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Soil Resistance to Burn Severity in Different Forest Ecosystems in the Framework of a Wildfire

Sara Huerta *, Víctor Fernández-García, Leonor Calvo and Elena Marcos

Area of Ecology, Department of Biodiversity and Environmental Management, Faculty of Biological and Environmental Sciences, Universidad de León, 24071 León, Spain; vferg@unileon.es (V.F.-G.); leonor.calvo@unileon.es (L.C.); elena.marcos@unileon.es (E.M.)

* Correspondence: shueg@unileon.es

Received: 23 May 2020; Accepted: 17 July 2020; Published: 19 July 2020

Abstract: Recent changes in fire regimes, with more frequent, extensive, and severe fires, are modifying soil characteristics. The aim of this study was to evaluate the effect of burn severity on the resistance of some physical, chemical, and biochemical soil properties in three different forest ecosystems affected by a wildfire in the northwest of the Iberian Peninsula. We evaluated burn severity immediately after fire using the Composite Burn Index (CBI) in three different ecosystems: shrublands, heathlands, and oak forests. In the same field plots used to quantify CBI, we took a composite soil sample to analyse physical (mean weight diameter (MWD)), chemical (pH; total C; total organic C (TOC); total inorganic C (TIC); total N; available P; exchangeable cations Na^+, K^+, Mg^{2+}, and Ca^{2+}; and cation exchange capacity (CEC)), and biochemical (β-glucosidase, urease, and acid phosphatase enzyme activities) properties. The resistance index of each property was then calculated. Based on our results, the values of the soil chemical properties tended to increase immediately after fire. Among them, total C, TOC, and exchangeable Na^+ showed higher resistance to change, with less variation concerning pre-fire status. The resistance of chemical properties was higher in the oak forest ecosystem. MWD decreased at high severity in all ecosystems, but soils in shrublands were more resistant. We found a high decrease in soil enzymatic activity with burn severity, with biochemical properties being the least resistant to change. Therefore, the enzymatic activity of soil could be a potential indicator of severity in forest ecosystems recently affected by wildfires.

Keywords: biochemical properties; burn severity; chemical properties; physical properties; soil resistance; wildfire

1. Introduction

Wildfires are important disturbance factors [1] that modify and shape Mediterranean ecosystems in the Iberian Peninsula [2,3]. In fact, fire is considered as an integral part [4,5] and a dominant ecological factor of these ecosystems [3]. However, climate change [6–8], changes in land use [4], and fire suppression policies [9] act to modify fire regimes, increasing fire frequency, burnt area and burn severity [3,10,11]. These are the main drivers that have modified fire regimes in Spain during the last decades [9], with the consequent ecological, socio-economic, and human impacts [12,13]. However, climatic conditions are highly related to recent changes in the Mediterranean regions [5,14], which are characterized by frequent, large [15], and severe fires [8,16], mainly as a consequence of fuel accumulation [5,9]. In fact, the 2017 wildfire season in the Iberian Peninsula was marked by the burnt area and severity of its fires [17]. In this context, burn severity is described as the loss of or change in organic matter above ground and below ground [18]. It measures the alteration of the burned area and, therefore, the effects of fire on ecosystems [19]. Burn severity is mainly determined by two components, fire intensity and residence time [20]. The former is the energy released from fire [18] and the latter is what produces the most impact on forest soils [21].

Fire effects also depend on the type of ecosystem. Different vegetation and soil characteristics [20,22], as well as temperature and moisture conditions of the environment, influence the amount of organic matter and soil quality [19]. In fact, fuel moisture and organic components affect the rate of energy released during a fire, which, along with other aspects such as topography, slope or altitude, and climatic conditions like temperature, humidity, rainfall, and wind, determine burn severity [21]. According to Doran and Parkin [23], soil quality measures the ability to function within the ecosystem, and it is related to its physical, chemical, and biological properties [24]. In this context, soil organic matter, or the organic carbon of soil, is considered as one of the most common indicators of soil quality, as it has a key role in most soil functions [25].

The effects of burn severity on soil's physical, chemical, and biochemical properties have been widely studied [26–31]. One of the main effects of high burn severity is the loss of soil structure due to the destabilization of aggregates [32]. Soil structure controls hydrological processes and erosion risk [33]. High severity fires tend to degrade soil organic matter by combustion, therefore, a destruction of the aggregates may occur in soils where aggregation depends mainly on organic matter [1,34]. This often leads to a reduction in the volume of macropores [35], which affects infiltration capacity, increasing surface runoff and soil erosion [36,37]. Wildfires also affect soil chemical properties [19,38]. For instance, soil pH tends to increase after fire due to the release of basic cations by combustion and the addition of ash [20,39–41]. The destruction of clay particles and organic matter (humus) by heat can affect soil cation exchange capacity, as it depends on the negative charges of these organic and inorganic soil colloids [38]. In terms of soil nutrients, the combustion of vegetation and soil organic forms modifies the nutrient biogeochemical cycles [38] and increases the availability of soil nutrients for vegetation [38,42]. All these effects mainly occur in the first few centimetres of the surface horizon [2,26,43]. Nutrient cycles are influenced by soil enzymes [44,45], with β- glucosidase, urease, and acid phosphatase being among the main enzymatic activities related to the C, N, and P cycles, respectively [2,46]. Enzymes in soils, which catalyse biological reactions [44], are of animal, plant, and microbial sources [45,47]. They have potential as indicators of soil microbial activity [48] and its quality [49] and have a decisive role in the maintenance of soil fertility [2].

Thus, wildfires largely affect soil's physical, chemical, and biochemical properties, which are closely related to soil quality [50–52]. Many indicators describe the capacity of soil to function [53], while changes in soil quality due to disturbances can be evaluated in terms of resistance and resilience [54,55]. Resistance is known as the capacity of the system to withstand a disturbance, while resilience can be defined as the ability to absorb this change [56] and recover after perturbations [57,58]. Both allow the stability of a system to be measured [57], i.e., its response to disturbance [59], which determines the capacity to continue functioning under changing conditions [60]. Resistance is considered as a function of soil properties, so these properties can be used to assess whether soil function changes after disturbance [53]. Resistance and resilience can be measured by indices that enable the control and disturbed samples to be controlled at the same time [60–62]. This approach takes into account changes in the control samples over time [63].

