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Vegetation dynamics and soil properties following low-intensity wildfire in loblolly pine (*Pinus taeda* L.) planted forest in Northern Iran

Mahdiyeh Nezhadgholam-Zardroodi[1], *Hassan Pourbabaei*[1], *Mehrdad Ghodskhah-Daryaei*[1], *Ali Salehi*[1], *Shahram Enayati-Charvadeh*[2], *Jamshid Eslamdoust*[3]*

[1]*Department of Forestry, Natural Resources Faculty, University of Guilan, Sowmehsara, Guilan, Iran*
[2]*Department of Natural Disaster Management, Faculty of Environment, University of Tehran, Tehran, Iran*
[3]*Department of Forestry, Faculty of Natural Resources and Marine Science, Tarbiat Modares University, Tehran, Iran*

**Corresponding author: j.eslamdoust@modares.ac.ir*

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Abstract: Vegetation dynamics, soil properties, and the correlation between them following a wildfire are crucial to understanding the recovery of forest (natural or planted forest) ecosystems. We compared species composition and soil properties in two burned (Br) and unburned (UBr) sites of loblolly pine (*Pinus taeda* L.) stand in Northern Iran. We detected 39 plant species including 22 (56.4%) species that were common in both sites, 13 (33.3%) species specifically in the Br site, and 4 (10.3%) species specifically in the UBr site. Although species abundance was significantly higher in the UBr site, species richness was significantly higher in the Br site. Species composition was significantly different (F = 16.25, P-value = 0.001) between Br and UBr sites. Rarefaction-extrapolation revealed consistently and significantly higher species diversity in Br site compared to UBr site for all three Hill numbers. Only sand (t = 2.23, P-value = 0.030), pH (t = 2.44, P-value = 0.018) and electrical coductivity (t = 2.98, P-value = 0.004) were significantly higher (P-value ≤ 0.05) in the Br site due to the demobilization of base cations in burnt vegetation. In the Br site, the wildfire did not cause any marked changes in C and N stocks.

Keywords: forest management; ground flora; litter thickness; soil organic matter; species composition

Globally, the overall area of planted forests has been estimated at 294 million ha, which is 7% of the world forest area, and it increased by 123 million ha between 1990 and 2020 (FAO 2020). Planted forests are crucial renewable sources of raw material, both in environmental as well as social and economic terms. Also, tree plantations can provide other significant ecosystem services like regulation of water flow, improvement in soil fertility, and carbon sequestration (Humpenöder et al. 2014). One of the main reasons for planted forests around the world is to reduce the pressure on natural forests (slow or reverse deforestation and forest degradation) by providing profitable wood products for

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the market, e.g. lumber, fuelwood, pulp, and paper (FAO 2001; Guedes et al. 2018). To this subject, fast-growing species are mainly considered because of their capability to produce a mean annual increment of at least ten cubic meters per hectare under favourable site conditions.

Loblolly pine (*Pinus taeda* L.) is one of the world's most important commercial species (Prestemon, Abt 2002). This species is native to the southeastern United States and is economically valuable because it can grow remarkably well outside its native range (Wallinger 2002) in differently textured soils, from deep sands to heavy-textured clays. It has rapid regeneration, substantial yields per hectare, and numerous marketable products (Fox et al. 2007). In addition to the United States (Perdue et al. 2017), *Pinus taeda* is widely cultivated in Brazil (Dobner, Campoe 2019), China (Jin et al. 2019), Mozambique (Guedes et al. 2018), Uruguay (Leites et al. 2013), Argentina (Martiarena et al. 2011), Iran (Picchio et al. 2020) as an exotic (non-native) fast-growing species. Therefore, as an exotic species planted in many countries, it is expected to be more impacted by natural disturbances.

