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Previous Land Use Affects the Recovery of Soil Hydraulic Properties after Forest Restoration

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Abstract: Knowledge of soil hydraulic properties after forest restoration is essential for understanding the recovery of hydrological processes, such as water infiltration. An increase of forest cover may improve water infiltration and soil hydraulic properties, but little is known about the response and extent to which forest restoration can affect these properties. The purpose of this study was to investigate the effect of forest restoration on surface-saturated soil hydraulic conductivity (K_s), and to verify the K_s recovery to the pre-disturbance soil conditions. We sampled field K_s at the surface in Campinas municipality, São Paulo State, Brazil, at 18 plots under three land-cover types: (i) a pasture; (ii) a restored forest using a high-diversity mix of plantings (85 regional native species) of 9 years of age; and (iii) a remnant forest patch. We used the Beerkan method for soil hydraulic characterization. Bulk density (ρ_b), soil organic carbon content (OC), soil porosity and particle size data were also sampled. We found considerable differences in soil hydraulic properties between land-cover classes. The highest K_s were observed in remnant forest sites and the lowest K_s were associated with pasture sites. The K_s recovery differs markedly between restored forests. Our results strongly suggest that soil attributes and K_s recovery are influenced by the duration and intensity of land use prior to forest restoration. Attention needs to be given to management activities before, during and after forest restoration, especially where the soil is still compacted and K_s is low.

Keywords: soil properties; saturated soil hydraulic conductivity; soil infiltration; Beerkan method

1. Introduction

The global forest restoration movement based on natural regeneration and tree plantations has increased tropical forest cover [1,2]. Nevertheless, soil hydraulic property responses in these restored forests are virtually unknown [3,4]. Soil water infiltration is a key hydrological process which, among others, influences groundwater recharge, soil erosion and surface runoff. Indeed, one of the best parameters for understanding and studying soil infiltration is the saturated soil hydraulic conductivity (K_s) [4,5]. The K_s is a soil property with the greatest spatial and temporal variability among soil properties. The K_s variability depends on many factors, such as soil types, land uses, soil depths, landscape positions, methods of measurement and physical and chemical soil attributes [6]. Despite this variation, the K_s is a useful and sensitive indicator of the effect of land-cover change on soil hydro-physical dynamics [7], which exerts a dominating influence on the partitioning of rainfall in vertical and lateral flow paths. Therefore, estimates of K_s are essential for describing and modelling hydrological processes [8].

The Atlantic Forest is one of the most important forest biomes of Brazil that has suffered intense pressure from human occupation, with approximately 12% of the original area remaining [9]. Recently, the Atlantic Forest Restoration Pact has emerged to restore large areas of degraded land. This is the largest forest-restoration initiative in Latin America with a target of restoring 15 million hectares of forest by 2050 [10]. These efforts have a substantial impact on soil hydraulic properties and can be expected to affect the hydrological processes in the restored ecosystems. However, these hydrological implications are rarely considered in studies of forest restoration [11]. Current literature reviews in tropical landscapes suggest that forest restoration can enhance surface K_s [12,13]. However, most studies on K_s recovery after forest restoration in tropical soils emphasize areas with natural regeneration or secondary succession [4,7,14–19].

Zimmerman et al. [17] found non-significant K_s recovery at surface and near-surface (12.5 and 20 cm soil depth) levels in Brazilian Amazônia during seven years of secondary succession after pasture abandonment. Recently, Leite et al. [19] by examining four sites of different ages in the Brazilian Caatinga—an abandoned pasture, a young forest (7 years), an intermediate forest (35 years), and an older forest (more than 55 years)—observed that forest regrowth promotes surface K_s recovery, increasing progressively over time. On the other hand, the effect of active restoration on K_s has been much less studied [20]. Zwartendijk et al. [11] compared surface K_s recovery between degraded lands, semi-mature forest, 2–10-year-old naturally regenerating vegetation and fallows that were actively reforested 6–9 years ago with 120 native species in Madagascar. They found higher K_s values in the semi-mature forest, followed by the active reforested sites, suggesting that active restoration may decrease the time it takes for the soil to recover hydraulic properties. Also, the impact of afforestation on K_s has been studied in teak (*Tectona grandis*) plantations at surface and near-surface (12.5 and 20 cm soil depth) levels in Brazilian Amazônia, where after 10 years the teak plantation shows K_s recovery from pasture conditions for all soil depths, but K_s values are still distant from pre-disturbance conditions [4]. Similarly, an increase in K_s after afforestation practices has been reported by several other tropical studies [21–23].

Tree planting to restore degraded lands is conducted in the expectation that soil hydraulic properties will be improved [13]. In order to understand the effect of forest restoration on K_s, we investigated the K_s recovery by field estimation under three land covers, namely pasture, 9-year-old restored forest, and remnant forest. To the best of our knowledge, no studies have investigated the K_s recovery after planting native mixed-species in the Brazilian Atlantic Forest and compared the results with pasture and remnant forest. We hypothesized that forest restoration can recover the surface K_s to the pre-disturbance soil conditions. The following questions were addressed: (1) Does forest restoration recover top-soil K_s values that characterize the remnant forest? (2) Are the measured soil attributes between the land covers similar?

2. Materials and Methods

2.1. Field Site

This research was carried out in the county of Campinas, São Paulo State, Brazil (22°54′ S, 46°54′ W). The area is located inside the sub-basin of Atibaia River (2800 km^2), which belongs to the Piracicaba River basin. This region has suffered over 200 years of historical landscape changes. In the Atibaia sub-basin, the main land covers are: native vegetation (33%), pasture lands (30%), and crops (17%), and the forest cover increased 5.7% in the last decade [24]. The mean annual precipitation is 1700 mm and the mean annual temperature is 20 °C, with rainy months generally concentrated between October and March. The native vegetation in the area is classified as seasonal semi-deciduous forest [25]. The two soil types found in the study sites are Ultisols and Entisols [26], related to the diverse geomorphology of the region, which is located at the transition between the Atlantic Plateau and the Peripheral Depression geomorphological provinces. The rocks in the Atalntic Plateau are mainly composed of granites and gneises, while the Peripheral Depression is characterized by sedimentary rocks. The elevation ranges from 600 m to 900 m with an undulating topography and the presence of slopes higher than 20% [27].