Therefore, the main objective of this study was to evaluate the effect of burn severity on soil resistance in different forest ecosystems. Specifically, we aimed to; (1) assess fire resistance of soil physical (mean weight diameter (MWD)), chemical (pH; total C; total organic C (TOC); total inorganic C (TIC); total N; available P; exchangeable cations Na^+, K^+, Mg^{2+}, and Ca^{2+}; cation exchange capacity (CEC)) as well as biochemical (β-glucosidase, urease, and acid phosphatase enzyme activities) properties to fire severity and, (2) compare the resistance of these soil properties to fire among three forestry ecosystems: *Genista hystrix* Lange shrublands, *Erica australis* L. heathlands and *Quercus pyrenaica* (Willd.) oak forests. Previous research in Mediterranean ecosystems found changes with increasing soil burn severity in soil organic carbon, pH, and enzymatic activities [30,31,64]. The importance of these changes was influenced by the property being measured and pre-fire soil status [64,65]. Therefore, we expect biochemical properties, explicitly soil enzymes, to be less resistant in all ecosystems, owing to the sensitivity of soil microbial biomass to burn severity [31] and the potential for enzyme denaturation [64,66]. However, changes in other properties such as soil carbon, pH, and nutrients could be conditioned by their initial content.

2. Materials and Methods

2.1. Study Area

The study was conducted in the Cabrera mountain range, León province (NW Iberian Peninsula) (Figure 1). On 21 August 2017, a wildfire occurred in this area, burning a total of 9.939 ha of forestry ecosystems dominated by *G. hystrix*, *E. australis*, and *Q. pyrenaica*. The elevation of the study area ranges from 836 to 1938 m.a.s.l. and the orography is mountainous. The area has a temperate climate, with dry temperate summers [67]. The average annual temperature in this zone is 9 °C, with an average annual precipitation of 758 mm [68].

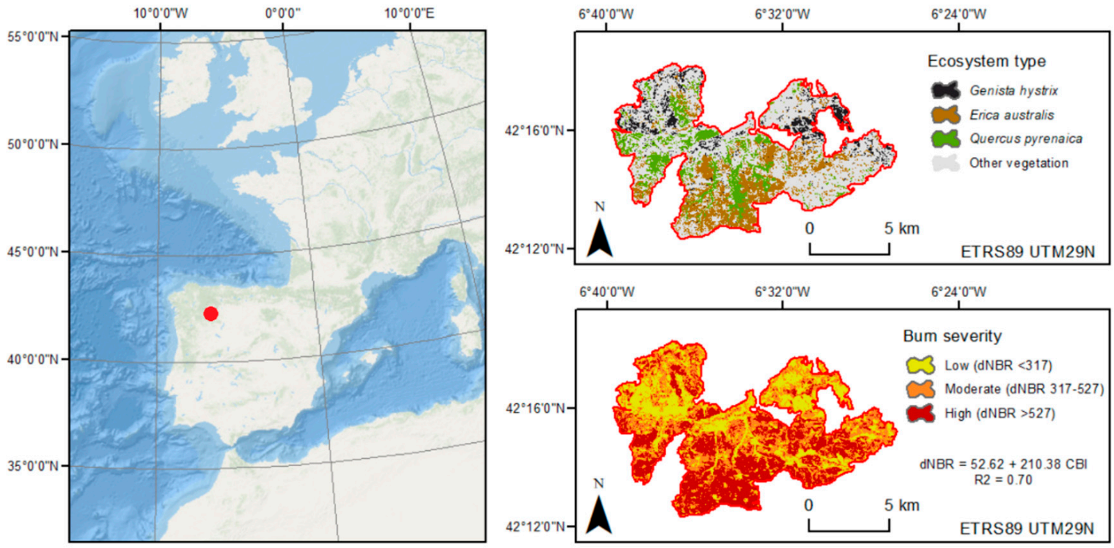

Figure 1. Location of the study area in the Iberian Peninsula (left). Panels on the right show the type of ecosystems in the fire perimeter (top) and burn severity levels (bottom) [69].

This area has acidic soils, developed mainly on slate, sandstone, and quartzite from the Ordovician period [70]. Soils, which have sandy loam and sandy clay loam texture, are mainly Lithic leptosols and Humic cambisols [71].

2.2. Field Sampling

During the months of September and October after the wildfire, 129 field plots (1 m × 1 m) were randomly established and sampled within the fire perimeter. The plots were fixed in the most representative ecosystems in the area and, within these, in all the severity categories previously defined by the dNBR index, that was validated and calibrated in the studied wildfire by Fernández- García et al. [69]. Therefore, soil burn severity was evaluated in each field plot using the Composite Burn Index procedure (CBI) adapted by Fernández-García et al. [28]. Two factors were visually evaluated from a score of 0 (unburned) to 3 (maximal burn severity): (1) percentage of litter and light fuel consumed, and (2) char depth and ash and mineral soil colour (Table 1).

Table 1. Rating factors and scale used to quantify soil burn severity (substrate stratum of the Composite Burn Index according to Fernández-García et al. [28]).

Rating Factors	Burn Severity Scale						
	Unburned	**Low**		**Moderate**		**High**	
	0	**0.5**	**1**	**1.5**	**2**	**2.5**	**3**
	Substrate						
Litter/light fuel consumed	None	<10%	10–20%	20–40%	40–80%	80–98%	98%
Char and colour	None	Blackened litter, no changes in soil		Charred remains, recognisable litter		Grey and white ash, grey soil	White ash, reddened soil

G. hystrix shrublands plots (n = 33): 16 at low severity, 15 at moderate severity and 2 at high severity. *E. australis* heathlands plots (n = 33): 11 at low severity, 9 at moderate severity and 13 at high severity. *Q. pyrenaica* oak forests plots (n = 63): 30 at low severity, 21 at moderate severity and 12 at high severity.

