



Impact of bonfires on soil properties in an urban park in Vilnius (Lithuania)

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ABSTRACT

Lighting bonfires in urban parks is a widespread practice. However, few studies have examined their impact on soil properties. The aim of this study was to analyze the impact of bonfires on the soil properties of an urban park in Vilnius, Lithuania. The properties studied were soil water repellency (SWR), aggregate stability (AS), soil organic matter (SOM) content, total nitrogen (TN), inorganic carbon (IC), pH, electrical conductivity (EC), extractable calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), aluminum (Al), manganese (Mn), iron (Fe), zinc (Zn), copper (Cu), boron (B), chromium (Cr), available phosphorus (P), silicon (Si) and sulfur (S). Soil ratios calculated were carbon/nitrogen ratio (C/N), $Ca + Mg/(Na + K)^{1/2}$ (SPAR), Ca:Al and Ca:Mg. Three areas were studied: Site 1) *Pinus sylvestris* L. and *Quercus robur* L.; Site 2) *Aesculus glabra* Wild.; and Site 3) *Pinus sylvestris* L. and *Acer plantanoides* L. At each site, 20 samples were collected (10 within the bonfire area, 10 from a control area). The results showed significantly higher values of SOM, IC, pH, EC, Ca, Mg, Na, K, P, Al, Zn, Cu, Cr, S, C/N ratio, Ca:Al ratio and Ca:Mg ratio in bonfire soils than in control unburned soils. In bonfire soils, significantly lower values were recorded of SWR, AS, TN, SOM, Al, Mn, Fe, Cr, S and SPAR comparing to control soils. Most affected area by the bonfire was Site 1, which presented a marked increase in heavy metal content comparing to the control. The impact of soil heating was evident in AS, IC, pH, extractable Ca, Mg, Na, K, P, Al, Zn, Cu, Cr, S, and in its ratios. Protective measures are needed to limit bonfires and prohibit the burning of hazardous materials.

1. Introduction

Lighting bonfires for cooking, transform raw materials and protecting from the cold and wild animals are a practice used for millennia since humans learnt to control fire in the Pleistocene. The control of fire and the usage of bonfires by early humans changed earth evolution and allowed us to survive as specie (Gibbs, 2015; Chazan, 2016; Sousa et al., 2018). Bonfires became an important part of human culture and are used in many religious rituals around the world (e.g. Magnunson, 2007; Ortiz et al., 2017; Walsham, 2017; Ahmed and Memmish, 2019). Often, they are used as well in street mass demonstrations and parades to protest against political decisions or in war areas (Schalliol, 2016; Martin, 2019). Overall, since pre-historic times, bonfires have been a form of human manifestation strongly embedded in our culture and traditions, assuming an important role in our evolution as a specie.

Bonfires are a common practice in urban parks for cooking or cultural activities (Sarah et al., 2015; Low and Iveson, 2016), however, these activities have detrimental in terms of both increased air pollution and the impact it has on human health, given the toxic smoke released

and the potential inhalation of PM_{2.5}, PM₁₀, carbon monoxide (CO), carbon dioxide (CO₂) and heavy metals. The release of these elements into the atmosphere is reported as being responsible for various diseases, including tuberculosis, asthma and rhinitis (Spiegel and Veiga, 2005; Vassura et al., 2014; Gautam et al., 2019). Waste burning is also responsible for the emission of toxic elements to the atmosphere in urban areas with important impacts on air quality (Yadav et al., 2017) and the contamination of soil and water resources (Wu et al., 2015; Ohajinwa et al., 2018). Very often, waste materials (e.g. tyres, metal cans, plastic bottles) are burned in bonfires, changing the chemical compositions of the materials released in the air, soil and water, when compared to biomass burning (Dyke et al., 1997; Liu et al., 2017; Borysiak et al., 2018; Gautam et al., 2019). Bonfires have also been reported as having a negative impact on vegetation structure (Zenin, 2012) and soil properties, as a consequence of prolonged soil heating (Stromgren, 1938; Dao et al., 2012; Golden et al., 2015).

Soil thermal conductivity is usually poor; however, bonfires can heat the soil for long periods at high temperatures radically changing soil properties (Golden et al., 2015). Indeed, prolonged periods of

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heating may result in the irreversible degradation of the soil affecting consequently vegetal recovery. It has been shown that soil heating during a bonfire is higher than that during a prescribed fire or wildfire, being comparable, in fact, to pile burns or smoldering fires when heating periods are of similar duration (Hubbert et al., 2015; Rhoades and Fornwalt, 2015). Nevertheless, the combustion process of untreated wood, treated wood, plastics, metals and other waste materials are different and may affect the flammability and the temperatures reached during the bonfire (Jin and Chung, 2016; Huron et al., 2017; Guo et al., 2018).

Soil heating is conditioned by several factors, including the contact period, the temperature attained, the type of soil, soil moisture, texture, organic matter content and the type of fuel combusted (Raison, 1979; Pereira et al., 2019a). Studies carried out in laboratory environment exposing soils to different temperatures indicate that heating has a variable impact on the concentration of soil elements. In a recent review of these studies, Pereira et al. (2019a) indicate that, normally, an increase in temperature and in the length of the contact period results in increased soil water repellency (SWR) up to temperatures of around 175–200 °C, while SWR is destroyed at temperatures above 270–300 °C. Additionally, they report that soil heating causes aggregate stability (AS) to either rise or fall and that it is destroyed at temperatures around 460 °C as a result of the combustion of organic matter. At temperatures higher than 500 °C new cement agents are formed including the thermal fusion of clay minerals and the transformation of aluminum (Al) and iron (Fe) oxides (Pereira et al., 2019a).

Soil total nitrogen (TN) and organic matter (SOM) fall with increasing temperature and contact time. At temperatures higher than 450 °C, the presence of TN and SOM is residual. The same trends are observed in the C/N ratio (Bará and Vega, 1983; Chandler et al., 1983). In contrast, inorganic carbon (IC) levels rise with increasing temperature (Aznar et al., 2016) and similar trends are observed in pH and electrical conductivity (EC), especially after exposure to temperatures of 300 °C (Henig-Sever et al., 2001; Pereira et al., 2019a). The major elements such as phosphorus (P) and sulfur (S) need high temperatures (> 550 °C) to be volatilized. This temperature there are no direct losses of these elements, however, their solubility seems likely to increase given that heating has a mineralization effect on organic matter, increasing its mobility. The solubility of these elements is, above all, dependent on pH levels and the presence of calcium carbonate (Qian et al., 2009; Pereira et al., 2019a).

Soil ratios serve as an important proxy for understanding soil conditions. For example, the C/N ratio is an indicator of organic matter decomposition (Ostrowska and Porebska, 2015) and normally decreases with exposure to heat as a consequence of a mineralization effect (Pereira et al., 2019a). The soil sodium and potassium adsorption ratio (SPAR), an indicator of soil salinity (Sarah, 2004), that normally increases in burned areas, as a consequence of organic matter mineralization and increased solute transport (Pereira et al., 2017). The Ca:Mg ratio is an indicator of soil structure and influences infiltration and the availability of other nutrients such as N, P and copper (Cu) (Favaretto et al., 2012; Lombini et al., 2003; Manimel Wadu et al., 2013). The Ca:Aluminium (Al) ratio is an indicator of soil fertility and ecosystem stress as a consequence of acid deposition. In addition, high levels of Al affect plant nutrient uptake (Cronan and Grigal, 1995), the distribution of species (Roem and Berendse, 2000) and earthworms (Dobson et al., 2017).

