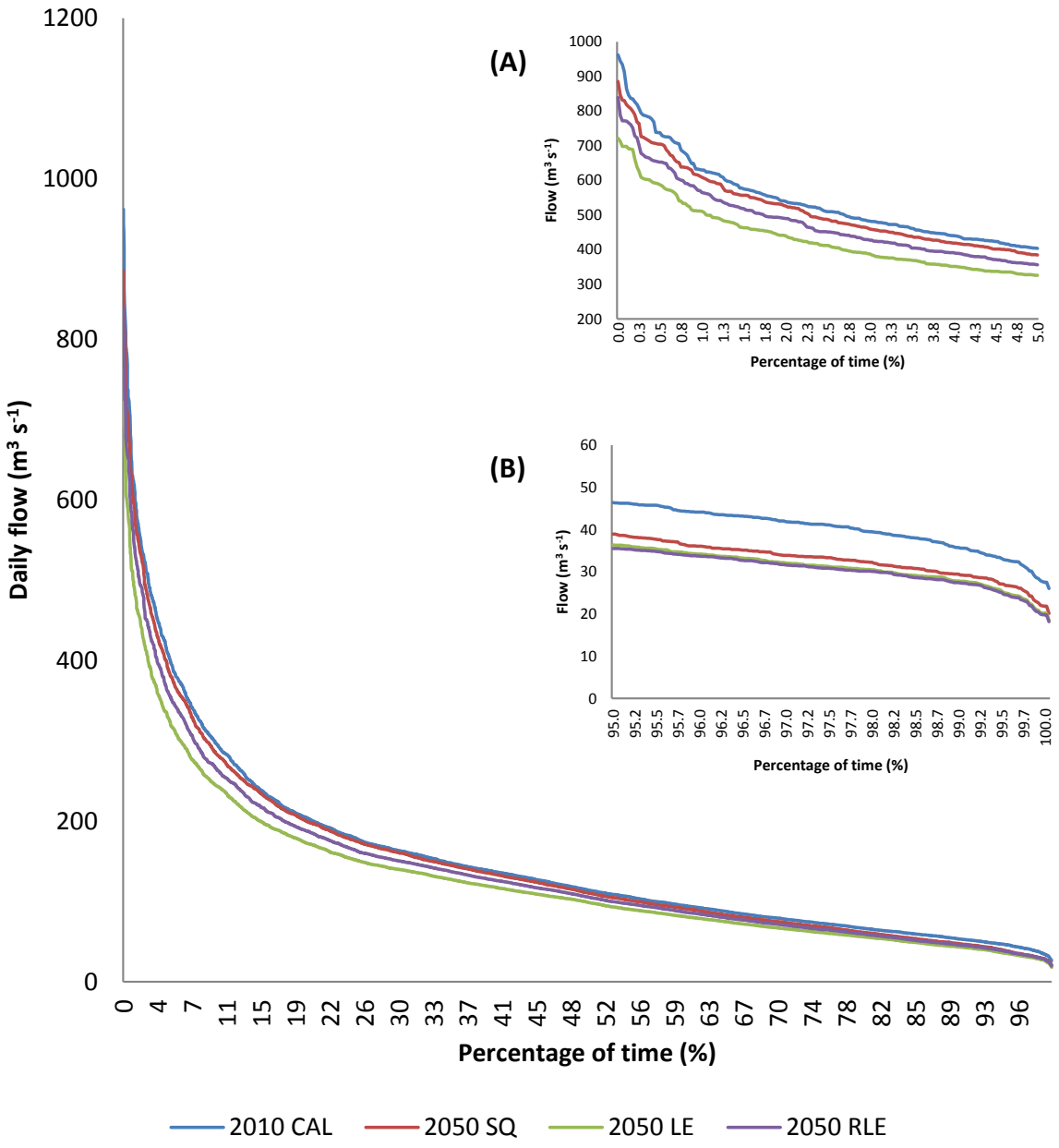


Figure 17 – Daily flow duration curve ($m^3 s^{-1}$) with a total of 11 years of processed data, with highlights for the 5% highest (A) and 5% lowest values (B), where CAL is calibrated, SQ is status quo, ND is no deforestation and RRE is riparian restoration enforcement

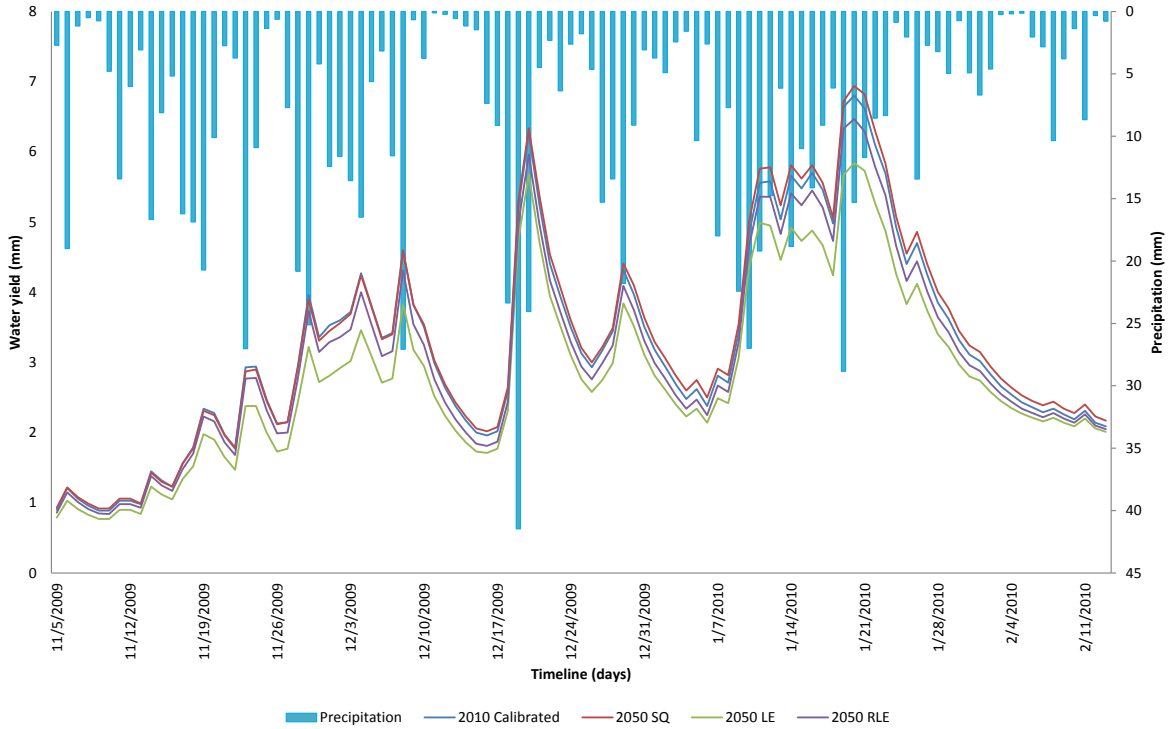


Figure 18 – Water yield (mm) and precipitation (mm) outputs of individual scenarios of the Piracicaba River basin for a selected 100 days of rainy season, where SQ is status quo, ND is no deforestation and RRE is riparian restoration enforcement

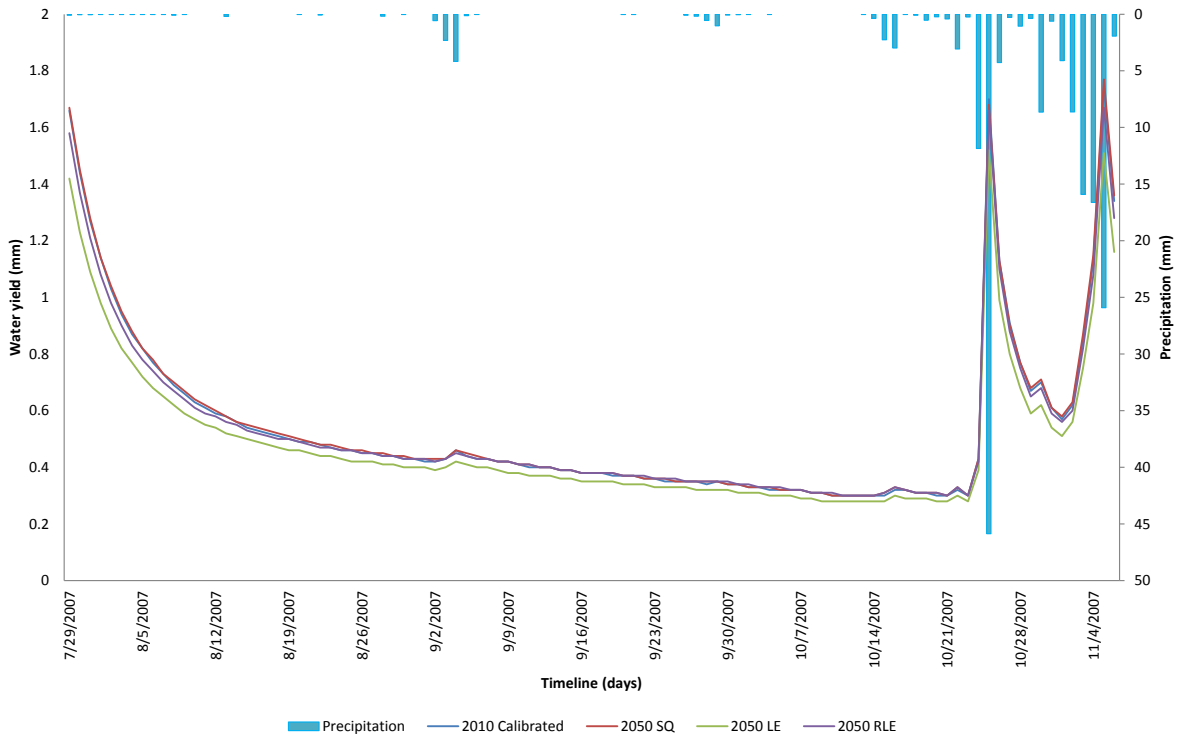


Figure 19 – Water yield (mm) and precipitation (mm) outputs of individual scenarios of the Piracicaba River basin for a selected 100 days of drought season, where SQ is status quo, ND is no deforestation and RRE is riparian restoration enforcement

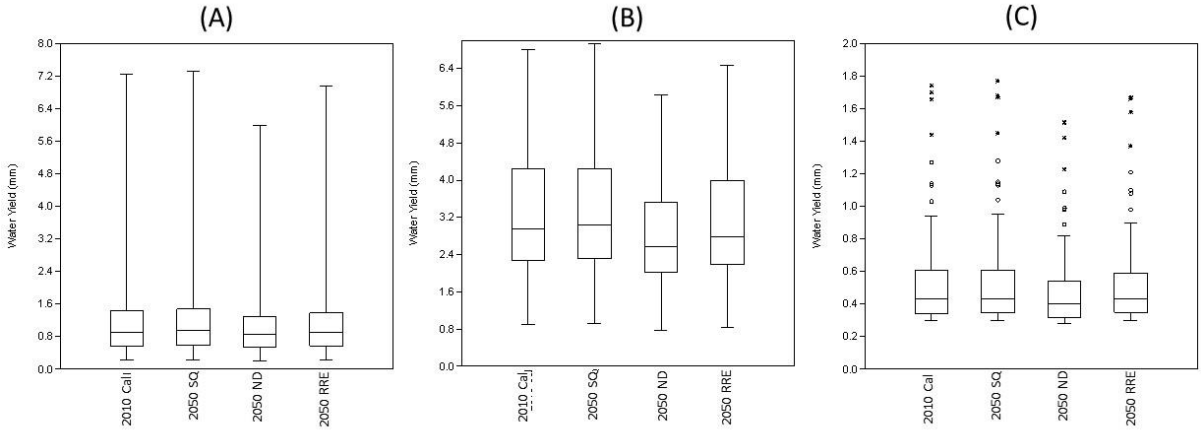


Figure 20 – Daily water yield (mm) analysis for the totality of data logs (A), 100 days of rainy season (B) and 100 days of drought season (C), where Cal is calibrated, SQ is status quo, ND is no deforestation and RRE is riparian restoration enforcement. Variance values for (A) were 0.89, 0.92, 0.63 and 0.78, (B) were 2.04, 2.14, 1.53 and 1.85 and (C) were 0.10, 0.10, 0.07 and 0.09, for each of the respective scenarios

3.4 Discussion

Landscape model validation

Calibration of our results showed that 90% of the transitions to NV occur by expansion and only 10% do so by formation of new patches. When analyzing similar regions in the Atlantic Forest, Teixeira et al. (2009) found similar values, resulting in 80% transitions by expansion and 20% by patches.

Our method of validation uses a multiple window adjustment and therefore is not a pixel to pixel validation method. The described method has the disadvantage of saturation in higher levels, generally when window sizes are higher than 13x13, as seen by Benedetti (2010). Fuzzy similarity has been shown to vary between 0.83 and 0.85 in urban regions (ALMEIDA et al., 2008), 0.87 and 0.9 in Amazonian deforested regions (XIMENES et al., 2008), 0.53 and 0.68 in native vegetation in the South of Brazil (ECKHARDT; REMPEL, 2013). Our lower values can be explained by the number of land cover classes, complexity of the transitions, generalization of the transitions in our model and especially due to all possible changes that occur and are not explained only by the selected variables. Even so, our calibration and validation were approved for use.

Landscape scenarios

The selected scenarios were developed to lead into distinct NV landscapes. SQ, a more conservative scenario, resulted in an increase of 0.3 pp in NV cover in the first decade, followed by another 0.3 pp in 2030, 0.1 pp in 2040 and a decrease in 0.1 pp in 2050. ND scenario resulted only in increases, as we expected, with 6.8 pp in the first decade, followed by 6.1 pp in 2030, 4.9 pp in 2040 and 4.2 pp in 2050. RRE scenario resulted in a total increase of 6.6 pp for a time span of 40 years, or 1.65 pp per decade.

As for forest maturity, in 2010, 60% of all forest cover was 30 years of age or older, 16.1% were around 20 year-old and 23.9% were 10 year-old patches. In SQ, this distribution resulted in, for the year 2050: 64.7% forest cover with at least 30 years of age, followed by 12.6% around 20 years of age and 22.7% around 10 years of age; and in LE, 71.3%, 12.1% and 16.6%, respectively.

As stated earlier, transitions were modelled only for the land covers CR, PL and NV as a way of simplifying the model. This simplification is also responsible for the difference between the observed transition indices obtained from the 2000-2010 period and the simulated periods, resulting in a less accentuated increase in NV over the years.

While endorsed as a more probable scenario, SQ results show how the forest regrowth tendency of the last decade is difficult to achieve in the future. Reasons for this could be due to the inexistence of areas in the future with the same variable weights observed in the 2000-2010 period. An example of this could be that CR and PL with higher slope and nearer to optimum physical & environmental conditions will tend to end since they could be already in the process of replacement by forests if have not been already replaced in the last decade. Since the model identified these features and variables as the most promising, similar to what was observed by Teixeira et al. (2009), NV increase tendency could have stabilized in a short period of time, not having enough power to increase over other regions of the landscape. In other words, the potential land abandonment observed in the 2000-2010 period could not exist anymore in the following decades.

It is possible that sugarcane mechanization, observed in the last decade and identified as responsible for land abandonment, may have already ceased in the region. Numbers show that mechanized harvesting for this crop has already reached

83% for the State of São Paulo by May 2014, which should reach 100% by 2017 (SÃO PAULO, 2014; UNIÃO DA INDÚSTRIA DE CANA-DE-AÇÚCAR, 2014). The abandonment of land due to mechanization is expected to occur in regions with slopes higher than 12%, which in turn is a potential region for forest regrowth, when in favorable environmental and climatic conditions, if not, the most likely future is degraded pasture (SEGATO et al., 2006; SÃO PAULO, 2014). Abandonment of pasture land could also be in its final days or even be cyclic but this assumption needs further studies to be conclusive. It would be necessary to model how these specific pasture landowners respond to government incentives or to other temporal reasons (HENDERSON; ANAND; BAUCH, 2013), for this may be a cause, leading to states of cyclic-forested landscapes, where suppression and regrowth happen every few years..

