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“Luiz de Queiroz” College of Agriculture

Forest cover and water quality in tropical agricultural watersheds

Kaline de Mello

Thesis presented to obtain the degree of Doctor in Science.
Area: Agricultural Systems Engineering

Piracicaba
2017

Kaline de Mello
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Versão revisada de acordo com a resolução CoPGr 6018 de 2011

Advisor:

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DEDICATION

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RESUMO

Cobertura florestal e qualidade da água de microbacias agrícolas tropicais

As florestas tropicais estão sob constante ameaça devido ao processo de desmatamento e fragmentação florestal impulsionado pelo crescimento das atividades econômicas, em especial, a agricultura. A substituição de áreas florestadas por outros usos do solo pode causar impactos severos na qualidade da água de rios, alterando suas características físicas, químicas e biológicas. A Mata Atlântica, em especial, teve sua cobertura original reduzida a cerca de 11%, sendo que a expansão de terras cultiváveis e urbanização ainda ameaçam esse importante ecossistema e os serviços ecossistêmicos prestados por ele. Nesse sentido, este estudo propôs investigar a relação da cobertura florestal com a qualidade da água de microbacias agrícolas tropicais. Para tanto, foram selecionadas seis microbacias experimentais com diferentes porcentagens de cobertura florestal na bacia do rio Sarapuí, Estado de São Paulo, Brasil, onde foram feitas coletas de amostras de água por um ano hidrológico para a obtenção de parâmetros que representassem alterações na água induzidas por atividades antrópicas. Inicialmente as microbacias foram classificadas em “florestadas” e “degradadas”, e modelos estatísticos multivariados foram aplicados para identificar diferenças entre os grupos. Em um segundo momento comparou-se a relação do uso e cobertura do solo na microbacia e na Área de Preservação Permanente (APP) com a qualidade da água utilizando-se modelos mistos e análise de redundância para identificar os principais fatores que influenciam a variabilidade da qualidade da água. Por último foi gerado um modelo hidrológico para simular o impacto da restauração da floresta ripária na qualidade da água da bacia do rio Sarapuí onde cada microbacia experimental desse estudo foi representada por uma sub-bacia do modelo. Os resultados mostram que as microbacias degradadas apresentam valores maiores de sólidos, turbidez, nutrientes e coliformes. Além disso, apresentam maior variabilidade temporal dos dados em relação às microbacias florestadas associada às alterações da vazão do rio. Em geral, a cobertura florestal foi relacionada à boa qualidade da água, enquanto que agricultura e ocupação urbana foram os usos do solo responsáveis pela degradação da qualidade da água. O uso pastagem apresentou impactos mistos, porém no geral não foi correlacionado à qualidade da água ruim. Os parâmetros de qualidade da água responderam de forma diferente quanto à influência dos padrões de uso e cobertura do solo na microbacia e na APP, porém, considerando-se todos parâmetros em conjunto, a qualidade da água é melhor explicada pela composição da paisagem da microbacia. Ainda assim, a simulação do modelo indicou que a restauração das APPs reduz a carga de sedimentos e nutrientes para o rio. Com isso, conclui-se que a floresta tropical tem papel fundamental na conservação dos recursos hídricos, reduzindo impactos das atividades humanas exercidas nas microbacias e que, apesar da importância das APPs na redução de poluentes para o rio, o manejo de bacias com estratégias de restauração florestal para toda a microbacia é extremamente importante para a manutenção da qualidade da água para abastecimento.

Palavras-chave: 1. Recursos hídricos 2. Conservação florestal 3. Manejo de bacias hidrográficas 4. Uso e cobertura do solo 5. Floresta ripária 6. Restauração florestal

ABSTRACT

Forest cover and water quality in tropical agricultural watersheds

Tropical forests are under continual threat due to deforestation and forest fragmentation processes which are driven by the economic activities growth, mainly agriculture. Replacing forest with other land uses can cause severe impacts on river water quality, altering its physical, chemical and biological characteristics. The Atlantic Forest, in particular, had its original vegetation cover reduced to about 11%, wherein the crop lands expansion and urban sprawl still threatening this important ecosystem and the ecosystem services that it provides. In this sense, the main objective of this study was to investigate the relation between forest cover and water quality of tropical agricultural watersheds. For that, six experimental watersheds with different percentage of forest cover were selected in the Sarapuí River watershed, State of São Paulo, Brazil. Water samples were collected during a hydrologic year to obtain water quality parameters that represent impacts induced by anthropic activities. According to the percentage of forest cover, the watersheds were denominated as “forested”, when they presented more than 55% of forest cover, and “degraded”, with less than 35%. Multivariate statistical models were applied to identify differences between these two groups. In a second moment, the relation of land use/land cover within the watershed and within its respective riparian zone, represented in this study by the Permanent Preservation Areas (PPA), with water quality was compared through mixed models and redundancy analysis to identify the main factors that influenced water quality variability. Lastly, a watershed simulation modeling was applied to verify the impact of riparian forest restoration on water quality of the Sarapuí River watershed, wherein each experimental watershed was represented by a sub-watershed in the model. The results showed that the degraded watersheds presented higher values of solids, turbidity, nutrients and coliforms, besides presenting greater temporal data variability compared to forested watersheds. This variation is associated with the stream flow changes during the year. In general, forest cover was related to good water quality, while agriculture and urban areas were responsible for the water quality degradation. Pasture presented mixed impacts, but it was not generally correlated with poor water quality. The water quality parameters responded differently to the influence of land-use/land-cover patterns in the watershed and riparian zone, but the overall water quality is better explained by the landscape composition within the watershed. Nevertheless, the watershed simulation indicated that PPA restoration reduces the sediment and nutrients loading into the river. Thus, it is possible to conclude that tropical forest plays a fundamental role in the water resources conservation, reducing impacts of human activities in watersheds and the watershed management with forest restoration strategies for the entire watershed is critical for the maintenance of water quality to water supply, despite the importance of the riparian zone.

Keywords: Water resources 2. Forest conservation 3. Watershed management 4. Land use and land cover 5. Riparian forest 6. Forest restoration.

1. INTRODUCTION

Natural ecosystems play a crucial role in producing natural resources and controlling physical, chemical and biological processes worldwide, generating benefits for humans, the so-called “ecosystem services”, such as food, fuel, fiber, climate regulation, pest control and drinking water. Water, in particular, is a natural resource imperative to life, to the natural cycles of the organisms, to the food production and the economic activities. This resource, however, is increasingly scarce due to the crescent demand associated with population growth, increasing in urbanization, agricultural frontier expansion, and the rapid development of economic activities. In addition, the intensification of natural ecosystems degradation is threatening the provision of water resources, especially drinking water.

Declining water quality has become a global issue of concern as industrial and agricultural activities expand, and climate change threatens to cause serious alterations to the hydrological cycles. Many areas worldwide are experiencing problems with lack of drinking water due to the water bodies’ pollution coming from agricultural runoff, domestic sewage, industrial effluents, and atmospheric inputs from fossil fuel burning and bushfires.

Land-use/land-cover composition has a strong influence on the water quality in watersheds, wherein different uses represent various degrees of risk to the water resources. The hydrological cycle and the water behavior reflects the land use, cover and conditions of the catchments, wherein surface runoff is a major source of non-point pollution and is primarily responsible for the relationship between land use/land cover and water quality. The conversion of forested areas to other uses, especially, has been associated with water quality deterioration. However, the non-point source water pollution is difficult to control, once it is not caused by an easily identified and regulated activity.

Agricultural activities are essential for the food provision to the population, feeding animals, and biofuel, however, it is one of the main factors of deforestation, especially in tropical regions. Brazil has reported the greatest annual net loss of forest area in the world due to agricultural expansion. Clearing forest areas to acreage expansion for new cultivation areas and inadequate agricultural practices generate direct and indirect impacts on stream water quality. Nonetheless, these impacts can vary according to the climate, hydrological characteristics, the region, and the physical aspects of the watershed.

In this context, watershed management is essential to ensure the water resources protection, striking a balance between human activities and natural habitats conservation

through, for example, the maintenance of forest areas within the watersheds. However, many questions regarding the relation of forest conservation and water quality have not been resolved. Forest cover in different landscapes can have different impacts on water quality, as well as the forest configuration. For example, the importance of the riparian forests is often mentioned, but empirical studies that test its effectiveness in maintaining water clean are scarce and contradictory.

We are facing the challenge of managing the immediate human needs and maintain long-term water supply capacity, as concern about the availability of this resource with good quality. Therefore, the water conservation depends on the conservation of other natural resources, and it requires plans that incorporate an integrated and holistic view of the ecosystem.

Thus, studies that include the knowledge of how and at which scale forest conservation acts positively on the maintenance of water quality are of utmost importance for the understanding of the ecosystem services provided by the natural habitats in agricultural watersheds. This information can help polices design, forest conservation efforts, and forest restoration projects.

This thesis comprises three chapters, which are complete manuscripts that address this issue. The first chapter gives us an overview of the relation between forest cover and water quality in agricultural tropical watersheds. The second chapter presents a comparison between the effects of land-use/land-cover patterns in the watershed and within the riparian zone on the water quality. The third chapter is a next step, where the effects of riparian restoration on water quality were modeled. Lastly, the final considerations, based on the results from the chapters.

2. INFLUENCES OF TROPICAL FOREST COVER ON WATER QUALITY IN AGRICULTURAL WATERSHEDS IN SOUTHEASTERN BRAZIL

Abstract

Tropical forests are under continued threat due to deforestation and fragmentation, mainly because of clear-cutting for agriculture and cattle ranching, and it can lead to water quality degradation. However, there is a need for understanding the role of forest areas in the current diverse landscape in tropical watersheds. Thus, this study aimed at evaluating the importance of maintaining forest cover for water quality in tropical agricultural watersheds. We selected six watersheds of low-order streams in Southeastern Brazil with different percentages of forest cover, where water samples were collected to obtain the following water quality parameters: dissolved oxygen (DO), turbidity (NTU), total suspended solids (TSS), inorganic suspended solids (ISS), organic suspended solids (OSS), total nitrogen (TN), total phosphorus (TP), total coliforms (TC), and fecal coliforms (FC). A land use/land cover (LULC) map was used to extract the LULC pattern of each watershed, and a distance metric was applied to explore the differences between them. The water quality parameters were analyzed by the yearly mean, data variation and the correlation between them. We compared forested (55% of forest cover or more) and degraded (35% or less) watersheds using MANOVA and PCA. Degraded watersheds showed high values of solids, turbidity, nutrients and coliforms. The most forested watershed showed the best water quality, whereas the watershed mainly covered by agriculture presented the poorest water quality and the greatest temporal variation. The forested watersheds showed significant difference with degraded watersheds, which presented higher value of all the parameters, excepted of DO. NTU, solids, and TP were correlated, showing that turbidity and solids represent the particles in the water, and they carry adsorbed phosphorus. Those parameters also responded to the streamflow variation. The results indicated that water quality degradation is due to the agricultural and residential areas, whereas the tropical forest was important to keep water clean. Thus, tropical forest cover plays an important role in minimizing the impacts of human activities on ecosystem services in agricultural watersheds, but better soil management practices and basic sanitation implementation are still needed to improve water quality.

Keywords: 1. Atlantic Forest 2. Water resources 3. Forest fragmentation 4. Watershed management 5. Land use 6. Tropical watersheds.

2.1. Introduction

Tropical forests are under severe threat from logging and clearing mainly for agriculture (WWF, 2011), threatening the ecosystem services that are provided by these areas, including soil and water protection. Although the rate of global forest area net loss has slowed in the past five years as more forests are better managed, global rates of tropical deforestation increased from 5.6 to 9.1 Mha per year over the last two decades, especially in Africa and South America (WWF, 2011; FAO, 2016). In this context, Brazil reported the greatest annual net loss of forest area between 2010 and 2015 (FAO, 2016). Atlantic Forest, in particular, is a biodiversity hotspot (Mittermeier et al., 2011) that has been reduced to only 11% of its original cover (Ribeiro et al., 2009).

According to Hosonuma et al. (2012), commercial agriculture is the largest driver of deforestation in developing countries, followed by subsistence agriculture. Replacing forest with agriculture decreases water quality due to the erosion from croplands, excess amount of fertilizer and manure applications, higher animal stocking densities, channel erosion, constructional activities and wastewater (Ometo et al., 2000; Gergel et al., 2002; Erol and Randhir, 2013; Gunawardhana et al., 2016; Huang et al., 2016). In this way, keeping forest areas in agricultural watersheds can control the biogeochemical cycles, providing protection against erosion, retention of pollutants, excessive nutrients runoff and increased water temperature (Sweeney et al., 2004; Mingoti and Vettorazzi, 2011; Schilling and Jacobson, 2014; Tanaka et al., 2016).

It is well accepted that watersheds covered with natural forests mostly provide better water quality than watersheds with other land uses. For example, Knee and Encalada (2014) found that watersheds that have streams in protected forests tended to have better water quality than areas which are not under protection. Wang et al. (2013) associated forestland to good water quality comparing to farm and urban lands. Huang et al. (2016) also found that water quality parameters are strongly positively correlated with the proportion of forest areas. However, tropical forests, including Atlantic Forest, are today a mix of forest at many degrees of conservation interspersed with patches of agriculture or other uses (Ribeiro et al., 2009; Singh and Mishra, 2014; Chazdon, 2017). Clément et al. (2017) also found that the landscape diversity appeared to have a negative impact on water quality, and it seems to occur at certain thresholds. According to the authors, even in areas of intensive farming, streams with a forest area that covers at least 47% of the watershed have a better water quality than others with fewer forest areas. Nonetheless, the effects of the forest cover on water quality in tropical watersheds with a mixed agricultural landscape have not really been evaluated (Singh and Mishra, 2014).

With an expected world population to reach 9.7 billion by 2050 (UN-Water, 2016), supplying water becomes a major issue related to drinking water in the world. We currently face the challenge of supplying the immediate population demands and, at the same time, ensure the long-term water supply capacity, assuring quantity and quality of this resource for the future generations.