2.2. Experimental Design

The sites were selected to capture variation in soil attributes. Also, the sites' accessibility was taken into account in this selection. We examined the following land-cover classes: pasture, restored forest, and remnant forest. In each class we selected two sites or toposequences (Figure 1), under pasture (P1 and P2), under restored forest (R3 and R4), and under remnant forest (F5 and F6).

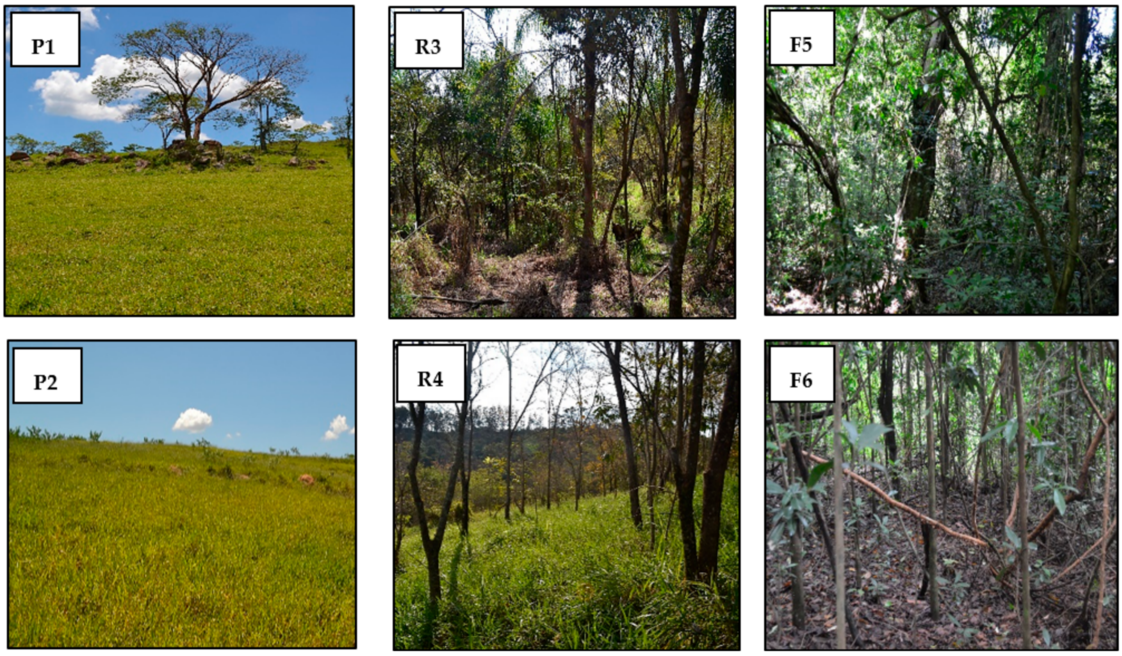

Figure 1. Pictures that represent the study sites in the seasonal semi-deciduous forest in Campinas, Brazil. Study sites are abbreviated with P1 and P2 for pasture, R3 and R4 for restored forest, and F5 and F6 for remnant forest.

The length of each toposequence was constrained by topography and varied between 100 m and 150 m. Each site was divided into three landscape positions (upslope (U), midslope (M) and downslope (D)). Within each landscape position, we located one plot (7 × 7 m in size), resulting in 18 plots altogether. Detailed characteristcs of the three land-cover classes are as follows.

The pasture site P1 (22°49′24″ S, 46°54′39″ W) and P2 (22°54′38″ S, 46°53′26″ W) was characterised by a dense cover of grass. The dominant grass species is *Urochloa brizantha*. Information obtained from landholders revealed that the pasture sites have been heavily grazed for more than 20 years and have a stocking rate between 1 to 1.5 animal units ha^{-1}. The measurements at these sites represent the K_s and soil attributes before forest restoration actions.

The restored forest sites (R3 and R4) were 9 years old when sampled and located in Fazenda Guariroba (22°53′48″ S, 46°54′28″ W). The forest-restoration process of an area of 300 ha began in 2007. The mixed plantation with a high-diversity mix of seedlings (85 regional native species), aimed to provide economical insurance and ensure successional processes for landowners [28,29].

Site preparation included grass control through herbicide applications and control of leaf-cutter ants by the distribution of insecticide baits. Direct seedling planting (3 × 2 m spacing) took place after conventional tillage. The mixed plantation also used fertilizer and irrigation at the time of planting and during the first year [28,30]. Aerial photographs and interviews with local peoples showed that land-use history differs between the restored forest sites. Both restored forests were originally deforested more than 100 years ago and planted with coffee (*Coffea arabica*) during the first decades of the 20th century. After the coffee plantation, the restored forest R3 was planted with eucalyptus (*Eucalyptus* sp.); this abandoned forest existed until 2006 without a commercial purpose, although a frequent grazing of cattle occurred, then it was harvested and grazing continued one year before the forest restoration. The eucalyptus harvest was made by motor–manual operations and a farm tractor forwarded the logs. The vegetation in the restored forest R3 prior to restoration activities consisted of low shrub and grasses. On the other hand, the restored forest R4 after the coffee plantation was used as pasture for livestock breeding until 1986, was subsequently planted again to coffee (*C. arabica*), and agricultural terraces were created with heavy track machinery. Then, the coffee plantation was replaced by pasture in 1996, which was similar to the pastures sites (P1 and P2), dominated by the grass species *U. brizantha*, and without natural regeneration.

