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Native forest cover safeguards stream water quality under a changing climate

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Abstract. Ensuring a sufficient and adequate supply of water for humans and ecosystems is a pressing environmental challenge. The expansion of agricultural and urban lands has jeopardized watershed ecosystem services and a changing climate poses additional risks for regional water supply. We used stream water quality data collected between 2000 and 2014, coupled with detailed precipitation and land cover information, to investigate the effects of landscape composition and short-term precipitation variability on the quality of water resources in the state of São Paulo, Brazil. The state is home to over 45 million people and has a long history of human landscape modification. A severe drought in 2014–2015 led to a major water crisis and highlighted the fragility of the regional water supply system. We found that human-dominated watersheds had lower overall water quality when compared to forested watersheds, with urban cover showing the most detrimental impacts on water quality. Forest cover was associated with a better overall water quality across the studied watersheds, with forested watersheds having low turbidity and high dissolved oxygen. High precipitation led to increased turbidity and fecal coliforms levels and lower dissolved oxygen in streams but these effects depended on watershed land cover. High precipitation diluted concentrations of nitrogen and dissolved solids in highly urbanized watersheds but exacerbated turbidity in pasture-dominated watersheds. Given the high costs of water treatment in densely populated regions, there is a pressing need to plan and manage landscapes in order to ensure adequate water resources. In tropical regions, maintaining or restoring native vegetation cover is a promising intervention to sustain adequate water quality.

Key words: forest restoration; land use cover changes; landscape management; precipitation; water provision; watershed ecosystem services.

Introduction

Fresh water is a crucial resource for human well-being and securing a sufficient supply of water to meet both human and ecosystem needs has become a major challenge to modern societies (Grimm et al. 2008, Vörösmarty et al. 2016). Water provision comprises both water quality and quantity, and human activities place pressure on both. Overconsumption from domestic, agricultural, and industrial use directly decreases water quantity and these activities are also generally detrimental to water quality (Gergel et al. 2002, Allan 2004, Brauman et al. 2007). Water quality is a key component of water security, especially under a changing climate, when the vulnerability of water provision can be further aggravated by long dry periods (Grimm et al. 2008). Consequently, land managers must develop plans to minimize the pervasive impacts of human activities on water resources while also ensuring water provision under future climates.

The conservation of native forests in human-modified watersheds may offer a viable and efficient nature-based solution to protect or improve water quality (Gregory et al. 1991, Lowrance et al. 1997, Allan 2004, Keesstra et al. 2018). Tree cover stabilizes the soil (Allan 2004, Pollen and Simon 2005) and reduces runoff of nutrients, sediments and debris to streams (Guilloy et al. 2000). In contrast, intensive human land uses, including urban areas and agricultural lands, are usually detrimental to water quality, and changes in land cover have been recognized as the leading cause of water quality

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degradation (Mello et al. 2020). Widespread fertilizer addition to crop fields result in nutrient runoff in agricultural lands, which can lead to eutrophication of water bodies (Schröder et al. 2004, Taniwaki et al. 2017). Erosion and sediment runoff are also major problems in agricultural lands, especially during high rainfall events (Ferreira et al. 2012, Souza et al. 2013). Even plantation forestry can have detrimental effects on water quality due to poor management practices and fertilizer use (Ferreira et al. 2012, Souza et al. 2013). Similarly, sewage disposal and increased runoff are the main sources of water pollution in highly urbanized areas and a great concern for city managers worldwide (Paul and Meyer 2001, Walsh et al. 2005).

A changing climate adds further urgency to watershed management issues (Pahl-Wostl 2007). Future climate change scenarios predict increasing variability in rainfall regimes in much of the world, including extended droughts and extreme rainfall events, putting in jeopardy future water provision (Feng et al. 2013, Mora et al. 2013, Greve et al. 2014). Understanding the joint effects that rainfall and land cover composition have on water resources is critical to improve watershed management practices. This is particularly important considering that only a handful of studies have explicitly investigated the effects that these interactions have on water resources (e.g., Uriarte et al. 2011).

In this study, we explore the consequences of landscape composition and short-term precipitation variability on water quality in the state of São Paulo in southeastern Brazil, a highly developed and densely populated tropical region. We use a comprehensive water quality data set including over 230 watersheds and 15 yr, coupled with detailed land use cover and precipitation data, to explore these relationships at both the watershed and riparian buffer scales. We also investigate how the interaction between these variables affect water resources in the region, as rainfall can either exacerbate or alleviate the effects of different land covers on water quality (Uriarte et al. 2011).

The state of São Paulo is an ideal region to explore the relationships between landscape management, climate, and watershed ecosystem services provision. The frequency and intensity of extreme rainfall events has increased in this region over the past few decades (Cetesb, Companhia Ambiental do Estado de São Paulo 2012) and regional climate models also predict an increase in the frequency of these events (Marengo et al. 2009). A major water crisis in the region during a severe drought in 2014–2015 exposed the precariousness of the water supply system in the region (Nobre et al. 2016, Soriano et al. 2016). A number of mitigation actions were implemented, including the adoption of a forest restoration initiative to improve water quality in heavily polluted watersheds (Programa Nascentes 2014). In light of these policies, our study presents a practical framework for integrating scientific research into land use decisions and public policy.