Also, ND although interesting, is highly impracticable as a 43.2% of NV cover in the basin is almost twice today's cover and over the 30% suggested by legislation, considering 20% of legal reserve land and additional 10% composed by permanent preservation areas. On the other hand, RRE seems plausible from a legislative point of view, endorsing the 30% also suggested by Banks-Leite et al. (2014). Considering that riparian buffers were made for a scale of 1:50,000, the same as land cover maps, and that certain features were not taken into account, the 28.4% NV cover could be even higher. Nonetheless, RRE also functions as a reminder that the Piracicaba River basin has at least 6.6% of its riparian buffer zone unprotected and therefore in need for restoration according to current local law (MARTINELLI, 2011; ZAKIA; PINTO, 2013; SOARES-FILHO et al., 2014).

Landscape structure

Observations about landscape structure were divided into large scale (Piracicaba River basin) and smaller scale (regional level) for identifying regional differences in NV structure. This division is important because of the nature of landscape dynamics that although may have resulted in an overall NV increase, may also have changed local structure over the years, not only increasing or decreasing cover but also altering patch and edge density, patch size, core area, and proximity between patches. A total of 70 samples, with 25 km^2 each, were processed for each observed and simulated scenario.

Mean NV cover resulted in a small decrease for the SQ scenario while ND and RRE presented an increase. Although SQ tendency for the selected samples is of decrease, the observed 1990-2010 tendency is even lower. This result shows that a SQ situation may not be positive, but has had improvements over the simulations.

When observed by regions, SQ scenario reveals that the higher and lower Piracicaba River basin have almost identical tendencies to their respective observed tendencies, with lower Piracicaba showing a small decrease, but both similarly stable. In the middle Piracicaba, SQ tendency reveals a decrease but with better perspectives when compared to the observed data. In both ND and RRE scenarios, the regional tendencies are of increase, led by ND.

Mean patch density resulted in overall decrease over the decades in all scenarios, although tendencies are more stable in SQ, followed by ND and RRE. The higher number of patches in SQ and ND may be the result of new patch formation or of the fragmentation of existing patches (only SQ), leading to very small patches. RRE resulted in the lowest value, as expected, due to the contiguity of the riparian buffers. In following decades, the tendency of decrease continues for all scenarios, confronting mean vegetation cover and therefore suggesting that even though the number of patches has decreased over time, total area is increasing or at least stable. Consequently, patch size in the basin may be increasing, while it is losing stepping stones, features that have been identified in earlier studies as important for habitat and biodiversity (RANTA et al., 1998; UEZU; BEYER; METZGER, 2008; RIBEIRO et al., 2009). This loss of stepping stones could also suggest that even though the model identifies small abandoned areas in CR and PL, and that these could become NV patches, in a time frame of a decade or more, these same patches could again be deforested and be converted into some other land use due to a pattern in the model to subtract small sized patches. In other words, there could be a cycle in the model where stepping stones are created due to potential land abandonment but later removed since there is little potential for small patches to exist.

When observed by regions, at landscape level, this metric reveals little or no difference. The only exception is the higher Piracicaba region, which stabilizes in the SQ scenario while all other scenarios and regions show a decrease.

Mean patch size showed an overall tendency to increase, led by ND and followed by RRE and SQ. Here, it is possible to confirm that although NV cover and density are

decreasing in SQ, patch size is increasing, indicating once more the possibility of loss of small patches through a gain in patch size. In a regional perspective, it is possible to observe how middle Piracicaba has the smallest patch sizes independently of the proposed scenario. Mean patch size for 2050 in a SQ scenario reveals a value of only 0.5 ha as compared to 4.0 ha in the same year for both ND and RRE scenarios. In the Atlantic Forest, 83.4% of all fragments are smaller than 50 ha and all together account for 20.2% of all remnants (RIBEIRO et al., 2009).

Mean edge density resulted in a decrease tendency for SQ, similar to the observed data, but in a slower pace. This decrease in SQ could be linked to the loss of stepping stones and expansion of existing larger patches, strengthening once again potential habitat and biodiversity losses. ND and RRE resulted in opposite tendencies and suggested more uneven patches or even newer patches. Increase in forest edge can be a result of forest fragmentation and can lead to a more exposed forest to anthropogenic landscape (HARPER et al., 2005), but, in this case, what could be observed is probably a growth of patches in an irregular way.

Regional results reveal similar behavior at the landscape level. An interesting but unexplained result is that higher Piracicaba, in all scenarios, reveled similar values for all studied periods.

Mean core area index resulted in a stable situation for SQ scenario while ND had a progressive increase and RRE had a small decrease but remained mostly stable when compared to the observed data. This situation suggests that SQ and RRE patches are not growing in all directions and therefore not gaining more core area, contrary to ND. Thus, SQ could be resulting in NV patches equivalent to riparian buffer zones, more stretched, rather than round or squared. When observed by regions, SQ tendencies differ, having higher Piracicaba tending to decrease while mid and lower Piracicaba tend to increase core area. It is interesting that mid Piracicaba values are much lower when compared to any other scenario and region. A plausible explanation is that patches that were lost had significant core area. Larger core area lead to larger fragments that promote more conservation if also associated with forest maturity. Large mature forest patches have their importance for seed dispersal, allowing neighboring patches to exchange genetic material and even recolonization of surrounding abandoned land (RODRIGUES et al., 2009).

Mean shape index reveals that observed data had a high tendency to increase in value and therefore become more elongated, similar to riparian buffer zones.

Scenario results showed that SQ and ND had similar tendencies of decrease, although ND revealed a more stable situation when compared to 2010 data. RRE resulted in the highest values, as expected, due to the inclusion of riparian buffers. Similar observations are made at regional levels. Even so, all values, except RRE, are below 1.4 and therefore more squared. This may seem a good observation at first, but when we take into account that patches have small core area and therefore are of small size, habitat and biodiversity may lose their effect (TURNER; CORRETT, 1996).

Proximity results for observed data show a stable situation while all of the modeled scenarios reveal a tendency to both increase in size and become closer to each other. Although they all show the same tendency, it is greater for ND and RRE while SQ is more similar to the tendency observed in the last decade. When observed by regions, mid Piracicaba reveals an accentuated decrease tendency, unlike other scenarios. Again, results suggest a tendency for increase in habitat conditions and biodiversity but when other metrics are also compared, regional division suggests that the situation tends to become better at higher Piracicaba, followed by lower and middle Piracicaba.

Mean distance to nearest neighbor reveals a small tendency of increase in observed maps, a result that reveals that patches are either being lost or growing and connecting with each other, and thus becoming one. The SQ scenario resulted in a high tendency to increase distances between patches, up to 375 m while ND and RRE resulted in a small decrease, both maintaining values between 65 m and 75 m, similar to what was observed. Regional results follow the landscape level tendencies, with a highlight to mid and lower Piracicaba final values that reveal the highest mean distances, suggesting once again the loss of stepping stones and possible concentration of forest patches. When compared to values found by Ribeiro et al. (2009), these values are considerably smaller, since they reported an average distance of 1,400 m between Atlantic Forest fragments, considering also small fragments.

Implications for supporting ecosystem services

Results presented here suggest that, in the SQ scenario, landscape is losing small forest patches and therefore, stepping stones, which would act as connections between larger patches in the landscape. Stepping stones have been identified in

earlier studies as important features for habitat and biodiversity (RANTA et al., 1998; UEZU; BEYER; METZGER, 2008; RIBEIRO et al., 2009). Since forest cover is still similar, a plausible assumption is that small patches are being lost while larger patches are increasing size, in a stretched pattern, sometimes merging. Subsequently, larger patches are being protected and even increasing, strengthening their high potential for conservation (RIBEIRO et al., 2009; RODRIGUES et al., 2009). The other two scenarios are both more promising for supporting ecosystem services, when compared to SQ. ND reveals no forest suppression effect and thus stepping stones only tend to grow and even merge, promoting more habitat and potentially more biodiversity in the long run. As for RRE, the riparian forest acts as corridors for terrestrial and aerial wildlife and habitat for aquatic wildlife (SCHLOSSER, 1991; SEMLITSCH; BODIE, 2003; MEYER et al., 2007; GILLIES; CLAIR, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; UEZU; BEYER; METZGER, 2008). As numbers have shown, with forest cover increase, followed by aging and maturity, an increase in biodiversity is expected, especially in circumstances where there is growth in core area (METZGER et al., 2009) and biological fluxes and genetic exchange through seed dispersal are possible (GUEVARA; LABORDE, 1993; GALINDO-GONZÁLEZ; GUEVARA; SOSA, 2000).

Results also suggest that there is a high degree of spatial heterogeneity among these patches. Higher Piracicaba is by far the most promising of the regions for biodiversity and habitat, with highest forest cover, patch core area, connectivity, and larger patches. On the other hand, mid Piracicaba showed the lowest values, presenting the lowest forest cover, along with more stretched patches, and cases of connectivity loss and potential stepping stones, thus resulting in a longer distance between patches.

Mid Piracicaba is inserted in one of the main regions of sugarcane production in Brazil (EMBRAPA, 2014). Sugarcane plantations are well known for their traditional harvesting which involves pre-burning followed by manual harvest (SEGATO et al., 2006). Over the years, this practice has been pointed out as harmful for many reasons, especially regarding environment and health related concerns (CANCADO et al., 2006; MARTINELLI; FILOSO, 2008). Legislation and the “Green Ethanol” Protocol led to the extinction of pre-burning and also extinction of sugarcane plantations in fields with slope greater than 12% (EMBRAPA, 2014; SÃO PAULO, 2014), what has eventually promoted abandonment of small portions of land and

potential forest regrowth. It is possible to state that the impacts of sugarcane production could have affected biodiversity and habitats (MARTINELLI; FILOSO, 2008) in a way that there is little to no resilience potential left, different from what was observed by Cheung, Liebsch and Marques (2010) in pasture land.

Hydrological indicators

SQ resulted in a small decrease in NV cover (0.5 pp) and some aging, where NV older than 30 years were accounted for 9.5% cover, which reflected on an increase of 12 mm in mean annual water yield. In this same scenario, PL and CR suffered an inversion in land cover predominance around the year 2030, thus with a tendency for sugarcane plantations to increase while pastures decreased. Although not analyzed in this study, sugarcane increase could potentially be affecting water consumption and consequently interfering with water yield, although more probable alterations to the environment reflect on soil degradation, deteriorations of aquatic systems and riparian ecosystems (MARTINELLI; FILOSO, 2008). In this scenario, by 2050, surface runoff increased by 0.1 pp, as did lateral soil (0.05 pp) and groundwater (0.73 pp) while ET decreased by a total of 0.85 pp, when compared to a calibrated scenario of 2010 and in relation to total precipitation.

LE scenario resulted in a projection of over 38% NV cover and a decrease in both PL and CR but still achieving an inversion between both. Age proportion for NV resulted in a total of 25.7% cover for 2050, with 30 years or more of age, as compared to 8.1% in 2010. Despite this result in land cover, mean annual water yield decreased 45 mm by 2050, contemplating also an increase in mean vegetation age increase. In this scenario, by 2050, surface runoff decreased by 2.07 pp, as did lateral soil (0.39 pp) and groundwater (0.81 pp). ET, on the other hand, increased by a total of 3.26 pp, when compared to a calibrated scenario of 2010 and in relation to total precipitation.

RRE scenario increased NV cover up to around 24% while mean annual water yield dropped 13 mm. Age was purposely set to be 30 years or more for the totality of the cover, in 2050, when hydrological simulation was run. CR and PL values decreased when compared to 2010 but remained similar to each other, indicating that when vegetation was buffered around riparian areas, both CR and PL were occupying this land. In this scenario, by 2050, surface runoff decreased 0.92 pp, as did lateral soil

(0.16 pp). Groundwater and ET increased 0.12 and 1.75 pp, respectively, when compared to a calibrated scenario for 2010 and in relation to total precipitation.

In all scenarios and time frames, water yield was found to be between 28% and 33% of total precipitation, which was 1,376.9mm. Surface runoff values were between 8% and 11%, lateral soil between 5% and 6%, groundwater between 15% and 17% and ET values were between 66% and 71% of total precipitation. These values comply with classic findings from Hewlett and Hibbert (1967), although they were made for smaller watersheds in temperate humid climate.

NV cover, when correlated to ET, runoff and water yield resulted in R^2 above 0.91, indicating that the model responded adequately to increases and decreases in forest cover. In a general analysis, it is possible to infer that for this basin, in these conditions, forest gain provides less water yield, corroborating with historic findings (HIBBERT, 1965; BOSCH; HEWLETT, 1982; BURT; SWANK, 1992) and contradicting popular myths that imply that more forest cover provides more water yield (ANDRÉASSIAN, 2004; CALDER, 2004, 2007).

When analyzing the daily flow duration curve (Figure 17), it is possible to visually interpret and conclude that SQ scenario resulted in similar outputs for the 2010 calibrated landscape, except for the 5% lowest values. RRE resulted in lower values of flow, followed by LE, indicating once again the influence of vegetation cover and possibly vegetation maturity on water production (BROWN et al., 2007, 2013). From Figure 18, we can clearly distinct differences, observing that LE, with around 40% forest cover, produces shorter peaks when compared to 2010 calibrated data in the rainy season. SQ, which resulted in higher values of mean annual water yield, correspondingly resulted in higher values when analyzed on a daily resolution, with occasional peaks higher than calibrated 2010. Furthermore, variance values from both the totality of daily data and rain season daily data show us that ND suffers more regulation when compared to any other scenario and calibrated 2010. Thus, these results support indications once again that forest cover act to attenuate potential flash floods, promoting more interception and infiltration, delaying or even extinguishing surface runoff (PIERCE, 1967; ARCOVA; DE CICCO; ROCHA, 2003; DE FREITAS et al., 2013).

When analyzing the drought period hydrograph (Figure 19), we observe once again the effects of forest cover over water production when comparing the ND scenario to others. ND resulted in a means of 0.49 mm of water yield for the selected 100 days,

while calibrated 2010 resulted in 0.54 mm, SQ in 0.54 mm and RRE resulted in a means of 0.53 mm. Variance for this period was also lower for ND (0.07), followed by RRE (0.09) when compared to calibrated 2010 (0.10) and SQ (0.10). Hence, these results support an indication that for these conditions, higher quantities of forest cover react in a way to lower water production in the drought season while maintain substantial water regulation. A composition of studies analyzed by Calder (2004) show that the popular belief that afforestation increases drought season flows is at the same time true and false as these effects are likely to be very site specific. Therefore, it is possible that at a sub-basin level, some regions could have opposite results.

Implications for regulating ecosystem services

NV cover increase in the basin can have both positive and negative side effects and implications for regulating ecosystem services questioned in this study. On one hand, forest increase proved to be regulating water flow in both the rainy and the drought season, with amplitudes and variances complying with this result. On the other hand, forest increase resulted in lower water yields considering annual means, denoting that water production tends to be lower with forest increase. The SQ scenario increased mean annual water yield by as much as 2.8%, while ND decreased by as much as -10.3% and RRE decreased by -3.0%.

Although forest may seem as the most important land cover for regulation of water yield, other vegetation may as well be significant. However, we did not question in this study, the consequences of other land cover changes over hydrological indicators. SWAT has been identified as very sensitive to applied crop rotation and sometimes even small variations of crop management practices (ULLRICH; VOLK, 2009). This implies that there is even more potential for water production and regulation through a combination of agricultural practices that in a long run can improve even better the results presented here.

We question if the inland region, known as the upriver Cantareira Transposition System, should be focused on gaining forest or simply maintaining the present state cover and improving only riparian corridors. The Cantareira Transposition System is already a water regulating measure (CHIODI; SARCINELLE; UEZU, 2013) and upriver forest cover should act as water quality control, especially for sediments, delivering higher amounts of water to the system.