Previous studies have highlighted the finding that bonfires increase the heavy metal content of soils and soil acidity (Dao et al., 2012). Indeed, some have specifically focused on heavy metals and magnetic susceptibility (Dao et al., 2012; Golden et al., 2015), while others have examined the impact of bonfires on soil physical properties, major cation concentrations and soil ratios (Arnesen, 1999). Nevertheless, the bonfires studied did not consider the impact of the different fuel's characteristics and ignored the time after the bonfire. This is important since vegetation have different vulnerabilities to fire and produce

different temperatures and impacts on soil. In addition, the time after the bonfire is key to understand the direct impacts (soil heating), and this can be detected in the immediate period after the bonfire, an aspect that was overlooked by the previous works. The direct impacts of bonfire are key to understand the type of ash and char produced and therefore the indirect effects (ash incorporation). The severity of burning will determine the type and amount of nutrients released in solution and will affect soil quality and water resources (Pereira et al., 2019a). More research is needed to understand the impact of lighting bonfires, a common practice in urban parks and recreation areas, but a recognized source of soil pollution (Zenin, 2012). There is clearly a need to understand these impacts to determine just how these practices can affect soil environment. Here, the aim of our study is to analyze the direct impact (soil heating) of recent bonfires on soil properties. More specifically, we seek to understand how bonfires can modify a) a soil's physical characteristics, b) its chemical properties and c) soil ratios in areas of different vegetation cover.

2. Material and methods

2.1. Study area and sampling design

The study area is located in Vingis park in Vilnius, Lithuania (54°41'N, 25°14'E and 130 m a.s.l.), the largest urban park in the city, occupying an area of 162 ha. (Fig. 1). Although illegal, bonfires are quite usual occurrences, especially on weekends and during festivals held in the park. They are especially common in spring and summer period when the weather conditions are more favorable for outdoor recreation activities. The park's geology comprises both pre-Quaternary parent material, composed primarily of chalk from the Cretaceous period, and post-Quaternary parent material formed by sand and alluvial sediments from the Holocene. Vilnius has a mean annual temperature of 8.8 °C and a mean annual rainfall of 735 mm (Pereira et al., 2014). The park's soils are classified as Albeluvisol (WRB, 2006). The park's most common tree species is *Pinus sylvestris* L.; however, other species are found there, including, *Quercus robur* L., *Acer plantanoides* L. and *Aesculus glabra* Wild. In this study, three different sites were selected and sampled. For this study, we selected fresh bonfires, lightened during the weekend prior to the sampling date. Samples were collected on Monday 19 June 2017 and bonfires burning the previous weekend. No rainfall fell in this period. All the sites had a similar topography (flat area). Bonfires were visually assessed for the presence of waste material (e.g. plastic bottles, cans). This was identified in the site 1. The bonfires selected for this work had a diameter between 2 and 4 m' radio. At each site, ten undisturbed soil samples were collected within the area affected by the bonfire and a further ten samples from the sites corresponding control (not burned previously) area (0–5 cm depth) with a steel cylinder (5 × 10 cm). Soils were sampled in the middle of the bonfire (where the impacts are high) with a distance of 20 cm between them. Previous to sampling, the burned organic material was removed in order to assess only the direct impacts of the burning, avoiding the impacts of ash and/or char. The control plot was established 8 m from the edge of the bonfires, in an area with the same vegetation. Soil sampling procedure was the same carried out in the bonfires plot. In total, we collected 60 samples. The plots were located as follows: Site 1: mixed forest of *Pinus sylvestris* L. and *Quercus robur* L.; Site 2: *Aesculus glabra* Wild. trees; and Site 3: mixed forest of *Pinus sylvestris* L. and *Acer plantanoides* L. (Fig. 1).

2.2. Laboratory analysis

Soil samples were dried for 7 days at room temperature (23 °C) and, then, sieved using three different mesh-sizes: 4.8 mm and 4 mm to separate out soil aggregates and 2 mm to analyze fine aggregates. SWR was analyzed using the water drop penetration time (WDPT) (Wessel, 1988) and classified according to Doerr (1998) into four categories:

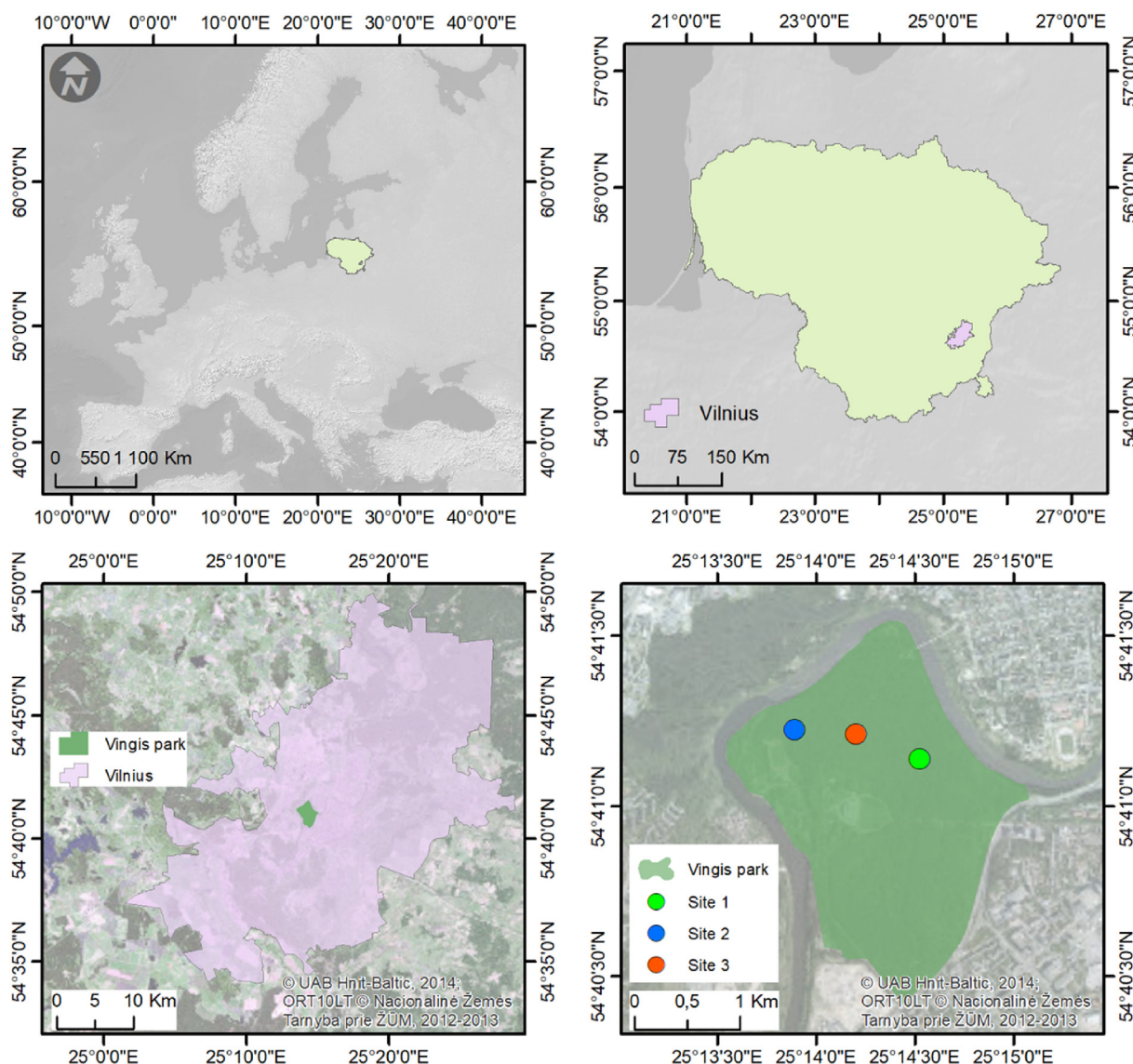


Fig. 1. Study area.