Accordingly, studies that evaluate how the tropical forests contribute positively to the water quality maintenance in the current diverse landscape are needed to help to understand the ecosystem services that are provided by the natural vegetation in agricultural watersheds, and its importance for the population. This information can assist the plans and future actions of forest conservation and restoration, and to design policies for water sources protection.

In this context, this study aimed at evaluating the importance of maintaining forest cover for water quality in tropical agricultural watersheds. The specific aims were: (1) to evaluate the water quality in watersheds with different percentages of forest cover; (2) to explore the relationship between forest cover and water quality variability; and (3) to assess the importance of forest cover in agricultural watersheds in keeping water clean.

2.2. Materials and Methods

2.2.1. Study area

The study area is the Sarapuí River basin that includes six experimental watersheds. Sarapuí River is a tributary of Tiete River, located in the São Paulo State, southeastern Brazil (Figure 1), and it supplies four cities, providing water for domestic, agricultural and other purposes. It was originally covered by Atlantic Forest, where Dense Ombrophilous Forest is the predominant forest type (Oliveira-Filho and Fontes, 2000). The forest patches remaining are within a complex matrix composed by agriculture (at large and small scale), pasture, planted forest (*Eucalyptus* and *Pinus*), and urban areas. Agriculture is the backbone of the economy, especially the production of grains, fruits, and vegetables.

The region is under the influence of Cwa climate (humid temperate with dry winters). Annual precipitation is between 1354.7 mm and 1807.7 mm (CEPAGRI, 2014), and the rain mostly falls from October to March.

Figure 1. Atlantic Forest distribution and the sampling sites location in the Sarapuí River basin, São Paulo State, Brazil.

Six watersheds (numbered S1 to S6) were selected based on physical characteristics and percentage of forest cover to establish the sampling sites. We selected watersheds of low-order streams (3rd order) with similar area, shape, average slope, and soil types, but with varying forest cover, including three forested watersheds (more than 55% of forest cover – S1, S2 and S5) and three degraded watersheds (35% or less – S3, S4 and S6). According to Vannote et al. (1980), low-order streams (orders 1-3) are strongly influenced by terrestrial inputs.

2.2.2. Data sources

River network and a 5m-resolution Digital Elevation Model (DEM) derived from official topographic information (IGC, 1:10.000 scale) were used to delimit the watersheds. We employed the standard tools for hydrology and watershed, which are available in the GIS to delineate the six 3rd-order watersheds, basing on the sampling sites and the watershed areas upstream.

Land-use/land-cover (LULC) data was extracted from SPOT satellite images (2.5m-spatial resolution; panchromatic band, year: 2010) supplied by the Environment Secretariat of the São Paulo State, Environmental Planning Coordination (SMA-CPLA) by on-screen digitizing (1:8,000 scale) with ArcGIS 10.2 software. The follow LULC classes were pre-defined, based on the technical manual on the land use of the Brazilian Institute of Geography and Statistics (IBGE, 2013): water, wetlands, forest, eucalyptus, agriculture, pasture and urban. In this study, agriculture represents the fast-growing vegetables, e.g. onion, potato, pumpkin, strawberry, and lettuce; urban represents the residential areas; pasture comprises grassland destined to livestock activity, even without cattle in the studied period; and forest represents the areas covered by native forest (Atlantic Forest).

The map accuracy and agreement were measured through the confusion matrix and the kappa coefficient (Congalton and Green, 1998), based on 121 ground control points distributed along the six watersheds. We obtained 105 overall hits, which represents 87% of agreement, and a kappa coefficient of 0.83, indicating a very good accuracy of the LULC map (Rosner, 2006).

Nine water quality parameters, including dissolved oxygen (DO), turbidity (NTU), total suspended solids (TSS), inorganic suspended solids (ISS), organic suspended solids (OSS), total nitrogen (TN), total phosphorus (TP), total coliforms (TC), and fecal coliforms (FC), were chosen to represent the water quality in the sampling sites. The samples were collected at bi-weekly intervals during a hydrologic year (from October 2013 to October 2014) summing up 24 observations for each site.

DO was measured through an *in-situ* water quality detector (YSI 556 multiparameter system). Water samples were collected in duplicate to determine NTU, TN, TP, TSS, and OSS, which were kept refrigerated and transported to the laboratory for advanced analysis, following standard methods (APHA, 2005).

NTU was obtained using an automatic turbidimeter (MS TEC – TB 1000). TN was determined by Kjeldahl digestion method (APHA, 2005) using an automatic digester (Buchi – K449). The spectrophotometric determination was applied to measure TP (APHA, 2005). The gravimetric analysis was used to obtain TSS, ISS and OSS (APHA, 2005), where 500 mL were filtrated for each sample. TSS is the total residue portion on the filter, ISS is the total solid portion that remains after the calcination, and OSS is the portion of the solids that is lost in the calcination process. TC and FC were detected by the multiple-tube technique with 100 mL sample (CETESB, 1993), and the results are given in Most Probable Number (MPN). We used Lactose Broth for the presumptive identification of coliforms, and the Brilliant Green Bile Broth for the confirmation and mensuration.

We also measured the streamflow (Q) using the current-meter method, that divides the stream channel cross section into various vertical subsections (Santos et al., 2001). In each subsection, the area was obtained by measuring the width and depth, and the water velocity was determined using a current meter (Global Water Flow Probe – 201). The total discharge was computed by summing the discharge of each subsection.

2.2.3. Statistical analyses

We applied a distance metric (Manhattan distance) to explore the differences in the LULC patterns between watersheds through the vegan package of the statistical software R (R Development Core Team, 2014). Manhattan distance between two items is the sum of the absolute values of the differences of their corresponding components (Shahid et al., 2009), and it is calculated by the equation 1.

$$d(jk) = \sum_{i=1}^n (x[ij] - x[ik]) \quad (1)$$

where d is the distance between two watersheds (j and k) and x is the value of the LULC percentage for each watershed.

Mean, standard deviation and coefficient of variation ($CV = SD/x \cdot 100\%$, where SD is the standard deviation and x is the mean) were calculated to obtain the water quality variables

pattern and variation between watersheds. The range of data values was categorized into three classes based on the natural breaks method to build a map of the water quality spatial variation.

We performed a Pearson's correlation to check for relations between water quality variables. The variables strongly correlated ($r > 0.9$) were discarded from the next analysis. A linear regression was also applied to evaluate the relation between water quality and streamflow. The water quality variables were checked for normality and transformed, when it was necessary, using logarithm forms.

A multivariate analysis of variance (MANOVA) using the Hotelling-Lawley Trace was employed to check if the water quality was different between forested and degraded watersheds. The Hotelling-Lawley Trace is the multivariate equivalent of the t -test, whether the two vectors of means for the two groups (forested and degraded) are sampled from the same sampling distribution (Carey, 1998).

The principal component analysis (PCA) was applied to check for groups between forested and degraded watersheds, also to identify the water quality variables that directed this result and the relation between them.

The statistical analyses were performed using the software RStudio (RStudio Team, 2014) and MVSP 3.22 (Kovach Computing Services, 2007).

2.3. Results

Among the forested watersheds, S5 showed the largest forest cover (75%) followed by S2 (57%) and S1 (55%) (Figure 2). On the other hand, S4, S6, and S3 presented, respectively, 35%, 29%, and 25% of forest cover. Likewise, S4 had the largest agricultural area (54%), followed by S6 (33%), S3 (29%), S1 (27%) and S2 (23%), whereas the S5 had only 16%. S3 showed the largest urban area, representing 11,5% of the watershed, whereas the other watersheds had less than 5%.

Figure 2. Percentage of land use/land cover (LULC) of the study sites, in the Sarapuí basin watershed, São Paulo State, Brazil.

In the same way, the Manhattan distance revealed that the most forested site (S5) is also the most distant watershed according to the LULC pattern, followed by S4, which has the watershed predominantly covered by agriculture (Table 1). This analysis also indicated that S1 and S2 had similar LULC pattern, both classified as forested watershed and with a secondary predominance of agriculture. S3 and S6 also presented short distance between them considering the LULC pattern. These watersheds had a close percentage of forest, agriculture, and pasture, but they differed in the percentage of urban area (Figure 2).

Table 1. Manhattan distance between sites considering the land-use/land-cover (LULC) pattern, in the Sarapuí River basin, São Paulo State, Brazil.

group	site	F	F	D	D	S5
		S1	S2	S3	S4	S5
F	S2	11.43				
D	S3	59.00	64.94			
D	S4	60.70	64.60	69.76		
F	S5	43.00	37.11	101.93	82.33	
D	S6	57.11	64.67	23.57	62.41	95.62

where F = forested watersheds; D = Degraded watersheds.

In general, the forested watersheds (S1, S2, and S5) showed better values of water quality parameters than the degraded watersheds (S3, S4, and S6) (Table 2). The spatial differences in water quality linked to LULC patterns among the watersheds are presented in Figure 3.

Table 2. Mean, standard deviation (Sd) and coefficient of variation (CV) for water quality variables for each site in the Sarapuí River basin, São Paulo State, Brazil.

Water quality variable	Statistical parameters	F S1	F S2	D S3	D S4	F S5	D S6
DO (mg.L ⁻¹)	Mean	7.75	8.03	6.77	7.91	6.91	6.96
	Sd	0.73	0.74	0.98	0.61	0.72	0.71
	CV	0.09	0.09	0.14	0.08	0.10	0.10
NTU	Mean	13.95	17.4	17.82	29.66	11.68	18.03
	Sd	3.59	9.28	6.19	24.5	1.53	7.78
	CV	0.26	0.53	0.35	0.83	0.13	0.43
TSS (mg.L ⁻¹)	Mean	5.15	11.51	10.51	21.13	6.98	13.15
	Sd	3.03	6.29	6.48	28.41	2.02	7.35
	CV	0.59	0.55	0.62	1.34	0.29	0.56
ISS (mg.L ⁻¹)	Mean	2.8	7.22	6.44	15.14	3.14	7.89
	Sd	2.35	4.97	5.3	23.31	1.48	5.58
	CV	0.84	0.69	0.82	1.54	0.47	0.71
OSS (mg.L ⁻¹)	Mean	2.35	4.29	4.07	5.97	3.83	5.27
	Sd	0.90	1.47	1.92	5.35	1.05	2.46
	CV	0.38	0.34	0.47	0.90	0.27	0.47
TN (mg.L ⁻¹)	Mean	0.22	0.21	0.26	0.30	0.17	0.17
	Sd	0.15	0.15	0.12	0.32	0.11	0.15
	CV	0.68	0.71	0.46	1.07	0.65	0.88
TP (µg.L ⁻¹)	Mean	49.66	56.52	67.32	88.95	37.21	59.83
	Sd	35.43	36.68	45.36	85.25	23.84	45.32
	CV	0.71	0.65	0.67	0.96	0.64	0.76
TC (MPN)	Mean	298	540.37	1245.75	1353.87	164.75	699.82
	Sd	415.37	544.54	637.64	535.97	145.41	546.45
	CV	1.39	1.01	0.51	0.40	0.88	0.78
FC (MPN)	Mean	106.44	101.86	330.13	519.75	42.69	90.13

Sd	133.62	133.67	451.41	602.88	72.41	95.80
CV	1.26	1.31	1.37	1.16	1.70	1.06

where F = forested watersheds, D = degraded watersheds, DO = dissolved oxygen, NTU = turbidity, TSS = total suspended solids, ISS = inorganic suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, TC = total coliforms, and FC = fecal coliforms.

Figure 3. Spatial variation of the water quality parameters among the sites in the Sarapuí River basin, São Paulo State, Brazil. Where DO = dissolved oxygen, NTU = turbidity, TSS = total suspended solids, ISS = inorganic suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, TC = total coliforms, and FC = fecal coliforms.

Notwithstanding its high value of DO, the S4 presented the poorest water quality among all the sites, showing the highest value for all the other parameters (Table 2). Moreover, it was also the site with the largest temporal variation, with high values of Sd and CV, followed by S3 and S6.

S3 was the second watershed with the highest values of nutrients (TN and TP) and coliforms (TC and FC), whereas S6 was the second one with highest values of NTU, TSS, ISS and OSS. In the case of TC and FC, S3 and S4 presented values much higher than the other sites.

On the other hand, S5 showed the best values of water quality parameters, followed by S1 and S2, respectively.

Concerning the DO concentration, the watersheds showed mean values above 6mg.L^{-1} , but S3 was the only watershed which presented samples below 6mg.L^{-1} (reaching 4.4mg.L^{-1}), and it was also the site with the lowest annual mean, whereas S2 showed the highest value (Table 2).

There was a correlation between NTU, TSS and ISS (Table 3), because those variables represent the particles in the water. OSS was also related to those variables, but not with the same magnitude.

Table 3. Pearson's correlation among water quality variables in the Sarapuí River basin, São Paulo State, Brazil.

	DO	NTU	TSS	ISS	OSS	TN	TP	TC
NTU	0.07							
TSS	0.00	0.93						
ISS	0.01	0.93	0.99					
OSS	-0.05	0.82	0.89	0.86				
TN	0.18	-0.13	-0.17	-0.17	-0.15			
TP	-0.11	0.54	0.57	0.55	0.54	0.23		
TC	-0.15	0.27	0.23	0.25	0.16	0.16	0.29	
FC	-0.04	0.29	0.25	0.25	0.23	0.19	0.42	0.48

where DO = dissolved oxygen, NTU = turbidity, TSS = total suspended solids, ISS = inorganic suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, TC = total coliforms, and FC = fecal coliforms.

Another relation was TP with TSS, with an $r = 0.57$. Moreover, the linear regression showed that TSS was the water quality variable more related to streamflow ($R^2 = 0.21$,

$p < 0.0001$) (Figure 4), followed by NTU ($R^2 = 0.19$). Although this relation was significant, it is not high, which denotes that streamflow influenced the water quality, but it alone did not explain the water quality variation.

