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Environmental Disturbances and Soil Water Infiltration: A study case of fire and post-fire plant communities

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Dissertação de Mestrado

Environmental Disturbances and Soil Water Infiltration: A study case of fire and post-fire plant communities

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Environmental Disturbances and Soil Water Infiltration: A study case of fire and post-fire plant communities

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"I know it seems hard sometimes but remember one thing. Through every dark night, there's a bright day after that. So, no matter how hard it gets, stick your chest out, keep your head up and handle it."

Tupac Shakur

Summary

ABSTRACT

The Brazilian Savanna (Cerrado) is the second-largest Brazilian biome with an area of approximately 2 million km². Because of its high diversity of plants and vertebrates, many of them endemics, this biome is considered a global biodiversity hotspot. However, due to agricultural expansion and urban development, up to 45% of its original area has been deforested, and 34% of the remaining native areas is likely to be cleared by 2050. Among the most frequent disturbance sources in this biome, fire and invasive species stand out. Fire is a fundamental component of the Cerrado biome. However, human actions are changing its behavior. As a consequence, fire regime changes can foster vegetation transitions such as biological invasions. In its turn, invasive plants suppress natives, impacting many ecosystem compartments. Also, they can alter the frequency and severity of fires, which may further promote the dominance of invaders on native ecosystems. These subjects have been studied in the context of Cerrado and the majority of researches addresses its effects on the diversity of native plants. On the other hand, its ecohydrological impacts on ecosystems remain largely unknown. Therefore, this dissertation tries to fill part of this gap presenting two separated chapters. The first presents a systematic review regarding the fire effects on soil water infiltration and the second addresses the effects of invasive plants (*Pteridium arachnoideum*) on soil water infiltration of riparian zones within the Cerrado biome.

Keywords: fire; hydraulic conductivity; systematic review; Cerrado

GENERAL PRESENTATION

The Brazilian Savanna (regionally known as Cerrado) is the second-largest Brazilian biome occupying an approximate area of 2 million km² (Ribeiro and Walter, 2008; Sano et al., 2019). This huge biome comprises a mosaic of vegetation types, forming a structural gradient from grasslands through savanna woodland to forests (Durigan and Ratter, 2016). As a consequence, the Cerrado biome presents a high diversity of plants and vertebrates, many endemics, which makes it a global biodiversity hotspot (Strassburg et al., 2017). Despite its importance for biodiversity conservation, up to 45% of its native vegetation cover has been removed, and only 7.5% of the biome is legally protected (Sano et al., 2019; Strassburg et al., 2017). Due to its extreme fragmentation, many of these areas do not guarantee the maintenance or representativeness of plant and animal species (Latrubesse et al., 2019). Thus, the combination of limited protection and agricultural expansion explains the projections that up to 34% of the remaining native area is likely to be deforested by 2050 (Soares-Filho et al., 2016; Strassburg et al., 2017). All of these factors lead to the conclusion that Cerrado may be the most threatened tropical savanna in the world (Sano et al., 2019).

Among the most frequent disturbance sources in this biome, fire, and invasive species stand out (Durigan et al., 2007) and, therefore, these events are the main focus of the present dissertation. Fire is a natural component of this biome and plays a fundamental role in its processes (Durigan and Ratter, 2016). However, anthropogenic activities, such as the use of fire for agricultural purposes, have changed its natural frequency (Arruda et al., 2018; Bowman et al., 2009). Consequently, when fire regimes are altered vegetation transitions such as invasive plants spreading can occur (Brooks et al., 2004; Pierson et al., 2011). After setting in a new habitat, invasive plants suppress native ones and disrupt multiple ecosystem compartments (Catford, 2017; Miatto et al., 2011). Additionally, invasive plants can alter the quality and quantity of fuel load and, therefore, the frequency and severity of fires (Castro-Díez and Alonso, 2017), which may further promote the dominance of invaders.

Naturally, both fire and biological invasions have been studied in the context of the Cerrado biome (Arruda et al., 2018; De Oliveira Xavier et al., 2019; Durigan and Ratter, 2016; Guerin and Durigan, 2015; Miatto et al., 2011; Xavier et al., 2016). Nevertheless, most of the research addresses its effects on plant diversity (Arruda et al., 2018), whereas its ecohydrological impacts on ecosystems remain unknown. Therefore, trying to fill this gap, this dissertation is structured in two chapters. The first is titled “How does fire affect the soil water infiltration? A systematic review” and its objective is to identify, compile, and evaluate the results of articles published worldwide regarding the fire effects on soil water infiltration. Finally, the second chapter is entitled “Understanding the effect of *Pteridium arachnoideum* invasion on soil permeability within riparian zones” and its objective was to evaluate the effects of plant invasions after three major wildfires on soil permeability in a riparian zone within the Cerrado biome.

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CHAPTER 1 - HOW DOES FIRE AFFECT THE SOIL WATER INFILTRATION? A SYSTEMATIC REVIEW

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Abstract:

Fire is a phenomenon of global distribution and essential for many ecosystems. On the other hand, when it is intense and uncontrolled, it becomes one of the disturbances with the most significant potential impact on the dynamics of ecosystems. In addition to impacts on vegetation, fire can generate profound changes in the biological, chemical, and physical-hydric properties of soils, including infiltration capacity, a fundamental process of the hydrological cycle. Therefore, we aimed to conduct a systematic review that identifies the effects of fire on the soil water infiltration capacity, in order to compile and evaluate the results of papers published on the subject worldwide. Our results demonstrate that the theme is studied mainly in the United States and is practically unknown in regions where the fire is a crucial ecosystem agent, such as tropical savannas. Studies use several methods to estimate water infiltration capacity in fire-affected soils. However, the comparison between different methods should be cautious since the equipment has different precision and scales that directly influence their results. Naturally, research results are varied, as the effects of fire on soils are dynamic and do not follow a linear pattern. Most studies point to the negative effect of fire on soil infiltration capacity. Still, some studies indicate that their effects are positive or neutral. Several causes are pointed as responsible for the changes of infiltration after the fire, such as the removal of vegetation cover and the organic layer, the production of ash and hydrophobic compounds, as well as changes in the physical attributes of soils. Among all, vegetation cover seems to be the most important factor. Finally, climate change is known to increase the frequency and severity of fires globally. Thus, it is essential to increase the understanding of the relationship between fire and soil infiltration capacity, so that it is possible to mitigate and prevent the major impacts of this disturbance on ecosystems and human lives.

Keywords: wildfire; prescribed fire, hydraulic conductivity; soil properties.

INTRODUCTION

Wildfires are common processes in many ecosystems, being a natural and fundamental component of forest ecology (Alcañiz et al., 2018). This is a globally distributed phenomenon and represents one of the disturbances with the greatest potential impact on ecosystems, mainly when it occurs uncontrollably and affects their resilience (Brady and Weil, 2013; Chapin III et al., 2011; Ebel and Moody, 2013; Zavala et al., 2014).

In recent decades, the increase in the occurrence and severity of wildfires has become a major concern worldwide, mainly due to the increase in population in areas prone to its development, as their effects can cause considerable economic and catastrophic damages to ecosystems and human health (Lohman et al., 2007; Moody et al., 2013). Currently, the estimated annual burned area in the world is between 300 and 450 million hectares, most of which occur on pastures and savannas in Africa, Australia, Asia, and South America (Flannigan et al., 2013; Meng et al., 2015). Climate change is expected to affect the severity, distribution, occurrence, and behavior of these fires, significantly impacting natural and human resources as well as ecosystem functions (Doerr and Santín, 2016; Westerling, 2016). However, there is still a gap in understanding the role of fire in terrestrial ecosystems, as well as its interaction with global climate change (Bowman et al., 2009).

The extent of fire effects on soils is variable as many factors control it and their responses vary according to the fire severity and environmental conditions before and after their occurrence (Certini, 2005; Pierson et al., 2008a; Shakesby and Doerr, 2006). In general, the action of fire modifies the bioavailability of nutrients and the presence of microorganisms in the soil, as well as their physical and hydrical properties such as porosity, bulk density, structure, aggregate stability, infiltration capacity, and water retention (Carballas et al., 2009; Doerr and Santín, 2016; Fernández and Díaz-raviña, 2015; González-Pelayo et al., 2010; Martín and Vila, 2012). Usually, the combination of all these factors results in more friable, erodible, and hydrophobic soils, which increases the vulnerability of the landscape to the development of erosion processes (Brady and Weil, 2013; Neary et al., 1999; Robichaud et al., 2016; Santín and Doerr, 2016; Shakesby and Doerr, 2006). However, research results are still variable although some has shown a reduction of infiltration rates after fires (Are et al., 2009; Cerdà and Doerr, 2005; Robichaud et al., 2016) some found an infiltration increase (Langhans et al., 2016; Wieting et al., 2017). Several factors are responsible for these changes, such as the removal of vegetation and the litter layer (Pierson et al., 2008a; Robichaud et al., 2016), reduction of soil organic matter content (Fernández et al., 2019, 2012), ash production (Are et al., 2009; Woods and Balfour, 2010) and increased production of hydrophobic organic compounds (Larson-Nash et al., 2018; Mataix-Solera et al., 2013). The extent and duration of these changes are dynamic in time and space as hydrological processes are typically nonlinear and depend on vegetation recovery time, weather conditions before and after fires, and its severity (Cerdà, 2009; Ebel et al., 2012; Moody et al., 2013).

Despite its importance, the relationship between fire and soil infiltration capacity remains mostly unknown, especially in regions of higher occurrence and vulnerability to wildfires such as tropical savannas. Thus, considering the increased global distribution of fires, especially during the drier years that are predicted by climate change, it is essential to better understand post-fire effects and their relationship to soil water infiltration capacity in order to prevent and mitigate their potential impacts (Bowman et al., 2009; Ebel and Moody, 2013; Krawchuk et al., 2009; Moritz et al., 2012; Pechony and Shindell, 2010).

In this context, we carried out a systematic review that quantify the effects of fire on the water infiltration capacity of soils, in order to compile and evaluate the results of papers published on the topic worldwide.