Methods

Study area

Our study area includes the south, central, and eastern portions of the state of São Paulo, Brazil (Fig. 1), a densely populated landscape with intense agricultural and industrial activity. The region also has the largest network of water monitoring stations and availability of water quality data within the state, which was an important criterion for the selection of our study area. Covering just about 3% of Brazil, the state is home for 42 million people or 22% of the country`s population, including the São Paulo Metropolitan Area, with a population of 21 million. It is also the most economically developed region in the country, responsible for over 30% of the Brazilian GDP. Agricultural and industrial activities account for almost 80% of the water use in the state, with the remaining 20% allocated for household use (Dufek and Ambrizzi 2008). The state underwent a long history of land use modification since the 1500s (Lira et al. 2012, Joly et al. 2014), with strong industrialization and urbanization processes occurring in the early 20th century. The once-dominant forest cover has been drastically reduced and fragmented (Ribeiro et al. 2009, Lira et al. 2012). Climate in our study area is predominantly tropical and sub-tropical, with average annual temperatures varying between 15°C and 22°C. Temperatures can reach above 30°C during the hot summer months and winters are relatively mild, with minimum temperatures ranging between 5°C and 13°C. Rainfall varies considerably throughout the year in the region, with a well-defined summer rainy season between December and March, with average monthly precipitation above 200 mm, and dry winter with averages below 50 mm (INMET, Instituto Nacional de Metereologia 2020). Annual rainfall ranges from 1,000 to almost 4,000 mm (Appendix S1: Fig. S1).

Water quality and precipitation data

Our study used data from 229 water quality sampling stations monitored by state agencies. These stations represent the range of land cover and land uses in the state (Fig. 1). Water quality data was obtained from the São Paulo State Environmental Agency (CETESB). Samples were collected once every two months from January 2000 through December 2014, covering some of the driest months of the 2014–2015 drought (January–March 2014). For this analysis, we selected six commonly used metrics (Mello et al. 2018*b*): dissolved oxygen (mg/L), a measure of water aeration and photosynthetic activity; total nitrogen (mg/L); total phosphorus (mg/L); turbidity (NTUs), a metric that captures soil runoff, fine organic matter inputs, and in-stream production; total dissolved solids (mg/L); and number of thermotolerant fecal coliforms (units/mL). The number of sampling stations with data for each individual metric ranged from 191 (total dissolved solids) to 223 (dissolved oxygen)

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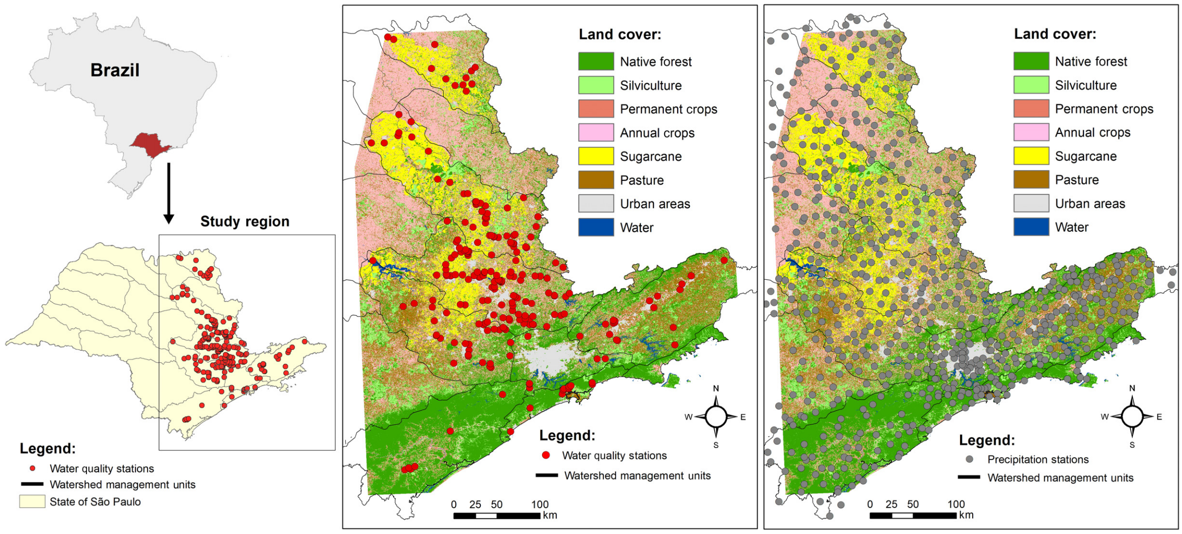

FIG. 1. Location of the study region, selected water monitoring stations, precipitation stations, and land use cover in the state of São Paulo.

and the number of samples varied from 8,276 (total dissolved solids) to 13,166 (turbidity; Table 1). Methods for determining the water quality parameters and ranges for each metric are listed on the Appendix S1: Table S1. Precipitation data was obtained from the São Paulo State Water and Electricity Department (DAEE). To investigate the effects of short-term precipitation on water quality metrics, we collected daily precipitation data from a network of 349 stations located within the study area (Fig. 1) and matched each water quality station to the closest precipitation station (average distance of 4.35 ± 0.12 km [mean ± SD]). Annual precipitation within our studied watersheds was highly variable, ranging from 1,200 mm to over 2,800 mm, with a mean of 1,500 mm (Appendix S1: Fig. S1).

Watershed delineation

We delineated the drainage area for each water sampling station based on 1:50,000 sub-watersheds maps (São Paulo, Secretaria de Infraestrutura e Meio Ambiente 2013) and adjusted the boundaries through visual interpretation of the digital elevation model ASTER GDEM 2 images (NASA 2009). Watershed size for the water quality sampling stations ranged from 0.4 to 32,667 km^2 (mean = 2,141; Appendix S1: Table S2).

Land use cover data

We generated land use cover maps by using supervised classification of Landsat-TM images with 30-m spatial resolution for the years 2003, 2006, and 2011, which represent dates close to the start, mid and end points of our study period (2000–2014). This was done to determine if significant changes in land cover occurred in the studied watersheds between 2000 and 2014. We also selected these dates due to field data availability for validation of the land cover maps. Our land use cover maps contained the following classes: (1) native forest; (2) silviculture (mainly *Eucalyptus*); (3) perennial crops; (4) sugarcane;

TABLE 1. Number of monitoring stations, number of samples, probability distribution, model comparison results and R^2 calculations for most parsimonious model for each water quality metric.