Figure 4. Linear regression of total suspended solids (TSS) as function of streamflow (Q) in the Sarapuí River basin, São Paulo State, Brazil.

Both groups (forested and degraded watersheds) showed a significant difference in water quality according to the multivariate analysis (Hotelling-Lawley's $\lambda = 0.47$; $F = 5$; $P = 0.0003$), as illustrated in Figure 5. It is possible to see that, besides DO concentration, degraded watersheds showed higher values of the water quality parameters than the forested ones, and the greatest temporal variation (Figure 5).

Figure 5. Box plot of the water quality parameters comparing forested and degraded watersheds in the Sarapuí River basin, São Paulo State, Brazil, where DO = dissolved oxygen, NTU = turbidity, TSS = total suspended solids, ISS = inorganic suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, TC = total coliforms, and FC = fecal coliforms.

PCA analysis strengthened the relation between degraded watersheds with higher values of water quality parameters, except with DO (Figure 6). The first PCA axis explained 33.3%, where the main variables that drove the separation between forested and degraded watersheds were TSS, OSS, TP and FC. Only TN was not correlated with PCA axis 1. The second PCA axis represented 31.8% of the data variability, which means a total of 65.1% of explanation.

Figure 6. PCA analysis based on water quality parameters in the six sites in the Sarapuí River basin, São Paulo State, Brazil, where DO = dissolved oxygen, TSS = total suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, and FC = fecal coliforms.

2.4. Discussion

The LULC patterns indicated that the natural ecosystem, in the study area, was replaced by the expansion of production systems as agriculture and livestock (Figure 2), that were the primarily responsible for the deforestation of the Atlantic Forest biome (Silva et al., 2016). The reduction of forest cover affects negatively the water quality, once the most forested watershed showed the best water quality among the watersheds as shown in Table 2 and Figure 3.

The two watersheds that had the most different LULC pattern from the others (S5 and S4) (Table 1) also presented the best and the poorest water quality, respectively (Table 2). It indicates that the LULC pattern had a significant role in controlling the water quality. Besides it had the lowest values of pollutants, the most forested watershed (S5) also presented the lowest temporal variation, whereas the most agricultural watershed (S4) presented the highest values of annual average and temporal variation of the water quality parameters (Table 2).

Great temporal variation in water quality is also an indicative of watershed degradation, once during the rainy season, especially in tropical watersheds, the water degradation in streams with no forest cover is raised due to the other LULC being more prone

to direct surface runoff and, consequently, to transfer pollutants into the river. That demonstrates how important the Atlantic Forest is for the surface water quality (Pinto et al., 2013; Huang et al., 2013; Ou et al., 2016). According to Baker and Miller (2013), forest cover in a watershed promotes the decrease in runoff and increase in interception and soil infiltration, reducing sediment discharge and nutrients and pollutants loading into the river in storm events.

In this context, our results indicated that NTU and TSS responded to streamflow variation (Figure 4) but this relation was not strong, showing that, besides the hydrological influence, the LULC pattern had a great impact on water characteristics. Similarly, Carroll et al. (2013) and Chea et al. (2016) found that spatial variation (between sites) was greater than the temporal variation of water quality. Seasonal variation in precipitation, surface runoff, interception, and abstraction have a strong influence on river discharge, and consequently variations in water quality (Huang et al., 2016), but these effects are controlled by the LULC pattern, once degraded watersheds showed greater temporal variation (Table 2).

Turbidity and suspended material represent the particles in the water, then it was expected that those parameters were correlated, as found by Wang et al. (2013) and Wickramaarachchi et al. (2013). Phosphorus outputs, in turn, are dependent on sediment (Huang et al., 2016), because it is easily adsorbed on mineral particles (Silva et al., 2009) which are represented by the correlation between TP and TSS (Table 3). The S4 presented much higher values of those parameters than the other sites, and it can be related to the non-point sources pollution from the agriculture lands. The forest cover of 35% in this watershed was not enough to control this impact. It was the only stream that reached values close to 100 NTU.

Great values of NTU lead to the decrease of photosynthesis in the water body, affecting the aquatic community, especially in low order streams. The increase of sediment represents an increment in nutrients, toxic metals and other pollutants carried by the soil particles (Nasrabadi et al., 2016; Sharma et al., 2017). Throughout the world, the increase in the sediment load of streams is one of the major causes of degradation on water quality from agricultural watersheds (Gonzales-Inca et al., 2015; Restrepo et al., 2015; Crabit et al., 2016).

Phosphorus is usually the most limiting nutrient in freshwater aquatic ecosystems, leading to eutrophication when it is in excess of what the organisms can use, which is one of the most important causes of water degradation (Huang et al., 2016; Sharma et al., 2017). In this way, S3 and S4 can face problems due to the high values of TP compared to the other watersheds, mostly because of the wastewater from residential areas and runoff from agricultural lands (Figure 3), the same found by Ding et al. (2013) and Gonzales-Inca et al. (2015) in agricultural watersheds. In the same way, S3 and S4 showed the highest values of TN

(Table 2 and Figure 3), indicating the water degradation from residential areas without sewage collection and excess of fertilizer use in agricultural lands (Calijuri et al., 2015; Gallo et al., 2015; Sharma and Sharma, 2015; Giri and Qiu, 2016).

Phosphorus and nitrogen fertilizers accumulate in agricultural soils, these surplus fertilizers may either remain in soils or be exported into surface waters by erosion or leaching. The use of fertilizers and pesticides is common and, in many cases, intense in the Sarapuí River basin (Schnaider and Costa, 2013). In the São Paulo State, nitrogen, phosphorus, and potassium consumption are 181.7 kg per hectare/year, which is above the national average (IBGE, 2012). According to the Brazil National Health Surveillance Agency (ANVISA), Brazil represents 86% of the pesticide and fertilizers consumption in Latin American. Examining some of the watersheds studied here, Schnaider and Costa (2013) found that almost all the farmers (96%) use pesticides for controlling pests, diseases, and invasive plant species and synthetic fertilizers. This uncontrolled use of agrochemicals in these watersheds threat the water quality of the Sarapuí river and its tributary, Pirapora river, that together supply four cities in the region. We can see that, even still presenting 35% of forest cover, S4 showed the highest levels of nutrients in the surface water, which is a result of the agricultural activities in the watershed.

The presence of FC in the water indicates contamination with fecal material of humans or other animals from the residential and agricultural areas, wherein domestic sewage is usually the primary source of FC loading (Reder et al., 2015). Our study area is characterized by agricultural land comprised of small farms with isolated houses also without sewage collection, where the contaminants from the wastewater are directed discarded on the soil or into the river (Costa et al., 2011). Therefore, the residential areas and agricultural lands in S3 and S4 were responsible for the highest values of FC (Table 2), following the pattern found for nutrients. According to the CONAMA Framework Resolution 357/2005 that fixed the conditions for establishing water quality categories in Brazilian aquatic systems to human use, S3 is classified as Class 2, and S4 as Class 3, which request a more complex water treatment than rivers classified as Class 1.

Although it does not affect the drinking water quality directly as the other water quality parameters here, DO is one of the most important water characteristics for the biological integrity of the aquatic environment, and it affects the chemical reactions that occur within the water body (Brooks et al., 2003). In this study, almost all the streams maintained good values of DO concentration except S3, which reached values above 5 mg.L⁻¹. According to Welch and Lindell (2004), warm water (tropic rivers) requires DO levels of at least 5 mg.L⁻¹ to sustain fish

populations. These low values of DO concentration in the S3 can be related to the eutrophication process due to the wastewater coming from the urban area, but also to its low streamflow registered during the year, once DO is directly associated with streamflow (Uriarte et al., 2011). The human activities, in this case, can be responsible also for the alterations in water yield, affecting the mixing processes that contribute to water aeration.

We highlight that, even presenting agriculture areas, the forested watersheds showed better water quality than degraded watersheds, indicating the importance of the forest cover for these systems to minimize sediments, nutrients and coliforms loads into the river, as shown by the results of Hotelling-Lawley Test (Figure 4), providing drinking water and simplifying the water treatment for water supply. On the other hand, the forest cover below 35% in the degraded watersheds was not enough to reduce the pollutants as it was for the watersheds predominantly forested (55% or more). Ding et al. (2013) pointed out that the water quality in stream catchments is severely influenced by the non-point source pollution resulting from farmland and residential land in agriculture watersheds, whereas forest can effectively mitigate water quality degradation. Knee and Encalada (2014) stated that forested streams consistently have the lowest pollutant concentration in a rural tropical forest region similar to our study area.

Accordingly, degraded watersheds were grouped due to the higher values of TSS, OSS, TP and FC (Figure 5), whereas forested watersheds were associated with high values of DO, the same pattern found by Huang et al. (2016) using PCA analysis. Thus, mechanisms such as better soil management practices in agriculture, forest cover restoration projects, and riparian vegetation restoration efforts are some of the essential steps needed to improve water quality in agricultural watersheds, decreasing the runoff, sediment losses, and loading rates of nutrients and coliforms (Gunawardhana et al., 2016).

Second, it might also be considered that wastewater from the residential and agricultural lands was combined with the pollutants from the croplands, which aggravated the water quality deterioration. It is important to control the point source pollution of wastewater from these areas. Therefore, parallel to forest restoration plans, there is a needed for the implementation of basic sanitation in this region, avoiding the sewage disposal on the soil and into the river, which is still a huge problem in tropical areas like South America and Africa (UNICEF and WHO, 2015).

2.5. Conclusion

Watersheds predominantly covered by forest have lower pollutant concentration than watersheds primarily covered by other uses. Forest cover has influence in reducing sediment, nutrients, and coliforms loads into the river. On the other hand, degraded watersheds, especially those covered by agricultural and urban lands, present the highest values of pollutants and the greatest temporal variance of water quality parameters, which is also an indicative of human impact.

Besides de LULC impact on water quality, stream flow also contributes to its variation, increasing turbidity, sediments and phosphorus in storm events.

Our findings suggest that the impacts sources on water quality in agricultural watersheds mainly come from runoff from croplands and residential wastewater, wherein the tropical forest cover plays an important role in minimizing the impacts of human activities on watershed ecosystem services.

Forest restoration, better soil management practices, basic sanitation implementation, and point source pollution control are all needed to improve water quality in tropical agricultural watersheds.

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3. EFFECTS OF LAND USE/LAND COVER ON WATER QUALITY OF LOW-ORDER STREAMS IN SOUTHEASTERN BRAZIL: WATERSHED VERSUS RIPARIAN ZONE

Abstract

Natural habitats conversion into agricultural and urban lands is one of the major factors of water quality degradation. Especially in low-order streams, water quality is strongly influenced by the land-use/ land-cover (LULC) pattern. However, this relation may be different for various water quality parameters at different spatial scales. Here, we evaluated the LULC effects on water quality of low-order streams, comparing the watershed and riparian zone influences, to identify the key factors determining water quality variability. We collected water samples in low-order streams in Brazil to obtain data of nutrients, sediments and fecal coliforms. We also used streamflow, precipitation and temperature data. The water quality parameters were analyzed separately and together by statistical models and multivariate analysis. We also investigated the influence of the seasonal effect. The results indicated that the forest cover plays an important role in keeping water clean, while agriculture and urban areas lead the water quality degradation. Pasture had mixed effects, but in general was not correlated with poor water quality, once the livestock activities are not intensive in the area. Dissolved oxygen, total phosphorus, total suspended solids and fecal coliforms responded better to the LULC composition within the watershed, while total nitrogen and organic matter were more affected by the landscape composition in the riparian zone. Besides the spatial variation, the water quality also varies with seasonal changes, especially with streamflow and temperature variability. The overall water quality variation is explained better by the LULC composition within the watershed than in the riparian zone. It suggests that the water management and forest restoration planning need to adopt a multi-scale perspective, but the actions must be directed to the whole watershed, not only to the riparian zone. Although the importance of the riparian zone, the watershed management in the low-order streams is extremely necessary for the water quality maintenance, aiming at the water resources conservation downstream.

Keywords: 1. Surface water quality 2. Land use pattern 3. Riparian buffer 4. Agricultural watershed 5. Forest cover 6. Landscape composition.

3.1. Introduction

Natural habitats conversion into anthropogenic landscapes due to the increasing human demand for resources is one of the main factors behind the alterations on water quality (Giri and Qiu, 2016; Su et al., 2016). Agricultural and urban lands increase has been described as one of the greatest contributors to the increment of nutrients and sediments in freshwater ecosystems worldwide (Uriarte et al., 2011; Huang et al., 2016). However, non-point sources pollution is tough to identify due to the complex and diffuse nature of the interaction between hydrologic and landscape patterns (Chiwa et al., 2012). Also, the relationship between land use/land cover (LULC) and water quality can occur at different spatial scales, from local to regional effects (Wang et al., 2013; Tanaka et al., 2016).

Low-order streams (1st to 3rd order) dominate the riverine landscape and they maintain the function, health, and biodiversity of entire river networks (Vannote et al., 1980; Wipfli et al., 2007). According to Vannote et al. (1980), they are strongly influenced by terrestrial inputs, which makes them fragile ecosystems, which can suffer dramatic impact from land-use changes.

The relationship between LULC and water quality in low-order streams, despite its importance for the watershed, it is not well documented (Ding et al., 2016). It is crucial to understand those interactions in low-order streams once they are responsible for the water, organic matter, sediments and nutrients transportation downstream (Gomi, 2002).

In this way, the riparian zones can play an important role in maintaining water quality, once they represent the aquatic-terrestrial ecotone. Riparian zones exert important influences on the waterways by mediating the bi-directional flow of matter and energy between the water body and the surrounding hinterland (Hanser et al., 2010). For example, the riparian forest can reduce nitrates, phosphorus, and sediment loading into the river (Krutz et al., 2005; Oliveira et al., 2010; Batson et al., 2012; Saalfeld et al., 2012; Gonzales-Inca et al., 2015; Ou et al., 2016) and it can also influence the energy balance (Tanaka et al., 2016). Replacing riparian forest by other land-cover types leads to a decrease in water quality due to the bank erosion, consequently increasing nutrient and sediment loads into the river (Ding et al., 2013, Ou et al., 2016; Yang et al., 2016).

