University of São Paulo
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Recovery of soil hydraulic properties after forest restoration in the Atlantic Forest

Sergio Esteban Lozano Baez

Thesis presented to obtain the degree of Doctor in Science. Area: Forest Resources. Option in: Conservation of Forest Ecosystems

Piracicaba
2019

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versão revisada de acordo com a resolução CoPGr 6018 de 2011

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EPIGRAPH

"Faites que le rêve dévore votre vie afin que la vie ne dévore pas votre rêve"

Antoine de Saint-Exupéry, Le Petit Prince (1943)

SUMMARY

RESUMO

Recuperação das propriedades hidrícas do solo após da restauração florestal na Mata Atlântica

O conhecimento sobre as florestas em processo de restauração florestal ao redor do mundo está cada vez mais em evidência, devido principalmente à sua importância nas funções ecossistêmicas relacionadas à água, tais como a promoção da infiltração. Contudo, apesar de existirem muitos estudos sobre áreas em restauração, abordando sua biodiversidade e algumas funções ecossistêmicas, o papel do solo nessas florestas em restauração permanece ainda pouco conhecido, por exemplo, poucos trabalhos têm analisado o efeito sobre o solo das diferentes estratégias de restauração (e.g., plantio de espécies nativas e regeneração natural). Nesse contexto, o objetivo desse trabalho foi avaliar e obter uma melhor compreensão dos efeitos de diferentes metodologias de restauração florestal na recuperação das propriedades físicas e hídricas do solo, mais especificamente na recuperação da infiltração de água no solo. Na primeira parte desse estudo (Capítulo 2) foi realizada uma revisão sistemática da literatura científica, reportando e discutindo os resultados de trabalhos sobre infiltração de água no solo em florestas em processo de restauração nos Trópicos, por meio do plantio de árvores. Os resultados desses trabalhos mostraram que houve um aumento da infiltração após o plantio de árvores; também observamos que a recuperação da infiltração foi mais rápida quando a agricultura era o uso anterior do solo; que solos mais argilosos (>30% argila) tenderam a exibir maiores aumentos na infiltração após plantio de árvores; e que as florestas em restauração com 10 ou mais anos evidenciaram valores de infiltração mais similares com as condições pré-distúrbio do solo (e.g., floresta natural de referência). Os dois capítulos restantes do trabalho (Capítulos 3 e 4) foram realizadas em uma área em processo de restauração florestal, com plantio de espécies nativas e elevada diversidade, no municipio de Campinas, São Paulo, Brasil. No Capítulo 3 foi investigado o efeito da restauração florestal na condutividade hidráulica do solo (K_s), verificando a recuperação da K_s até as condições pré-distúrbio. A K_s foi amostrada no campo em três tipos de uso do solo: (i) pastagem; (ii) área em restauração com 9 anos de idade; e (iii) florestal natural remanescente. Os resultados desse capítulo mostraram que a recuperação da K_s diferiu entre as áreas em processo de restauração; e que os atributos do solo e a recuperação da K_s foram influenciados pela intensidade e tempo de uso do solo anterior à restauração florestal. No Capítulo 4 foi avaliado o efeito do histórico de uso do solo na recuperação da K_s, dos atributos do solo e da vegetação, comparando as estratégias de restauração ativa vs. passiva (e.g., restauração assistida). Nesses capítulos concluímos que as ações de restauração florestal podem melhorar as propriedades físicas e hídricas do solo, porém, a recuperação de valores próximos aos valores de referência tem grande dificuldade, especialmente quando a área em restauração teve um histórico de uso intenso do solo. É fundamental entender como ocorre a recuperação do solo nas áreas em processo de restauração ecológica em diferentes tipos de climas, florestas e solos. Assim, fica claro a necessidade de pesquisas de longo prazo que foquem no movimento da água no perfil do solo, visando entender como a restauração florestal recupera o processo de infiltração da água no solo, inclusive na escala de paisagem (e.g., bacia hidrográfica).

Palavras-chave: Restauração florestal; Infiltração; Hidrologia; Propriedades do solo

ABSTRACT

Recovery of soil hydraulic properties after forest restoration in the Atlantic Forest

Knowledge about forests undergoing restoration across the world is becoming increasingly essential due to the benefits of restoring forest for ecosystem functions related to water, such as water infiltration. Although there is a growing literature regarding the biodiversity and some ecosystem functions in forest undergoing restoration, soil responses in these forests remain virtually unknown. Moreover, few works have analyzed the effects on soil of different restoration approaches (e.g., planting of native species and natural regeneration). In this context, the main objective of this work was to evaluate and gain a better understanding of the effects of different forest restoration methodologies on the recovery of soil physical and hydraulic properties, more specifically on water infiltration. In the first part of this study (Chapter 2) was conducted a systematic review of scientific literature, reporting and discussing the infiltration measures in tropical forests undergoing restoration by tree planting. The results of this review indicated that infiltration was likely to increase after tree planting; that infiltration recovery was faster when agriculture was the prior land use; that clayey soils (>30% clay) tended to exhibit greater increases in infiltration after tree planting; and that restored forests after 10 years evidenced more similar infiltration values with the pre-disturbance soil conditions (e.g., natural reference forest). The following two parts of the thesis (Chapter 3 and 4) were based on a restoration program using a high-diversity mix of native plantings in the county of Campinas, São Paulo, Brazil. In the Chapter 3 was investigated the effect of forest restoration on saturated soil hydraulic conductivity (K_s), verifying the K_s recovery to the pre-disturbance soil conditions. We sampled field K_s under three land-cover types: (i) a pasture; (ii) a restored forest of 9 years of age; and (iii) a remnant forest patch. Our results showed that K_s recovery differ markedly among the forests undergoing restoration; and that soil attributes and K_s recovery are influenced by the duration and intensity of land use prior to forest restoration. In the Chapter 4 we assessed the effects of land use history on the recovery of K_s, soil and vegetation attributes, comparing active vs. passive restoration (e.g., assisted restoration). In these chapters we conclude that forest restoration actions may improve soil physical and hydraulic properties, but in some cases a complete recovery to reference levels can be difficult, especially when land use was more intense prior to forest restoration actions. It is very important to understand soil recovery in forests undergoing restoration on different climate, forest and soil types. Thereby, in future research long-term studies are essential, which should focus in the water movement through the soil profile and aiming to understand how the forest restoration can recover the infiltration process, also including landscape scale (e.g., watershed).

Keywords: Forest restoration; Infiltration; Hydrology; Soil properties

1. INTRODUCTION

Recent forest restoration initiatives are playing a fundamental role in forest cover expansion across the planet (Chazdon, 2008; Keenan et al. 2015), through ambitious international (e.g., Bonne Challenge; New York Declaration on Forests and Goal 15 in U.N. Sustainable Development Goals), regional (e.g., 20 x 20 and AFR100) and national initiatives (Chazdon et al. 2017). In Brazil, the Pact for the Restoration of the Atlantic Forest has emerged to restore 15 million hectares of forest by 2050 (Rodrigues et al. 2009) and the National Plan for the Restoration of the Native vegetation has a goal of restoring 12 million hectares in the next 20 years. These restoration efforts are fundamental for human well-being, especially when considering the recovery of biodiversity and ecosystem functions (Sabogal et al. 2015; Aronson et al. 2017). In this context, the soils have critical relevance, providing ecosystem services through their functions (Mendes et al. 2018). Seven ecosystem functions have been highlighted for which soils are critical (Keesstra et al. 2016): (i) biomass production; (ii) storing, filtering and transforming nutrients, substances and water; (iii) biodiversity pool, such as habitats, species and genes; (iv) physical and cultural environment for humans and human activities; (v) source of raw material; (vi) acting as carbon pool; (vii) archive of geological and archaeological heritage. Additionally, since restoration ecology emerged, the importance of soil has been increasingly recognized as fundamental to reach the goals of forest restoration programs (Heneghan et al. 2008). Nevertheless, in forest undergoing restoration around the world, most studies have focused on establishing aboveground plant communities, but belowground environment (e.g., soil physical and hydraulic properties), and aboveground-belowground linkages have been neglected. For that reason, some authors stated that forest restoration has been "phytocentric" (Kardol & Wardle, 2010; Ohsowski et al. 2012). This situation is evident in the Brazilian Atlantic Forest, where a recent literature review emphasizes that the monitoring in most of the restoration projects are not considering any soil indicator (59% of 152 published works), and they are not including information about reference sites (e.g., old-growth forest and degraded lands) (Mendes et al. 2018). The noticeable soil data gaps are a great challenge to forest restoration practice and a test to soil ecology knowledge, which according with (Bradshaw, 1987) statement, it could provide an "acid test" (Heneghan et al. 2008).

Considering the aforementioned information, the effects of restoration efforts on soil still requires a better understanding. In particular, water infiltration, which is considered as a key hydrological process that influences a large number of essential ecosystem processes, such as groundwater recharge, soil erosion and surface runoff (Zimmermann et al. 2006; Neary et al. 2009), as well as growth and plant productivity (Thompson et al. 2010), have been poorly explored in forest restoration contexts. Water infiltration is defined as the process of water entry into the soil surface (Brutsaert, 2005). In this way, during rainfall events the water can move through the soil profile both vertically and horizontally by the combined effect of gravity and capillary, also depending on the soil type and slope of the terrain (Elsenbeer, 2001). It is important to note that water infiltration is a complex process which depends on a large number of factors, thus, a vast literature has been produced in the last century to understand the physics and dynamics of this process

(Assouline, 2013). Since early work of Darcy (1856), it is possible to find in scientific literature several approaches developing mathematical and numerical tools to quantitatively describe the infiltration process (e.g., Buckingham, 1907; Richards, 1931; Philip, 1957 among other studies).

A crucial parameter to characterize water infiltration is the saturated hydraulic conductivity (K_s), which determines how much water will move through the soil under saturated conditions (Elsenbeer, 2001; Hassler et al. 2011). Although K_s is reported to have the greatest variability among soil physical and hydraulic properties (Deb, 2012), it is also considered a key indicator to evaluate the effects of land use change on soil hydrology (Zimmermann et al. 2006; Zimmermann & Elsenbeer, 2008). For instance, removal of native forest cover followed by settlement of different land uses (e.g., agriculture and pastures) results into soils with low values of K_s, which could be associated with increasing erosion, surface runoff and floods (Ziegler et al. 2004; Nyberg et al. 2012). On the contrary, when the land use change occurs from degraded soils to forested land, using different restoration approaches (e.g., active and passive restoration), the K_s may eventually reach the high values found in pre-disturbance soil conditions. But this recovery may take decades (Bonell et al. 2010; Ghimire et al. 2014) and can be extremely difficult in some cases (Filoso et al. 2017). While most studies measuring soil physical and hydraulic properties in forest undergoing restoration have focused in areas under natural regeneration, actively restored forests and comparisons between restoration approaches have rarely been considered (Ghimire et al. 2014; Crouzeilles et al. 2017).

Currently, it is clear that there are several knowledge gaps for soil recovery process after forest restoration, and soil is still regarded as a "black box" by restoration practitioners (Heneghan et al. 2008). To bridge these gaps and continue opening the "black box", our study aims to evaluate and gain a better understanding of the effects of different forest restoration methods on the recovery of soil physical and hydraulic properties, more specifically on water infiltration. Having that in mind, our work is based on a forest restoration program in the Brazilian Atlantic Forest, planed and implemented by the Forest Ecology and Restoration Laboratory (LERF) from "Escola de Agricultura Luiz de Queiroz" (ESALQ), University of São Paulo, in partnership with the Fazenda Guariroba, located in the county of Campinas, São Paulo State, Southeast Brazil. This program started in 2007 over an area of 300 hectares, testing different restoration approaches (e.g., active and passive restoration). Before restoration actions the area was formerly covered mainly by low-yielding pastures and presents slope percentages greater than 20%. In particular, tree plantings were implemented as mixed plantation with high-diversity-mix of seedlings (85 regional native species), aiming to provide economical insurance for landowners and to ensure successional processes. For detailed information's of this restoration model see Preiskorn et al. (2009). Also, it is important to highlight that active restoration was implemented on highly degraded soil, with an intense land use history and poor potential for autogenic restoration, and passive restoration was promoted on a slightly degraded soil, with a previous second-growth forest and high potential for autogenic restoration (Rodrigues et al. 2011).

Given the previous context, this thesis consists of five chapters. This first chapter is an overall introduction, containing the contextualization of the topic, main objectives, questions and hypotheses that have been addressed during the research. The second chapter presents a systematic review of scientific literature reporting and discussing infiltration measures in tropical forests undergoing restoration by tree

planting. In this chapter we also analyzed the potential effect of restoration age and soil texture on infiltration responses after tree planting. The third chapter investigates the K_s recovery by field estimation under three land covers, namely pasture, 9-year-old undergoing restoration forest, and remnant forest. We hypothesized that forest restoration can recover surface K_s values to the pre-disturbance soil conditions, also the following questions were addressed: (1) Does forest restoration recover top-soil K_s values that characterize the remnant forest? and; (2) Are the measured soil attributes between land covers similar? The fourth chapter compares the K_s, soil physical and hydraulic properties recovery between soils under active vs. passive restoration approach. This chapter evaluates whether differences in land use history led to differences in soil and vegetation attributes. Specifically, we hypothesized that K_s would vary with intensity of land use history, considering that the active restoration site had a more intensive land use history, we expected that K_s recovery and vegetation attributes will be higher in the passive restoration. Finally, the fifth chapter presents a summary of the main results of this study, providing the main conclusions and recommendations for future researches.

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2. TREE PLANTING EFFECTS ON INFILTRATION CAPACITY IN THE TROPICS: A SYSTEMATIC AND CRITICAL REVIEW

This manuscript was submitted in the journal *Forest Ecology and Management*.

Abstract

Infiltration of rainfall is one of the most important hydrological processes with important influence on soil erosion, runoff, soil moisture content and groundwater recharge in ecosystems. This is particularly important in forest restoration contexts, considering the increasing number of restoration initiatives around the world promoting tree planting and consequently increasing forest cover. Nevertheless, a comprehensive overview of the effects of tree planting on infiltration and the factors controlling these effects is lacking. Here, we conducted a systematic review of scientific literature reporting infiltration measures in restored forests in the tropics. We found eleven studies representing 67 data comparisons in eight countries. Overall results indicate that infiltration is likely to increase after tree planting in forest restoration. Infiltration recovered quickly when agriculture was the prior land use, whereas recovery was slower when trees were planted into pastures and bare soils. There was a trend in restored forests with soils having >30% clay to exhibit greater increases in infiltration, contrasting with the smaller increases for sandy soils (<30% clay). Restored forests after 10 years evidenced similar infiltration values than the pre-disturbance soil conditions. Our findings emphasize the need to monitor water infiltration in restored forests under different soil conditions and over time. Finally, we identified six knowledge gaps requiring future research efforts, that aim to improve our understanding of when and why forest restoration may promote recovery of infiltration in tropical soils.

Keywords: Agriculture; Forest; Forest restoration; Pasture; Saturated hydraulic conductivity

2.1. Introduction

Recent global assessments have shown that forest cover is expanding through forest restoration efforts, resulting in an increase of forest cover at rates of 2.5 million ha yr^{-1} around the world from 2010 to 2015 (FAO, 2015; Keenan et al., 2015). This increase is usually associated with positive effects on many ecosystem functions that are essential to sustain ecosystem services for human wellbeing such as infiltration of rainwater (Chazdon, 2008; Filoso et al., 2017). Infiltration is a key hydrological process in ecosystems to maintain productive soil-water-plant interactions but also with strong effects on soil erosion, runoff, soil moisture content and groundwater recharge (Ziegler et al., 2004; Gageler et al., 2014).

There have been limited efforts to summarize the effects of forest restoration on infiltration in the tropics. The early reviews by Bruijnzeel (2004) and Scott et al. (2005) studied to some extent the impacts of forest cover expansion on infiltration. Subsequently, Ilstedt et al. (2007) provided a meta-analysis of the increase in infiltration after tree planting in formerly agricultural fields. A more recent review by Filoso et al. (2017) focusing in the impacts of forest restoration on water yields, showed that infiltration was a positive

outcome after restoration activities. However, infiltration is a complex soil attribute with high variability in both space and time (Deb and Shukla, 2012), and it is affected by a number of factors, such as soil texture and structure, past land use, soil type and vegetation, among others (Zimmermann et al., 2006; Leite et al., 2017). Indeed, the influence of these factors on the recovery of infiltration after tree planting is poorly understood and has been neglected by previous reviews (Sun et al., 2018).

Here, we conducted a systematic review of scientific literature with the primary objective of gathering, synthesizing and discussing the available information on the recovery of infiltration in tropical forests restored by tree planting in sites under different prior land uses. We also analyzed the potential effect of restoration age and soil texture on infiltration responses after tree planting.

2.2. Material and methods

2.2.1. Literature Searching

A systematic literature search was made in December 2017, using ISI Web of Science and SciELO, without any restriction on publication year and including peer-reviewed articles published in English, Spanish or Portuguese. We used the following combination of terms: ("forest restoration" or "afforestation" or "reforestation" or "forestation") and ("water conductivity" or "infiltrat*" or "hydraulic conductivity" or "soil properties"). These keyword terms found 502 hits. To be included in the analysis, studies had to meet the following inclusion criteria; (1) focus on forest restoration actions using trees (e.g., not shrublands or arid ecosystems); (2) report field measurements on infiltration rate or saturated hydraulic conductivity in the surface soil (0-10 cm); (3) compare infiltration in the restored forest with either degraded or reference conditions; (4) reported mean values, standard deviations and sample size; (5) study site located in the tropical region, between 23.5º N and 23.5º S, and thus excluding temperate and boreal forests.

We only found eight papers that met all criteria. Thus, as suggested by Ilsted et al. (2007) and to increase the number of observations in the database, we included the following three subtropical studies: Gilmour et al. (1987), Gageler et al. (2014) and Ghimire et al. (2014), which are slightly outside the tropical region but met all other criteria.