				DIC			R^2		
Water quality metric	No. stations	No. samples	Probability distribution	Watershed model	Buffer model	Best model scale	Fixed effects	Random effects	Full model
Dissolved oxygen (mg/L)	223	12,269	normal	39,782	**39,770**	buffer	0.07	0.51	0.58
Total nitrogen (mg/L)	212	11,705	log-normal	**11,694**	11,702	watershed	0.20	0.47	0.67
Total phosphorus (mg/L)	210	11,649	log-normal	−3,170	**−3,185**	buffer	0.15	0.41	0.56
Turbidity (NTUs)	218	13,166	log-normal	30,058	**30,056**	buffer	0.24	0.24	0.48
Total solids (mg/L)	191	8,276	gamma	**40,643**	40,654	watershed	0.10	0.48	0.58
Fecal coliforms (units/mL)	202	9,369	gamma	**10,979**	10,990	watershed	0.21	0.47	0.68

Notes: Water quality samples were collected once every two months from January of 2000 to December of 2014. Boldface type indicates best scale model for each metric.

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(5) annual crops; (6) pasture; (7) water; and (8) urban areas (Fig. 1). For more information on the mapping methodology, see Supplementary Methods (Appendix S1: Section S1). We matched each water quality sample with the land cover map with closest date to extract the land cover data for our statistical analysis. We observed minimal changes in landscape composition within the study watersheds between 2003 and 2011, although some watersheds experienced small changes in sugarcane cover throughout the study period (Appendix S1: Table S2). Land cover can influence stream conditions at multiple scales (Allan 2004, Uriarte et al. 2011, Mello et al. 2018*a*). To account for potential effects of scale, we calculated the percentage of every land use cover class within the whole catchment area for each watershed and also along a 60-m riparian buffer, which is protected under the Brazilian environmental legislation (Soares-filho et al. 2014, Brancalion et al. 2016), allowing us to conduct our analysis at two different spatial scales (Fig. 2).

Statistical analyses

We used generalized linear mixed models to determine the effects of landscape composition and short-term precipitation variability on water quality. We ran separate models with each of the six water quality metrics (sampled once every two months) as response variables (Table 1). All models included the percentage of different land use cover classes and short-term (2-d) antecedent precipitation as fixed covariates. We checked for collinearity between predictor variables using pairwise linear correlation (Pearson's *r*) and highly correlated predictors (>60%) were not included in the same model (Appendix S1: Fig. S2). Post-hoc multicollinearity analysis using VIF calculation was also performed and all results were below a threshold of 3 (Zuur et al. 2010) (Appendix S1: Table S5). Final models included five land cover categories: urban, pasture, sugarcane, silviculture and forest. To investigate the effect of short-term precipitation on water quality metrics, we included antecedent cumulative precipitation as a predictor variable in our models. We calculated short-term antecedent precipitation for each sample for periods of 2, 4, and 7 d and preliminary analyses showed water quality metrics had the strongest response to 2 d of cumulative precipitation, which was selected for the final models, mirroring previous studies in smaller catchments (Mello et al. 2018*a*). We also included watershed area as a predictor variable to the models. However, it was not a significant predictor of water quality metrics ($P > 0.05$) and was excluded from final models. Finally, we examined the interactive effects of landscape composition and precipitation on water quality by adding interaction terms of significant predictors in our models. To account for the stream network structure of our watersheds, river and sampling station were included as random effects, with sampling stations nested within river. Because the effects of land use cover on water quality can be scale dependent (Uriarte et al. 2011, Molina et al. 2017, Mello et al. 2018*a*), we conducted our statistical analysis at two distinct spatial scales, the entire watershed and the 60-m riparian buffer scales (Fig. 2).

All response variables were either normally distributed or were normalized using logarithmic transformation (Table 1). Models were fitted using the lme4 package in R statistical software (R Development Core Team 2008, Bates et al. 2015). To determine which scale, watershed or riparian buffer, better predicted water quality, models were compared using DIC (Deviance Information Criterion). To facilitate interpretation, all predictor variables were standardized by *z*-score transformation (Gelman and Hill 2007). Significant predictors, including interaction terms, had $P < 0.05$. Goodness of fit was determined using the marginal and conditional R^2 calculated with the function r.squaredGLMM of the package MuMIN (Barton 2019). Finally, we also used ANOVA analysis to investigate the effects of the drought on water quality metrics by comparing the values for each metric in 2014 in relation to all other years in our study period.

Results

Our models showed that landscape composition and precipitation have marked effects on water quality, with distinct effects across metrics. Total nitrogen, dissolved solids, and fecal coliforms were better predicted at the watersheds scale (models with lower DICs), while dissolved oxygen, total phosphorus, and turbidity responded more strongly to landscape composition in the riparian buffers (Table 1). Our final models accounted for 48–68% of the variance in water quality metrics. Fixed effect covariates alone (land use cover classes and antecedent precipitation) explained 7–24% of the variance in water quality, while site specific random effects accounted for 24–51% of the variance. Results from the models including the standardized coefficient estimates, *P* values, DIC, and R^2 are summarized in Tables 1 and 2 and Appendix S1: Table S5.

Stream turbidity was better predicted at the riparian buffer scale (lower DIC; Table 1). Not surprisingly, high rainfall increased turbidity levels across all watersheds (Fig. 3). Landscape composition also influenced turbidity levels: turbidity was lower in watersheds with forested riparian buffers and higher in streams surrounded by pasture. The remaining classes were not significant predictors of stream turbidity ($P > 0.05$).