In order to analyse the effects of burn severity on soil properties immediately after fire, one composite soil sample (each one composed of four subsamples) was collected from all 129 burned plots (1 m × 1 m), as well as 32 control plots within each forestry ecosystem (12 in shrublands, 12 in heathlands, and 8 in oak forests). Samples were taken using an auger (7 cm diameter × 3 cm depth) after removal of post-fire residues (ash and scorched debris) or unburned litter. One part of each sample was air dried, sieved at <2 mm and stored in the laboratory (20 °C) for the analysis of physical and chemical properties. Another part was frozen (−18 °C) for the subsequent analysis of enzymatic activities.

2.3. Soil Analysis

Soil physical (mean weight diameter (MWD)), chemical (pH; total C; total organic C (TOC); total inorganic C (TIC); total N; available P; exchangeable cations Na^+, K^+, Mg^{2+}, and Ca^{2+}; cation exchange capacity (CEC)), and biochemical (β-glucosidase, urease, and acid phosphatase enzyme activities) properties were analysed in each sample.

MWD was used to measure the distribution of aggregate sizes and the average size of stable aggregates. Soil samples were dry-sieved through 1, 0.25, 0.1, and 0.05 mm sieves for 120 s in an electromechanical shaker [72]. MWD was calculated according to Equation (1):

$$\mathrm{MWD} = \sum\nolimits_{i=1}^{n} \mathrm{X}_i\, \mathrm{W}_i \tag{1}$$

where X is the average particle size (mm) and W the weight of each soil fraction (%).

We analysed soil pH by the potentiometric method, using a suspension of soil:deionised water (1:2.5 *w*/*v*) at 25 °C. Total C, TOC, and total N were determined following the combustion method of Dumas [73], using a EuroVector EA3000 elemental analyser (Eurovector SpA, Radovalle, Italy), whereas TIC was calculated arithmetically by the difference between total C and TOC. Available P was measured at 882 nm wavelength according to the Olsen et al. [74] procedure, using a UV Mini 1240 spectrophotometer (Shimadzu Corporation, Kyoto, Japan). We used AcONH4 1N pH 7 to extract soil exchangeable cations (Na^+, K^+, Mg^{2+}, and Ca^{2+}) and barium chloride 0.1 M to extract CEC. After that, both properties were determined through inductively coupled plasma-optical emission spectroscopy (ICP-OES).

β-glucosidase (β-D-glucoside glucohydrolase, EC 3.2.1.21) and acid phosphatase (phosphate-monoester phosphohydrolase, EC 3.1.3.2) activities were analysed following the method described by Tabatabai [75], whereas the procedure of Kandeler and Gerber [76] was used to measure the urease (urea amidohydrolase, EC 3.5.1.5) activity. We incubated the soils with each enzyme substrate: *p*-nitrophenyl-β-D-glucopyranoside for the β-glucosidase, *p*-nitrophenyl phosphate for the acid phosphatase, and urea in the case of urease activity. The absorbance of the *p*-nitrophenol (*p*-NP) released by β-glucosidase and acid phosphatase activities was measured at 400 nm wavelength, and the absorbance of the $NH4^+$ produced by urease activity was determined at 690 nm wavelength. We used a UV-1700 PharmaSpec spectrophotometer (Shimadzu Corporation, Kyoto, Japan) to measure these products colorimetrically.

2.4. Resistance Calculations

To quantify soil property resistance to burn severity just after fire, we applied the resistance index developed by Banning and Murphy [62], following Equation (2):

$$\text{Resistance}: \ R_0 = -100\left[\frac{C_0 - P_0}{C_0}\right] \text{at } t_0 \tag{2}$$

where C_0 is the value of the soil property in the control and P_0 the value in the burned soil immediately after the wildfire (t_0). An R_0 value equal to zero indicates maximal resistance and, therefore, no differences between the control and the burned soil. Minimal resistance is obtained when the R_0 index is −100, which means that the value of the burned soil property is zero. The R_0 index does not have a positive limit. Values above zero are obtained when the property in the control is lower than in the burned soil. Thus, the resistance index (R_0) represents the % of difference between the property in the burned soil and in the control immediately after disturbance.

We calculated the resistance index for all soil properties in each ecosystem separately, considering the three burn severity levels (low, moderate, and high). For the control values (C_0) we calculated an average for each property and ecosystem.

2.5. Data Analysis

We applied a two-way analysis of variance (ANOVA) to compare forest ecosystems and burn severity levels for each soil property. The predictor variables (fixed factors) were the types of ecosystems (shrubland, heathland, and oak forest) and burn severity categories (low, moderate, and high) calculated using the CBI substrates scoring. The response variables in the models were the resistance indices (R_0) calculated for the following soil characteristics: (1) MWD, (2) pH, (3) total C, (4) TOC, (5) TIC, (6) total N, (7) available P, (8) exchangeable Na^+, (9) exchangeable K^+, (10) exchangeable Mg^{2+}, (11) exchangeable Ca^{2+}, (12) CEC, (13) β-glucosidase, (14) acid phosphatase, and (15) urease activities. We considered significant differences at a p value < 0.05. All data analyses were developed with SPSS Statistics 26.0 software.

3. Results

3.1. General Values

Soil characteristics for each forest ecosystem are presented in Table 2. Oak forest soils present higher available nutrient content than shrubland and heathland soils, as well as a greater cation exchange capacity, carbon content, and enzymatic activity. Between the non-arboreal ecosystems, heathland soils have more total carbon, nitrogen, and phosphatase activity than shrubland soils, which are poorly developed in this region.

Table 2. Average values of each soil property in the control plots for each forest ecosystem type: shrubland (n = 12), heathland (n = 12), and oak forest (n = 8).