The forest sites (F5 and F6), used as a reference for soil attributes, were located in Ribeirão Cachoeira forest (22°50′13″ S, 46°55′58″ W), the second largest natural remnant forest of 245 ha in the county of Campinas. The forest presents a high tree species diversity, with an average canopy stature of 15 m and emergent trees reaching up to 35 m tall [31].

2.3. Soil Sampling and Measurements

The first field campaign started in February and ended in March 2017. A total of four disturbed soil samples were collected per plot to determine the soil particle size distribution (PSD) and the soil organic carbon content (*OC*). The PSD was determined by the hydrometer method and soil texture was classified according to the US Department of Agriculture (USDA) standards [32]. The *OC* was determined by the Walkley–Black method [33]. In addition, four undisturbed soil cores (0.05 m in height and 0.05 m in diameter) were also collected per plot at the depth of 0–0.05 m to determine soil macroporosity (*Mac*) and microporosity (*Mic*), using the Richards pressure chamber with the application of 6 kPa suction [34].

Soil infiltration measurements were taken in a second field campaign during the month of June 2017 (dry season). We conducted a K_s characterization using the Beerkan method [35], referred to as BEST. We chose the BEST test because it is a simple, fast and inexpensive method [36–38]. At each plot, we carried out seven infiltration runs using a steel ring with an inner diameter of 0.16 m inserted approximately 0.01 m into the soil surface, with a minimum distance between measurements of 2 m. Before the ring's insertion, the litter was removed and, if necessary, the grass and ground cover were cut in order to expose the soil surface. Sampling-point selection was influenced by suitable ground conditions for measurment and constraints such as tree roots, rocks and variations in microtopography. For each infiltration run, we collected one undisturbed soil core (0.05 m in height and 0.05 m in diameter) at the 0–0.05 m depth. We used the undisturbed soil cores to determine the initial volumetric soil water content (θ_i), the soil bulk density (ρ_b) and total soil porosity (*Pt*) assuming a particle density of 2.65 g cm^{-3} [39]. In each measurement, a known volume (150 mL) was repeatedly poured into the cylinder and the time needed for the complete infiltration of this volume was logged. We repeated the procedure until the difference in infiltration time between two or three consecutives trials became negligible. At the end of each infiltration test, we collected a disturbed soil sample inside the ring area to determine the saturated gravimetric water content, and thus the satured volumetric water content (θ_s) was calculated using the ρ_b. A total of 126 experimental cumulative infiltrations, $I(t)$ (L), versus time, t (T), were then deduced, 42 for each land cover, 21 for each site, and 7 for each plot.

2.4. Estimating and Selecting the BEST Algorithm

The BEST-steady algorithm by Bagarello et al. [40] was used to obtain the K_s (K_{sB}, the subscript B is used to indicate BEST-steady). This choice was made since it allows a higher success percentage of the infiltration runs to be obtained compared with other possible algorithms, such as BEST-slope [41] and BEST-intercept [42], whose data require fitting to the transient stage of the infiltration run. Another expected advantage of the BEST-steady algorithm is that the possible problems associated with the use of the transient infiltration data are avoided. The BEST-steady expresses the K_{sB} with the following equation [43]:

$$K_{sB} = \frac{Ci_s}{Ab_s + C} \tag{1}$$

where i_s (L T^{-1}) and b_s (L) are, respectively, the slope and the intercept of the regression line fitted to the data describing steady-state conditions on the cumulative infiltration I (L) versus t (T) plot. Taking into account that BEST focuses on the Brooks and Corey relationship for hydraulic conductivity [44], the A (L^{-1}) and C constants are defined as follows [35]:

$$A = \frac{\gamma}{r(\theta_s - \theta_i)} \tag{2}$$

$$C = \frac{1}{2\left[1 - \left(\frac{\theta_i}{\theta_s}\right)^{\eta}\right](1-\beta)} \ln\left(\frac{1}{\beta}\right) \tag{3}$$

where γ and β are infiltration coefficients commonly set at 0.75 and 0.6 as explained by Lassabatere et al. [3,7,16,19], r (L) is the radius of the disk source, η is a shape parameter that is estimated from the capillary models [45], and θ_i and θ_s are the initial and final water contents, respectively. Note that θ_i should not exceed 0.25 θ_s; however, Di Prima et al. [43] showed that BEST-steady can be applied in initially wetter soil conditions (θ_i > 0.25 θ_s) without an appreciable loss of accuracy in the predictions of K_s. Therefore, as suggested by Cullotta et al. [46], the θ_i was not considered to affect the reliability of the predicted K_s. On the other hand, the BEST-steady algorithm failed in some sampling points, providing negative K_s values and affecting the reliability of measured K_s. For this reason, we also estimated K_s for the whole data set by the near steady-state phase of a Beerkan infiltration run (SSBI—K_{sS}, the subscript S is used to indicate steady-state) [47]. This method is attractive for a simple soil hydraulic characterization, but testing the ability of this procedure to estimate K_s is necessary. Indeed, in scientific literature there is no exhaustive testing of the performances of the SSBI method, notwithstanding that this method has a noticeable practical interest. This method estimates K_s through a simple Beerkan infiltration test and an estimate of the so-called sorptive parameter, α^* (L^{-1}), expressing the relative importance of gravity and capillary forces during a ponding infiltration process [48,49]. With this method K_{sS} is estimated by the following equation [47]:

$$K_{sS} = \frac{i_s}{\frac{\gamma\gamma_w}{r\alpha*} + 1} \tag{4}$$

where γ_w is a dimensionless constant related to the shape of the infiltration front and is set at 1.818 [50]. In this investigation, we considered α^* as a constant and equal to 0.012 mm^{-1}, since it was found to be usable in tropical soils [47,51]. The reasons for this choice was that we did not find in the literature other specific support for using a different α^* value for tropical soils. Following Bagarello et al. [47], the BEST-steady algorithm was chosen to check the SSBI method by comparing K_{sB} and K_{sS} in terms of factors of difference (*FoD*), calculated as the highest value between K_{sB} and K_{sS} divided by the lowest value between K_{sB} and K_{sS}. Differences between K_{sB} and K_{sS} not exceeding a factor of two were considered indicative of similar estimates [49].