As for the lower region of the basin, the recent Piracicaba, Capivarí and Jundiaí Watershed Plan (AGÊNCIA DAS BACIAS-PCJ, 2011) states that the $Q_{7,\ 10}$ for the Piracicaba River is 35.76 $m^3\ s^{-1}$ while $Q_{available}$ (for water harvesting) is 32.10 $m^3\ s^{-1}$. According to our simulations, for the calibrated reality in 2010, in 99.7% of the time, there would be water available in these amounts. As for scenario simulations for 2050, this amount would only be available in 98% of the time for SQ, 96.8% for ND and 97.7% for RRE. Also, according to Agência Das Bacias-Pcj (2011), there is a tendency of 7.4% population growth from 2014 to 2020, but also a 7.4% increase in urban demand for water, 7.2% in industrial demand and 3.4% in agricultural demand. At these rhythms, conflicts on demands for water are soon to be noticed.

Summary for implications on ecosystem services

A summary of the main observations and implications for each scenario are presented in Figure 21.

Scenarios	Observed Occurrences	Implications on Supporting Services	Implications on Regulating Services
SQ	Forest cover of 22.4% along with some aging; when analyzed without inland, cover decreased to 16.7%; loss of small patches; mean patch size of 0.5 ha; patch increase tendency is less elongated when compared to 2010; patches become farther apart; mean annual water yield increase; higher daily water yield variance	Loss of stepping stones; consequent potential loss of biodiversity and habitat; negative influence on provisioning and regulating services	Increase in water availability; decrease in flow regulation;
ND	Forest cover of 43.2% along with substantial aging; when analyzed without inland, increase to 38.3%; mean patch size of 4 ha; patch increase tendency is similar to observed in 2010; patches unify and become closer; mean annual water yield decreased; lower daily water yield variance	Improved connectivity between patches; improved core area; consequential potential gain in biodiversity and habitat; positive influence on provisioning and regulating services	Decrease in water availability; increase in flow regulation
RRE	Forest cover of 28.4% along with total aging; when analyzed without inland, cover was of 23.8%; riparian corridors were all afforested; mean patch size of 4 ha; elongated patches; mean annual water wield decreased; lower daily water yield variance	Improved connectivity between patches; improved core area; consequential potential gain in biodiversity and habitat; positive influence on provisioning and regulating services	Decrease in water availability; increase in flow regulation

Figure 21 – Summary of observed occurrences and implications for ecosystem services found for each of the proposed scenarios

3.5 Concluding remarks

Our results have shown that future forest scenarios show discrepancy in forest cover due to the formulation of drivers. Also, an SQ approach showed that it is plausible that tendencies found for the 2000-2010 period may not be as promising in the next periods due to the inexistence of similar trajectories and conditions (e.g. high slope regions in PL or CR, near forests and water). Although the SQ scenario has not resulted in high increases of NV cover as have ND and RRE, many other drivers may

be explaining transitions. Also, economic, cultural and social trends change over time and could revolution future scenarios. Even so, mean forest age has progressively increased, potentially resulting in more mature forests as compared to 2010.

For the ND scenario, the numbers show how a drastic measure of law enforcement can potentiate forest cover within one or two decades, reaching desired scenarios of forest cover and maturity. Likewise, RRE reveals the potential riparian reforestation scenario with the aid of legislation and also with a desired final cover and age.

Regarding ecosystem services related to biodiversity and habitat, results suggest that SQ landscape is losing small patches of forests that act as connections between larger patches in the landscape. ND and RRE have increased patch size and even have improved structural connection. Regional observations of these features suggest that the higher Piracicaba basin is by far the region with larger patches, more connections, and greatest proximity between patches, therefore, being a positive sign that biodiversity and habitat conservation could be stronger. The middle Piracicaba basin, where sugarcane plantation industry is predominant, presents the smallest forest cover, along with smaller patches, generally more stretched with small or inexistent core areas. It also resulted in the greatest loss of what could be stepping stones, what means longer distances between patches and certainly a loss for habitat and biodiversity potentials.

Regulating ecosystem services, related to water production, revealed that the NV cover increase in the basin can have implications. One of the consequences show that forest cover increase, in either the rainy or drought seasons, can be more effective for the regulation of water yield when measured through variance. They also indicated that flow peaks can be attenuated, lowering the power of potential flash floods. On the other hand, forest increase resulted in lower water yields considering annual means, which implies that water production tends to be lower with forest increase, even in the drought season.

All scenarios presented different outcomes. It is necessary, now, to weigh these differences and apply the available knowledge into practical planning and management of the landscape, contemplating expected results that improve necessary ecosystem services.

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4 CONCLUSIONS

Forest transition is indeed occurring, although partially restricted to certain regions of the landscape such as the sub-basins Jaguarí and especially Atibaia, comprehending the Higher Piracicaba, a region that had larger quantities of forest cover, mean annual precipitation and less extensive agricultural systems. Even so, the observed transitions can only be associated with the past decade and respective variables. Observed forest suppression was indeed associated with anthropogenic characteristics while forest regrowth was related to environmental and physical conditions, both influenced by multiple drivers, working in synergy. Forest suppression has targeted both older mature and newer less biodiverse forests.

Future projections, made considering matrix transitions, reveal that forest gain may come in a slow pace, followed by specific ecosystem losses or lag in increase, due to continuous trends of older mature forest loss.

Land abandonment in both CR and PL may be a leading cause for forest regrowth, stimulated by other factors such as legislation or social causes. Understanding the drivers of native forest increase is essential in order to comprehend the potential for landscape restoration and potential reestablishment of ecosystem services.

When landscape structure was analyzed, these differences in regions and in scenarios were clear, especially when comparing SQ to ND and RRE. Discrepancy in cover was due to specific drivers and regional characteristics. Landscape structure was affected especially by patch density and connectivity, revealing that SQ scenario has a tendency to loose small forest patches while larger patches increased in size, in a stretched pattern. In the LE, as expected, NV cover tended to increase, but localized, with connections between patches and emergence of new patches, almost always with a stretched pattern. The RRE also revealed, as expected, a stretched pattern but with greater potential for connectivity between larger patches.

Also, an SQ approach revealed that it is plausible that tendencies found for the 2000-2010 period may not be as promising in the next periods due to inexistence of similar trajectories and conditions (e.g. high slope regions in pasture or crop land, near forests and water). Even so, aging of forest patches was observed to a point where 64.7% of all NV in 2050 would be older than 30 years. The ND and RRE approaches showed how strict law enforcement allied to effective surveillance have potential to drastically increase forest cover in only one or two decades, reaching a desired cover of 30%, with 63.8% and 76.8% of all forests older than 30 years in 2030.

Potential effects on ecosystem services related to biodiversity and habitat were all considered structural. Our results suggest that the SQ scenario is losing small patches of forest that act as connections between larger patches in the landscape, also known as stepping stones. ND and RRE, on the other hand, only gain forest cover, promoting better patch connection and core area increase.

When observed by regions, SQ reveals that the Higher Piracicaba basin has larger patches, more connections, and greater proximity between patches, therefore being a positive sign that biodiversity could be higher and habitat conditions could be improved. As for the middle Piracicaba basin, region where sugarcane plantation industry is predominant, forest cover is smaller, along with smaller patches, generally more stretched with small or inexistent core areas. It also resulted in the greatest loss of stepping stones and longer distances between patches, implicating in a loss for habitat and biodiversity potentials.

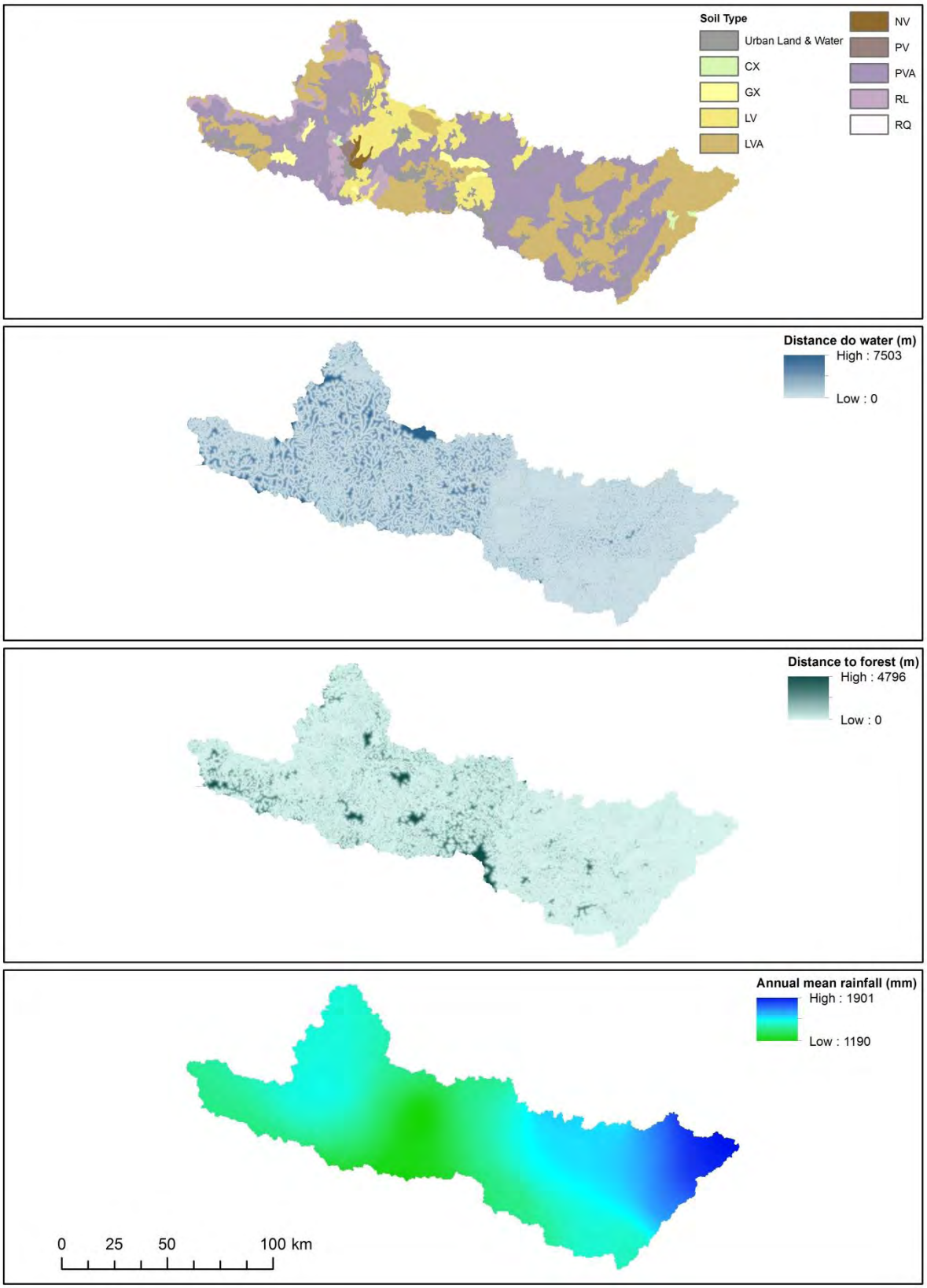
As for hydrological indicators, mean annual water yield decreased with forest cover increase although regulation was observed. Mean annual water yield increased in the SQ scenario by as much as 2.8%, decreased by as much as -10.3% in the ND and decreased it by -3.0% in the RRE. We also observed lowering of flow peaks in the ND and RRE scenarios, while, in some cases, there was an increase in flow peak in the SQ.

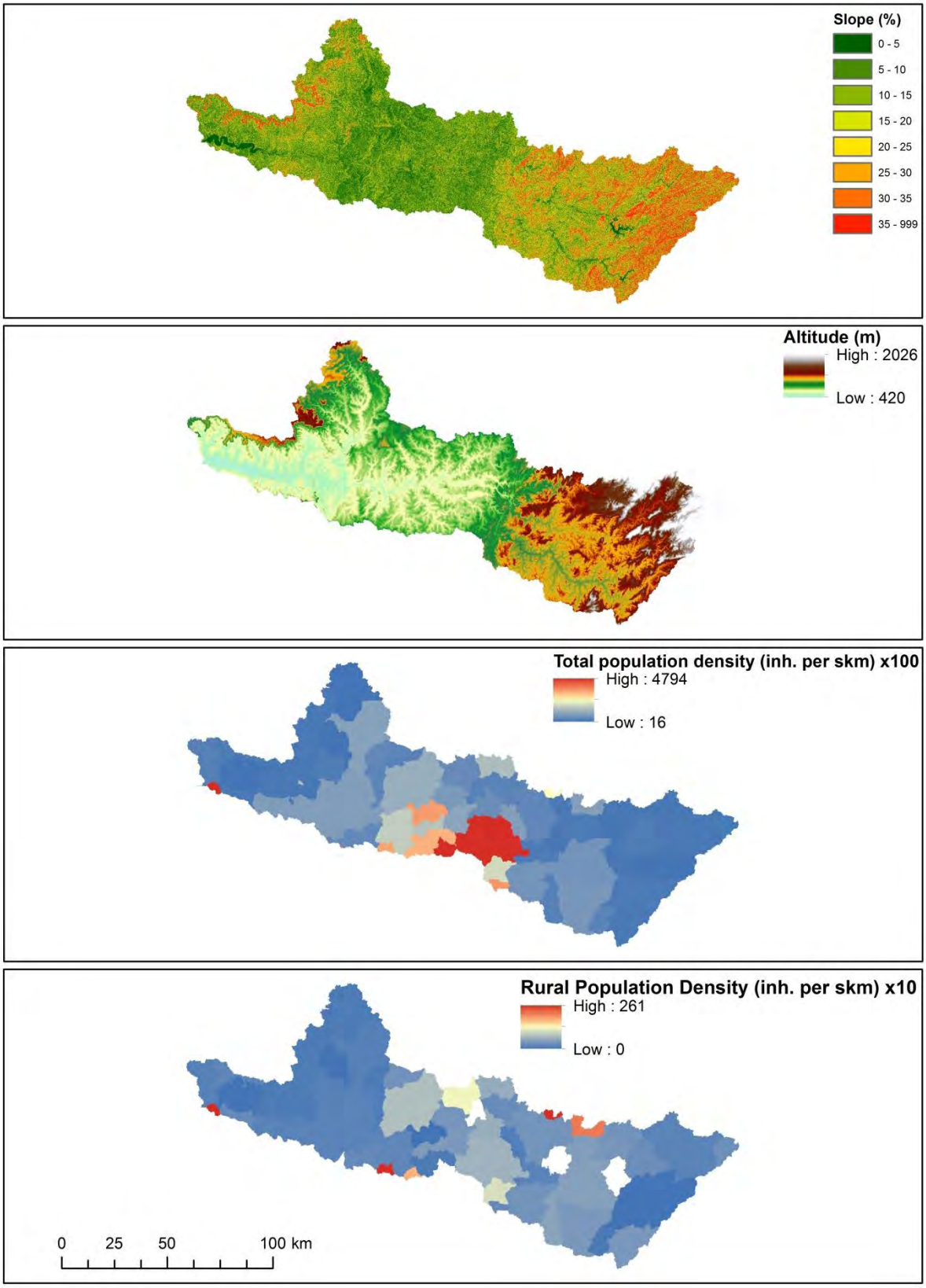
Regulating ecosystem services, related to water production and regulation, revealed that NV cover increase in the basin can have thresholds, showing the greatest benefits for a selected time period and need. Forest cover increase, either in the rainy or in the drought season, can be more effective for water yield regulation. It also indicates that flow peaks can be attenuated, making potential flash floods less likely to occur. Forest cover increase can also result in lower water yields considering annual means, implying that water production tends to be lower with specific forest cover thresholds, even in the drought season.