Soil Property	Shrubland	Heathland	Oak Forest
Physical			
MWD (mm)	1.39 (0.38)	1.66 (0.48)	1.50 (0.45)
Chemical			
pH	4.88 (0.17)	4.79 (0.25)	5.45 (0.29)
Total C (%)	2.90 (1.40)	6.79 (2.62)	8.56 (1.81)

Table 2. *Cont.*

Soil Property	Shrubland	Heathland	Oak Forest
TOC (%)	2.70 (1.35)	6.38 (2.61)	8.12 (1.67)
TIC (%)	0.20 (0.29)	0.42 (0.53)	0.44 (0.35)
Total N (%)	0.17 (0.06)	0.35 (0.14)	0.56 (0.11)
Available P (mg kg^{-1})	4.54 (2.53)	4.97 (2.17)	11.15 (4.95)
Na^+ (cmol kg^{-1})	0.04 (0.04)	0.05 (0.03)	0.04 (0.02)
K^+ (cmol kg^{-1})	0.16 (0.04)	0.29 (0.10)	0.54 (0.16)
Mg^{2+} (cmol kg^{-1})	0.23 (0.09)	0.47 (0.32)	2.60 (0.59)
Ca^{2+} (cmol kg^{-1})	0.94 (0.58)	1.03 (0.79)	8.11 (2.26)
CEC (cmol kg^{-1})	1.63 (0.63)	2.77 (1.39)	12.30 (2.88)
Biochemical			
β-glucosidase (μmol *p*-NP g^{-1} dw soil h^{-1})	2.11 (0.75)	2.56 (1.10)	4.35 (1.15)
Urease (μmol N-NH_4^+ g^{-1} dw soil h^{-1})	4.83 (2.08)	3.78 (1.71)	12.20 (6.93)
Acid phosphatase (μmol *p*-NP g^{-1} dw soil h^{-1})	6.16 (2.26)	12.59 (5.56)	18.98 (4.64)

The values in brackets correspond to the standard deviation. MWD (mean weight diameter); TOC (total organic C); TIC (total inorganic C); CEC (cation exchange capacity).

3.2. Physical Properties

Mean weight diameter (MWD) was relatively resistant to low and moderate burn severity in shrublands and heathlands, but not in oak forests, where MWD actually increased (Table 3). At high burn severity, all three ecosystems experienced major declines in MWD compared to control soils (Table 3; Figure 2; Table S1).

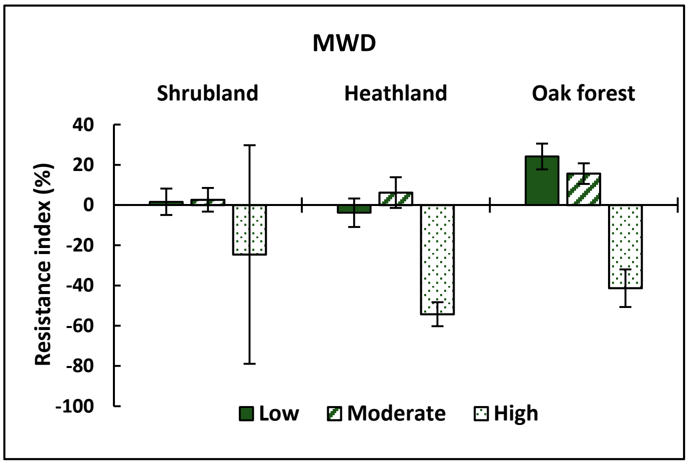

Figure 2. Average of mean weight diameter (MWD) resistance index values and standard error for each forestry ecosystem (shrubland, heathland, and oak forest) and burn severity level (low, moderate, and high) measured by the Composite Burn Index (CBI).

Table 3. Results of the two-way ANOVA showing the effects of the factor ecosystem (shrubland, heathland, and oak forest), the effects of the burn severity variable (low, moderate, and high), and the interaction between ecosystem and burn severity (*), on the resistance index of each soil property (response variable). Significant *p* values are in bold face. NS (not significant).

Soil Property	Treatment	*F* Value	*p* Value
Physical			
MWD	Ecosystem	3.514	**0.033**
	CBI	16.694	**0.000**
	Ecosystem*CBI	1.012	NS

Table 3. *Cont.*

Soil Property	Treatment	*F* Value	*p* Value
Chemical			
pH	Ecosystem	3.379	**0.037**
	CBI	27.607	**0.000**
	Ecosystem*CBI	4.558	**0.002**
Total C	Ecosystem	1.739	NS
	CBI	8.643	**0.000**
	Ecosystem*CBI	4.611	**0.002**
TOC	Ecosystem	2.131	NS
	CBI	7.932	**0.001**
	Ecosystem*CBI	5.085	**0.001**
TIC	Ecosystem	4.196	**0.017**
	CBI	2.972	NS
	Ecosystem*CBI	6.975	**0.000**
Total N	Ecosystem	14.505	**0.000**
	CBI	4.594	**0.012**
	Ecosystem*CBI	5.127	**0.001**
Available P	Ecosystem	0.119	NS
	CBI	3.614	**0.030**
	Ecosystem*CBI	3.626	**0.008**
Na^+	Ecosystem	3.917	**0.023**
	CBI	1.675	NS
	Ecosystem*CBI	0.161	NS
K^+	Ecosystem	16.634	**0.000**
	CBI	4.269	**0.016**
	Ecosystem*CBI	4.738	**0.001**
Mg^{2+}	Ecosystem	21.649	**0.000**
	CBI	1.750	NS
	Ecosystem*CBI	2.019	NS
Ca^{2+}	Ecosystem	10.106	**0.000**
	CBI	3.057	NS
	Ecosystem*CBI	2.838	**0.027**
CEC	Ecosystem	11.636	**0.000**
	CBI	2.807	NS
	Ecosystem*CBI	2.085	NS
Biochemical			
β-glucosidase	Ecosystem	6.018	**0.003**
	CBI	30.664	**0.000**
	Ecosystem*CBI	2.398	NS
Urease	Ecosystem	3.987	**0.021**
	CBI	6.077	**0.003**
	Ecosystem*CBI	0.666	NS
Acid phosphatase	Ecosystem	1.427	NS
	CBI	15.533	**0.000**
	Ecosystem*CBI	5.202	**0.001**

MWD (mean weight diameter); TOC (total organic C); TIC (total inorganic C); CEC (cation exchange capacity).

