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Multi-scale assessment of forest cover in an agricultural landscape of Southeastern Brazil: Implications for management and conservation of stream habitat and water quality

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ABSTRACT

Forest cover has important functions for streams. Consequently, deforestation and forest degradation due to agricultural activities tend to have negative impacts on stream ecosystems. We related forest cover to stream variables, expecting to find better habitat and water quality conditions in catchments with better forest cover conditions in order to evaluate forest cover as indicator of stream health in agricultural landscapes. We sampled stream variables and quantified forest cover and physical variables in 60 small agricultural catchments in Southeast Brazil. We used redundancy and regression analysis to relate the landscape predictors to the channel responses. Percent forest cover had low to intermediate values in the spatial scales evaluated. Forest cover was fragmented and mostly located in riparian and steep slope areas. Redundancy analysis showed little influence of forest cover on the response variables, which were more influenced by catchment physical variables. Regression analysis showed that forest cover in the reach and forests located closer to the sampled reach are positively related to wood, habitat diversity, and dissolved oxygen, and negatively related to channel depth, volume, and temperature. We also found that forest cover fragmentation is negatively related to pH, potassium, water acidity, and temperature. Although many of these relationships were fairly weak, it appears that naturally regenerated forest cover is at least moderately effective in protecting streams in agricultural landscapes in the region.

1. Introduction

Headwater streams are strongly dependent on the surrounding terrestrial landscape (Richardson and Danehy, 2007), and their structure and function are regulated by complex hydrological and ecological processes operating at different spatial and temporal scales (Ward, 1989; Wiens, 2002). These ecosystems are influenced by catchment physical characteristics, such as elevation, slope, geology, and precipitation patterns (Allan and Castillo, 2007; Bisson et al., 2007), and also by forest cover (Richardson and Danehy, 2007). Forest cover influences runoff processes and regulates sediment and nutrient inputs to the streams (Allan and Castillo, 2007; Taniwaki et al., 2016). Local riparian forests are also important for ecological processes in headwater streams by controlling light incidence and primary productivity, reducing channel margin erosion, controlling the inputs of fine sediments and nutrients, and supplying food and structural resources to the aquatic fauna (Richardson and Danehy, 2007; Paula et al., 2011; Ferreira et al., 2012; Tanaka et al., 2016).

Natural forests have been extensively modified by humans since the development of agriculture, with reductions in forested area, increases in forest fragmentation, and modification of forest patch configuration (Forman, 1995). These modifications affect stream and water characteristics at different spatial scales. Local scale deforestation increases channel and bank erosion and channel lateral migration, causing complex changes to stream width and depth (Hickin, 1984; Sweeney et al., 2004; Allmendinger et al., 2005). It also increases water temperature and stream productivity (McTammany et al., 2007; Fernandes et al., 2014), decreases the amount of large wood (LW) and reduces the

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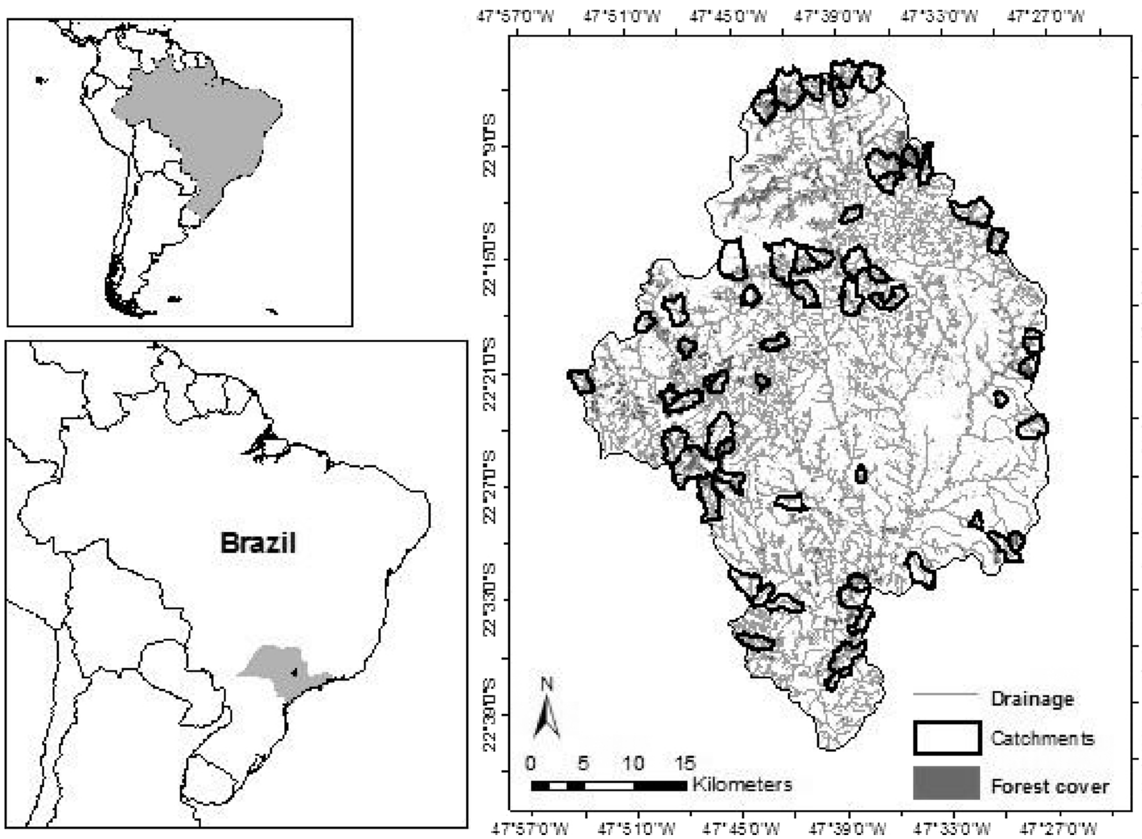

Fig. 1. Location of the 60 study sites in the Corumbataí River basin, São Paulo State, Brazil.

structural complexity of stream habitats (Paula et al., 2011; Leal et al., 2016). At broader spatial scales, forest cover alterations increase channel sedimentation and alter the amount and characteristics of channel habitat units and reduce substrate heterogeneity (Roth et al., 1996; Sutherland et al., 2002; Iwata et al., 2003; Liébault et al., 2005). Changes in forest cover at larger scales also affect hydrology and water chemistry by increasing nutrient and dissolved organic carbon concentrations and suspended sediments (Roth et al., 1996; Johnson et al., 1997; Liébault et al., 2005; Mori et al., 2015; Leal et al., 2016; Taniwaki et al., 2016).

Primary forest cover has been substantially reduced in tropical countries (FAO, 2016), whereas secondary forest cover has increased in recent decades (FAO, 2010). The literature show that secondary forests play an important role for carbon sequestration in terrestrial ecosystems (Guariguata and Ostertag, 2001; Chazdon, 2014) and retention of sediments, control of light and temperature, and input of organic material in stream ecosystems (Heartsill-Scalley and Aide, 2003; Iwata et al., 2003; Paula et al., 2011). However, these forests have reduced capacity to provide the full range of ecological processes and services provided by primary forests (Guariguata and Ostertag, 2001; Iwata et al., 2003; Chazdon, 2014). Therefore, it is important to evaluate the extent to which these regenerated and secondary forests are effective in performing their ecological functions.

Brazil has around 50–60% of its territory covered by forests (Serviço Florestal Brasileiro, 2013; FAO, 2016). The Atlantic Forest originally covered the coast and the interior regions of Brazil, but today only 11.7% of this forest remains. It is highly fragmented, composed of a few large patches of primary forest located mainly in steep slope areas and many small patches of secondary forests at different sussecional stages (Ribeiro et al., 2009). The interior Atlantic Forest covers the Corumbatai River basin, which is located in an area of intense economic development in São Paulo state. This basin bears a long and extensive history of land use conversion and intensification, with deforestation and forest degradation spanning nearly 200 years (Victor et al., 2005). The result is a landscape dominated by agricultural activities (Valente and Vettorazzi, 2003) and fragmented (Valente and Vettorazzi., 2002, 2005) and degraded forest cover (i.e., forests characterized by small diameter trees, fire exposure, understory trampling by cattle, and high density of woody vines; Rodrigues, 1999; Paula et al., 2011). Since the implementation of the Brazilian Forest Code in 1965, the 30 m riparian zones of all ≤10 m wide streams and steep slope areas (gradient over 45°) are legally excluded from agriculture and are designated as Permanent Preservation Areas (Brasil, 1965). In practice, some of these areas were never harvested (but were selectively logged), many others still are agricultural, while others have been abandoned for natural forest regeneration (Ferraz et al., 2014; Molin et al., 2017). However, most of these unharvested or regenerated forests are still subject to degradation, and their role in protecting stream ecosystems is uncertain.

