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RESEARCH ARTICLE

Long-term effect of fire on herbaceous species diversity in oriental beech (*Fagus orientalis* Lipsky) forests in northern Iran

Mohammad Naghi ADEL[1*], Hassan POURBABAEI[1], Ali OMIDI[2], David POTHIER[3]

[1] Department of Forestry, Natural Resources Faculty, University of Guilan, Somehsara, P. O. Box 1144, Guilan, Iran
[2] Department of Forestry, Natural Resources Office, Guilan, Iran
[3] Centre d'étude de la forêt (CEF) et Département des sciences du bois et de la forêt, Pavillon Abitibi-Price, 2405 rue de la Terrasse, Université Laval, Québec, QC G1V 0A6, Canada

Abstract We studied the long-term impacts of natural burning on herbaceous species diversity 37 years after a fire occurred in the Roudbar forests of Guilan Province, northern Iran. Numerous studies have examined short-term changes in understory vegetation following wildfire; however, very few long-term studies are available or changes inferred from retrospective studies based on chronosequences. For this study, 170 ha of forest (85 ha, burned areas; 85 ha, unburned areas) were surveyed. Because the 1000 m^2 plots were too large for detailed measurements of herbaceous species, we determined a sub-sample size according to the Whittaker's nested plot sampling protocol and minimal areas method. Hence, sub-plots of 32 m^2 were used for herbaceous species measurements, which consisted of percent cover of each species based on the Domin criterion. We measured plant diversity (Shannon-Wiener index), species richness (Margalef's index), and evenness (Smith-Wilson index). Mean percent cover, together with diversity, richness and evenness, increased markedly in burned areas compared to unburned controls. This suggests that the biodiversity of these forests could be restored within 37 years after fire. However, the abundance of invasive species such as *Rubus fruticosus* and *Bromus benekenii* increased significantly in burned areas, but these could be controlled by relevant silvicultural operations.

Key words fire, diversity, richness, evenness, beech forest, northern Iran

1 Introduction

Despite a growing awareness that the herbaceous layer serves a special role in maintaining the structure and function of forests, this stratum remains an underappreciated aspect of forest ecosystems. Because species diversity is highest in the herb layer among all forest strata, forest biodiversity is largely a function of the herb layer community (Gilliam, 2007). Disturbances can cause major changes in plant communities depending on their intensity, extent, frequency, seasonality and the resilience of the component species (Herath et al., 2009). Fire is one of the most important disturbance factors in natural ecosystems throughout the world (Moretti and Barbalat, 2004). The degree of change from a pre-fire to post-fire community is affected by the intensity, severity, periodicity and seasonality of fire (Wright and Bailey, 1982), together with other factors such as precipitation cycles (Anderson et al., 1968; Moore et al., 2006) and grazing patterns (Zimmerman and Neuenschwander, 1984). Vegetation recovery following fire depends on the ability of plant populations to tolerate fire, to regrow from surviving tissues, or to establish from viable seeds that have remained in the soil, in the canopy seed bank, or in unaffected adjacent populations. This recovery is determined to a certain extent by the fire regime, in terms of both its temporal and spatial aspects and by the physical characteristics of fires (e.g., Lloret and Zedler, 2009).

Forests cover 12 million hectares of the land surface of Iran (8% of the total land area), of which about 1.8 million hectares are located in the northern part of the country, i.e., the Hyrcanian or Caspian Forest ecoregion. This forest type is situated on the northern slopes of the Alborz mountains overlooking the Caspian Sea

*Author for correspondence. E-mail: mn.adel87@yahoo.com

(Sagheb-Talebi et al., 2004). Extant forests of northern Iran consist mostly of broadleaf deciduous stands, but some areas are locally covered by Mediterranean-type vegetation. Moreover, the distribution of forest types in northern Iran is heterogeneous, with forest productivity following a decreasing west-east gradient. Caspian forests appear to be very similar to broadleaf forests of central Europe, northern Turkey and the Caucasus (Mohadjer, 2006). Fire is the main natural disturbance in the forests of northern Iran and burns 300–400 ha annually (Banj-Shafiei et al., 2010). These fires normally occur during the short drought season, which is caused by hot, dry winds that desiccate the forest floor. These conditions have mostly resulted in surface fires, which rarely exceed 10–30 cm in flame height under normal fuel and humidity conditions.