Some studies have shown that LULC is a better predictor of water quality in the riparian zone than within the watershed (Tran et al., 2010; Shen et al., 2015), while others have found that the LULC patterns at the watershed scale better account for the variability in river water quality (Zhou et al., 2012; Ding et al., 2016). Uriarte et al. (2011) and Tanaka et al. (2016) also mentioned that the water quality indicators had presented different response for LULC patterns, when they were evaluated at different spatial scales. Consequently, authors have highlighted the importance of the multiscale analysis, specially, aiming at the understanding of the impacts of LULC on water quality (Uriarte et al., 2011; Zhou et al., 2012; Tanaka et al., 2016).

In this context, this study aimed at to evaluate the land-use/land-cover (LULC) effects on water quality of low-order streams, comparing the watershed and the riparian zone influences. The specific objectives were: (1) to identify the key factors that influence the water quality variability; (2) to evaluate the relationship between LULC patterns and water quality within the whole watershed and in the riparian zone; (3) to identify which LULC pattern has the strongest influence on water quality in low-order streams.

3.2. Material and Methods

3.2.1. Study area

The study area is the Sarapu River basin located in the So Paulo State, southeastern Brazil (Figure 1). The Sarapu River is a tributary of Tiete River, and it supplies four cities in the State, providing water for domestic, agricultural and other purposes.

Most of the soil types in the Sarapu River basin are red or yellow tropical soils, mainly Latosols, and low-activity clays predominate (Oliveira et al., 1999; Coelho et al., 2003). The watershed was originally covered by Atlantic Forest, with Dense Ombrophilous Forest as the predominant forest type, which has been replaced by agriculture, pasture, eucalyptus and urban areas. Agriculture is the backbone of the economy, especially the production of grains, fruits and vegetables.

The region is under the influence of Cwa climate (humid temperate with dry winters), with the annual precipitation between 1354.7 mm and 1807.7 mm (CEPAGRI, 2014), and the rain mostly falling from October to March.

Six 3rd-order streams (numbered S1 to S6) were selected with similar area, shape, average-slope and soil types, but with different LULC patterns.

Figure 1. Study area location and sampling sites in the Sarapu River basin, So Paulo State, Brazil.

3.2.2. Conceptual approach

The conceptual approach for this study includes the LULC types within the watershed and riparian zone, water quality data, streamflow and, water temperature data. All data were used to build statistical models for each water quality parameter and a multivariate analysis integrating all the water quality parameters (Figure 2).

Figure 2. Conceptual approach used in this study.

3.2.3. Watershed and riparian zone delineation

Map manipulations and spatial analysis were performed using the Geographical Information System (GIS) ArcGIS 10.2.

River network and a 5m-resolution Digital Elevation Model (DEM) derived from official topographic information (IGC, 1:10,000 scale) were used to delimit the watersheds. We employed the standard tools for hydrology and watershed, which are available in the GIS to delineate the six 3rd-order watersheds, basing on the sampling sites and the watershed areas upstream.

We adopted the Permanent Preservation Area (PPA), according to the Brazilian Forest Code (Brasil, 2012), as riparian zone. Therefore, we established a 30m buffer along the river networks, and a 50m buffer around springs.

3.2.4. Land-use/land-cover maps

The watershed LULC maps were obtained by on-screen digitizing (1:8,000 scale) of SPOT images (2.5m-spatial resolution; panchromatic band, year: 2010), that were supplied by the Environment Secretariat of the São Paulo State, Environmental Planning Coordination (SMA-CPLA).

The follow LULC classes were pre-defined, basing on the technical manual on land use of the Brazilian Institute of Geography and Statistics (IBGE, 2013): water, wetlands, forest, eucalyptus, agriculture, pasture and urban.

In this study, agriculture represents the fast-growing vegetables (short cycle crops), e.g. onion, potato, pumpkin, strawberry, and lettuces. Urban represents the residential areas, once there are no large cities or industrial areas in the study area. Pasture comprises grassland destined to livestock activity, even without cattle in the studied period. Forest represents the areas covered by native forest (Atlantic Forest).

The map accuracy and the agreement were measured through the confusion matrix and the kappa coefficient (Congalton and Green, 1998), based on 121 ground control points distributed along the six watersheds. We obtained 105 overall hits, which represent 87% of agreement, and a kappa coefficient of 0.83, indicating a very good accuracy of the LULC map (Rosner, 2006).

The LULC eucalyptus, water, and wetlands were not used in the models, due their low incidence in the watersheds.

3.2.5. Water sampling and measurement

The following water quality indicators were chosen to represent impacts produced by anthropogenic activities: dissolved oxygen (DO), total nitrogen (TN), total phosphorus (TP), total suspended solids (TSS), organic suspended solids (OSS), and fecal coliforms (FC). The samples were collected at bi-weekly intervals from October 2013 to October 2014 for the six watersheds, when we also measured the temperature (T) and streamflow (Q).

DO and T were measured through an *in-situ* water quality detector (YSI 556 multiparameter system). Water samples were collected in duplicate to determine TN, TP, TSS and OSS, which were kept refrigerated and transported to the laboratory for advanced analysis, following standard methods (APHA, 2005). TN was determined by Kjeldahl digestion method, TP by spectrophotometry, and TSS and OSS by using gravimetric analysis (APHA, 2005). FC

were detected by the multiple-tube technique (CETESB, 1993), and the results are given in Most Probable Number (MPN).

Q was measured using the current-meter method, which divides the stream channel cross section into numerous vertical subsections (Santos et al., 2001). In each subsection, the area was obtained by measuring the width and depth, and the water velocity was determined using a current meter (Global Water Flow Probe – 201). The total discharge was computed by summing the discharge of each subsection.

The daily precipitation data for the study period were obtained from our own Weather Station (Davis Vantage Pro2) close to the Itupararanga Reservoir.

3.2.6. Data analysis

The variables were checked for normality and transformed, when it was necessary, using logarithm forms. We applied Pearson's correlation to determine the strength and directions of the relationships between LULC patterns and the individual water quality parameters.

A multivariate analysis of variance (MANOVA) using the Hotelling-Lawley Trace was used to check differences between rainy (October – March) and dry season (April – September).

After this preliminary analysis, we performed separate generalized linear mixed models (Zuur et al., 2009), to obtain the optimal equation for each water quality parameter through the lme4 package of the statistical software R (R Development Core Team, 2014). Firstly, we performed, for the riparian zone and watersheds, a full model for each water quality parameter, including the water quality parameters as dependent variables and, as predictors, the percentage of forest, pasture, agriculture and urban cover types. Based on previous findings, models also included Q as predictor and T as covariate to include the season effect (McDowell and Asbury 1994; Uriarte et al., 2011). Watershed and time (monthly) were used as random components. We also tested a 3-day and a 7-day precipitation period prior to each water quality sampling, in order to predict streamflow.

The Akaike Information Criteria (AIC) and Bayesian Information Criteria (BIC) were used for model selection (Akaike, 1998). The backwards stepwise removal of predictors was employed to assess which individual predictors were important drivers for each water quality parameters. The final models were selected based on the lowest value of AIC and BIC, but only predictors with significance level of 0.05 were considered.

A Pearson's correlation between the LULC types was used to avoid variables strongly correlated. The forest cover was not included as predictor in the final model because it was strongly negatively correlated with the other LULC types.

We calculated model goodness of fit as the proportion of explained variance (R^2) using the method described by Xu (2003), that computes the residual variance of the full model against the residual variance of a fixed intercept-only null model.

The redundancy analysis (RDA), which is an extension of multiple regression to model multivariate response data (Zuur et al., 2007), was applied to evaluate the global descriptions about the influences of LULC pattern on water quality, considering the watershed and riparian zone. The RDA is a form of constrained ordination that examines how much of the variation in the set of independent variables explains the variation in the set of dependent variables. This analysis allowed us to simultaneously examine the influences of the LULC types on all of the water quality parameters. For this analysis, the logarithm transformation was applied for the annual average of each water quality parameter. A Monte Carlo permutation test (999 permutations) was used to determine the statistical validity of the RDA. We used the RDA function in the vegan package of the statistical software R.

All the statistical analyses were performed using the software RStudio (RStudio Team, 2014).

3.3. Results

3.3.1. Land use/land cover composition

The proportion of forest cover at the watershed scale was greater than land uses in S1, S2, and S5, wherein the last one showed the greatest forest cover (75%), followed by S2 (57%) and S1 (55%) (Figure 3 and Table 1).

Conversely, S4, S6 and S3 presented, respectively, 35%, 29%, and 25% of forest cover in the watershed. Likewise, S4 had the largest agricultural area (54%), followed by S6 (33%), S3 (29%), S1 (27%) and S2 (23%), while the S5 had only 16%. S6 and S3 presented the highest values of pastureland (32% and 28%), S1 and S2 showed similar values of pasture cover (between 11% and 12%), while S4 and S5 had only 3%. S3 showed the largest urban area, representing 11,5% of the watershed.

Figure 3. Land use/land cover for the six watersheds and their riparian zone, in the Sarapuí River basin, São Paulo State, Brazil.

Considering the LULC in the riparian zone, S1, S2 and S5 showed forest cover greater than 70%, while S3 showed the lowest value (39%), followed by S6 (45%) and S4 (56%) (Table 1). S1 and S2 showed the lowest value of agriculture in the riparian zone (between 6% and 8%). S4 showed the highest value for agricultural lands (26%) and S3 for urban areas (10%), the same pattern from the LULC within the entire watershed.

Table 1. Land use/land cover (%) for the six watersheds (W) and their riparian zone (RZ), in the Sarapuí River basin, São Paulo State, Brazil.

Site	Location	Water	Agriculture	Eucalyptus	Forest	Pasture	Urban	Wetland
S1	W	0.50	26.66	3.28	55.16	11.49	1.14	1.77
	RZ	2.06	7.66	2.66	70.39	9.64	1.56	6.04
S2	W	0.75	23.14	1.92	57.40	11.91	3.47	1.40
	RZ	2.61	6.64	0.06	76.85	6.76	2.15	4.93
S3	W	0.54	29.31	3.68	25.17	27.96	11.45	1.90
	RZ	2.32	19.66	5.37	39.40	16.60	10.13	6.52
S4	W	0.81	54.39	0.92	35.21	3.47	3.31	1.88
	RZ	3.99	26.43	0.56	56.01	3.25	2.37	7.39
S5	W	1.76	16.62	0.82	75.01	3.34	0.82	1.63
	RZ	4.99	10.64	0.00	78.11	0.75	0.42	5.09
S6	W	1.64	32.97	2.31	28.96	32.09	1.25	0.79
	RZ	5.90	22.34	3.31	45.70	19.36	0.63	2.76

An important characteristic of the study area are the agricultural lands, which are comprised by small farms, having families living, in many cases, in the property (Ribeiro et al., 2011). Consequently, the agriculture class revealed isolated houses (Figure 4). The predominant fast growing vegetables found in the study area were onion, potato, pumpkin, cucumber, leek, strawberry, artichoke, and lettuce.

Figure 4. Detail of one small farm, with houses, in site 4, in the Sarapuí River basin, São Paulo State, Brazil.

3.3.2. Impacts of land use/land cover on water quality

Water quality parameters were correlated with LULC patterns in accordance with the results of Pearson's correlation (Table 2), and mixed models (Table 3).

Forest cover was the main land cover associated with good water quality, while urban and agricultural lands were related to water quality degradation, considering both, watershed and riparian zone. After forest cover, the next LULC type that showed the lowest impact on water quality was pasture.

The proportion of forest cover was positively correlated with DO and negatively correlated with solids (TSS and OSS), TP, TN and FC within the watershed and riparian zone. Although, the correlation between forest and DO was strongest at the riparian zone. Conversely, urban area was negatively correlated with DO and positively with the other parameters. Agriculture was positively correlated with solids, TP, TN and FC, but also with DO at the watershed scale.

Table 2. Pearson's correlation between the percentage of land use/land cover (LULC) and water quality parameters, in the Sarapuí River basin, São Paulo State, Brazil.

Location		DO (mg.L ⁻¹)	TSS (mg.L ⁻¹)	OSS (mg.L ⁻¹)	TN (mg.L ⁻¹)	TP (µg.L ⁻¹)	FC (MPN)
Watershed	Forest	0.20	-0.55	-0.51	-0.47	-0.74	-0.59
	Agriculture	0.34	0.88	0.70	0.76	0.94	0.87
	Pasture	-0.55	-0.06	0.09	-0.21	0.07	-0.12
	Urban	-0.32	0.12	0.06	0.51	0.39	0.50
Riparian zone	Forest	0.47	-0.49	-0.55	-0.28	-0.59	-0.47
	Agriculture	-0.25	0.79	0.80	0.48	0.78	0.73
	Pasture	-0.45	-0.06	0.03	-0.13	0.12	-0.08
	Urban	-0.34	0.04	0.14	0.53	0.35	0.47

Where: DO = dissolved oxygen, TSS = total suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, and FC = fecal coliforms.

There was a difference between rainy and dry seasons in the water quality (Hotelling-Lawley's $\lambda = 0,26$; $F = 2,85$; $P = 0.015$), due to the changes in precipitation, temperature, and streamflow.

Streamflow varied with precipitation, and the 3-day precipitation period prior to each water sampling had better value than the 7-day ($R^2 = 0.61$ and 0.58 , respectively). Almost all the models for water quality included streamflow as predictor (Table 3), showing that water quality parameters varied with changes in water discharge. This variable showed a better prediction than previous precipitation in the models. Temperature was also a predictor for DO and TN for both the watershed and for the riparian zone models.