2.2.2. Data Extraction and Database Building

For each study, we extracted the country, mean annual precipitation, latitude and longitude where the infiltration measurements took place. Response variables that could influence infiltration were compiled, such as soil type, soil texture, method of measurement (e.g., disc permeameter, minidisk infiltrometer, simple-ring or double-ring), planting type (e.g., monoculture or diverse), restoration age (as the number of years since forest planting was established), and land use prior to tree planting (e.g., agriculture, bare soil, pasture or reference forest). Soil type was classified according to the USDA classification system, and because of the important role that soil texture can have on infiltration, soils was grouped in two contrasted

groups based on clay content: less than 30 percent clay (<30% clay, hereafter "sandy soils") and greater than 30 percent clay (>30% clay, hereafter "clayey soils") (Basche and DeLonge, 2017). Furthermore, the values of infiltration rate or saturated hydraulic conductivity in the surface soil were recorded and standardized in $mm.h^{-1}$. Since several studies suggest that infiltration recovery might take more than one decade after tree planting (Zimmermann and Elsenbeer, 2008; Bonell et al., 2010), a 10-year threshold was used to detect if water infiltration could recover to pre-disturbance levels a decade after forest planting. Therefore, restoration age was classified in two main groups, namely "older restored forests" (>10 years) and "younger restored forests" (<10 years).

Previous agricultural lands were areas that have been heavily transformed for agricultural activities, included slash-and-burn agriculture and different croplands (e.g., annual crops). Bare soil included degraded areas where the topsoil was removed and have been exposed since the disturbance without vegetation cover. Pastures were areas that were highly disturbed by grazing livestock, where the vegetation cover is dominated by grass and herb species. Reference forests were naturally conserved or minimally disturbed areas that never have been totally cleared. These forests contain large trees, understory vegetation and a litter layer that covers the soil surface.

2.2.3. Data Analysis

The log response ratio (LRR) was used to quantify the effects of tree planting on infiltration relative to a control. The LRR was calculated using the following equation (Hedges et al., 1999):

$$\mathrm{LRR} = \ln \frac{X_{rest}}{X_{ref}} \qquad (1)$$

Where X_{rest} is the infiltration value in the restored forest, and X_{ref} is the infiltration value in the land use prior to tree planting or in the control. The mean effects with 95% confidence intervals were calculated for each land use prior to tree planting. Effect size estimates were considered significantly different from zero if their 95% confidence intervals did not cross zero (Benayas et al., 2009). In order to facilitate the interpretation the LRR were back transformed and converted to percentages (Basche and DeLonge, 2017) in figures, as:

$$\text{Percent change} = [\mathrm{Exp}(\mathrm{LRR}) - 1] * 100 \qquad (2)$$

In addition, mean effects of land use prior to tree planting was analyzed according to soil clay content and years since forest planting was initiated. However, given the limited amount of data for bare soil (four comparisons from one study), this land use was not considered in the analysis. Also, for agricultural land uses (clayey soils and >10 years after tree planting), the small number of studies did not allow a suitable comparison.

2.3. Results

2.3.1. Infiltration Recovery after Tree Planting under Different Prior Land Uses

Our review included eleven studies, yielding 67 comparisons (Appendix A), which were distributed across eight countries. Five studies were conducted in Asia, followed by South America, Africa and Oceania, each one with two studies. In those studies, annual rainfall ranged from 730 to 5,610 mm. The average infiltration values in restored forest was 434 $mm.h^{-1}$, varying from 11 to 2,592 $mm.h^{-1}$. The most studied soil types were the young soils Inceptisols, with four studies, followed by the highly weathered Oxisols and Ultisols, with three and two studies respectively. Additionally, two studies were conducted on fertile soils (Alfisols), one study on a volcanic soil (Andisol) and one study on a soil with low permeability (Vertisols). Most studies (75%) planted monocultures whereas a small number (25%) planted diverse tree arrangements. Monocultures commonly involved *Pinus* sp. species and *Tectona grandis*. Conversely, diverse plantings involved mixes of two (Perkins et al., 2012), five (Gageler et al., 2014), and more than 120 species (Zwartendijk et al., 2017). Field methods to determine infiltration were predominantly disc permeameters and double-ring infiltrometers, also, single-ring infiltrometers were used in two studies and a mini disk infiltrometer in one study (Figure 1).

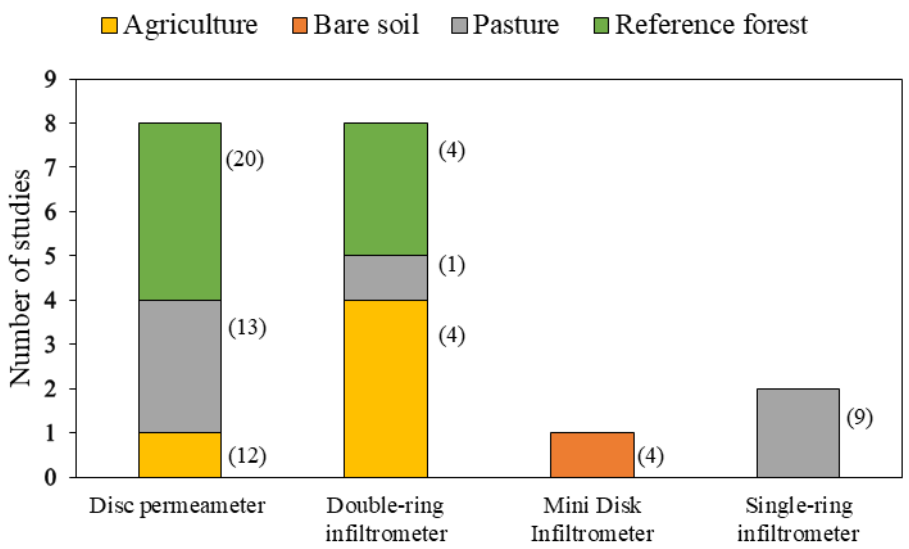

Figure 1. Number of **s**tudies by field method used to measure infiltration after tree planting in soils with different land use. Number of comparisons in each category are indicated in parentheses.

Most studies (91%) reported an increase in infiltration after tree planting, and only one study reported lower infiltration in restored forest when compared to pasture sites (Ghimire et al., 2014). Changes in infiltration after tree planting ranged widely and varied depending on prior land use (Figure 2). Overall results showed that infiltration values in the restored forests were 39% lower than in reference forests, but, higher than in agricultural sites (291%), pastures (182%) and bare soil (57%).

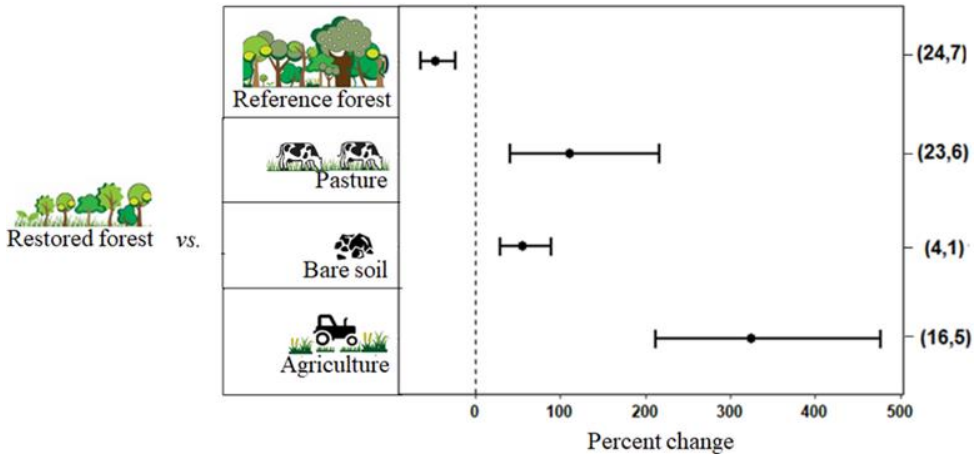

Figure 2. Effects of tree planting on infiltration showing the comparisons between restored forests and different prior land use types. Numbers in parentheses indicate the sample size followed by the numbers of studies. The percent change is significantly different from zero (vertical dashed line) if the 95% confidence interval does not overlap with it.

2.3.2. Effect of Soil Texture and Restoration Age on Infiltration after Tree Planting under Different Prior Land Uses

Results showed a trend for soils with higher clay content to exhibit higher infiltration (Figure 3). When compared with the reference forest, the restored forests had significant lower infiltration values in sandy soils (-66%), but not significant difference in clayey soils (mean percent of -24%). A similar situation was found when comparing restored forest and pastures, where clayey soils presented significant higher infiltration (302%) than sandy soils (90%). Additionally, in the former agricultural sites, tree planting in sandy soils showed a positive trend to improve infiltration (305%).

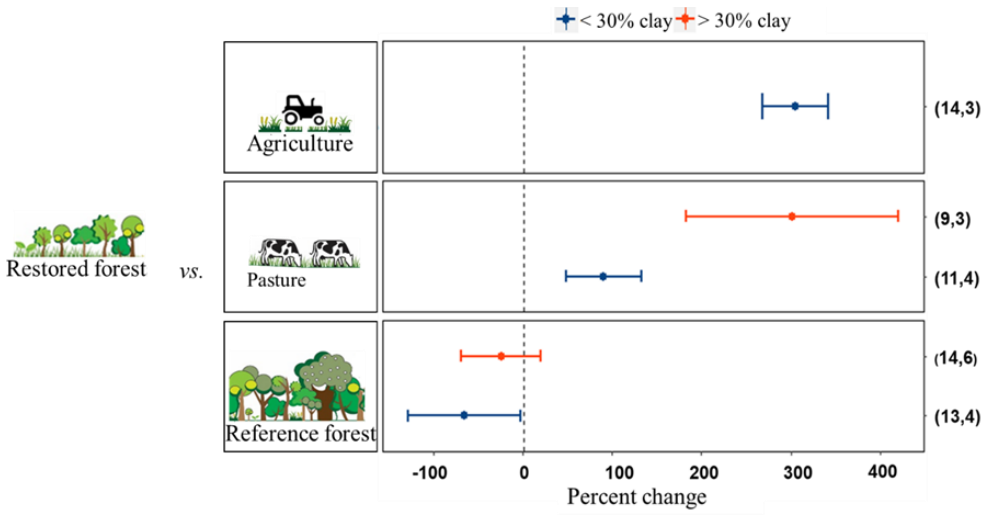

Figure 3. Changes in infiltration after tree planting under different prior land use types grouped by soil clay content. Results for clayey soils in agricultural sites are not shown due to no available data. Numbers in parentheses indicate the sample size followed by the numbers of studies. Error bars represent 95% confidence intervals. The percent change is significantly different from zero (vertical dashed line) if the 95% confidence interval does not overlap with it.

Mean restoration age was 12 years, ranging from three to 36 years. In general, older restored forests showed higher infiltration values than younger restored forests, indicating a clear effect of restoration age on infiltration recovery. When compared with the reference forest, infiltration levels as determined by the LRR were not significantly different in older and younger restored forests, with mean infiltration 32% and 63% percent less than the reference forest respectively (Figure 4). A similar relationship was observed in the comparison of restored forests versus pastures, where the mean percent increase in infiltration was 228% in the older restored forests and 82% in the younger restored forests. In the case of the comparison between younger restored forests and agriculture the mean percent increase in infiltration was 330%.

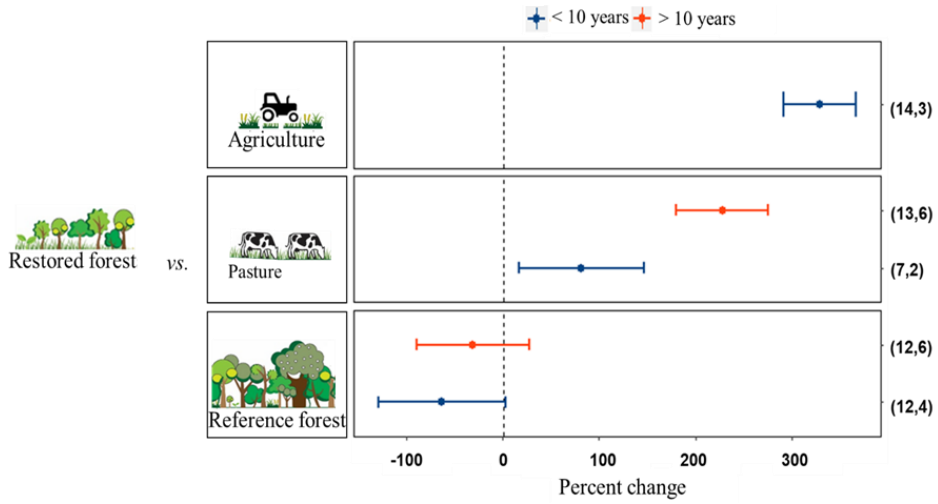

Figure 4. Changes in infiltration after tree planting under different prior land use types and grouping by years since forest planting was started. Results for restored forests > 10-year-old in agricultural sites are not shown due to no available data. Numbers in parentheses indicate the sample size followed by the numbers of studies. Error bars represent 95% confidence intervals. The percent change is significantly different from zero (vertical dashed line) if the 95% confidence interval does not overlap with it.

2.4. Discussion

2.4.1. Infiltration Recovery after Tree Planting under Different Prior Land Uses

Our systematic review shows that tree planting during forest restoration has positive effects on infiltration in the tropics. This result is consistent with several meta-analyses in a wide range of ecosystem types analyzing other hydrological issues (Ilstedt et al., 2007; Filoso et al., 2017; Sun et al., 2018). However, these finding should be interpreted with caution because most of the studies examined in our systematic review were conducted in planted monocultures with little information for diverse planting. Furthermore, in our systematic review only Ghimire et al. (2014) reported a restored forest presenting similar infiltration values than pasture sites. In this case, the restored forest was 25-year-old and used *Pinus* species (mainly *P. roxburghii* and *P. patula*) in Nepal, moreover, tree planting did not improve infiltration due to removal of litter, grazing and fuelwood harvesting that decreased the inputs of organic matter in the restored forest.

Thus, infiltration recovery after tree planting is affected by factors such as; soil structure, the increase of pore connectivity, root turnover, inputs of organic matter and growth of vegetation (Godsey and Elsenbeer, 2002; Ziegler et al., 2004).

Our results are in line with previous studies showing that infiltration recovery is strongly dependent on the prior land use before tree planting (Ziegler et al., 2004; Zimmermann et al., 2006; Hassler et al., 2011; Zwartendijk et al., 2017; Lozano-Baez et al., 2018). For example, when agriculture was the prior land use a higher degree of infiltration recovery was observed in restored forests. Similarly, Ilstedt et al. (2007) found that conversion from agriculture to restored forest in the tropics increased infiltration approximately three-fold. In contrast, trees planted into bare soil and pasture site produced lower infiltration recovery. In the case of conversion from bare soil to restored forest, the interpretation of results is limited to one study with four observations, thus, our focus will be to compare infiltration responses with agriculture and pasture as prior land uses. One possible explanation in the marked differences between agriculture and pastures is the increased soil compaction expected due to grazing animal traffic, where the pressure exerted by the animals can be higher than the pressure exerted by farm machinery (Silva et al., 2003). Furthermore, high soil compaction levels have been associated with a reduction in infiltration in several studies (Martínez and Zinck, 2004; Hamza and Anderson, 2005).

In all cases infiltration after tree planting was not fully recovered to reference forest levels. This result could be associated with the preservation of natural soil conditions in reference forests, such as better soil structure, higher organic matter, higher macroporosity, greater soil faunal activity, higher plant diversity and more complex root systems that benefit the infiltration process (Bruijnzeel, 2004; Leite et al., 2017). The increase of infiltration after tree planting could be related to plant species and individual tree effects. A recent experiment on natural savannas in West Africa (Ilstedt et al. 2016) found four-fold higher infiltrability under trees than open areas, suggesting that an intermediate tree cover could maximize groundwater recharge and infiltration. In this sense, tree densities and many plant characteristics such as; root architecture, shade and litter could influence infiltration. However, our review found a lack of studies dealing with species and tree effects on infiltration in restored forests. There are many studies in natural ecosystems investigating soil-water dynamics (Oliveira et al., 2005; Cooper et al., 2012; Oliveira et al., 2014; Zenero et al., 2016). This knowledge will be crucial to apply in forest restoration. It is equally important to realize the soil impacts that forest restoration could be having during tree planting. For example, if the trees are planted by hand or shovel the disturbance is minor, but if tree planting with bulldozer or tractor is used then there is much more disturbance and compaction, affecting the water infiltration negatively (Löf et al., 2012).

2.4.2. Effect of Soil Texture and Restoration Age on Infiltration after Tree Planting under Different Prior Land Uses

Results from the available studies indicated a clear effect of soil texture on infiltration capacity, showing differences in infiltration recovery between sandy and clayey soils. In the first case, sandy soils tend

to have a low water retention capacity and a high water infiltration (Oades, 1993; Regelink et al., 2015). The aggregating forces in these soils are weak, this is due mainly to the high sand content, the low biological activity and low organic matter content (Fisher and Binkley, 2000). Consequently, it is possible to expect in sandy soils a slower recovery of infiltration after tree planting (Ampoorter et al., 2007). On the other hand, clay soils with smaller soil particles (e.g., clay and colloidal particles), interact intensively with cementation agents (e.g., organic matter and iron oxides) improving soil aggregation and increasing the shrink-swell forces that, positively affect the formation of soil structure (Fisher and Binkley, 2000; Regelink et al., 2015). Recovery of soil structure and its stability in clayey soils is essential for the faster recovery of infiltration after tree planting (Oades, 1993; Ampoorter et al., 2007; Regelink et al., 2015; Basche and DeLonge, 2017), as was evident in the comparisons between restored forests versus pastures and reference forests.

Changes in infiltration after tree planting in forest restoration are related to time (restoration age), thereby, several studies have suggested that to achieve the pre-disturbance soil conditions it will probably take more than a decade (Zimmermann and Elsenbeer, 2008; Bonell et al., 2010; Ghimire et al., 2014). For example, Bonell et al. (2010) reported a small increase in infiltration during the first 10 years in *Acacia auriculiformes* planting in India, however, infiltration values remain low when compared with reference forests. Ziegler et al. (2004) argued that infiltration recovery is a process that occurs gradually over the years, with lower values in the beginning that will increase with the maturation of the forest. This remark is corroborated with our results, comparing the younger restored forests and older restored forests. The fact that infiltration in younger and older restored forests did not produce significant differences with reference forests highlights that the first decade of forest planting could recover the infiltration to a similar pre-disturbance level. However, it will probably take decades to achieve a full infiltration recovery. In addition, the observed trend between older restored forests and reference forests showed an infiltration recovery, but the infiltration values are still far from the reference forests, indicating that to achieve full infiltration recovery will be difficult. On the other hand, our results reflect the short duration of most studies (12 years of mean restoration age) and as Zimmermann and Elsenbeer (2008) highlighted, the recovery process is still insufficiently understood and multi-decadal effects of tree planting has not been captured in most of the studies (Bonell et al., 2010). Furthermore, in the literature the scarcity of long-term studies measuring infiltration after tree planting is a limitation that need further research.