Numerous studies have examined short-term changes in understory vegetation following wildfire (e.g., Bernhardt et al., 2011); however, very few long-term studies are available or changes inferred from retrospective studies based on chronosequences (e.g., Lecomte et al., 2005; Bataineh et al., 2006). In addition, analyses of wildfire effects are rare because it is difficult to study random events with robust, replicated experiments (van Mantgem et al., 2001). Therefore, most studies of forest fire effects have focused on prescribed burns that are more easily controlled and manipulated (e.g., Reinhardt and Ryan, 1988; Moreira et al., 2003; Fernandez-Abascal et al., 2004). Knowledge regarding the effects of wildfire would greatly increase our understanding of the role of this form of disturbance in forests and serve to improve management decisions (Laughlin et al., 2004).

Forests dominated by oriental beech (*Fagus orientalis*) cover about 565000 ha and represent the total area of indigenous forests in Guilan Province. These forests thus form a major carbon pool (Hall et al., 2001) and are important economic, soil protection and recreation resources (Wardle, 1984). Oriental beech is poorly adapted to fire and even low temperature burns can eliminate it from the community. From 1992 to 2010 inclusive, 856 fires occurred in northern Iran, burning 2953 ha of forest in Guilan Province. Despite these numerous fire events, the impacts of forest fires on plant diversity have never been studied in this region. Studying and surveying fire effects on the composition of both the forest overstory and understory layers would yield important information on forest vegetation recovery, which is necessary to forecast forest composition and succession. Such studies could help determine the intensity of forest disturbance and suggest methods that aim at reestablishing the former forest composition. The general objective of this study was to investigate the impact of natural fires on herbaceous species diversity in Hyrcanian forests, 37 years after fire. We hypothesized that fire has no effects on different measures of plant diversity.

2 Materials and methods

2.1 Study area

The study area is located in Zilaki, near Roudbar City, which is in the southern part of Guilan Province, northern Iran (36°54′30″–36°56′06″ N, 49°46′24″–49°51′17″ E). Elevation within the study area ranges from 1010 to 1560 m a.s.l., with 30% – 40% slopes that generally face northward. Common forest soils are deep and brown and have a heavy texture and weak acidic pH. Parent materials include lime silt, sandstone, siltstone and shill. The climate, based on the Emberger classification (Daget, 1977), is very humid with mean annual precipitation of 1560 mm at the nearest meteorological station (Rasht City). The annual mean maximum temperature has been recorded in August (29.3°C) and mean minimum temperature in February (2.7°C). The usual harvest method employed in the Hyrcanian forest is a single-tree selection system, but logging has never occurred in the study area, despite the existence of forest management planning. The lack of harvesting activity can be attributed to the absence of a network of forest roads. Consequently, the forest is uneven-aged and is composed of mixed deciduous broadleaf stands dominated by *Fagus orientalis*. A very severe fire occurred at the beginning of December 1973, covering 85 ha in seven days.

2.2 Data collection

This study included both burned (B) and unburned (UB) areas, each covering 85 ha. To avoid fire effects, the unburned area was separated from the burned area by a buffer of at least 100 m. B and UB areas shared similar elevations, slopes and aspects. In each area, we used a random systematic 150 m × 200 m gird sampling plan to establish 30 1000-m^2 circular plots in July 2010, i.e., 37 years after fire. In each plot, we recorded habitat factors such as slope percentage, geographical aspect, elevation, crown canopy percentage and percent cover of each herbaceous species. In addition, litter depth was measured at five locations within each plot. Because the 1000-m^2 plots were too large for detailed measurements of herbaceous species, we determined a sub-sample size according to the Whittaker nested plot sampling protocol and minimal areas method (Cain, 1938). Hence, sub-plots of 32 m^2 were used for herbaceous species measurements, which consisted of percent cover of each species based on the Domin criterion.

2.3 Data analysis

To evaluate the effect of fire on different aspects of herbaceous biodiversity, we used three indices. First, species diversity was assessed with the Shannon-Wiener index (Magurran, 1988):

$$H' = -\sum_{i=1}^{s} p_i \ln p_i \tag{1}$$

where p_i is the relative frequency of the i^{th} species.

Second, species richness was estimated according to the Margalef index (Ludwig and Reynolds, 1988):

$$R_1 = \frac{S-1}{\ln N} \tag{2}$$

where S is the total number of species and N is the total number of individuals.