The models explained 49% to 72% of the observed variance in water quality parameters (Table 3). Those parameters showed different responses related to both the whole watershed and the riparian zone. In the case of DO, TSS, TP and FC, they can be explained by the proportion of LULC at the whole watershed. On the other hand, models that considered LULC in the riparian zone were better predictors of in-stream concentration of TN and OSS.

Table 3. Parameter estimates for the best models of water quality variation, in the Sarapuí River basin, São Paulo State, Brazil.

Variable	Location	AIC	BIC	R ²	Int.	flow	T	Agri	Past	Urban
DO (mg.L ⁻¹)	W	249.4	271.7	0.72	8.64**	2.46*	-0.13**	1.24*	-2.41*	
TSS (mg.L ⁻¹)	W	229.7	250.5	0.60	0.79**	15.21**		1.69**		6.42**
OSS(mg.L ⁻¹)	RZ	171.3	192.1	0.51	0.55**	9.69**		1.75**		2.46*
TN (mg.L ⁻¹)	RZ	319.5	336.8	0.49	4.43**	-0.06*	-0.09**			5.67**
TP (µg.L ⁻¹)	W	206.0	226.1	0.67	3.03**	5.67**		1.64**		4.86**
FC (MPN)	W	340.4	355.8	0.65	2.26**			4.91**		5.04*

Where: W = Watershed, RZ = riparian zone, Agri = Agriculture, Past = Pasture, DO = dissolved oxygen, TSS = total suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, and FC = fecal coliforms. *Significance level of 0.05; ** Significance level of 0.01.

The DO concentration showed the lowest value in the watershed with the highest percentage of urban area (S3). Concerning the other LULC types, the proportion of forest increases the DO concentration, while it decreases with pasture. The watershed model also considered agriculture as a predictor of increase of DO. This water quality parameter responded better to the composition of LULC in the whole watershed than the riparian zone. As expected, DO concentrations increased with streamflow and decreased with temperature (Table 3).

Similarly, the parameters TSS, TP, and FC also showed better response for the LULC composition in the whole watershed, wherein agriculture was the most important predictor of the increase of those variables, followed by urbanization. Inversely, forest cover represented decrease of them. Not surprisingly, TSS increased with streamflow, as well as TP. However, streamflow did not represent relation with FC variation.

The LULC in the riparian zone were better predictors of TN and OSS concentrations. Urban areas in the riparian zone were the most important LULC type for the increase in TN, and agriculture was also important for the input of OSS. We can also highlight two results, that watersheds covered by forest presented lower levels of TN than the others, and that TN and

OSS showed negative relation with pasture. In the case of TN concentration, it decreases with the increase of temperature, but decreases with rise in streamflow.

The RDA results showed that, considering all the water quality parameters simultaneously, the variation was better explained by LULC composition within the whole watershed than by the riparian zone configuration (Figure 5). The first canonical axis (Figure 5) explained the most of the water quality variation. In this scenario, the RDA model explained 82% of the variation for the whole watershed and 75% for the riparian zone composition.

Considering the LULC in the watershed, the first axis separated sites with the highest percentage of agriculture and urban lands (S3 and S4) from those with the highest forest cover (S5, S1 and S2) (Figure 5.A). This first axis consistently displayed a water degradation gradient, where sediments, nutrients and fecal coliforms were negatively correlated with forest cover and they were also positively correlated with urban and agricultural lands. It is possible to note that S5, which has the highest forest cover than the other sites, was associated with the best water quality. On the other hand, S3 and S4, which have the highest percentages of urban and agriculture areas, respectively, were associated with poor water quality. Although it has not an extensive forest cover, S6 did not show the same pattern as S3 and S4. This watershed presents the highest percentage of pasture between the sites.

Figure 5. Biplot of the RDA based on water quality and land cover percentages of the six sampling sites for the watershed (A) and the riparian zone (permanent preservation areas - PPA) (B), in the Sarapuí River basin, São Paulo State, Brazil. Where: Agri = Agriculture, Past = Pasture, Urb = Urban, For = Forest, DO = dissolved oxygen, TSS = total suspended solids, OSS = organic suspended solids, TN = total nitrogen, TP = total phosphorus, and FC = fecal coliforms.

3.4. Discussion

The LULC maps indicated that some watersheds are predominantly covered by agriculture and pasturelands due to the agricultural occupation process that occurred in this region (Schneider and Costa, 2013). The S4 is an example with more than 50% of the watershed covered by agriculture. Vettorazzi and Valente (2016) also observed that watersheds which suffered an increase in agricultural areas had a drastic reduction of forest cover. This fact is not suitable for the water quality conservation, once the reduction of forest cover and the increase in agricultural activities lead to a rise in sediments and nutrients loading into the river (Table 2).

Conversely, the studied sites presented their riparian zone with a higher percentage of forest than the whole watershed, reflecting the efforts of conservation of riparian zones, and also the difficulty in expanding agricultural activities over those areas due to the steep relief or predominance of wetlands (Mello et al., 2014).

The forest cover, both for watershed and for riparian zone, was related to good water quality, and it plays a major role in keeping water clean in different watersheds in the world (FAO, 2008; Wang et al., 2013; Oliveira et al., 2016). Precisely, because sediments, nutrients, and fecal coliforms were positively related to the agricultural and urban uses and negatively to forest cover. Studies have shown that agriculture and urban areas represent the most important LULC that induce the water quality degradation (Zhou et al., 2012; Ding et al., 2016; Huang et al., 2016; Ou et al., 2016). In the agricultural lands, excessive fertilizers, runoff and soil erosion led to the increase in sediment, nutrients, chemical contaminants, and organic matter into the water body (Poudel, 2016). In urban areas, the increase of wastewater and pollutants accumulate on the impervious surfaces resulting in the water deterioration (Lee et al., 2009; Ding et al. 2016), and although small in percent coverage, it exerts a disproportionately large influence on water quality (Zhou et al. 2012),

In our study, urban use is composed of residential areas and agriculture comprises small farms, and they represent the most important vectors of water degradation, as found by Ding et al. (2013) and Gunawardhana et al. (2016). On the other hand, forest cover behaves as a filter, controlling and decreasing the sediment and pollutants, that are carried in surface runoff (Ding et al., 2013), also increasing DO concentration (Table 2 and Figure 5).

Unlike other studies that showed negative impacts of the pasture on water quality (Uriarte et al., 2011; Tanaka et al., 2016; Ou et al., 2016), our results did not associate this land use with water degradation for most of the water quality parameters. Pasture only had an adverse impact on DO. Ding et al. (2013) and Wang et al. (2013) also found a positive

correlation between water quality and grassland, which means that both forest and grassland can effectively mitigate water quality degradation (Wang et al., 2013; Ding et al., 2013; Ding et al., 2016). It is important to note that our study area do not have large cattle ranches, and many of those areas are abandoned pasture lands. In this way, adverse impacts from the presence of the animals on the water quality may not occur in the same magnitude as found by other studies. According to Latawiec (2015), the low-productivity pasturelands with few cattle are a great opportunity for forest restoration in the tropical agricultural landscapes. The increase in pasture productivity could provide new areas of land for restored forest.

Besides the spatial variation in water quality, we also observed a temporal variation linked to the seasonal changes. According to Huang et al. (2016), seasonal variations in precipitation surface runoff, interception, and abstraction have a strong influence on river discharge and, consequently, variations in sediment and nutrient concentrations are a seasonal phenomenon. Oliveira et al. (2016) highlighted that in regions with a well-defined rainy season, like many parts of Brazil, the non-point pollution sources usually have a reduced effect on the stream water quality due to the decrease in runoff during the dry season. Thereat, the addition of temporal and hydrological data was essential to determine the effect of the landscape pattern on water quality in the region.

The percentages of forest, agriculture and urban lands were the most important LULC that explain the spatial variations of water quality parameters (Table 3), wherein urbanization and agriculture were the main predictors of the increase of water quality parameters. Recent studies also showed a reduction of independent variables of the models with LULC class mainly because of the existing collinearity between the LULC classes (Gonzales-Inca et al., 2015; Li et al., 2015; Oliveira et al., 2016). According to Oliveira et al. (2016), this approach allows to easily identify the main classes responsible for the variations in water quality.

Our results showed that DO concentration responded better to LULC within the watershed than in the riparian zone, and the forested watersheds have the highest values for DO, alike found by Uriarte et al. (2011) and Ding et al. (2016). However, the correlation analysis showed that the riparian forest had an important role in increasing DO concentration, which is greater than we had considered for the forest in the whole watershed. According to Abell and Allan (2001), forested streams have cooler water and higher oxygen concentration levels. DO concentration was negatively correlated with temperature, once the DO demand increases with the increase in temperature, while the oxygen solubility in the water decreases (Manahan, 2004). Thus, the negative impact of pasture on DO can be connected to the fact that these lands led to an increase of water temperature. The discharge of organic matter into the

water body from urban areas also leads to low DO concentrations, especially in areas that do not have sewage collection and treatment (Calijuri et al., 2015). On the other hand, increasing streamflow raises the water oxygenation, which can occur in agricultural areas (Uriarte et al., 2011; Wang et al., 2013).

The results also indicated that TP and TSS were strongly explained by the LULC composition in the watershed and weakly in the riparian zone, and it was the same pattern found by Uriarte et al. (2011) and Ding et al. (2016). We can highlight, in this context, that agriculture represented the main source of sediments and phosphorus, followed by urban land. The suspended solids reflect the intense agricultural activity, where the farmers do not implement conservation practices, which leaves the soil constantly exposed and subject to erosion, as so found by Calijuri et al. (2015). Also, the hill slopes in the study area can enhance the huge soil erosion from the exposed soil, as pointed by Sharma and Sharma (2015) on watersheds covered by shifting cultivation.

The parameters TP and TSS also increased with stream discharge. In rain events, the soil erosion is potentiated, and sediments with nutrients are transported into the river (Poudel, 2016). In this way, this study highlighted that the agricultural practices in fast-growing vegetable crops need to adopt strategies to minimize soil erosion and runoff.

FC were also better explained by the percentage of agriculture and urban area in the watershed, what is common in the literature, specially linked to human activities and animal waste (Meays et al., 2004). However, in our study, they were much more correlated to agriculture than with urbanization and, it can be related with the fact that agriculture represents, in the sites, higher percentage of the landscape than urban areas (Ou et al., 2016). In addition, the agricultural lands in the study area are comprised of small farms with isolated houses without sewage collection (Costa et al., 2011). The sewage is often discharged directly into the watercourse, on the soil or in a cesspit without structure to avoid soil and groundwater contamination, called “black cesspit”, a common problem in rural areas in Brazil (Pinto et al., 2012). It brings the importance of the basic sanitation services in rural areas, and the biodigester has been an alternative in some agricultural watersheds in Brazil (Costa and Guilhoto, 2014).

Contrary to the other water quality parameters, TN and OSS were stronger correlated with the LULC composition in the riparian zone than in the watershed (Table 3), where agriculture and urban areas were responsible for the in-stream nitrogen and organic matter, and forest cover represents decrease in these parameters. Sewage from domestic and food processing sources contains a wide variety of pollutants, including organic pollutants

(Manahan, 2004). In the same way, Gonzales-Inca et al. (2015) found that riparian vegetation is important explaining nitrate concentrations due to its potential of nutrient pollution mitigation in agricultural watersheds. The nitrogen cycle is complex, and it depends on many factors within the aquatic, terrestrial and aerial environment (Manahan, 2004). This chemical element is in constant transformation and it is used by many organisms in the waterbody and in the riparian ecosystem (microorganisms and plants) (Wetzel, 1993; Brooks et al., 2003). Due to these characteristics, it is understandable that our riparian zone model presented better results than the model for the watershed. OSS represents the organic matter in the water, and it is also strongly linked to the riparian composition and its interaction with the aquatic ecosystem (Wetzel, 1993).

As expected, NT concentration in the water decreases with the increase in temperature, because the microorganisms' activities in the water and soil are affected by temperature (Mello et al., 1983; Wetzel, 1993). However, in contrast to other studies that showed a positive relation between streamflow and nitrogen (Uriarte et al., 2011), our study showed a negative relation. The dilution effect played an important role for the sampling sites located in small watersheds, which means abundant discharge reduced the concentration of TN, as it was so pointed by Wang et al. (2013). According to Dellagiustina (2000) and Gallo et al. (2015), the agricultural activities may be not enough to overcome the dilution effect of the precipitation events. Gallo et al. (2015) also pointed out that these sites are more sensitive to shifts in hydrologic partitioning in response to land-cover change than those associated with climate change.

Concerning the RDA results, we ascertained that the LULC pattern at watershed was better than at riparian zone when it was related to the overall water quality variation as found by Ding et al. (2016) and Gonzales-Inca et al. (2015) for low-order streams. However, the mixed models showed that different parameters of water quality varied in their responsiveness to the different scales when analyzed separately, as reported elsewhere (Uriarte et al., 2011; Zhou et al., 2012). This multiplicity point of view suggests that the water management and forest restoration planning need to adopt a multi-scale perspective, but the actions must be directed to the whole watershed. Although the confirmed importance of the riparian zone, our study demonstrates that the watershed management in low-order streams is extremely necessary for the water quality maintenance.

It is important to note that our study considered a 30m-riparian buffer along rivers and 50m-spring buffer as riparian zone (PPA) following the Brazilian law. Ou et al. (2016), studying the influence of the buffer zone width on water quality, highlighted that the 50m-riparian zone accounts for a low proportion of the whole watershed area, which weakens at this scale the

influence of landscape features on river conditions. The authors found that the 100m-riparian zone had the largest effect on river water quality. It brings the question about the effectiveness of the 30m-buffer for the water sources conservation. In our study, the forest conservation and the anthropic activities at the watershed scale in general had greater importance for the water quality maintenance than the 30m-riparian zone. The studies which reported that the LULC pattern in the riparian zone plays greater importance than the LULC within the watershed used different buffer widths. For example, Tran et al. (2010) considered 200m of riparian zone along the rivers in USA, while Shen et al. (2015) considered a buffer zone of 100m in China.