3.3. Chemical Properties

We observed that burn severity had significant effects over pH, total C, TOC, total N, and available P ($p < 0.05$), because their resistance declined at high severity levels (Table 3; Figure 3). However, the resistance pattern was different depending on the soil property considered. Available P and pH appeared to be less resistant (greater percentage of change after fire) to high severity than total C and TOC, whereby a minor depletion was detected (Table S1).

The soil resistance pattern of pH ($p < 0.05$), TIC ($p < 0.05$) and total N ($p < 0.001$) was also influenced by the type of forest ecosystem (Table 3; Figure 3). In this sense, the heathland ecosystem showed a different and significant behaviour at moderate severity for pH ($p < 0.001$) and total N ($p < 0.001$). TIC presented a different response for each type of forestry ecosystem ($p < 0.001$). We were able to obtain an interaction ($p < 0.05$) between type of ecosystem and burn severity (Table 3; Figure 3) in all soil properties.

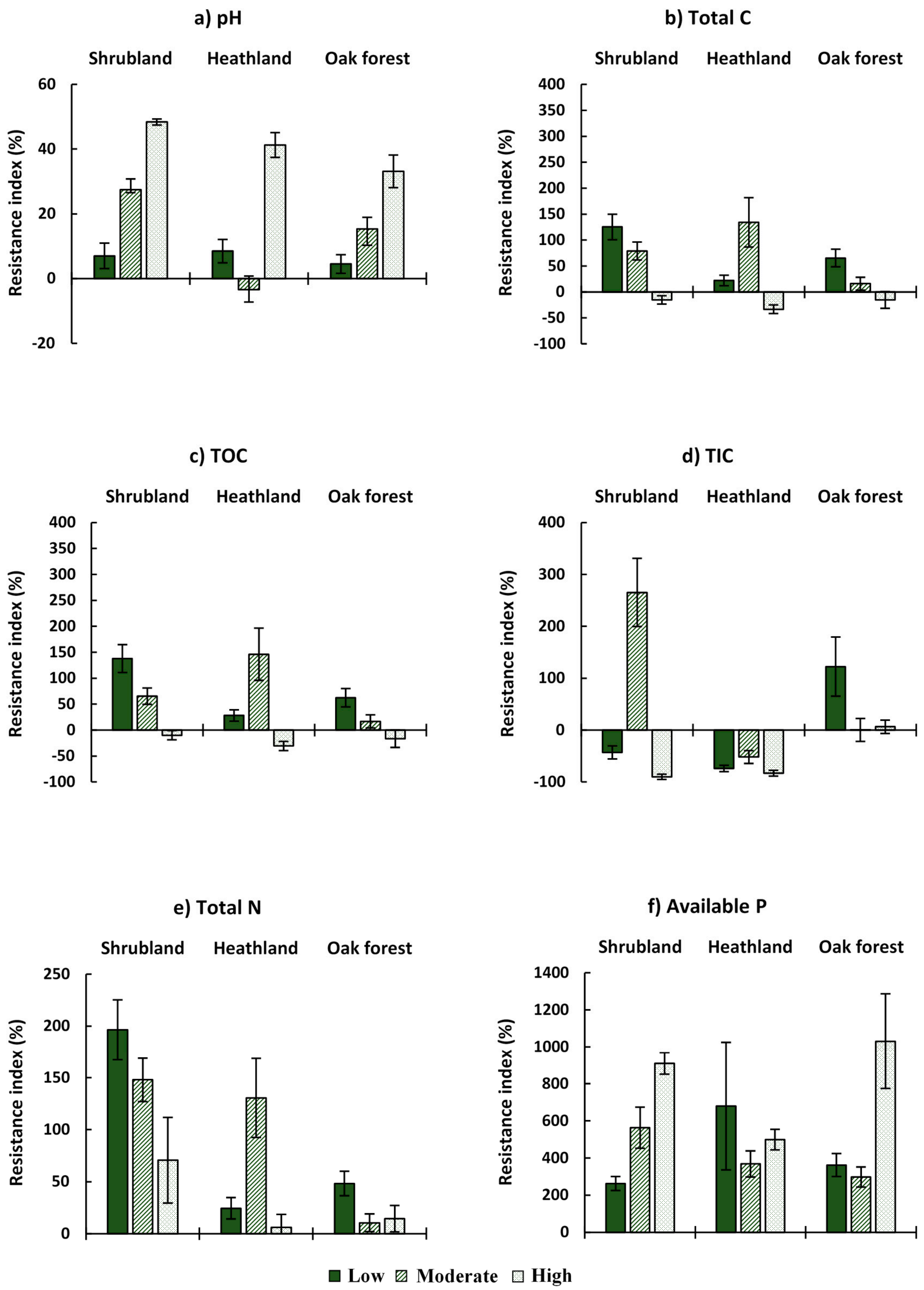

Figure 3. Average of (**a**) pH, (**b**) total C, (**c**) total organic carbon (TOC), (**d**) total inorganic carbon (TIC), (**e**) total N, and (**f**) available P resistance index values and standard error for each forestry ecosystem (shrubland, heathland, and oak forest) and burn severity level (low, moderate, and high) measured by the Composite Burn Index (CBI).

Exchangeable cations and CEC resistance patterns were influenced by forestry ecosystem type ($p < 0.05$). Although exchangeable cations increased immediately after the wildfire (Table S1), the response of each type of ecosystem was different at moderate severity level ($p < 0.05$). We only observed a significant effect of burn severity ($p < 0.05$) on K^+ resistance (Table 3; Figure 4), which was higher at low severity in heathlands and moderate severity in oak forests. We also detected significant interactions between ecosystem and severity for K^+ and Ca^{2+} ($p < 0.05$) resistance indices (Table 3).