Natural disturbances such as fire, insect outbreaks, ice storms, or windthrow are globally increasing mainly due to current and feature global changes (climate changes, land use and land cover changes, social changes) (Dobner, Campoe 2019). Wildfires are one of the most significant primary abiotic disturbances in different ecosystems (Barreiro, Díaz-Raviña 2021) and are the result of high fuel availability, low humidity, high temperature, and high wind speed (Leite et al. 2015). In many territories of the world, wildfires are influential contributing factors that explain habitat structure, ecosystem functioning, and community composition (Bond et al. 2005). Fire influences natural ecosystems by burning plants and changing the succession pattern. Its impact on the ecosystem plays a determining role in the present species and dynamics of forest ecosystems. However, in many cases, surface fire (the height of flames is roughly 10 cm to 30 cm under standard humidity and fuel conditions) occurs and specifically impacts understorey plant species and soil in the planted forests (Peterson, Reich 2008).

Planted forests supply habitats for wildlife and provide conditions for recolonization by native plant species that help the conservation of biological diversity (Chamshama et al. 2009). Biological diversity or biodiversity is crucial for the functionality of ecosystems. Nutrient cycling, resilience, and succession are meaningful ecological roles of plant diversity in forest ecosystems (Poorbabaei, Poor-Rostam 2009). Biodiversity provides essential food and habitat resources for many wild species, including insects, birds, and deer (Felton et al. 2018). The loss of vegetation is the most direct impact of a wildfire, but additionally, the soil ecosystem is also severely affected by the fire and hence there may be a loss of soil quality. The physical and chemical properties of soil are affected by fire severity, which is related to several environmental factors such as moisture content of dead and alive combustibles, wind speed, and site topography (Certini 2005). The principal direct effect of fire on soil physical properties is related to the combustion of organic matter (Mataix-Solera et al. 2011).

Recently, different reviews have addressed the impact of fire on vegetation (Stavi 2019) and soil physicochemical properties (Minervini et al. 2018). It emphasizes the importance of predicting the impact of natural factors and management regimes on biodiversity and soil (Balvanera et al. 2006). Although there is a great deal of published information on the initial stages and long-term fire recovery of understorey species in *Pinus taeda* planted forests (e.g. Iglay et al. 2014; Matusick et al. 2020; Westlake et al. 2020), there is less published information on the two years recovery after wildfire compared with non-fire affected *Pinus taeda* planted forests. From this point of view, our study looked for the influence of surface wildfire on soil properties and understorey vegetation. Specific objectives of the study were (*i*) how species abundance and richness change two years after wildfire, (*ii*) how soil properties are affected by low-intensity wildfire, (*iii*) what of the stand or soil properties mainly determines the species abundance and richness.

MATERIAL AND METHODS

Study area. The study area is located in Saravan Forest, Guilan Province (37°09'12''N, 49°35'40''W – Northern Iran), in the temperate climate conditions which are characterized by high precipitation with a strong seasonal pattern. Mean annual rainfall is 1 189 mm. 67% (796 mm) of the rain occurs in the autumn-winter periods and 33% (393 mm) during spring and summer. Mean annual temper-

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ature is 15.9 °C and the air temperature typically ranges from 14.5 °C (mean minimum temperature of the coldest month) to 28.3 °C (mean maximum temperature of the hottest month). A forest site was identified which was affected by low-intensity wildfire on July 21st, 2017, with an area of approximately 5 ha. Simultaneously, the burnt (Br) site was situated in close proximity to the unburned (UBr) site selected as control. Site conditions (topography, altitude, soils, etc.) were similar among the selected sites. The age of plantations with the same planting distance (4 m × 4 m planting distance) was 24 years. Table 1 shows the plantation characteristics of tree density, basal area, *DBH* (diameter at breast height) and canopy cover.

Data collecting. Sixty plots (1 000 m^2) were established using a random systematic 100 m × 200 m grid. Edge effect was minimized by placing plots at least 30 m from the site boundary and roads. Within each plot, a subsampling technique was employed based on the minimal area method and Whittaker nested plot sampling protocol. Thus, the percent cover of each ground flora was measured using 8 m^2 subplots (centre of each plot). All ground flora individuals were identified, and those that could not be identified to species in the field were transported to the laboratory where they were pressed, dried, and identified by experts. Live standing trees larger than 9 cm in diameter at breast height (*DBH*, 1.37 m) were recorded in each plot. Basal area ($m^2 \cdot ha^{-1}$) was calculated from standing trees > 9 cm in diameter at *DBH*. The forest survey and vegetation measurements were conducted from May to August during the growth season, two years after the wildfire.