In this study we asked the following question: Is the present structure of forest cover (composition and configuration) in the catchments a good indicator of stream health (i.e. does it reduce the structural and physicochemical impacts of agricultural activities on streams)? To address this question, we assessed the structure of forest cover at the landscape level and collected a suite of water quality and stream habitat metrics from catchments in the Corumbatai River basin that span a gradient of forest cover condition. We then tested the relationships between channel and landscape variables (forest cover and catchment physical variables). We hypothesized that if the current structure of forest cover is protecting the streams, we will detect relationships between indicators of forest cover and the stream water quality and habitat metrics, i.e., low suspended sediments, low sand substrate, high dissolved oxygen, and more channel habitat units and substrate diversity. Our objective was to evaluate the extent to which these regenerated, fragmented, and often degraded forest patches protect the stream ecosystem in order to promote stream conservation in agricultural landscapes. Also, identifying and understanding these complex relationships in these old established agriculture areas can help to formulate better environmental policies and management/conservation programs in new managed areas around the world that face severe threats and need urgent conservation/management planning, like the Amazon region, where agriculture, minning, and infrastructural development are fast expanding in the last decades over natural areas (Gardner et al., 2013; Ferreira et al., 2014).

2. Methods

2.1. Study area

The Corumbatai River basin is located in the east of São Paulo state, Brazil (approximately between parallels 22°04′06"S and 22°41′28"S, and meridians 47°26′23"W and 47°56′15"W) (Fig. 1) and covers 170 km^2. Elevation varies from 500 m in downstream regions to 1000 m at headwaters (Ceapla, 2010). The basin occurs on stratigraphic units from Paleozoic, Mesozoic and Cenozoic origins and is comprised predominantly of Ultisols (44%) and Oxisols (22%) (Koffler, 1993). The climate is subtropical with distinct rainy and dry seasons. Annual average temperature and precipitation is 19.5 °C and 1400 mm, respectively (Ceapla, 2010).

Pre-settlement forest cover was composed mainly of semi-deciduous seasonal forest (approximately 40% of the trees are deciduous to some degree during the dry season) and was characterized by an irregular canopy, reaching 15–20 m in height, including high emergent trees (25 m–35 m; Rodrigues, 1999). The native forest was extensively removed and replaced by coffee plantations after the intensive colonization process in São Paulo state started at the end of 18th century (Victor et al., 2005). The remaining forested areas are very fragmented and have an undefined canopy, low basal area, high density of small diameter trees, high abundance of woody vines and high tree mortality (Rodrigues, 1999; Paula et al., 2011). In the beginning of 20th century, coffee areas were replaced by sugarcane and pastures, which are now the dominant land uses (the lower basin is 25.57% sugarcane, while the upper basin is 43.68% pasture). The remaining forest areas (11%) are mostly at early to intermediate stages of succession, are highly fragmented, and are mostly restricted to hillsides and riparian areas (Rodrigues, 1999; Valente and Vettorazzi, 2002, 2003, 2005).

2.2. Sampling design and data collection

Channel and water variables were sampled at 60 catchments (1st to 4th order) by Gerhard (2005). Catchments were selected considering the dominant land use types (pastures, sugar-cane, and forest) in the whole watershed and in the drainage network. This sampling addressed a gradient of forest cover in the landscape, except for the catchment scale (highest value = 62.3%) as agriculture is the dominant matrix of land cover in the Corumbatai River basin. Catchment boundaries, drainage network and sampled reaches (150 m) were digitized using a Geographic Information System based on 1:10,000 topographical maps (IGC/SP, 1979). Field work was conducted twice in each catchment, once in the dry season (June to September) and once in the rainy season (January to March) in 2003 and 2004. Precipitation during the study period was typical for the rainy season (540 mm in 2003 and 469 mm in 2004) and dry season (55.5 mm in 2003 and 135.2 mm in 2004).

Channel and water variables (Table 1) were collected at base flow conditions in a 150 m sample reach located at the downstream end of each catchment. In each reach we counted the number of habitat units (pools, riffles and runs) and visually estimated the surface area of the habitat units and of the large wood inside the channel. For the habitat units we calculated the pool:riffle ratio (units/units and area/area). We measured channel wetted width, depth and substrate size category (on the Wentworth scale) at 38 linear transects set 4 m apart; at each transect, these cross-section measurements were taken at different intervals according to the channel wetted width (Fitzpatrick et al., 1998). We calculated channel area and volume from the average cross-section width and depth measurements (Bisson et al., 2007). We also calculated substrate size diversity using the Simpson's diversity index based on samples collected in 30 random quadrats, 50 cm on each side, from within the sample reach (Gordon et al., 2004).

We took five measures of water temperature and dissolved oxygen along the stream reach using a YSI probe (YSI, Yellow Springs, Ohio, USA) at approximately the same time in the morning of each sampling event. We collected water samples in mid-channel at the upstream end of the reach in the afternoon (after data collection was complete). Samples were stored in 1 L polyethylene bottles and kept on ice during transport to the laboratory for analysis. Water chemistry variables measured were pH, total alkalinity, ammoniacal nitrogen, total phosphorus, total potassium, total calcium, total magnesium, conductivity, total acidity, hardness, carbon dioxide, total suspended sediments, color and turbidity. The water samples were analyzed following APHA standard methods (APHA, 1975).

2.3. Forest cover mapping and landscape structure quantification

Forest cover was hand digitized by photo interpretation (color aerial photographs − year 2000, scale 1:30,000) using previous knowledge of the study area and ground-truthed in selected locations (Jensen, 2000). Studies have shown that the forest cover in agricultural landscapes of São Paulo State (including our study area) are stable or slowly increasing, with 1.7% growth reported in the period from 2000 to 2010 (Molin et al., 2017) and 2–5% in the period from 2000 to 2008 (Ferraz et al., 2014). Therefore, for the time interval from aerial imaging (2000) to field data collection (2003 and 2004) in this study, we assume that change in native forest cover was minimal. Forest cover in the study area is composed mostly of secondary forests at different successional stages. For this study, the forest cover maps are composed of secondary vegetation at initial successional stages (dominated by small diameter trees and shrubs) and intermediate-advanced successional stages (dominated by trees). Initial stages composed predominantly by shrubs-grass were excluded. Catchment elevation and slope were extracted from a digital elevation model based on 20 m interval isopleths from topographic maps (scale 1:50,000). Reach scale slope was extracted from a digital elevation model based on 5 m interval isopleths from topographic maps (scale 1:10,000). Percentage of soils in the catchment was extracted from a digital soil map (Oliveira and Prado, 1989; scale 1:100,000). Soil classes were assigned to four groups: Lithic Orthents, Orthoxic Quartzipsamments (referred in this work as sandy soils), Oxisols and Ultisols, (USA, 1999). Percent forest cover was obtained at three spatial scales: whole catchment, 30 m buffer along the drainage network and 30 m buffer along the 150 m study reach. We also calculated several "forest position" variables that integrated information on forest cover with forest location within the catchment and drainage network scales (Paula et al., 2013). To calculate these variables, we created a map of hydrological distances for each catchment, using the digital elevation model and the outlet as a starting point. These maps were divided by the maximum value of distance in each catchment to normalize the distance due to differences in catchments size. We then reclassified all pixels in each catchment into one of three categories (near, intermediate and distant) using the equal interval method in ArcGis and calculated the percent forest cover in each of these categories. Patch density was obtained by dividing the number of patches by the area of the landscape (McGarigal et al., 2002). Forest cover fragmentation was obtained by the Matheron index: total forest perimeter divided by the square root of total forest area multiplied by the square root of total landscape area (Boschetti et al., 2004). The distance of the largest forest patch in the landscape (catchment and drainage network scale) to the sampled stream reach was obtained by measuring the length of the drainage network segment that connects the largest patch to the sampled reach (this measure is important

Table 1
Descriptive statistics for stream variables.