Ding et al. (2016) highlight that opposite results in studies regarding the LULC impact on water quality might be because different methods adopted to delineate the local buffer zone, differences in regional settings and water quality parameters. Ou et al. (2016) also linked these differences to the sampling strategy and data resolution. The higher or lower resolution of the land-use data may miss some crucial landscape features of the land-use composition and configuration. In our study, we considered seasonal variation, and we used high-resolution spatial data, which can differ from studies that only considered rainy season or used low-resolution data. Other differential in our work is that the agricultural land was comprised by small farms, with isolated houses, without sewage collection, which was strongly connected to water quality degradation.

Nevertheless, our findings highlight the forest cover as a key landscape feature to prevent deterioration of water quality. Additionally, we found that grassland (pasture with few cattle or abandoned pasture) does not lead to water degradation as agriculture and urbanization, and it could be a good opportunity for forest restoration plans. This information can help managers to establish best alternatives for the water quality improvement in the headwater streams. We showed that the environmental planning must consider the whole watershed to design public policies aiming at the water resources conservation, not only the riparian zone, although the riparian forest restoration can be prioritized as a short-term action aiming at the water quality improvement (Vettorazzi and Valente, 2016).

According to Rodrigues et al. (2011), better agricultural and cattle production practices can provide new areas for restoration actions in rural watersheds. Low-order streams, often located on steeper slopes, are more sensitive to nonpoint source pollutants loading than flat areas in the watershed (Yang et al., 2016; Yu et al., 2016). Then, the environmental planning of these areas is crucial to ensure water with high quality downstream.

3.5. Conclusion

We conclude, based in our results, that forest cover is the most important LULC type to maintain the water quality in the low-order streams, and agriculture and urban areas are responsible for the water quality degradation.

Sewage from residential areas, sediments and nutrients loading from the short-cycle crops lead to the nonpoint pollution into the small rivers. Grassland has mixed impact on water quality, and in general does not lead to water quality degradation, once they represent low-productivity or abandoned pasturelands.

Besides the spatial variation, water quality also varies with seasonal changes, especially with streamflow and temperature variability. Thus, streamflow and temperature are also important predictors to explain some of the water quality parameters variation.

The water quality parameters have different responses regarding the landscape composition within the watershed and the riparian zone. However, the overall water quality variation is better explained by LULC composition at watershed scale than at riparian zone. It suggests that the water management and forest restoration planning need to adopt a multi-scale perspective, but the actions must be directed to the watershed management. Although the importance of the riparian zone, our study demonstrates that the watershed management in low-order streams is extremely necessary for the water quality maintenance aiming at the water resources conservation downstream. Nevertheless, the riparian restoration can be prioritized as a short-term action aiming at the water quality improvement.

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4. RIPARIAN RESTORATION FOR PROTECTING WATER QUALITY IN TROPICAL AGRICULTURAL WATERSHEDS

Abstract

Land-use change in riparian zones is one of the most significant threats to water quality in stream ecosystems. Riparian forests play an important role in protecting water quality, and there is a need to assess the role of riparian restoration in reducing nutrients and sediment loading. This study uses watershed simulation modeling to evaluate impacts of riparian forest restoration on water quality in agricultural watersheds. Soil Water Assessment Tool (SWAT) is used to simulate streamflow, suspended sediment and nutrients of the Sarapuí River basin, southeastern Brazil. We found a spatial and temporal variation in water quality impacts resulting from variation in land use/land cover (LULC) and rainfall patterns. Subbasins with agricultural and residential lands showed a higher sediment and nutrients loads than others covered by forest and pasture, and this relation is strongest during the rainy season. Riparian restoration reduced suspended sediment (by 9.26%), total nitrogen (22.6%) and total phosphorus (7.83%). However, forest conservation in the whole watershed cannot be substituted by protecting only the riparian zone preservation, as forested subbasins showed better water quality than those covered by other LULC types. The results show the importance of restoration plans in the riparian zone, providing a scientific understanding of the role of riparian forest for the water quality improvement by reducing nutrients and sediments loading into the river.

Keywords: 1. Watershed modeling 2. Land use 3. Riparian zone 4. Forest restoration 5. Water resource 6. SWAT.

4.1. Introduction

The riparian zone is the interface between terrestrial and aquatic ecosystems and can provide many ecosystem functions, as well as playing an important role on nutrients and sediments transfer (Kuglerová et al., 2014; Ou et al., 2016). Although the riparian forest can contribute to the water quality protection, it is one of the most degraded ecosystems in the world (Nilsson and Berggren, 2000; Kuglerova et al., 2014). Replacing riparian forest by other land uses decreases water quality due to bank erosion, increasing nutrient and sediment loads into the river (Ding et al., 2013, Ou et al., 2016; Yang et al., 2016a). Therefore, land use change in riparian zones is one of the most significant threats to water quality in stream ecosystems, bringing the importance of understanding the benefits from their restoration (Zhang et al., 2013; Yang et al., 2016b).

Riparian forest cover can play a major role in the biogeochemical cycles in watersheds. The vegetation provides protection against erosion, retention of pollutants, excessive nutrients runoff and increased water temperature (Sopper, 1975; Sweeney et al., 2004; Lima and Zakia, 2006; Mingoti and Vettorazzi, 2011; Schilling and Jacobson, 2014; Tanaka et al., 2016) and benefits both the quality of human life and survival of aquatic species (Saalfeld et al., 2012;

Ding et al., 2013; Yang et al., 2016a; Tanaka et al., 2016). Especially in the currently context in that agriculture and urban growth have been appointed as the primary cause of water quality degradation (Uriarte et al., 2011; Lin et al., 2015; Huang et al., 2016).

Riparian buffers have been pointed as effective in reducing nutrients and sediment loading into streams (Uusi-Kämpä and Jauhiainen 2010; Gonzales-Inca et al., 2015; Monteiro et al. 2016; Ding et al., 2016). Cho et al. (2010) observed that intact riparian forest reduced 20.5 % of sediment load in a watershed in USA. In this context, the proximity to surface water has been used as a criterion for prioritizing areas for forest restoration in watersheds (Vettorazzi and Valente, 2016). However, the effectiveness of the riparian buffers depends on the hydrogeological condition, and it is influenced by numerous factors and varies from local to regional scales (Wang et al., 2013; Gonzales-Inca et al., 2015).

Land use impacts on water quantity and quality are assessed using hydrological models in urban and agricultural watersheds (Wangpimool et al., 2013; Pereira et al., 2014; Lin et al., 2015; Wagner et al., 2016; Zhao et al., 2016). Lin et al. (2015) assessed the hydrologic and water quality impacts of agricultural land changes in North America. Nejadhashemi et al. (2012) used models to evaluate land use change impacts in agricultural watersheds in Midwest.

Studies have also reported the impacts of riparian land use on water quality worldwide using hydrological models. Gitau et al. (2006) used a model in evaluating a combination of best management practices that include riparian buffers in reducing phosphorus loads. Moriasi et al. (2011) used a hydrological model to quantify the effect of riparian forest buffer and Bermuda grass filter strip on sediment yield in a reservoir in USA. Yang et al. (2016b) examined the water quality effects of riparian wetland losses and restoration in Canada.

However, the impacts of riparian restoration in tropical river basins are not well documented. Monterio et al. (2016) observed that riparian restoration to comply with Brazilian Forest Code could reduce sediment load by 29.4% in a catchment in the Brazilian Cerrado. The Atlantic Forest, in particular, is a biodiversity hotspot (Mittermeier et al., 2011) highly threatened since it has been reduced to only 11% of its original cover (Ribeiro et al., 2009). Its reduction has affected the water quality and, consequently, water supply (Strassburg et al. 2016).

There is a need to understand the importance of tropical forest cover to water quality and how the riparian forest restoration can contribute to the water sources conservation. Here, we use a watershed hydrological model to simulate the effects of riparian zone restoration on water quality in tropical agricultural watersheds. It presents a comparative evaluation of watershed-wide and riparian zone forest restoration on water quality for policy considerations.

This is important in using riparian zone management within the broader context of watershed systems. The specific objectives were: (1) to model hydrological processes for streamflow and water quality in watershed system; (2) to evaluate the impacts of agricultural and residential lands on sediment and nutrient loadings; (3) to assess the importance of the tropical forest cover on water quality; and (4) to simulate a riparian zone restoration scenario and its impacts on agricultural watershed system.

4.2. Material and Methods

4.2.1. Study Area

The study area is the Sarapuí River basin (1,550 km²), located in the São Paulo State (between the coordinates UTM 23S 195,000 m and 265,000 m; 7,360,000 m and 7,420,000 m), southeastern Brazil (Figure 1).

Figure 1. Land use/land cover of Sarapuí River basin and the six experimental watersheds, Brazil.

The main rivers of the basin flow from east to west. The drainage network is particularly dense in the eastern portion of the basin, where the highest elevations can be found. The density of the drainage network decreases in the central and west part. Most of the soil types in the Sarapuí River basin are red or yellow tropical soils, mainly Latosols, and low-activity clays predominate. However, other soil types also occur in the basin, as Gleysols and young soils like Regosols and Fluvisols. Gleysols occur more on the central and west portion of the basin. The region is under the influence of Cwa climate (humid temperate with dry

winters). Annual precipitation is between 1355 mm and 1808 mm (CEPAGRI, 2014), and the rain mostly falls from October through March. Between December and February are the maximum precipitations (around 200 mm monthly) (CEPAGRI, 2014). Temperatures range from 5°C to 32°C, with an annual average of 20°C. January and February are the hottest months (monthly temperature average of 24°C), and June and July are the coldest months (monthly temperature average of 16°C).

Sarapuí River is a tributary of Tiete River, one of the main rivers in Brazil. Tiete river showed the lowest values of the national water quality index in 2014 (SMA, 2014). Sarapuí River basin is a particular case of an agricultural watershed close to very high-density urban regions like Sorocaba and Sao Paulo. Sarapuí River supplies four cities in the São Paulo State, and its proximity to those regions represent trends for agricultural and urban sprawl due to the demands from the big cities. It was originally covered by Atlantic Forest, which has been replaced by agriculture, pasture, eucalyptus and residential (urban) areas. Pasture and agriculture lands represent, together, 53% of the basin. Agriculture is characterized by the production of grains, fruits, and vegetables. Despite the agricultural activities, the basin still presents 37% of forest cover.

We selected six 3rd-order streams (numbered S1 to S6) (Figure 1) within the Sarapuí River basin to be our experimental watersheds and to establish the sampling sites, where we collected water samples during 2013 and 2014 for comparing with simulated data. We selected watersheds with similar physical characteristics: area, shape, average slope and soil types, but with different LULC patterns: three forested watersheds (more than 55% of forest cover) and three degraded (less than 35%). S5 has 75% of forest cover followed by sites 1 and 2 (both with 60% of forest cover). S3 presents only 25% of forest cover, 30% of agriculture and the largest residential area (12%). S4 has the largest agricultural area (54%) and 35% of forest cover. S6 is 29% covered by forest, 32% by agriculture and 32% by pasture.

4.2.2. Conceptual Model

The conceptual framework for the Sarapuí River Basin Model consists of a system including the watershed hydro-physical characteristics, land use/management and conservation actions (riparian restoration), with water quality as the effect (Figure 2). We used slope, hydrology, climate, soil and LULC information to calibrate a hydrological model for the Sarapuí River basin to assess the water quality characteristics and their relation to LULC pattern. Then, we created a new LULC map simulating a 100% riparian restoration. The new scenario was used in the model to simulate water quality responses of riparian forest restoration.

Figure 2. Conceptual model for the Sarapuí River basin, Brazil.

4.2.3. SWAT Model Development

Models have long been used in water resources management to guide decision-making and improve understanding of the watershed system. Hydrological and water quality data can be modeled at different spatial and temporal scales. Integration between water quality monitoring and modeling has been suggested to understand surface water quality better and to make appropriate and effective decisions for nonpoint source pollution control (Davenport et al., 2008). Models calibration enables to simulate alternative scenarios and evaluate the dynamics of the watershed (Pessoa et al., 1997).

SWAT (Soil and Water Assessment Tool) model was used to model streamflow, sediment yield (total suspended solids) and nutrients load (total nitrogen and total phosphorus). SWAT is a non-point source pollution model developed by Dr. Jeff Arnold for the USDA Agricultural Research Service (ARS) (Neitsch et al., 2011). SWAT was developed to predict the impact of land management practices on water, sediment and chemical yields in watersheds with varying soils, land use and management conditions over long periods of time (Neitsch et al., 2011). It is a continuous-time, semi-distributed, process-based river basin model (Arnold et al., 2012). Simulation of the hydrology of a watershed in the model is separated into two major divisions: land phase, that controls the amount of water, sediment, nutrient and pesticide loadings to the main channel; and routing phase, defined as the movement of water and the

other components through the channel network of the watershed to the outlet (Neitsch et al., 2011). It has been applied to understand the sediment losses and nutrient loadings in the watershed around the world (Marshall and Randhir, 2008; Reungsang et al., 2009; Wilson and Weng, 2011; Poudel et al., 2013; Lin et al., 2015).

The SWAT model divides a basin into subbasins connected by stream networks. Each subbasin is further divided into hydrologic response units (HRUs) that are unique combinations of different LULC types, soils, and surface slopes. Within each subbasin, the areas with similar LULC, soil types, and surface slopes are lumped together into a single HRU. The model requires several data, which encompasses LULC map, soil map, elevation map, precipitation, temperature, humidity, streamflow, and water quality input. Those data are necessary for model calibration and validation, the two steps for developing a hydrological model.

In this study, the Sarapuí River basin was divided into 178 subbasins, including our six experimental watersheds. Each experimental watershed was represented by a subbasin in the model.