Soil sampling and analysis. Soil samples were taken from the first 20 cm of the topsoil with a 7.5 cm diameter soil core at the centre of each plot (a total of 60 soil samples for both sites). To ensure homogeneity in soil samples, we manually removed the soil organic layer before sampling the mineral layer, allowing for a direct comparison of the soil properties between the UBr and Br sites. Soil samples were placed in Ziploc bags before transfer to the lab. Litter depth was measured at the four corners of 8 m^2 subplots. All soil samples were sieved through a 2 mm sieve to remove roots and rocks and air-dried for physical and chemical analysis. The hydrometric technique was used to determine the percentages of clay, sand, and silt (Bouyoucos 1962). A subsample was oven-dried at 105 °C for 24 h to calculate the soil sample water content. Soil samples were soaked in distilled water with soil : water of 1 : 2.5 was stirred sufficiently and left to sit for 30 min until translucent suspensions. Soil pH and EC (electrical conductivity) were determined using an Orion Ionalyzer Model 901 EC and pH meter (Thermo Orion, USA). Soil bulk density was measured by the clod method (Soleimany et al. 2021). Soil porosity was calculated by the formula [1 – (bulk density/particle density)]. Organic carbon (OC) was measured by dichromate oxidation (Allison 1965) and N by wet oxidation using the Kjeldahl method.

Data analysis. All statistical analyses were conducted in R (Version 3.6.3, 2020), and significance was accepted at P < 0.05. To analyse the effects of wildfire on species communities, we used the general linear model (GLM) in the MASS package (Version 7.3-55, 2021). We used non-metric multidimensional scaling (NMDS) to visualize and test differences within the species community composition between Br and UBr sites (Oksanen et al. 2013). To compare diversity between Br and UBr sites, we used a framework published by Chao et al. (2014), implemented in the R package "iNEXT" (Hsieh et al. 2016). This framework unifies interpolation and extrapolation approaches with Hill numbers, which allows analysing rare to dominant species in a common framework. We calculated diversity for the diversity of rare (q = 0), common (q = 1), and dominant species (q = 2) (Chao et al. 2014). Significant differences in estimated diversity between sites were judged by non-overlapping confidence intervals (Schenker, Gentleman 2001). Two-sample t-test was used to compare the means of soil variables between sites. All figures are made by using the ggplot2 package (Version 3.3.5, 2021). Significant differences are indicated in the tables by P-values along with the significance level (*significant at 5%, **significant at 1%, ***significant at 0.1%).

Table 1. Stand characteristics (mean ± SD) of burned (Br) and unburned (UBr) sites

Variables	Br	UBr	*P*-value
Tree density ($n \cdot ha^{-1}$)	265.2 ± 32.8	277.3 ± 31.4	0.153
DBH (cm)	22.8 ± 4.7	24.2 ± 6.3	0.124
Basal area (m^2)	10.9 ± 3.0	13.2 ± 5.6	0.062
Canopy cover (%)	45.8 ± 11.4	53.4 ± 15.7	0.072

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RESULTS

We detected 39 plant species including 22 (56.4%) species that were common in both sites, and 13 (33.3%) species specifically in the Br site, and 4 (10.3%) species specifically in the UBr site. Species abundances (Figure 1A) were significantly higher in the UBr site, whereas species richness (Figure 1B) was significantly higher in the Br site (Table 2).