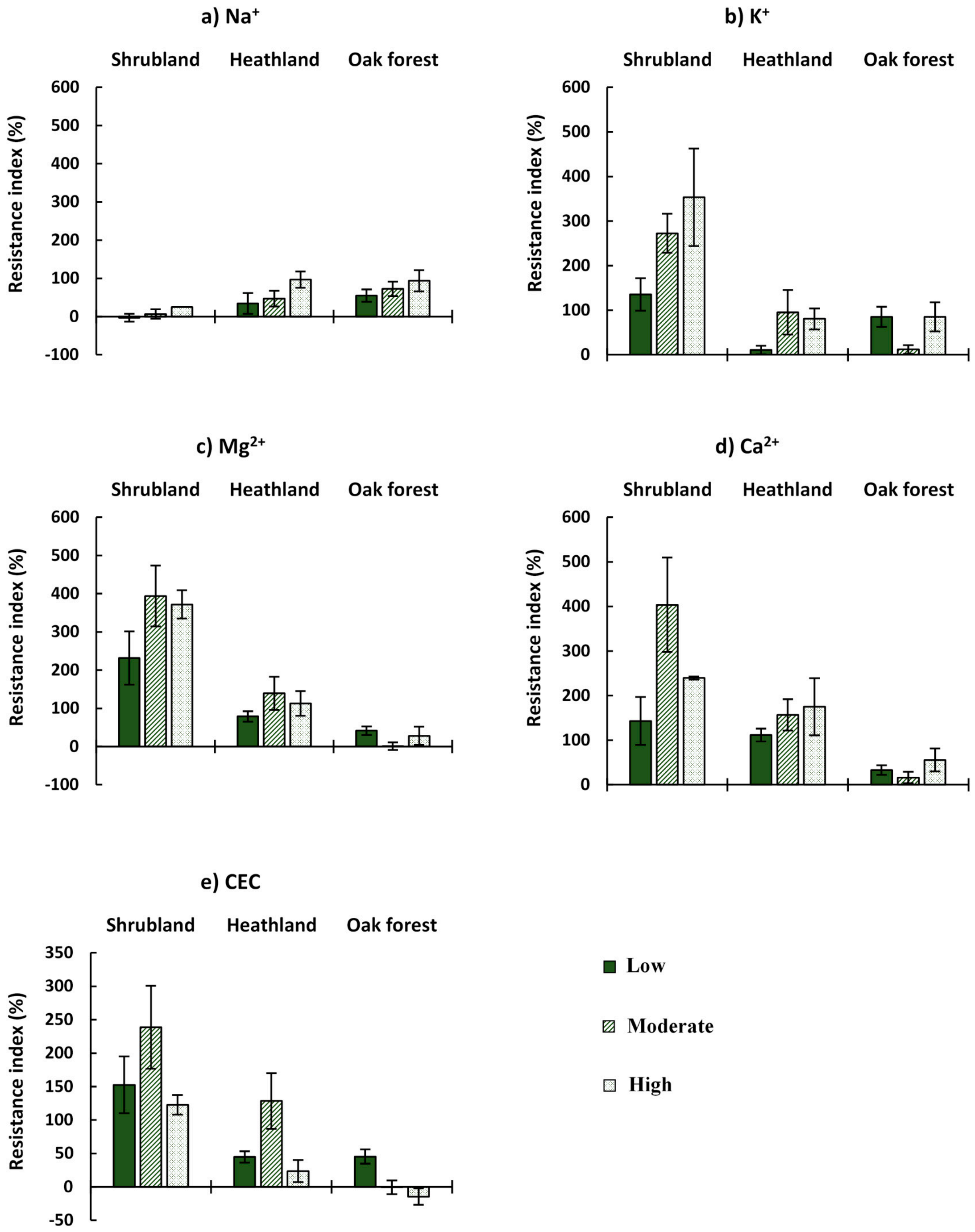

Figure 4. Average of (**a**) Na^+, (**b**) K^+, (**c**) Mg^{2+}, (**d**) Ca^{2+}, and (**e**) cation exchange capacity (CEC) resistance index values and standard error for each forestry ecosystem (shrubland, heathland, and oak forest) and burn severity level (low, moderate, and high) measured by the Composite Burn Index (CBI).

3.4. Biochemical Properties

All enzymatic activities decreased (p < 0.05) with burn severity (Table 3; Table S1), but β-glucosidase and acid phosphatase were the least resistant, completely disappearing at high severity (Figure 5). Only β-glucosidase and urease showed significant differences (p < 0.05) among forestry ecosystems (Table 3). We found that both enzymatic activities were highly resistant to change in low severity shrublands (Figure 5).