2.5. Conclusions

2.5.1. Final Remarks and Future Research Directions

Our study reinforces the conclusion that infiltration is likely to increase after tree planting in forest restoration areas in the tropics. However, it must be noted that the available scientific evidence is severely limited. Based on this systematic review we have identified six knowledge gaps requiring future research efforts to advance our understanding of infiltration after tree planting in forest restoration:

1) Although infiltration is an important indicator of the possible pathways that water could take in the soil after rainfall, there is a need to study the water movement through the soil profile. Only a few studies in restored forests measured water percolation in depths greater than one meter. Moreover, most of the studies only considered saturated soil conditions in the infiltration measurements with little information on unsaturated soil conditions. Thus, tension infiltrometer measurements are recommended to describe the infiltration in these soil conditions (Salako et al., 2001).

2) The interactions between infiltration and other soil attributes are poorly understood. Therefore, more experiments in restored forests should be performed, considering detailed information about soil structure (e.g., soil porosity), also taking into account the influence of soil faunal activity on infiltration. In this sense, there are studies in natural ecosystems that would allow a possible comparison with restored forests (Juhász et al., 2006; Cooper et al., 2012; de Pierre Castilho et al., 2016; Zenero et al., 2016).

3) Details on the effects of forest restoration techniques and the level of disturbance during tree planting on infiltration are severely lacking. For example, the role that soil preparation, plant species, diversity of plants or tree densities could have on infiltration is virtually unknown. Indeed, most studies that measured infiltration were conducted in monoculture plantings, missing information for diverse plantings and native species. Therefore, more research comparing infiltration between different restoration techniques, considering the level of disturbance during tree planting, the effect of plant species and tree cover per se should be conducted.

4) Many studies show that land use history has a great influence on infiltration recovery after tree planting (Zimmermann et al., 2006; Hassler et al., 2011), however, the majority of these studies are made without considering the type, duration, and intensity of land use prior to forest restoration. Thus, future research should examine this issue.

5) The scarcity of long-term studies is a limitation. It is important to highlight that most studies that measured infiltration were short-term experiments (mean restoration age 12 years). This could be problematic and misleading when in some cases the findings in short-term experiments are extrapolated to long-term evaluations.

6) There is scant information about recovery rate of infiltration after tree planting. The great majority of studies have focused in one measurement over time. Our results suggest that monitoring of infiltration in restored forests should be made at fixed time intervals, in order to understand the temporal variability of infiltration recovery.

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3. PREVIOUS LAND USE AFFECTS SOIL HYDRAULIC PROPERTIES RECOVERY AFTER FOREST RESTORATION

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Lozano-Baez, S., Cooper, M., Ferraz, S., Ribeiro Rodrigues, R., Pirastru, M., and Di Prima, S. 2018. "Previous Land Use Affects the Recovery of Soil Hydraulic Properties after Forest Restoration". *Water*, 10(4):453.

Abstract

Knowledge of soil hydraulic properties after forest restoration is essential to understand the recovery of hydrological processes, such as water infiltration. The 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 at 18 plots under three land use types: (i) a pasture; (ii) a restored forest using high-diversity-mix of plantings (85 native species) of 9 years of age; and (iii) a remnant forest patch, in Campinas municipality, São Paulo State, Brazil. We used the Beerkan method for the 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 use classes. The highest K_s were observed in remnant forest sites and the lowest K_s were associated with pastures 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: Beerkan method; Saturated soil hydraulic conductivity; Soil properties; Water infiltration

3.1. Introduction

The global forest restoration movement based on natural regeneration and tree plantations has increased the tropical forest cover [1,2]. Nevertheless, soil hydraulic properties responses in these restored forests are virtually unknown [3,4]. Infiltration is a key hydrological process, which, among others, influences groundwater recharge, soil erosion and surface runoff. Indeed, one of the best parameters to understand and study 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 depend on many factors, such as soil types, land uses, soil depths, landscape positions, methods of measurement, physical and chemical soil attributes [6]. Despite this variation the K_s is a useful and sensitive indicator of the effect of land use 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, remaining approximately 12% of the original area [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]. Though, most studies on K_s recovery after forest restoration in tropical soils are emphasized in areas with natural regeneration or secondary succession [4,7,14–19]. Zimmerman et al. [17] at surface and near-surface (12.5 and 20 cm soil depth) in Brazilian Amazônia found non-significant K_s recovery 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 with 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 to recover the soil 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) in the 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 uses, namely pasture, 9-year-old restored forest and remnant forest. From our best 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 uses similar?

3.2. Materials and Methods

3.2.1. 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 (2,800 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 uses are: native vegetation (33%), pasture lands (30%) and crops (17%), as well as the forest cover increased 5.7% in the last decade [24]. The mean annual precipitation is 1,700 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 Atlantic Plateau are mainly composed of granites and gneises, while the Peripheral Depression is characterized by sedimentary rocks. The elevation ranges from 600 to 900 m with an undulating topography and the presence of slopes higher than 20% [27].

3.2.2. Experimental Design

The sites were selected to capture variation in soil attributes. Also, the sites accessability was taken into account in this selection. We examined the following land use 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). The maximum separation distances among these sites was 15 km in a straight line.

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 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 x 7 m in size), resulting in 18 plots altogether. Detailed characteristcs of the three land use 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) were characterised by a dense cover of grass. The dominant grass species is *Urochloa brizantha*. Information obtained with 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 high-diversity-mix of seedlings (85 native species), aimed to provide economical insurance and ensure successional processes to 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 x 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 differ between restored forest sites (Appendix B). 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 commercial purpose, although a frequent grazing of cattle occurred, then was harvested and grazing continued one year before the forest restoration. The eucalyptus harvest was made by motor-manual operations and farm tractor forwarded the logs. The vegetation in the Restored Forest R3 prior to restoration activities consisted in low shrub and grasses. On the other hand, the Restored Forest R4 after the coffee plantation was used as pastures for livestock breeding until 1986, subsequently was planted again to coffee and agricultural terraces were made with heavy track machinery. Then, the coffee plantation was replaced by pasture in 1996, which was similar to pastures sites (P1 and P2), dominated by grass species *U. brizantha* and without natural regeneration.

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

3.2.3. Soil Sampling and Measurements

The first field campaing 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 classified

according to the 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 to 0.05 m to determine soil macroporosity (*Mac*) and microporosity (*Mic*), using the Richards pressure chamber with application of 6kPa suction [34].

Infiltration measurements were taken in a second field campaing 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 ring 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 to 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 use, 21 for each site and 7 for each plot.

3.2.4. 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 to obtain a higher success percentage of the infiltration runs, as compared with other possibles 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]:

$$K_{sB} = \frac{C\, i_s}{A\, b_s + C} \qquad (1)$$

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]:

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

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

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], 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 failure 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]:

$$K_{sS} = \frac{i_s}{\frac{\gamma\gamma_w}{r\alpha^*} + 1} \qquad (4)$$

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 of these choice were 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].

3.2.5. Data Analysis

Data sets were summarized by calculating the mean and the associated coefficient of variation (CV). Following similar investigations [37,52], a unique value of clay, silt, sand, *OC*, ρ_b, total porosity, macroporosity, microporosity and θ_i was 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 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, e.g., 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].

3.3. Results

3.3.1. 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 to 35.37 g Kg^{-1} under Pastures (P1 and P2), from 10.46 to 24.60 g Kg^{-1} under Restored Forests (R3 and R4) and from 17.53 to 48.59 g Kg^{-1} under Remnant Forests (F5 and F6). The ρ_b values ranged between 1.12 and 1.40 g cm^{-3} in the pastures, for the restored forests the values ranged from 1.09 to 1.52 g cm^{-3}, while in the remnant forests the values ranged from 0.88 to 1.25 g cm^{-3}. The *Pt* varied from 0.47 to 0.58 cm^3 cm^{-3} in the pastures, from 0.43 to 0.59 cm^3 cm^{-3} in the restored forests and 0.53 to 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 and 0.37 cm^3 cm^{-3} and the soil was significantly wetter in plots P2M, R4U and R4M (Table 1).

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^3cm^{-3}), macroporosity (*Mac* in cm^3cm^{-3}), microporosity (*Mic* in cm^3cm^{-3}) and initial volumetric soil water content (θ_i in cm^3cm^{-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	19.6[a]	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	27.4[a]	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	17.4[a]	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	1.27[a]	1.24a	1.22a	1.29a	1.18b	1.33a	1.23a	1.29a	1.22a	1.33a	1.34a	1.42[a]	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	0.53[a]	0.53a	0.54a	0.51a	0.55a	0.50a	0.54a	0.51a	0.54a	0.50a	0.50a	0.47[a]	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	0.16[a]	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	0.36[a]	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	0.32[a]	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 (e.g., P1, P2, R3, R4, F5 and F6), means that do not share a letter are significantly different according to the Tukey test ($p = 0.05$).

3.3.2. Estimating and Selecting the BEST Algorithm

Overall the Beerkan method used in this study was found to be robust to measure the K_s in the field. However, the BEST-steady algorithm yielded physically plausible estimates (e.g., positive K_s values) in 108 of 126 infiltration runs (85.7% of the cases). The percentage of successful runs was 40 of 42 runs (95.2%) both in the pasture sites and restored forest. With reference to the Remnant Forest (F5 and F6), BEST-steady led to failure rate value of 33.3%, leading to 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 (e.g., positive K_s values).

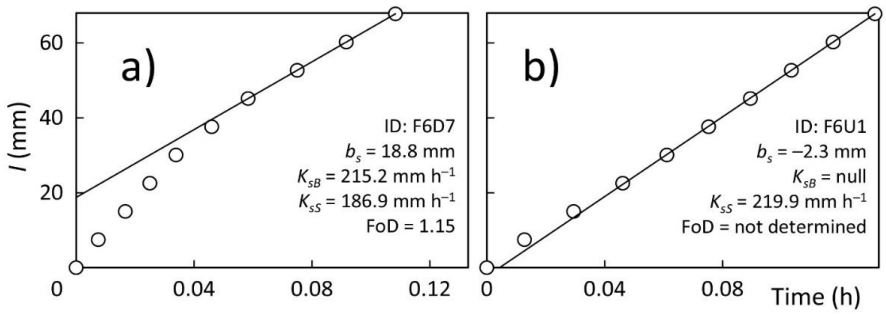

Figure 2. Illustrative examples of the influence of the shape of the cumulative infiltrations on the discrepancies occurring between BEST-steady and the SSBI method. (**a**) Concave-shaped cumulative infiltration curve in which the intercept, b_s (mm), of the straight line interpolating the last I vs. 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}.

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 (e.g., 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 the 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.

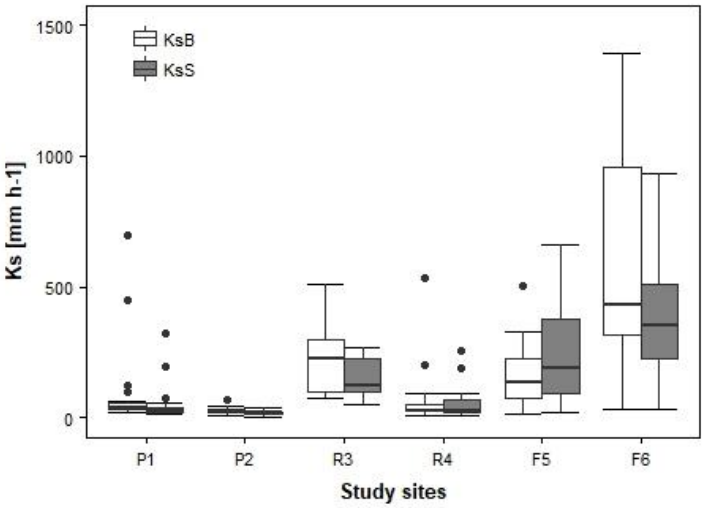

Figure 3. Comparison between K_s estimated with BEST-steady, K_{sB}, and the SSBI method, K_{sS}. Study sites are abbreviated with P1 and P2 for pasture, R3 and R4 for restored forest and F5 and F6 for remnant forest.

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 a 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 and 934 mm h^{-1}, with a high variability inside all study sites.

3.3.3. 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 pastures plots, for example, in Pasture P1 the K_s ranged from 10 to 320 mm h^{-1}, and in pasture P2 K_s ranged from 4 to 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 to 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 to 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 to 256 mm h^{-1} and average of 54 mm h^{-1}) was significantly different from the Remnant Forest Plot F5M (average of 387 mm h^{-1}), furthermore, all the plots in the Restored Forest R4 had low K_s variability, similar to the pasture land use. Finally, clay loam and loam textures showed the same comparison among land uses, characterized by marked differences between lower K_s in the 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 to 934 mm h^{-1}) in the study sites (Figure 4).

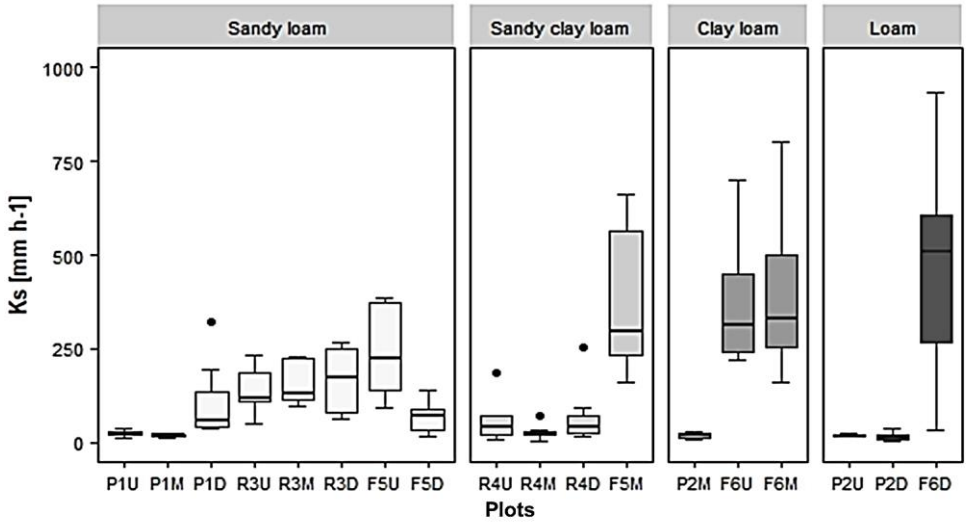

Figure 4. K_s estimated with SSBI method, K_{sS}, per plots and grouping by soil texture (USDA classification system). Study sites are abbreviated with P1 and P2 for pasture, R3 and R4 for restored forest, and F5 and F6 for remnant forest. The subscript letter refers to the landscape position (Upslope,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

3.4. Discussion

3.4.1. 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 use 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 factor for K_s variability over time [15,17], as well as the cattle grazing intensity could have influenced the K_s variability in the pasture plots [7]. Otherwise, the lower soil faunal activity and organic matter in pasture land uses are important factors when analyzing the soil attributes [15,56]. Especially, 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 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 pasture plots, which could be attributed in part to the higher sand contents 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), 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 the 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 abandoned eucalyptus forest with a canopy structure for 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, 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 Pastures Sites (P1 and P2), considering the low spatial K_s variability observed in the pasture land use. Thus, the Restored Forest R4 showed no significant differences in K_s with 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, lead to greater soil exposure, also, trampling pressure and 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 decrease with increasing land use intensity, as well as 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 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 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 the 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 pastures plots in the sandy loam, clay loam and loam textures allowed to detect 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, 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 to develop a better soil structure and store 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 of Hassler et al. [7], who attributed the K_s variability in Panama forest soils to overland flows that results in erosion [19]. Other factors that might have caused the K_s spatial variability in remnant forest plots are the steepness of sample plots and the soil distribution in the landscape positions (U, M, D) influenced by contrasting slope and topography.

3.4.2. 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 include collection of soil compaction and K_s data to understand the initial compaction degree and water infiltration, reinforcing the need to compare these values with the pre-disturbance soil conditions. After assessing soil compaction and infiltration at the restored forests, management practices could be implemented to alleviate soil compaction, such as mechanical loosening techniques (e.g., deep ripping and subsoiling), which may improve plant growth [73,74]. In addition, some technical methods in forest restoration that have shown to aid the natural regeneration and soil recovery are: suppressing 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 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 introduction of silvopastoral systems during the pasture management. Moreover, there are an increasing number of reports regarding different tropical land uses, suggesting that lowers 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 the pastures and restored forests to avoid the soil erosion, conserve water and create opportunities to enhance water infiltration [75].

3.5. Conclusions

In this study, the hypothesis that the 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 use classes: pasture, restored forest and remnant forest. They also bring out the inverse relationship between K_s and ρ_b, where the K_s increase as a result of a decrease in ρ_b, consequently, the dominance of macropores over micropores, which facilitate the water infiltration.

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4. RECOVERY OF SOIL HYDRAULIC PROPERTIES FOR ASSISTED PASSIVE AND ACTIVE RESTORATION: ASSESSING HISTORICAL LAND USE AND FOREST STRUCTURE

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Abstract

Tree planting and natural regeneration are the main approaches to achieve global forest restoration targets, affecting multiple hydrological processes, such as infiltration of rainfall. Our understanding of the effect of land use history and vegetation on the recovery of water infiltration and soil attributes in both restoration strategies is limited. Therefore, we investigated the recovery of top-soil saturated soil hydraulic conductivity (K_s), soil physical and hydraulic properties in five land use types: (i) a secondary old-growth forest; (ii) a forest established through assisted passive restoration 11 years ago; (iii) an actively restored forest, with a more intensive land use history and 11 years of age; (iv) a pasture with low-intensity use; and (v) a pasture with high-intensity use, in the Brazilian Atlantic Forest. For these land use types, we determined the historical land use patterns and conducted soil sampling, using the Beerkan method to determine K_s values in the field. We also measured tree basal area, canopy cover, vegetation height, tree density and species richness in forest covers. The K_s decreased when land use was more intense prior to forest restoration actions. Our results indicate that land use legacy is a crucial factor to explain the current difference in soil and vegetation attributes among study sites.