Data set	Variable Description	Code	Rainy season				Dry season			
			Mean	ST (±)	Min.	Max.	Mean	ST (±)	Min.	Max.
Water chemistry	Total alkalinity (mg L^{-1})	Alk	29.28	33.54	4.00	212.80	38.18	53.69	3.40	251.40
	Ammoniacal nitrogen $-NH_3N$ (mg L^{-1})	NH3-N	0.36	0.25	0.10	1.30	0.27	0.15	0.20	1.20
	Total phosphorus (mg L^{-1})	P	0.07	0.09	0.02	0.56	0.04	0.08	0.01	0.63
	Total potassium (mg L^{-1})	K	1.78	1.11	0.10	6.00	2.28	1.90	0.20	8.00
	Total calcium (mg L^{-1})	Ca	6.73	9.23	0.60	48.50	7.73	11.51	0.60	64.70
	Total magnesium (mg L^{-1})	Mg	3.51	7.05	0.30	36.70	3.61	6.83	0.30	43.50
	Total suspended sediments (mg L^{-1})	TSS	115.55	404.53	4.00	2868.30	29.13	44.18	4.30	238.70
	Total acidity (mg L^{-1})	Acid	12.27	11.28	1.00	76.90	5.06	5.80	0.50	37.70
	Hardness (mg L^{-1})	Hard	31.19	51.10	2.80	271.10	34.12	55.88	2.70	340.60
	Carbon dioxide (mg L^{-1})	CO2	3.78	2.92	0.90	15.10	3.31	4.32	0.80	32.10
	Color (PtCo)	Col	160.27	400.10	1.00	3125.00	43.05	41.89	1.00	167.00
	Turbidity (FTU)	FTU	70.63	178.84	2.00	1275.00	17.97	16.06	2.00	66.00
	Conductivity (mS.cm^{-1})	Cond	0.10	0.07	0.05	0.40	0.09	0.08	0.04	0.45
	Potentiometric hydrogen ion concentration	pH	7.05	0.46	6.00	8.30	7.16	0.49	6.20	8.30
	Dissolved oxygen (mg L^{-1})	DO	7.64	0.99	3.27	8.79	8.40	1.33	2.51	10.19
	Temperature (°C)	Temp	22.45	1.66	19.16	27.64	17.50	2.57	11.82	24.52
Channel structure and morphology	Number of habitat units (pool, run, riffle)	Hab	14.33	6.29	1.00	32.00	14.25	5.67	3.00	30.00
	Pool/riffle ratio (units)	PR	0.29	0.22	0.00	1.00	0.30	0.21	0.00	0.80
	Pool/riffle ratio (m^2)	APR	0.27	0.36	0.00	2.06	0.47	0.76	0.00	4.03
	Large wood (m^2)	LW	6.16	7.89	0.00	46.75	5.00	5.19	0.00	19.70
	Channel width (m)	Wid	248.41	106.08	110.13	721.05	203.40	81.68	106.53	504.08
	Channel depth (cm)	Dep	18.94	10.84	3.21	45.98	13.81	7.72	1.80	35.61
	Channel area (m^2)	Area	372.62	159.13	165.20	1081.58	305.11	122.52	159.79	756.12
	Channel volume (m^3)	Vol	69.24	49.63	14.69	213.06	41.47	28.43	5.43	161.87
	Substrate diversity	Subs	0.61	0.22	0.13	0.90	0.64	0.16	0.24	0.85
	Sand substrate (%)	Sand	41.40	30.81	0.00	98.55	33.86	27.44	0.00	94.83

Table 2
Descriptive statistics for landscape variables.

Landscape Data	Variable Description	Code	Mean	SD (±)	Min.	Max.
Forest cover – catchment scale	Forest Percentage	FPC	23.01	15.16	0.75	62.38
	Forest Fragmentation	FFC	0.06	0.02	0.02	0.11
	Patch Density (n.100 ha)	PDC	5.09	3.18	0.49	19.14
	Forest Position/Near (%)	FNC	26.24	19.12	1.49	73.80
	Forest Position/Intermediate (%)	FIC	23.94	18.80	0.69	94.04
	Forest Position/Distant (%)	FDC	19.73	19.26	0.00	69.68
	Large Patch Distance (m)	LDC	1078.86	1087.89	0.00	4909.80
Catchment Physical characteristics	Catchment Area (ha)	AREA	367.90	208.90	73.14	1076.77
	Catchment Elevation (m)	EL	706.54	100.61	542.09	949.52
	Catchment Slope (%)	SL	6.45	2.87	0.82	14.58
	Soil Orthoxic Quartzipsamments(%)	QUA	11.97	22.33	0.00	73.44
	Soil Oxisols (%)	OXI	34.08	35.27	0.00	100.00
	Soil Ultisols (%)	ULT	34.62	33.32	0.00	100.00
	Soil Lithic Orthents (%)	LIO	19.30	20.42	0.00	92.38
Forest cover – network scale	Forest Percentage	FPN	49.36	20.37	10.66	97.65
	Forest Fragmentation	FFN	0.10	0.04	0.01	0.24
	Patch Density (n.100 ha)	PDN	29.94	15.42	3.72	80.92
	Forest Position/Near (%)	FNN	48.63	26.91	0.00	100.00
	Forest Position/Intermediate (%)	FIN	50.04	23.42	6.71	100.00
	Forest Position/Distant (%)	FDN	38.96	29.42	0.00	100.00
	Large Patch Distance (m)	LDN	953.84	1042.38	0.00	4909.80
Reach scale	Forest Percentage – FPR	FPR	45.15	36.05	0.00	100.00
	Slope (%)	SLR	2.82	3.03	0.01	12.17

because forest patches close to deforested reaches have the potential to ameliorate water quality (Harding et al., 2006; Fernandes et al., 2014), increase habitat diversity and decrease fine sediments (Jones et al., 1999)). The landscape variables and the spatial scales considered in this study are shown in Table 2. These procedures were conducted in ArcGis 9.3 (Environmental Systems Research Institute, Redlands, CA, USA).

2.4. Statistical analysis

Non-normal response and predictor variables were transformed to improve normality; predictor variables were also transformed to improve the linearity of the relationships (Quinn and Keough, 2002). The data-set of channel response variables was split into four sub-sets: channel structure in the rainy season (n = 60), channel structure in the dry season (n = 57), water chemistry in the rainy season (n = 60), and water chemistry in the dry season (n = 58). The sub-sets were analyzed separately. We first used Pearson correlation analysis to examine the correlations between forest and physical data-sets. We then used two major analytical approaches: (1) redundancy analysis (RDA) and (2) linear regression. We used RDA to quantify the unique variation of each landscape data-set (forest cover at whole catchment, catchment physical variables, forest cover at 30 m buffer in the drainage network and forest cover at 30 m buffer in the stream reach) and also the variation explained jointly by the landscape data-set on each channel response data-set (Borcard et al., 2011). We used linear regression to test the relationship between individual response variables and individual forest predictor variables, as well as combinations of one forest and one physical landscape predictor.