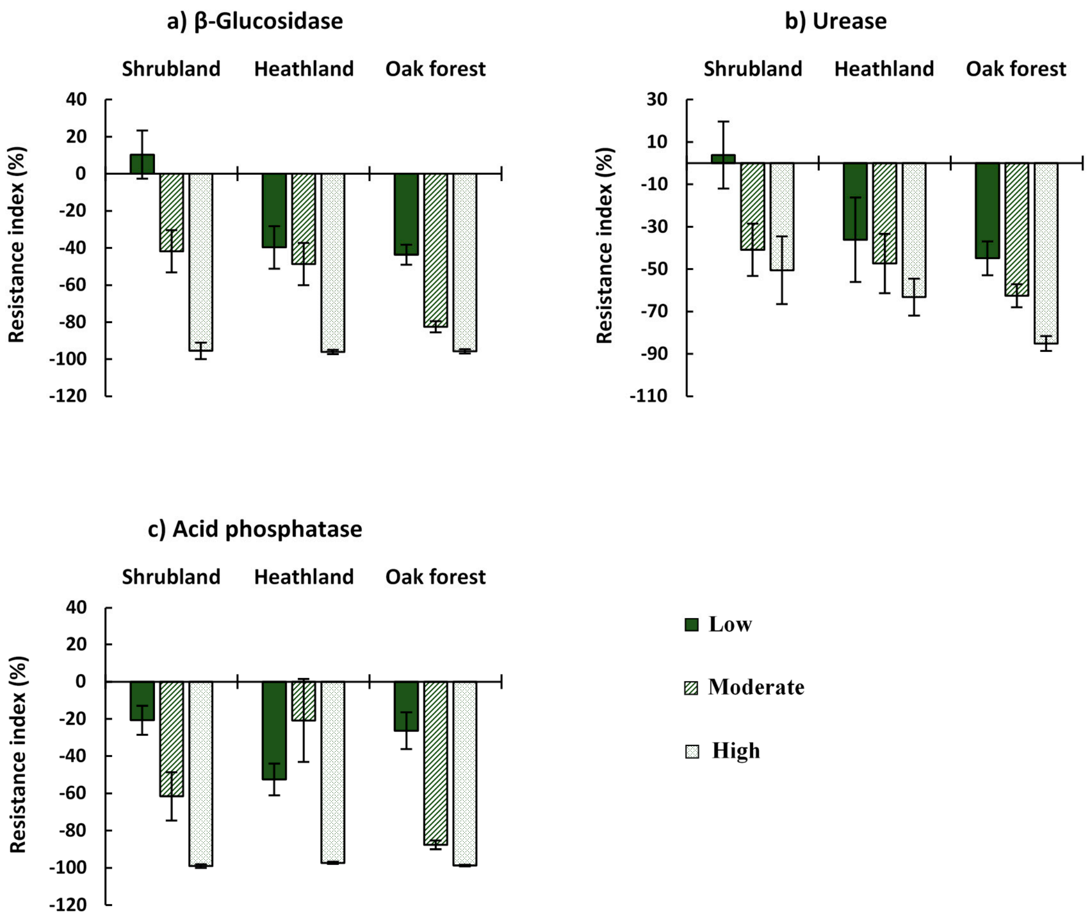

Figure 5. Average (**a**) β-glucosidase, (**b**) urease, and (**c**) acid phosphatase resistance index values and standard error for each forestry ecosystem (shrubland, heathland, and oak forest) and burn severity level (low, moderate, and high) measured by the Composite Burn Index (CBI).

A significant ecosystem and severity interaction (p < 0.05) was identified for acid phosphatase (Table 3) as at moderate severity its resistance was higher than the others.

4. Discussion

In the present study, we explored the resistance of soil properties to wildfire severity shortly after a major burn event covering three fire-prone ecosystems of the Iberian Peninsula in Southwestern Europe. Based on our results, the resistance of soil properties to wildfire varied with the ecosystem studied and with the severity level when sampled 1–2 months post-burn.

In the case of MWD, we found decreases at high severity in all ecosystems. Some authors have documented the decrease of the aggregates size, because of the increase in temperature both under controlled laboratory conditions [37,77], and after a wildfire [30,34,78]. The reduction of MWD is related to the loss of organic matter by combustion [37,78,79] since it acts as a binding agent of soil particles [1]. Therefore, organic matter has an especially important function in the stabilization of soil structure [80]. This could explain the larger MWD reduction found in high severity heathlands and oak forests and the higher resistance of the shrubland ecosystems. Soil organic matter content in heathlands and oak forests soils in general is higher than in shrublands, whose soils are poorer, so the effect of fire is greater. Despite the reduction in MWD at high severities, Varela et al. [77] observed that the aggregate size hardly changed at low severity, whereas Fernández-García et al. [30] documented an MWD increase at low and moderate severities. When temperatures are not high enough to destroy organic matter [79], hydrophobic substances may contribute to the formation of aggregates in soils where particles are bound mainly by the organic matter [1,37]. In the present study, we found that MWD tends to increase at low and moderate severity, with shrublands being the most resistant ecosystem to change.

In general, the resistance of soil chemical properties is more influenced by the type of ecosystem than by the burn severity level. However, soil resistance changes for total C and TOC only depend on the burn severity level, with a negative resistance index at high severity for all ecosystems. Total C of soil includes the fraction present in soil biomass, the organic C, and the carbon that is part of the inorganic compounds. Total C and TOC had the same behaviour pattern, since they constitute most of the carbon pools of soil [33]. Fire affects the chemical composition of organic matter, as well as the rates of decomposition, and these changes are conditioned by severity [38]. In fact, low and moderate severity fires may increase soil C due to the incorporation of unburned or partially burned materials that remain after incomplete combustion, when temperatures do not allow organic matter to oxidize [81]. The deposit of dry leaves and burnt plant materials in fires increases the organic matter content of soil [82,83], and compensates for losses by combustion [84]. This can be observed in heathlands affected by moderate severity. Nevertheless, severe fires tend to reduce the thickness of the organic soil horizon [85] and, therefore, the organic matter content [20]. Lombao et al. [86] also found that fire negatively affected total C of soil, and the reduction in organic C content with increased burn severity was also observed in other studies [26,30,64]. Like total C and TOC, available P resistance was only conditioned by burn severity level. Available P increased after fire, mostly at high severity in shrublands and oak forests. Many authors have documented that the available P enhances immediately after fire [22,43,87,88] and under high severity conditions [30,64]. This is because of the deposit of ash on the soil surface [38,89]; and the mineralization of organic P to orthophosphate, the form available to plants [43,90]. The interaction between the forest ecosystem and burn severity was significant for all these parameters due to the behaviour of heathlands at moderate severity, which shows lower temperatures in this ecosystem than in the others. Thereby, the heathland ecosystem showed less resistance for total C and TOC at moderate severity, while greater resistance was observed in *Quercus* ecosystems.