2.5. Data Analysis

Data sets were summarized by calculating the mean and the associated coefficient of variation (CV). Following similar investigations [37,52], unique values of clay, silt, sand, *OC*, ρ_b, total porosity, macroporosity, microporosity and θ_i were determined for each plot by averaging the measured values, considering the small size of the sampled areas [52]. The hypothesis of normal distribution of both the untransformed and the log-transformed K_s data were tested by the Lilliefors test [53]. The other parameters were assumed to be normally distributed and, thus, no transformation was performed on these data before statistical analysis [54,55]. Treatment means were calculated according to the statistical distribution of the data, i.e., geometric means for K_s (log-normal distribution) and arithmetic means for all other parameters (normal distribution) [56]. According to Lee et al. [55], the appropriate CV expression for a log-normal distribution was calculated for the geometric means, and the usual CV was calculated for the arithmetic means. Statistical comparison was conducted using two-tailed *t*-tests, whereas the Tukey honestly significant difference test was applied to compare the data sets. The ln-transformed K_{sS} was used in the statistical comparison. A probability level, $p = 0.05$, was used for all statistical analyses. All analyses were carried out in the statistical programming software R [57].

3. Results

3.1. Differences in Soil Attributes among Study Sites

The PSD showed considerable differences among the soils. Most of the sampled plots presented sandy loam (P1U, P1M, P1D, R3U, R3M, RD, F5U and F5D) and sandy clay loam textures (R4U, R4M, R4D and F5M), and the rest clay loam (P2M, F6U and F6M) and loamy textures (P2U, P2D and F6D). The *OC* ranged from 14.76–35.37 g Kg^{-1} under pastures (P1 and P2), from 10.46–24.60 g Kg^{-1} under restored forests (R3 and R4), and from 17.53–48.59 g Kg^{-1} under remnant forests (F5 and F6). The ρ_b values ranged between 1.12–1.40 g cm^{-3} in the pastures, for the restored forests the values ranged from 1.09–1.52 g cm^{-3}, while in the remnant forests the values ranged from 0.88–1.25 g cm^{-3}. The *Pt* varied from 0.47–0.58 cm^3 cm^{-3} in the pastures, from 0.43–0.59 cm^3 cm^{-3} in the restored forests, and from 0.53–0.67 cm^3 cm^{-3} in the remnant forests. In general, the highest soil *Mac* values were observed in the remnant forests, the intermediate values in restored forests, and the lowest values in the pastures. In contrast, the soil *Mic* was greater in the pastures, intermediate in the restored forests, and lower in the remnant forests. The mean θ_i at the time of the Beerkan infiltration run varied between 0.16–0.37 cm^3 cm^{-3} and the soil was significantly wetter in plots P2M, R4U and R4M (Table 1).

3.2. Estimating and Selecting the BEST Algorithm

Overall, the Beerkan method used in this study was found to be robust for measuring the K_s in the field. However, the BEST-steady algorithm yielded physically plausible estimates (i.e., positive K_s values) in 108 of 126 infiltration runs (85.7% of the cases). The percentage of successful runs was 95.2% (40 of 42 runs) both in the pasture sites and restored forest. With reference to the remnant forest (F5 and F6), BEST-steady led to a failure rate value of 33.3%, leading to a lack of estimates in 14 of 42 infiltration runs. In these cases, convex cumulative infiltration-shaped data always produced a negative intercept of the straight line fitted to the data describing steady-state conditions, which yielded negative K_s values (Figure 2). On the other hand, the SSBI method always yielded physically plausible estimates (i.e., positive K_s values) and small differences were found between the K_{sB} and K_{sS} estimates (Figure 3). The means of K_{sS} differed from the corresponding values of K_{sB}, by a factor not exceeding 1.81. The individual determination (i.e., point by point) of the factors of difference, *FoD*, did not exceed 2.37 (mean of *FoD* is equal to 1.51) and they were less than 2 and 1.5 in 90% and 53% of the cases, respectively. Therefore, it can be argued that the BEST-steady and SSBI method led to similar estimates, given that the individual *FoD* values were lower than two in almost all cases.

Table 1. Comparison between the mean and coefficient of variation (CV) of the clay (%), silt (%), sand (%), soil organic carbon content (*OC* in g Kg^{-1}), soil bulk density (ρ_b in g cm^{-3}), total porosity (*Pt* in cm^3 cm^{-3}), macroporosity (*Mac* in cm^3 cm^{-3}), microporosity (*Mic* in cm^3 cm^{-3}) and initial volumetric soil water content (θ_i in cm^3 cm^{-3}), values for the 18 sampled plots in the landscape positions upslope (U), midslope (M) and downslope (D).