Dissolved oxygen (DO) concentrations in stream water also had a stronger response to landscape composition within the 60-m riparian buffer (lower DIC; Table 1). Dissolved oxygen was lower at higher precipitation and in watersheds draining urban areas (Fig. 3). On the other hand, DO was higher in areas where riparian buffers were dominated by pasture, sugarcane fields or native forest cover, with native forests having the strongest positive effect on DO.

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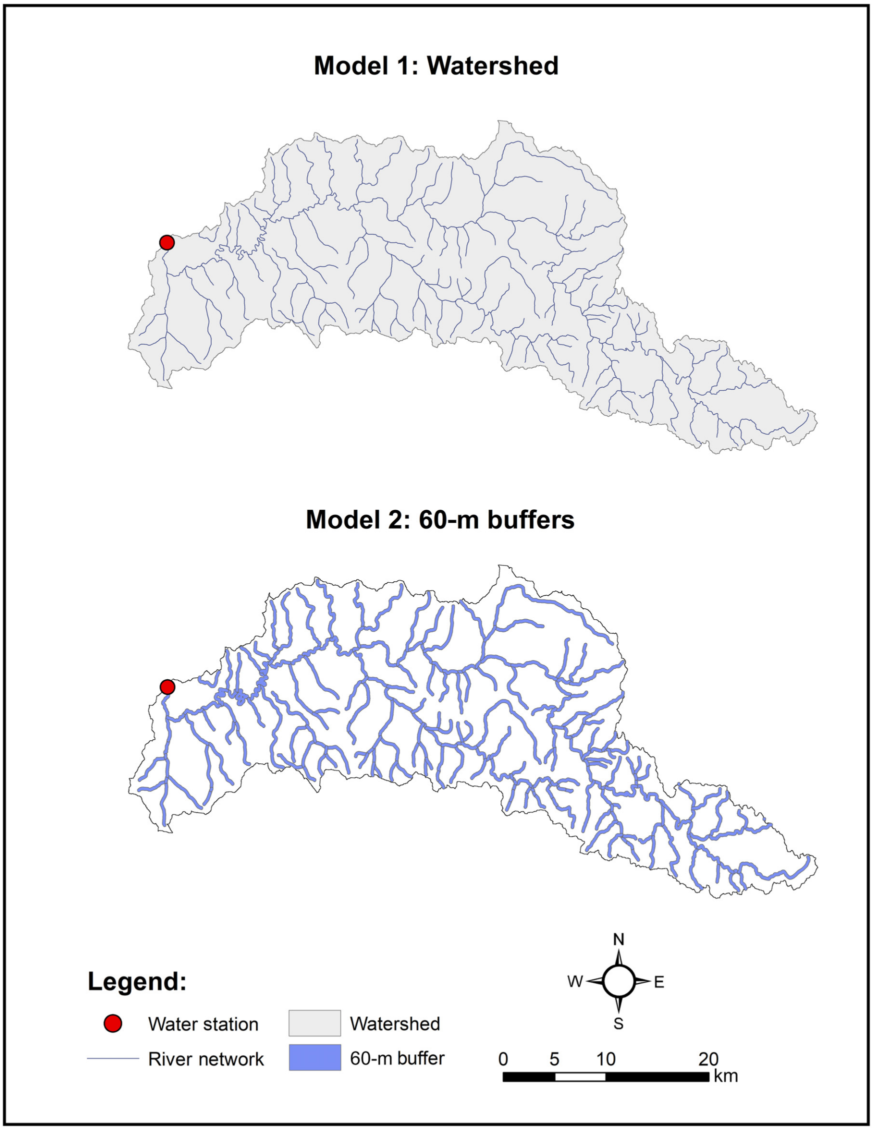

Fig. 2. Schematic illustrating the two spatial scales used in the statistical analysis.

Table 2. Parameter estimates and P values for all the significant interactions between precipitation and land cover for water quality metrics.

Water quality metric	Best model scale	Interaction	Coefficient estimates	P
Dissolved oxygen	riparian buffer	precipitation × forest	0.04 (0.01)	0.004
Dissolved oxygen	riparian buffer	precipitation × urban	0.14 (0.01)	<0.001
Total nitrogen	watershed	precipitation × urban	−0.12 (0.006)	<0.001
Turbidity	riparian buffer	precipitation × forest	−0.09 (0.01)	0.001
Turbidity	riparian buffer	precipitation × pasture	0.09 (0.01)	<0.001
Total solids	watershed	precipitation × urban	−0.06 (0.006)	<0.001

Note: Standard errors are provided in parentheses.

Urban land use cover was the most important variable explaining total stream phosphorus (P) concentrations, with higher P in watersheds dominated by urban cover (Fig. 3). Watersheds dominated by native forest, sugarcane or forest plantations had lower total P concentrations when compared to highly urban ones. Precipitation had no influence on total P ($P > 0.05$).