Keywords: Beerkan method; Forest restoration; Infiltration; Natural regeneration; Pasture

4.1. Introduction

Forest restoration strategies are being implemented around the world through ambitious international (e.g., Bonne Challenge and New York Declaration on Forests), regional (e.g., Initiative 20×20 and AFR100) and national initiatives such as restoration plans in many countries [1]. Consequently, secondary forests have expanded in tropical regions [1,2]. In Brazil, the location of our study area, the "Atlantic Forest Restoration Pact" aims by 2050 to increase the current forest cover from 17% to at least 30%, with a restoration target of 15 million hectares [3]. These initiatives include both passive and active restoration strategies. Passive ecological restoration refers to spontaneous recovery of tree species in an ecosystem that has been damaged, while assisted passive restoration involves human interventions to assist natural regeneration [4,5]. This can include introduction of propagules and removal of invasive species and

persistent disturbances, for example, fire or livestock grazing [4]. On the other hand, active restoration requires a higher human intervention through planting of tree seedlings to accelerate the recovery process [6,7].

Both restoration approaches have been shown to impact positively the provision of ecosystem services, as well recovering biodiversity and ecosystem functions [8]. However, most restoration research around the world has focused on aboveground plant communities, whilst the belowground environment (e.g., soil physical and hydraulic properties) has been poorly studied [9,10]. For example, the response of the infiltration process, and soil physical and hydraulic properties after forest restoration is virtually unknown [11]. A crucial parameter in the infiltration process is the soil saturated hydraulic conductivity (K_s), which influences water percolation through the soil matrix [12,13]. It is well known from previous studies that K_s is highly variable compared to other soil physical properties [14,15]. In fact, the K_s depends strongly on the highly variable soil structure, and it is known to vary several orders of magnitude [16,17], especially on forested soils [18,19]. In general, K_s recovery and soil hydraulic properties have been reported separately in passive [20–24] and active [25–28] restoration, but few comparisons between both restoration strategies have been conducted. Lozano-Baez et al. [28] investigated the surface K_s recovery under a nine-year-old actively restored forest in the Atlantic Forest of Brazil and observed that the land use prior to forest restoration influences the K_s recovery. Moreover, the few recent comparisons between active and passive restoration show contradictory results. For instance, K_s at 12.5 cm depth in Brazilian Amazônia was higher under a 15-year-old passively restored forest than a 10-year-old tree plantation [11]. In contrast, other authors in Madagascar found much lower surface K_s in 2–10-year-old naturally regenerating fallow than actively restored forest of 6–9 years of age [29]. Most previous studies have assessed the recovery of soil physical and hydraulic properties without addressing the relationships among soil, vegetation and land use history. These relationships are fundamental to better understand the recovery process (e.g., resilience of the ecosystem) and successional trajectories after forest restoration [30,31]. Foster et al. [32] argued that the imprints of past land use on ecosystems may persist for decades to centuries. In particular, after forest restoration, such imprints of past land use on soil (e.g., K_s, soil physical and hydraulic properties) may persist for a time frame of more than a decade, as suggested by several studies [12,26,33]. However, the above-mentioned mechanisms and relationships that affect the recovery process are poorly understood.

As part of a larger research effort investing the effects of forest restoration on K_s, this study aimed to extend the work of Lozano-Baez et al. [28] at a new location. Apart from presenting new K_s data for pastures with different land use intensities and a secondary old-growth forest, this paper includes the first measurements of K_s for a forest established through assisted passive restoration in the Brazilian Atlantic Forest. We further quantified and compared the K_s, soil physical and hydraulic properties recovery of active vs. assisted passive restoration strategies from the same restoration program described by Lozano-Baez et al. [28]. We examined whether differences in land use history led to differences in these soil attributes (e.g., K_s, bulk density, soil organic carbon content, soil porosity, initial and saturated soil water content) and vegetation attributes (e.g., basal area, canopy cover, vegetation height, tree density and species richness). We studied five land-cover types: (i) a secondary old-growth forest, used as a reference forest (hereafter, RF);

(ii) a forest established through assisted passive restoration (hereafter, APR); (iii) an actively restored forest (hereafter, AR); (iv) a pasture with low-intensity use (hereafter, LiP); and (v) a pasture with high-intensity use (hereafter, HiP). In forest stands, we associated the recovery of K_s, soil physical and hydraulic properties with the vegetation attributes. We hypothesized that K_s would vary with intensity of land use in the past among land-cover types as follows: RF > APR > AR > LiP > HiP. As the AR site had a more intensive land use history, we expected that K_s recovery and vegetation attributes would be higher in the APR.

4.2. Materials and Methods

4.2.1. Study Area

The study area is located in the county of Campinas (22°53' S, 46°54' W), São Paulo State, Southeast Brazil (Figure 1). The climate in this region is classified as Cwa according to the Köppen classification mean annual precipitation is 1,700 mm and mean annual temperature is 20 °C, with dry winters and wet summers [34]. Our study sites are located at the transition between the Atlantic Plateau and the Peripheral Depression geomorphological provinces [35]. The soils are classified as Ultisols [36] and the original vegetation in this area is a seasonal semi-deciduous forest, belonging to the Atlantic Forest biome. This region is highly fragmented, because of 200 years of historical landscape changes [37]. In particular, our study area is located inside the sub-basin of the Atibaia River where the main land uses are native vegetation and pastureland, occupying 33% and 30% of the sub-basin, respectively. The native vegetation includes Atlantic Forest remnants with different sizes and ages [38].

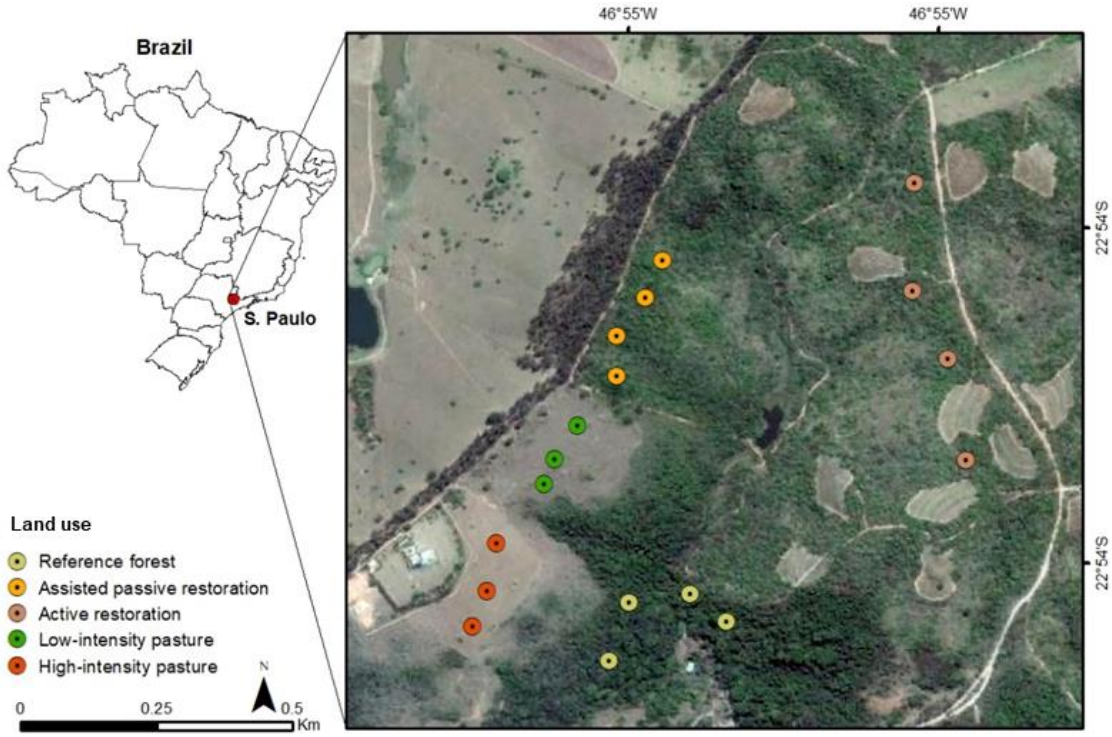

Figure 1. Location of the 18 study plots in the state of São Paulo, Southeast Brazil.

Within this area, we selected five land uses to measure soil physical and hydraulic properties, vegetation structure and diversity (Figure 2). In general, the deforestation of our study area already existed at the beginning of the 19th century, with the objective of introducing coffee (*Coffea arabica*) plantations. However, after the crisis in coffee cultivation during the early 20th century, the plantations were gradually replaced by pastures.

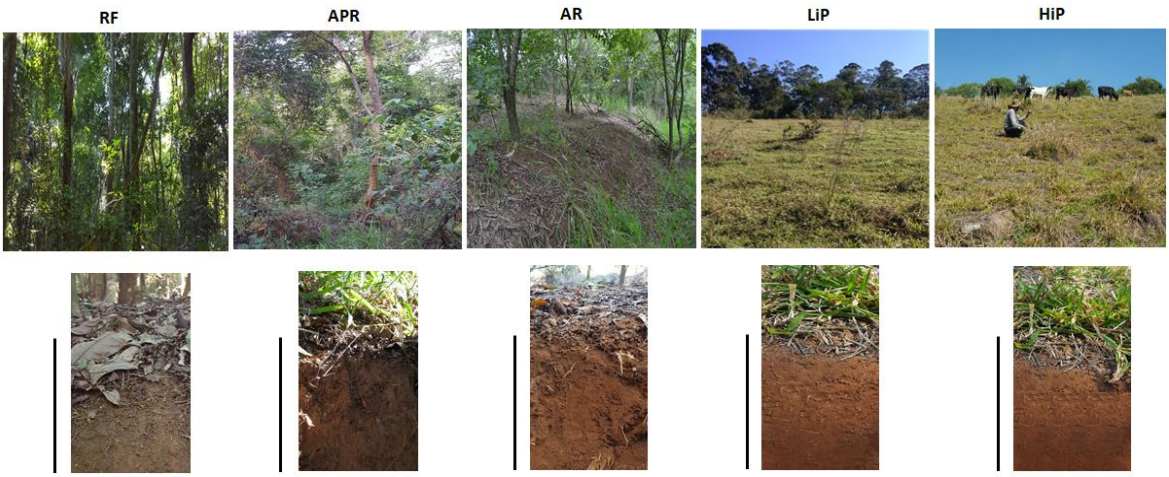

Figure 2. Pictures showing the vegetation cover and the Ultisol top-soil profile for each study site. The black lines in each top-soil profile represent 0.2 m scale. RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture.

Land use history for the study sites was reconstructed based on interviews with the local population and aerial photographs taken in 1968, 1978, 1994, 2005 and 2017. The RF site is a secondary

old-growth forest characterized by having the highest slope between the study sites, which was 28.8 ± 4.9% (SD). The slope was measured in the study plots with laser distance meter. Site RF was used as a control area to assess reference values for soil physical and hydraulic properties. According to interviews, in the early 20th century, this site was affected by natural fire disturbances and it was partially cleared at least once in the past for agricultural purposes. Moreover, aerial photographs showed that forest cover in most of the area was established and has increased since 1968. In this context, all RF plots were located in sites with forest cover in the last 40 years.

Site APR is located adjacent to the LiP. The slope (28.1 ± 2.8%) was similar to the RF. From 1968 to 1994, it was used for milk cattle grazing. Then, the area was abandoned and remained without a specific land use until 2007, leaving the forest to naturally regrow over 12 years. In 2007, to decide the best restoration strategy for the area, the "Diagnostic" protocol proposed by Rodrigues et al. [37] was implemented. This protocol allowed identifying the initial environmental situation and evaluating the potential of autogenic restoration of the area. Considering that this site evidenced favorable abiotic and biotic conditions (e.g., naturally regenerating native plants) for native plant establishment (Appendix C), the restoration diagnosis was of fair potential for autogenic restoration. Thus, forest restoration techniques included the encouragement of regenerating individual native trees and shrubs by manual and chemical control of invasive grasses. Moreover, enrichment plantings with native tree species were also implemented in patches without natural regeneration. In this regard, our measurements reflect the effect of 11 years of APR on a soil with a previous second-growth forest.

At AR site, the slope (22.8 ± 1.7%) was lower than the RF and APR. Initially, this site was used for dairy cattle grazing from 1968 to 1986. Later, it was replaced by coffee plantations until 1994. It is important to emphasize that, at the beginning of the coffee plantation phase, widespread terracing was implemented. Then, in 1994, the coffee was replaced by pastures with *Urochloa brizantha* for beef cattle grazing. In 2007, the "Diagnostic" protocol mentioned previously was implemented. Given that AR site evidenced very few spontaneously regenerating seedlings and degraded environmental conditions that limited the passive restoration strategy, the restoration diagnosis in this area was of very low potential for autogenic restoration. Thus, AR was implemented through a restoration model that aimed to provide economical insurance and ensure successional processes to landowners [37]. Restoration plantings were implemented as mixed plantation with high-diversity-mix of seedlings (>50 native trees species). During the planting, these species were organized in fourth groups (e.g., initial, filling, middle and final species) according to the rate of growth and commercial value. Initial species (e.g., *Croton floribundus*, *Senegalia polyphylla* and *Schinus terebinthifolius*) can be harvested for fuel production in 10–15 years, and are characteristically fast-growing, providing fast soil coverage and beneficial initial conditions for other species growth. Filling species (e.g., *Croton urucurana*, *Gochnatia polymorpha* and *Trema micrantha*) are also fast-growing species planted in the same line as the species. Middle species (e.g., *Astronium graveolens*, *Gallesia integrifolia* and *Machaerium stipitatum*) can be harvested during Years 20–25 and are more valuable wood species that will replace the initial and filling species. Final species (e.g., *Aspidosperma polyneuron*, *Cariniana estrellensis* and *Cariniana legalis*)

are narrow canopy and slow-growing species that can be harvested during Years 40–45 for luxury and finished carpentry. The species planted are listed in Appendix D. The total density of seedling was 1,660 ind·ha^{-1}, in a 3 m × 2 m spacing, using mechanized soil preparation. Before planting, invasive grasses were controlled through herbicide application. Fertilizers and irrigation were applied at the time of planting and during the first year [37,39]. The initial environmental conditions for AR are presented in Appendix E. As a result, our measurement in this restoration site represent the effect of 11 years of active restoration on highly degraded soil, with an intense land use history.

Site LiP with a slope of 22.7 ± 2.1% is located adjacent to the HiP, and both sites share a similar land use history until 2008. Since this year, in the LiP, grazing has been intermittent and with low productivity (e.g., stocking rate lower than two livestock units per hectare). During our field campaign, the vegetative cover in the LiP was dominated by the same grass species (*U. brizantha*), with a mean height about 50 cm and isolated native trees, shrub species and non-native grasses scattered in the area were also evident (Figure 2). Consequently, our results reflect the influence of 40 years of grazing, with a lower land use intensity in the last decade.

At site HiP, the slope was 23.3 ± 3.2%. This site was covered by a coffee plantation until 1968. Afterwards, the coffee was replaced by pasture, planting *U. brizantha* as grass species. Since 1978, this area has been heavily grazed with dairy cattle, supporting a stocking rate greater than two livestock units per hectare, with regular application of fertilizers and other inputs. As a result, our measurements at this site represent the effect of 40 years of continuous grazing.

A graphical summary of the land use history for the five land uses described previously is provided in Figure 3.

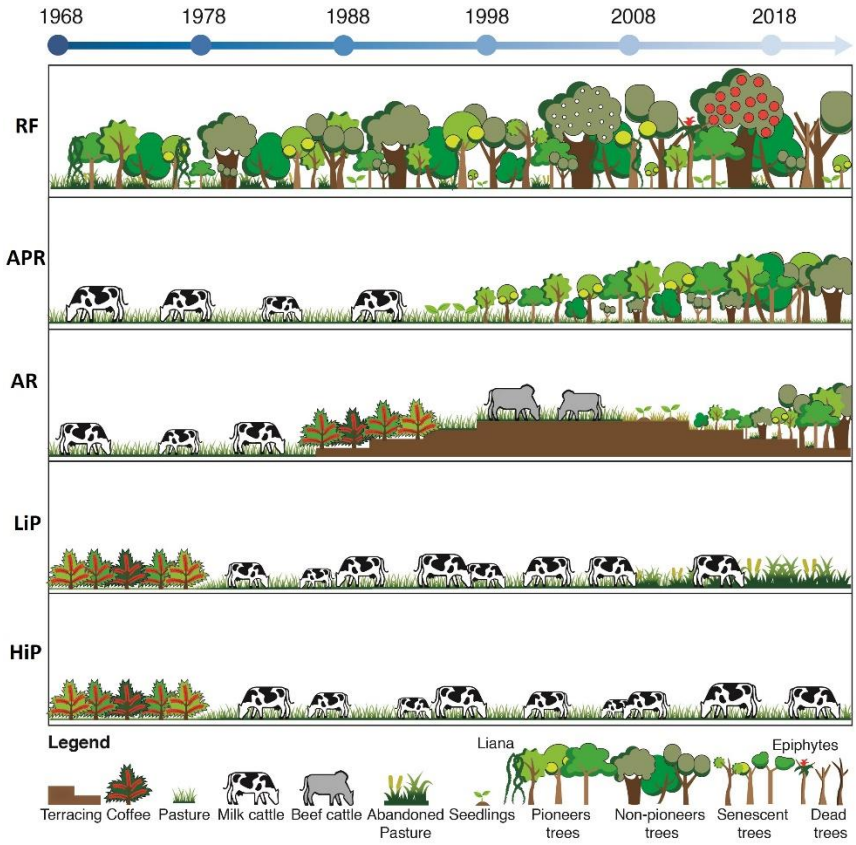

Figure 3. Land use history for each land use type. RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture.

4.2.2. Experimental Design

The study sites were located in a similar landscape position along the hillslope gradient and were selected to have the same soil type following Zwartendijk et al. [23]. In forest stands, we established four plots, and for the pasture sites three plots were established. Sampling the same number of plots per land uses was impossible due the restricted accessibility in pasture sites, resulting in 18 plots altogether (Figure 1). For sampling vegetation and soil attributes, the size of each plot was 500 m^2 (50 m long and 10 m wide), a total area of 2,000 m^2 for each site. The size and number of plots were chosen according to similar investigations aimed at evaluating vegetation structure and composition in tropical forest restoration projects [40–42].