Variable	Statistic	Plots																	
		Pasture 1			Pasture 2			Restored Forest 3			Restored Forest 4			Remnant Forest 5			Remnant Forest 6		
		U	M	D	U	M	D	U	M	D	U	M	D	U	M	D	U	M	D
Clay	Mean	19.6a	10.2b	9.5b	25.0b	31.7a	21.6b	11.2b	12.1b	19.0a	26.1a	21.0b	21.9b	18.3b	23.3a	19.0b	30.2a	31.0a	24.4b
	CV	7.9	2.8	12.9	5.6	3.9	13.1	17.3	11.7	4.3	5.1	3.9	2.9	14.4	2.0	5.7	3.6	4.4	2.0
Silt	Mean	27.4a	20.2b	22.3b	30.7a	32.5a	29.9a	20.7b	21.2b	27.7a	22.7a	19.4b	19.9b	25.4a	26.3a	26.7a	34.3b	33.9b	39.7a
	CV	6.6	6.8	6.5	7.1	6.4	11.6	14.6	12.8	6.8	2.1	2.5	6.5	10.2	8.5	8.6	5.8	2.5	1.2
Sand	Mean	53.0a	69.6a	68.3a	44.4a	35.8a	48.5a	68.1b	66.8b	53.3a	51.2a	59.7b	58.3b	56.3b	50.4a	54.3b	35.6a	35.2a	35.9b
	CV	4.1	1.7	3.8	3.5	2.5	13.0	6.8	4.6	4.9	3.3	2.1	2.0	8.0	5.1	6.2	8.3	1.7	2.3
OC	Mean	30.0a	20.1b	17.8b	32.1a	32.1a	25.6b	14.5a	17.3a	21.3a	22.0a	19.0a	17.4a	30.9a	34.8a	33.8a	31.1a	34.2a	27.8a
	CV	5.9	2.4	18.6	11.7	4.9	12.7	22.6	33.7	9.8	12.2	14.8	17.0	41.4	18.5	12.6	12.3	11.0	25.4
ρ_b	Mean	1.27a	1.24a	1.22a	1.29a	1.18b	1.33a	1.23a	1.29a	1.22a	1.33a	1.34a	1.42a	1.05b	1.03b	1.15a	1.02a	1.05a	0.99a
	CV	5.7	5.1	3.8	5.8	4.9	5.0	7.1	3.3	5.9	3.4	6.0	5.3	5.4	7.1	7.1	9.7	7.6	8.3
Pt	Mean	0.53a	0.53a	0.54a	0.51a	0.55a	0.50a	0.54a	0.51a	0.54a	0.50a	0.50a	0.47a	0.60b	0.61b	0.57b	0.61b	0.60b	0.63b
	CV	4.8	4.6	3.5	6.1	4.5	4.7	6.0	3.3	5.4	4.1	6.4	6.0	3.1	4.4	5.0	6.4	5.3	4.6
Mac	Mean	0.12b	0.18a	0.21a	0.11a	0.07b	0.06b	0.20a	0.19a	0.16b	0.18a	0.14b	0.16a	0.24a	0.26a	0.15b	0.19b	0.22a	0.18b
	CV	5.0	12.0	15.5	25.2	15.4	40.8	10.2	4.3	28.1	12.0	37.0	28.6	15.9	3.1	18.1	13.3	13.9	25.5
Mic	Mean	0.36a	0.32b	0.32b	0.50a	0.51a	0.50a	0.28b	0.29b	0.34a	0.35a	0.33a	0.31b	0.31b	0.29b	0.37a	0.37a	0.35a	0.37a
	CV	16.8	5.7	16.5	3.7	1.1	4.8	12.1	8.4	12.3	7.7	11.4	5.9	3.1	5.1	7.8	6.5	2.3	8.0
θ_i	Mean	0.32a	0.19b	0.16b	0.25b	0.37a	0.25b	0.19a	0.19a	0.19a	0.37ab	0.34a	0.23b	0.15a	0.17b	0.32a	0.21a	0.23a	0.17b
	CV	12.8	29.3	28.4	23.2	5.2	23.6	14.9	17.0	11.2	15.9	14.0	14.1	12.8	9.0	26.9	9.9	4.5	14.7

For a given variable and site (i.e., P1, P2, R3, R4, F5 and F6), means that do not share a letter are significantly different according to the Tukey test (p = 0.05).

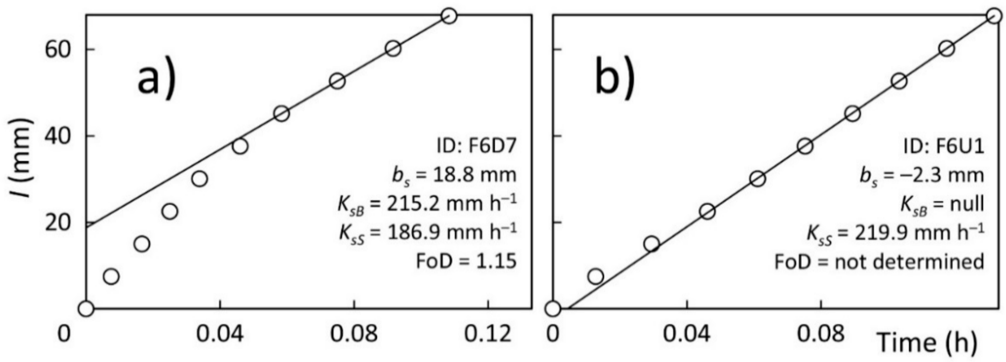

Figure 2. Illustrative examples of the influence of the shape of the cumulative infiltrations on the discrepancies occurring between the Beerkan method (BEST-steady) and the steady-state phase of a Beerkan infiltration (SSBI) method. (**a**) Concave-shaped cumulative infiltration curve in which the intercept, b_s (mm), of the straight line interpolating the last I vs. t data points is positive and the FoD between the saturated soil hydraulic conductivity values estimated with BEST-steady (K_{sB}) and the SSBI method (K_{sS}) is small. (**b**) Convex-shaped cumulative infiltration curve with a negative intercept yielding null K_{sB}.