METHODS

In this research, we followed the guidelines suggested in the PRISMA Platform for the elaboration of systematic reviews (Moher et al., 2015). We performed searches in the advanced search fields of Scopus and Web of Science. To do so, we used the following combinations of terms to search titles, summaries, and keywords: {TI=("soil properties" OR "hydraulic conductivity" OR infiltration)} AND {TS=(burn* OR fire OR wildfire)}. The search was conducted in April 2019 without restriction as to the year of publication, being considered only scientific articles published in English. To be included in the analysis, studies had to meet the following eligibility criteria: (1) address the effects of fire on soils; (2) explicitly present the values of infiltration capacity or saturated hydraulic conductivity on the surface (0-10 cm); (3) compare the results of control areas with areas impacted by fire action; (4) present the measurement method used and (5) present the central tendencies values and sample size.

After removing the duplicate records, two independent reviewers selected the publications by reading the content of their titles and abstracts, considering the above eligibility criteria. When both reviewers selected an article for exclusion, it was removed from the systematic review. When only one of the reviewers elected an article for exclusion, a third reviewer was consulted. After this stage, the selected articles have been read entirely to assess their suitability for eligibility criteria, and we extracted the information mentioned above.

RESULTS

The search resulted in 294 publications in the Web of Science database and 373 in Scopus. After removing duplicate articles and selecting based on eligibility criteria, we selected 35 articles (Figure 1 and Table 1).

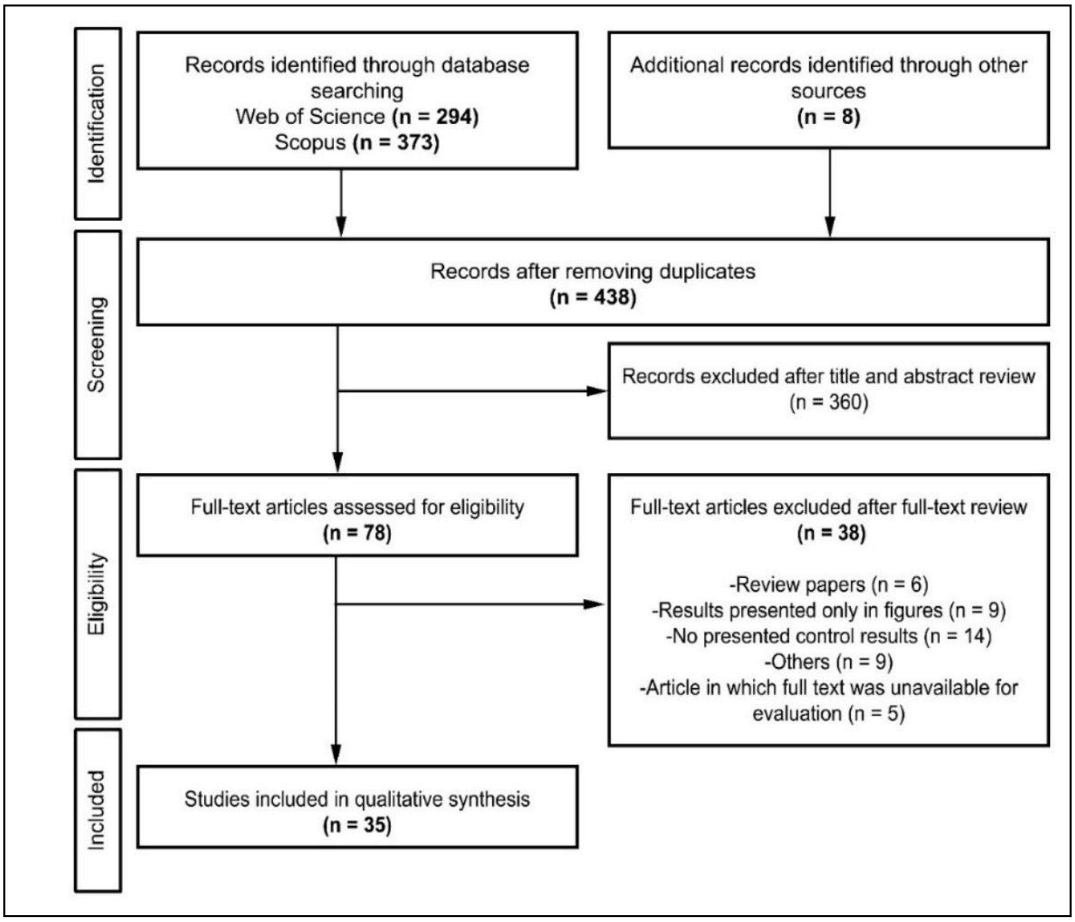

Figure 1. Flow diagram of the selection process of the publications included in the systematic review

The first articles on the subject were published in 1956 (Scott, 1956; Scott and Burgy, 1956). Then, after 22 years, the subject was again addressed in two publications in 1978 (Roundy et al., 1978; Ueckert et al., 1978). Between the 1980s and 1990s, only five articles on the topic were published (Boelhowers et al., 1996; Knight et al., 1983; Lal and Ghuman, 1989; Mallik et al., 1984; Valzano et al., 1997). We observed a slight increase in the annual frequency of publications on the subject between 2000 and April 2019, during which 26 articles were published (Figure 2). Since 2008, the annual amount of publications has been two or more, especially in 2016, which presented the highest number of publications on the topic (Figure 2).

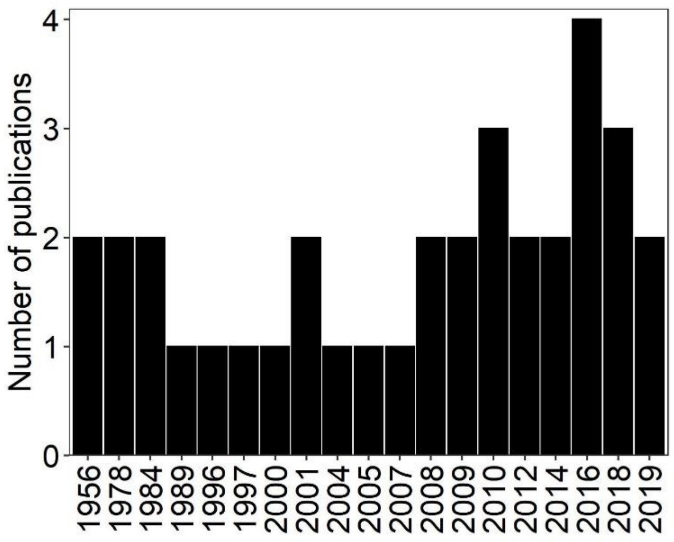

Figure 2. Number of publications per year of studies on the effects of fire on the soil water infiltration**.**

All studies found were performed in only seven countries, most of them were conducted in the United States of America (n = 20), followed by Spain (n = 5) and Australia (n = 4) (Figure 3). Regarding the distribution of sample areas within countries, we observed that, in the United States, they were concentrated in the Midwest. In Spain, the areas were more widely distributed, and in other countries, there was a low distribution of the study areas due to the low number of experiments.

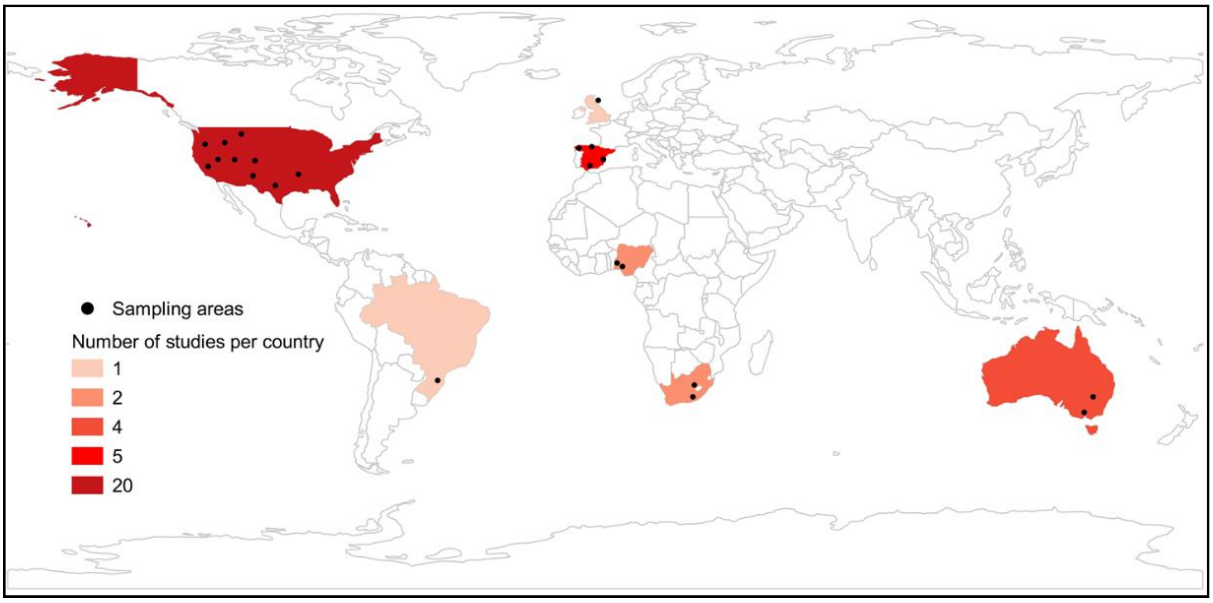

Figure 3. Sampling areas and number of studies per country on water infiltration in fire-affected soils

Most studies took place under prescribed fire conditions (c.a. 60%), whereas the remaining were under wildfire (c.a. 40%). Differences in water infiltration capacity between treatment and control were significant ($p < 0.05$) in 59% of the results. In these cases, when evaluating the type of fire effect, most experiments (76%) indicated a negative effect, whereas 24% indicated a positive effect on infiltration.

Regarding the methods used, the rainfall simulator was the most used equipment (n = 17), followed by the mini-disk infiltrometer (n = 7) and double-ring infiltrometer (n = 5). Finally, the average number of samples was approximately 12, and the largest sample size was 548.

Table 1. Description of studies on the effects of fire on the soil water infiltration.