wettable (< 5 s), low (6–60 s), strong (61–600 s) and severe (601–3600 s). Six drops of deionized water (0.05 ml) were dropped on to the top of each sample (< 2 mm) and the time required for each drop to completely penetrate the soil was recorded. The average time recorded by the six drops was calculated as the WDPT for each sample. To analyze soil AS, the ten drop impact (TDI) method was used on aggregates measuring between 4 and 4.8 mm (Low, 1954). For each sample, ten air-dried aggregates were placed over a 2.8-mm sieve. Each aggregate was weighed before being subjected to the impact of ten drops of deionized water (0.1 ± 0.001 g) from a height of 1 m. The disaggregated material passing through the sieve was then weighed and dried at 105 °C for 24 h. Results are expressed as a percentage of the aggregate weight that remained stable after the TDI test.

TN was analyzed using a Flash 112 Series (Thermo-Fisher, Milan) and data calculations were carried out with Eafar 300 software (Pereira et al., 2012). SOM and IC were measured using the loss-on-ignition method (Heiri et al., 2001). For each sample, 1 g of pulverized soil was used and dried at 105 °C for 24 h in a muffle furnace. To determine SOM, dried samples were subjected to a temperature of 550 °C for 4 h. To calculate IC, the samples were exposed to a temperature of 950 °C for 2 h. Soil pH [1:2.5] and EC (expressed in $\mu\text{S}/\text{cm}$) [1:2.5] were analyzed with an extraction of deionized water. Extractable major

cations (Ca, Mg, Na and K) were analyzed using an extraction [1:20] of ammonium acetate (Knudsen et al., 1982). Soil extractable Al, manganese (Mn), Fe, zinc (Zn), Cu, boron (B), chromium (Cr), silicon (Si), and S were analyzed using an extraction [1:20] of ammonium acetate. Available P was analyzed following the Olsen Gray method (Olsen et al., 1954). Extractable cations and P were expressed in mg/Kg of soil and analyzed by inductively coupled plasma mass spectrometry (ICP-MS), using a PerkinElmer Elan-6000 Spectrometer and a PerkinElmer Optima-3200 RL Spectrometer.

The soil C/N ratio was calculated as the proportion of organic carbon to TN. Organic C content in SOM was calculated as follow: $\text{Organic C} = \text{SOM}/1.724$ (Al-Gburi et al., 2017). Soil SPAR was calculated according to the formula proposed by Sarah (2004) i.e. $\frac{(\text{Extractable Na} + \text{Extractable K})}{(\text{Extractable Ca} + \text{Extractable Mg})^2}$, the concentration being expressed in mmol/l. The Ca:Al and Ca:Mg ratios were also calculated.

2.3. Statistical analyses

Data normality and homogeneity of the variances were assessed using the Shapiro-Wilk and Levene's tests. Data were considered normally distributed and heteroscedastic at $p > 0.05$. We applied a one-way ANOVA test in those cases where data followed a Gaussian

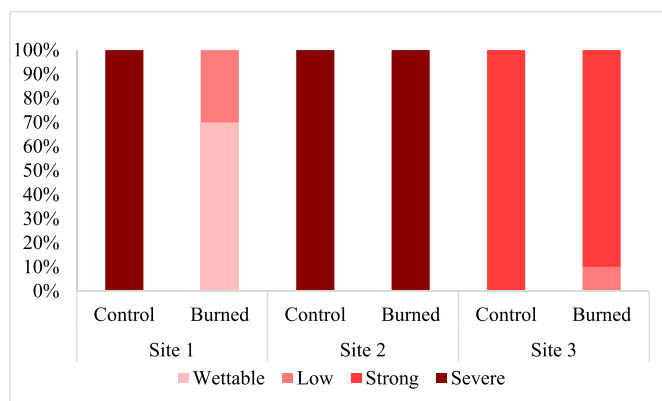


Fig. 2. Relative frequency of SWR at the study sites.

distribution and respected the homogeneity of the variances, while in those cases in which the data did not respect normality and homogeneity, we applied the non-parametric Mann-Whitney test. If significant differences were identified at $p < 0.05$, a Tukey post-hoc test was applied in order to identify differences within the soil affected by bonfires and the control. A principal component analysis (PCA) was carried out to identify the relations between the variables. Statistical analyses were implemented using SPSS 23.0 and XLSTAT 2017.

3. Results

3.1. Soil water repellency, aggregate stability, total nitrogen, soil organic matter, inorganic carbon

Soil water repellency ranged between strong and severe at all three control sites. Bonfires reduce SWR, above all at Site 1 (Fig. 2), and in all three cases SWR was significantly higher in soil controls than in those affected by bonfires. Soil AS was significantly higher in the control soils of Sites 1 and 3 than in their bonfire soils; but, there was no significant difference in AS at Site 2. Soil organic matter was significantly higher in the bonfire soils than in the corresponding controls at Site 1. At Site 2, SOM resulted significantly higher in control than in bonfire soils. No significant differences were observed at site 3 in SOM. Soil IC was significantly higher in the area affected by the bonfire at Site 1, whereas there were no significant differences in IC between areas at Sites 2 and 3. Total nitrogen was significantly higher in control soils than in bonfire soils at Site 2; however, at Sites 1 and 3 there were no significant differences between control and bonfire soils (Table 1).

3.2. pH, electrical conductivity, major elements and available phosphorus

Soil pH and EC were significantly higher in the areas affected by the bonfire than in their control soils at Sites 1, 2 and 3. Extractable Ca, Mg and Na were significantly higher in bonfire than control areas, but only significantly at Site 1. At Sites 2 and 3 there were no significant differences in extractable Ca, Mg and Na. Finally, extractable K and available P were significantly higher in bonfire than in control soils at Sites 1 and 2, while at Site 3 no significant differences were recorded (Table 2).

3.3. Minor elements

At Site 1, extractable Al was significantly higher in bonfire than in its control soils, but significantly lower in bonfire than in control soils in Sites 2 and 3. Soil extractable Mn was significantly higher in control than in bonfire soils at Site 2, while there were no significant differences between control and bonfire soils at Sites 1 and 3. Soil extractable Fe was significantly higher in control than in bonfire soils at Sites 1 and 2; no significant differences were recorded at Site 3. Soil Zn and Cu

were significantly higher in bonfire than in control soils at Site 1, while there were no significant differences in either soil property between areas at Sites 2 and 3. No significant differences in extractable B and Si were recorded at any of three sites. Soil extractable Cr and S were significantly higher in soils affected by bonfires than those of their respective controls at Sites 1 and 3 but significantly higher in the control than in bonfire soils at Site 2 (Table 3).