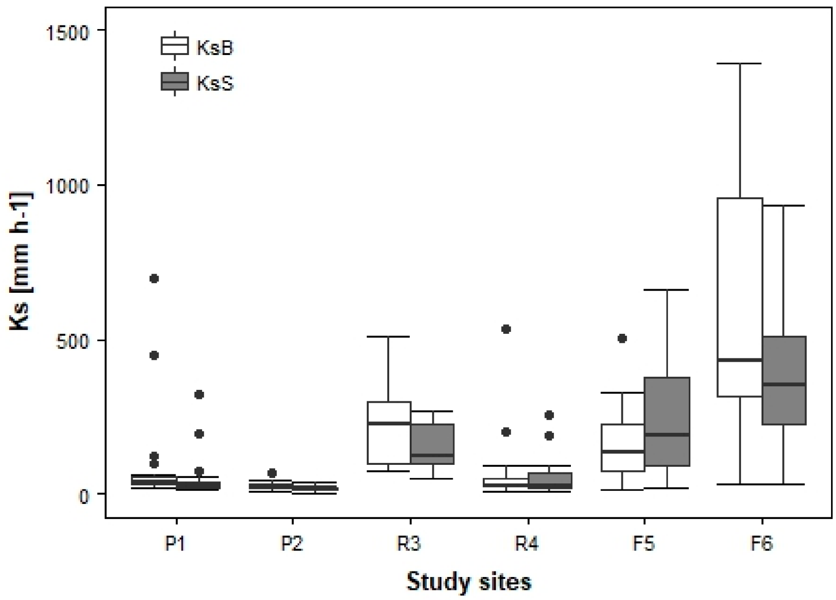

Figure 3. Comparison between K_s estimated with BEST-steady, K_{sB}, and the SSBI method, K_{sS}. Study sites are abbreviated with P1 and P2 for pasture, R3 and R4 for restored forest, and F5 and F6 for remnant forest.

The failure in the BEST-steady algorithm is reported by several studies in subtropical soils, where OC exceeds 5%. This failure is normally related to the occurrence of hydrophobic conditions [43,46,58]. Nevertheless, our soils showed lower OC values (less than 5%). In addition, the soil hydrophobicity is a complex property and poorly studied in tropical soils [59,60]. Other factors that probably contributed to the BEST-steady algorithm failure are the heterogeneous soil structure, changes in soil structure during measurement, initial soil moisture, and temperature [61,62]. For these reasons, the failure of the BEST-steady algorithm should be addressed in detail by future studies, considering detailed physical, chemical and mineralogical analyses. Hereafter, for the sake of reliable K_s values and comparison across study sites, only the K_{sS} values estimated using the SSBI method were considered. This choice was supported by the fact that the SSBI method allowed us to maintain the integrity of the dataset.

In addition, the K_{sS} values ranged between 3 mm h^{-1} and 934 mm h^{-1}, with a high variability inside all study sites.

3.3. Saturated Soil Hydraulic Conductivity (K_s) Characterization

Evaluating the surface K_s values by soil texture, greater K_s variation was found in soils with higher clay content, contrasting with lower variation in soils with higher sand content. Also, soils with higher sand content did not show the higher K_s. In general, the lowest K_s values occurred in pasture plots, for example, in pasture P1 the K_s ranged from 10–320 mm h^{-1}, and in pasture P2 K_s ranged from 4–37 mm h^{-1}, whereas the highest K_s values were observed in most remnant forest plots. The sandy loam texture highlighted the large differences between K_s in the restored forest R3 and pasture plots (P1U and P1M). In this case, the K_s in the restored forest R3 varied from 49–267 mm h^{-1}, with the higher K_s evidenced at the restored forest plot R3D (average of 180 mm h^{-1}); moreover, the K_s was similar to the pasture plot P1D (average of 110 mm h^{-1}) and most remnant forest plots. For the remnant forest F5, the K_s varied from 18–660 mm h^{-1}, showing the higher K_s at remnant forest plot F5U (average of 247 mm h^{-1}), which differs from pasture plots (P1U and P1M) but not from restored forest R3. In contrast, the K_s at remnant forest plot F5D (average of 68 mm h^{-1}) exhibited a similar K_s in relation to pasture and restored forest plots. For the sandy clay loam texture, the K_s in the restored forest R4 (from 6–256 mm h^{-1}) was significantly different from the remnant forest plot F5M (average of 68 mm h^{-1}); furthermore, all the plots in the restored forest R4 had low K_s variability, similar to the pasture land cover. Finally, clay loam and loam textures showed the same comparison among land covers, characterized by marked differences between lower K_s in pasture 2 and higher K_s in the remnant forest F6. In particular, the remnant forest F6 evidenced the higher K_s variability (from 33–934 mm) in the study sites (Figure 4).