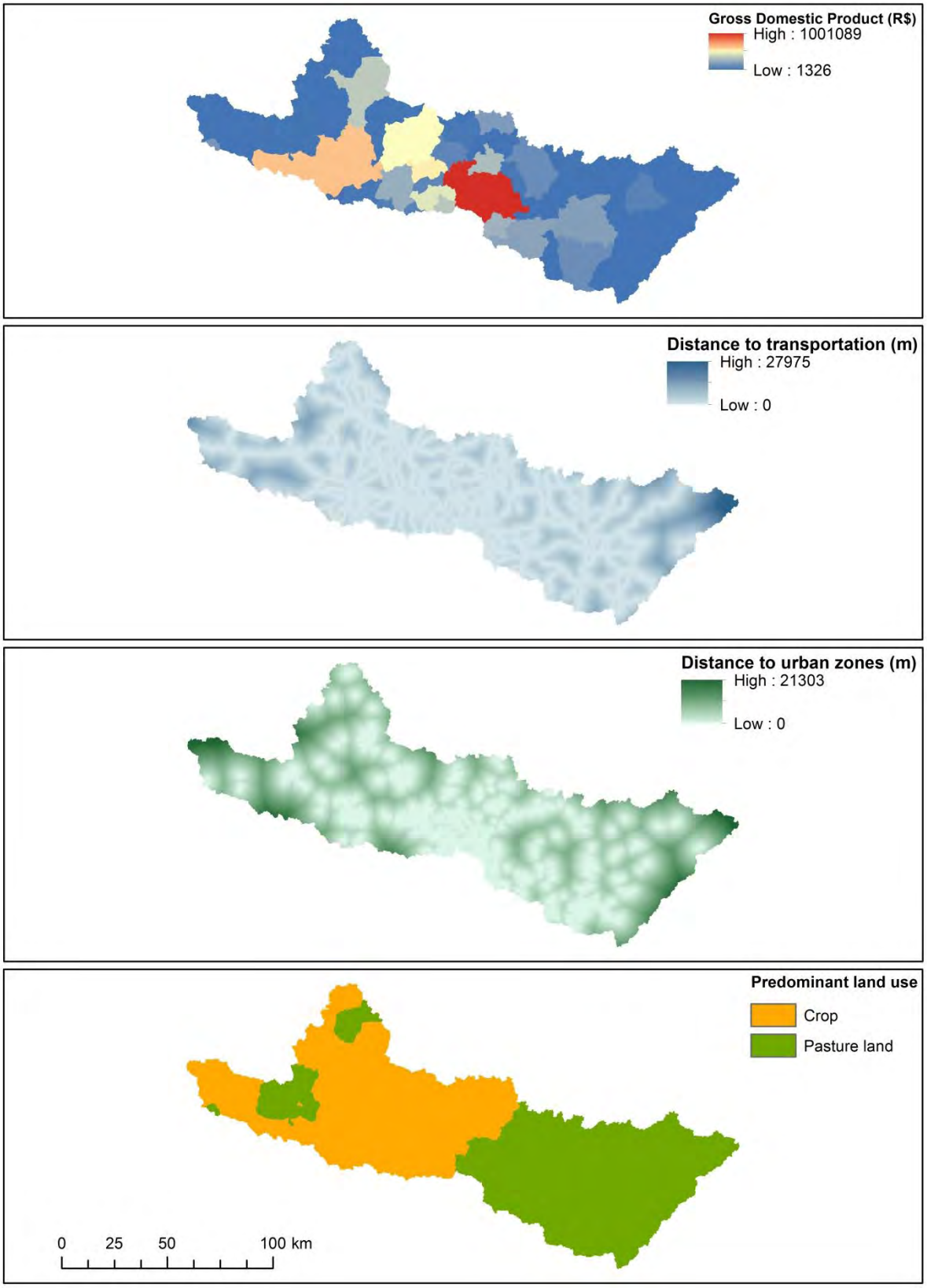
It is important now to question which mechanisms will guarantee that forests that regrow are not once again deforested in the future; which approaches should be taken to guarantee equal chances for forest transition in all regions of the basin; if there are additional management strategies to improve specific ecosystem services; and if management strategies should be taken at basin scale or at sub-basin scale.

APPENDIX

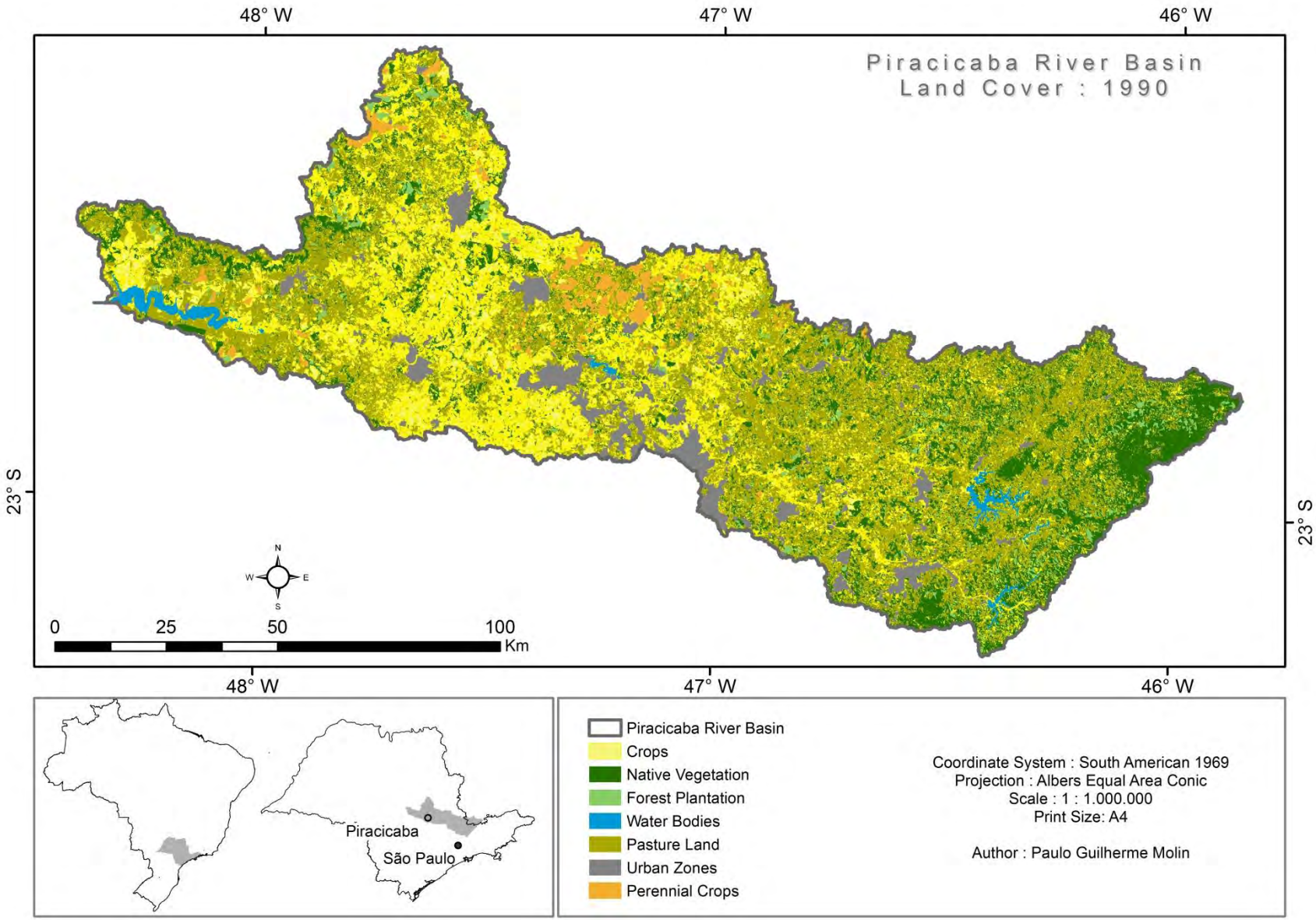
Appendix A – Dinamica EGO variables for dynamic landscape modeling



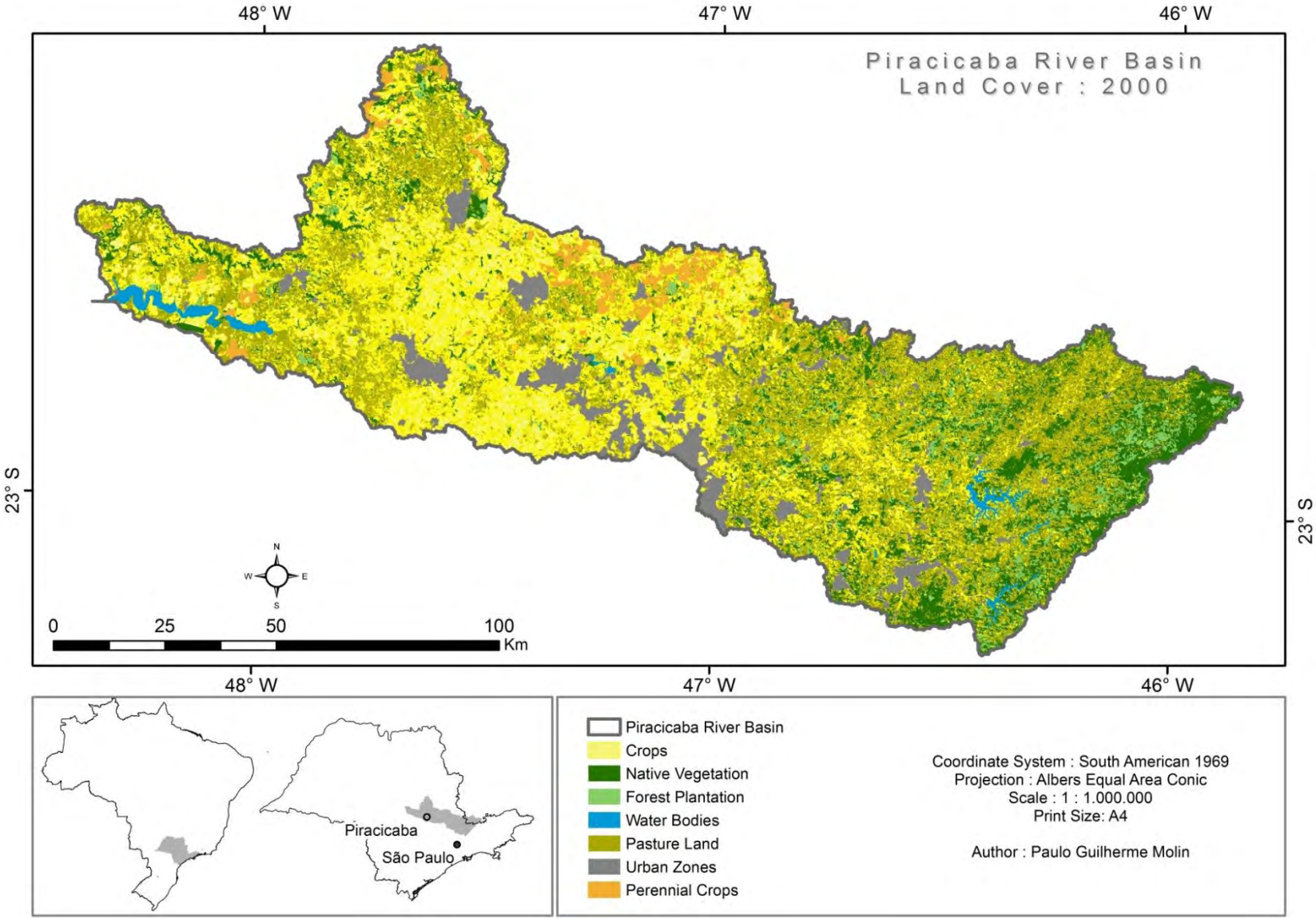

Slope (%)
0 - 5
5 - 10
10 - 15
15 - 20
20 - 25
25 - 30
30 - 35
35 - 999
Altitude (m)
High : 2026
Low : 420
Total population density (inh. per skm) x100
High : 4794
Low : 16
Rural Population Density (inh. per skm) x10
High : 261
Low : 0
0 25 50 100 km


Gross Domestic Product (R$)
High : 1001089
Low : 1326
Distance to transportation (m)
High : 27975
Low : 0
Distance to urban zones (m)
High : 21303
Low : 0
Predominant land use
Crop
Pasture land
0 25 50 100 km

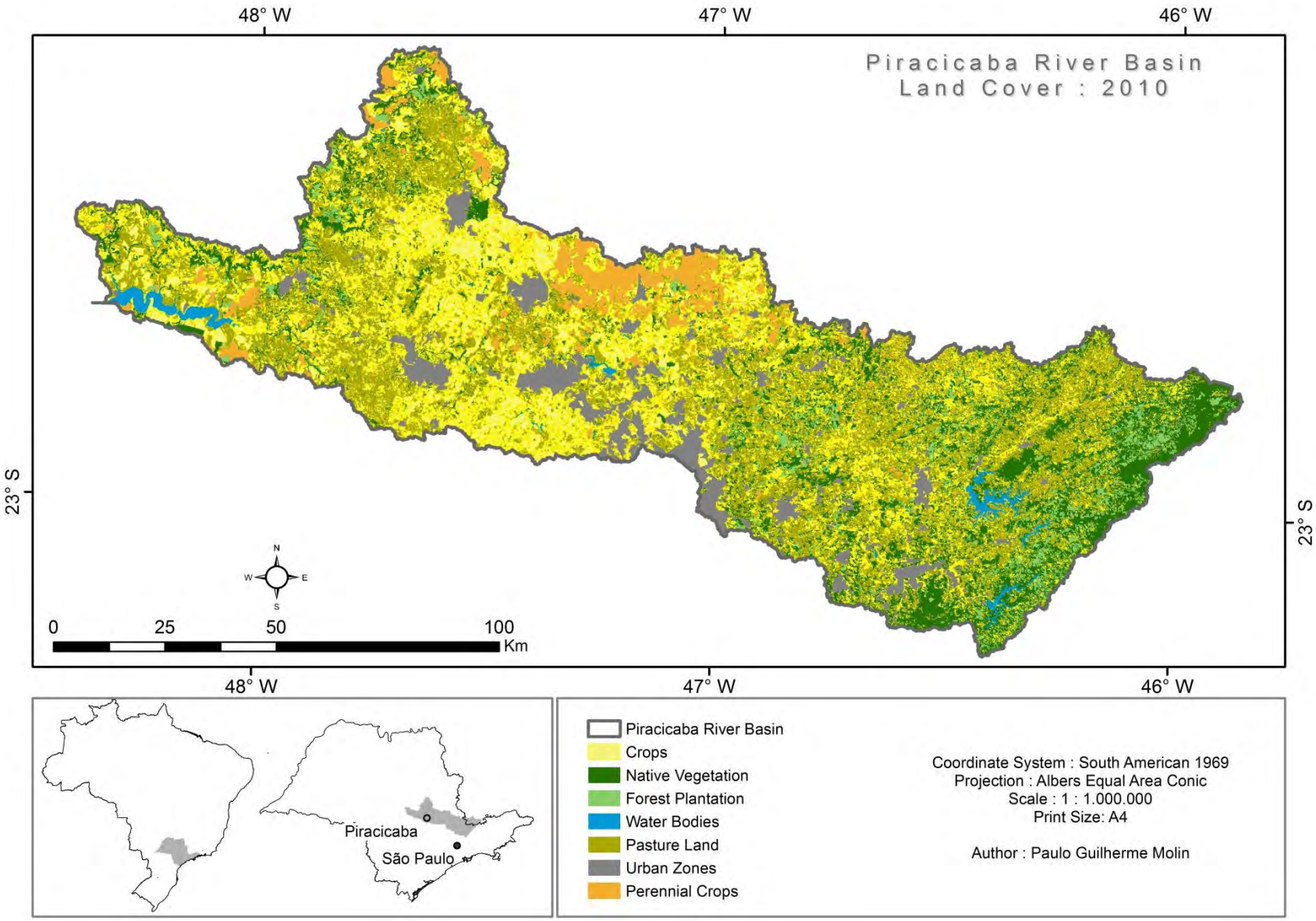
Appendix B – Land cover thematic map of the Piracicaba River basin for the year 1990



Appendix C – Land cover thematic map of the Piracicaba River basin for the year 2000