RDA is a form of direct gradient analysis that describes the variation between two multivariate data-sets, where a matrix of predictor variables is used to explain variation in a matrix of response variables. In RDA, the site scores from a Principal Component Analysis are regressed iteratively against one set of predictor variables, and the fitted values of the regression become new site scores (ter Braak and Prentice, 1988). We checked collinearity among predictors in each landscape data-set by computing the variance influence factor (VIF), which measures the proportion by which the variance of a regression coefficient is inflated in the presence of other predictor variables (Borcard et al., 2011). Strong collinearity (VIF ≥ 20) was found within each data-set and reduction in the number of predictors was necessary. We conducted variable selection using the forward selection method (ordistep function of vegan package) and the double stopping criterion (Borcard et al., 2011). This function selects the variables on the basis of their permutational *p* values and on AIC in case of ties. The double stopping criterion consists of the calculation of R^2_{adj} of RDAs that incorporate the variables in their order of inclusion by ordistep and then by checking when the cumulative R^2_{adj} of the global model is exceeded. When this happens, the selection stops and the last variable entered should be excluded. After forward selection, we checked the significance ($\alpha = 0.05$) of each RDA using permutation tests (1000 permutations). To avoid the probability of type I error increase with multiple tests, we corrected the *p*-values using the sequential Bonferroni correction (Borcard et al., 2011). After this correction, the results showed that only the physical data-set was significant for water variables in both seasons, while for channel data, all the data-sets were significant in both seasons. We then performed one parsimonious RDA for each channel response data-set using only the selected predictors (Borcard et al., 2011). We again conducted permutation tests on each of the parsimonious RDA models to assess the significance (at $\alpha = 0.05$) of the analysis (Borcard et al., 2011). Linear dependencies among predictors in each parsimonious model were also assessed by VIF. To assess for spatial autocorrelation in each response data-set, we computed a dissimilarity matrix (Euclidean) of the response and a dissimilarity matrix (Euclidean) of the geographic coordinates of the sites and tested the correlation between them by a Mantel test (Borcard et al., 2011). Finally, we partitioned the variance explained in each of the channel data-sets.

We then used linear regressions to relate stream channel and water responses to landscape predictors. For each stream variable, we constructed candidate models consisting of one measure of forest cover alone and a combination of one measure of forest cover and one physical variable (Appendices A to D). We limited candidate models to only two predictor variables because our main focus was on comparing the measures of forest cover alone and the association of forest cover and catchment physical characteristics. We evaluated candidate models by Akaike's Information Criterion (AIC) adjusted for small sample size (AICc; Burnham and Anderson, 2002); models with the lowest AICc are considered the most parsimonious in a candidate model set (Burnham and Anderson, 2002). We identified models with high support (ΔAICc ≤ 2) and considerably less support (ΔAICc > 2 and ≤ 4). We extended our selection criteria to models with ΔAICc > 2 and ≤ 4 in order to increase our assessment of the influence of forest cover predictors (predictors of high interest in this work), even if their influence was low (Grueber et al., 2011). For these selected models, we calculated Akaike weights (which can be interpreted as the probability that a model is the best-supported of those in the candidate set; Burnham and Anderson, 2002) and calculated standardized model averaged coefficients. We calculated associated 85% confidence intervals for the predictors included in the final set of candidate models. We also assessed residual spatial autocorrelation with a Mantel test of the dissimilarity matrix (Euclidean) of the residuals of the averaged models versus the dissimilarity matrix (Eucliden) of the geographic coordinates of the sites. The final step was to assess the importance of the selected predictors (forest cover and physical) for each response by variance partitioning. All the analysis were done in R (R Development Core Team, 2016) using the vegan and MuMIn packages.

3. Results

Descriptive statistics for all forest cover and physical landscape variables are presented on Table 2. Catchments presented low values of percent forest cover at the catchment scale and intermediate values at the drainage network and reach scales. At the catchment scale, only 5 catchments had more than 50% of their area in forests; 27 catchments were majority forest at the drainage network scale and 24 at the reach scale. Forest cover fragmentation was higher at the drainage network scale, but decreased with increasing percent forest cover at both spatial scales evaluated (PFC and FFC − $r = -0.27$; PFN and FFN − $r = -0.42$; PFN and PDN − $r = -0.48$). Percent forest cover was low for all distance categories at catchment scale (means less than 30%), whereas drainage network scale presented intermediate values for all categories (means varying from 50 to 38%). The largest forest patch was most distant from the sampled reach. Percent of soils LIO and QUA increased with catchment slope ($r = 0.49$ and $r = 0.29$, respectively). Percent of soils ULT decreased with catchment elevation ($r = -0.43$).

The results of correlation analysis between forest cover and physical variables (Table 3) showed that forest cover at the catchment scale was positively related with elevation and slope. Forest fragmentation at the catchment scale also was related to soil types in the landscape (FFC positively related with ULT and negatively related with OXI; PDC negatively related with OXI) and positively related with slope.

Redundancy analysis showed that the physical variables (QUA + EL + AREA + SL and EL + ULT + AREA + LIO for rainy and dry season, respectively) explained 16% ($p = 0.001$) and 17% ($p = 0.001$) of the variance in the water data-set in the rainy and dry season, respectively. For channel data-sets, the RDA variance components are shown in Fig. 2. The results show that much of the variation in the channel data was explained by the physical data while forest cover data at all spatial scales explained very little of the variation for the channel data-sets in either season. The Mantel tests showed no spatial autocorrelation for the channel data-sets ($r = 0.002$ for the rainy and $r = 0.007$ for the dry season) and weak spatial autocorrelation for the water data-sets ($r = 0.09$ for the rainy and $r = 0.08$ for the dry season).

Model averaging showed that the majority of stream response variables had an influence of forest predictors (Tables 4 and 5); the exceptions were Ca, Mg, Hard, Cond, Subs, and Sand in the rainy season and Alk, Ca, Mg, Hard, CO2, pH, Wid, Area, Subs and Sand in the dry

Table 3
Results of Pearson correlation analysis between forest cover and physical predictors. Values in bold indicate $r \geq 0.3$.

	EL	SL	AREA	QUA	OXI	ULT	LIO
FPC	**0.30**	**0.71**	−0.03	**0.31**	−0.06	−0.21	0.26
FFC	−0.19	0.02	−0.11	−0.04	**−0.44**	**0.45**	0.08
PDC	0.04	**0.30**	0.07	0.12	**−0.43**	**0.31**	0.26
FNC	0.23	**0.36**	−0.04	**0.34**	0.00	−0.23	0.07
FIC	**0.30**	**0.65**	0.03	0.07	0.04	−0.19	0.27
FDC	0.16	**0.71**	−0.05	**0.42**	**−0.30**	0.02	0.24
LDC	0.03	0.16	**0.34**	0.01	−0.14	0.09	0.22
FPN	0.10	0.17	−0.10	0.27	0.03	−0.15	−0.08
FFN	−0.20	**−0.32**	−0.07	−0.11	0.06	0.03	−0.11
PDN	0.05	0.16	−0.08	0.19	−0.10	−0.08	0.17
FNN	0.09	0.07	−0.10	**0.30**	0.01	−0.15	−0.06
FIN	0.19	0.29	0.00	0.20	0.02	−0.10	−0.03
FDN	−0.07	**0.34**	−0.11	0.27	**−0.34**	0.16	0.03
LDN	0.06	0.05	**0.30**	−0.12	−0.01	0.07	0.09
FPR	0.02	0.08	−0.17	0.29	−0.01	−0.15	−0.04