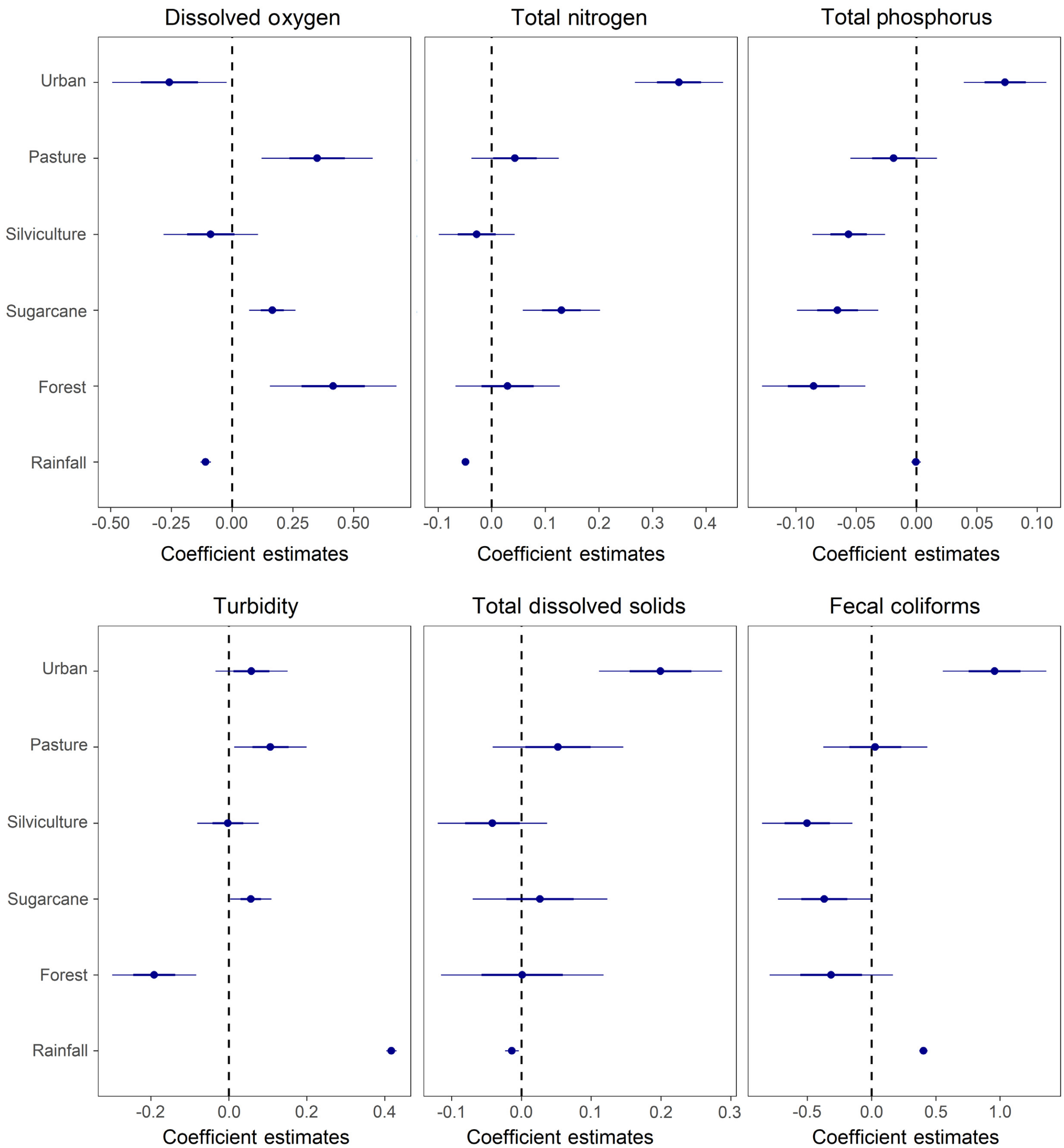

FIG. 3. Standardized parameter estimates (±2 SE) for effects of landscape composition and precipitation covariates on water quality. Results are relative to the best model/scale for each metric. Note difference in *x*-axis ranges.

The remaining water quality metrics (total nitrogen, dissolved solids and fecal coliforms) were all better predicted at the watershed scale (lower DIC; Table 1). Total nitrogen concentrations in stream water increased dramatically in urban dominated landscapes (Fig. 3). High percentages of sugarcane plantations within the watersheds also led to elevated total N levels in streams while pasture, native forest and silviculture were not significant predictors for stream N. Precipitation decreased stream total N concentrations.

Dissolved solids concentrations in streams were lower during periods of high precipitation (Fig. 3). Urban cover was the only significant land use predictor of this metric with greater concentrations of dissolved solids in heavily urbanized watersheds ($P < 0.001$). Fecal coliform counts were also significantly higher ($P < 0.05$) in watersheds draining urban landscapes (Fig. 3). In contrast to dissolved solids, however, fecal coliforms were elevated during periods of high precipitation across all watersheds. Watersheds dominated by sugarcane and

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eucalyptus plantations had low fecal coliform counts, while pasture and native forest cover were not significant predictors of this metric ($P > 0.05$).

Precipitation and land use cover interact to affect water quality. In watersheds dominated by urban cover, high rainfall resulted in twofold decrease in dissolved oxygen concentration relative to streams draining watersheds with little urban area, while the decrease in DO was almost fourfold at low precipitation levels (Table 2; Fig. 4). Similarly, the effects of native forest cover on DO was far more pronounced at low precipitation levels than during high rainfall conditions. We also found a significant interaction between precipitation and urban cover on total N ($P < 0.001$), where higher levels of N in urban watersheds occurred at lower precipitation levels, suggesting that precipitation has a diluting effect on the concentrations of these nutrients in watersheds with high urban cover (Table 2; Fig. 5). Precipitation also had a slight dilution effect on total dissolved solids in heavily urbanized watersheds (Table 2; Fig. 5). In addition, turbidity in watersheds dominated by pasturelands was exacerbated during high rainfall events (Table 2; Fig. 6). On the other hand, the effect of rainfall on turbidity was weaker in watersheds with high native forest cover.

Finally, the results from the ANOVA analysis suggest that the drought in 2014 had a significant negative impact on most of the water quality metrics (Appendix S1: Table S4). Concentrations of total nitrogen and phosphorus were significantly higher during the drought year (2014) relative to the rest of the study period ($P < 0.001$). Dissolved solids concentrations were also higher during the drought ($P = 0.037$). Stream turbidity, on the other hand, was significantly lower in 2014 ($P < 0.001$). We did not find a significant difference in dissolved oxygen between the drought and non-drought years ($P = 0.48$). There was no data available on fecal coliforms for the year of 2014.