Appendix D – Land cover thematic map of the Piracicaba River basin for the year 2010

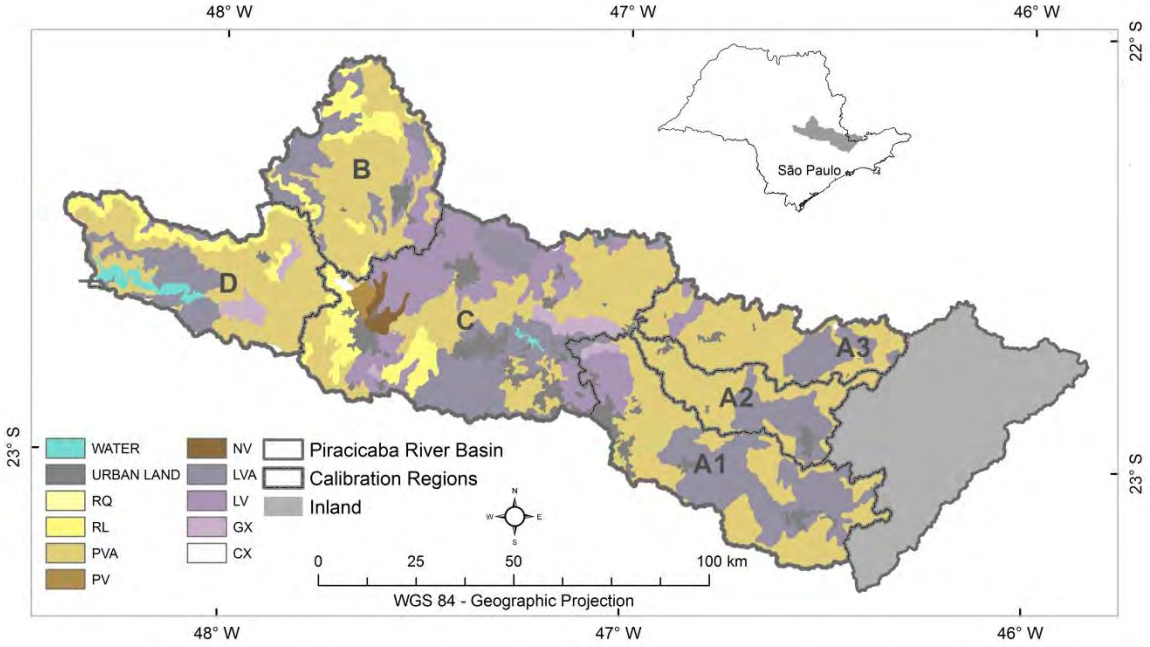


Appendix E – Transition matrices, in percentage, for both 1990-2000 and 2000-2010 periods of the Piracicaba River basin

			2000							
			1	**2**	**3**	**4**	**5**	**6**	**7**	**Total**
	Crops	**1**	53.7	7.5	1.5	0.5	32.4	1.6	2.8	100
	Native Vegetation	**2**	17.7	56.3	6.3	0.2	18.3	0.5	0.7	100
	Forest Plantations	**3**	16.8	34.8	30.5	0.0	15.8	0.3	1.8	100
1990	**Water Bodies**	**4**	4.9	4.0	0.0	89.7	1.1	0.2	0.1	100
	Pasture	**5**	32.4	8.9	2.1	0.1	53.3	1.3	1.9	100
	Urban Zones	**6**	1.8	0.1	0.0	0.0	0.2	97.8	0.0	100
	Perenial Crops	**7**	26.2	6.2	1.0	0.0	22.4	1.0	43.1	100

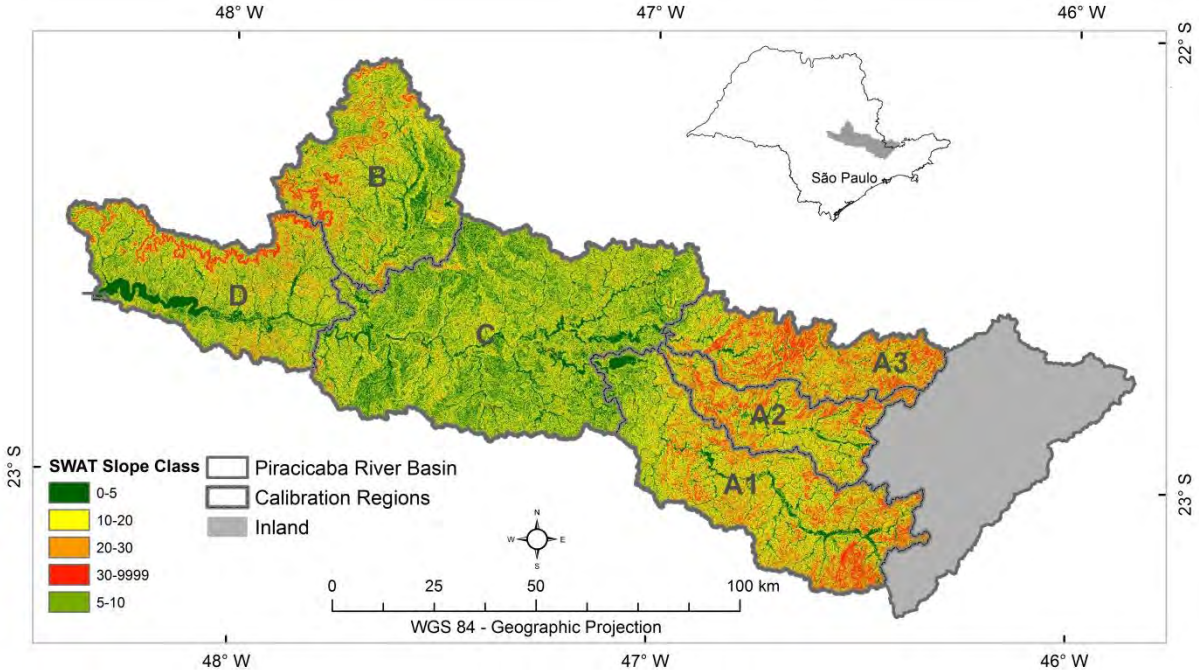
			2010							
			1	**2**	**3**	**4**	**5**	**6**	**7**	**Total**
	Crops	**1**	47.5	7.0	1.4	0.5	38.8	1.4	3.4	100
	Native Forests	**2**	9.9	73.7	1.8	0.5	12.7	0.2	1.1	100
	Forest Plantations	**3**	3.6	14.0	78.4	0.0	3.1	0.0	0.9	100
2000	**Water Bodies**	**4**	3.7	4.7	0.0	90.0	1.4	0.1	0.1	100
	Pasture	**5**	28.1	11.9	2.0	0.2	54.0	0.8	2.9	100
	Urban Zones	**6**	1.3	0.2	0.0	0.0	0.4	98.0	0.1	100
	Perenial Crops	**7**	14.5	2.9	0.4	0.1	9.3	0.3	72.5	100

Appendix F – Soil map for the Piracicaba River basin used for calibration of SWAT model

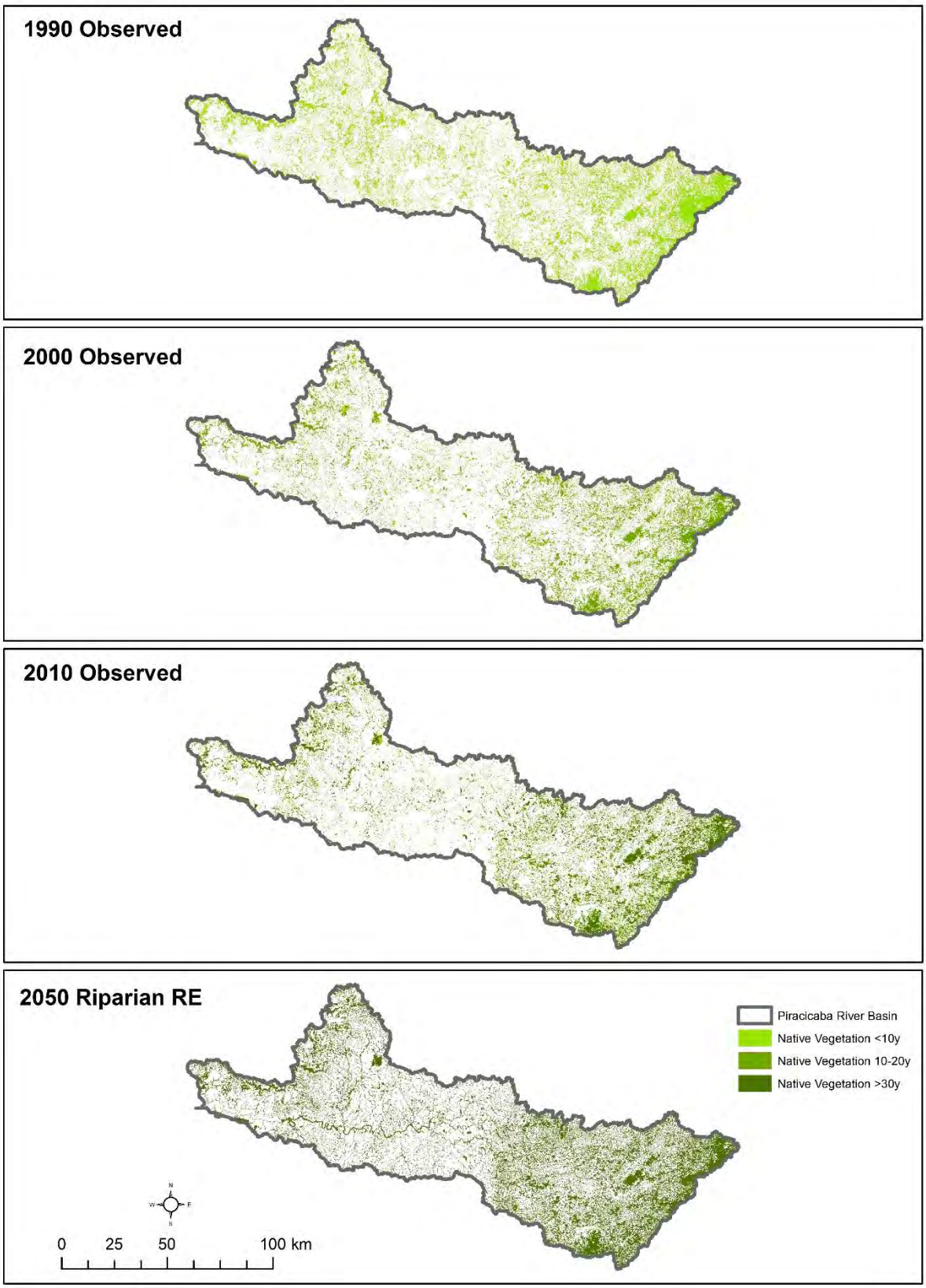


Soil Types	Percentage of Watershed (%)
Argisslo Vermelho Amarelo (PVA)	48.12
Latossolo Vermelho Amarelo (LVA)	24.37
Latossolo Vermelho (LV)	11.77
Neossolo Litólico (RL)	6.5
Urban Land	5.15
Gleissolo Haplicos (GX)	2.01
Water	0.87
Nitossolo Vermelho (NV)	0.64
Argisslo Vermelho (PV)	0.39
Cambissolo Haplicos (CX)	0.12
Neossolo Quartzenico (NQ)	0.05

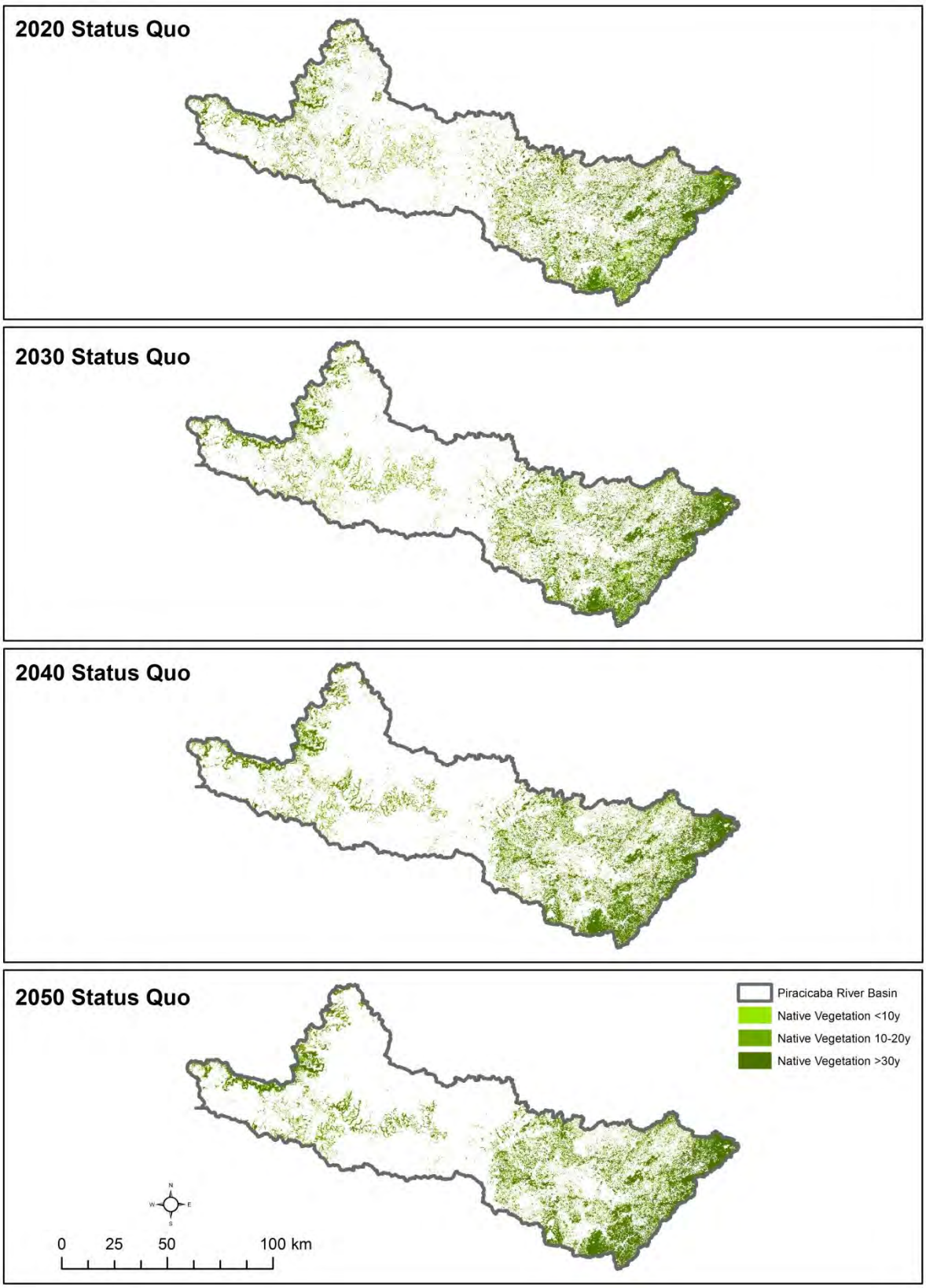
Appendix G – Slope map for the Piracicaba River basin used for calibration of SWAT model