4.2.3. Vegetation Sampling

Vegetation sampling was conducted from September to November 2017 in the RF, APR and AR plots. In each 500 m^2 plot, we identified and sampled all living trees and shrubs with height ≥ 50 cm and diameter at breast height (DBH) > 5 cm. Additionally, we installed a 200 m^2 (50 m long and 4 m wide) subplot at the center of each plot, to identify and measure all trees and shrubs with DBH < 5 cm and height ≥ 50 cm. For all sites considered in this investigation, we measured the following vegetation attributes: (1) tree basal area; (2) canopy cover; (3) vegetation height; (4) tree density; and (5) species richness. These are

key ecological indicators, useful to evaluate vegetation structure and composition in tropical forest restoration projects, also, they are being recommend in Atlantic Forest monitoring protocols [42,43]. As suggested by Viani et al. [42], the percentage of canopy cover in each plot was measured by an adaptation of the line interception method [44], installing a 50 m line transect in the middle of each study plot. Vegetation height was measured with a 5 m measuring stick, and the remaining height of trees taller than this was estimated visually. As suggested by Suganuma and Durigan [40], for tree density, we analyzed: (1) density of trees (DBH > 5 cm); and (2) density of saplings (DBH 1–5 cm). In the same way, for species richness, we analyzed: (1) total richness (all individual sampled); (2) overstory richness (DBH > 5 cm); and (3) richness of saplings (DBH 1–5 cm). For all previous ecological indicators, we calculated the mean values per study site. In addition, for all sampled individuals we classified the species origin as: native or nonnative to the study region. Specifically, in the AR site, we evaluated planted tree mortality.

4.2.4. Soil Sampling

Soil sampling was conducted during the dry season in April 2018. In the middle of each plot, we installed a 50 m transect along the hillslope gradient. At intervals of 15 m, three disturbed soil samples at 0–5 cm depth were collected. Before soil sampling, the litter and a small layer of soil (e.g., organic horizon) of less than 1 cm was removed. We determined the soil particle size distribution (PSD) with sand particle separation, the particle density (*Pd*) and soil organic carbon content (*OC*). The PSD analyses were carried out according to the hydrometer method [45]. Then, soil textures were classified following the USDA standards. The *Pd* was determined using the helium gas pycnometer method [46], and the *OC* analyses were performed following the Walkley–Black method [47]. In the same transects, at 7 m intervals, soil hydraulic measurements were conducted. In the specific case of AR site, to minimize spatial variability and possible induced effects by tillage, we placed the seven sampling points in the inter-plant space of planting lines. At each sampling point, we performed the Beerkan method [48,49]. A total of 126 Beerkan experiments were carried out, using a steel ring with an inner diameter of 16 cm inserted to a depth of about 1 cm into the soil surface. In each infiltration point, a known volume of water (150 mL) was repeatedly poured into the cylinder at a small height above soil surface (e.g., a few cm) and the energy of the water was dissipated with the hand fingers to minimize the soil disturbance. Then, the time needed for complete infiltration was logged. This procedure was repeated until the difference in infiltration time between two or three consecutives trials became negligible. Following a procedure commonly used for Beerkan method, at the beginning of each infiltration run, and near the steel ring, we collected one undisturbed soil core (100 cm^3) at 0–5 cm depth to determine the bulk density (ρ_b) and the initial volumetric soil water content (θ_i). Saturated soil hydraulic conductivity (K_s) values were estimated by the Steady version of the Simplified method based on a Beerkan Infiltration run (SSBI method) [50]. According to previous a investigation carried out by Lozano-Baez et al. [28] on the same area, this method was chosen to avoid uncertainties due to a specific shape of the cumulative infiltration [51,52].

In addition, the undisturbed soil cores were used to determine total soil porosity (*Pt*), soil microporosity (*Mic*) and soil macroporosity (*Mac*). The *Pt* was calculated using ρ_b and mean *Pd* of each plot [53]. The *Mic* was estimated using a tension table with application of 6 kPa suction, and *Mac* was obtained by the difference between *Pt* and *Mic* [54]. Finally, according to Lassabatere et al. [48], at the end of each infiltration test, a disturbed soil sample was collected to determine the saturated gravimetric water content, and ρ_b was used to calculate the saturated volumetric soil water content (θ_s).

4.2.5. Data Analysis

The hypothesis of normal distribution of both the raw and the log-transformed data was tested by the Kolmogorov–Smirnov test. One-way analysis of variance (ANOVA) was performed with raw or log-transformed data. When ANOVA null hypothesis was rejected, we used multiple comparisons to detect differences between pairs by applying the Tukey's honestly significant difference test. The related *p*-values were computed and compared to the level of significance of 0.05. Alternative non-parametric tests (Kruskal–Wallis ANOVA) were used when even the log-transformed data were non-normally distributed. In this case, when ANOVA null hypothesis was rejected, multiple comparisons between pairs were made with the Bonferroni method (adjusted *p*-values). Variables means were calculated for soil attributes according to the statistical distribution of the data, e.g., geometric mean for log-normal distributions and arithmetic means for normal distributions [55]. According to Lee et al., 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 [56]. Pearson's correlation coefficient was calculated to identify correlation among the selected soil attributes: *Pt*, *Mic*, *Mac*, *OC*, θ_i, ρ_b and K_s across all study sites. Furthermore, to compare the soil and vegetation attributes among land use types, Principal Component Analysis (PCA) was performed on standardized variables. All statistical analyses were carried out using the Minitab© computer program (Minitab Inc., State College, PA, USA).

4.3. Results

4.3.1. Vegetation Attributes

A total of 541 saplings and 646 trees distributed in 38 families, 92 genera, and 138 species were sampled. For non-native species, we found 62 saplings and 147 trees, representing 11% and 23% of the total, respectively (Appendix F and Appendix G). Although the basal area and vegetation height of trees were much higher in RF, these did not differ statistically with both restored forests. The canopy cover showed significant differences in AR with RF and APR. We highlight the higher similarity between RF and APR for density of trees and saplings (Table 1), as a result of the high density of non-native trees *Psidium guajava* and *Tecoma stans*, which represented 24% and 5% of trees in APR, respectively. In contrast, non-

native trees in the RF and AR represented 18% and 16%, respectively, of all tree individuals sampled in each site. Additionally, the total richness, the density and richness of trees and saplings were markedly lower in AR, where 14% of planted trees (e.g., initial species) were dead yet there was a higher presence of grasses (e.g., *U. brizantha*) observed in all plots.

Table 1. Mean vegetation attributes (± standard error, n = 4) sampled in the forest stands. RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture.

	RF	APR	AR
Basal area ($m^2 ha^{-1}$)	26.4 ± 4.49 [a]	20.8 ± 2.53 [a]	12.5 ± 3.32 [a]
Canopy cover (%)	95.8 ± 2.17 [a]	91.3 ± 1.49 [a]	77.5 ± 3.11 [b]
Vegetation height of trees (m)	10.1 ± 1.16 [a]	7.79 ± 0.57 [a]	7.00 ± 0.11 [a]
Density of trees ($ind \cdot ha^{-1}$)	1,325 ± 137 [a]	1,300 ± 72 [a]	610 ± 72 [b]
Density of saplings ($ind \cdot ha^{-1}$)	3,950 ± 172 [a]	1,963 ± 959 [ab]	850 ± 119 [b]
Total richness (tree and non-tree)	82 ± 4 [a]	62 ± 1 [a]	38 ± 2 [b]
Overstory richness	50 ± 2 [a]	41 ± 1 [a]	30 ± 1 [b]
Richness of saplings	62 ± 2 [a]	39 ± 3 [b]	15 ± 1 [b]

Note. Different superscript letters denote statistically significant differences between land use types, according to the Tukey's test ($p < 0.05$), except for the basal area and vegetation height of trees where Kruskal–Wallis test ($p < 0.05$) was applied.

4.3.2. Soil Physical and Hydraulic Properties

The texture of the upper layers of the soil (0–5 cm) was clay loam in APR and pasture sites, while it was sandy clay loam in RF and AR sites. The clay content at the study sites ranged between 21% and 44%, but only the RF with lower values of clay differed significantly from the other study sites. Moreover, soil samples taken from the HiP showed the highest clay content. The silt ranged between 18% and 37%, with higher silt values in APR that differed significantly from other land-covers. The sand content varied between 31% and 55%, and was significantly lower in APR compared with other study sites (Table 2).

Table 2. Mean values for soil particle size distribution, and textural class according to the USDA classification of the depth 0–5 cm for each land use type. RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture.

Land Use	Clay (%)	Silt (%)	Sand (%)	Sand (%)					Textural Class
				Very Fine	Fine	Medium	Coarse	Very Coarse	
RF	24.8 [b]	25.9 [b]	49.3 [a]	6.23 [a]	14.0 [abc]	12.8 [a]	9.21 [a]	6.99 [a]	Sandy clay Loam
APR	30.2 [a]	31.9 [a]	37.9 [c]	6.28 [a]	12.1 [c]	9.11 [c]	5.54 [c]	4.84 [b]	Clay loam
AR	30.0 [ab]	23.9 [b]	46.1 [ab]	5.73 [a]	14.2 [ab]	11.7 [ab]	7.67 [b]	6.68 [a]	Sandy clay Loam
LiP	31.7 [a]	22.6 [b]	45.7 [ab]	6.60 [a]	15.1 [a]	12.6 [ab]	6.87 [bc]	4.44 [b]	Clay loam
HiP	33.6 [a]	23.2 [b]	43.1 [bc]	5.82 [a]	12.8 [bc]	10.7 [bc]	7.43 [b]	6.34 [ab]	Clay loam

Note. Number of soil texture samples: RF = 12, APR = 12, AR = 12, LiP = 9 and HiP = 9. Different superscript letters denote statistically significant differences between land use types, according to the Tukey's test ($p < 0.05$).

Comparisons of ρ_b values between study sites revealed higher similarity between RF and APR, while AR presented similar ρ_b values with both pasture sites. The *Pd* had similar values in all study sites, ranging from 2.61 to 2.71 g cm^{-3}. The *OC* varied significantly among sites (from 4.6 to 25.6 g kg^{-1}), with

higher values in HiP and markedly lower values in AR. The soil *Mac* ranged from 0.16 to 0.38 $cm^3\ cm^{-3}$, with greater values observed in RF, followed by the APR, AR, LiP and HiP. Similar results were obtained for *Pt*, which varied from 0.48 to 0.66 $cm^3\ cm^{-3}$. In contrast, the soil *Mic* (from 0.21 to 0.43 $cm^3\ cm^{-3}$) was much higher in pasture sites and decreased in forest land-covers, with lower values in RF and AR. In addition, the θ_i at the time of the Beerkan infiltration run varied between 0.12 and 0.32 $cm^3\ cm^{-3}$, with significant lower values in the RF. The θ_s varied between 0.29 and 0.75 $cm^3\ cm^{-3}$ with significant lower values in restored forests. The K_s ranged from 4 mm h^{-1} to a maximum of 1,121 mm h^{-1} among the study areas. The higher K_s was evidenced in APR, which was only similar with RF and significantly different from other three land uses. The K_s values obtained in the RF were lower than APR. In contrast, the K_s of AR between 15 and 1,121 mm h^{-1} was similar to RF and differed significantly for the other three land uses. In addition, across the five land uses, K_s values varied least and differed significantly from each other at the LiP and HiP (Table 3).

Table 3. Mean and associated coefficient of variation (CV, in parenthesis) of soil bulk density (ρ_b in g cm^{-3}), soil particle density (*Pd* in g cm^{-3}), soil organic carbon content (*OC* g kg^{-1}), saturated soil hydraulic conductivity (K_s in mm h^{-1}), microporosity (*Mic* in $cm^3\ cm^{-3}$), macroporosity (*Mac* in $cm^3\ cm^{-3}$), total soil porosity (*Pt* in $cm^3\ cm^{-3}$), initial volumetric soil water content (θ_i in $cm^3\ cm^{-3}$) and saturated volumetric soil water content (θ_s in $cm^3\ cm^{-3}$), of the depth 0–5 cm for each land use type. RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture.

Land Use	ρ_b	*Pd*	*OC*	K_s	*Mic*	*Mac*	*Pt*	θ_i	θ_s
RF	1.04 ^{b}	2.66 ^{ab}	16.2 ^{a}	215 ^{ab}	0.29 ^{ab}	0.32 ^{a}	0.61 ^{a}	0.18 ^{c}	0.48 ^{ab}
	(7.06)	(1.17)	(24.3)	(90.2)	(14.6)	(9.82)	(4.54)	(12.6)	(22.2)
APR	1.04 ^{b}	2.68 ^{a}	16.4 ^{a}	351 ^{a}	0.31 ^{bc}	0.29 ^{a}	0.60 ^{a}	0.24 ^{a}	0.45 ^{b}
	(6.50)	(1.11)	(21.4)	(58.4)	(9.12)	(9.18)	(2.53)	(14.1)	(19.4)
AR	1.19 ^{a}	2.68 ^{a}	10.3 ^{b}	163 ^{b}	0.29 ^{c}	0.25 ^{b}	0.56 ^{b}	0.20 ^{bc}	0.38 ^{c}
	(7.20)	(0.49)	(35.5)	(135.5)	(12.8)	(10.1)	(4.49)	(13.7)	(14.7)
LiP	1.14 ^{a}	2.65 ^{ab}	15.1 ^{ab}	32.6 ^{c}	0.33 ^{ab}	0.22 ^{c}	0.57 ^{b}	0.22 ^{ab}	0.54 ^{a}
	(7.12)	(0.82)	(12.4)	(155.0)	(10.8)	(11.7)	(5.31)	(10.8)	(15.0)
HiP	1.18 ^{a}	2.64 ^{b}	18.6 ^{a}	10.4 ^{d}	0.34 ^{a}	0.20 ^{c}	0.55 ^{b}	0.22 ^{ab}	0.50 ^{ab}
	(12.0)	(0.67)	(28.4)	(82.9)	(11.9)	(9.90)	(9.59)	(27.0)	(15.5)

Note. For ρ_b, K_s, *Mic*, *Mac*, *Pt*, θ_i and θ_s numbers of soil sample: RF = 28, APR = 28, AR = 28, LiP = 21 and HiP = 21. For *Pd* and *OC* number of soil samples: RF = 12, APR = 12, AR = 12, LiP = 9 and HiP = 9. Different superscript letters denote statistically significant differences between land use types, according to the Tukey's test ($p < 0.05$).

Within-site plots, high variability in K_s was observed in the RF plots and within the two restored forest classes. In contrast, smaller variations were evidenced in pasture sites. Figure 4 includes the results of the of the Tukey's test for all sampled plots. The grouping information highlights the significant and non-significant comparisons for all sampled plots. In the first group, the forest plots evidenced not significant differences due to the high K_s variability within these plots (e.g., K_s means from 104 to 407 mm h^{-1}). Then, the second group (RF1, RF3, RF4, AR1, AR3, LiP1 and LiP2) showed significant differences with pasture sites, which were grouped in a third (LiP1, LiP2, LiP3, HiP1 and HiP2) and fourth group (LiP3, HiP1, HiP2

and HiP3). In general, the LiP and HiP plots were similar, and mean K_s values altogether (e.g., from 8 to 47 mm h^{-1}) were very low (Figure 4).

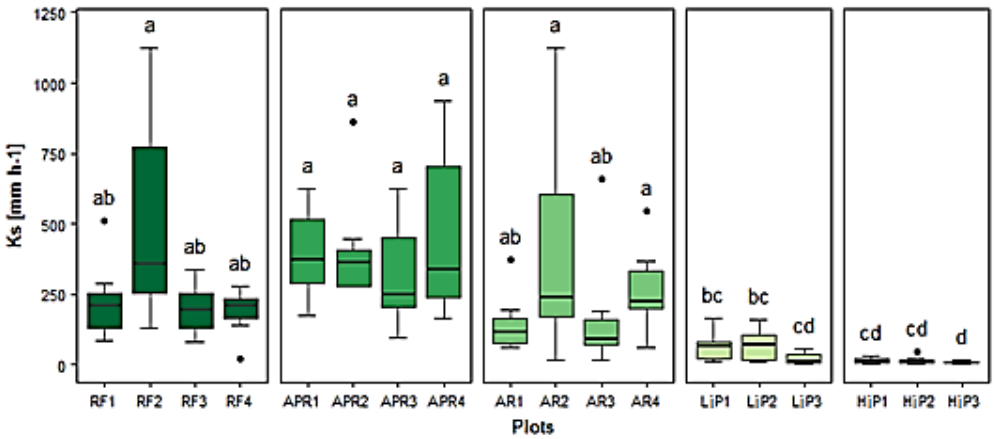

Figure 4. K_s at the surface by individual plots. Different letters above boxplots indicate significant difference based on Tukey's test ($p < 0.05$). RF, Reference Forest; APR, Assisted Passive Restoration; AR, Active Restoration; LiP, Low-intensity Pasture; HiP, High-intensity Pasture. The subscript number refer to plot numbers.

According to the Pearson's correlation coefficient among selected soil attributes across all study sites, significant positive correlations were found for *Pt* vs. *Mac* (0.60) and K_s vs. *Mac* (0.67). In contrast, significant negative correlations were found for ρ_b vs. *Pt* (−0.99), ρ_b vs. *Mac* (−0.58) and K_s vs. *Mic* (−0.49) (Appendix H).

The first and second axis of the PCA for the soil attributes explained 43.0% and 29.3%, respectively, of the variation among all study sites. This analysis revealed a gradient of land-cover types from pastures to forest covers. As expected, the pasture sites were separated from the forest covers due to the higher *Mic* and ρ_b. Similarly, the higher ρ_b values in AR plots contributed to separating the study site. Then, APR plots were more similar to RF plots, and both forest covers were associated with higher K_s, *Mac*, *Pt*, θ_i and *OC* values (Figure 5A). The PCA correlating the soil and vegetation attributes showed a clear segregation among forest cover sites, explaining 55.2% and 16.5% of the variation in the first and second axis, respectively. This analysis showed that RF plots were mainly related with larger trees, evidencing higher correlation with vegetation attributes such as height of trees, density of saplings, basal area, canopy cover and overstory richness, also it was evidenced intermediate values of *Mic*. Considering the vegetation attributes in APR plots, the PCA showed positive correlation with total richness of species, density of trees and richness of saplings, also positive correlation and higher values of K_s, θ_i, *Mac* and *Pt* were found in these plots. By contrast, the separation of AR plots was driven by the higher ρ_b values and lower vegetation attributes, since AR site had a more intensive land use history compared to RF and APR sites. In particular, among AR plots, AR3 was the most different plot, composed by few and smaller trees growing in a compacted soil (Figure 5B).