The model was set to define multiple HRUs with a 2% threshold level for each subbasin – that is, any LULC or soil type that covered 2% or more of the subbasin was defined as an HRU.

4.2.3.1. Data

The inputs for the SWAT model of the Sarapuí River basin are derived from different sources, and they were reprojected in UTM SIRGAS 2000.

We used a 30-m Digital Elevation Map (DEM) from the Environmental Planning Coordination of the São Paulo State (CPLA, 2013). SWAT model produces a slope map using the DEM. For the slope classes, we used the Brazilian classification from Embrapa (1999). We used the stream network data from the Brazilian Institute of Geography and Statistics (IBGE, 2015) (1:50,000 scale). The LULC map was extracted from our own thematic map (1:50,000 scale) obtained by on-screen digitizing of a Landsat TM-5 scene from 2010 (Figure 1). We used our own map of soil types, produced by sensor data and soil samples collected in 2014. The soil types found in the Sarapuí River basin are Gleysol, Yellow Latosol, Red-yellow Latosol, Regosol, Fluvisol, and Cambisol. The soil properties like 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 were collected from the Agronomic Institute of Campinas (IAC) (Oliveira, 1999) and specialized literature (Juhasz et al., 2006; IBGE, 2007; Gomes and Pereira, 2009). The hydrological classification followed Sartori et al. (2005) for

Brazilian soils. We use an equation proposed by Baumer (1990) for albedo, and for soil erodibility factor (USLE – K) an equation proposed by Williams (1995).

Daily weather information was collected from four official stations, provided by the National Meteorological Institute (INMET): São Paulo – Mirante do Santana, São Miguel Arcanjo, Sorocaba, and Iguape. We used the following weather information: solar radiation (MJ/m²/day), wind speed, maximum and minimum air temperature (°C) and relative humidity. The other parameters are derived from those data, using the equations presented on SWAT 2012 manual (Winchell et al., 2013).

Precipitation data for the 1955-2014 period was obtained from five gauge stations from the National Water Agency (ANA, 2015): São Miguel Arcanjo, Tatuí – Sarapuí, Piedade, Sorocaba and São Paulo – Mirante do Santana. Streamflow, suspended solids and nutrients data were collected from ANA at two gauge stations: Sarapuí River and Pirapora River (Figure 1), and from our own data of the experimental watersheds. Daily loads of suspended solids and nutrients were estimated using the product of the daily streamflow volume and the instantaneous water quality measurements.

4.2.3.2. Model Calibration and validation and sensitivity analysis

The calibration and sensitivity analysis were performed by optimization methods using the Sequential Uncertainty Fitting algorithm (SUFI-2) (Abbaspour, 2011) in the SWAT-CUP software (Abbaspour, 2015). We calibrated the model starting with streamflow, followed by sediment and then nutrients as suggested for calibration procedure (Abbaspour, 2011). The streamflow calibration is a first step necessary to obtain the water quality parameters.

Streamflow was calibrated and validated against daily, monthly and yearly streamflow, for a period of 33 years (1980-2012), with three years as a warm up period. The streamflow data from the Sarapuí River gauge were used for observed values for calibration. Streamflow data from the Pirapora river gauge were used for validation. Sediment and nutrients were calibrated for the period of seven years (2004-2010) for the Sarapuí River basin and validated using the six experimental watershed data.

We parameterized the model following suggestions in SWAT manual (Winchell et al., 2013) for three levels: basin, subbasin and HRU levels. We also used parameters identified in the literature that has the potential to influence on streamflow, sediment yield and nutrients loading (Arnold et al., 2012; Andrade et al., 2013; Kushwaha and Jain, 2013; Abbaspour et al., 2015; Fukunaga et al., 2015).

Numerical and graphical performance criteria were used to evaluate model performance during calibration and validation. The numerical performance criteria include coefficient of determination (R^2) (Coffey et al., 2004), Nash-Sutcliffe efficiency (NS) (Nash and Sutcliffe, 1970) and percent bias ($PBIAS$) (Gupta et al., 1999), as shown in Equations 1 to 3:

$$R^2 = \frac{[\sum_i (Q_{m,i} - \bar{Q}_m)(Q_{s,i} - \bar{Q}_s)]^2}{\sum_i (Q_{m,i} - \bar{Q}_m)^2 \sum_i (Q_{s,i} - \bar{Q}_s)^2} \quad (1)$$

where Q is a variable (e.g., discharge), and m and s stand for measured and simulated, i is the i th measured or simulated data.

$$NS = 1 - \frac{\sum_i (Q_m - Q_s)_i^2}{\sum_i (Q_{m,i} - \bar{Q}_s)^2} \quad (2)$$

where Q is a variable (e.g., discharge), and m and s stand for measured and simulated, respectively, and the bar stands for average.

$$PBIAS = 100 \times \frac{\sum_{i=1}^n (Q_m - Q_s)_i}{\sum_{i=1}^n Q_{m,i}} \quad (3)$$

where Q is a variable (e.g., discharge), and m and s stand for measured and simulated, respectively. Percent bias measures the average tendency of the simulated data to be larger or smaller than the observations. The optimum value is zero, where low magnitude values indicate good simulations. Positive values indicate model underestimation and negative values indicate model overestimation (Gupta et al., 1999).

We conducted sensitivity analysis to check which input or calibration parameters have the strongest influence on the model results. A t -test was used to identify the relative significance of each parameter. The sensitivities are estimates of the average changes in the objective function resulting from changes in each parameter, while all other parameters are changing. This gives relative sensitivities based on linear approximations and, hence, only provides partial information about the sensitivity of the objective function to model parameters. In this analysis, the larger, in absolute value, the value of t -stat, and the smaller the p -value, the more sensitive is the parameter (Abbaspour, 2011).

4.2.4. Land-use/land-cover patterns and water quality

After calibrated, the model was applied to evaluate the impact of LULC patterns (agriculture, residential areas, and forest) on water quality for the Sarapuí River basin in a current scenario. We compared subbasins with different LULC types and their water quality

responses. Forested subbasins were compared to agricultural ones regarding sediment and nutrients loading.

The spatial distribution of annual (30-year average) values for water quality attributes was mapped for each subbasin. The values of each water quality parameter were reclassified by natural breaks (Jenks) method into three categories to represent low, medium and high levels.

4.2.5. Riparian forest restoration impacts on water quality

We simulated the riparian forest restoration impacts on the water quality for the Sarapuí River basin. We considered Riparian Zone the Permanent Preservation Area (PPA), according to the Brazilian forest code (law 12651 of 2012). The law sets a buffer of 30m from the river banks and 50m around the springs. FAO recommendation for water quality protection is also 30m, as well as proposed by Welsch (1991). We created a riparian-scenario besides the current scenario to simulate the PPA restoration, adding forest cover in all the PPA buffer in the current LULC map. The PPA buffer was created from the stream network using ArcGIS (ESRI). The new LULC map was added to the calibrated model to run the riparian-scenario results.

We simulated the impact of the riparian forest restoration on total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP). Annual and average loads during 1983-2014 were calculated under the current and riparian scenarios.

4.3. Results and Discussion

4.3.1. Streamflow, sediment and nutrient calibration

The parameters and their calibrated values for streamflow, sediments and nutrients are listed in Table 1. It also presents the results of the sensitivity analysis.

Table 1. Parameters used for calibration of streamflow, sediment and nutrients and their respective method of calibration (r =relative and v=replace), calibrated value and sensitivity analysis (p-value) for the Sarapuí River basin, Brazil.

			Calibrated	Sensitivity
Parameter	Definition	Method	Value	(p-value)
Streamflow				
GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to			0.9779
	occur (mm H2O)	v	30	

GW_REVAP.gw	Groundwater revap coefficient	v	0.14204	0.8252
SURLAG.bsn	Surface runoff coefficient	v	23.09	0.7832
REVAPMN.gw	Threshold depth of water in the shallow aquifer for "revap" or percolation to the deep aquifer to occur	v	317	0.4732
SOL_AWC.sol	Available water capacity of the soil layer (mm H2O/mm soil)	r	-0.1212	0.4248
RCHRG_DP.gw	Deep aquifer percolation fraction	v	0.93	0.1692
GW_DELAY.gw	Groundwater delay time	v	170.28	0.1134
ESCO.hru	Soil evaporation compensation factor	v	0.75	0.0547
SOL_K.sol	Saturated hydraulic conductivity (mm/hr)	r	0.5408	0.0091
CH_N2.rte	Manning's "n" value for the main channel	v	0.0822	1*10 ⁻⁸
LAT_TIME.hru	Lateral flow travel time (days)	v	11.88	1*10 ⁻¹⁰
CH_K2.rte	Effective hydraulic conductivity in main channel alluvium (mm/hr)	v	19	1*10 ⁻¹⁸
CN2.mgt	Manning's "n" value for the main channel	v	0.0822	1*10 ⁻²⁰
ALPHA_BF.gw	Baseflow alpha factor (1/days)	v	0.941	1*10 ⁻³¹
Sediment				
	Exponent parameter for calculating sediment reentrained in channel			
SPEXP.bsn	sediment routing	v	1.202	0.8599
CH_COV2.rte	Channel cover factor	v	0.28034	0.7797
RSDCO.bsn	Residue decomposition coefficient	v	0.0263	0.6781
	Linear parameter for calculating the maximum amount of sediment that can be reentrained during channel sediment routing			
SPCON.bsn		v	0.00476	0.5674
CH_COV1.rte	Channel cover factor	v	0.28034	0.2987
HRU_SLP.hru	Average slope steepness (m/m)	r	0.1932	1*10 ⁻⁷
	USLE equation soil erodibility (K) factor (units: 0.013 (metric ton m2 hr)/(m3-metric ton cm))			
USLE_K.sol		r	0.1	1*10 ⁻¹¹
SLSUBBSN.hru	Average slope length (m)	r	-20.76	1*10 ⁻¹¹
USLE_P.mgt	USLE equation support practice factor	v	0.75	1*10 ⁻⁴²
Nutrients				
SOL_ORGP.chm	Initial organic P concentration in the soil layer (mg P/kg soil or ppm)	v	53	0.9822
SOL_ORGN.chm	Initial organic N concentration in the soil layer (mg N/kg soil or ppm)	v	71.8	0.8934
SHALLST_N.gw	Initial concentration of nitrate in shallow aquifer (mg N/L or ppm)	v	394	0.4403
RCN.bsn	Concentration of nitrogen in rainfall (mg N/L)	v	6.33	0.4284
BIOMIX.mgt	Biological mixing efficiency	v	0.446	0.0903
PPERCO.bsn	Phosphorus percolation coefficient (10 m3/mg)	v	10.135	0.0001
PHOSKD.bsn	Phosphorus soil partitioning coefficient (m3/mg)	v	195.3	1*10 ⁻⁹
CMN.bsn	Rate factor for humus mineralization of active organic nutrients (N and P)	v	0.0025	1*10 ⁻¹¹
NPERCO.bsn	Nitrate percolation coefficient	v	0.002	1*10 ⁻¹⁴

The observed and simulated streamflow for Sarapuí River basin are presented in Figure 3. It was calibrated daily, monthly and yearly, and the comparisons for the observed and the simulated streamflow are shown in Figure 3. The model performance statistics were better for monthly calibration (Table 2), as reported by Lin et al. (2015) in the USA and Schmalz et al. (2015) in China.

The streamflow showed an R^2 greater than 0.6 for calibration and validation process (Table 2), with a value of 0.85 and 0.79 for monthly calibration and validation, respectively. Schmalz et al. (2015) obtained close values of R^2 for calibration (0.88) and validation (0.85).

We obtained an acceptable value of NS (>0.36) considering daily (0.65) and yearly (0.71) calibration and a value equal to 0.81 for monthly calibration, which is seen as good according to the classification proposed by Liew et al. (2007). Lin et al. (2015) obtained close values of NS for monthly calibration (between 0.75 and 0.82). At validation process, the NS kept an acceptable value, with an NS equal to 0.58 for monthly streamflow average. Pereira et al. (2014), also studying a river basin in Southern Brazil, found an NS value of 0.65 and 0.70 for calibration and validation period, respectively. Glavan et al. (2011) also found better results for the calibration (0.62) than for validation (0.47).

$PBIAS$ presented a good value for calibration ($<15\%$) and satisfactory for validation ($<25\%$). Similar results were found by Fukunaga et al. (2015) modeling streamflow in a watershed in the Espírito Santo State, Brazil, who obtained better results for calibration ($NS = 0.75$, $PBIAS = 11\%$) than for validation ($NS = 0.67$, $PBIAS = 22\%$).

According to Liew et al. (2007), when NS is lower than 0.36 and $PBIAS$ is higher than 25%, the model is considered unsatisfactory. Thus, our model was considered able to predict streamflow of the Sarapuí River Watershed.

Table 2. Daily, monthly and yearly streamflow calibration statistics for the Sarapuí River Basin Model, Brazil.

	R^2	NS	$PBIAS$
<i>Calibration</i>			
Day	0.67	0.65	-12.7
Month	0.85	0.81	-14.1
Year	0.82	0.71	-12.1
<i>Validation</i>			
Day	0.78	0.56	-19.1
Month	0.79	0.58	-17.6

Figure 3. Comparisons of the observed and simulated streamflow for the Sarapuí River Basin Model, Brazil: A) Daily, B) Monthly, and C) Yearly.

The comparison between observed and simulated sediment and nutrients is presented in Figure 4. Sediment and nutrients calibration showed acceptable value for *NS* and *PBIAS*. We obtained an R^2 of 0.68, 0.54 and 0.62 for TSS, TN, and TP, respectively. Considering *NS* coefficient, TSS showed a value of 0.65 while TN and TP showed values of 0.47 and 0.46. For *PBIAS*, we obtained values of 6.85, 0.18 and 24.45 for TSS, TN, and TP, respectively. Yang et al. (2016a) in China, obtained *NS* values > 0.55 and $R^2 > 0.7$ for TN and TP calibration. In another way, Ferrant et al. (2011) found that SWAT model performed poorly in simulating nitrogen loads, with an *NS* coefficient of 0.15.