3.4. Soil ratios

The C/N ratio was significantly higher in soils affected by a bonfire than in its corresponding control soils at Site 1, while at this same site SPAR was significantly higher in control than in bonfire soils. No significant differences in C/N and SPAR were recorded at Sites 2 and 3. Significantly higher values of the Ca:Al ratio were recorded in the bonfire soils at Sites 1, 2 and 3 compared to their respective control soils. Significantly higher values of the Ca:Mg ratio were also observed in the bonfire than in control soils at Sites 1 and 2. No significant differences were observed in the Ca:Mg ratio between bonfire and control soils at Site 3 (Table 4).

3.5. Multivariable analysis

Principal component analysis allows us to determine just how different each soil is at the three sites following exposure to the bonfire. Factor-1 in the PCA explains 36.9% of the variance while Factor-2 explains 24.7%, with 61.6% of the total variance being explained. The variables with the greatest explanatory capacity are SOM, TN, SPAR, Ca and pH, while the properties with the least explanatory capacity are B, EC and Mg (Fig. 3). Thus, at Site 1, PCA clearly separates the soils affected by bonfire from its controls given the high number of differences between these two areas of the site. At Site 2, PCA also separates the respective areas but not as clearly as at Site 1. Finally, at Site 3, given the few differences between the properties of its burnt and control soils, the PCA arrows appear very close together. C/N, Cu, IC, Zn, Cr, Ca, Ca:Mg and P are closely associated with the burned area of Site 1, while no soil properties are associated with its control area. Mg, K, Al and S are closely associated with the burned soils of Site 2, while AS, SWR, Mn, EC, Na, SOM, TN and Fe are associated with its control soils. Finally, soil pH, Si, Ca:Al and B are associated with both areas in Site 3.

4. Discussion

4.1. Soil water repellency, aggregate stability, total nitrogen, soil organic matter, inorganic carbon

Soil water repellency presented significantly higher values in control soils than in soils affected by bonfires irrespective to the vegetation cover. Soil water repellency occur naturally and it has been reported that can reach high values in acid soils covered by *Pinus sylvestris* (Iovino et al., 2018). Combustion process can increase or decrease soil water repellency. Soils heated until 175–200 °C increase soil water repellency, being destroyed at temperatures higher than 270–300 °C (Pereira et al., 2019a). Doerr et al. (2004) determined that the removal of repellent compounds is caused by the volatilization or oxidation of these compounds. Heating normally decreases repellency in water repellent soils. In addition, long heating periods decrease faster SWR than short ones. To our knowledge there were no studies about the impact of pile burns or bonfires on SWR, however we hypothesize that the decrease of soil water repellency in bonfire affected soils was a consequence of the high temperatures reached (> 270–300 °C) and the long period of contact. According to Tylecote (1962), the normal temperature during bonfire is 400 °C, however a previous reports found that high temperatures can be reached (600–900 °C) (Golden et al., 2015).

Aggregate stability fell significantly as a result of the bonfires at

Table 1Descriptive statistics of physico-chemical characteristics. Different letters represent significant differences at a $p < 0.05$. $N = 10$.

Soil property	Study site	Plot	Mean	SD	df	F value	p value
Soil Water Repellency (Sec.)	Site 1	Bonfire	3.49	3.27	18	469323.57	** ^{a,b}
		Control	10813.50	49.79			
	Site 2	Bonfire	6043.20	136.54	18	9945.28	** ^a
		Control	10846.90	67.52			
	Site 3	Bonfire	113.90	44.77	18	356.42	** ^{a,b}
		Control	426.25	27.07			
Aggregate Stability (%)	Site 1	Bonfire	74.71	7.28	18	38.01	** ^a
		Control	93.12	6.02			
	Site 2	Bonfire	97.19	3.59	18	0.27	n.s. ^a
		Control	97.81	1.27			
	Site 3	Bonfire	69.34	8.43	18	14.09	** ^a
		Control	84.03	9.05			
Soil Organic Matter (%)	Site 1	Bonfire	13.57	4.89	18	17.73	** ^a
		Control	6.20	2.59			
	Site 2	Bonfire	31.97	1.12	18	42.49	** ^a
		Control	34.98	0.97			
	Site 3	Bonfire	3.22	0.70	18	1.11	n.s. ^a
		Control	3.61	0.82			
Total Nitrogen (%)	Site 1	Bonfire	0.24	0.08	18	1.59	n.s. ^a
		Control	0.32	0.15			
	Site 2	Bonfire	2.10	0.19	18	10.09	** ^a
		Control	2.35	0.14			
	Site 3	Bonfire	0.25	0.09	18	0.64	n.s. ^a
		Control	0.29	0.11			
Inorganic Carbon (%)	Site 1	Bonfire	7.22	3.27	18	45.73	** ^{a,b}
		Control	0.21	0.26			
	Site 2	Bonfire	0.44	0.47	18	0.03	n.s. ^a
		Control	0.51	1.06			
	Site 3	Bonfire	0.52	0.39	18	0.21	n.s. ^a
		Control	0.45	0.36			

Three different sampling sites. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. The degrees of freedom (df) = the total number of subjects in the experiment – the number of groups.

^a One-way ANOVA.

^b Mann-Whitney.

Sites 1 and 3. Heating can decrease or increase AS. High temperatures are responsible for the combustion of SOM and AS reduction (Mataix-Solera et al., 2011). Soil aggregates are destroyed at the temperature of 220 °C, close to the temperature that SOM starts to be combusted (200 °C). Nevertheless, when soils are exposed to temperatures above 500–700 °C, there is an increase of soil aggregation due to the transformation of Al and Fe minerals and the collapse/destruction and thermal fusion of clay minerals (Pereira et al., 2019a). The causes for AS reduction in sites 1 and 3 were very likely attributed to SOM combustion and showed us that the temperatures reached by these bonfires were not high enough to increase AS. Aggregate stability depends on SOM content that is an important cement agent (Mataix-Solera et al., 2011). In the site 1 there was a significant increase of SOM on the soils affected by the bonfire compared to the control (discussed below). These results are a paradox. We hypothesized that despite the increase of SOM, the remaining organic matter was poor in carbohydrate content, essential to AS (Díaz-Fierros et al., 1987; Soto et al., 1991). Previous works highlighted that heating decreases importantly soil carbohydrate content and SOM stability (Knicker et al., 2005; Miesel et al., 2015). In relation to the site 2, the lack of significant differences in AS between control and bonfire affected soils may be attributed to the fact that heating temperatures and the consequent decrease of SOM (from 34.98 to 31.97%, discussed below) were not high enough to reduce significantly AS. The sites 1 and 3 were the impacts on soil AS were high and this may be attributed to the usage of *Pinus sylvestris* L., *Quercus robur* L. and *Acer plantanoides* L. in these bonfires. *Pinus sylvestris* L. is a flammable specie as consequence of the high presence of oils and resins (Adamek et al., 2016). Despite the lower values of flammable compounds comparing to *Pinus* species, *Quercus robur* L. is considered also a flammable specie (Nunez-Regueira et al., 2004). To our knowledge no study was carried out about *Acer Plantanoides* L. Species and materials

with high flammability have been reported to have a high heat release rate and peak heat release rate (Borysiak et al., 2006; Das et al., 2017) and therefore a high capacity to heating the soil, despite the fact that the great majority of the heat during a fire is transferred to the atmosphere by convection than into the soil, since soil is a poor conductor of energy (Smits et al., 2016). This may explain the high impacts of the bonfires in site 1 and 3 on AS. To our knowledge, no research was carried out on the flammability of *Aesculus glabra* Wild., nevertheless, we hypothesized that it may be lower than *Pinus sylvestris* L. and *Quercus robur* L.