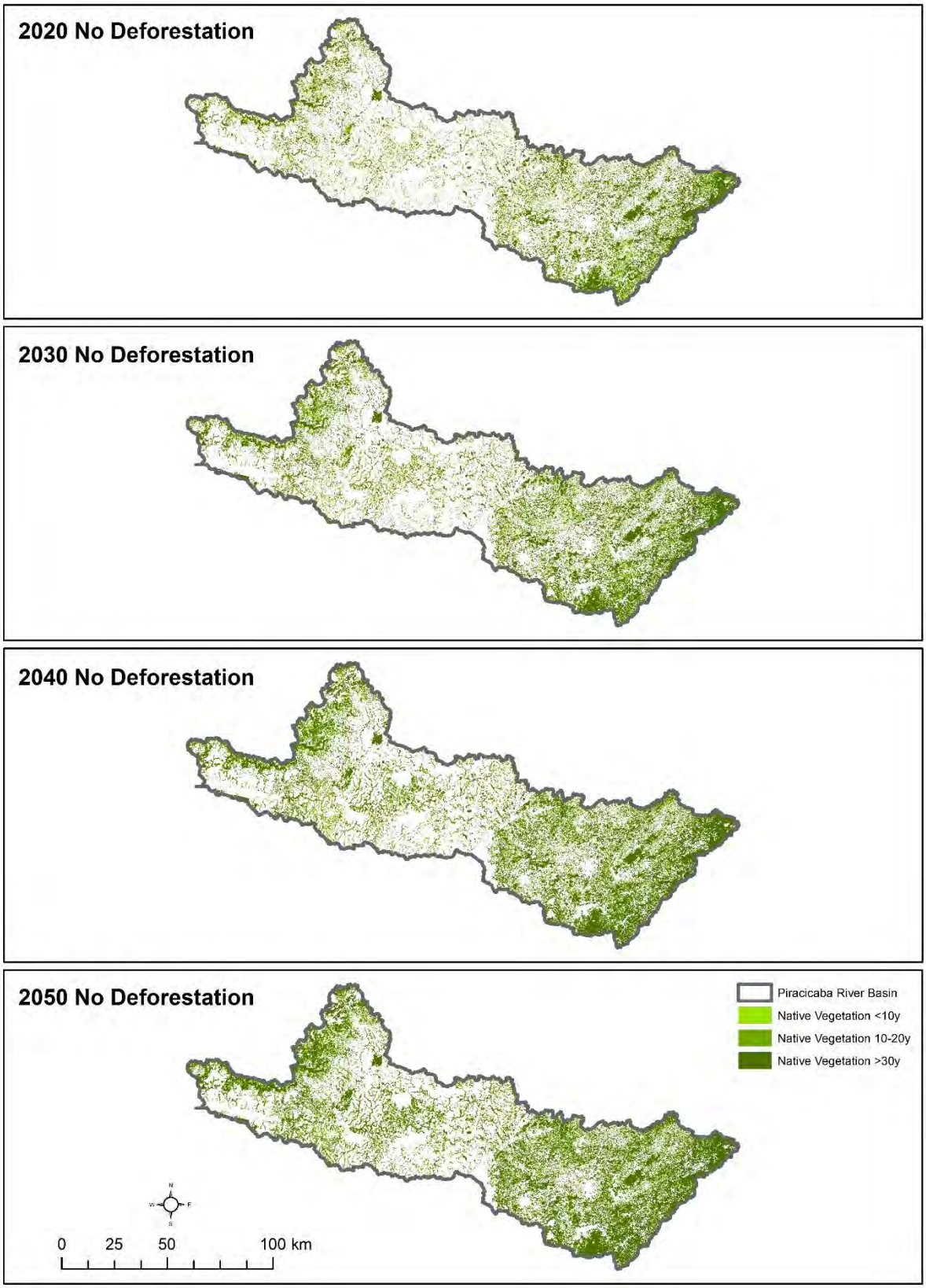
Appendix H – Native vegetation cover maps for the observed dates of 1990, 2000 and 2010 as well as the simulated 2050 Riparian Restoration Enforcement scenario



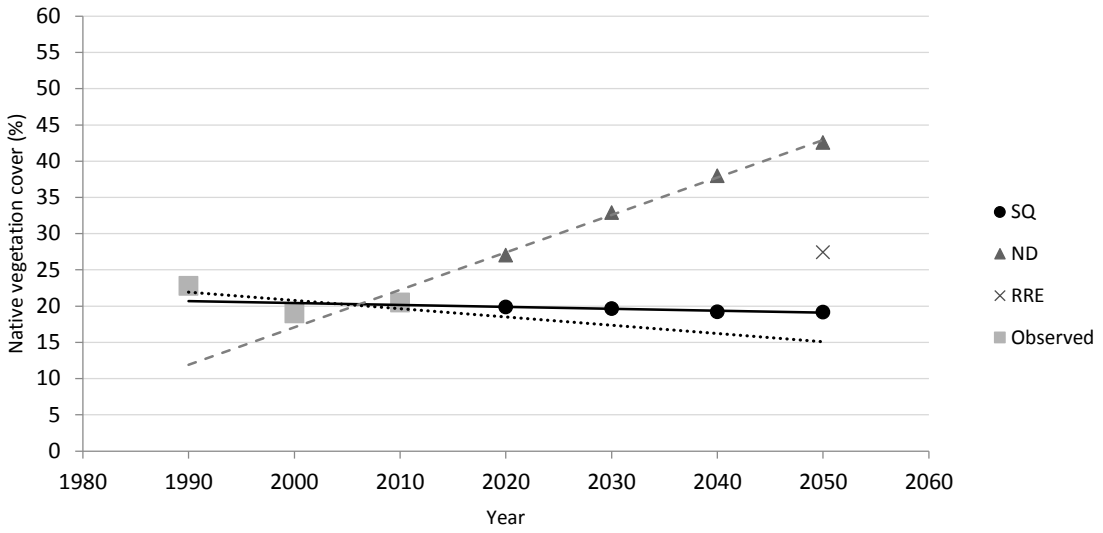
Appendix I – Native vegetation cover maps for the simulated scenarios of Status Quo for the dates of 2020, 2030, 2040 and 2050



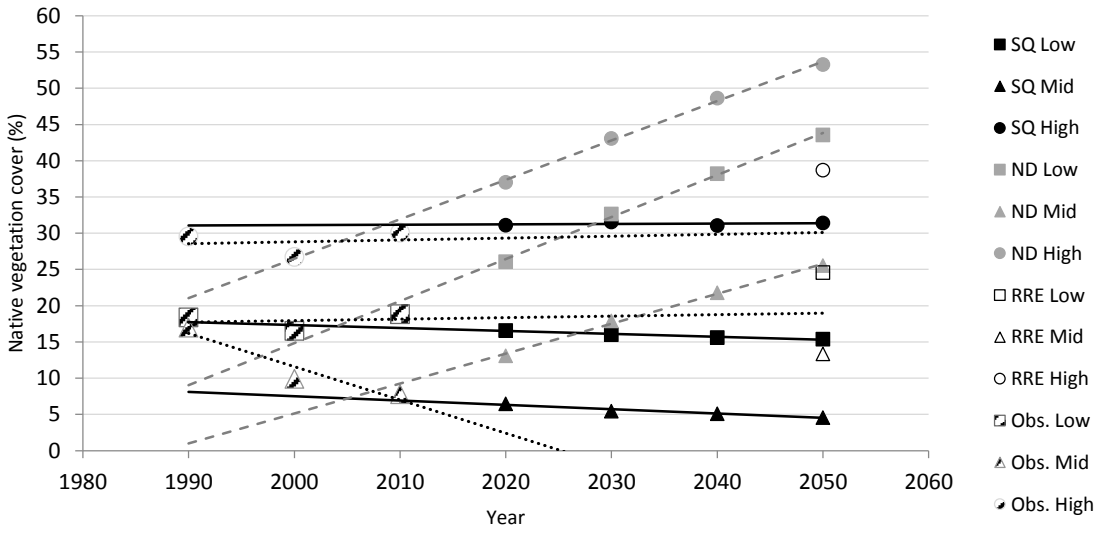
Appendix J – Native vegetation cover maps for the simulated scenarios of No Deforestation for the dates of 2020, 2030, 2040 and 2050



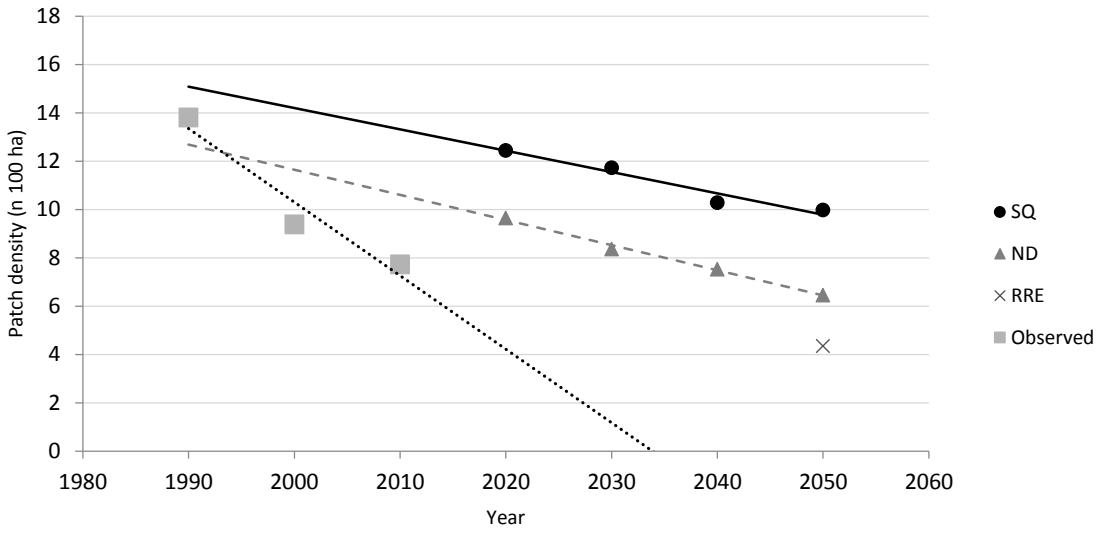
Appendix K – Mean native vegetation cover of the 70 samples representing the Piracicaba River basin, here presented in percent cover (%)



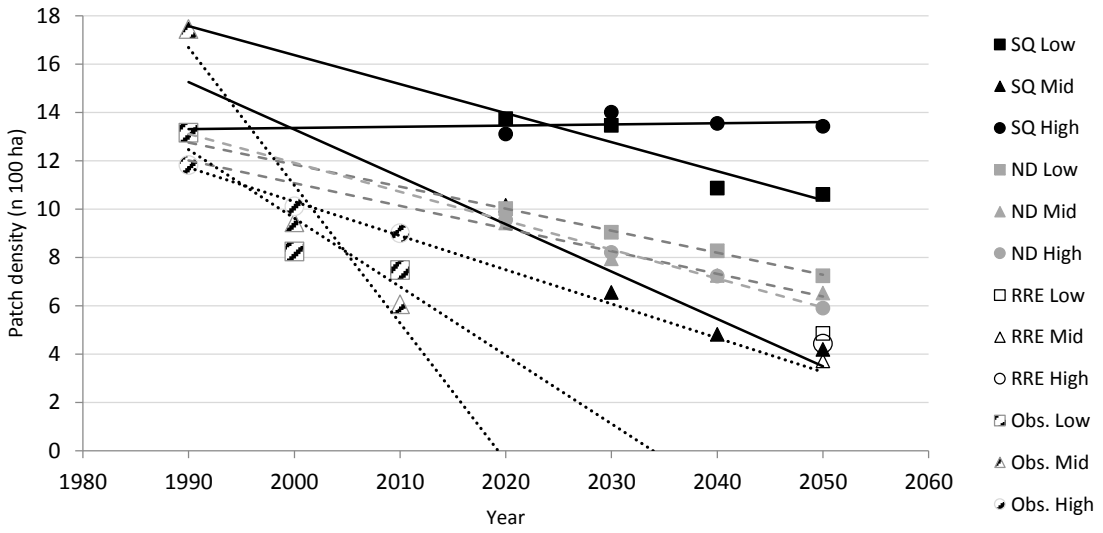
Appendix L – Mean native vegetation cover the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in natural logarithm (Ln %)



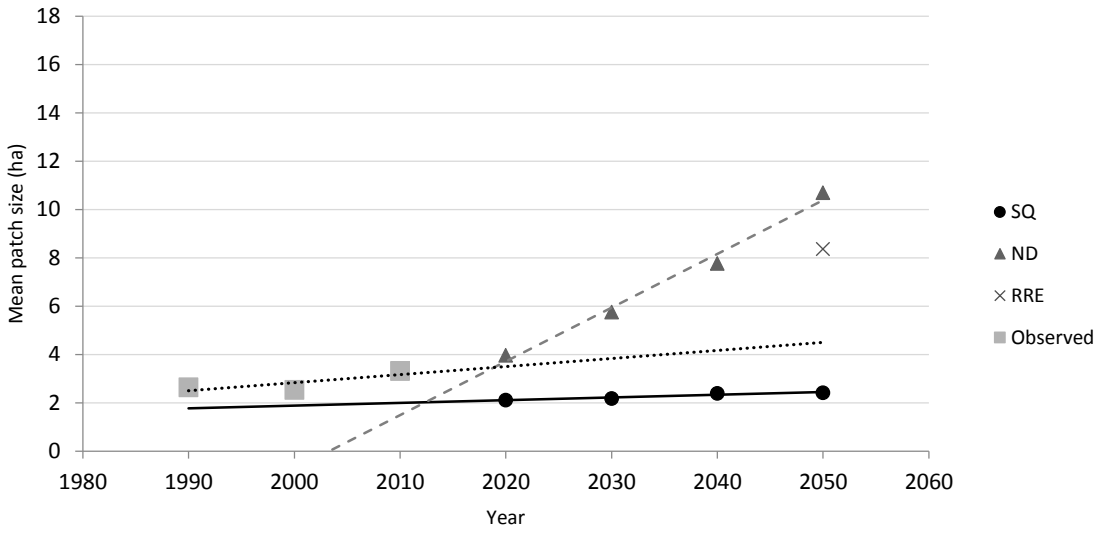
Appendix M – Mean patch density of the 70 samples representing the Piracicaba River basin, here presented in number of fragments per 100 ha (n 100ha^{-1})



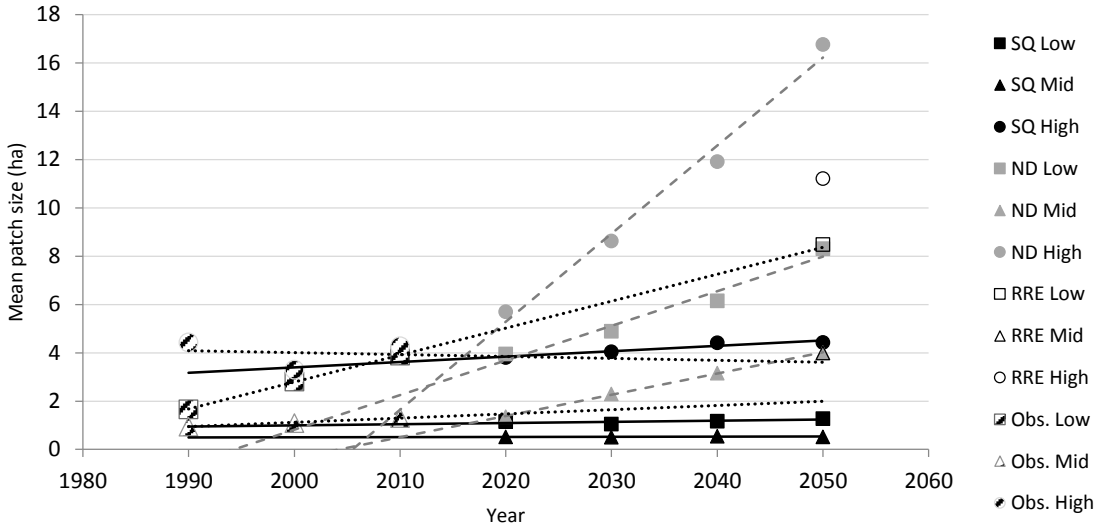
Appendix N – Mean patch density of the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in number of fragments per 100 ha (n 100ha^{-1})