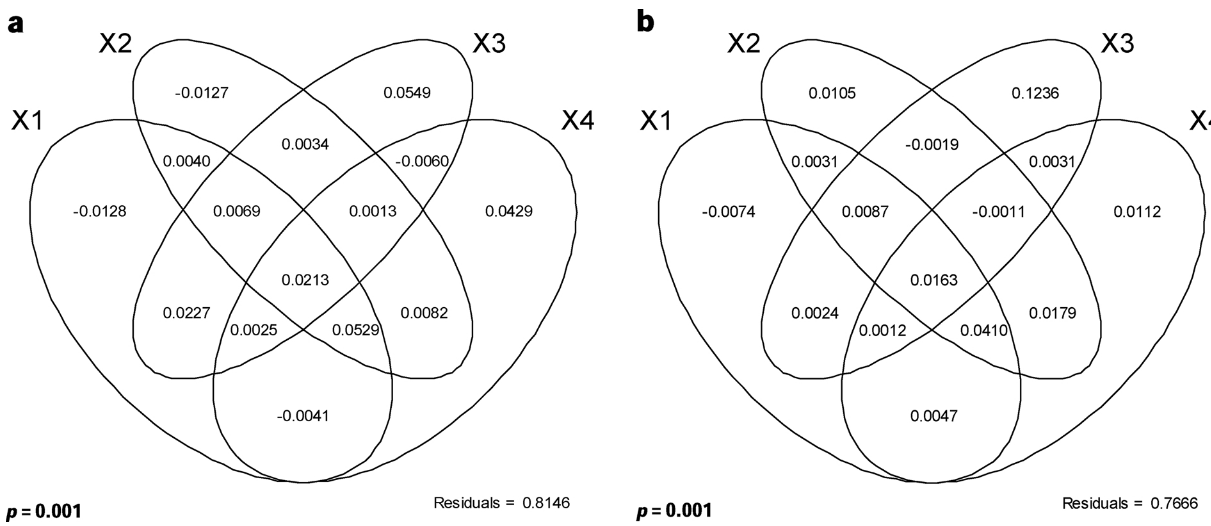

Fig. 2. Results of RDA variation partitioning for channel data-sets in the rainy (a) and dry (b) seasons. The negative signal means no effect (Borcard et al., 2011). Data sets: X1–Forest cover at catchment scale, X2–Forest cover at network scale, X3–Catchment physical variables, and X4–Forest cover at reach scale. Forward selected predictors in each component of the data-sets: a) X1 = FNC + FFC, X2 = FNN + FDN, X3 = QUA + LIO + SL + OXI; b) X1 = FNC + FFC, X2 = FPN + FDN + LDN, X3 = QUA + ULT + EL + LIO.

season. However, the variation partitioning showed that in the majority of models analyzed the physical predictor explained more variance than the forest cover predictors. In the cases where the forest cover predictor had a higher variation explained (APR, Dep, K, CO2, DO, and Temp) the variance explained varied from 8% to 50%. Only 16 variables showed residual spatial autocorrelation (r varying from 0.08 to 0.22).

In the rainy season, the stream variables Alk, K, TSS, Acid, CO2, Col, DO, Temp, PR, LW, Wid, and Area had positive relationships with the forest predictors, while CO2, pH, DO, PR, APR, LW, Dep, and Vol had negative relationships with the forest predictors. In the dry season, the stream variables NH3-N, K, TSS, Col, FTU, Cond, Temp, PR, APR, LW, and Dep had positive relationships with the forest predictors, while DO, Temp, Hab, Dep, and Vol had negative relationships with the forest predictors.

In general, the results showed that a high percentage of forest cover (especially in stream reach), the proximity of the forest to the sampled reach, and a good conservation level (less fragmented) were related to higher quality stream conditions. An important exception was observed for TSS, which was positively related to forest cover in the landscape and to the proximity of the forest cover.

4. Discussion

4.1. Effects of forest cover on channel structure and water chemistry

Our regression analyses showed positive relationships between forest cover metrics and characteristics of the studied streams generally associated with stream ecosystem health. We found that more forest in the reach or closer was related to increased wood, habitat diversity, and dissolved oxygen, and to decreased channel depth, volume, and temperature. This was expected, as the presence of forest at the sampling reach guarantees a continuous supply of wood and shade (Richardson and Danehy, 2007; Paula et al., 2011; Leal et al., 2016). Shade controls water temperature, and hence, dissolved oxygen (Allan and Castillo, 2007). Wood is an important structural resource that contributes for habitat diversity by creating dammed and erosive pools (Paula et al., 2011). Forest cover in the reach also retains sediments from the uplands (Leal et al., 2016), although we did not find the expected negative relationship between forest cover and TSS. In fact, many of the correlations between forest and response variables were relatively weak in our averaged models, and might have been even weaker if we had included candidate models with combinations of non-forest variables. Moreover, our redundancy analysis showed minimal effect of forest cover on response variables after accounting for physical variables.

Nevertheless, the relationships we observed between forest cover and stream habitat and chemistry metrics were (with the exception of TSS) consistent with expectations. Poor forest cover at local scales increases the rates of channel erosion. The rates of lateral migration are lower in forested reaches as dense tree roots reduce the rate of cutbank migration (Hickin, 1984; Allmendinger et al., 2005). Hence, the high erosion rates can erode stream margins, creating larger and deeper lateral pools at points of marginal erosion and channel migration (Fig. 3c). In addition, the high erosion rates can erode the stream bed, increasing channel depth and creating larger and deeper pools in the middle of the channel (Fig. 3a and b). An alternative explanation is that the channel is becoming deeper by high floodplain accretion (accumulation of sediments in the floodplain) caused by capture and retention of sediments by the grass vegetation in the floodplain after floods events. As these sediments accumulations increase, the floodplain is buried leaving deeper channels (Allmendinger et al., 2005). Other studies also found that forest cover increased the amount of pool habitats (Lammert and Allan, 1999), habitat diversity, bed roughness (Jones et al., 1999; Sutherland et al., 2002; Sweeney et al., 2004), stream health (Tanaka et al., 2016), and improved water features (Fernandes et al., 2014; Leal et al., 2016; Taniwaki et al., 2016). Differently from our results, Leal et al. (2016) found that local deforestation was associated with wider and shallower channels.

We also found that less fragmented forest cover in the landscape is related to higher pH and reduced K, water acidity, and temperature. Forest cover fragmentation in the landscape increase the erosion in upland areas and the input of fine sediments and nutrients to the channel, which are carried to downstream reaches and change water features (Liébault et al., 2005; Allan and Castillo, 2007; Taniwaki et al., 2016). Also, poor management of agricultural activities can increase the amount of sediments produced in areas with poor forest cover. For example, the filtering capacity of forested buffers can be easily suppressed by gully formation that connect the agricultural fields directly to the stream, generating sediments and nutrients that make their way to the stream channels (Lowrance et al., 1997). These gullies are common in the study area, especially in the sugarcane fields and high slope pasture areas (personal observations). Trails created by cattle in the riparian forest to access stream water can have the same effect. All these degradation processes can be intensified by fragmented and very narrow forested buffers in the network since their filtering capacity can be greatly reduced by decreasing buffer width (Wenger, 1999; Tanaka et al., 2016).

Table 4
Results of variance partitioning for channel data after model averaging and predictors selection.

Variable	Predictor1	Predictor2	Shared	Total	Residuals
Rain					
Hab	–	ALT(−) = 0.10	–	0.1	0.9
Hab	–	AREA(−) = 0.09	–	0.09	0.91
Hab	–	QUA(−) = 0.06	–	0.06	0.94
PR	LDN(+) = 0.05	–	–	0.05	
PR	FPR(−) = 0.08	–	–	0.08	
APR	FNN(−) = 0.10	EL(−) = 0.03	0.01	0.14	0.84
APR	FPR(−) = 0.15	EL(−) = 0.05	0	0.2	0.79
LW	LDN(−) = 0.22	ULT(−) = 0.09	0.01	0.32	0.66
LW	FPR(+) = 0.31	OXI(+) = 0.04	0	0.35	0.65
LW	FPR(+) = 0.26	ULT(−) = 0.06	0.04	0.36	0.62
Wid	FPC(+) = 0.04	AREA(+) = 0.10	0	0.14	0.85
Wid	PDC(+) = 0.05	EL(+) = 0.07	0	0.12	0.86
Wid	PDC(+) = 0.04	AREA(+) = 0.09	0	0.13	0.85
Wid	FDC(+) = 0.05	EL(+) = 0.05	0.02	0.12	0.86
Wid	FDC(+) = 0.09	AREA(+) = 0.11	(−)0.01	0.19	0.8
Dep	FNC(−) = 0.26	AREA(+) = 0.15	0.01	0.42	0.56
Dep	FPR(−) = 0.28	SL(−) = 0.11	0.02	0.41	0.57
Dep	FPR(−) = 0.24	AREA(+) = 0.10	0.06	0.4	0.58
Area	FPC(+) = 0.04	AREA(+) = 0.10	0	0.14	0.85
Area	PDC(+) = 0.05	EL(+) = 0.07	0	0.12	0.86
Area	PDC(+) = 0.04	AREA(+) = 0.09	0	0.13	0.85
Area	FDC(+) = 0.05	EL(+) = 0.05	0.02	0.12	0.86
Area	FDC(+) = 0.09	AREA(+) = 0.11	(−)0.01	0.19	0.8
Vol	FPR(−) = 0.20	AREA(+) = 0.23	0.08	0.51	0.47
Subs	–	QUA(−) = 0.10	–	0.1	0.9
Subs	–	OXI(+) = 0.04	–	0.04	0.96
Sand	–	QUA(+) = 0.28	–	0.28	0.72
Dry					
Hab	FDN(−) = 0.07	EL(−) = 0.07	(−)0.01	0.13	0.87
PR	FDC(+) = 0.05	QUA(−) = 0.21	(−)0.06	0.2	0.8
APR	PDC(+) = 0.04	EL(−) = 0.14	(−)0.01	0.17	0.82
APR	LDC(+) = 0.05	ULT(+) = 0.10	0	0.15	0.82
APR	LDC(+) = 0.08	EL(−) = 0.14	(−)0.01	0.22	0.78
LW	FPR(+) = 0.50	OXI(+) = 0.08	(−)0.02	0.56	0.43
LW	FPR(+) = 0.50	SL(−) = 0.05	(−)0.02	0.53	0.46
Wid	–	AREA(+) = 0.20	–	0.2	0.8
Dep	FNC(−) = 0.25	AREA(+) = 0.10	(−)0.01	0.34	0.65
Dep	LDN(+) = 0.24	SL(−) = 0.11	(−)0.02	0.33	0.66
Area	–	AREA(+) = 0.20	–	0.2	0.8
Vol	FNC(−) = 0.21	AREA(+) = 0.30	(−)0.02	0.49	0.5
Subs	–	QUA(−) = 0.07	–	0.07	0.93
Subs	–	OXI(+) = 0.09	–	0.09	0.91
Sand	–	QUA(+) = 0.20	–	0.2	0.8