Discussion

Our analyses of 15 yr of data in the state of São Paulo identified clear associations between water quality and landscape composition and short-term precipitation. Although site specific conditions were important drivers of water quality in our analysis, our landscape approach allowed us to identify significant and generalizable patterns over a large number of watersheds that could be applicable to watershed management and policy making in other highly developed regions. Nonetheless, we highlight that other important factors such as soil type, topography, agricultural management practices, and point source pollution (Paula et al. 2018) are also important determinants of water quality and should be considered in watershed management decisions (Paula et al. 2018, Mello et al. 2020).

Our results mirror those from previous studies conducted at local and regional scales using smaller catchments and shorter periods, synthesized by Mello et al. (2020). In general, anthropogenic land uses had a negative impact on water quality when compared to native forest cover. However, the effects of landscape composition on water quality were highly idiosyncratic for each metric. Our analysis showed that urban areas were, by far, the most detrimental land use for water quality, negatively affecting five of the six water quality metrics. Although urban areas usually comprise only a small portion of the entire catchment area, they exert a disproportionately large influence on water quality (Mello et al. 2020). Poor water quality in urban streams can be mainly attributed to domestic and industrial discharge, two major sources of pollution in cities worldwide (Paul and Meyer 2001, Allan 2004, Walsh et al. 2005). In our study, streams in highly urbanized watersheds were characterized by having low dissolved oxygen and high concentration of nutrients, dissolved solids, and fecal

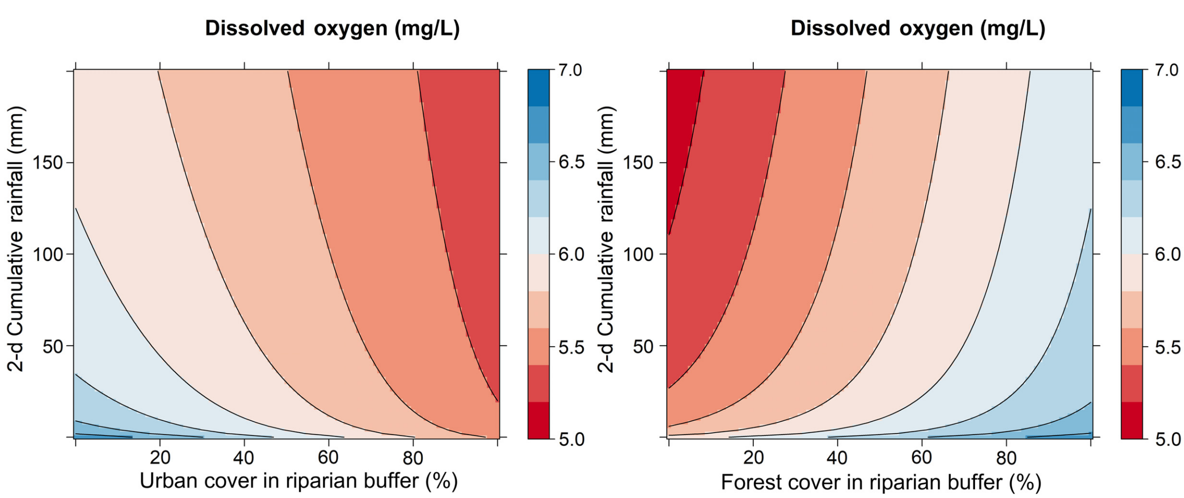

Fig. 4. Predicted effects of urban and forest cover interacting with precipitation on dissolved oxygen. Data shown are predicted values for dissolved oxygen obtained from the model, at a range of observed precipitation and land cover values at the study region.

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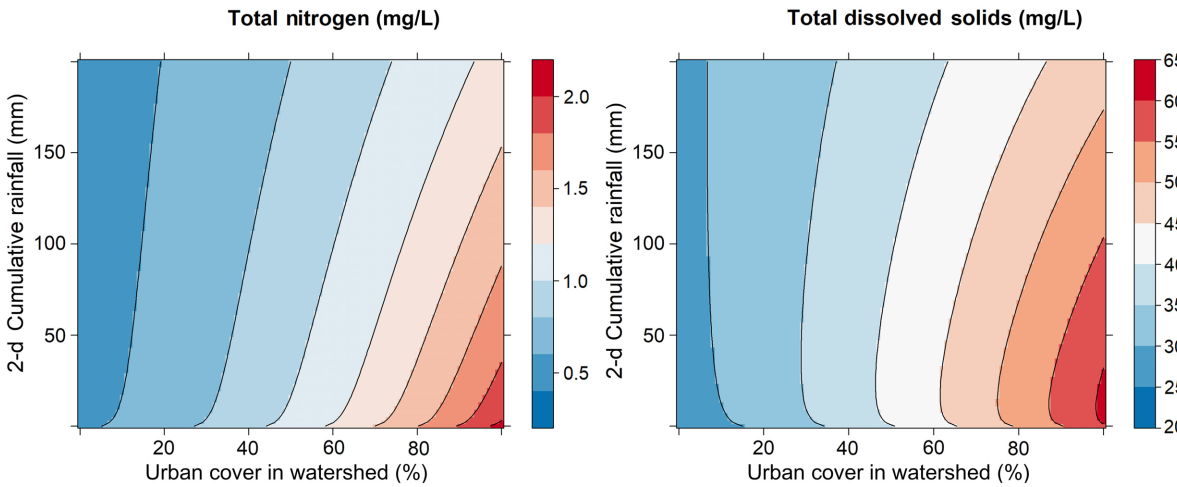

FIG. 5. Predicted effects of urban cover interacting with precipitation on total nitrogen (left) and total dissolved solids (right). Data shown are predicted values for both metrics obtained from the model, at a range of observed precipitation and land cover values at the study region.