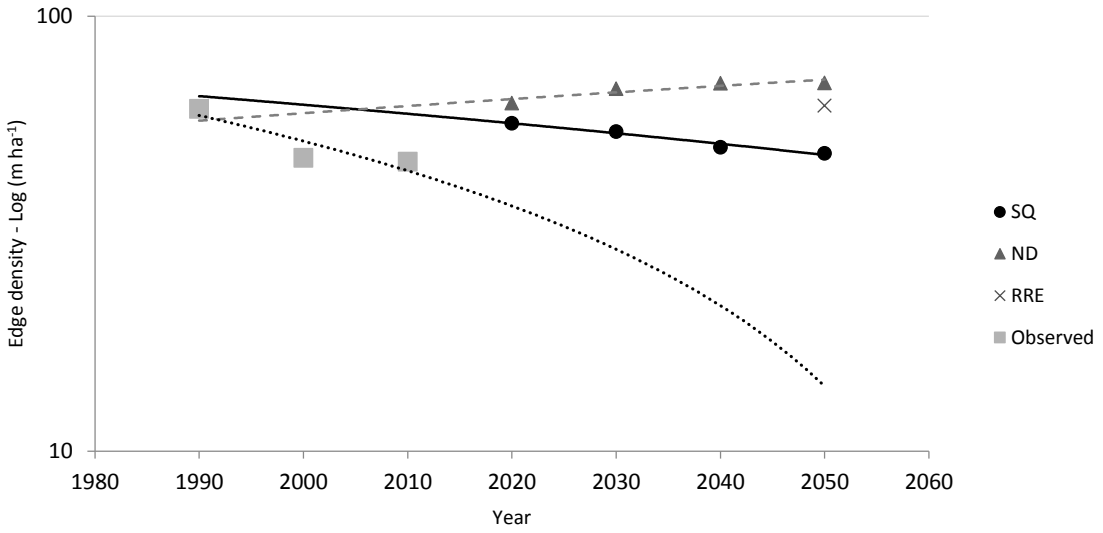
Appendix O – Mean patch size of the 70 samples representing the Piracicaba River basin, here presented in hectares (ha)



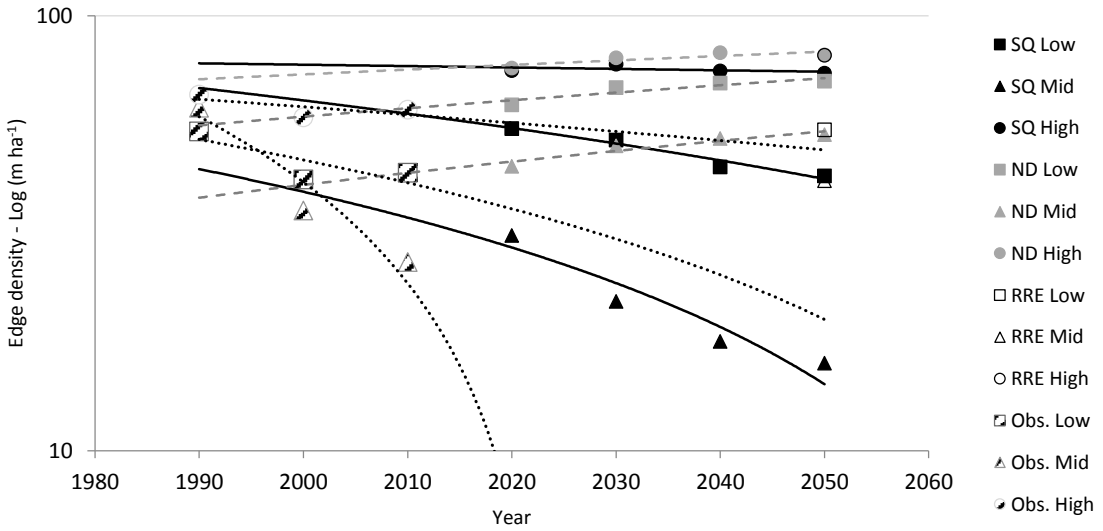
Appendix P – Mean patch size of the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in in hectares (ha)



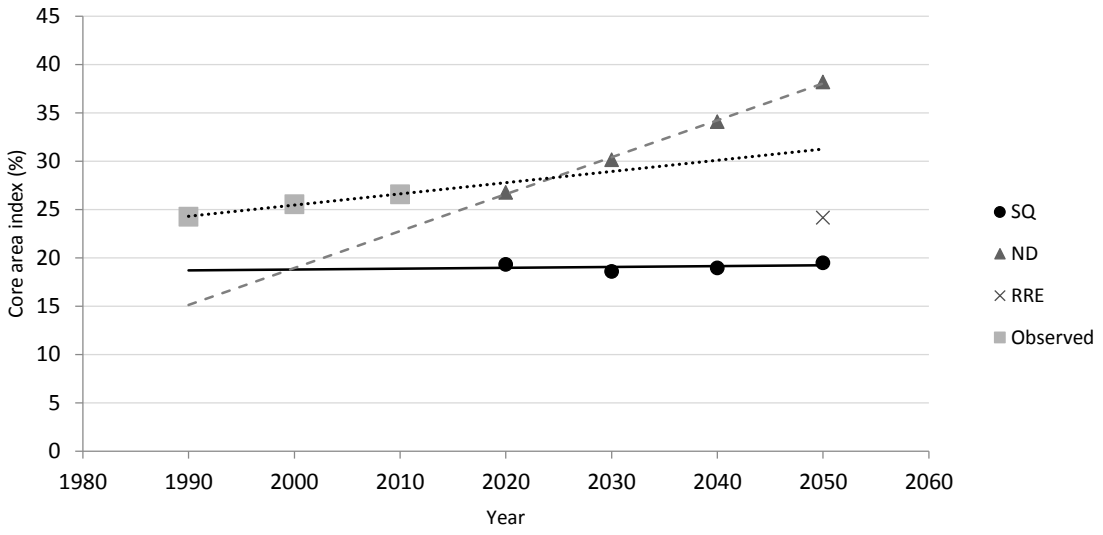
Appendix Q – Mean edge density of the 70 samples representing the Piracicaba River basin, here presented in decadic logarithm – Log (m ha^{-1})



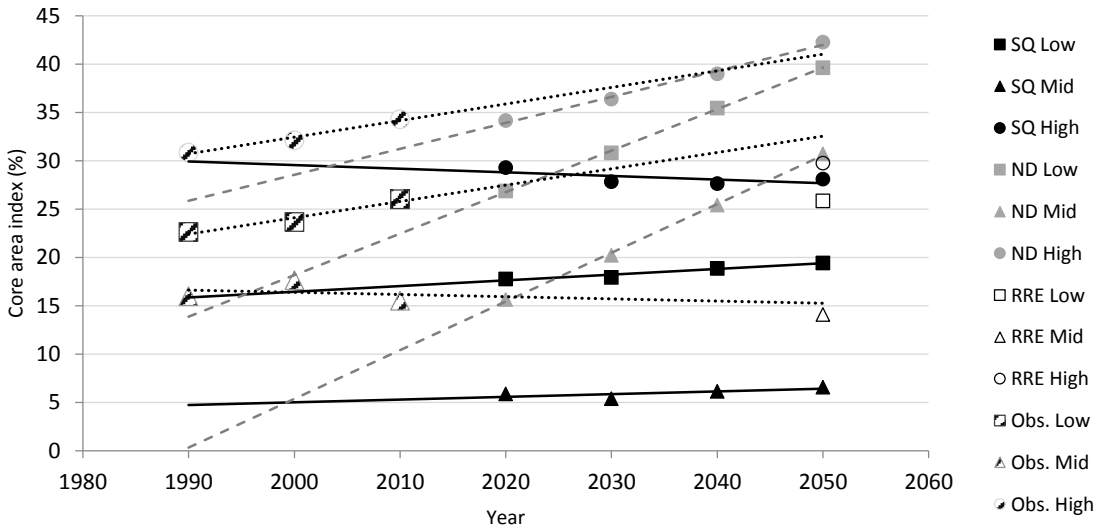
Appendix R – Mean edge density of the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in decadic logarithm – Log (m ha^{-1})



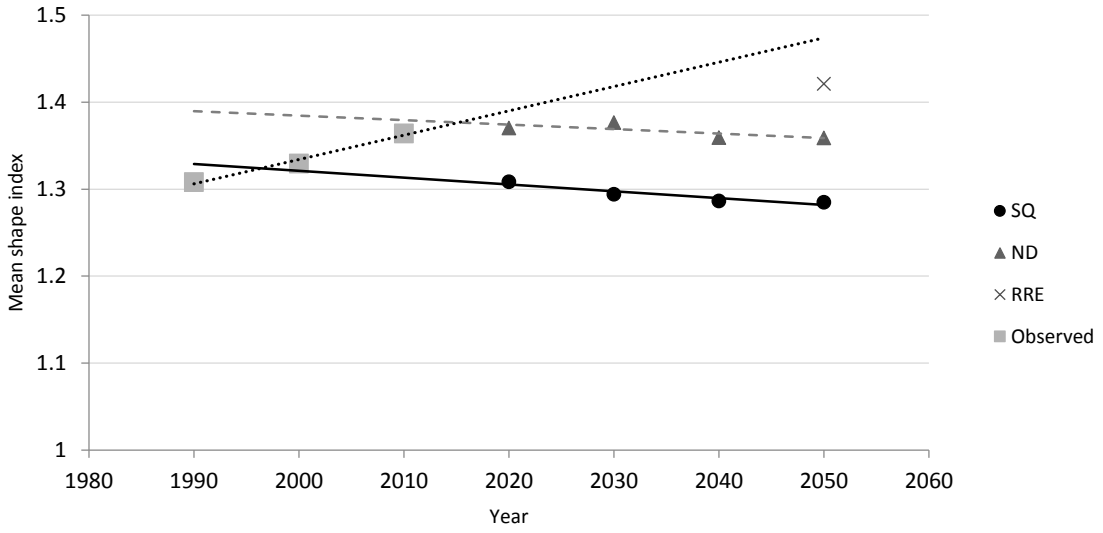
Appendix S – Mean core area index of the 70 samples representing the Piracicaba River basin, here presented in percentage (%)



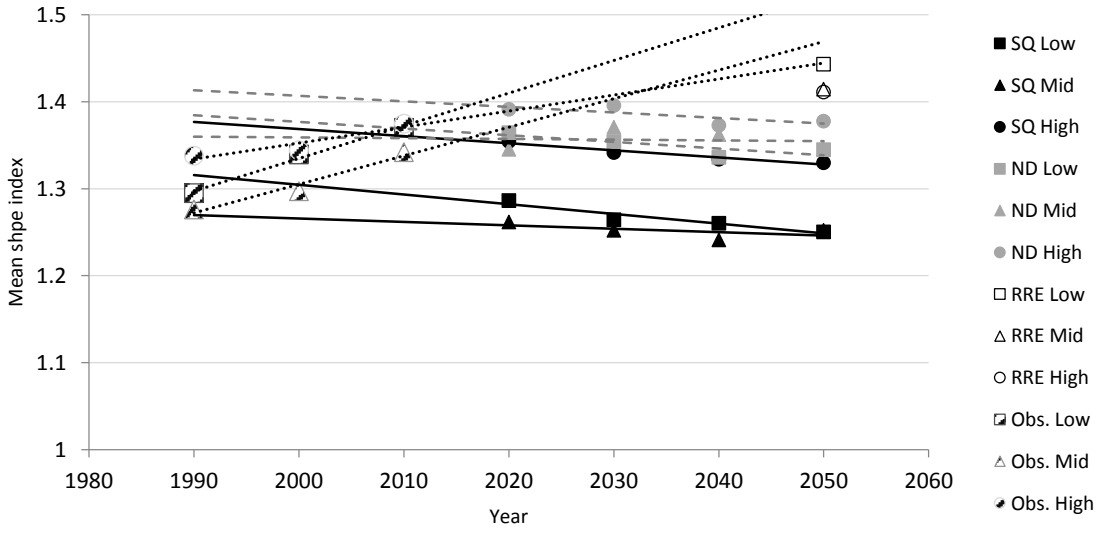
Appendix T – Mean core área index of the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in percentage (%)



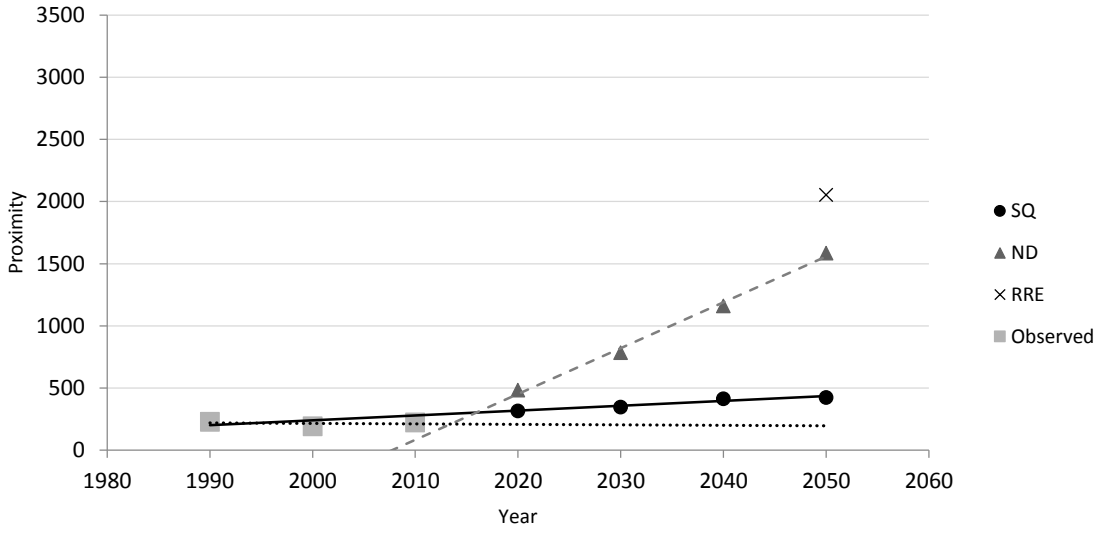
Appendix U – Mean shape index of the 70 samples representing the Piracicaba River basin



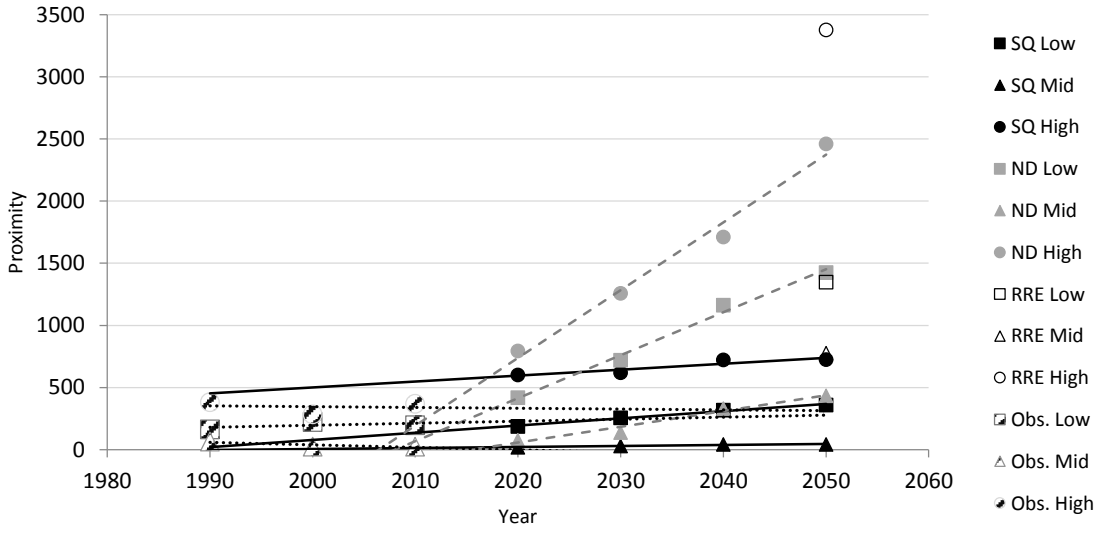
Appendix V – Mean shape index of the lower (low), middle (mid) and higher (high) Piracicaba River basin