The signal beside the predictor shows the direction of the relationship. The signal beside the numbers means no effect (Borcard et al., 2011).

Table 5
Results of variance partitioning for water data after model averaging and predictors selection.

Variable	Predictor1	Predictor2	Shared	Total	Residuals
Rain					
Alk	FIC(+) = 0.05	EL(−) = 0.32	(−)0.06	0.31	0.68
Alk	LDC(+) = 0.06	EL(−) = 0.27	(−)0.01	0.32	0.66
Alk	LDN(+) = 0.04	EL(−) = 0.28	(−)0.01	0.31	0.69
K	FFC(+) = 0.08	OXI(−) = 0.09	0.15	0.32	0.66
K	FFC(+) = 0.08	ULT(+) = 0.08	0.15	0.31	0.67
K	FFC(+) = 0.17	EL(−) = 0.10	0.06	0.33	0.66
K	FDN(+) = 0.07	ULT(−) = 0.18	0.04	0.29	0.69
Ca	–	EL(−) = 0.29	–	0.29	0.71
Mg	–	EL(−) = 0.23	–	0.23	0.77
TSS	FPC(+) = 0.08	LIO(−) = 0.13	(−)0.05	0.16	0.83
TSS	FNC(+) = 0.02	QUA(+) = 0.07	0.05	0.14	0.84
TSS	FNC(+) = 0.09	LIO(−) = 0.10	(−)0.01	0.18	0.81
TSS	FNN(+) = 0.03	QUA(+) = 0.07	0.05	0.15	0.83
TSS	FNN(+) = 0.07	LIO(−) = 0.07	0	0.14	0.83
TSS	FPR(+) = 0.03	QUA(+) = 0.07	0.05	0.15	0.83
TSS	FPR(+) = 0.08	LIO(−) = 0.08	0	0.16	0.82
Acid	PDN(+) = 0.09	–	–	0.09	0.91
Hard	–	EL(−) = 0.27	–	0.27	0.73
CO2	LDC(+) = 0.08	SL(−) = 0.10	(−)0.03	0.15	0.84
CO2	PDN(+) = 0.08	EL(−) = 0.06	(−)0.01	0.13	0.86
CO2	PDN(+) = 0.11	SL(−) = 0.11	(−)0.03	0.19	0.81
CO2	FDN(+) = 0.08	SL(−) = 0.13	(−)0.06	0.15	0.84
CO2	LDN(+) = 0.07	SL(−) = 0.08	(−)0.01	0.14	0.85
CO2	FPR(−) = 0.06	SL(−) = 0.06	0.01	0.13	0.86
Col	FFN(+) = 0.05	–	–	0.05	0.95
FTU	–	QUA(+) = 0.05	–	0.05	0.95
Cond	–	EL(−) = 0.21	–	0.21	0.79
pH	PDN(−) = 0.02	QUA(−) = 0.03	0.01	0.06	0.92
pH	PDN(−) = 0.07	LIO(+) = 0.08	(−)0.03	0.12	0.87
DO	LDC(−) = 0.05	–	–	0.05	0.95
DO	FDN(−) = 0.06	–	–	0.06	0.94
DO	LDN(−) = 0.06	–	–	0.06	0.94
DO	FPR(+) = 0.08	–	–	0.08	0.92
Temp	LDC(+) = 0.05	QUA(+) = 0.06	0	0.11	0.87
Dry					
Alk	–	EL(−) = 0.32	–	0.32	0.68
NH3-N	FIC(+) = 0.07	EL(−) = 0.08	(−)0.04	0.11	0.89
K	FFC(+) = 0.08	ULT(+) = 0.10	0.16	0.34	0.63
K	FDN(+) = 0.12	ULT(+) = 0.18	0.08	0.38	0.6
Ca	–	EL(−) = 0.31	–	0.31	0.69
Mg	–	EL(−) = 0.29	–	0.29	0.71
TSS	FPN(+) = 0.05	OXI(−) = 0.06	0	0.11	0.87
TSS	FIN(+) = 0.05	OXI(−) = 0.06	0	0.11	0.88
Hard	–	EL(−) = 0.31	–	0.31	0.69
CO2	–	EL(−) = 0.06	–	0.06	0.94
Col	PDC(+) = 0.02	ULT(+) = 0.03	0.03	0.08	0.9
Col	PDC(+) = 0.06	EL(−) = 0.07	0	0.13	0.86
Col	FIC(+) = 0.04	ULT(+) = 0.09	(−)0.02	0.11	0.88
Col	FIC(+) = 0.06	EL(−) = 0.11	(−)0.05	0.12	0.86
Col	PDN(+) = 0.03	OXI(−) = 0.04	0	0.07	0.9
Col	PDN(+) = 0.07	ULT(+) = 0.09	(−)0.02	0.14	0.85
Col	PDN(+) = 0.05	EL(−) = 0.07	0	0.12	0.87
FTU	FIC(+) = 0.09	AREA(−) = 0.11	(−)0.01	0.19	0.8
Cond	LDC(+) = 0.04	EL(−) = 0.23	0	0.27	0.72
DO	LDC(−) = 0.08	EL(+) = 0.06	0	0.14	0.85
DO	LDN(−) = 0.13	EL(+) = 0.06	0	0.19	0.8
Temp	FFC(+) = 0.06	EL = 0.06	(−)0.02	0.1	0.9
Temp	FFC(+) = 0.05	AREA(+) = 0.09	(−)0.02	0.12	0.87
Temp	FFN(−) = 0.04	AREA(+) = 0.04	0.02	0.1	0.88
Temp	FDN(+) = 0.09	EL(+) = 0.04	(−)0.01	0.12	0.86
Temp	FDN(+) = 0.09	AREA(+) = 0.07	0	0.16	0.84

The signal beside the predictor shows the direction of the relationship. The signal beside the numbers means no effect (Borcard et al., 2011).

Some other stream variables were influenced only by physical variables, like Alk, Ca, Mg, Hard, CO2, FTU, Cond, pH, Hab, Wid, Area, Subs, and Sand. Catchment elevation is expected to be negatively related to water chemistry as low elevation reaches are depositional areas of materials carried from upstream reaches. This increases nutrients and suspended sediments in stream water and changes water features on downstream reaches (Welch et al., 1998; Allan and Castillo, 2007). Streams draining through sandy soils will naturally have a high percent of sand substrate because most of bed materials come from eroded hillslopes or stream margins (Richards et al., 1996; Allan and Castillo, 2007). Therefore, these relationships represent more general patterns in stream ecology.