Figure 4. Observed and simulated total suspended solids (sediment yield) (A), total nitrogen (TN) (B) and total phosphorus (TP) (C) for the Sarapu River Basin Model, Brazil.

In our study, the values for sediment and nutrients calibration were lower than those for streamflow, which is expected according to Abbaspour et al. (2015), but it is important to consider that the lack of data, especially in Brazil, can contribute with the difficult to calibrate water quality parameters (Bressiani et al., 2015). Another point to be considered is the fact that the model did not consider the contributions of point source pollution, that exist in the Sarapu river basin. Abbaspour et al. (2015) pointed that the model uncertainties can be related to processes occurring in the watershed, but not include in the program or processes that are

included in the program, but their occurrences in the watershed are unknown to the modeler or unaccountable because of data limitation.

4.3.2. Land-use/land-cover patterns and water quality

The results showed that the highest levels of sediment and nutrients occur in the upper half of the Sarapuí River basin (Figure 5), the region that presents the steepest slopes and many of the agricultural lands (Figure 1). Despite being at hillslope, some of the subbasins in the Upper Sarapuí basin showed low values of sediment and nutrients. Those subbasins presented a high percentage of forest cover. It is the case of sites 1, 2 and 5. S5 showed the lowest value for sediment and nutrients (in the samples collected in 2013 and 2014), and the sites 3 and 4 presented the highest value between the six experimental watersheds. The simulated data showed the same pattern, with sites 3 and 4 presenting the highest level of sediments and nutrients.

Subbasins covered by agriculture and residential areas in this study showed higher levels of sediment and nutrients than subbasins covered by forest and pasture (Figure 5). Forested subbasins showed the best values for water quality parameters. Studies show that urban and agricultural lands lead to a decreased in water quality (Ometo et al., 2000; Gergel et al., 2002; Uriarte et al., 2011; Zhou et al., 2012; Erol and Randhir, 2013; Huang et al., 2016). Lin et al. (2015) used a SWAT Model to assess the impact of bioenergy policies on the water-quality in the USA. The results showed that an increase of planting areas for corn and soybean over forest and pasture losses increases sediment and nutrient loading.

In our study, watersheds covered by forest followed by those covered by pasture showed better water quality than the watersheds with agriculture and residential areas. In the same way, Wang et al. (2013) found that forest and farmland cover types play a significant role in determining the surface water quality.

Figure 5. Spatial distribution of water quality attributes for the Sarapu River basin, Brazil.

Besides the spatial variation, we also observed a temporal variation of water quality. Rainy season showed higher nutrients and sediments levels than the dry season for the Sarapu River basin. The same pattern was observed by other studies (Uriarte et al., 2011; Gonzales-Inca et al., 2015; Ou et al., 2016). Surface runoff wash away the pollutants and soil into the

river during the rainy season, carrying out sediments and nutrients (Hasan et al., 2015). Gonzales-Inca et al. (2015) pointed out that the relationship between agricultural use and nutrient losses is strongest during high flow periods. Our model showed the highest sediment and nutrients loading between October and March (Figure 4), the rainy season in the region. The agricultural areas in the watersheds worked as sources of sediment and nutrients in the rainy season.

We observe the relationship between streamflow, sediment, and nutrients especially when we analyze the phosphorus dynamics. A significant relationship was found between TP and TSS following the variation on streamflow. This relationship demonstrates a high explanatory power (92%). Randhir and Tswetkova (2009) also observed a similar relationship, with an explanatory power of 83%. Our results from experimental conditions at the six experimental watersheds showed a similar relationship, with an explanatory power of 57% for a data collection of one year. Urban and agricultural development frequently increase nitrogen concentrations in streamflow and result in the greatest problems of phosphorus loading to water bodies. Those nutrients are transported by sediment, especially in rainfall events. Transported sediment from watersheds composed of different land uses can carry high levels of nutrients. Sediment transport of phosphorus can reduce the chemical quality of surface water and result in substantial changes in aquatic ecosystems (Brooks et al., 2003). Sediment loading represents the major problem for water quality in other tropical rivers (Hasan et al., 2015).

4.3.3. Riparian forest restoration impacts on water quality

Forest cover in the Sarapuí River basin has increased by 14% after riparian forest restoration (Table 3). Pasture and agriculture have decreased by 8% and 10%, respectively.

Table 3. Land-use/land-cover (LULC) changes after the riparian restoration simulation in the Sarapuí River basin, Brazil.

LULC classes	% current scenario	% riparian scenario	% of variation
forest	37.06	42.33	14.22
pasture	28.41	26.08	-8.20
agriculture	24.42	21.97	-10.03
eucalyptus	5.89	5.73	-2.72
residential area	3.52	3.23	-8.24
industrial area	0.37	0.33	-10.81
water	0.33	0.33	0

Figure 6 shows the annual and average loads under the current and riparian scenarios. The general trend is that the sediment and nutrients loads were greater under the riparian scenario than in the current scenario. TSS average reduced by 9.26%, TN by 22.6% and TP by 7.89%. Studies show that riparian forest helps to reduce nitrogen, nitrate, phosphorus, pesticide and sediment (Krutz et al., 2005; Oliveira et al., 2010; Batson et al., 2012; Saalfeld et al., 2012; Randhir e Ekness, 2013; Ding et al., 2013, Ou et al., 2016). Simulating the wetland restoration at the riparian zone, Yang et al. (2016b) found a decrease of 35% of sediment, 28% of TN and 37% of TP. Monteiro et al. (2016) found a sediment reduction of 29.4% by simulating the riparian restoration according to the Brazilian law in a catchment in the Cerrado.

These results show that the riparian restoration could be an efficient alternative to improve water quality against water treatment processes and wastewater treatment projects. The costs associated with water and wastewater projects are enormous and in many cases, they are not considered to be implemented in rural areas in Brazil. According to Honey-Rosés et al. (2013), the riparian forest restoration could generate economic savings for water treatment managers in the range of €57,000-€156,000 per year in a river in Spain.

Figure 6. Annual and annual average loads of (A) total suspended solids (sediment yield), (B) total nitrogen (TN) and (C) total phosphorus (TP) under two different scenarios in the Sarapuí River basin, Brazil.

According to Hanser et al. (2010), riparian zones play an important role in mediating the bi-directional flow of matter and energy between the water body and the surrounding land. Forest cover at the riparian corridor can influence stream nutrient concentrations, physical characteristics and energy balance (Tanaka et al., 2016). Ou et al. (2016) found that the first 100 m of the riparian zone has the largest effect on river water quality. FAO recommendation for water quality protection is 30 m, as well as proposed by Welsch (1991) and used in our study. In Victoria, Australia, the minimum width recommendations for riparian zones were established for different levels of land use intensities (Hanser et al., 2010). For low intensity, the recommendation to improve water quality is 30m and for high-intensity land use, the recommendation is 60m.

This is an interesting discussion because the role of the Riparian Zone can be different for different land use intensities. In our study, we followed the Brazilian Forest Code that establishes a 30m riparian zone protection that does not consider the landscape around. According to Monteiro et al. (2016), the reduction of the 30-m buffer to 5-m riparian corridor

only, as currently discussed in the São Paulo State, reduces the potential of the riparian zone in reducing sediment into the river.

It is recognized that there will always be competing social and economic issues that will ultimately influence decisions about riparian zone (Hanser et al., 2010). However, the scientific understanding of riparian zone as it relates to ecological function is important to guide the decision making. In this way, our results provide a scientific understanding of the role of riparian forest for the water quality improvement by reducing nutrients and sediments loading into the river. Future restoration plans must incorporate riparian zone as priorities areas in order to mitigate water quality problems, once our results showed that the riparian forest restoration can reduce sediment and nutrients loads into the river by 8 to 23%, respectively.

Especially in tropical regions, the natural regeneration can improve the provision of ecosystem services including water purification (Strassburg et al., 2016; Chazdon and Uriarte, 2016). Considering that many degraded areas in the tropics are composed by low-productivity or abandoned pasturelands, those areas are a great opportunity for forest restoration in the tropical agricultural landscapes (Latawiec et al., 2015). According to Strassburg et al. (2016) the natural regeneration of abandoned pasturelands would reduce the costs of purifying the water. In the Sarapuí River watershed, pasture is the second predominant land use, representing 28.4% of the watershed. Better cattle ranching techniques could create new areas for forest restoration in this area.

In this way, riparian restoration can be a good alternative to improve water quality in tropical watersheds where land use is predominantly for agricultural activities, but it cannot replace forest conservation plans for the entire watershed, once forested watersheds showed better water quality than those covered by other LULC types in this study. Our results further demonstrate the importance of stopping forest degradation and the establishment of forest restoration plans by prioritization of riparian restoration to improve water quality. It is important to consider sustainable agricultural practices combined with forest conservation plans to provide current and future water supply.

4.4. Conclusions

We propose a modeling approach to evaluate the impacts of riparian forest restoration on water quality in agricultural watersheds in Brazil using a watershed simulation model (SWAT).

The Sarapuí River Basin Model is satisfactory to simulate streamflow and water quality parameters. Water quality has a spatial and temporal variation correlated to LULC patterns and precipitation events. There is a high correlation between sediment and phosphorus loading following streamflow variation. Sediment and nutrients increase in high streamflow periods. Forested subbasins show better water quality than subbasins covered by anthropogenic uses. Subbasins covered by agricultural and residential lands have a high contribution to sediment and nutrients loading into the river and this relation is strongest in the rainy season. The riparian restoration, with the increase of forest cover and a decrease of agriculture and pasture, contributes to the decrease of sediment, nitrogen and phosphorus loading.

Future conservation plans in agricultural watersheds can use our results to locate areas for restoration in order to improve water quality. Riparian restoration can be a good alternative to improve water quality in watersheds where land use is predominantly related to agricultural activities, but a forest conservation plan for the entire watershed cannot be discarded. The management practices must have a watershed perspective not only a local focus on properties or towns. Future management plans and environment policies should consider the landscape planning at the whole watershed, indicating the riparian restoration as the first step to water quality improvement, especially in regions where available areas for forest restoration are scarce.

The model provides satisfactory results for water parameters simulation and enables to create future scenarios, but the lack of available data can be a challenge to apply it in some regions. Future studies about land-use change and the impacts on water quality must be conducted in tropical regions where agricultural activities are in expansion. The contribution of the point source pollution must be considered in future studies about watershed management. Riparian zone shows a high importance for water quality, and different vegetation types, the degree of forest conservation and buffer width can be tested for riparian restoration in future researches.

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5. FINAL CONSIDERATIONS

Throughout the chapters, this study has showed the importance of forest cover maintenance to the water conservation, in tropical agricultural watersheds, once it provides sediment, nutrients and other pollutants retention to the water bodies, whereas agricultural and urban lands are the major land uses responsible for water quality degradation.

The soil constantly exposed, present in short cycle crops, as in our study area, are heavily subject to erosion, increasing runoff in storm episodes, especially in the rainy season in tropical areas. The sediment from these areas carries nutrients and other pollutants into the river, mainly total phosphorus (TP), which is easily adsorbed by the soil particles. Thus, the land-use impacts on water quality are controlled by the hydrologic conditions and, in this study, the watersheds that presented the greatest agricultural area showed the highest pollutant concentrations and the greatest temporal variation because of this process. On the other hand, even with agricultural areas, watersheds with high percentage of forest cover present better water quality than watersheds mostly covered by other land uses, which demonstrates that forest cover contributes to minimizing the negative impacts from the agricultural activities.

The urban areas, represented in this study by small residential areas scattered within the agricultural lands or by small neighborhoods of the town, which do not have sewage collection, have a strong influence on the river water quality despite their low percentage in the landscape, regarding the production of organic matter (organic suspended solids – OSS), nutrients (TP and TN), and fecal coliforms (FC) and decreasing dissolved oxygen (DO) concentrations. Besides, the isolated houses present in the agricultural areas also seem to lead water quality degradation, especially in the FC production.

In contrast to agriculture and urban areas, in our study area, where the livestock activities are not intensive, pasture (or abandoned pasture) does not generate water quality degradation, in general, showing that the grassland cover contributes to minimizing runoff and, consequently, the pollutants loading into the river. However, forested areas have a higher potential of pollutant retention.

The water quality parameters have different responses regarding the landscape composition within the watershed and the riparian zone, which requires a multi-scale perspective of the decision-makers regarding the water resources management. Nonetheless, the land-use and land-cover pattern within the watershed exerts greater influence on water quality than considering only the riparian zone, which emphasizes the importance of the

watershed management in an effort to order the land use and occupation guaranteeing the ecosystem services provision in keeping water clean.

Nevertheless, the riparian restoration can reduce by up 23% the sediment and nutrients loading into the river, which shows off the advantage in keeping the forest cover along the buffer zone surrounding the rivers. Therefore, the riparian restoration can be prioritized as a short-term action aiming at the water quality improvement, especially in watersheds mostly covered by agriculture and in regions where available areas for forest restoration are scarce. However, to ensure water with good quality downstream, proper management need to be implemented on the low-order streams.

Forest restoration and conservation plans, better soil management practices from agricultural activities and sewage collection need to be adopted for the entire watershed aiming at the water quality improvement. In this regard, this study can contribute to planning future actions and policies design of forest and water resources management and conservation. It can also contribute to future studies on water management strategies and negative impacts of deforestation and forest degradation.

The crescent demand for drinking water requires strategies of long-term water quality maintenance, which involves a holistic view of the ecosystem, and the understanding of the processes in the watershed that are a mix of natural and anthropic factors. The balance between environmental conservation and economic expansion is needed to the sustainability of human activities and the other species survival that inhabit the same planet. To achieve this goal, science and policy need to walk side by side, and researches like the present study are important to bring science closer to critical environmental, social and economic issues that we are currently facing.