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
SCOTT, 1956	California, USA	Redding Soil, Aiken soil, Sierra Soil, Hugo Soil and Vista Soil	Ceanothus, Manzanita, Oak and grass	1.85	24	2.34	24	in/hr	Single-ring infiltrometer	0	Prescribed	Not Reported	Yes	Positive
				5.28	24	3.98	24			1			Yes	Negative
				14.71	24	18.06	24			0			Yes	Positive
				17.65	24	19.36	24			1			Yes	Positive
				4.84	24	13.73	24			0			Yes	Positive
				14.35	24	17.50	24			1			Yes	Positive
				5.37	24	14.06	24			0			Yes	Positive
				6.66	24	8.30	24			1			Yes	Positive
				7.33	24	24.06	24			0			Yes	Positive
				6.51	24	15.33	24			1			Yes	Positive
				7.57	10	9.26	10			0			Yes	Positive
				19.45	24	21.01	24			0			Yes	Positive
SCOTT; BURGY, 1956	California, USA	Hugo Soil	Ceanothus and Chamise	0.81	8	1.72	8	in/h	Constant head (Simulation)	Not Reported	Prescribed	Not Reported	Yes	Positive
		Aiken Soil		0.91	8	0.91	8						No	-
ROUNDY; BLACKBURN; ECKERT JR, 1978*	Nevada, USA	Aridic Argixerolls	Scattered pinyon-juniper community (Lower site)	8.07	6	7.71	6	cm/h	Rainfall simulator	1	Prescribed	Not Reported	No	-
				7.76	6	6.76	6						Yes	Negative
				5.76	6	4.53	6						No	-
				3.75	6	2.25	6						Yes	Negative
			Scattered pinyon-juniper community (Upper site)	8.15	6	7.84	6						No	-
				7.82	6	6.71	6						Yes	Negative
				4.12	6	5.97	6						Yes	Positive
				1.95	6	3.85	6						Yes	Positive
UECKERT; WHIGHAM; SPEARS, 1978	Texas, USA	Typic Chromustert	Mesquite-Tobosagrass	98	9	46	9	mm/h	Double-ring infiltrometer	0	Wildfire	Not Reported	Yes	Negative
				130	9	79	9						Yes	Negative
KNIGHT; BLACKBURN; SCIFRES, 1983	Texas, USA	Pachic Argiustolls	Whitebrush	12.4	10	14.2	10	cm/h	Rainfall simulator	0	Prescribed	Not Reported	No	-
		Ustollic Cambothids	Running Mesquite	12.8	10	12.8	10						No	-
MALLIK; GIMINGHAM; RAHMAN, 1984	Aberdeen, Scotland	Brown podzolic soil	Herbaceous vegetation	78.42	8	29.6	8	$ml.cm^{-2}$ / h	Double-ring infiltrometer	0	Prescribed	High	Yes	Negative
				76.15	8	30.6	8			0			Yes	Negative
				50.56	8	32.66	8			0			Yes	Negative
				48.76	8	34.18	8			0			Yes	Negative
				53.72	8	31.42	8			1			Yes	Negative
				52.95	8	20.3	8			1			Yes	Negative
				37.29	8	9.88	8			1			Yes	Negative
				21.84	8	11.35	8			2			Yes	Negative
				43.44	8	17.22	8			2			Yes	Negative
				39.59	8	29.88	8			2			Yes	Negative
LAL; GHUMAN, 1989	Nigeria	Not reported	Tropical Rainforest	13.4	1	105.8	1	cm/h	Double-ring infiltrometer	0	Prescribed	Not Reported	Yes	Positive

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
BOELHOUWERS; DE GRAAF; SAMSODIEN, 1996	Cape Town, South Africa	Sandstone mixed with siltstone and shale materials	Eucalyptus forest	Un101.8	7	7.3	14	mm/h	Rainfall simulator	0	Wildfire	Not Reported	Not Reported	Negative
VALZANO; GREENE; MURPHY, 1997	Cowra, Australia	Xeric Alfisol	Cropland	33.9	5	15.5	5	mm/h	Disc permeameter	0	Prescribed	Low	Yes	Negative
ROBICHAUD, 2000*	Montana, USA	Typic Cryoboralf and Dystric Cryochrept	Douglas-fir and lodgepole forest	79	2	74.5	8	mm/h	Rainfall simulator	0	Prescribed	Low	No	-
	Idaho, USA	Typic Cryumbrept	Ponderosa pine and Douglas-fir forest	49	3	36.5	3					Low	No	-
MARTIN; MOODY, 2001	New Mexico, USA	Volcanic soil	Ponderosa pine	170	5	26	8	mm/h	Rainfall simulator	0	Wildfire	High	Not Reported	Negative
			Mixed conifer	>260	3	97	3						Not Reported	Negative
	Colorado, USA	Granitic soil	Ponderosa pine	120	3	45	3						Not Reported	Negative
PIERSON; ROBICHAUD; SPAETH, 2001	Nevada, USA	Pachic Haploxerolls	Pine Forest	48.1	10	57.6	20	mm/h	Rainfall simulator	0	Wildfire	High	No	-
				66	10	72.4	20			1			No	-
MILLS; FEY, 2004	Kruger National Park, South Africa	Rhodic Lixisols, Paraplinthic Cambisols, Dystric Regosols and Paraplinthic Lixisols	Savanna	302	3	127	3	mm/h	Laboratory infiltration	Not Reported	Prescribed	Not Reported	Yes	Negative
				234	3	219	3						Yes	Negative
				318	3	203	3						Yes	Negative
				259	3	130	3						Yes	Negative
				271	3	179	3						Yes	Negative
				238	3	168	3						Yes	Negative
				172	3	105	3						Yes	Negative
				188	3	134	3						Yes	Negative
				44	2	16	2		Rainfall simulator				Yes	Negative
				43	2	28	2						Yes	Negative
				44	2	27	2						Yes	Negative
				32	2	14	2						Yes	Negative
				42	2	14	2						Yes	Negative
				35	2	26	2						Yes	Negative
				32	2	14	2						Yes	Negative
				13	2	15	2						Yes	Negative
WUEST et al., 2005*	Oregon, USA	Typic Haploxeroll	Wheat plantation	2.34	-	0.52	-	mm/min	Single-ring infiltrometer	Not Reported	Prescribed	Not Reported	Yes	Negative
				1.73		0.28							Yes	Negative
SHERIDAN; LANE; NOSKE, 2007	Victoria, Australia	Acidic Eutrophic Red Dermosols	Eucalyptus forest	1001	27	715	18	mm/h	Single-ring infiltrometer	Not Reported	Wildfire	Not Reported	No	-

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
FERNÁNDEZ et al., 2008*	Pontevedra, Spain	Alumi-umbric Cambisols	Mixed Heathland	58.2	12	36.7	12	mm/h	Rainfall simulator	0	Prescribed	High	Yes	Negative
PIERSON et al., 2008	Idaho, USA	Ultic Haploxerolls	Big sagebrush and Idaho fescue	59.3	10	57.2	10	mm/h	Rainfall simulator	0	Wildfire	High	No	-
						55.5	10			1			Yes	Negative
		Typic Argixerolls and Pachic Haploxerolls	Big sagebrush	33.9	6	26.6	6			0	Prescribed		No	-
				72.3	6	72.7	6			0			No	-
	Nevada, USA	Pachic Haploxerolls	Big sagebrush, Idaho fescue and bluebunch wheatgrass	48.1	10	57.6	20			0	Wildfire		No	-
				66	10	72.4	20			0			No	-
				56.4	10	63.8	20			0			No	-
ARE et al., 2009	Ibadan, Nigeria	Typic Kanhaplustalf	Fallow (4 yrs)	67.2	13	24	13	cm/h	Double-ring infiltrometer	0	Prescribed	Not Reported	Yes	Negative
	Akure, Nigeria		Fallow (35 yrs)	57.6	13	30	13						Yes	Negative
MOODY; KINNER; ÚBEDA, 2009	Colorado, USA	Typic Usorthents and Typic Cryorthents	Not Reported	3×10^{-3}	2	2×10^{-3}	3	cm/s	Mini-disk infiltrometer	2	Wildfire	Not Reported	Not Reported	Negative
				0.86×10^{-3}	2	1.9×10^{-3}	2			2			Not Reported	Positive
GLENN; FINLEY, 2010	Idaho, USA	Not reported	Sagebrush steppe (Shrub)	9.9	12	6.6	46	ml/min	Mini-disk infiltrometer	0	Wildfire	Low	Yes	Negative
						4.5	65					Moderate	Yes	Negative
						5.7	42					High	Yes	Negative
			Sagebrush steppe (Grass)	5.3	12	5.4	23			0		Low	No	-
						4.3	16					Moderate	No	-
						7.6	19					High	Yes	Positive
GONZÁLEZ-PELAYO et al., 2010	Valencia, Spain	Rendzic Leptosol	Shrubland Vegetation	19.6	12	17.2	12	mm/h	Mini-disk infiltrometer	0	Prescribed	High	No	Negative-
				22.3	12	21.5						Moderate		Negative
				166.2	12	79.3	12			1		High	Yes	Negative
				119.4	12	86.1						Moderate		Negative
WOODS; BALFOUR, 2010	Montana, USA	Winkley Soil Series (G)	Pine Forest	91	-	35	-	mm/h	Rainfall simulator	0	Prescribed	High	Yes	Negative
						66				1			Yes	Negative