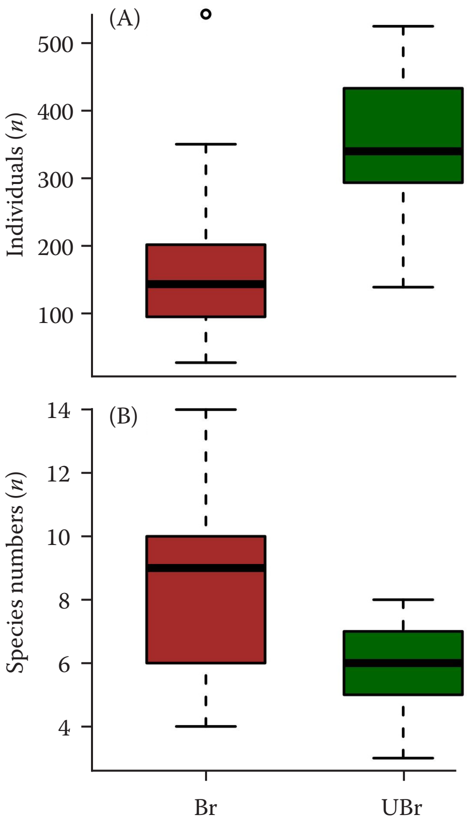

Figure 1. Mean individuals (A) and species numbers (B) per plot sampled across 60 Br and UBr plot

Br –burned; UBr – unburned

Table 2. Results of generalized linear model with abundance and species richness as response variable and sites as predictor

Response		Estimate	SE	*Z*-value	*P*-value
Abundance	intercept	5.106	0.014	359.40	< 0.001***
	sites (UBr)	0.788	0.017	46.00	< 0.001***
Species richness	intercept	2.128	0.062	33.78	< 0.001***
	sites (UBr)	–0.358	0.098	–3.65	< 0.001***

***significant at the 0.001 level (2-tailed); UBr – unburned

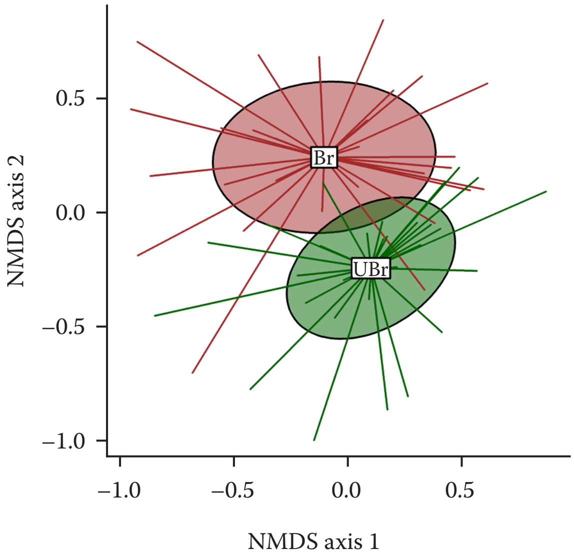

Figure 2. Non-metric multidimensional scaling of species composition sampled in Br and UBr sites (stress = 0.245)

Br –burned; UBr – unburned; NMDS – non-metric multidimensional scaling

Table 3. Pearson correlation between species abundance and richness, and variable types of stands and soil in two Br and UBr sites

Variables	Br		UBr	
	abundance	richness	abundance	richness
Abundance	–	0.661**	–	0.569**
Canopy cover	–0.495**	–0.777**	–0.334	–0.540**
Clay	0.219	0.106	–0.069	0.160
Silt	–0.247	–0.159	0.001	–0.203
Sand	0.011	0.122	0.155	0.142
Saturation moisture	–0.116	–0.019	0.096	0.043
Temperature	–0.312	–0.280	–0.291	–0.080
Balk density	–0.002	0.087	–0.219	–0.097
Porosity	–0.069	–0.235	0.192	0.085
EC	0.046	0.042	0.274	0.003
pH	–0.331	–0.417*	0.351	0.136
Organic carbon	–0.144	–0.064	–0.127	–0.165
N	0.046	0.042	0.274	0.003
C/N	–0.128	–0.002	–0.208	–0.099
Carbon stock	–0.175	–0.041	–0.158	–0.178
N stock	0.036	0.077	0.268	–0.002
Litter thickness	–0.467**	–0.734**	–0.375*	–0.511**