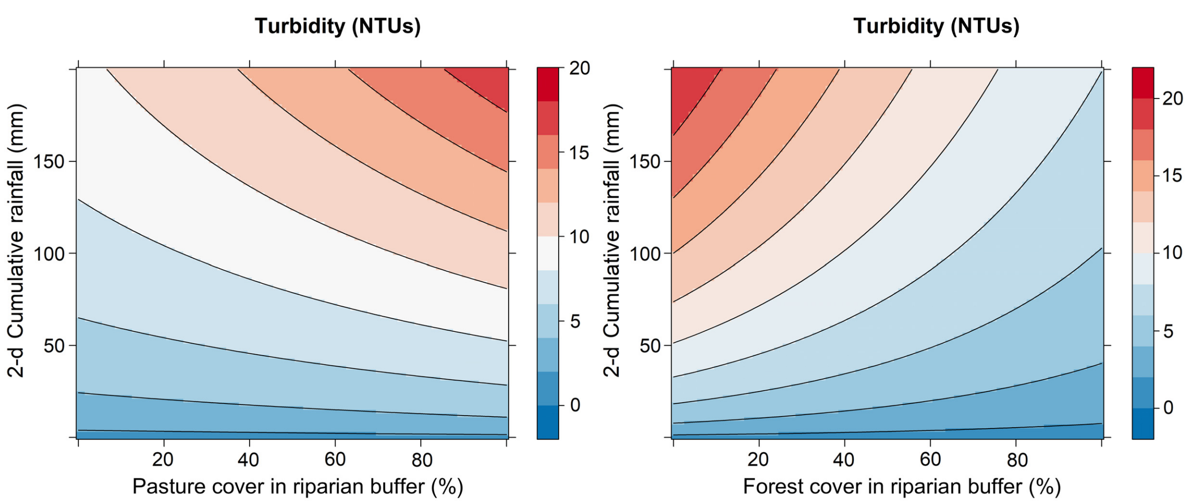

FIG. 6. Predicted effects of pasture and forest cover interacting with precipitation on turbidity. Data shown are predicted values for turbidity obtained from the model, at a range of observed precipitation and land cover values at the study region.

coliforms, all common pollutants in urban streams (Cunha et al. 2011, 2016, Uriarte et al. 2011, Mello et al. 2018*a*). Our findings further reinforce previous findings in the state of São Paulo, suggesting that insufficient and/or inadequate sewage treatment and inappropriate wastewater disposal may be largely responsible for the poor quality of surface waters in urban catchments (Martinelli et al. 2002, Cunha et al. 2011, Mello et al. 2020).

Native forest cover was associated with better overall water quality across watersheds mirroring previous studies conducted in the region (Mello et al. 2018*a*, *b*). The presence of native forests in riparian buffers proved to be particularly important for water quality, as watersheds with high riparian forest cover had greater dissolved oxygen concentrations and lower turbidity and total phosphorus levels, the three metrics that were better predicted at the buffer scale. In fact, riparian restoration in small catchments in the region yielded positive results in the past, reducing suspended sediments, total phosphorus, and nitrogen in streams (Mello et al. 2017). These findings emphasize the importance of restoring riparian zones to improve water quality, as they not only maintain the integrity of aquatic ecosystems but can also prevent sediment influx from surface runoff and erosion (Guilloy et al. 2000, Ferreira et al. 2012, Souza et al. 2013, Tanaka et al. 2016).

The efficacy of forested riparian buffers to provide watershed ecosystem services, however, depends on a number of factors. Size, tree diversity, and canopy

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structure of forested riparian buffers are all important determinants of stream dissolved oxygen and macroinvertebrate assemblages in agricultural watersheds (Tran et al. 2010, Tanaka et al. 2016). Inadequate soil conservation practices can severely compromise the ability of forested riparian buffers to prevent erosion and stream sedimentation (Tanaka et al. 2016, Guidotti et al. 2020) so that better management practices in agricultural lands are key in supporting the effectiveness of watershed restoration initiatives. Finally, native forests in agricultural landscapes are commonly exposed to different types of disturbance, including fragmentation and degradation (Hosonuma et al. 2012), which can reduce their eco-hydrological functioning (Ferraz et al. 2014). Although we did not assess the conservation status of native forest cover in our analysis, we emphasize that it should also be considered in restoration programs aiming to improve water quality.

Nonnative tree cover, namely eucalyptus plantations, did not have significant negative impacts on water quality. This is not surprising since sediment export from planted forests is common only during harvesting periods (Van Dijk and Keenan 2007) or in landscapes where roads are built to facilitate timber extraction (Rodrigues et al. 2019). A recent review of studies conducted in Brazil suggests that the lower impact of plantation forestry on water quality compared to other crops could mostly be attributed to improved management practices (Mello et al. 2020). Furthermore, eucalyptus plantations only accounted for a modest percentage of the total watershed area in our study region (Appendix S1: Table S2) and the management of these plantations have little effect on water quality, especially if these are not located within or near riparian areas (Binkley et al. 1999, Rodrigues et al. 2019).

Agriculture is a major cause of water quality degradation in Brazil (Mello et al. 2020), and our results demonstrate that agricultural land cover has detrimental effects on water quality at a landscape scale. Watersheds dominated by pastures had high turbidity, likely the result of exposed soil and erosion, leading to higher sediment inputs to streams (Mori et al. 2015, Tanaka et al. 2016). Our landscape approach also showed that sugarcane plantations are significant sources of nitrogen into streams, corroborating the results of previous local-scale studies in the region (Mori et al. 2015, Taniwaki et al. 2017). Nitrogen leaching from sugarcane processing byproducts and fertilizers are major sources of pollutants from this agricultural activity (Filoso et al. 2015), which is a very important component of the landscape in rural areas of the state of São Paulo, as the region is the main producer of sugar and biofuel (ethanol) in the country (Martinelli et al. 2013). Watersheds dominated by sugarcane, as well as eucalyptus plantations, had low fecal coliforms concentrations in streams when compared to other land cover classes, a finding that could be attributed to the absence or lower density of animals within these intensive monocultures. Our findings highlight the importance of including native forest restoration as an effective watershed management practice to increase water quality in watersheds where agricultural activity are important components of the landscape.