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
CHIEF; YOUNG; SHAFER, 2012*	Nevada, USA	Courville Soil Series (B)		24	-	26	-	cm/s	Mini-disk infiltrometer	0	Prescribed	High	No	-
						30				1			No	-
		Broland-Yody	Sagebrush steppe and Pinyon-Juniper	1.14×10^{-3}	20	1.13×10^{-3}	20			0			Yes	Negative
						6.63×10^{-4}	16			0			Yes	Negative
						6.23×10^{-4}	20			1			Yes	Negative
				2.08×10^{-4}	20	3.52×10^{-4}	18			0			No	-
						4.52×10^{-4}	18			0			No	-
						3.91×10^{-4}	18			1			No	-
FERNÁNDEZ; VEGA; FONTURBEL, 2012	Orense, Spain	Alumic-umbric Regosols	Shrubland	50.7	32	50.2	16	mm/h	Rainfall simulator	0	Prescribed	Not Reported	No	-
				64.1									Yes	Negative
	Santander, Spain	Humic Cambisols		56.9	32	38.6	16						Yes	Negative
				45									No	-
GORDILLO-RIVERO et al., 2014	Calañas, Spain	Lithic Leptosols	Shrubland and woodland forest	7.3	40	7.2	40	ml/m	Mini-disk infiltrometer	0	Wildfire	Low	Yes	Negative
						6	40					Moderate	Yes	Negative
						5	40					High	Yes	Negative
PIERSON et al., 2014	Nevada and Utah, USA	Not Reported	Single-leaf piñon and Utah juniper	60	7	54	8	mm/h	Rainfall simulator	1	Prescribed	Low/Moderate	No	-
				93	5	81	4			1			No	-
				49	4	56	8			2			No	-
CAWSON et al., 2016	Victoria, Australia	Not reported	Eucalyptus forest	12	17	0	36	mm/h	Pouded infiltrometer	0	Prescribed	Low	Yes	Negative
				0	23	0	26						No	-
				107	16	47	48						Yes	Negative
LANGHANS et al., 2016	Victoria, Australia	Not reported	Dry forest	78.50	45	30.60	147	mm/h	Disk infiltrometer	0	Prescribed	Not Reported	Not Reported	Negative
				52.70	24	81.40	48			1	Prescribed		Not Reported	Positive
				19.20	4	4.40	8		Rainfall simulator	0	Prescribed		Not Reported	Negative
				26.20	4	14.50	3			1	Prescribed		Not Reported	Negative
			Wet Forest	89.9	3	88.9	3			1.5	Wildfire		Not Reported	Negative
				82.40	3	64.20	3			2	Wildfire		Not Reported	Negative
				56.8	3	32.8	3			3	Wildfire		Not Reported	Negative
				476	2	439.10	6		Single-ring infiltrometer	1.5	Wildfire		Not Reported	Negative
				732.20	6	1350.50	6			2	Wildfire		Not Reported	Positive
				248.70	6	354.60	6			3	Wildfire		Not Reported	Positive
ROBICHAUD et al., 2016	Montana, USA	Andic–dystrocryept	Subalpine fir and Fool's Huckleberry Forest	44	21	31	60	mm	Rainfall simulator	0	Wildfire	High	Yes	Negative
				45	21	38	59			1			No	-
				48	21	37	59			2			No	-
				47	20	84	46			5			Yes	Positive

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
WILLIAMS et al., 2016	Idaho, USA	Not Reported	Shrubland Vegetation: Shrub coppice (Dry-Run)	48	8	46	8	mm/h	Rainfall simulator	1	Prescribed	Moderate/High	No	-
						47	6			2			No	-
						55	6			5			No	-
			Shrubland Vegetation: Interspace (Dry-Run)	36	8	44	8			1			No	-
						33	6			2			No	
						48	6			5			Yes	Positive
			Shrubland Vegetation: Shrub coppice (Wet-run)	68	8	68	8			1			No	-
						52	6			2			No	-
						72	6			5			No	-
			Shrubland Vegetation: Interspace (Wet-run)	41	8	55	8			1			Yes	Positive
						35	6			2			No	-
						62	6			5			Yes	Positive
WIETING; EBEL; SINGHA, 2017	Colorado, USA	Aquic arguidolls	Bare soil (Low-temperature)	3.7×10^{-5}	3	2×10^{-5}	3	cm/s	Mini-disk infiltrometer	0	Prescribed	Low	Yes	Negative
						14×10^{-5}	3			0		High		
DOS SANTOS et al., 2018	Santa Catarina, Brazil	Clayey Nitisol	Bare soil (High-temperature)	60.2	12	1.1	12	cm/h	Double-ring infiltrometer	0	Prescribed	Not Reported	Yes	Negative
			Pine Plantation	34.5	12								Yes	Negative
			Crop-livestock Integration (CLI)	3.4	12								Yes	Negative
LARSON-NASH et al., 2018	Idaho, USA	Typic Cryorthent and Typic Xerochrept	Subalpine fir and Douglas-fir Forest (20% slope)	61	10	20	10	mm	Rainfall simulator	0	Wildfire	High	Yes	Negative
				67	10	18	10			1			Yes	Negative
				70	10	19	10			2			Yes	Negative
				83	10	27	8			5			Yes	Negative
			Subalpine fir and Douglas-fir Forest (60% slope)	35	10	33	10			0			No	-
				42	10	33	10			1			No	-
				45	10	33	10			2			No	-
				73	8	62	10			5			No	-
DESROCHERS et al., 2019*	Arkansas, USA	Glossaquic Fraglossudalf	Soybean Conventional Tillage	16.1	-	5.3	-	cm/h	Mini-disk infiltrometer	11	Prescribed	Not Reported	Yes	Negative
FERNÁNDEZ; FONTÚRBEL; VEGA, 2019	Pontevedra, Spain	Alumi-umbric Cambisols	Pine forest (Organic Layer)	93.57	12	25.7	12	mm/h	Rainfall simulator	0	Wildfire	Low	Yes	Negative

Reference	Location	Soil Taxonomy	Soil Cover	Mean (Unburned)	Samples Unburned	Mean (Burned)	Samples Burned	Units	Method	Time since fire (years)	Type of fire	Burn Severity	p< 0.05	Effect
			Pine forest (No Organic Layer)	33.01		10.46				0			Yes	Negative

*Not all treatments considered

DISCUSSION

Historical trends

Our results indicated an interest increase on the subject over the last decades since there has been a rise in related publications quantity. The most substantial amount of research in the United States is probably due to its increased research funding compared to other countries (Grueber and Studt, 2016). In this country, research started more than 60 years ago and is concentrated in the western region, where fires are still increasingly frequent and intense (Westerling, 2016; Westerling et al., 2006). In turn, countries where fire is an fundamental ecosystem agent, such as Spain and Australia, present a reasonable number of publications. However, this number is much lower than in the United States. Finally, given the low number of papers found in the regions with the highest occurrence of fires such as central South America, Central Africa, the Gulf of Mexico, and Southeast Asia (Meng et al., 2015), we found that this subject still poorly understood in these regions. That is also the case of the tropical savannas where fire is frequently present. As mentioned earlier, the lower number of articles in these countries is probably also due to the lower research funding (Grueber and Studt, 2016).

Infiltration Responses

Post-fire infiltration changes do not follow a simple temporal model of progressive recovery where infiltration capacity return to pre-fire levels over some time (Cerdà and Robichaud, 2009). The responses of burned areas are transient, often lasting less than seven years, depending on various aspects, notably the rate of vegetation recovery, post-fire weather conditions, burn severity, litterfall rate, soil water repellency, sediment availability and basin morphology (Moody et al., 2013; Pierson et al., 2008b). For instance, Wieting et al. (2017) found higher infiltration capacity in high-temperature burned compared to low-temperature conditions which is uncommon since most of the prior researches showed opposite results (Ebel et al., 2012; Moody et al., 2016; Nyman et al., 2010; Robichaud, 2000).

Also, the extent and duration of fire-effects are controlled by environmental factors that affect the combustion process, such as amount, nature and moisture of live and dead fuel, air temperature and humidity, wind speed, and topography of the site (Certini, 2005). Therefore, the geographical location of the studies included in our review may explain the high variability of infiltration capacity, as they are under the influence of climatic and geomorphological conditions which, in turn, are determinant for soil water infiltration (Brandão et al., 2006; De Morais, 2012; Robichaud et al., 2016). Furthermore, the hydrological responses of burned soils vary according to the type of fire (prescribed or wildfire) and, consequently, of its severity and duration (Cerdà, 2009; Ebel and Moody, 2017). This might occur because, unlike wildfires, prescribed burning occurs under controlled conditions and is usually used when the soil is moderately moist. Thus, prescribed fires have lower severity and, therefore, its impacts are prone to be lower when compared to wildfires (Alcañiz et al., 2018; Certini, 2005).

The characteristics of the surface horizon are crucial for establishing water infiltration in the soil. Therefore, post-fire changes in the upper horizons are relevant to understand the hydrological processes in the burned areas (González-Pelayo et al., 2010). Several causes may explain such changes, and they may be interrelated. For example, removal of vegetation cover (Pierson et al., 2008a; Robichaud et al., 2016), the type of land use (Desrochers et al., 2019; Dos Santos et al., 2018; Sun et al., 2018), quality and quantity of organic matter (Fernández et al., 2019, 2008), water repellency (Cawson et al., 2016; Larson-Nash et al., 2018), the presence of ashes on the surface (Are et al., 2009; Woods and Balfour, 2010), and soil physical characteristics (e.g., texture, density, and porosity) (Scott and Burgy, 1956).

Vegetation

Although many variables control infiltration rates, the vegetation seems to be dominant, especially in forests (Cerdà, 2009; Robichaud et al., 2016). Under unburned conditions, vegetation increases litter that protects the soil from direct impact of the raindrops and the subsequent destruction of the soil aggregates and the filling and clogging of pores (Robichaud et al., 2016), which allows it to absorb and store larger quantities of water and, consequently, enhancing the infiltration (Cerdà, 2009). In this sense, vegetation recovery is crucial to increase water infiltration into the soil (Muñoz et al., 2017). This occurs because the vegetation cover promotes roots' development and soil biological activity, which increases soil porosity and, consequently, water infiltration (Robichaud et al., 2016). For example, some authors found that five years after fire occurrence, the infiltration capacity of burned areas can be even higher than pre-fire conditions as vegetation cover is recovered (Robichaud et al., 2016; Williams et al., 2016).

On the other hand, as fire consumes above-ground biomass, burned soils become unprotected against the impact of raindrops, which reduces their infiltration capacity (Cerdà, 2009; Cerdà and Doerr, 2005; Pierson et al., 2008a, 2008b; Robichaud et al., 2016). Some researchers suggested that a thicker and moister organic layer enhances the water absorption capacity, which reduces the consumption and temperature of fires on the mineral surface of the soil, minimizing its impacts on infiltration capacity (Desrochers et al., 2019; Dos Santos et al., 2018; Fernández et al., 2019). In this regard, Fernández et al. (2008; 2012) suggested that fuel management treatments, such as chopping or plant clearing, may limit the adverse effects of fire on the environment. The authors found that the infiltration capacity of the areas under these treatments was higher than areas under prescribed fires and organic layer and soil moisture content showed to be critical for runoff and erosion process. Mallik et al. (1984) also found that residue burning reduced the soil water infiltration up to 74%. The authors attributed this to the clogging of soil pores by the ash particles. Similarly, Desrochers et al. (2019) found that the infiltration under unburned conditions was over 300% greater than in the burn treatment and suggested that this difference may be due to the hydrophobic properties of ash reducing infiltration.