*;**significant at the 0.05 and 0.01 level respectively (2-tailed); EC – electrical conductivity; Br – burned; UBr – unburned

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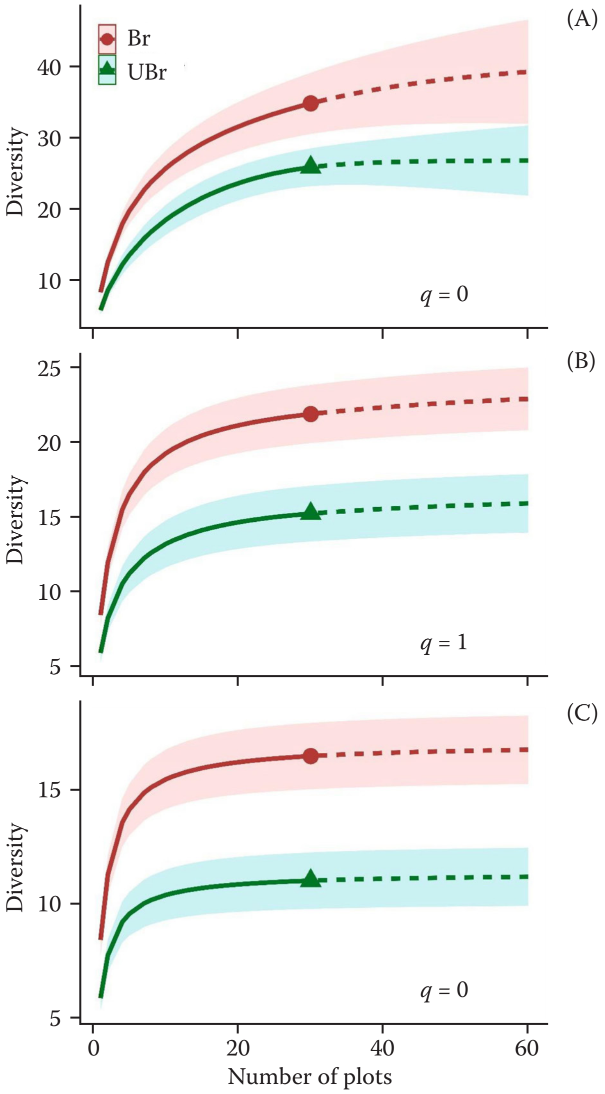

Figure 3. Sample-based rarefaction (solid lines) and extrapolation (dotted lines, up to twice the actual sample size) species diversity in Br and UBr sites, for (A) rare ($q = 0$), (B) common ($q = 1$), and (C) dominant ($q = 2$) species