In the case of TIC, exchangeable Na^+, Ca^{2+}, Mg^{2+}, and CEC, resistance to change is determined by the type of ecosystem. TIC occupies a minimum part of the total C of soil. In this study, TIC resistance decreased in shrublands and heathlands, with higher resistance in the oak forest ecosystem. Inorganic C can be associated with the total combustion of litter and organic matter, which takes place at high severities, producing ash rich in carbonates [27]. Furthermore, exchangeable soil cations resist high temperatures and are not easily volatilized [38]. The mineralization of soil and vegetation organic matter mobilizes these cations to a soluble fraction [91], so ash deposited during fire contains high concentrations of these elements [19]. The increase in Ca^{2+} and Mg^{2+} in shrubland and heathland ecosystems is related to greater combustion of organic matter [92] than in oak forest. The same pattern was observed in the CEC, with a reduction at high severity, also observed by Franklin et al. [93]. This may be due to a greater alteration of colloidal organic matter whose negative charges favour the uptake of positive ions [38]. In the case of Na^+, higher resistance was found in shrubland ecosystems and its resistance tended to decrease as the severity rose [64].

The resistance of pH, total N, and exchangeable K^+ depended on both severity level and type of ecosystem. We observed that pH rose with burn severity, in agreement with other studies conducted in shrublands [64,94], oak forest [26,66,86], and pine forests [30], with the highest resistance at low severity level. However, we found a slight reduction in pH (depletion) in moderate severity heathlands, that was also observed by Marcos et al. [95] in heathlands and Pereira et al. [96] in grasslands, which could be attributed to the nitrification process after fire [97]. In the case of K^+, results showed a greater influence of the type of ecosystems and it followed the same pattern as Ca^{2+} and Mg^{2+}. Total N decreased at high severity level due to volatilization, which depends on destroyed organic matter [38] and temperatures reached [98]. However, Alcañiz et al. [22] observed that total N increase after prescribed fires, which they associated to low temperatures in this type of forest fire.

Soil biochemical properties showed no resistance after wildfire [99], decreasing with burn severity in all the ecosystems. Many authors found a decrease in enzymatic activity immediately after fire [2,88,100–102], as well as a decrease in the activity of these extracellular enzymes with fire burn severity [30,31,46,64] due to the thermal denaturation of soil enzymes [2,66], which occurs above 60–70 °C [44]. The increase in soil pH has also been related to the decrease in β-glucosidase [103] and acid phosphatase [104] activities. Besides, vegetation losses are also associated with low values of enzymatic activities in burned soils [44], so post-fire vegetation recovery will influence soil enzymes [45]. Knowledge of potential soil enzymatic activity reflects its capacity for biochemical processes, which are necessary for maintaining fertility [2]. β-glucosidase and urease activities showed high resistance in shrubland when burn severity was low, which could be due to the lower temperatures reached in this ecosystem.

Our results indicated that the resistance of soil properties is not only conditioned by burn severity level but also by the type of ecosystem. In most cases, pre-fire soil characteristics determined the response to burn severity, and ecosystems with poorer soils showed less resistance to change.

In addition to evaluating the behaviour of soil properties immediately after fire, it would be of special interest to develop medium-long term studies that provide information on the ecosystem resilience. Therefore, the study of the resistance of soil properties to change, as well as of their recovery after perturbation, would offer a broader vision of the ecosystems response and the magnitude of that change. In this context, it is known that burn severity also affects soil properties in the medium- [31] and long-term [27] after fire. The response of burned areas to fire is mainly conditioned by post-fire weather conditions and vegetation recovery [105], and it is highly related to fire burn severity [106]. In fact, precipitation is considered a key factor that controls many post-fire processes [107]. According to Inbar et al. [108], changes in soil properties caused by fire can modify its structure and cause soil losses under rainfall conditions. Besides, these changes in soil properties, along with the vegetation removal, may affect runoff and erosion processes [1,78,105,109], with important effects on the hydrological cycle [110].

Mediterranean ecosystems are highly fire-prone [111], and they are able to cope with fire [3]. However, taking into account the fire regime forecasts in this region [16], this approach could be useful for the development of a landscape management focused on the reduction of negative impacts of fire [112] and the resilience of the ecosystems in these areas [113].

5. Conclusions

This study furthers our knowledge on the resistance of soil properties to burn severity across three ecosystems common to the region. Soil biochemical properties are the least resistant to burn severity (mainly acid phosphatase), depleting even at low severity. Within soil chemical properties, we identified total C, TOC, and Na^+ as the properties with a higher resistance index. MDW is more resistant to low and moderate severities, mainly in shrublands and heathlands, but not to high severity levels.

In general, and for the same severity level, oak forest ecosystems were the most resistant to the change in soil chemical properties, but not for physical and biochemical properties, which proved to be more resistant in shrubland ecosystems.

Therefore, the short-term identification of the less resistant ecosystems to burn severity could be a useful tool for pre- and post-fire management, especially in areas where fires are becoming more frequent and severe.

Supplementary Materials: The following are available online at http://www.mdpi.com/1999-4907/11/7/773/s1, Table S1: Average values of each soil property in the burned plots for each forest ecosystem type (shrubland, heathland, and oak forest) and severity level (low, moderate, and high).

Author Contributions: S.H., L.C. and E.M. conceived and designed the experiment; V.F.-G. and L.C. obtained the soil samples in the field; S.H. and V.F.-G. analysed the soil properties in the laboratory; S.H. analysed the data and wrote the manuscript; V.F.-G., L.C., and E.M. revised the manuscript; L.C. and E.M. coordinated the study. All authors have read and agreed to the published version of the manuscript.

Funding: This study was financially supported by the Spanish Ministry of Economy and Competitiveness in the framework of the FIRESEVES (AGL2017-86075-C2-1-R) project; and by the Regional Government of Castilla and León in the framework of the SEFIRECYL (LE001P17) project. The European Regional Development Fund also provided funding for the present study. S.H. was supported by a predoctoral fellowship from the Regional Government of Castilla and León and the European Social Fund (EDU/574/2018).

Conflicts of Interest: The authors declare no conflict of interest.

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