Water Repellency

Soil water repellency or hydrophobicity ia an important soil property that affect or interrupts water infiltration, impacts plant growth and potentially leads to soil erosion (Mao et al., 2019). Several articles suggested that soil water repellency is the primary cause of reduced infiltration after burning (Cawson et al., 2016; Doerr et al., 2009; González-Pelayo et al., 2010; Langhans et al., 2016). This occurs because, during combustion, hydrophobic organic compounds in the litter and topsoil are volatilized and released upwards to the atmosphere and downwards into the soil profile along a temperature gradient (Olorunfemi et al., 2014). These molecules penetrate into the cooler underlying soil layers and condense around soil particles forming a distinct water-repellent layer below the surface (Brady and Weil, 2013; DeBano, 2000a). However, soil hydrophobicity is not caused only by fire, but it is also related to soil texture, organic matter content, land use and location (Doerr et al., 2009; Olorunfemi et al., 2014; Robichaud et al., 2016) (see the section “soil physical properties”). Additionally, the soil moisture content is a crucial parameter affecting water repellency, and generally, soils under drier conditions exhibit the highest levels of repellency (Doerr et al., 2006). Hence, it is not completely clear why some soils exhibit soil water repellence, and others do not (Doerr et al., 2009). Also, soil hydrophobicity is not temporally stable and depends upon a set of attributes such as soil characteristics (e.g., structure, density, clay mineralogy and its ratio to organic matter), the growth of specific plant species, the activity of soil microorganisms, decomposition of organic matter, and soil heating by fires (Cerdà and Robichaud, 2009; DeBano, 2000a, 2000b; Jiménez Morillo et al., 2017; Keesstra et al., 2017; Mataix-Solera et al., 2013; Olorunfemi et al., 2014). Likewise, fire can have various effects on soil hydrophobicity. It can create, strengthen, or destroy soil water repellency (Cawson et al., 2016). Consequently, soil water infiltration is directly related to fire severity (Certini, 2005; Gordillo-Rivero et al., 2014). Therefore, this can also explain the high infiltration capacity variability we found in the studies included in our review. However, the temperature limits in which the repellency is strengthened or destroyed still unclear. According to DeBano (2000b), water repellency changed little at soil temperatures below 175 °C, increased considerably between 175 and 200 ºC, and was destroyed when temperatures between 280 and 400 °C. Similarly, Glenn and Finley (2010) found that low infiltration and water repellent soils are likely to be found in areas of moderate fire severity. The authors suggested that the increase in infiltration from moderate to high fire severity may be due to the destruction of hydrocarbons by high temperatures at the surface. On the other hand, other authors found that higher soil temperature was directly related to greater repellency even when they exceeded temperatures that have been associated with the destruction of water repellency (Cawson et al., 2016; Gordillo-Rivero et al., 2014).

Ashes

Another attribute highlighted by many papers is the presence of ashes on the soil surface. However, its effects on soil water infiltration are variable and the reasons, once more, seem not be well defined (Woods and Balfour, 2010). The most common view is that ash contributes towards increased runoff after fire by causing surface sealing and by creating a hydraulically smoother surface (Are et al., 2009; Mallik et al., 1984; Valzano et al., 1997) which limits water infiltration and increase the risk of erosion development. On the other hand, other field-based studies reported that thicker ash layers increased infiltration, primarily by intercepting and storing, resulting in the reduction or prevention of runoff (Cerdà, 1998; Leighton-Boyce et al., 2007; Woods and Balfour, 2010, 2008). Nevertheless, Woods and Balfour (2010) concluded that the hydrological responses also depend upon soil texture since the ash is more prone to clog the pores of a coarse soil than a fine. Therefore, the variability of the results of these experiments may be due to the relation between ash thickness with different soil textures. However, more research is necessary to clarify this relationship and its effects.

Fire effects on soil physical properties

Fire can substantially alter soil characteristics both directly during burning and indirectly during the post-fire recovery period (Santín and Doerr, 2016). The principal direct effect of fire on soil physical properties is related to the combustion of organic matter, since this component may be lost at relatively low temperatures (Mataix-Solera et al., 2011; Neary et al., 2005; Úbeda and Outeiro, 2009; Zavala et al., 2014). Naturally soil structure is affected because organic matter is essential for the formation and stabilization of soil aggregates, particurlarly in the surface horizons (Varela et al., 2015). In turn, the loss of soil structure increases the soil bulk density and reduces its porosity and, consequently, changes in soil water infiltration may occur (Neary et al., 1999; Úbeda and Outeiro, 2009). The magnitude and duration of these changes are dependent on fire severity and the most significant changes in soil physical properties occur when severe fires burn for extended periods (Mataix-Solera et al., 2011; Neary et al., 2005). However, even under lower severity fires, some changes can occur. For instance, some authors found a reduction of soil water infiltration after low severity fires and attributed this to the blockage of soil pores by ashes(Are et al., 2009; Valzano et al., 1997). Furthermore, the heat transfer in the soil profile can promote changes on soil water repellency by the condensation of the organic compounds affecting soil water infiltration (see the section “Water Repellency”) (Neary et al., 2005). Generally, the disruption of most minerals occurs at temperatures over 500 °C (Certini, 2005). In this sense, the components of soil texture (sand, silt, and clay) are not affected by fire unless they are subjected to high temperatures at the mineral soil surface (Neary et al., 2005). In this respect, Scott (1956) and Scott and Burgy (1956) found an increase in water infiltration after high temperature burnings and attributed this to fire-caused physical changes on soil surface, especially the reduction of soil aggregation.

Limitations

We emphasize that our systematic review may have limitations, since studies that did not use fire-related keywords such as burn, wildfire, and fire in their topics may not have been included. Furthermore, a significant problem in characterizing infiltration in fire-affected soils is that some measurement methods appear to bias results (Ebel and Moody, 2013). Most experiments used the rainfall simulator, possibly because it allows the comparison of rain events with the same characteristics (i.e., duration and intensity) in different treatments. However, due to the limited area, their estimates may not accurately represent the characteristics of natural rainfall and the process of larger scale infiltration (Fernández et al., 2012; Robichaud et al., 2016). The mini-disk infiltrometer, in turn, though portable, can have problems on burned soils given it is difficult to maintain a reliable interface between the equipment and the soil surface (Ebel and Moody, 2013; González-Pelayo et al., 2010; Larson-Nash et al., 2018). Likewise, ground saturation measurement methods such as disk infiltrometers can overestimate real infiltration by increased water pressure and lack of trapped air (Cerdà et al., 2009) along with the fact of the potential of not dectecting water repellency. The infiltration values obtained by these types of equipment can be almost five times higher than those of a rainfall simulator and catchment-scales estimates and, in some cases, results in burned soils are close to the control areas (Cerdà, 1996; Cerdà et al., 2009; Ebel and Moody, 2013; Langhans et al., 2016; Nimmo et al., 2009; Nyman et al., 2010). Therefore, the comparison between the results of different methods should be cautious since each equipment has different degrees of precision and scale (Bertoni and Lombardi Neto, 2014; Doerr et al., 2006; Moody et al., 2013, 2009).

FINAL REMARKS

The infiltration process is complex and depends on intrinsic factors related to soil, vegetation, as well as climatic, and geomorphological characteristics of watersheds. Greater complexity is added to the system when a fire occurs, as its effects depend on several environmental factors, making them highly dynamic in time and space.

Predicting these impacts is increasingly challenging. Therefore, it is necessary to improve the modeling of this variable, since researchers used several different methods. Thus, it is essential to increase the number of experiments on the subject, especially in the tropical savannas, in order to clarify how the relationship between fire and water infiltration affects ecosystem processes on larger scales.

Finally, we emphasize that climate change tends to increase the occurrence of extreme drought events. As a result, the frequency and severity of wildfires should increase globally. In this sense, it is essential to enlarge the understanding of their effects in order to minimize and prevent major impacts on ecosystems and human lives. In the present research, we conclude that the results of the studies are varied and there are still several gaps to be filled. Therefore, we expect the number of researches to increase in the coming years.

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CHAPTER 2 - UNDERSTANDING THE EFFECT OF *PTERIDIUM ARACHNOIDEUM* INVASION ON SOIL PERMEABILITY WITHIN RIPARIAN ZONES

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Abstract:

Biological invasion is a major cause of biodiversity loss. In this context, plants of the genus *Pteridium* stand out for being aggressive, cosmopolitan, and having attributes that make them present in all continents, being one of the greatest threats to tropical ecosystems. However, its ecohydrological effects are virtually unknown, especially in riparian zones. Therefore, the main objective was to evaluate the effect of *Pteridium* invasion on soil permeability of a riparian zone within the Brazilian Savanna. Thus, we selected two riparian zones that occur under the same soil type and with different degrees of disturbance, one invaded by *Pteridium arachnoideum* (INV) and the other conserved (CSV). We tested the hypothesis that the presence of these plants reduces soil permeability in the invaded areas. Infiltration, hydrophobicity and penetration resistance tests were performed to evaluate soil permeability as well as their possible forcing. As a result, we found significant differences in all variables assessed ($p<0.01$). The average infiltration capacity in the invaded riparian zone was 72.1 ± 96.4 mm.h^{-1} and in the conserved one was 565.8 ± 219.9 mm.h^{-1}. We attribute this reduction to increased hydrorepellency and penetration resistance in the invaded riparian zone. Although we did not find similar studies, we conclude that in the long run, *Pteridium* plants may affect the hydrology of the riparian zones and watersheds under invasion.

Keywords: Biological Invasion; Bracken Fern; Infiltration; Hydrology; Brazilian Savanna; Cerrado.

INTRODUCTION

Biological invasions are a major threat to biodiversity and ecosystems functioning worldwide (Brooks et al., 2004; Mack et al., 2000; Roos et al., 2010; Vitousek, 1996). Disturbances such as increased atmospheric CO2 concentration and nitrogen deposition, climate change, habitat fragmentation, and altered fire regimes favor the success of invasive species (Dukes and Mooney, 1999). Therefore, biological invasion is related to global environmental changes induced by human action and represents one of the main drivers for increasing species risk of extinction (Dukes and Mooney, 1999; Miatto et al., 2011; Pejchar and Mooney, 2010).