Br –burned; UBr – unburned

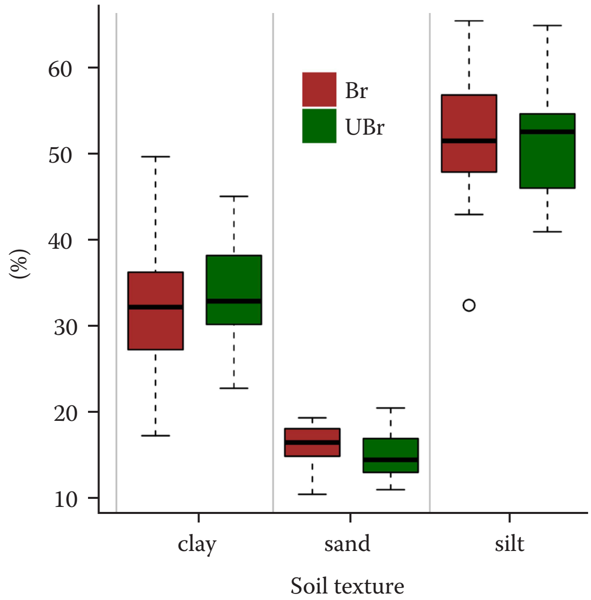

Figure 4. Soil clay, sand, and silt percentage in Br and UBr sites

Br –burned; UBr – unburned

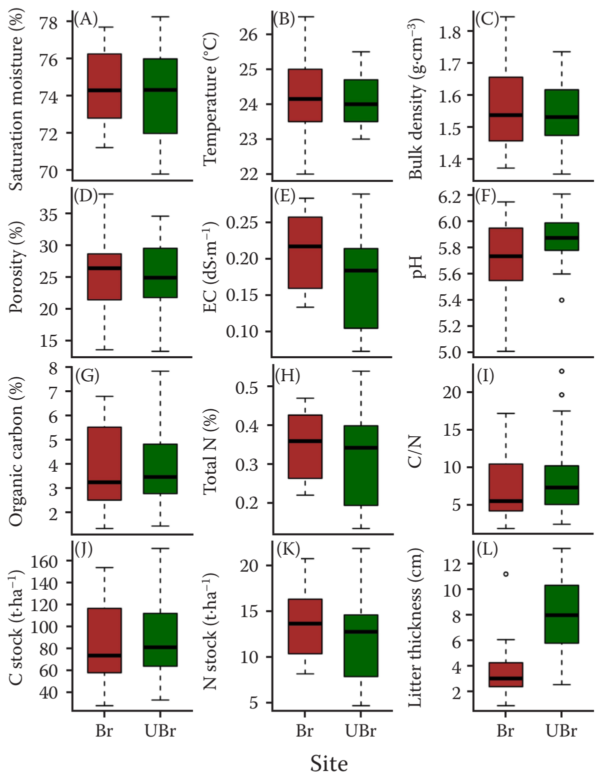

Figure 5. Soil variables of (A) saturation moisture, (B) temperature , (C) bulk density, (D) porosity, (E) EC, (F) pH, (G) organic carbon, (H) total N, (I) C/N, (J) carbon stock, (K) N stock and (L) litter thickness in Br and UBr sites

Br – burned; UBr –unburned

Species composition was significantly different between the Br and the UBr site ($F = 16.25$, P-value = 0.001) (Figure 2). Rarefaction-extrapolation revealed a higher species diversity in Br site compared to UBr site consistently and significantly (non-overlapping confidence intervals of the rarefaction and extrapolation curves) for all three Hill numbers (Figure 3A–C). Soil texture shows insignificantly higher clay content in the Br site and silt content in the UBr site. However, sand content was significantly higher in the Br site ($t = 2.23$, P-value = 0.030) (Figure 4).

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Soil variables of saturation moisture (Figure 5A), temperature (Figure 5B), bulk density (Figure 5C), porosity (Figure 5D), EC (Figure 5E), pH (Figure 5F), organic carbon (Figure 5G), total N (Figure 5H), and N stock (Figure 5K) were higher in the Br site compared to the UBr site but differences were significant only for pH (t = 2.44, P-value = 0.018) and EC (t = –2.98, P-value = 0.004). Although C/N (Figure 5I) and carbon stock (Figure 5J) were slightly higher in the UBr site, these differences were not significant. Litterfall thickness was significantly higher in the UBr site compared to the Br site (t = 6.99, P-value = 0.000) (Figure 5L).

A negative relationship was found between species abundance with canopy cover and litter thickness in the Br site and only litter thickness in the UBr site (Table 3). Although species richness was positively related to the species abundance in both sites, it was negatively correlated with canopy cover, pH, and litter thickness in the Br site and canopy cover and litter thickness in the UBr site (Table 3).