Soil organic matter increased significantly in bonfire respect to unburned soils in the site 1, decreased in site 2 and no differences were observed in site 3. Soil heating induces a decrease of SOM as observed in previous works (Varela et al., 2010; Armas-Herrera et al., 2016). Soil organic matter volatilize at temperatures around 200 °C and at 450 °C, most of the SOM is consumed. Longer is the period of exposition lower is the temperature required to combust SOM (Pereira et al., 2019a). The significant reduction of SOM in site 2 (covered by *Aesculus glabra* Wild.), is explained by the impacts of soil heating. Nevertheless, a different dynamic was observed in site 1 (*Pinus sylvestris* L., and *Quercus robur* L.) and site 3 (*Pinus sylvestris* L. and *Acer plantanoides* L.). The increase of SOM in site 1 can be attributed to the increase of very fine and microscopic particles ash (Gonzalez-Perez et al., 2004; Mastrolonardo et al., 2017) that could be incorporated during or immediately after the combustion process. Is very likely that this material is especially inorganic (discussed below in with IC results). We removed manually all ashes, however, it is possible that microscopic particles may have remained and be incorporated into the soil. These particles were incorporated into the soil matrix and may have increased the content of non-mineral material. In this context, this increase do not reflect the increase of SOM, but residues of the combustion process. In

Table 2

Descriptive statistics of pH, electrical conductivity, major elements and available phosphorus characteristics. Different letters represent significant differences at a $p < 0.05$. $N = 10$.

Soil property	Study site	Plot	Mean	SD	df	F value	p value
pH	Site 1	Bonfire	7.72	0.38	18	124.42	*** ^b
		Control	5.18	0.61			
	Site 2	Bonfire	4.15	0.54	18	5.15	* ^a
		Control	3.75	0.18			
	Site 3	Bonfire	6.74	0.61	18	6.07	* ^a
		Control	6.11	0.54			
EC ($\mu\text{S}/\text{cm}$)	Site 1	Bonfire	472.40	167.09	18	30.99	*** ^b
		Control	142.90	84.36			
	Site 2	Bonfire	4318.50	2865.94	18	15.89	*** ^b
		Control	696.20	204.96			
	Site 3	Bonfire	274.40	116.68	18	13.00	*** ^b
		Control	137.40	28.68			
Extractable Ca (mg/Kg)	Site 1	Bonfire	22860.70	4437.20	18	224.32	*** ^b
		Control	1767.20	383.80			
	Site 2	Bonfire	5296.70	3299.62	18	3.12	n.s. ^a
		Control	3328.20	143.35			
	Site 3	Bonfire	2593.90	672.73	18	1.76	n.s. ^a
		Control	2181.70	717.14			
Extractable Mg (mg/Kg)	Site 1	Bonfire	316.20	143.07	18	12.80	** ^b
		Control	149.60	34.85			
	Site 2	Bonfire	282.50	143.35	18	0.05	n.s. ^a
		Control	270.60	89.06			
	Site 3	Bonfire	263.30	67.07	18	3.37	n.s. ^a
		Control	208.00	68.33			
Extractable Na (mg/Kg)	Site 1	Bonfire	121.20	34.19	18	4.55	* ^a
		Control	91.80	27.00			
	Site 2	Bonfire	183.00	46.11	18	0.15	n.s. ^a
		Control	175.90	35.99			
	Site 3	Bonfire	104.20	31.05	18	1.97	n.s. ^a
		Control	81.60	40.37			
Extractable K (mg/Kg)	Site 1	Bonfire	420.50	287.17	18	10.24	*** ^b
		Control	127.60	35.67			
	Site 2	Bonfire	454.60	190.69	18	11.83	** ^b
		Control	236.40	62.21			
	Site 3	Bonfire	192.40	32.76	18	3.58	n.s. ^a
		Control	161.10	40.77			
Available P (mg/Kg)	Site 1	Bonfire	325.50	240.96	18	8.84	** ^b
		Control	95.40	42.42			
	Site 2	Bonfire	106.80	31.71	18	26.50	*** ^b
		Control	46.20	19.50			
	Site 3	Bonfire	35.70	24.06	18	0.13	n.s. ^a
		Control	31.70	25.63			

Three different sampling sites. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. The degrees of freedom (df) = the total number of subjects in the experiment – the number of groups.

^a One-way ANOVA.

^b Mann-Whitney.

the study area, the soils are rich in sand and therefore the incorporation of these particles was facilitated (Pereira et al., 2013).

Soil total nitrogen fell significantly after the bonfire at Site 2 but did not change at sites 1 and 3. In all the cases there was a reduction, however, this was more evident in the bonfire carried out in *Aesculus glabra* Wild. cover, though the differences were minimal. The fall in TN appears to be the result of the volatilization of the organic nitrogen present in soil (Johnson et al., 2009). The volatilization of soil nitrogen has been reported to be initiated at 200 °C, and to fall further at higher temperatures (Chandler et al., 1983). At temperatures higher than 500 °C the presence of TN in soils is residual (Pereira et al., 2019a). This is an evidence that despite the significant decrease of TN in site 2, it is very likely that the temperatures did not reach the 500 °C in the first cm of soil in any of the bonfires studied. The incorporation of micro-particles of ash and charred material, could mitigate the losses or increase TN content in site 1. Nevertheless, is very likely that they were poor in this element. The levels of IC increased markedly following the bonfire at Site 1 but underwent no change at Sites 2 and 3. The increase in site 1 (*Pinus sylvestris* L. and *Quercus robur* L.) was attributed to the

incorporation of ash rich in IC (Úbeda et al., 2009). Soil heating can increase the amount of IC as consequence of the transformation of organic carbon (Aznar et al., 2016). Nevertheless, this only occurs at high temperatures of combustion (> 450–500 °C) (Pereira et al., 2019a), and as stated previously is very likely that these temperatures were not registered in the soils affected these bonfires. This can explain the lack of differences observed in the sites 2 and 3.

4.2. pH, electrical conductivity, major elements and available phosphorus

Soil pH and EC increased significantly after fire at all three bonfire sites and this was attributed to the mineralization of the organic matter and to the denaturing of organic acids during fire (Johnson et al., 2011; Pereira et al., 2019a). Similar results were observed in slash pile burning (Jimenez-Esquilin et al., 2007). Soil pH increases at temperatures higher to 300 °C, lower temperatures do not affect soil pH (Pereira et al., 2019a). Soil heating produced the increment of pH attributed to colloids dehydration and then falls in soil buffer capacity (Masyagina et al., 2016). The higher increase of soil pH in site 1 was very likely attributed to ash incorporation (Pereira et al., 2019a). In the case of EC, the increment in bonfire soils is due to the release of anions and major cations from the combustion of organic matter and its mineralization (Knicker, 2007). Soil EC increase at moderate temperatures between 300 °C and 450 °C (Quintana et al., 2007; Úbeda et al., 2009). Soil extractable Ca, Mg and Na increased significantly in the soils affected by the bonfire at Site 1, however, levels of these elements increased but not significantly at Sites 2 and 3. In all the cases there was an increment, nevertheless, this was significant in the bonfire carried out in *Pinus sylvestris* L. and *Quercus robur* L. cover. This was attributed to the increase of soil pH that increased the mobility of these major elements. In the site 2 and 3, we did not observed any significant difference as consequence of the low pH (Table 2) (Neary et al., 2005).