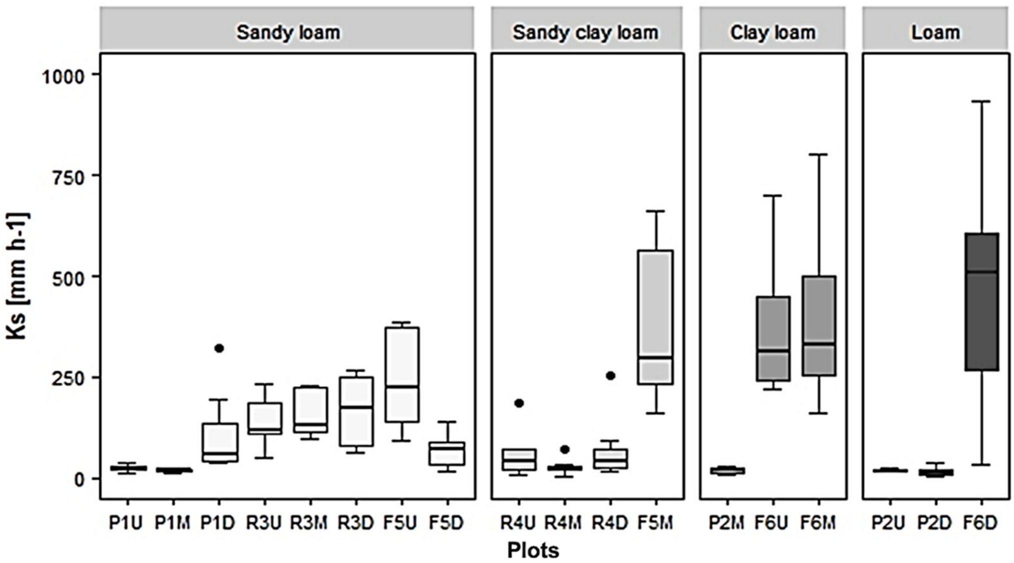

Figure 4. K_s estimated with SSBI method, K_{sS}, per plots and grouping by soil texture (US Department of Agriculture (USDA) classification system). Study sites are abbreviated with P1 and P2 for pasture, R3 and R4 for restored forest, and F5 and F6 for remnant forest. The subscript letter refers to the landscape position (Upslope, Middleslope and Downslope) in each site.

Statistical comparision of K_{sS} revealed no significant differences between the restored forest (R3) and the remnant forest (F5 and F6). However, significant differences between the restored forest (R4) and remnant forest were detected, indicating similarity with the pastures (P1 and P2) (Table 2).

Table 2. Results of the Tukey honestly significant difference test (p = 0.05) for the ln-transformed saturated hydraulic conductivity values estimated with the SSBI method (K_{sS}). The grouping information highlights the significant and not significant comparisons.

Variable	**Grouping Information (Plots)**																	
	F6M	**F6U**	**F5M**	**F6D**	**F5U**	**R3D**	**R3M**	**R3U**	**P1D**	**F5D**	**R4D**	**R4U**	**R4M**	**P1U**	**P1M**	**P2U**	**P2M**	**P2D**
K_{sS}	a	a	a	a	a	a	a	a										
					b	b	b	b	b									
						c	c	c	c	c	c							
							d	d	d	d	d	d						
									e	e	e	e	e	e				
										f	f	f	f	f	f	f	f	
												g	g	g	g	g	g	g

4. Discussion

4.1. Effects of Land Use on Soil Attributes and K_s

Although the soils in the study area showed some variability, this was overcome by choosing sites and landscape positions within the different land uses that presented similar soil textural classes in the surface horizon. This approach allowed us to group and compare the soil attributes and K_s (Figure 4). In general, important differences were observed in the soil attributes and K_s between land-cover classes. These differences could be related to many factors such as intensity of past land use [4,23], spatial and topographic variations in soil types along the toposequences [63,64], density and diversity of plants, root system, vegetation type, canopy cover and soil faunal activity, among others [19]. Unfortunately, the influence of these factors on soil attributes and K_s after forest restoration is poorly understood and needs to be included in future studies.

Pasture. As expected, K_s was significantly lower under pasture plots than restored forest and remnant forest plots. This result was directly related to the highest ρ_b found in the study pastures, which influences the higher soil *Mic* and lower soil *Mac* values [65]. Similar findings have been reported be several authors [13–15]. An exception to this was related to pasture plot P1D, which showed similar K_s values compared to the restored forest and remnant forest in the sandy loam texture, suggesting lower soil compaction, and consequently higher soil *Mac*. Also, the highest sand content found in this plot could help to explain this result. Moreover, the present results illustrate the K_s spatial variability in two different pasture sites, characterized by a low variation in K_s values. This could be due to the soil compaction [4,13], and the duration of pasture use in the land-use history, which is one of the most important factors for K_s variability over time [15,17], as well as the cattle-grazing intensity that could have influenced the K_s variability in the pasture plots [7]. Moreover, the lower soil faunal activity and organic matter in pasture land covers are important factors when analyzing the soil attributes [15,56]. Notably, the pasture plots P1U, P2U and P2M, had *OC* similar to the remnant forest. These similarities are closely linked to carbon inputs in pastures sites where the root system of grasses, the animal-derived organic matter and the application of fertilizers might have increased the organic substrate [20,65]. In contrast, pasture plots P1M, P1D and P2D showed the lowest *OC* values in the pasture plots, which could be attributed in part to the higher sand content in these plots.

Restored forest. The soil texture, understory vegetation (Figure 1) and intensity of past land use were different in the restored forest sites (R3 and R4), and these are the most likely reasons for the differences in soil attributes and K_s values between both restored forests [23,66]. Also, it is important to underscore that this result could have been influenced by possible soil compaction during mechanized soil preparation during the forest restoration [67]. The most important soil attributes of the K_s differentiation between restored forest sites was the ρ_b and *OC*. For example, the higher K_s in the restored forest plot R3D was associated to the lowest ρ_b and higher *OC* values. Overall, the restored forest R3 with higher sand content (sandy loam texture) exhibited lower *OC*, lower soil *Mic*, lower ρ_b, higher soil *Mac* and higher K_s than restored forest R4. The higher K_s in restored forest 3, relative to pasture plots with similar soil texture (P1U and P1M), is consistent with the results under teak plantation in Brazilian Amazônia [4] and pine plantation in Nepal [23]. Furthermore, plots in the restored forest R3 showed no significant differences in K_s from most remnant forest plots. These results can be linked to the land-use history in the restored forest R3, where the presence of an abandoned eucalyptus forest with a canopy structure of more than 50 years influenced the low trampling pressure and machinery traffic intensities, suggesting a litter accumulation that could have protected the soil surface during this period [68,69].