The positive relationship between forest cover measures and TSS was an unexpected result as forest cover is expected to decrease fine sediments. This result needs to be further investigated because if the forest cover is not performing its filtering function in full, there is the risk of excessive sedimentation and habitat degradation which can imposes serious threats to the ecosystem and the biodiversity (Harding et al., 1998; Sutherland et al., 2002, Iwata et al., 2003Sutherland et al., 2002, Iwata et al., 2003). Another important issue in our work is that our sampling design did not cover a complete gradient of forest cover for the catchment scale due to the predominancy of agriculture in this landscape. This may weaken some of the results found since the percent of forest cover considered to be high is not enough to provide enough protection for some stream features, like sediments for example.

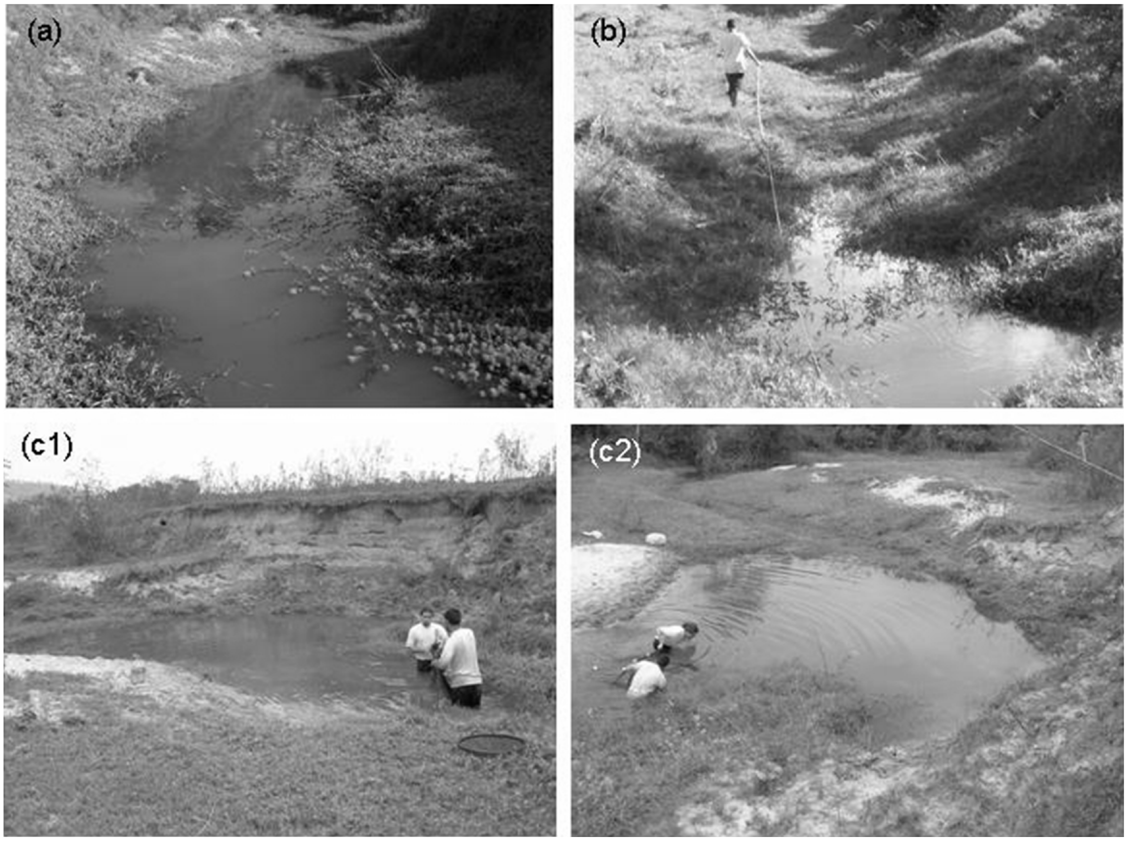

Fig. 3. Large pools in deforested streams. Large mid-channel pools formed possibly by channel narrowing (erosion of channel bed) or floodplain accretion (A-B). Large lateral pool formed by erosion of channel margins (C1-C2). (Photo Credits: P. Gerhard, A. Ferreira).

4.2. Ecological and management implications

Deforestation and forest fragmentation have several ecological implications for stream ecosystems. In this study, the riparian deforestation at the reach and the forest cover fragmentation at the landscape level are associated with changes in the habitat structure and water features, respectively. Because forested headwater streams are generally small and shaded (Richardson and Danehy, 2007), the increase in light associated with forest alterations (Fig. 3) changes stream habitats to conditions more typical of riverine habitats (more water column habitat, warmer temperatures, and less dissolved oxygen). In turn, this can impact fish communities by favoring the establishment of exotic and riverine species (Casatti et al., 2009; Gerhard and Verdade, 2016). In the study area, the deforested reaches have more fish richness and diversity than forested reaches, probably due to the addition of exotic species (*Tilapia rendalli* Boulenger, *Oreochromis niloticus niloticus* Linnaeus, and *Poecilia reticulate* Peters) and species that inhabit larger streams (*Astyanax altiparanae* Garutti & Britski; Gerhard and Verdade, 2016). Other alterations associated with local deforestation were already reported in this study area, like changes on the feeding behavior of two nektonic fishes (Ferreira et al., 2012) and no input of LW and less LW habitat (Paula et al., 2011) on these deforested streams.

Our results indicate that the secondary and degraded forest cover present in our study area provides some protection of the stream environment in these agricultural landscapes. Therefore, it is beneficial to restore the forest cover where it is deforested, especially in the riparian zones. Forest cover regeneration has the potential to rehabilitate stream ecosystems (Iwata et al., 2003; Liébault et al., 2005) and it seems that this is happening in our study area. However, it is not clear if forest regeneration has been protecting stream ecosystems from sedimentation in this area, as observed for TSS and high sedimentation observed in some streams (Fig. 4). We found a high percent of sand substrate ($>$ 70%) in catchments with high percent of forest cover at both the drainage ($>$ 70% forested, n = 6) and catchment ($>$ 40% forested, n = 5) scales. More specifically, we observed a high percent of sandy substrate ($\geq$95%) in two highly forested catchments (60% and 98% of forest in the 30 m buffer of the drainage), even though they have a low percent of sandy soil in the catchment (18% and 10%, respectively). Also, we observed values above 70% for sand substrate in 11 and 14 streams in the dry and rainy season, respectively, and eight streams had values above 70% in both seasons. These highly sedimented streams also presented the lowest values for channel depth, volume, pools number, and substrate diversity. Although heavy sedimentation has affected only some of the streams we sampled, heavy sedimentation represents a concern in our study area high erosive potential for agricultural activities (Cavalli et al., 2001). Some of these streams are highly forested (drainage network and the reach scales), but they were located predominantly in areas covered by pastures in steep slope and erosive areas or by sugarcane (soil remains unprotected during the rainy season after harvesting). Some of them had a high quantity of sand substrate in both seasons, suggesting that this material probably is not being transported annually or that it is being continuously replenished. Therefore, while the large proportion of riparian forest in the drainage network of these channels suggests that these streams are being protected in this agricultural landscape, they may actually be substantially degraded by the effects of extensive sedimentation. Therefore, further studies, especially in the long term, are necessary to confirm these observations.