After setting in a new habitat, invasive plants can proliferate and suppress native species, modify ecosystem processes and impact the structure and genetic diversity of the community (Mack et al., 2000; McKinney and Lockwood, 1999; Miatto et al., 2011). Its effects also can be highlighted by changes in nutrient cycling, fire regimes, and especially on soil physical-hydric attributes (e.g. aggregate stability, bulk density, organic matter content, erosion, subsurface and surface flows, water retention, and infiltration capacity) which may further promote the dominance of invaders (Brooks et al., 2004; Catford, 2017; Guerin and Durigan, 2015; Mack et al., 2000; McKinney and Lockwood, 1999; Miatto et al., 2011). Thus, biological invasions can disrupt multiple ecosystem compartments affecting hydrological services in their distribution and availability (Enright, 2000; Huddle et al., 2011; MacDonald, 1989; Nie et al., 2012; Ruwanza et al., 2013; Simberloff and Rejmánek, 2011; Strayer et al., 2006; Vitousek, 1990).

Among the invasive plant species, those of the genus *Pteridium* stand out (De Oliveira et al., 2018; Jatoba, 2016; Matos et al., 2002; Miatto et al., 2011; Roos et al., 2010). These plants are aggressive pioneers adapted to a wide range of environments that, due to their physiological and ecological attributes, make them the most dispersed group of vascular plants in the world (De Oliveira et al., 2018; Guerin and Durigan, 2015; Marrs and Watt, 2006). Besides the competition for nutrients, water, and sunlight, these plants produce allelopathic substances that inhibit the germination of other species, thus representing a great challenge for biodiversity conservation and restoration (De Oliveira et al., 2018; Guerin and Durigan, 2015; Jatoba, 2016; Matos and Pivello, 2009; Miatto et al., 2011; Roos et al., 2010). Additionally, plants of this genus are rich in oil and waxes that make their organic matter resistant to physical and chemical degradation and thus reduce their decomposition rate (Baker and Gaskin, 1987; Doerr et al., 2000; Guo et al., 2018; Halarewicz and Szumny, 2010). Consequently, there is a large accumulation of dry biomass in areas colonized by

Pteridium that increases the occurrence, duration, and severity of wildfires (De Silva and Matos, 2006; Guerin and Durigan, 2015; Matos and Pivello, 2009). Currently, several physical, chemical and biological methods have been tested for the control and restoration of the invaded areas. Combined cutting with or without herbicides has been the most used restoration procedure (Måren et al., 2008; Pakeman et al., 2002; Roos et al., 2011). Another method proved successful is transporting soil with its seedbank from natural environments to the restoration area which can be performed with soil harrowing (Brandão et al., 2017; Carvalho et al., 2019; De Oliveira Xavier et al., 2019), as well as nucleation techniques with fast-growing plants, since *Pteridium* does not respond well to shading (Brandão et al., 2017; De Oliveira Xavier et al., 2019).

Despite the knowledge about the *Pteridium* biology, ecology, and management in temperate regions, its effects on neotropical savannas ecosystems, such as the Brazilian Cerrado are still poorly understood (Matos and Pivello, 2009; Miatto et al., 2011; Silva Matos et al., 2014). Moreover, most of *Pteridium* studies focus on its impacts on plant structure and biological diversity so, the mechanisms that connect this invasive weed to soil hydrology and ecosystem services remain largely unknown (Charles and Dukes, 2008; Vilà and Hulme, 2017).

The objective of the present study was to evaluate the effects of *Pteridium arachnoideum* colonization on soil permeability in a riparian zone within the Cerrado biome, after three major wildfires. As mentioned earlier, biological plant invasions alter the ecohydrological functions of native ecosystems (Catford, 2017; Nie et al., 2012; Zou et al., 2014). Due to the high production of lipid organic matter (Baker and Gaskin, 1987; Doerr et al., 2000; Guo et al., 2018; Halarewicz and Szumny, 2010), our hypothesis is a decrease in soil permeability in the area invaded by *Pteridium arachnoideum*.

METHODS

STUDY AREA

Our experiment was conducted in the riparian zones of Taquara and Pitoco streams, located in the Distrito Federal - Brazil within the domains of RECOR (Ecological Reserve of the Brazilian Institute of Geography and Statistics). This reserve is fully inserted in the Brazilian Savanna and has the largest concentration of scientific researches related to ecology, botany, and zoology of this biome, which makes this reserve a reference in understanding the structure, biodiversity, and functioning of the Brazilian Savanna (IBGE, 2011).

According to the Köppen-Geiger climate classification, the regional climate is Aw: Tropical savanna climate with dry-winter characteristics (Alvares et al., 2013). The average annual temperature is 20.6 ºC, and the climate has two well-defined seasons, summer (wet) and winter (dry), which differ by rainfall volume. The average annual rainfall in the area is 1430 mm and is concentrated between October and March when 84% of the total rainfall occurs. During the other months, the precipitation decreases considerably and the lowest values occur between June and August. In recent years, three major wildfires occurred at RECOR in 1994, 2005, and 2011 (Figure 1). These fires impacted, respectively, 60, 50, and 95% of the reserve area, destroying a part of its experiments and being responsible for increasing the distribution of invasive plant species, especially those of the genus *Pteridium*.

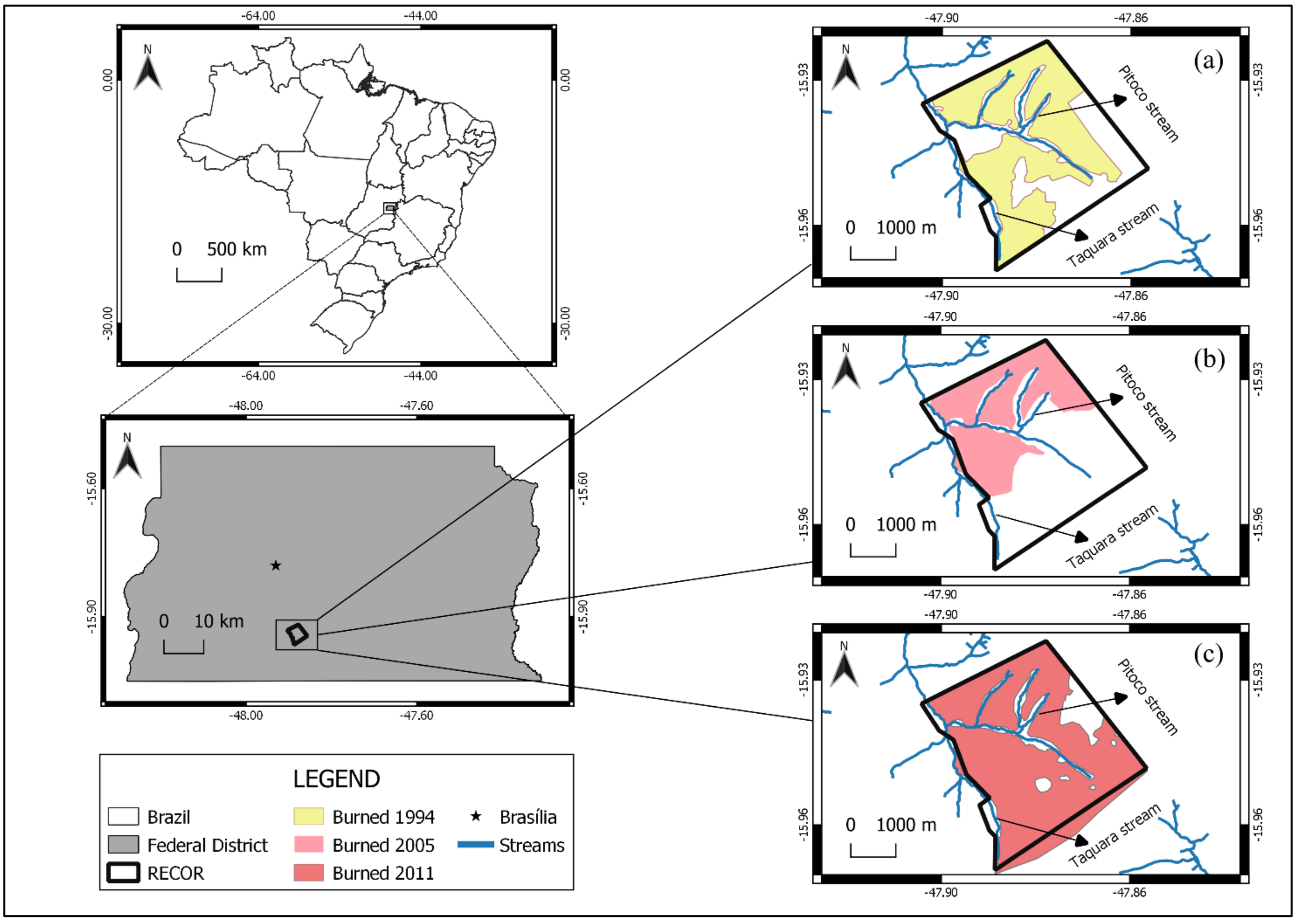

Figure 1. Study area and wildfires extension during three events (a) 1994, (b) 2005 and (c) 2011.

Currently, the riparian zones of Taquara (INV) and Pitoco (CSV) streams, objects of our study, present different disturbance degrees created by the invasion of *Pteridium arachnoideum*. While the former has large forest gaps and a high degree of biological invasion, the latter is preserved and the presence of the invaders does not occur. These areas are approximately two kilometers apart and they have similar topography and the same soil type: Haplic Histosol (Mamede et al., 2005). Based on soil analyses, no differences were detected regarding particle size and organic matter content. Both areas have clayey textured soils and the organic matter content varies between 3.9 and 4 $dag.Kg^{-1}$ (Table 1).

Table 1. Particle size analysis, organic matter content of soil samples collected at 20 cm of depth.

Sample	Sand (%)	Silt (%)	Clay (%)	Texture	O.M ($dag.Kg^{-1}$)
INV 1	16.4	29.8	53.8	Clay	3.9
INV 2	15.7	29.1	55.2	Clay	3.9
INV 3	14.0	32.3	53.7	Clay	4.0
CSV 1	6.5	35.3	58.2	Clay	4.0
CSV 2	6.3	36.8	56.9	Clay	3.9
CSV 3	5.4	35.9	58.7	Clay	4.0

SAMPLING DESIGN

We selected the study areas following a vegetation cover analysis based on the RGB (red green blue) color composition of a Landsat sensor scene. In this sense, the identification of land cover classes in the images was based on the spectral interpretation criteria that depend on the targets' reflectance behavior (ARAÚJO FILHO et al., 2007). Therefore, the result obtained in the color composition determined the linear transects sampling arrangement in both treatments: low disturbance riparian zone or control (CSV - Conserved) and high disturbance riparian zone (INV - Invaded).