In the second situation, the restored forest R4 with higher clay content (sandy clay loam texture) exhibited higher *OC*, higher ρ_b and lower K_s than restored forest R3. In particular, the lower K_s compared to the remnant forest plot (F5M) with a similar soil texture clearly indicates that the full return to pre-disturbance conditions is still far off [22]. On the other hand, the sandy clay loam texture did not include pasture plots; however, pasture K_s in this soil texture could be assumed to be similar to the pasture sites (P1 and P2), considering the low spatial K_s variability observed in the pasture land cover. Thus, the restored forest R4 showed no significant differences in K_s with the pasture sites. This result can be related to past land-use intensity in the restored forest R4, in which the combination of coffee plantation and pastures led to greater soil exposure, and also trampling pressure and the construction of agricultural terraces could have caused erosion and soil compaction before the forest restoration. The present results agree with several studies [4,16,23], which suggest that K_s decreases with increasing land-use intensity, and that K_s recovery will be longer in view of the intensive land use. Filoso et al. [13] argued that in some cases the recovery of soil infiltration after forest restoration may be extremely difficult, because of the absence of natural understory vegetation. This research did not directly quantify the herbaceous cover, but in the field we observed that natural regeneration in the restored forest R4 is impeded by the dominance of an invasive grass species (*U. brizantha*), which is also associated with the open canopy conditions. Conversely, restored forest R3 presented visually a canopy structure with greater understory vegetation. Indeed, the canopy cover determines the interception rainfall, reducing raindrop impact and surface sealing, which could enhance the K_s [19]. Additionally, the higher ρ_b values in restored forest R4 are an indication of lower root and soil organism presence [70]; this may reduce plant seed germination, reduce root growth and decrease soil oxygen availability, becoming an ecological filter in the natural regeneration processes [71,72]. Zimmerman et al. [17] reported that invasive species could delay the K_s recovery in Brazilian Amazônia after a decade of pasture abandonment.

Remnant forest. Comparing remnant forest plots and pasture plots in the sandy loam, clay loam and loam textures allowed the detection of significantly higher K_s in remnant forest plots. In the case of the sandy clay loam texture, the remnant forest plot F5M showed significantly higher K_s than plots in the restored forest R4. In contrast, the sandy loam texture showed no significant differences between plots of restored forest 3 and remnant forest plot F5U. These results are related to the lowest ρ_b and higher *Mac* values that favor the K_s, suggesting a higher soil pore connectivity. In the specific case of remnant forest plot F5D in the sandy loam texture, no significant differences were found in relation to pasture plots (P1U and P1M). This result can be associated with the high ρ_b and a consequent increase in the soil *Mic* that was noted in the remnant forest plot F5D. The soil attributes and K_s values in remnant forest sites could be explained by the longer time that these forests have remained undisturbed, which allows a better soil structure to develop and the storage of more soil carbon [19,66]. These findings are in agreement with those reported by several other studies in the Atlantic Forest [63,65]. Additionally, the K_s spatial variability observed in both remnant forests is in line with previous work by Hassler et al. [7], who attributed the K_s variability in Panama forest soils to overland flows that result in erosion [19]. Other factors that might have caused the K_s spatial variability in remnant forest plots were the steepness of the sample plots and the soil distribution in the landscape positions (U, M, D) influenced by contrasting slope and topography.

4.2. Management Implications

The fact that restored forests R3 and R4 showed clear differences in K_s recovery and soil attributes may provide evidence that, in some cases, simply planting trees is not, in itself, enough to recover the soil attributes to the pre-disturbance soil conditions [23]. Attention needs to be given to management activities before, during and after forest restoration, especially where the soil is still compacted and K_s is low. From this point of view, it is therefore important that monitoring forest restoration programs includes collection of soil compaction and K_s data to understand the initial compaction degree and soil infiltration, reinforcing the need to compare these values with the pre-disturbance soil conditions. After assessing soil compaction and soil infiltration at the restored forests, management practices could be implemented to alleviate soil compaction, such as mechanical loosening techniques (i.e., deep ripping and subsoiling), which may improve plant growth [73,74]. In addition, some technical methods in forest restoration that have been shown to aid natural regeneration and soil recovery are the suppression of weedy vegetation and maintenance and enrichment planting [28].

If the pasture sites (P1 and P2) presented here represent the planted pastures of the Atlantic Forest, we could observe that water infiltration is drastically affected in most cases, regardless of the soil texture. This result and the negative effects of pastures that have been heavily grazed are well documented [4,15,17] and have also been confirmed in the present research. Indeed, according to Martínez and Zinck [15] pasture degradation can be improved by rotational grazing and the introduction of silvopastoral systems during pasture management. Moreover, there is an increasing number of reports regarding different tropical land covers, suggesting that lower K_s may lead to less groundwater recharge and increases in overland flow frequency [3,7,16,19]. Thus, our results reinforce the need for better management practices in pastures and restored forests to avoid soil erosion, conserve water and create opportunities to enhance water infiltration [75].

5. Conclusions

In this study, the hypothesis that forest restoration can recover the surface K_s to the pre-disturbance soil conditions was not supported for both restored forest sites (R3 and R4). We found two different situations with marked differences in soil attributes and K_s recovery between restored forest sites. Our results strongly suggest that soil attributes and surface K_s recovery are influenced by the duration and intensity of land use prior to forest restoration: while the restored forest R3 with a previous lower intensity of land use showed similar K_s to the remnant forest sites, the K_s recovery in restored forest R4 is still far-off from these remnant forest sites due to greater exposure of the soil and trampling pressure during the land-use history.

The present results further illustrate that the measured soil attributes were different between land-cover classes: pasture, restored forest and remnant forest. They also bring out the inverse relationship between K_s and ρ_b, where the K_s increases as a result of a decrease in ρ_b, and, consequently, the dominance of macropores over micropores, which facilitate soil water infiltration.

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