Soil extractable K and available P increased significantly in the bonfire soils at Sites 1 *Pinus sylvestris* L. and *Quercus robur* L.) and 2; however, there were no differences in Site 3. The increment of extractable K and available P were recorded in all sites but was only significant in site 1 (cover by *Pinus sylvestris* L. and *Quercus robur* L.) and in site 2 (cover by *Aesculus glabra* Wild.). The increase of extractable K and available P in the in site 1 was due the increase of soil pH. Nevertheless, the increase in site 2 is not explained only by the pH since the values are rather low, despite the significant, but small increase (Table 1). Potassium is a very mobile element and increases their extractability with pH. Phosphorous is less mobile than K, since at low pH levels precipitate with Al and Fe oxides and hydroxides and at high pH precipitates with carbonates (Gerardo et al., 2015; Xiaoyang et al., 2018). The increase of extractable K and available P may be related also with the decrease of IC, despite the fact that was not statistically significant. Previous works observed that carbonate surfaces have the capacity to capture K (Kassa et al., 2019) and P (Mihoub et al., 2016; Jia et al., 2017) in solution, reducing their presence in water extracts. In this context, the reduction of IC in the burned soils in site 2, may have increased the presence in solution of these two elements. Finally, the absence of significant changes in site 3, may be attributed to the residual increase of pH level and decrease of IC.

4.3. Minor cations

Soil Al increased significantly at Site 1 and fell at Sites 2 and 3 after the bonfires. In site 2 and 3 this was attributed to the increase of soil pH. Previous studies observed a decrease of extractable Al with pH increase (Clapman and Zibilske, 1992). The temperatures of combustion for Al volatilization are extremely high (> 2467 °C) (Pereira et al., 2019b), therefore is very likely that reduction is only attributed to pH effects. The increase of extractable Al in site 1 are very likely to the presence of cans (Fig. 4) in this bonfire, which increased the levels of extractable Al in soil. Heavy metals content in soils can increase after

Table 3Descriptive statistics of minor element characteristics. Different letters represent significant differences at a $p < 0.05$. $N = 10$.

Soil property	Study site	Plot	Mean	SD	df	F value	p value
Extractable Al (mg/Kg)	Site 1	Bonfire	51.15	47.28	18	5.10	** ^b
		Control	17.07	6.61			
	Site 2	Bonfire	28.76	12.31	18	13.77	** ^a
		Control	45.66	7.47			
	Site 3	Bonfire	3.62	0.76	18	14.38	* ^a
		Control	5.06	0.94			
Extractable Mn (mg/Kg)	Site 1	Bonfire	47.03	12.73	18	3.05	n.s. ^a
		Control	36.55	14.05			
	Site 2	Bonfire	47.02	39.25	18	15.08	*** ^b
		Control	219.38	134.77			
	Site 3	Bonfire	63.01	9.44	18	0.18	n.s. ^a
		Control	59.48	24.30			
Extractable Fe (mg/Kg)	Site 1	Bonfire	1.78	0.96	18	23.40	*** ^b
		Control	10.93	5.90			
	Site 2	Bonfire	40.74	27.93	18	11.35	** ^a
		Control	76.71	18.96			
	Site 3	Bonfire	2.81	0.65	18	0.33	n.s. ^a
		Control	3.05	1.11			
Extractable Zn (mg/Kg)	Site 1	Bonfire	880.02	425.69	18	42.03	*** ^b
		Control	7.25	2.08			
	Site 2	Bonfire	23.70	9.84	18	3.81	n.s. ^a
		Control	16.08	7.43			
	Site 3	Bonfire	3.48	0.92	18	0.04	n.s. ^a
		Control	3.38	1.14			
Extractable Cu (mg/Kg)	Site 1	Bonfire	40245.81	26672.26	18	22.69	*** ^b
		Control	66.06	85.02			
	Site 2	Bonfire	21.12	10.30	18	1.14	n.s. ^a
		Control	17.04	6.37			
	Site 3	Bonfire	2.38	0.42	18	4.66	n.s. ^a
		Control	2.23	1.18			
Extractable B (mg/Kg)	Site 1	Bonfire	31.80	9.83	18	0.28	n.s. ^a
		Control	36.00	23.32			
	Site 2	Bonfire	17.06	10.22	18	0.08	n.s. ^a
		Control	18.38	10.57			
	Site 3	Bonfire	37.80	22.73	18	0.03	n.s. ^a
		Control	35.97	28.01			
Extractable Cr (mg/Kg)	Site 1	Bonfire	4.44	2.45	18	30.02	*** ^b
		Control	0.18	0.16			
	Site 2	Bonfire	0.27	0.21	18	44.07	*** ^a
		Control	1.09	0.50			
	Site 3	Bonfire	0.93	0.30	18	6.52	* ^a
		Control	0.59	0.30			
Extractable Si (mg/Kg)	Site 1	Bonfire	54.30	12.55	18	0.03	n.s. ^a
		Control	52.80	25.45			
	Site 2	Bonfire	34.20	15.40	18	3.22	n.s. ^a
		Control	48.50	19.95			
	Site 3	Bonfire	94.80	37.34	18	0.63	n.s. ^a
		Control	80.50	43011			
Extractable S (mg/Kg)	Site 1	Bonfire	223.10	94.38	18	30.73	*** ^b
		Control	56.00	13.37			
	Site 2	Bonfire	127.50	51.68	18	13.26	** ^b
		Control	197.30	31.66			
	Site 3	Bonfire	59.40	18.69	18	19.54	*** ^a
		Control	28.00	12.47			

Three different sampling sites. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. The degrees of freedom (df) = the total number of subjects in the experiment – the number of groups.

^a One-way ANOVA.

^b Mann-Whitney.

bonfires (Dao et al., 2012).

Soil extractable Mn levels did not vary between bonfire and control soils of Sites 1 and 3; but fell significantly after the bonfire at Site 2. The significant decrease of extractable Mn can be attributed to the increase of pH and SOM decrease. As in the case of extractable Al, extractable Mn decreases with pH increase in the soil (Wang et al., 2016; DeGroot et al., 2018). Previous works observed that extractable Mn normally is correlated with SOM content (Moreira et al., 2016). The temperature of Mn volatilization is very high (> 2467 °C) (Pereira et al., 2019b), therefore, it is very likely that this decrease of extractable Mn is attributed to soil pH fluctuation.