DISCUSSION

The results show that the fire caused an increase in species richness, but it decreased the species abundance after two growing seasons. These results are in line with the findings of Brewer (2016) and Ürker et al. (2018), indicating that the richness index is higher in the burned stands. Also, Karimi et al. (2019) found that many pioneer species appeared in areas 1 to 3 years after the fire. Barefoot et al. (2019) reported slightly higher species richness in burned than in unburned areas. The species abundance in the Br site was significantly lower than in the UBr site, but it can gradually increase based on Delitti et al. (2005). Furthermore, fire altered the species community composition. Similarly, Xiang et al. (2015) found significantly different species compositions in burned and unburned forests. Richter et al. (2019) found a positive correlation between the species richness and average thickness of litterfalls. It matches our findings since litter thickness in the Br site is significantly lower compared to the UBr site. Although these differences could also be linked to the density of tree overstorey, we observed a similar density of overstorey trees. However, it has been reported as a significant determinant of understorey variation in many studies due to its influence on light transmittance, soil insolation, moisture content, and nutrient cycling (Felton et al. 2010).

The main link between plant species and soil properties is the quantity and quality of nutrient sources. In this study, the fire altered the soil texture by increasing the sand and silt contents and reducing the clay content. We believe the fire intensity was not high enough to cause significant changes in soil texture. Changes in soil texture can occur at temperatures above 250 °C (Giovannini, Sun 2012) by forming silt and sand particles from fine clay particles, which might not happen in this study. Similarly to our result Kara and Bolat (2009) found a higher sand content and lower clay content in burnt soil in Turkey.

In this study, the fire increased soil bulk density but it was not significantly different in comparison with the UBr site. Similarly, Hubbert et al. (2006) found the increased soil bulk density after a fire in an oak forest in Middle Tennessee, USA. Arunrat et al. (2021) indicate that lower clay content is mainly related to higher soil bulk density, which was the case of our study. Soil pH is a critical soil feature that determines the availability of plant nutrients. Therefore, pH changes will have subsequent impacts on soil nutrients (Prendergast-Miller et al. 2017). Wildfire increased soil pH and EC two years after the fire that were significantly different from the levels observed in the UBr site. Muñoz-Rojas et al. (2016) found a significant increase in pH and EC after the fire. Similarly, Fachin et al. (2021) mentioned that soil pH increased after the fire. These results are related to the high amounts of ash and wood charcoal and the slow release of alkaline cations into the soil (Certini 2005). The findings of Knicker et al. (2007) comply with our results, reporting significantly higher EC values in the burnt compared to the unburnt plots, mainly due to increased levels of major cations in the soil. Although our result indicates these variations two years after the fire, Fonseca et al. (2017) observed that pH and EC values after thirty-six months from fire were similar to those seen before the fire.

The soil moisture variations were not significant in both studies, perhaps due to the recovery period after the fire. On the other hand, Arunrat et al. (2022) indicated that the reason for non-significant changes in soil moisture was most likely because the fire intensity was low to medium as it took place in the present study. Based on Keeley (2009), fire intensity refers to the energy or heat released during various phases of fire. Santín et al. (2016b) mentioned that moisture content and fire inten-

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sity depend on the extent of litterfall consumption by fire. Although the soil temperature was higher in the Br site compared to the UBr site, but with a non-significant difference, in our observation, the surface in the Br site can absorb more sunlight due to dark looks because of necromass such as burned leaves, litter, and partially burned branches. Accordingly, Alexis et al. (2007) found an insignificant increase in soil temperature in prescribed fires.