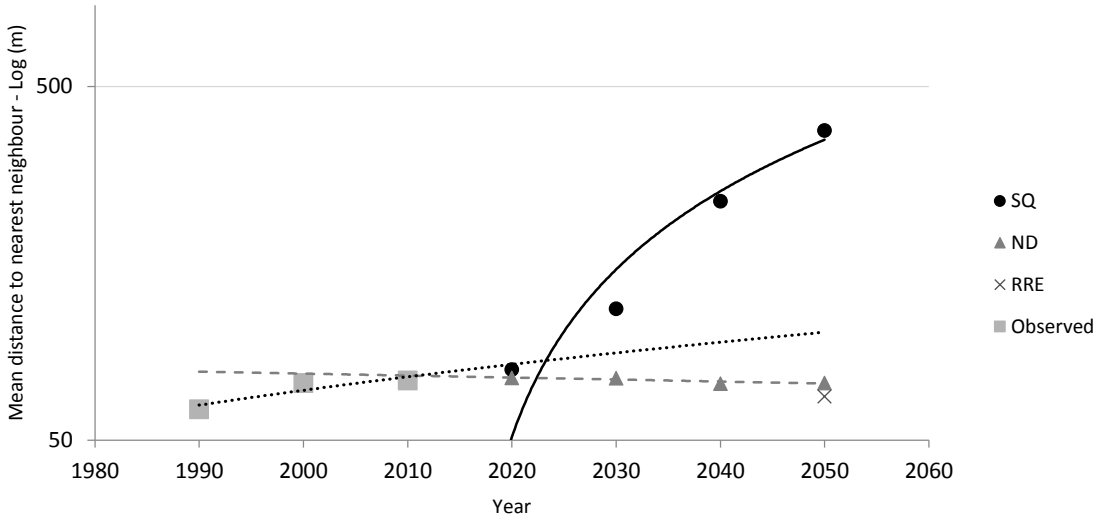
Appendix W – Mean proximity index of the 70 samples representing the Piracicaba River basin



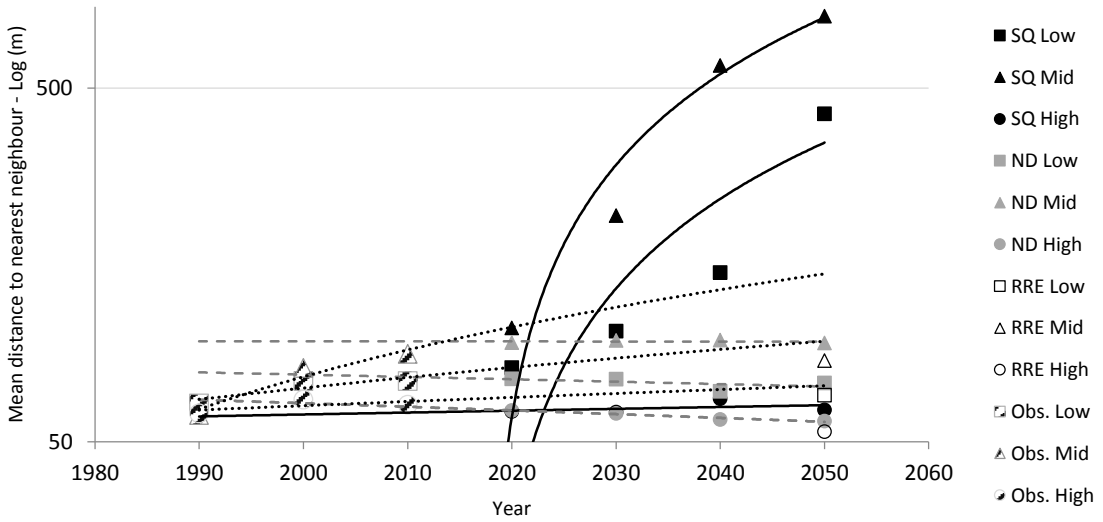
Appendix X – Mean proximity index of the lower (low), middle (mid) and higher (high) Piracicaba River basin



Appendix Y – Mean distance to nearest neighbor index of the 70 samples representing the Piracicaba River basin, here presented in decadic logarithm Log (m)



Appendix Z – Mean distance to nearest neighbor index of the lower (low), middle (mid) and higher (high) Piracicaba River basin, here presented in decadic logarithm Log (m)



ANNEX

Annex A – Error matrix for the 1990, 2000 and 2010 Piracicaba River basin land cover maps

			Reference Data							
			1	**2**	**3**	**4**	**5**	**6**	**7**	**Total**
1990 Classified Data	Crops	**1**	**27**	3		1	4			35
	Native Vegetation	**2**	2	**28**	3		2			35
	Forest Plantation	**3**	4	8	**18**		5			35
	Water Bodies	**4**				**35**				35
	Pasture Land	**5**	2	1		1	**29**	1	1	35
	Urban Zones	**6**						**35**		35
	Perennial Crops	**7**	6	7					**22**	35
		Total	41	47	21	37	40	36	23	**245**
	Omission Error		0.66	0.60	0.86	0.95	0.73	0.97	0.96	0.79
	Comission Error		0.77	0.80	0.51	1.00	0.83	1.00	0.63	0.14
	Kappa									0.76
			1	**2**	**3**	**4**	**5**	**6**	**7**	**Total**
2000 Classified Data	Crops	**1**	**21**	2		1	10	1		35
	Native Vegetation	**2**		**33**	2					35
	Forest Plantation	**3**	2		**30**		3			35
	Water Bodies	**4**			1	**34**				35
	Pasture Land	**5**	2				**31**	1	1	35
	Urban Zones	**6**						**35**		35
	Perennial Crops	**7**	2	2			1		**30**	35
		Total	27	37	33	35	45	37	31	245
	Omission Error		0.78	0.89	0.91	0.97	0.69	0.95	0.97	0.87
	Comission Error		0.60	0.94	0.86	0.97	0.89	1.00	0.86	0.14
	Kappa									0.85
			1	**2**	**3**	**4**	**5**	**6**	**7**	**Total**
2010 Classified Data	Crops	**1**	**25**		1		7	1	1	35
	Native Vegetation	**2**	1	**27**	6		1			35
	Forest Plantation	**3**		1	**34**					35
	Water Bodies	**4**	1			**34**				35
	Pasture Land	**5**	4		1	1	**27**	1	1	35
	Urban Zones	**6**						**35**		35
	Perennial Crops	**7**	3				1		**31**	35
		Total	34	28	42	35	36	37	33	**245**
	Omission Error		0.74	0.96	0.81	0.97	0.75	0.95	0.94	0.87
	Comission Error		0.71	0.77	0.97	0.97	0.77	1.00	0.89	0.14
	Kappa									0.85

Annex B – Total area and percentage for individual sub-basin land cover

1990	ATIBAIA		CAMANDUCAIA		CORUMBATAÍ		JAGUARÍ		PIRACICABA		TOTAL	
	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%
Crops	44857	15.97	15606	15.01	59512	34.88	61893	18.77	118968	32.09	300836	23.95
Native Vegetation	84213	29.98	26982	25.96	36859	21.60	92792	28.14	66183	17.85	307028	24.44
Forest Plantations	9108	3.24	1802	1.73	6681	3.92	7801	2.37	3772	1.02	29164	2.32
Water Bodies	3173	1.13	26	0.03	151	0.09	4445	1.35	10323	2.78	18119	1.44
Pasture Land	112702	40.13	54930	52.84	56407	33.06	140947	42.74	138268	37.29	503254	40.07
Urban Zones	26664	9.49	3176	3.06	5549	3.25	8033	2.44	28482	7.68	71904	5.72
Perenial Crops	154	0.05	1290	1.24	5109	2.99	13628	4.13	4071	1.10	24253	1.93
TOTAL	280871	100.00	103950	100.00	170618	100.00	329778	100.00	370782	100.00	1255999	100.00

2000	ATIBAIA		CAMANDUCAIA		CORUMBATAÍ		JAGUARÍ		PIRACICABA		TOTAL	
	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%
Crops	54423	19.38	22340	21.49	69904	40.97	84302	25.56	161609	43.59	392579	31.26
Native Vegetation	76882	27.37	24182	23.26	29545	17.32	77173	23.40	44520	12.01	252301	20.09
Forest Plantations	16866	6.01	3820	3.67	3615	2.12	16840	5.11	2456	0.66	43597	3.47
Water Bodies	3592	1.28	118	0.11	233	0.14	4820	1.46	10035	2.71	18798	1.50
Pasture Land	99437	35.40	47797	45.98	55172	32.34	118582	35.96	111208	29.99	432195	34.41
Urban Zones	29562	10.53	3553	3.42	5964	3.50	10217	3.10	34518	9.31	83814	6.67
Perenial Crops	109	0.04	2003	1.93	5834	3.42	17606	5.34	5721	1.54	31272	2.49
TOTAL	280871	100.00	103950	100.00	170618	100.00	329778	100.00	370782	100.00	1255999	100.00

2010	ATIBAIA		CAMANDUCAIA		CORUMBATAÍ		JAGUARÍ		PIRACICABA		TOTAL	
	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%	Area (ha)	%
Crops	47946	17.08	27101	26.07	59533	34.89	74832	22.69	131480	35.46	340892	27.14
Native Vegetation	92900	33.09	24533	23.60	30955	18.14	81525	24.72	43305	11.68	273217	21.75
Forest Plantations	19804	7.05	3709	3.57	6297	3.69	17863	5.42	5406	1.46	53080	4.23
Water Bodies	4034	1.44	278	0.27	434	0.25	5368	1.63	10740	2.90	20854	1.66
Pasture Land	83777	29.84	41114	39.55	59322	34.77	106463	32.28	131705	35.52	422381	33.63
Urban Zones	32101	11.43	3886	3.74	6100	3.58	12117	3.67	37709	10.17	91914	7.32
Perenial Crops	211	0.08	3199	3.08	7629	4.47	31369	9.51	9736	2.63	52144	4.15
TOTAL	280773	100.00	103950	100.00	170618	100.00	329778	100.00	370782	100.00	1255901	100.00

Annex C - Calibrated parameters of expander and patcher algorithms for the individual segments of the Piracicaba River basin, where CR is crops, NV is native vegetation, PL is pasture land

		Transitions			
		CR->NV	NV->CR	NV->PL	PL->NV
		Transition Parameters [a]01BPIR			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.58	0.44	0.44	0.58
Patch	Patch Size Variance (ha)	1.06	0.88	0.88	1.06
Patch	Patch Isometry	0.5	0.5	0.5	0.5
Expand	Mean Patch Size (ha)	0.58	0.44	0.44	0.58
Expand	Patch Size Variance (ha)	1.06	0.88	0.88	1.06
Expand	Patch Isometry	1.5	1.5	1.5	1.5
		Transition Parameters [b]02JAAP			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.48	0.54	0.54	0.48
Patch	Patch Size Variance (ha)	0.96	1.08	1.08	0.96
Patch	Patch Isometry	0.5	0.5	0.5	0.5
Expand	Mean Patch Size (ha)	0.48	0.54	0.54	0.48
Expand	Patch Size Variance (ha)	0.96	1.08	1.08	0.96
Expand	Patch Isometry	1.5	1.5	1.5	1.5
		Transition Parameters [c]03CORU			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.68	0.58	0.58	0.68
Patch	Patch Size Variance (ha)	1.359	1.16	1.16	1.359
Patch	Patch Isometry	1	1	1	1
Expand	Mean Patch Size (ha)	0.68	0.58	0.58	0.68
Expand	Patch Size Variance (ha)	1.359	1.16	1.16	1.359
Expand	Patch Isometry	1.5	1.5	1.5	1.5
		Transition Parameters [d]04BAAT			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.47	0.35	0.35	0.47
Patch	Patch Size Variance (ha)	0.94	0.7	0.7	0.94
Patch	Patch Isometry	1	1	1	1
Expand	Mean Patch Size (ha)	0.47	0.35	0.35	0.47
Expand	Patch Size Variance (ha)	0.94	0.7	0.7	0.94
Expand	Patch Isometry	1.5	1.5	1.5	1.5
		Transition Parameters [e]05BCJA			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.35	0.36	0.36	0.35
Patch	Patch Size Variance (ha)	0.7	0.72	0.72	0.7
Patch	Patch Isometry	1	1	1	1
Expand	Mean Patch Size (ha)	0.35	0.36	0.36	0.35
Expand	Patch Size Variance (ha)	0.7	0.72	0.72	0.7
Expand	Patch Isometry	0.5	0.5	0.5	0.5
		Transition Parameters [f]06AAJA			
Expand	% transition by expansion	0.9	0.9	0.9	0.9
Patch	Mean Patch Size (ha)	0.45	0.36	0.36	0.45
Patch	Patch Size Variance (ha)	0.9	0.72	0.72	0.9
Patch	Patch Isometry	1	1	1	1
Expand	Mean Patch Size (ha)	0.45	0.36	0.36	0.45
Expand	Patch Size Variance (ha)	0.9	0.72	0.72	0.9
Expand	Patch Isometry	0.5	0.5	0.5	0.5

[a] 01BPIR represents a lower region of the Piracicaba sub-basin,
[b] 02JAAP is a composition of a the higher section of the Piracicaba sub-basin and lower Jaguarí sub-basin,
[c] 03CORU is the Corumbataí sub-basin,
[d] 04BAAT is the lower Atibaia sub-basin,
[e] 05BCJA is the middle Jaguarí and Camaducaia sub-basins,
[f] 06AAJA is a composition of both higher Atibaia and higher Jaguarí sub-basins

Annex D – Fuzzy similarity index calculated through constant and exponential decay function, with a multiple windows for each individual segment simulation

		Simulation Models (Subdivisions)											
		[a]01BPIR		[b]02JAAP		[c]03CORU		[d]04BAAT		[e]05BCJA		[f]06AAJA	
	Decay Function	**Const.**	**Expon.**	**Const.**	**Expon.**	**Const.**	**Expon.**	**Const.**	**Expon.**	**Const.**	**Expon.**	**Const.**	**Expon.**
Windows	**1x1**	0.25	0.25	0.29	0.29	0.28	0.28	0.26	0.26	0.22	0.22	0.30	0.30
	3x3	0.40	0.35	0.41	0.37	0.41	0.37	0.42	0.37	0.38	0.33	0.45	0.40
	5x5	0.53	0.41	0.52	0.42	0.53	0.42	0.58	0.44	0.55	0.40	0.58	0.46
	7x7	0.63	0.44	0.60	0.45	0.63	0.45	0.69	0.48	0.68	0.44	0.67	0.49
	9x9	0.71	0.46	0.65	0.46	0.70	0.47	0.76	0.49	0.76	0.46	0.73	0.50
	11x11	0.76	0.47	0.70	0.46	0.76	0.47	0.81	0.50	0.82	0.47	0.77	0.51

[a] 01BPIR represents a lower region of the Piracicaba sub-basin,
[b] 02JAAP is a composition of a the higher section of the Piracicaba sub-basin and lower Jaguarí sub-basin,
[c] 03CORU is the Corumbataí sub-basin,
[d] 04BAAT is the lower Atibaia sub-basin,
[e] 05BCJA is the middle Jaguarí and Camaducaia sub-basins,
[f] 06AAJA is a composition of both higher Atibaia and higher Jaguarí sub-basins

Annex E – Kruskal-Wallis statistical test with Mann-Whitney pairwise comparisons to identify significant differences between landscape metrics for the observed 2010 native vegetation and the 2050 simulated scenarios

		2010	2050		
		Observed	**Status Quo**	**No Deforestation**	**Riparian Restoration Enforcement**
2010	**Observed**	-	C**, E**, F**	A**, B*, C**, D**, E**, G**	A**, B**, C**, D**, F**, G**, H**
2050	**Status Quo**	-	-	A**, B**, C**, D**, E**, F**, G**, H*	A**, B**, C**, E*, F**, G**, H**
	No Deforestation	-	-	-	A**, B**, D*, E**, F**, G*
	Riparian Restoration Enforcement	-	-	-	-

(A) native vegetation cover
(B) patch density
(C) mean patch size
(D) edge density
(E) core area index
(F) mean shape index
(G) proximity
(H) mean distance to nearest neighbor

Note: When values were <0.05, the difference was found to be significant (*)
When values were <0.01, the differences were found to be very significant (**)