In relation to extractable Fe a significant reduction was observed in

the sites 1 and 2 and this is due to the pH increase, especially in site 1 (Neary et al., 2005). As in the case of Mn, the temperatures of volatilization are very high (> 3000 °C), therefore direct losses of Fe might be limited (Pereira et al., 2019b) as well. Soil extractable Zn, Cu and Cr increased importantly in the site 1, while in the others no differences were observed (with exception of site 2, where it was observed a decrease on extractable Cr). This was attributed to the same reasons described above in the extractable Al, the presence of cans in this bonfire. High concentrations of these elements in bonfires were observed previously by Dao et al. (2012) and Golden et al. (2015) in an urban park located in Galway (Ireland). Zinc, Cu and Cr are considered metals of urban origin (Karim et al., 2015; Qing et al., 2015). The concentrations

Table 4
Descriptive statistics of soil ratios characteristics. Different letters represent significant differences at a $p < 0.05$. $N = 10$.

Soil property	Study site	Plot	Mean	SD	df	F value	p value
C/N ratio	Site 1	Bonfire	32.70	8.17	18	28.35	*** ^a
		Control	12.70	10.15			
	Site 2	Bonfire	8.89	1.30	18	0.31	n.s. ^a
		Control	8.68	0.94			
	Site 3	Bonfire	8.11	3.32	18	3.03	n.s. ^a
		Control	10.10	2.71			
SPAR	Site 1	Bonfire	0.05	0.04	18	58.43	*** ^a
		Control	0.23	0.06			
	Site 2	Bonfire	0.26	0.07	18	0.24	n.s. ^a
		Control	0.24	0.05			
	Site 3	Bonfire	0.21	0.05	18	0.09	n.s. ^a
		Control	0.21	0.04			
Ca:Al	Site 1	Bonfire	717.00	430.65	18	19.29	*** ^b
		Control	115.96	42.50			
	Site 2	Bonfire	273.80	379.12	18	2.82	* ^a
		Control	71.98	21.37			
	Site 3	Bonfire	728.91	191.76	18	16.15	*** ^a
		Control	432.27	133.13			
Ca:Mg	Site 1	Bonfire	84.93	36.52	18	39.79	*** ^b
		Control	12.02	1.62			
	Site 2	Bonfire	18.80	5.36	18	12.68	** ^b
		Control	12.29	2.16			
	Site 3	Bonfire	9.93	1.26	18	1.37	n.s. ^a
		Control	10.48	0.77			

Three different sampling sites. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. The degrees of freedom (df) = the total number of subjects in the experiment – the number of groups.

- ^a One-way ANOVA.
- ^b Mann-Whitney.



Fig. 4. Detail view of the Site 1 bonfire where cans and other hazardous materials were burned.

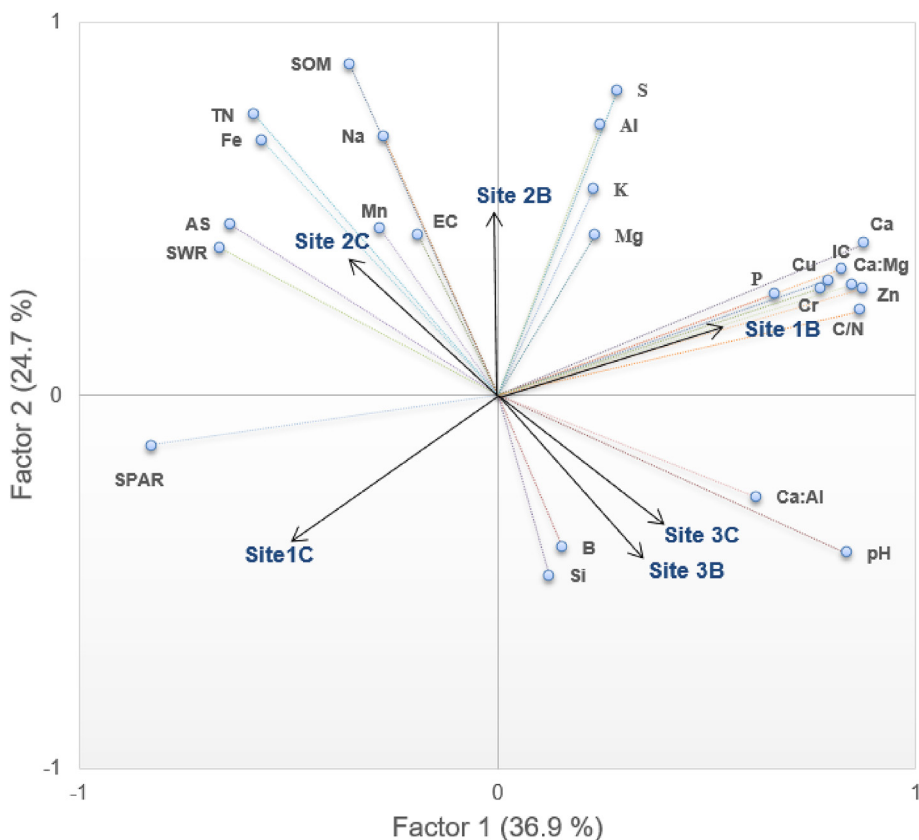


Fig. 3. PCA showing Factor 1 and 2 loadings. Soil Water Repellency (SWR), Aggregate Stability (AS), Soil Organic Matter (SOM) content, Total Nitrogen (TN), Inorganic Carbon (IC), pH, Electrical Conductivity (EC), extractable Calcium (Ca), Magnesium (Mg), Sodium (Na), Potassium (K), Phosphorus (P), Aluminum (Al), Manganese (Mn), Iron (Fe), Zinc (Zn), Copper (Cu), Boron (B), Chromium (Cr), Silicon (Si), Sulfur (S), Carbon/Nitrogen ratio (C/N), SPAR ratio, Calcium:Aluminum ratio (Ca:Al) and Calcium:Magnesium ratio (Ca:Mg). Site 1 Bonfire soils (1B), Site 1 Control unburned soils (1C), Site 2 Bonfire soils (2B), Site 2 Control unburned soils (2C), Site 3 Bonfire soils (3B), Site 3 Control unburned soils (3C). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

of Zn and Cu are extremely high, above the recommended concentrations in soil (400 mg/kg for Zn and 200 mg/kg for Cu) (Toth et al., 2016). Even with the increase of soil pH, the levels of these metals were extremely high, therefore is very likely that their total amount in this bonfire site are much high. Despite the fact that Zn, Cu and Cr are considered essential elements for plants, in high amounts can be considered as toxic (Tiecher et al., 2016; Ertani et al., 2017) and have impacts on human health (Gu et al., 2016). The significant decrease of Cr in site 2 was likely attributed to the increase of pH values. The levels of Cr in solution decrease with pH increase (Imai and Glonya, 1990). As in previous heavy metals, the losses of Cr in solution (site 2) are only due pH fluctuations, since the volatilization temperature of this element is very high (1227 °C) (Pereira et al., 2019b). Soil extractable Cr and S increased significantly in the bonfire soils at Sites 1 and 3 but decreased in those soils at Site 2. Khanna et al. (1994) observed a reduction of extractable S after heating due to at least the 50% of this element is dissolved rapidly and incorporated into the soil, that explained the dynamic of S in bonfire soil from site 2. This dynamic is also explained by the increment of pH in bonfire soils that facilitates the solubility of soil S (Pereira et al., 2019a) as occurred in the present study in site 2. In this sense, the increment of soil extractable S and Cr in bonfire soils from sites 1 and 3 can be produced by two different reasons. Firstly, sites 1 and 3 are cover by *Pinus sylvestris* L. contain a great quantity of oils and resins that after ignition can be partially incorporated into the soil (Adamek et al., 2016). Secondly, *Pinus* is a flammable specie (Nunez-Regueira et al., 2004) and then hold a great capacity to heating the soil (Das et al., 2017). In the case of bonfire soils of site 1, the increment of Cr was also stronger due to the ignition of cans during bonfire and the pyrolyzed traces of these materials found on the topsoil. Pereira et al. (2012) observed high quantities of S in ash slurries and the incorporation of microscopic ashes could increment the soil extractable S in bonfire soils from site 1. The soil response to bonfire can be explained by heating and the incorporation of micro particles of ash, by the different vegetation cover in each site and by the ignition of contaminant materials during the bonfire.