Soil organic matter is a significant soil property that increases soil resource availability and water-holding capacity and improves the soil structure. (Reynolds et al. 2003). Meanwhile, fires can modify the amount of soil organic matter and change the structure and composition of plant communities (Nghalipo, Throop 2021). We found no significant changes in OC between Br and UBr sites, most probably because of the low fire intensity (the heat was not strong enough to impact significantly) and low mineralization rates. Our findings are consistent with Nave et al. (2011), who observed that the prescribed fire had no significant effects on OC. Lucas-Borja et al. (2019) highlighted the low-intensity fire to explain the stable levels of OC and N stocks. An increment of total carbon concentration in soils of boreal forests was reported by Santín et al. (2016b) after a fire occurred. In other ecosystems, Nichols et al. (2021) reported a decrease of C after a fire in sagebrush steppe ecosystems of the Columbia Basin. Soil C is a strong predictor of plant communities due to its influence on soil water-holding capacity and nutrient retention and availability. Our results indicate that the low-intensity fire has no impact on ecosystem functioning through changes in the soil C stock. An insignificant slight decrease of the C stock was observed at the Br site two years after the fire, most likely it was associated with leaching from this upper to deeper soil layers (Jones et al. 2020). All subsequent changes emanated from the movement of carbon compounds under the influence of water. In contrast to our findings, Fairman et al. (2022) found a significant decrease in carbon stocks at 10 cm soil depth in a eucalypt forest after a wildfire. We observed insignificantly higher N and N stock in the Br site. These findings are consistent with the study of Xiang et al. (2015), who reported a non-significantly higher content of available nitrogen one-year after a wildfire in a Chinese boreal forest. Alteration of soil N cycling was found following the fire disturbance in different ecosystems (Ball et al. 2010; Stephan et al. 2015; Prendergast-Miller et al. 2017). The N cycle in forest soil is an internal cycle between the vegetation and the pool of N. Arunrat et al. (2022) mentioned that fire is not the main factor affecting N, but also N was likely impacted by plant uptake.

It is reasonable to observe a decrease in the litter thickness of the Br site after the fire. We found a significant 57.6% reduction of litter thickness in the Br site compared to the UBr site. Litter is the primary fuel for starting and spreading fire, especially in low-intensity surface fires (Volkova et al. 2019). Although we did not examine the litter biomass, Espinosa et al. (2018) found a reduction between 59 and 77% of litter biomass in mixed stands of *Pinus nigra* and *Pinus pinaster* and pure stands of *Pinus nigra* in central-eastern Spain after prescribed burning. Nevertheless, depending on the forest ecosystem type (natural or planted forest), litterfall reaches pre-fire levels in two to seven years (Dymov et al. 2017).

Dymov et al. (2021) indicated that the soil chemical composition was similar to that before the fire two years after the fire. This can probably be explained by the renewal of ground cover plants and the influx of fresh litter contributing to an increase in the mobility of organic compounds in soil. An important aspect to be considered with post-fire recovery and longevity of fire impacts is the type of ecosystem involved (Prendergast-Miller et al. 2017). The differences in C or N stock in the mineral layer are significantly affected by species difference between aboveground litterfall inputs and decomposition, primarily controlled by the quality of the litter (Li et al. 2012; Rafiei Jahed et al. 2017). The effects of tree species in a planted forest on soil C and N stock changes after the afforestation were reported by Li et al. (2012). Consequently, more C accumulates in the mineral layer for conifer species, notably pine (Li et al. 2012). However, previous studies have reported a wide variability in soil chemicals and nutrients within and among different ecosystems (Allen et al. 2011; Santín et al. 2016a), mainly due to fire intensity (Bird et al. 2015).

CONCLUSION

This study research provides a better understanding of dynamics and changes in vegetation cover and soil properties which have a relevant role in forest management planning. Here we studied the species diversity and composition, and

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most fire-affected soil properties following wildfire in *Pinus taeda* stands as an alien species in the temperate area of Northern Iran, Central Asia. Our findings indicated that fire altered the species composition. Moreover, although species abundance was negatively affected in the Br site, species richness increased dramatically within two years after the fire. There were significant differences in species diversity based on rarefaction-extrapolation between Br and UBr sites. Due to the influence of a low-intensity fire on soil, the number of soil properties (saturation moisture, temperature, bulk density, porosity, EC, pH, organic carbon, total N, and N stock) increased within two years after the fire. However, only pH and EC were found significantly different between Br and UBr sites. We showed that a single low-intensity fire in *Pinus taeda* stands improved species richness and diversity by reducing litterfall thickness. Finally, as we have not found a significantly negative impact of wildfire after two years, we suggest the prescribed fire (considering fire intensity) as a potentially effective mechanism to decrease the litterfall thickness and promote species richness and diversity in similar ecosystems. However, long-term monitoring of vegetation and soil fertility in the planted forest in future studies is needed.

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