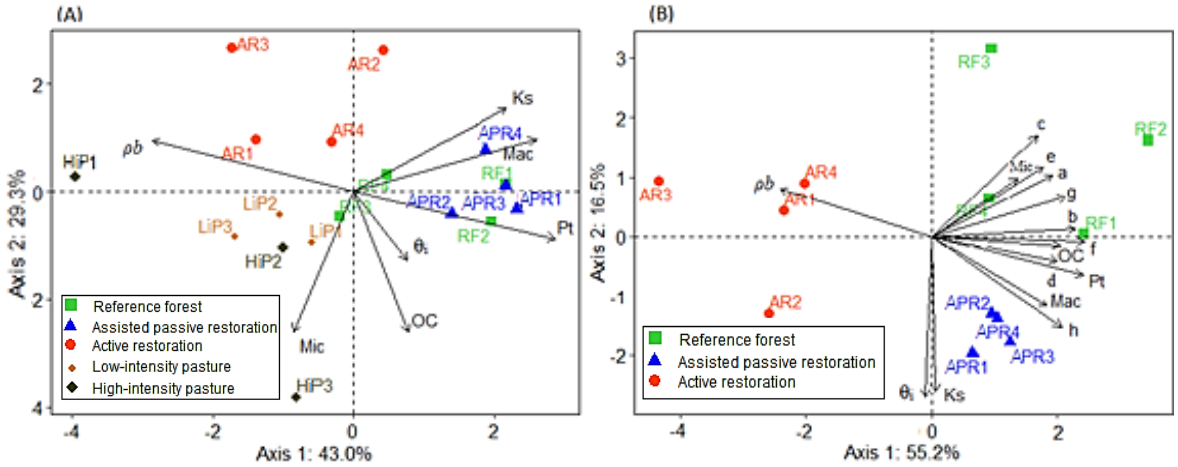

Figure 5. Principal component analysis (PCA) biplot based on soil attributes (A); and PCA correlating soil and vegetation attributes (B). Symbols represent plot sites for each land-cover type: Reference Forest (RF), Assisted Passive Restoration (APR), Active Restoration (AR), Low-intensity Pasture (LiP) and High-intensity Pasture (HiP). The soil physical and hydraulic are indicated in the vectors as follow: ρ_b, bulk density; θ_i, initial volumetric soil water content; K_s, saturated soil hydraulic conductivity; *OC*, soil organic carbon content; *Mac*, soil macroporosity; *Mic*, soil microporosity; and *Pt*, total soil porosity. The vegetation attributes are indicated in the vectors by the letters: (a) basal area; (b) canopy cover; (c) vegetation height of trees; (d) density of trees; (e) density of saplings; (f) total richness of species; (g) overstory richness; and (h) richness of saplings.

4.4. Discussion

4.4.1. Effects of Land-Cover Type and Land Use History on Soil Physical and Hydraulic Properties

Assessing the K_s recovery, soil physical and hydraulic properties of different forest restoration strategies, and investigating their relationships with land use history, vegetation structure and composition, provided the opportunity to identify the extent to which these forest restoration strategies contribute to supplying ecosystem functions as infiltration of rainwater. The variation of K_s among land-cover types was not as we expected, due to the higher K_s evidenced in APR and lower in RF. However, our results supported the first study hypothesis, namely that a more intensive land use history in AR resulted in a lower K_s recovery compared to APR. Importantly, despite both restored forest types being located in the same soil type and landscape position, it was not clear from our measurements whether APR resulted in a faster recovery of K_s compared to AR, due to the high variability in land use history. Similar situations have been reported in several tropical studies [26,29]. In addition, the K_s recovery in APR could be associated with improved soil physical and hydraulic properties, which suggest a higher soil pore connectivity. Hassler et al. [12] found similar K_s at 0–6 cm depth between 100-year-old and 12–15-year-old secondary forests in Panama. Similarly, Leite et al. [24], at Brazilian Caatinga, obtained no significant differences for surface K_s between old-growth forest (more than 55 years) and young secondary forest (7 years). This K_s recovery to pre-pasture levels was also detected at 12.5 cm depth after 15 years of pasture abandonment for an Oxisol in the Brazilian Amazon [11] and by other studies carried out in tropical environments [20,21]. In contrast, after 10 years of natural

regeneration in Ecuador, no significant changes of K_s were reported at 12.5 cm depth for an Inceptisol, which was related with invasive species delaying the K_s recovery [33].

Our K_s in the RF plots can be compared with those for Lozano-Baez et al. [28], as both investigations in the same forest biome estimated the K_s with the SSBI method, on the same soil texture (e.g., sandy clay loam) and at the same soil depth used an identical measurement technique (e.g., Beerkan method) and instruments. Our mean K_s in the RF (215 mm h^{-1}) was close to reported value (387 mm h^{-1}) by Lozano-Baez et al. [28] under similar soil conditions. This difference can be explained by the more conserved soil conditions in the study area of Lozano-Baez et al. [28], for example, as their remnant forest was never burned or cultivated. In contrast, our RF was partially cleared and was affected by natural fire disturbances. This observation is in line with several studies, which suggest that in old-growth tropical forests the K_s can be affected by past soil degradation and intensity of forest use [11,15,29,57]. The K_s values obtained in RF plots (e.g., from 23 to 1,122 mm h^{-1}) showed the high spatial variability of the infiltration process under forest cover, which can be associated with the heterogeneous soil structure, lower ρ_b and higher *Mac* [57–59]. Another factor to consider is the spatial heterogeneity of our RF, where different landscape conditions such as higher slope and vegetation attributes among sample plots could have influenced the K_s variability. In this sense, we believe that the true reference soil condition could be represented by RF plot RF2, due to observed low soil disturbance in this plot, which is consistent with the higher tree basal area (39.4 m^2 ha^{-1}), vegetation height (average of 12.9 m), species richness (41 trees and non-tree; Table 1 and Appendix I), *OC* and soil porosity (Table 3 and Appendix J). Unfortunately, it was not possible to find similar forests in the study area, but we could expect that infiltration capacity in other Brazilian Atlantic Forest patches will be directly related with the forest age and forest conditions, which has been shown by several studies [12,24,60]. Therefore, the K_s values in RF could be limited by the number of measurements (n = 28), which should be increased in future studies, considering the gradient and spatial heterogeneity of forest cover.

Our results highlight the importance of land use legacy on K_s recovery during forest restoration. The restoration diagnosis in APR and AR based on the "Diagnostic" protocol [37], allowed evidencing significant differences in the initial environmental situations (e.g., naturally regenerating native plants) and land use history between both sites. In fact, the initial differences in the initial environmental situations in each restored site allowed the restoration practitioners to identify and select at the beginning of the restoration project the most suitable restoration strategy. Figure 3 provides a graphical summary of these differences between APR and AR. Our findings are also in agreement with other studies that reported lower K_s when land use was more intense prior to forest regrowth [11,21,27]. In this sense, our AR site with a more intensive land use history, resulted in significant lower K_s, which could be attributed mainly to greater soil exposure and soil compaction during the land use history. Our mean K_s in the AR site (163 mm h^{-1}) was considerably higher than the reported value (54 mm h^{-1}) by Lozano-Baez et al. [28] on the same soil texture (e.g., sandy clay loam). This difference can be explained by the higher ρ_b values found in the actively restored forest of Lozano-Baez et al. [28].

Despite no statistically significant differences being found between AR and other forest plots, we stress that the specific past land use intensity and management in each plot could also have played an important role in soil degradation. Thus, we found AR plots (AR1 and AR3), where the K_s values were below the mean of other forest plots. The lower K_s in these plots are consistent with their higher ρ_b and *Mic*, lower *Mac* and *OC* (Appendix J) and possibly a more intensive past management, suggesting that these plots still retain the "memory" from the previous land use. Similar results have been reported by Bonell et al. [26], for a 10-year-old *Acacia* plantation in India growing in Ultisols and Oxisols, with lower K_s when compared to less disturbed forests. On the other hand, the higher K_s in restored plots AR2 and AR4 is closer to the RF and APR plots, indicating that in some cases after 11 years the active restoration could reach the infiltration recovery target defined by the reference conditions. This finding agrees with recent literature reviews, which show that K_s recovery after tree planting in the tropics occur across a wide range of soil conditions [61,62] and probably after more than one decade [26,33]. In addition, it is important to underscore that before tree planting at AR site, trampling pressure occurred for 13 years over abandoned agricultural terraces, causing terracing failure. Several previous studies have reported an increase in soil loss, surface runoff, ρ_b and reduction of infiltration rates after terrace abandoning [63,64]. Another important factor that might have influenced the current K_s in AR site is the possible soil compaction during soil preparation, tree planting using bulldozers or tractors is associated with high levels of soil disturbance, and the effects of this soil preparation can persist long after tree planting [65]. Overall, these circumstances suggest that initial soil conditions before forest restoration actions at AR site were more degraded than in APR. Nevertheless, in our study the lack of soil measurements in each moment of the land use history precludes a stronger understanding of the relative impacts of historical land management on K_s recovery, thus future studies should consider the role of previous land use, comparing sites with a truly identical history.

The significantly lower K_s observed in pasture sites compared to forest land-cover types was consistent with several previous studies [20,33,66], supporting the importance of preserving the forest cover and promoting forest restoration actions in the landscape to maintain the infiltration process. This result can be attributed to higher ρ_b and *Mic* in both pasture sites (Table 3). In the present study, we observed a significant higher K_s in the LiP compared to HiP. The differences in K_s between LiP and HiP are mainly related to factors such as cattle-grazing intensity and the duration of pasture cover, which has been similarly reported in several other tropical studies [11,12,66]. Additionally, the similar *OC* between pasture sites and forest covers (RF and APR) is a trend that has been previously noted by other authors [9,22,28,67], suggesting that such similarities are linked to the accumulation of organic matter by the root system of grasses, the animal-derived inputs and application of fertilizers.

4.4.2. Relationships between Soil, Vegetation and Land Use History

When evaluating vegetation attributes in forest cover sites, our study revealed the significantly lower values in AR and higher values in RF. The higher values of vegetation attributes in the RF could be

explained as a consequence of the longer time that this old-growth forest (Figure 3) has remained undisturbed [68,69]. It is interesting to note that basal area and vegetation height of trees in AR could reach statistically similar values to the RF. Similarly, Garcia et al. [70] found in the same biome no significant differences in the basal area between actively restored forests (12, 23 and 55 years old) and the reference condition. Furthermore, it is noteworthy in AR that, while K_s, basal area and vegetation height of trees have reached statistically similar values with the RF, other vegetation attributes, such as canopy cover, tree density and species richness, were significantly lower than those for the RF plots. For instance, the lower canopy cover, density and richness of saplings (e.g., lack of regenerating trees) in AR might compromise the future forest structure [68], which could hamper the recovery of soil hydraulic properties [62]. Although the herbaceous cover was not directly quantified, we observed a higher abundance of grasses (e.g., *U. brizantha*) in AR site, which could be related to the lower canopy cover. The open canopy conditions in AR may have favored the persistence of grasses, hindering the recruitment of new trees species [71]. In contrast, APR site had statistically similar K_s (Figure 4) and vegetation attributes to the RF (Table 1). However, the vegetation attributes in APR were mainly influenced by non-native trees, such as *P. guajava,* an aggressive pioneer species with allelopathic potential [72]. For these reasons, we suggest that both restored forests need management activities to improve soil and vegetation attributes. Canopy cover protects the soil from physical disturbance, and higher species richness and tree density with native species can produce a higher biomass and enhance the K_s [73,74].

One possible explanation for the different outputs between AR and APR are the initial soil conditions at each site. As mentioned above, when the forest restoration actions began in APR, the soil might have had better initial conditions (lower ρ_b, higher K_s, *OC* and soil porosity), and some degree of natural regeneration, which stimulated the potential recovery of the site and facilitating the recruitment of new trees species [3,37]. In AR site, the more intensive land use history probably led to an area with low resilience, a more compacted soil and poor *OC*. In particular, the intensity of past land use has been reported as the main factor affecting tropical forest recovery, for example, the vegetation in pastures with a long-lasting land use will regenerate more slowly relative to pastures used less intensively [75,76]. Both restoration approaches (APR and AR) have an important role in the process of restoring degraded ecosystems in tropical landscapes and can be used complementary to enhance the chances of restoration success [37,68]. To understand the soil hydraulic recovery after forest restoration, it is important that future studies consider the role of the duration and intensity of the previous land use, including parameters to assess the land use legacy effects [11,77] and more measurements over time in deeper soil layers, which may reveal further differences among restoration strategies.

The correlation results for *Pt* vs. *Mac* and K_s vs. *Mac* are in agreement with several other studies in the Atlantic Forest [30,58], which reported the positive influence of *Mac* for the pore space and infiltration process. This finding is consistent with the inverse relationships between K_s vs. ρ_b, K_s vs. *Mic* and *Mac* vs. ρ_b, also found by some studies [9,23,28,78]. High *OC* contributes to the trend of increasing K_s, soil porosity and reducing ρ_b values [9]. However, the reverse occurred in the present study, which can be attributed to the high *OC* and ρ_b in pastures sites as well as low *OC* in AR but with a high K_s, suggesting that recovery of soil

physical and hydraulic properties is not only dependent on the *OC*. The result of the PCA indicate the importance of forest cover to promote the infiltration and better soil physical and hydraulic properties. This could be associated with the litter inputs, roots and higher soil faunal activity produced by the trees, which can influence positively the aggregate stability, *Mac* and *OC*, thereby K_s increase in forest covers [79,80]. Nevertheless, there is a need to further research the plant–soil interactions; for example, little attention has been paid to the effects of individual trees, richness and density of species on K_s and soil hydraulic properties [80].

4.5. Conclusions

The K_s recovery differed between AR and APR sites. As we expected, the knowledge of the land use history was crucial for understanding the current differences among the study sites for K_s, soil physical and hydraulic properties. This is consistent with our previous work [28] in the same forest restoration program. The K_s and vegetation attributes decreased when land use was more intense prior to forest restoration actions. The influence of land use intensity on soil physical and hydraulic properties could also be evidenced in the comparison between LiP and HiP. The present results further illustrate the positive correlation between K_s and vegetation attributes (tree basal area, vegetation height of trees and overstory richness) in forests undergoing restoration.

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5. FINAL CONSIDERATIONS

Although the importance of soil for forest restoration has been pointed by several studies, it is clear that soil is still regarded as a "black box" by restoration practitioners around the world. In line with previous studies and to continue opening this "black box", our work contributed to understand the effects of forest restoration on the recovery of soil physical and hydraulic properties, more specifically on water infiltration. In the first part of the study (Chapter 2), with a systematic review of scientific literature, we showed that infiltration was likely to increase in many tropical forests restored by tree planting; that infiltration recovery could be faster when the agriculture was the prior land use; that clayey soils (>30% clay) trended to exhibit greater increases in infiltration after tree planting; and that restored forests after 10 years evidenced more similar infiltration values with the pre-disturbance soil conditions. In the second part (Chapter 3), our study was carried out in a study area with great soil variability. To overcome this soil variability, we compared sites with similar soil textural classes in the surface horizon. In this context, our results showed two actively restored forests with marked differences in soil physical and hydraulic properties. In the first situation, the soil recovery could reach similar values to the pre-disturbance soil conditions. In the second situation, the soil recovery was not evidenced, and soil attributes were similar to pasture sites. Consequently, 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. In addition, the different soil responses in both restored forests could be related to many factors (e.g., topographic variations, soil texture, land use history, density and diversity of plants, among others). In order to understand and interpret adequately the influence of these factor on soil physical and hydraulic properties after forest restoration, in the last part (Chapter 4), a more rigorous experiment was designed to capture the least possible soil variation. In this part, we also compared the soil and vegetation recovery of active vs. assisted passive restoration strategies. As we expected, the active restoration site with a more intensive land use history, evidenced lower K_s recovery, soil and vegetation attributes. Our findings also indicated the positive correlation between K_s and vegetation attributes. In this respect it is important to highlight the role of vegetation attributes used in Atlantic Forest monitoring protocols, these vegetation attributes could reflect the good or poor soil conditions in forests undergoing restoration.

Based on the research findings of this thesis, we concluded that forest restoration actions may improve soil physical and hydraulic properties, but in some cases a complete recovery to reference levels can be difficult, especially when land use was more intense prior to forest restoration actions. Thus, we suggest that management activities should be implemented before and during forest restoration to avoid soil compaction and guarantee the soil recovery. Results reported here also indicated that pasture land use affects drastically the soil physical and hydraulic properties, therefore, it is highly recommended to implement best land management practices (e.g., rotational grazing and introduction of silvopastoral systems) to avoid the negative effects on the pastures. Our results have shown that more studies are needed to understand how infiltration and soil recovery occurs in both actively and passively restored forests. It

would be very interesting to study the soil recovery in forest undergoing restoration on different climates, forests and soils types. In this sense, there are important challenges to be addressed, for example, assessing the water movement through the soil profile. Most studies have focused on top-soil, and only few investigations have analyzed the effect of forest restoration on deeper soil layers. Other research priority is to determine the rates and extents, which forest restoration is recovery the infiltration process. In this regard, it is important to highlight that most research is focusing in young restored forest for measuring soil physical and hydraulic properties, if these results are extrapolated could create misleading conclusions, thus long-term studies with replication in time (e.g., undergoing restoration forests with different ages) are also necessary. Considering that hydrological processes occur in the landscape, it would be relevant upscaling the infiltration plot measurements to catchment level and investigating the relationships with other hydrological processes (e.g., groundwater recharge, surface runoff, evaporation among other). Future researches are also needed to understand the effects of individual trees, richness and density of species on soil physical and hydraulic properties. Finally, the role that water infiltration can have for the water quality deserve more attention.