The effects of precipitation differed among the six water quality metrics. High rainfall was associated with low dissolved oxygen and high turbidity and fecal coliforms, which can all be linked to increased surface runoff, intensification of erosive processes, and inflow of sediments and nutrients into streams (Kistemann et al. 2002, Mano et al. 2009, Fang et al. 2012). Interestingly, while previous studies showed that precipitation can increase stream nitrogen and dissolved solids, mostly due to surface runoff and inflow of sediment and nutrients to streams, our results showed the opposite, suggesting that precipitation can potentially have a diluting effect for both metrics at the scale of our analysis (Mano et al. 2009, Fang et al. 2012).

The interaction between landscape composition and precipitation had significant effects on water quality. Increased surface runoff during high rainfall events can severely exacerbate the effects of exposed soil in pasturelands on stream turbidity, once again reinforcing the importance of native vegetation cover in stabilizing the soil, preventing erosion, and slowing down surface runoff during extreme rainfall events (Mori et al. 2015). Similarly, the negative effect of high precipitation on DO and turbidity was stronger in watersheds with low native forest cover, highlighting the importance of vegetation cover at maintaining consistent water quality under variable rainfall circumstances (Tanaka et al. 2016). Our analyses also showed that precipitation can have an diluting effect on some water quality metrics such as nitrogen and dissolved solids in heavily urbanized areas, where sewage discharge is the main driver of poor water quality (Paul and Meyer 2001, Walsh et al. 2005).

Determining the scale at which landscape composition have a stronger influence on water resources has very important implications for effective watershed management planning and decision-making processes (Uriarte et al. 2011, Molina et al. 2017, Mello et al. 2018*a*). In our analysis, dissolved oxygen, turbidity, and total phosphorus responded better to land use cover at the riparian buffer than within the entire watershed, opposite to what was reported elsewhere (Uriarte et al. 2011, Mello et al. 2018*a*). Despite these differences, our results for dissolved oxygen and turbidity were not entirely unexpected, as previous studies have suggested that both metrics have a strong response to native vegetation and other land uses within the riparian buffers, often impacted by erosion and sediment input (Mello et al. 2018*a*, 2020). Total nitrogen, however, was better predicted at the watershed scale, in contrast with other studies. This finding could be attributed to a larger percentage of sugarcane and urban area in our watersheds than in those used in similar studies (Uriarte et al. 2011, Mello et al. 2018*a*). Finally, dissolved solids and

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fecal coliforms were better predicted at the watershed scale as found by Uriarte et al. (2011) and Mello et al. (2018*a*). The inclusion of a greater number of watersheds of considerably larger size and more heterogenous land use could explain these contrasting results.

But most importantly, our multi-scale analysis reinforced the value of forested riparian buffers for water quality (Mello et al. 2020), particularly for maintaining stable levels of dissolved oxygen concentrations and low turbidity and phosphorus, conditions that foster environmental stability for aquatic organisms (Souza et al. 2013, Tanaka et al. 2016). Watershed restoration initiatives could benefit from targeting the reestablishment and improvement of vegetated riparian buffers (Guilloy et al. 2000, Mello et al. 2017, Guidotti et al. 2020), especially in places where these are protected by law like in Brazil (Soares-Filho et al. 2014, Brancalion et al. 2016). However, previous studies have shown that the minimum riparian width protected by law in Brazil (30 m) is not always sufficient to secure water quality in agricultural watersheds (Valera et al. 2019, Mello et al. 2020), highlighting the need to consider the influence of land use cover beyond the riparian zone on water quality into watershed management (Mello et al. 2020). Furthermore, the complexity and distribution of the remaining forest cover in the landscape should also be included in landscape management policies that aim to safeguard stream water quality (Mori et al. 2015, Tanaka et al. 2016, Mello et al. 2020).

Our study uncovered clear impacts of land cover composition and short-term precipitation variability on water quality in a human-dominated tropical landscape. The interactive effects of these variables on water resources call attention to the need for jointly considering these two factors when developing watershed management plans. At a landscape scale, the presence of native forest cover within watersheds and riparian buffers mitigated the effects of anthropogenic land use on water quality under variable rainfall conditions, as water quality metrics in watersheds with low forest cover responded strongly to rainfall variability. This effect achieves greater importance under unpredictable future rainfall scenarios, where guaranteeing the quality and quantity of surface waters is crucial to secure water provision for both human and ecosystem use during extended dry periods. The poor quality of surface waters was one of the main aggravating factors that contributed to the water crisis in the state of São Paulo during the 2014–2015 drought (Soriano et al. 2016). Our results also show that drought negatively affected three out of the six water quality metrics, adding further urgency for water resource management in the region in the face of ongoing climate change. Between 1950 and 2003, the frequency and intensity of extreme rainfall events increased in this region (Cetesb, Companhia Ambiental do Estado de São Paulo 2012) and regional climate models also predict a further increase in these events (Marengo et al. 2009). Therefore, watershed restoration with native vegetation, can be an efficient and cost-effective nature-based solution to secure water resources in highly developed locations where anthropogenic activities have pervasive effects on water resources, resulting in a system with higher resilience to anthropogenic or climatic changes (Lafortezza et al. 2018).

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