It is important to mention that natural regeneration alone is not enough to fully recover forests in some situations, as forest recovery may be hindered by ongoing disturbance in many agricultural landscapes and also by land degradation (soil compaction, nutrient loss and low propagule availability) caused by the time and the intensity of past land use (Nepstad et al., 1991; Chazdon, 2008; Griscom et al., 2009). In

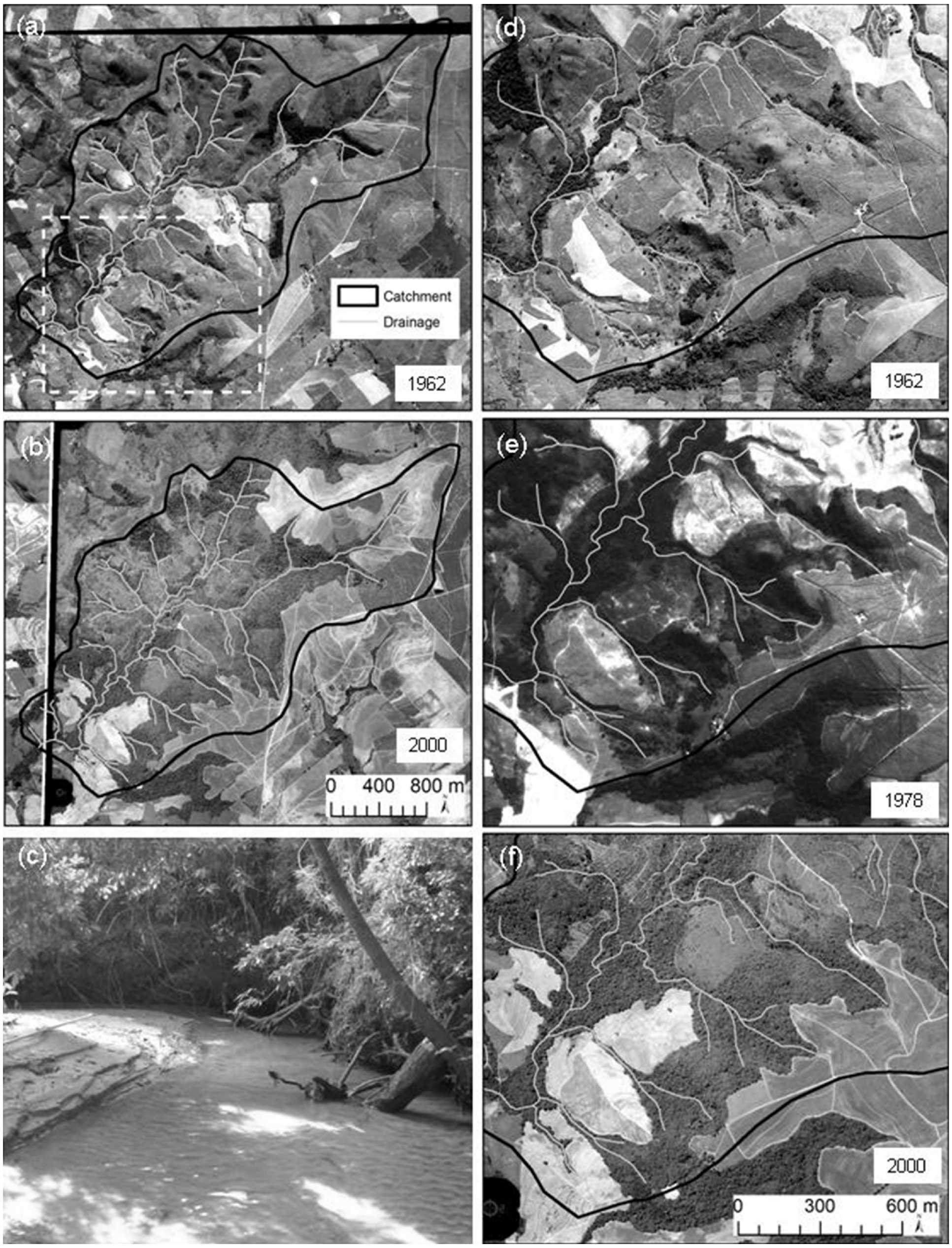

Fig. 4. Forest cover regeneration and stream sedimentation in an agricultural landscape. Forest cover regeneration in the last decades (A-B – the regenerated area represented by the white dotted square is showed in detail in D-F). Channel completely filled with sand in the sampled reach (C).

the study area, most of the forest regeneration has not been guided by any kind of restoration or management plan as farmers generally adopt land abandonment for natural forest regeneration since they do not have the financial resources to perform assisted restoration (human intervention to recover soil characteristics and plant communities by planting native tree species). However, assisted restoration may be required when the area is highly degraded or when unassisted forest regeneration has proven ineffective to recover the structural and functional attributes of the ecosystems (Chazdon, 2008; Clewell and Aronson, 2013).

Another important issue related to the conservation of streams is the management of agricultural activities. In the study area, the uplands have regions of sandy soils in their landscape and are dominated by poorly managed pastures that are subject to erosion. Low elevation catchments are dominated by sugarcane plantations. The recent expansion and intensification of sugarcane production in Brazil has been accompanied by an increase of fertilizers use (Martinelli and Filoso, 2008), which can lead to eutrophication of streams (Johnson et al., 1997; Silva et al., 2007). Moreover, sugarcane burning, heavy machinery, soil exposure, and agricultural use of limestone for soil pH correction (common practices in sugarcane plantations in Brazil) are sources of ions and sediments carried to the channel by runoff (Allan and Castillo, 2007; Silva et al., 2007). Some studies have found higher nitrate concentrations in some sugarcane catchments even though they have a high percent of forest cover in the drainage network (Cassiano, 2013; Mori et al., 2015; Taniwaki et al., 2016). Apparently, poor forest conservation in the landscape and poor land use management in the catchment have contributed to these results, as observed by Taniwaki et al. (2016), who found higher conductivity, nitrate and dissolved organic carbon in a sugarcane catchment with deforested springheads compared to one with preserved springheads.

The Brazilian Forest Code (Brasil, 1965, 2012) is an important instrument to promote forest conservation in agricultural areas of Brazil, but has proven challenging to enforce (Soares-Filho et al., 2014). Recent modifications that weakened these forest protection provisions may worsen the situation. Before the modifications, agricultural areas were required to have 30 m wide buffers (Permanent Preservation Areas – PPA) with natural forest along the entire drainage network for < 10 m wide streams. Recent modifications reduced the buffers width in agricultural areas established before 2008 (see Soares-Filho et al., 2014). These riparian areas were illegally deforested and should have been reforested with 30 m wide buffers, but with the new regulations, the buffer width to be reforested will vary from 5 to 15 m depending on the size of the property; property size classes vary according to the regions in the country (Brasil, 2012). This change has strong ecological implications for stream ecosystems because most Brazilian agricultural landscapes will have less protection provided by the riparian forests and may see their ecological functions decrease with smaller forested buffer width (Wenger, 1999). Also, the establishment of these 30 m wide forested buffers has been questionable because in practice they are vulnerable to disturbances in agricultural landscapes and they may not be enough to protect riparian forests and to ensure the regeneration process. Hence, these forests will become more vulnerable and less effective to protect streams.

Fortunately, the new Forest Code (2012) also introduced new mechanisms that contribute to reduced deforestation and to increased forest regeneration at larger scales, potentially bringing ecological benefits for surrounding ecosystems (see Soares-Filho et al., 2014). Also, improving the use of fire to clean the fields, controlling crop fertilization, strengthening environmental legislation and its enforcement, and creating cooperative programs to support farmers on conservation and restoration actions are extremely important to reduce forest and stream degradation. We believe that these actions will provide more protection for riparian and stream ecosystems in agricultural landscapes, conciliating agricultural production and nature conservation.

5. Conclusions

Our results indicate that the current forest cover in this agricultural landscape is a useful indicator of stream health. However, not all features of the stream ecosystem may be protected by the present forest cover. Some deforested areas should be reforested to reduce fragmentation, and as most of the forest cover in our study area is a result of natural regeneration, it seems that natural regeneration is a viable restoration action to restore forests and rehabilitate stream ecosystems. This is important because the finantial resources to support the active planting of trees is very limited. This is the reality for most tropical countries that face deforestation issues. However, it is extremely important that this action be continuously monitored in order to evaluate the effectiveness of the regeneration process as agricultural landscapes have disturbances that may hinder or delay the regeneration process. It is possible that using forest regeneration in conjunction with practicing a good management of the agricultural activities (to reduce the disturbances and enhance forest regeneration) can offer good results for riparian forests and stream protection in agricultural landscapes. This action can be especially important in recent deforested areas, as soil nutrients and tree seeds usually are promptly available to start the regeneration process, making the restoration process faster and cheaper. Enforcing environmental legislation is also important to preserve riparian forests.

Acknowledgments

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ecolind.2017.11.061.

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