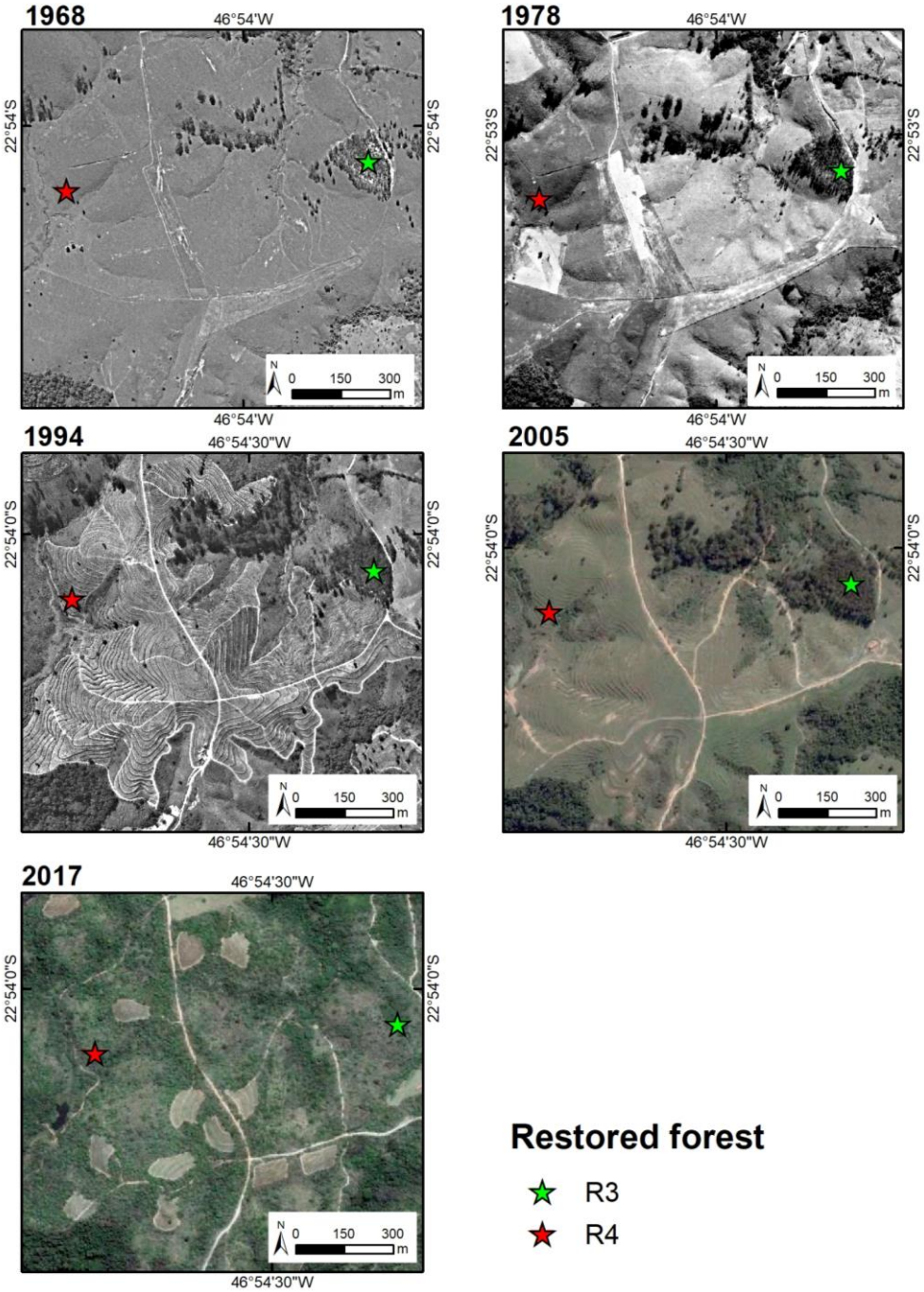
APPENDICES

APPENDIX A. Database used for the systematic review. [a]DP = Diverse planting; M = Monoculture; [b]A = Agriculture; BS = Bare soil; P = Pasture; RF = Reference forest.

Reference	Country	Latitude	Planting type[a]	Soil type	Clay (%)	Age of restoration (years)	Prior land use type[b]	Infiltration in other land use ($mm.h^{-1}$)	Infiltration in restored forest ($mm.h^{-1}$)	Response ratio
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	34	5	P	39	51	0.27
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	30	12	P	39	183	1.55
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	30	12	P	39	524	2.60
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	39	5	RF	370	51	-1.98
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	39	12	RF	370	183	-0.70
Gilmour et al. 1987	Nepal	27°00' N	M	Inceptisols	39	12	RF	370	524	0.35
Bertol and Santos, 1995	Brazil	27°50 S	M	Inceptisols	42	15	RF	380	310	-0.20
Bertol and Santos, 1995	Brazil	27°50 S	M	Inceptisols	42	15	A	40	310	2.05
Mapa, 1995	Sri Lanka	7°19 N	M	Ultisols	28	12	P	26	57	0.78
Mapa, 1995	Sri Lanka	7°19 N	M	Ultisols	28	12	A	29	57	0.68
Salako et al. 2001	Nigeria	7°30 N	M	Alfisols	21	6	A	80	135	0.52
Salako et al. 2001	Nigeria	7°30 N	M	Alfisols	21	6	RF	146	135	-0.08
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	RF	299	34	-2.17
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	RF	299	67	-1.50
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	RF	299	85	-1.26
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	14	34	0.89
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	14	67	1.57
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	14	85	1.80
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	7	34	1.58
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	7	67	2.26
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	7	85	2.50
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	22	34	0.44
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	22	67	1.11
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	22	85	1.35
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	13	34	0.96
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	13	67	1.64
Bonnell et al. 2010	India	15°00 N	M	Alfisols	18	8	A	13	85	1.88
Bonnell et al. 2010	India	15°00 N	M	Vertisols	70	27	RF	7	11	0.45
Bonnell et al. 2010	India	15°00 N	M	Vertisols	70	27	RF	4	11	1.01
Bonnell et al. 2010	India	15°00 N	M	Ultisols	20	27	RF	34	44	0.26
Bonnell et al. 2010	India	15°00 N	M	Ultisols	20	12	RF	34	31	-0.09
Perkins et al. 2012	United States	20°38' N	M	Andisols	19	14	P	1,368	2,592	0.64

Reference	Country	Latitude	Planting type[a]	Soil type	Clay (%)	Age of restoration (years)	Prior land use type[b]	Infiltration in other land use ($mm.h^{-1}$)	Infiltration in restored forest ($mm.h^{-1}$)	Response ratio
Perkins et al. 2012	United States	20°38' N	M	Andisols	19	14	P	1,368	1,980	0.37
Perkins et al. 2012	United States	20°38' N	DP	Andisols	19	14	P	1,368	2,412	0.57
Ghimire et al. 2013	Nepal	27°35' N	M	Inceptisols	19	25	P	25	26	0.04
Ghimire et al. 2013	Nepal	27°35' N	M	Inceptisols	19	25	RF	333	26	-2.55
Ghimire et al. 2014	Nepal	27°47' N	M	Inceptisols	40	36	P	33	114	1.24
Ghimire et al. 2014	Nepal	27°47' N	M	Inceptisols	39	36	P	33	117	1.27
Ghimire et al. 2014	Nepal	27°47' N	M	Inceptisols	40	36	RF	204	114	-0.58
Ghimire et al. 2014	Nepal	27°47' N	M	Inceptisols	39	36	RF	204	117	-0.56
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	17	3	P	220	657	1.09
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	19	3	P	220	495	0.81
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	27	4	P	220	219	-0.01
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	44	6	P	220	512	0.85
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	27	7	P	220	149	-0.39
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	17	8	P	220	1,043	1.56
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	32	11	P	220	1,884	2.15
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	24	11	P	220	1,135	1.64
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	44	12	P	220	734	1.21
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	34	20	P	220	896	1.40
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	17	3	RF	1,421	657	-0.77
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	19	3	RF	1,421	495	-1.06
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	27	4	RF	1,421	219	-1.87
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	44	6	RF	1,421	512	-1.02
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	27	7	RF	1,421	149	-2.26
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	17	8	RF	1,421	1,043	-0.31
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	32	11	RF	1,421	1,884	0.28
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	24	11	RF	1,421	1,135	-0.23
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	44	12	RF	1,421	734	-0.66
Gageler et al. 2014	Australia	26°45' S	DP	Ultisols	34	20	RF	1,421	896	-0.46
Marchini et al. 2015	Brazil	20°22' S	M	Oxisols	28	7	BS	196	299	0.42
Marchini et al. 2015	Brazil	20°22' S	M	Oxisols	28	7	BS	196	351	0.58
Marchini et al. 2015	Brazil	20°22' S	M	Oxisols	28	7	BS	196	361	0.61
Marchini et al. 2015	Brazil	20°22' S	M	Oxisols	28	7	BS	196	235	0.18
Zwartendijk et al. 2017	Madagascar	19°00' S	DP	Oxisols	41	7	A	68	460	1.91
Zwartendijk et al. 2017	Madagascar	19°00' S	DP	Oxisols	41	7	RF	855	460	-0.62
Zwartendijk et al. 2017	Madagascar	19°00' S	DP	Oxisols	41	7	RF	198	460	0.84

APPENDIX B. Aerial photographs for years 1968, 1978, 1994, 2005 and 2017, showing the differences in land use history between restored forest sites R3 and R4 in Campinas, Brazil.



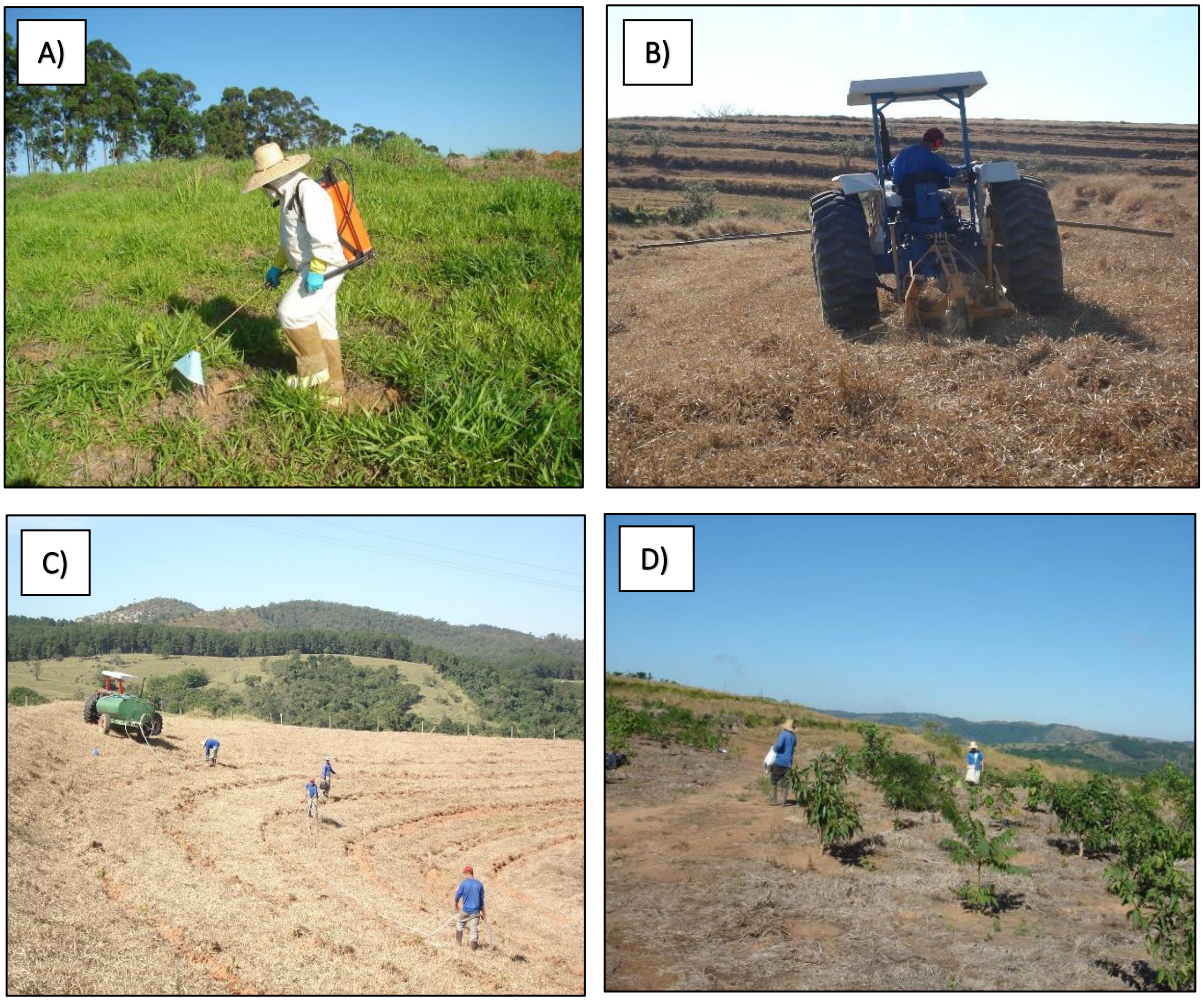
APPENDIX C. Photographs of the likely initial conditions of the assisted passive restoration forest. Both photographs show the enrichment plantings with native tree species in the area with high potential for natural regeneration.

APPENDIX D. List of species used in the Active Restoration site. Successional group: IN = Initial species; MD = Middle species; FL = Filling species; FN: Final species.

Species	Common name	Succesional group
Anacardiaceae		
Astronium graveolens Jacq.	Aroeira-paulista	MD
Astronium urundeuva (Allemão) Engl.	Aroeira-verdadeira	FN
Schinus terebinthifolius Raddi	Aroeira-pimenteira	IN
Apocynaceae		
Aspidosperma cylindrocarpon Müll.Arg	Peroba-poca	MD
Aspidosperma polyneuron Müll.Arg.	Peroba-rosa	FN
Aspidosperma subincanum Mart. ex A. DC.	Guatambu-amarelo	FN
Asteraceae		
Gochnatia polymorpha (Less.) Cabrera	Capitão	FL
Bignoniaceae		
Handroanthus chrysotrichus (Mart. ex A. DC.) Mattos	Ipê-amarelo	MD
Handroanthus impetiginosus (Mart. ex DC.) Mattos	Ipê-roxo	MD
Boraginaceae		
Cordia americana (L.) Gottschling & J.S.Mill.	Guajuvira	MD
Cordia trichotoma (Vell.) Arráb. ex Steud.	Louro-pardo	MD
Cannabaceae		
Trema micrantha (L.) Blume	Crindiúva	FL
Caricaceae		
Jacaratia spinosa (Aubl.) A.DC.	Jaracatiá	FL
Euphorbiaceae		
Alchornea triplinervia (Spreng.) Müll. Arg.	Tapiá	IN
Croton floribundus Spreng.	Capixingui	IN
Croton urucurana Baill.	Sangra-d'água	FL
Fabaceae		
Anadenanthera macrocarpa (Benth.) Brenan	Angico-vermelho	MD
Bauhinia forficata Link	Pata-de-vaca-de-espinho	FL
Centrolobium tomentosum Guill. ex Benth.	Araribá	MD
Copaifera langsdorffii Desf.	Copaíba	FN
Enterolobium contortisiliquum (Vell.) Morong	Tamboril	MD
Hymenaea courbaril L.	Jatobá	FN
Inga striata Benth.	Ingá-vera	FL
Lonchocarpus muehlbergianus Hassl.	Embira-de-sapo	MD
Machaerium stipitatum Vogel	Sapuva	MD
Myroxylon peruiferum L. f.	Cabreúva-vermelha	FN
Peltophorum dubium (Spreng.) Taub.	Canafístula	IN
Piptadenia gonoacantha (Mart.) J.F.Macb	Pau-jacaré	IN
Poecilanthe parviflora Benth.	Coração-de-negro	FL
Senegalia polyphylla (DC.) Britton & Rose	Monjoleiro	IN
Senna multijuga (Rich.) H.S. Irwin & Barneby	Pau-cigarra	IN

Species	Common name	Succesional group
Lamiaceae		
Vitex polygama Cham.	Tarumã	FL
Lecythidaceae		
Cariniana estrellensis (Raddi) Kuntze	Jequitibá-branco	FN
Cariniana legalis (Mart.) Kuntze	Jequitibá-rosa	FN
Malvaceae		
Ceiba speciosa A. St.-Hil.	Paineira	FL
Guazuma ulmifolia Lam.	Mutambo	IN
Heliocarpus americanus L.	Pau-jangada	IN
Luehea divaricata Mart. & Zucc.	Açoita-cavalo	IN
Meliaceae		
Cabralea canjerana (Vell.) Mart.	Canjerana	FN
Myrtaceae		
Eugenia uvalha Cambess.	Uvaia	FL
Petiveriaceae		
Gallesia integrifolia (Spreng.) Harms	Pau-d'álho	MD
Rhamnaceae		
Colubrina glandulosa Perkins	Saguaraji-vermelho	IN
Rosaceae		
Prunus myrtifolia (L.) Urb.	Pessegueiro-bravo	MD
Rutaceae		
Balfourodendron riedelianum (Engl.) Engl.	Pau-marfim	MD
Zanthoxylum hyemale A. St.-Hil.	Maca-de-porca	MD
Salicaceae		
Casearia sylvestris Sw.	Guaçatonga	FL
Sapotaceae		
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	Aguaí	MD
Urticaceae		
Cecropia pachystachya Trécul	Imbaúba	FL
Verbenaceae		
Citharexylum myrianthum Cham.	Pau-viola	IN
Aloysia virgata (Ruiz & Pav.) Juss.	Lixeira	FL

APPENDIX E. Photographs of the likely initial conditions of the actively restored forest. The active restoration included; A) Herbicide application. B) Mechanized soil preparation. C) Irrigation and D) Fertilization.



APPENDIX F. Species list of the trees with DBH > 5 cm sampled in the studies sites: Reference Forest, Assisted Passive Restoration and Active Restoration. *Non-native species.