Soil extractable B and Si did not vary between bonfire and control soils at any of the sites irrespective to the vegetation cover. The absence of changes is explained by the scarce impact of fire on these soil properties due to the high volatilization temperatures of the elements (1800 °C and 2230 °C) (Pereira et al., 2019b). Soil extractable B is reported not to vary after exposure to different temperatures and for different durations (Thomaz, 2017) and Pereira et al. (2019a) also found that soil B did not vary after heating. The absence of any changes in extractable B and All in all, few studies have examined the effect of heating on a soil's minor elements and heavy metals, although some research has been carried from the perspective of the contamination of burned materials and problems of soil pollution (see section 4.5 for detailed discussion).

4.4. Soil ratios

The soil C/N ratio increased in the burned soils at Site 1 but showed no changes at Sites 2 and 3. Pereira et al. (2011) observed a decrease in the ratio, its peak at 400 °C being lowered further when TN levels were high. High temperatures increase soil C/N ratio due to the volatilization of N (Neary et al., 2005), as occurred at Site 1. At temperatures below 400 °C, there is no major changes in C/N Blank et al. (1994) did not observe any changes in C/N content at temperatures below 300 °C, as occurred at Sites 2 and 3. The incorporation of microscopic ashes (mainly inorganic) into the soil (explained above) in the bonfire soils from site 1 can explain the increase of C/N ratio. Soil SPAR decreased significantly in the burned soils at Site 1 but remained unaltered at Sites 2 and 3. According to Pereira et al. (2017) low-severity fire does not significantly modify the SPAR ratio and this may explain the lack of differences observed in site 1 and 2. The increase of soil pH after a heating and the high amount of extractable Ca and Mg in comparison to

extractable Na and K can resulted in a decrease of SPAR, as occurred in site 1. The soil Ca:Al ratio increased significantly in the burned soils at all three sites, due to the rise in extractable Ca levels (Pereira et al., 2017), that were much higher than Al, reducing soil acidity. The soil Ca:Mg ratio increased significantly in the burned sites 1 and 2 but did not changed in Site 3 and this is because if the higher increase if extractable Ca than extractable Mg. Pereira et al. (2017) observed a significant fall in this ratio after fire due to the greater impact of fire on soil extractable Mg than on Ca. In this study (especially in site 1) occurred the opposite. To our knowledge, with the exception of Pereira et al. (2017) there is no previous works focused on the effects of fire on soil Ca:Al and Ca:Mg ratios More research is needed to understand the direct implications of soil heating on these soil ratios under different fire conditions.

4.5. Overall discussion of the effects of bonfires on soil and on human health and their implications for land management

Bonfires have been identified as an important source of metal contamination in urban areas (Golden et al., 2015). According to the European Environment Agency (2009), heavy metals are the main contaminant of European soils. In response to this, several remediation techniques (e.g. phytoremediation) have been applied to reduce heavy metals concentration (García-Carmona et al., 2017). The presence of contaminant materials (e.g. metal cans) in bonfires can increase importantly soil heavy metal accumulation, such as Zn (Fig. 4). In this regard, it has been shown that the time a fire is allowed to burn is critical, because even when it is of low intensity and severity, if the episode is of considerable duration the fire can be responsible for changes in soil properties (Francos et al., 2018). For example, Busse et al. (2010) report that soil productivity can change as a result of long-term burning if conditions are acute, such as the presence of high fuel loads. Heavy metals can accumulate in a soil after fire and, as such, represent a risk for plants and human health. Bonfires and their impact on soil fertility and the vegetation capacity to colonize these areas as consequence of the damages induced by heating on soil (García-Marco and Gonzalez-Prieto, 2008). Soil Cu and Zn are essential to plant nutrients in low amounts (Huang, 2005). but are toxic in high quantities Flemming and Trevors (1989). Here, the burned soils of Site 1 were contaminated with Cu and Zn. Bowen (1979) defined the upper critical limits for soil metal concentrations: Mn (1500–3000 mg/kg), Zn (70–400 mg/kg), Cu (60–125 mg/kg), B (5–10 mg/kg) and Cr (75–100 mg/kg).

Bonfires pose a threat to environment and human health (Dao et al., 2012). Although some heavy metals are essential for plants (Wintz et al., 2002), high concentrations they can be a cause of diseases, especially in children and elderly people * = Abraham et al. (2017) Likewise, Balkhair and Ashraf (2016) report that health can be affected by the consumption of vegetables cultivated in the proximity of heavy metal sources. The following effects of exposure to heavy metal have been documented: Mn can affect neuro-behavioral functions and the central nervous system (WHO, 2008); Cr can produce dermatitis, mouth ulcers, nasal septum atrophy and a variety of cancers (Offenbacher, 1994; Wright and Welbourn, 2002), and, finally observed that Zn can affect the renal and nerve systems (Kim et al., 2002).

In an attempt to control soil contamination by heavy metals, the European Union (EU) has ruled that contaminated areas posing a threat to human health and environment should be inventoried. The EU recommended the member countries to take the necessary measures to avoid heavy metal contamination (EUR-Lex, 2004) and recognizes, that human activities contaminate soils, resulting in the loss of fertility and soil degradation. Dao et al. (2012) recommended that local residents must be made aware of the health consequences recurrent from burning car tires and other hazardous metal-containing components. Burning these materials in bonfires should be forbidden and instead of burned, they should be recycled (Dao et al., 2013). All in all and based on the

impact of bonfires on the soil properties and on soil degradation, we can rank the sites as follows in terms of their noxious consequences: Site 1 > Site 2 > Site 3.

5. Conclusions

Soil heating resulted in an increase in pH, EC and Ca:Al and a decrease in SWR at all three sites. The bonfires lit at Sites 1 and 2 also affected SOM, IC, major nutrients, Zn, CU, Fe, C/N ratio and SPAR, none of which were affected by the Site 3 bonfire. The soils at Site 1 presented greater modifications than those at Site 2 in terms of IC, Ca, Mg, Na, AS and SPAR. Overall, our results indicate that the impact of the bonfires was greatest at Sites 1 and 2. The burning of waste materials as tins in the Site 1 bonfire seems to account for these greater modifications to the site's soil properties.

Despite the probable (or compelling evidence of) moderate intensity attained by the bonfires, Site 1 soils suffered the greatest impact because of the waste products thrown onto the fire and the longer ignition time to which it was exposed.

Clearly, protective measures need to be introduced in urban parks to prohibit the ignition of hazardous materials and to limit the period of time during which a bonfire can be allowed to burn. Urban park bonfires have to be subject to certain controls and, where necessary, forbidden if they pose a risk to human health. In the meantime, further studies are needed to analyze how the soil properties of urban parks exposed to recreational bonfires are modified over time and establish the temperatures needed for each change, being these an interesting future point to study.

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