We performed 30 infiltration tests in each treatment, distributed in three linear transects with ten infiltration points each (Salemi et al., 2013). The transects' location was approximately 30 meters away from the river channels. We chose the mini-disk infiltrometer (MDI) for measuring soil infiltration capacity (Decagon, 2012). The selection of this equipment was for its portability, simple maintenance, and efficient reading at low depth hydraulic conductivity, which allows a greater number of repetitions than other measurement methods (Fachin et al., 2016; González-Pelayo et al., 2010; Robichaud et al., 2008). During sampling, we used the 0 Kpa pressure in the MDI upper chamber to nullify the soil particles' matrix potential and to achieve the saturation conductivity that occurs at this pressure.

We performed the Water Drop Penetration Time (WDPT) test to determine the soil-water repellency (DeBano, 1981). This test consists of placing ten water drops on the soil surface to measure the average time until its complete infiltration (Fernández et al., 2019; Larson-Nash et al., 2018). Water repellency is classified as slight (drops remains 5 to 60 s), moderated (60 to 180 s), and severe (180 s or more) (Robichaud et al., 2016). In our work, we performed ten repetitions in each treatment, placing ten water drops in each measuring point, and its location followed the infiltration transects.

Additionally, we evaluated the soil mechanic resistance to penetration using an impact penetrometer (Stolf et al., 2014). Its functioning principle consists in the penetration of a metal-tipped rod into the ground by the impact of a piston of known mass at a constant height. (Carvalho et al., 2012). We performed ten repetitions with four impacts in each treatment following the infiltration transects.

Finally, using a Dutch Auger, we collected three composed soil samples in each riparian zone to analyze soil texture and organic matter content. The points were randomly selected within the infiltration transects limits. All sampling and analysis followed the guidelines of the Field Soil Description and Collection Manual (Santos et al., 2005) and the Soil Methods and Analysis Manual (Teixeira et al., 2017) published by Embrapa (Brazilian Agricultural Research Corporation).

STATISTICAL ANALYSIS

We used the Shapiro-Wilk statistic test to evaluate the soil's physical-hydric attributes normality. The infiltration capacity, resistance to penetration, and soil-water repellency variables presented a non-normal distribution. Thus, we performed the Mann-Whitney test to compare them in each treatment. All analysis was performed using the Paleontological Statistics software – PAST version 3.22 (Hammer et al., 2001).

RESULTS

In the invaded riparian zone (INV), the median infiltration capacity was 46.1 ± 96.4 $mm.h^{-1}$, whereas in the conserved riparian zone (CSV), it was 574.0 ± 219.9 $mm.h^{-1}$ (Figure 2). The infiltration capacity in INV was significantly different than CSV ($z= 6.44$; $p<0.01$).

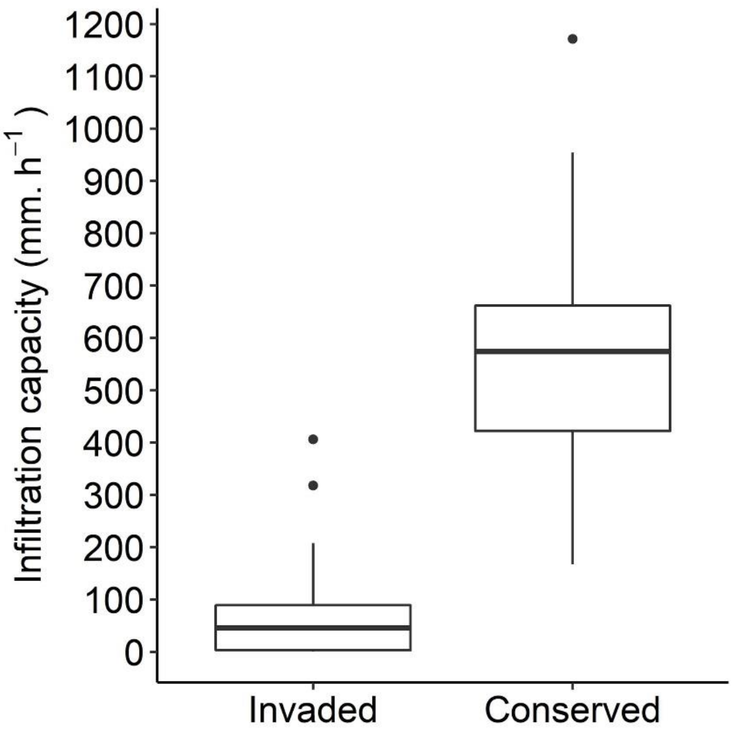

Figure 2. Boxplot exhibiting soil water infiltration capacity in a riparian zone invaded by *Pteridium arachnoideum* (INV) and another conserved (CSV). The horizontal line within the boxes represents the results' median. The boxes' horizontal boundaries represent the first and third quartiles. The ends of the vertical lines represent the maximum and minimum values, and the black dots represent the outliers.

Regarding the WDPT test, we found a significant difference between the two treatments (z= 3.69; p<0.01). The median time for the drops to completely penetrate the soil was 180.7 ± 13.3 s in INV and 98.4 ± 51.2 s in CSV (Figure 3). Following the hydrorepellence patterns established by Robichaud et al. (2016), the conserved area showed moderated soil water repellency whereas the invaded was severe. However, we point out that the values in INV are underestimated, as we stopped the time counting at the 180 seconds mark. In our work, 92% of INV's water drops and 38% of CSV's water drops remain on the soil surface for a time longer than 180 seconds. Usually, under these conditions, the water drops tend to evaporate before infiltrating into the soil (Hallett and Gaskin, 2007).

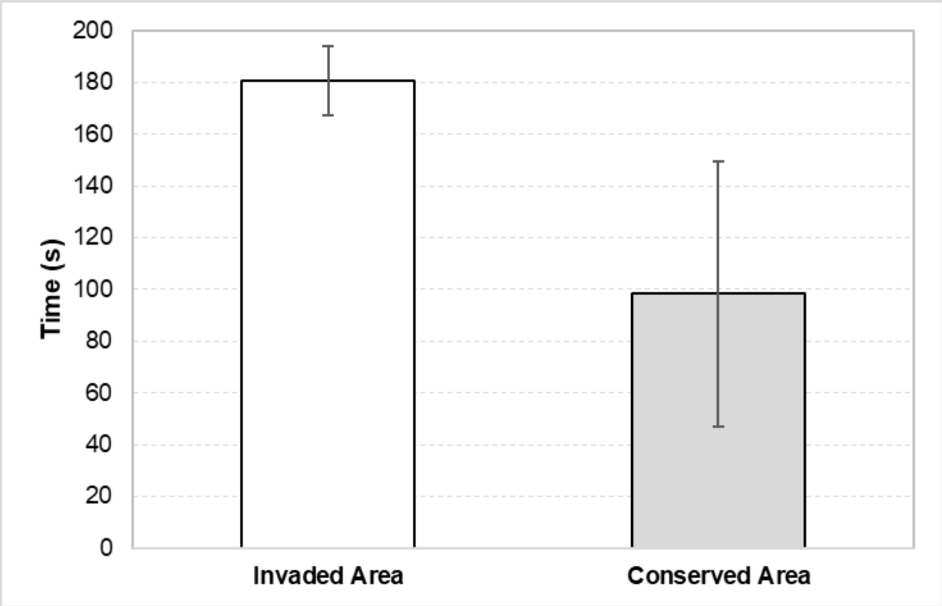

Figure 3. Average time for total infiltration of a waterdrop into the soils of a *Pteridium*-invaded riparian zone (INV) and a conserved riparian zone (CSV). The vertical lines within the bars represents the standard deviation of the samples.

Finally, regarding the soil penetration resistance, after the impacts 0, 1, 2, and 3, the median depth reached in INV was respectively 0.06, 0.12, 0.17, and 0.20 meters. In the CSV treatment, the average depth reached after the same impacts were equal to 0.10, 0.21, 0.29, and 0.36 meters (Figure 4). In all impacts, we observed that the depth reached in CSV was significantly higher than in INV ($z= 4.52$; $p<0.01$).

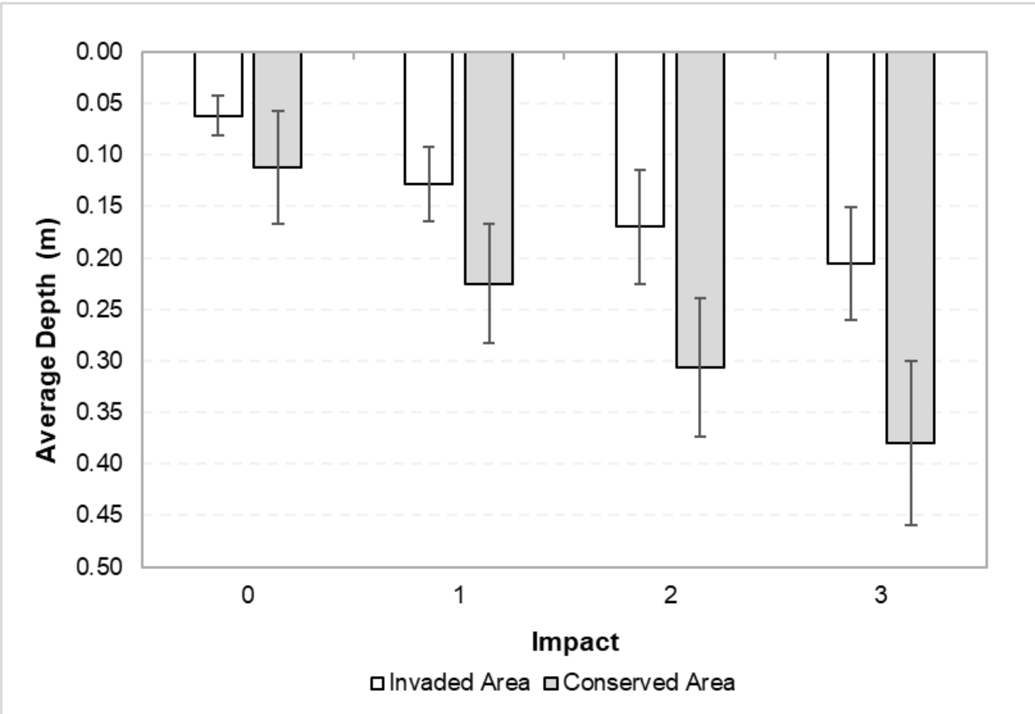

Figure 4. Depth reached (m) by penetrometer as a function of the number of impacts. The vertical lines within the bars represents the standard deviation.