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
Anacardiaceae				
Astronium graveolens Jacq.	1	7	10	18
Schinus terebinthifolius Raddi		20	12	32
Apocynaceae				
Aspidosperma cylindrocarpon Müll.Arg		1		1
Aspidosperma polyneuron Müll.Arg.	15			15
Arecaceae				
Syagrus romanzoffiana (Cham.) Glassman	5	5		10
Asteraceae				
Gochnatia polymorpha (Less.) Cabrera	3	19	3	25
Vernonanthura brasiliana (L.) H. Rob.			1	1
Vernonanthura phosphorica (Vell.) H. Rob.		1		1
Viguiera robusta Gardner	1			1
Bignoniaceae				
Handroanthus chrysotrichus (Mart. ex A. DC.) Mattos			3	3
**Tecoma stans* (L.) Juss. ex Kunth		12		12
Boraginaceae				
Cordia magnoliifolia Cham.	4			4
Cordia trichotoma (Vell.) Arráb. ex Steud.			4	4
Burseraceae				
Protium heptaphyllum (Aubl.) Marchand	1			1
Cannabaceae				
Trema micrantha (L.) Blume	1	5		6
Caricaceae				
Jacaratia spinosa (Aubl.) A.DC.			1	1
Celastraceae				
Maytenus floribunda Reissek	1			1
Erythroxylaceae				
Erythroxylum cuneifolium (Mart.) O.E. Schulz			1	1
Euphorbiaceae				
Alchornea glandulosa Poepp. & Endl.		5		5
Alchornea triplinervia (Spreng.) Müll. Arg.			2	2
Croton floribundus Spreng.	35	5	2	42
Croton urucurana Baill.		2	5	7
Sebastiania commersoniana (Baill.) L.B. Sm. & Downs	2			2
Tetrorchidium rubrivenium Poepp.	6			6
Fabaceae				
Anadenanthera colubrina (Vell.) Brenan		1		1
Anadenanthera macrocarpa (Benth.) Brenan			1	1
Bauhinia longifolia (Bong.) Steud.	6			6
Centrolobium tomentosum Guillem. ex Benth.		1	1	2

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
Dimorphandra mollis Benth.		2		2
Enterolobium contortisiliquum (Vell.) Morong			3	3
Holocalyx balansae Micheli	4			4
Inga edulis Mart.		1	1	2
Inga striata Benth.		1	4	5
Lonchocarpus muehlbergianus Hassl.	1			1
Machaerium hirtum (Vell.) Stellfeld	1			1
Machaerium nyctitans (Vell.) Benth.	1			1
Machaerium scleroxylon Tul.		5		5
Machaerium stipitatum Vogel	2			2
Peltophorum dubium (Spreng.) Taub.		9	3	12
Piptadenia gonoacantha (Mart.) J.F.Macb	41		1	42
Senegalia polyphylla (DC.) Britton & Rose		2	6	8
Senna multijuga (Rich.) H.S. Irwin & Barneby		2	7	9
**Tipuana tipu* (Benth.) Kuntze		4		4
Lauraceae				
Ocotea cf. beulahiae Baitello	3			3
Ocotea elegans Mez	4			4
Ocotea indecora (Schott) Mez	1			1
Lecythidaceae				
Cariniana estrellensis (Raddi) Kuntze	2			2
Cariniana legalis (Mart.) Kuntze	3			3
Magnoliaceae				
**Magnolia champaca* (L.) Baill. ex Pierre	1			1
Malvaceae				
Ceiba speciosa A. St.-Hil.	1			1
Guazuma ulmifolia Lam.		12	8	20
Luehea divaricata Mart. & Zucc.	1	9	11	21
Luehea paniculata Mart. & Zucc.		2		2
**Sida cordifolia* L.	1			1
Meliaceae				
Cedrela fissilis Vell.	9	1	2	12
Guarea guidonia (L.) Sleumer		1		1
Guarea kunthiana A.Juss	2			2
**Melia azedarach* L.			3	3
Trichilia clausseni C.DC.	3			3
Trichilia elegans A.Juss.	2			2
Moraceae				
Maclura tinctoria (L.) D.Don ex Steud.	3			3
Myrtaceae				
Campomanesia xanthocarpa Mart. ex O. Berg	1			1
Eugenia florida DC.	1			1
Eugenia paracatuana O.Berg		1		1
Myrciaria floribunda (H.West ex Willd.) O.Berg	3			3

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
**Psidium guajava* L.	47	61	16	124
Nyctaginaceae				
Guapira hirsuta (Choisy) Lundell	1			1
Phyllanthaceae				
Savia dictyocarpa Müll.Arg.	4			4
Phytolaccaceae				
Seguieria langsdorffii Moq.	1			1
Piperaceae				
Piper mollicomum Kunth	3			3
Piper sp.		1		1
Primulaceae				
Myrsine balansae (Mez) Otegui		1		1
Myrsine coriacea (Sw.) R. Br. ex Roem. & Schult.		10		10
Myrsine umbellata Mart.	9	1		10
Rhamnaceae				
Colubrina glandulosa Perkins			1	1
Rhamnidium elaeocarpum Reissek	7			7
Rosaceae				
Prunus myrtifolia (L.) Urb.		1		1
Rubiaceae				
Psychotria carthagenensis Jacq.		1		1
Randia armata (Sw.) DC.		3		3
Rutaceae				
Almeidea coerulea (Nees & Mart.) A. St.-Hil.	5			5
**Citrus* sp.		1		1
Esenbeckia leiocarpa Engl.	6			6
Zanthoxylum acuminatum (Sw.) Sw.		1		1
Zanthoxylum fagara (L.) Sarg.	2			2
Zanthoxylum rhoifolium Lam.		1		1
Salicaceae				
Casearia sylvestris Sw.	6	2		8
Sapindaceae				
Allophylus edulis (A.St.-Hil. et al.) Hieron. Ex Niederl.	1			1
Cupania vernalis Cambess.		1		1
Matayba elaeagnoides Radlk.	1			1
Solanaceae				
Acnistus arborescens (L.) Schltdl.			1	1
Cestrum sp.	1			1
Urticaceae				
Cecropia pachystachya Trécul			1	1
Verbenaceae				
Aloysia virgata (Ruiz & Pav.) Juss.		37	7	44
Citharexylum myrianthum Cham.			1	1

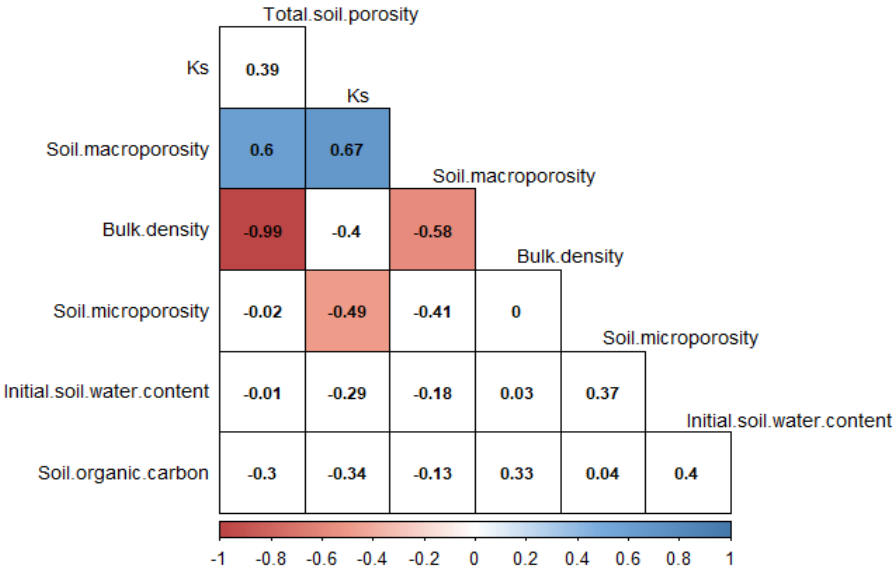
APPENDIX G. Species list of the trees with DBH 1-5 cm sampled in the studies sites: Reference Forest, Assisted Passive Restoration and Active Restoration. *Non-native species.

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
Anacardiaceae				
Astronium graveolens Jacq.	6	2		8
Schinus terebinthifolius Raddi		7	7	14
Apocynaceae				
Aspidosperma polyneuron Müll.Arg.	1			1
Aspidosperma subincanum Mart. ex A.DC.			1	1
Arecaceae				
Syagrus romanzoffiana (Cham.) Glassman	1			1
Asteraceae				
Baccharis dracunculifolia DC.			20	20
Chromolaena asperrima (Sch.Bip. ex Baker) R.M.King & H.Rob.		1		1
Chromolaena sp.		2		2
Cyrtocymura scorpioides (Lam.) H.Rob.		2		2
Gochnatia polymorpha (Less.) Cabrera		14	5	19
Piptocarpha rotundifolia (Less.) Baker			1	1
Piptocarpha sellowii var. balansiana Hieron.	1			1
Vernonanthura beyrichii (Less.) H.Rob.		1		1
Vernonanthura phosphorica (Vell.) H. Rob.			4	4
Vernonanthura brasiliana (L.) H. Rob.		2		2
Bignoniaceae				
Handroanthus impetiginosus (Mart. ex DC.) Mattos		1		1
**Tecoma stans* (L.) Juss. ex Kunth		3		3
Boraginaceae				
Cordia selloviana Cham.	1			1
Burseraceae				
Protium heptaphyllum (Aubl.) Marchand	1			1
Cannabaceae				
Trema micrantha (L.) Blume	1			1
Chrysobalanaceae				
Hirtella hebeclada Moric. ex DC.	1			1
Euphorbiaceae				
Actinostemon concolor (Spreng.) Müll.Arg	7			7
Alchornea glandulosa Poepp. & Endl.	2	3		5
Croton floribundus Spreng.	5	2		7
Sapium glandulosum (L.) Morong		2		2
Sebastiania commersoniana (Baill.) L.B. Sm. & Downs	11			11
Tetrorchidium rubrivenium Poepp.	8			8
Fabaceae				
Centrolobium tomentosum Guillem. ex Benth.	1			1
Dalbergia frutescens (Vell.) Britton	1			1

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
Dimorphandra mollis Benth.		1		1
Indigofera hirsuta L.		1		1
Inga edulis Mart.		1	1	2
Inga striata Benth.			1	1
Lonchocarpus muehlbergianus Hassl.		1		1
Luetzelburgia guaissara Toledo	2			2
Machaerium brasiliense Vogel	2			2
Machaerium scleroxylon Tul.		1		1
Machaerium stipitatum Vogel	4			4
Myroxylon peruiferum L.f.	2			2
Peltophorum dubium (Spreng.) Taub.		1	1	2
Piptadenia gonoacantha (Mart.) J.F.Macb	77	1		78
Senegalia polyphylla (DC.) Britton & Rose		3		3
Lauraceae				
Aniba firmula (Nees & Mart.) Mez	4			4
Nectandra megapotamica (Spreng.) Mez	1			1
Ocotea elegans Mez	1			1
Ocotea sp.	1			1
Malvaceae				
Luehea divaricata Mart. & Zucc.	1		1	2
Triumfetta rhomboidea Jacq.		1	1	2
Melastomataceae				
Miconia discolor DC.	1			1
Miconia sp.		3		3
Meliaceae				
Cedrela fissilis Vell.	7			7
Guarea guidonia (L.) Sleumer	1			1
Guarea kunthiana A.Juss	1			1
Trichilia casaretti C.DC.	1			1
Trichilia catigua A.Juss.	1			1
Trichilia clausseni C.DC.	5			5
Trichilia elegans A.Juss.	8			8
Trichilia pallida Sw.	2			2
Moraceae				
Maclura tinctoria (L.) D.Don ex Steud.		1		1
Myrtaceae				
Eugenia florida DC.	2			2
Myrciaria floribunda (H.West ex Willd.) O.Berg	4			4
Psidium cattleianum Sabine		1		1
**Psidium guajava* L.	17	20	21	58
Phyllanthaceae				
Savia dictyocarpa Müll.Arg.	1			1
Phytolaccaceae				
Seguieria langsdorffii Moq.	1			1

Species	Reference Forest	Assisted Passive Restoration	Active Restoration	Total
Piperaceae				
Piper amalago L.	4			4
Piper gaudichaudianum (Kunth) Kunth ex Steud.	7			7
Piper malacophyllum (C.Presl) C.DC.	13			13
Piper mollicomum (Kunth) Kunth ex Steud.	5			5
Piper sp.	1	29		30
Piper umbellatum L.		1		1
Primulaceae				
Myrsine coriacea (Sw.) R. Br. ex Roem. & Schult.	2	26		28
Myrsine umbellata Mart.	22			22
Rhamnaceae				
Rhamnidium elaeocarpum Reissek	4		1	5
Rubiaceae				
Psychotria carthagenensis Jacq.		2		2
Psychotria leiocarpa Cham. & Schltdl.	1			1
Psychotria myriantha Müll.Arg.	1			1
Randia nitida (Kunth) DC.		4		4
Rutaceae				
Almeidea lilacina A.St.-Hil.	9			9
Almeidea coerulea (Nees & Mart.) A. St.-Hil.	3			3
Esenbeckia febrifuga (A.St. -Hil.) A. Juss. ex Mart.			1	1
Metrodorea stipularis Mart.	8			8
Zanthoxylum fagara (L.) Sarg.	1	1		2
Zanthoxylum acuminatum (Sw.) Sw.		1		1
Salicaceae				
Casearia decandra Jacq.	1			1
Casearia sylvestris Sw.	18	2		20
Sapindaceae				
Allophylus edulis (A.St.-Hil. et al.) Hieron. Ex Niederl.	1	1		2
Cupania vernalis Cambess.	3			3
Matayba elaeagnoides Radlk	1			1
Sapotaceae				
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	2			2
Solanaceae				
Cestrum mariquitense Kunth		6		6
Cestrum strigillatum Ruiz & Pav.			1	1
Solanum swartzianum Roem. & Schult.	1			1
Verbenaceae				
Aloysia virgata (Ruiz & Pav.) Juss.	4	4		8
**Lantana camara* L.		1		1
Violaceae				
Hybanthus atropurpureus (A.St.-Hil.) Taub.	9	2		11

APPENDIX H. Correlogram showing the Pearson correlations coefficients between soil attributes across the study sites: Reference Forest, Assisted Passive Restoration, Active Restoration, Low-intensity Pasture and High-intensity Pasture. Positive significant correlation ($p < 0.05$) are indicated in blue, negative ones in red and no significant correlation in white.



APPENDIX I. Vegetation attributes across the study plots. Study sites are abbreviated with RF for Reference Forest, APR for Assisted Passive Restoration, AR for Active Restoration. The subscript number refer to plot numbers.

Site	Plot	Basal area (m^2 ha^{-1})	Canopy cover (%)	Vegetation height of trees (m)	Density of trees (ind. ha^{-1})	Density of saplings (ind. ha^{-1})	Total richness (tree and non-tree)	Overstory richness	Richness of saplings
Reference Forest	RF1	18.7	90	9.4	1,260	3,700	40	31	22
	RF2	39.4	100	12.9	1,080	3,900	41	27	21
	RF3	24.7	95	10.7	1,260	4,450	27	19	14
	RF4	22.9	98	7.3	1,720	3,750	28	20	17
Assisted Passive Restoration	APR1	20.8	90	7.0	1,260	600	26	7	21
	APR2	26.3	88	6.7	1,500	1,700	30	16	19
	APR3	14.1	92	8.1	1,160	4,750	34	22	20
	APR4	22.2	95	9.2	1,240	800	28	11	21
Active Restoration	AR1	8.5	84	7.1	720	1,000	22	7	17
	AR2	9.5	81	7.0	640	900	18	8	15
	AR3	9.5	70	6.7	400	500	14	6	10
	AR4	22.4	75	7.2	680	1,000	17	5	15

APPENDIX J. Mean for soil attributes in the depth 0-5 cm across the study plots. Particle size distribution (clay, silt and sand in %), soil bulk density (ϱ_b in g cm^{-3}), soil particle density (Pd in g cm^{-3}), soil organic carbon content (OC g Kg^{-1}), saturated soil hydraulic conductivity (K_s in mm h^{-1}), microporosity (Mic in cm^3 cm^{-3}), macroporosity (Mac in cm^3 cm^{-3}), total soil porosity (Pt in cm^3 cm^{-3}), initial volumetric soil water content (θ_i in in cm^3 cm^{-3}) and saturated volumetric soil water content (θ_s in cm^3 cm^{-3}). Study sites are abbreviated with RF for Reference Forest, APR for Assisted Passive Restoration, AR for Active Restoration, LiP for Low-intensity Pasture and HiP for High-intensity Pasture. The subscript number refer to plot numbers.

Site	Plot	Clay	Silt	Sand	ϱ_b	*Pd*	*OC*	K_s	*Mic*	*Mac*	*Pt*	θ_i	θ_s
Reference Forest	RF1	24.4	28.4	47.1	1.00	2.68	15.2	225	0.30	0.33	0.63	0.20	0.55
	RF2	25.2	24.7	50.0	1.01	2.62	21.6	524	0.31	0.30	0.62	0.17	0.51
	RF3	26.3	26.6	47.0	1.08	2.66	14.2	198	0.34	0.25	0.60	0.16	0.44
	RF4	23.1	24.0	52.9	1.08	2.66	13.7	188	0.32	0.27	0.59	0.19	0.43
Assisted Passive Restoration	APR1	31.6	31.3	37.0	1.01	2.68	20.4	397	0.30	0.28	0.62	0.22	0.43
	APR2	33.2	35.6	31.2	1.06	2.68	16.3	411	0.33	0.27	0.60	0.24	0.44
	APR3	30.6	31.3	38.1	1.03	2.68	15.4	326	0.29	0.30	0.61	0.28	0.54
	APR4	25.2	29.5	45.3	1.06	2.67	13.4	475	0.32	0.32	0.60	0.21	0.40
Active Restoration	AR1	27.7	28.8	43.5	1.21	2.70	9.8	149	0.32	0.24	0.55	0.21	0.40
	AR2	30.5	22.5	47.0	1.15	2.68	8.3	418	0.27	0.26	0.57	0.22	0.38
	AR3	31.9	21.6	46.4	1.27	2.69	8.3	176	0.27	0.25	0.53	0.20	0.37
	AR4	29.9	22.7	47.3	1.14	2.66	14.9	272	0.29	0.26	0.57	0.18	0.37
Low-intensity Pasture	LiP1	30.8	25.3	43.8	1.15	2.66	18.1	64	0.32	0.25	0.57	0.24	0.52
	LiP2	31.2	20.9	47.8	1.11	2.64	12.6	71	0.32	0.20	0.58	0.22	0.58
	LiP3	32.9	21.6	45.4	1.17	2.67	14.6	23	0.34	0.22	0.56	0.21	0.53
High-intensity Pasture	HiP1	30.4	24.9	44.7	1.34	2.65	16.1	16	0.33	0.21	0.49	0.16	0.44
	HiP2	29.5	23.0	47.5	1.09	2.63	15.4	15	0.32	0.19	0.59	0.21	0.53
	HiP3	41.0	21.9	37.1	1.12	2.64	24.1	9	0.37	0.21	0.58	0.29	0.51