DISCUSSION

We hypothesized that the riparian zone invaded by *Pteridium arachnoideum* (INV) would have lower soil permeability to water. In fact, our results confirmed it due to the significant reduction of soil infiltration capacity in INV compared to CSV. This reduction may be linked to two factors detected here: (a) higher soil penetration resistance, and (b) higher soil water repellency.

The higher soil penetration resistance in the invaded area indicates that there is a higher soil density and, consequently, lower porosity, and higher soil compaction. As a result, less water infiltrates into the soil in the invaded areas than in the conserved ones. Also, the conserved riparian zone presents higher species richness and, naturally, it has an extensive root distribution that increases the macropores quantity and decreases soil penetration resistance and, thus, allows more water to infiltrate into the soil (Fischer et al., 2015; Niemeyer et al., 2014; Su et al., 2018; Sun et al., 2018; Tabacchi et al., 2000). On the other hand, the soil penetration resistance increasing in INV may be related to two main factors: (i) the fire effects on soil and (ii) the characteristics of *Pteridium* radicular system. About the former, in general, changes in the physical attributes of soils are associated with the organic matter combustion, which is essential for maintaining soil structure (Úbeda and Outeiro, 2009). Usually, fire reduces soil particles' aggregation degree and this results in indirect impacts in the increasing of water repellency and the diminishing of soil water infiltration (Mataix-Solera et al., 2011; Scott et al., 2009; Úbeda and Outeiro, 2009). In some cases, soil penetration resistance of burned areas can be even five times greater than in conditions without fire (Wahlenberg et al., 1939). Besides that, the *Pteridium* roots morphology also may influence the change of soil penetration resistance. Because of its thin roots, its root system can thoroughly explore superficial soil layers forming a dense and intertwined layer of roots (Cody and Crompton, 1975; O'brien, 1963; Ziemer, 1981). Moreover, this plant has vigorous and long-lasting rhizomes (Lorenzi, 2008) that occupy large portions of the soil and promotes its displacement and densification. Results similar to ours are described by Boxell and Drohan (2009) that evaluating *Bromus tectorum* L. invasions after fires, found reduced infiltration capacity and increased soil penetration resistance in these areas. The authors concluded that water infiltration reduction occurred because invaded areas had less surface roughness due to loss of soil structure than natural ecosystems. Consequently, water flows faster across the surface, decreasing infiltration. Finally, they suggested that the increased penetration resistance may be related to differences in cementing agents such as organic matter and/or salts and calcium carbonate, and in the translocation of clay to deeper soil layers.

Besides all that we found that soil permeability reduction in the INV area is due to increasing soil hydrophobicity. Soil water repellency is caused by the covering of soil particles by organic substances that are, naturally, hydrophobic (Jiménez-Morillo et al., 2016; Zheng et al., 2016). These substances are common in vegetated soils and occur in all environments (Brady and Weil, 2013). Furthermore, its molecules are relatively resistant to physical, chemical, and biological degradation, which makes decomposition rate very slow (Brady and Weil, 2013; Doerr et al., 2000). It is important to highlight that soil hydrophobicity is not temporally stable and its origin is variable. In general, it depends on a set of attributes such as soil characteristics (e.g., structure, density, clay mineralogy and its ratio to organic matter), the growth of specific plant species, the activity of soil microorganisms, decomposition of organic matter, and soil heating by fires (Cerdà and Robichaud, 2009; DeBano, 2000a, 2000b; Jiménez Morillo et al., 2017; Keesstra et al., 2017; Mao et al., 2016; Olorunfemi et al., 2014). This phenomenon can be observed in diverse soil granulometry (Doerr et al., 2006, 2000; Leelamanie et al., 2010; Woche et al., 2005). Nevertheless, coarse-textured soils are more prone to hydrophobia development due to their smaller surface area compared to fine-textured soils (Doerr et al., 2006, 2000). In turn, organic soils, such as those observed in our study area, have a hydrophobic attribute that often results in reduced infiltration capacity (Loss et al., 2015; Pérez et al., 1998). Thus, it is possible to understand the reason why even the soils of the conserved riparian zone (CSV) showed some degree of water repellency.

It is noteworthy that hydrorepellency depends on specific combinations of organic compounds and their molecular arrangement in response to environmental conditions (Morley et al., 2005; Roy and McGill, 2000). Therefore, the difference between the hydrophobicity degree of the evaluated riparian zones occurs because some soil types and vegetation combinations are more prone to hydrorepellence development. Thus, the soils' hydrophobicity degree seems to be influenced more by organic matter quality than by its quantity (Cerdà and Robichaud, 2009; Olorunfemi et al., 2014). The increased soil water repellency in the INV area may be due to the *Pteridium* oiliness (Baker and Gaskin, 1987; Doerr et al., 2000; Guo et al., 2018) which, upon decomposition, releases these hydrophobic substances on the soil surface (Cerdà and Robichaud, 2009; Olorunfemi et al., 2014). According to the classification proposed by Robichaud et al. (2016), the soil hydrorepellency degree in the conserved riparian zone is moderate, while that of the invaded riparian zone is classified as severe. These results are similar to those found in other areas invaded by plants of the genus *Pteridium* (Jiménez-Morillo et al., 2016). Other experiments carried out in different vegetation types corroborate our results and highlight the hydrorepellency increasing after the establishment of other invasive species. For instance, Ruwanza et al. (2013) observed the hydrorepellency intensification and changes in the soil water infiltration in riparian zones invaded by *Eucalyptus camaldulensis*. In this case, the repellency increasing occurred by the accumulation of organic matter and eucalypt leaves, rich in phenolic acids and volatile oils, which are naturally hydrophobic (Eynard et al., 2004; Scott, 2000). In addition, periods of high temperatures and wildfires may exacerbate the volatilization and condensation of hydrophobic organic substances (Doerr et al., 2005; Malkinson and Wittenberg, 2011).

Despite the importance, the effects of plants of the genus *Pteridium* on riparian zones and watersheds hydrology remains virtually unknown. We found no studies similar to ours assessing the effect of these plants on soil infiltration capacity and, in most cases, researches address its effects solely on native species diversity (Guerin and Durigan, 2015; Jatoba, 2016; Matos and Pivello, 2009; Roos et al., 2010; Xavier et al., 2016). The work of Williams et al. (1987) is one of the few showing the effect of *Pteridium* on some hydrological aspects of invaded areas. According to the authors, the plants of this genus can have substantial implications for the hydrological cycle. For example, in the areas dominated by *Pteridium* plants the water loss through canopy interception can be 50% and, in some cases, depending on rainfall intensity can be up to 100%. By way of comparison, other studies concluded that canopy interception in riparian zones of the Brazilian Savanna ranges from 24.7 and 34.6% (Lima and Leopoldo, 2000; Távora, 2017), which may indicate the potential impacts of the *Pteridium* on hydrological processes. Furthermore, Williams et al. (1987) highlighted that water chemical composition after stemflow and litterflow are significantly different from rainfall water which impacts the nutrient cycling on ecosystems. Our study, in turn, indicates that water flowing through *Pteridium* canopy encounters unsuitable conditions to infiltrate and, therefore, long-term impacts may occur as the water that infiltrates into the soil replenishes groundwater aquifers that maintains the flow of watercourses during the drought period which makes improving on water infiltration conditions essential for increasing water availability and maintaining ecosystem services (Brandão et al., 2006; Pejchar and Mooney, 2009; Salemi et al., 2011). Moreover, the infiltration process is determinant for surface runoff and consequently interferes with the risk of erosions and floods. Thus, its knowledge is fundamental for soil and water quality management and conservation (Brandão et al., 2006).

Given the above, the changes promoted by species invasion in riparian zones are of great concern as these ecosystems perform key ecohydrological functions that are directly related to biological and biogeochemical cycles (Abe et al., 2016; Catford, 2017; Huddle et al., 2011; Ruwanza et al., 2013; Tundisi and Tundisi, 2010). Under natural conditions, riparian zones connect aquatic and terrestrial environments and perform key ecosystem functions, such as stabilization of riverbanks, protection against erosion e mass movements, reduction of soil particles transport (e.g. minerals, organic matter, and nutrients) for watercourses which influences aquatic communities, water quality, groundwater recharge, as well as increasing diversity within the aquatic environment (Salemi et al., 2016; Tickner et al., 2001). According to Pejchar and Mooney (2009), invasive plants alter the vegetation structural complexity by increasing water loss through interception and evapotranspiration and by the water storage in their tissues. Additionally, invasive plants with deeper radicular system absorb water in layers that natives cannot reach (Mooney, 2005; Pejchar and Mooney, 2009). Thus, the authors pointed out that these plants can reduce water production by up to 30%. Similarly, Tickner et al. (2001) reported that plants of the genus *Tamarix* significantly reduced the water table in arid regions. According to the authors, in addition to reducing water availability through the desiccation of watercourses, these plants can also affect water quality and impact species diversity.

Finally, it is important to highlight that the impacts generated by plant invasions go beyond biodiversity loss, however, its implications on hydrobiogeochemical cycles remain largely unknown (Huxman et al., 2005; Pierson et al., 2011; Zou et al., 2014). Therefore, it is critical that policymakers and scientists direct their attention to this major driver of global environmental change in the coming years.

CONCLUSION

We found that soil permeability to water reduced significantly after the invasion of *Pteridium arachnoideum* plants. We attribute these changes to two main factors: increased soil penetration resistance and hydrophobicity. In the long term, the presence or development of *Pteridium* is expected to intensify these impacts and further affect river basin hydrology, as this species is not restricted to riparian zones. Therefore, considering the colonization capacity of this invader, it is essential that eradication and control techniques are applied so that ecosystems may perform the ecohydrological functions.

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