Last, the Smith-Wilson index (Smith and Wilson, 1996) was used to calculate species evenness:

$$E_{\mathrm{var}} = \frac{2}{\pi \arctan\left\{\sum_{i=1}^{s}\left(\log_e(n_i) - \sum_{j=1}^{s}\log_e(n_j)/s\right)^2 \Big/ S\right\}} \tag{3}$$

where n_i is the number of individuals of the i^{th} species in a plot, n_j the number of individuals of the j^{th} species and S the total number of species in B and UB areas. All three indices were computed with software provided by Krebs (1989; Ecological Methodology for Windows, version 6.0).

Each index was computed for each sample within both B and UB areas. Kolmogorov-Smirnov tests were used to verify the normality of their distributions. Normality tests were followed by comparisons of means between B and UB, using two-sample *t*-tests or their non-parametric equivalents (Mann-Whitney *U*-tests) if the data were not found to be normally distributed. All statistical analyses were performed in SPSS (version 16.0, SPSS Inc., Chicago, USA). Significance levels were set to $p = 0.05$.

3 Results

Percent canopy cover, percent herbaceous cover and leaf litter depth differed significantly ($p < 0.05$) between B and UB areas, 37 years after fire (Table 1). While percent canopy cover and leaf litter depth were both greater in unburned areas, significantly greater herbaceous cover was associated with the burned area compared to the unburned area (Table 1).

Overall, 32 families and 37 species were found in the study area (Table 2). The burned area was represented by 28 families and 35 species, while the unburned area was represented by 26 families and 31 species. The most frequently represented family was the Lamiaceae (Labiatae) with three species, followed by three families, Aspleniaceae, Euphorbiaceae and Rosaceae, each represented by two species. All other families were each represented by only one species. *Asplenium adiantum*, *Bromus benekenii*, *Epimedium pinnatum*, *Rumex* sp., *Stellaria media* and *Pteris dentata* were confined to burned areas, while *Blechnum spicant* and *Platanthera bifolia* were only present in unburned areas. All remaining species were found in both B and UB areas.

The most frequently observed species in the burned area were *Polystichum woronowi* (28), *Carex* sp. (27), *Bromus benekenii* (24), *Asperula odorata* (24), *Rubus fruticosus* (23), *Viola sylvestris* (22) and *Euphorbia amygdaloides* (21). In the unburned area, the most frequently observed species were *Polystichum woronowii* (28), *Viola sylvestris* (27), *Lamium album* (22), *Euphorbia amygdaloides* (21) and *Cardamine* sp. (19). Five or fewer individuals were observed for 14 species in the burned area and for 12 species in the unburned area. In addition, six species were observed equally as frequently in B and UB areas: *Polystichum woronowii* (28), *Euphorbia amygdaloids* (21), *Calamintha officinalis* (16), *Hypericum androsaemum* (14), *Phylitis schlopendrium* (12) and *Tamus communis* (4) (Table 2).

Species exhibiting the highest percent cover in burned areas included *Euphorbia amygdaloides* (22.7%), *Bromus benekenii* (20.8%), *Asperula odorata* (18.1%), *Polystichum woronowii* (15.4%), *Primula heterochroma* (15.2%), *Carex* sp. (11.7%) and *Cardamine* sp. (10.1%), whereas in unburned areas, the species included *Euphorbia amygdaloides* (15.3%), *Asperula odorata* (13.9%), *Cardamine* sp. (11.4%) and *Polystichum woronowii* (10.2%). Six species in burned areas and seven species in unburned areas had percent covers < 1%. The percent cover of 10 species

Table 1 Characteristics of the unburned and burned study areas, 37 years after fire

Index	B	UB	*p*
Slope (%)	30.5	33.1	–
Elevation (m)	1295.6	1276.3	–
Canopy cover (%)	81.1	86.0	0.028[a]
Herbaceous cover (%)	50.1	47.4	0.004[a]
Leaf litter depth (cm)	7.05	8.82	0.014[a]

Note: B, burned; UB, unburned. *p*-values below 0.05 indicate significant differences between B and UB areas.

was significantly greater ($p < 0.05$) in the burned compared to the unburned area (Table 2; Fig. 1).

Estimates for all three biodiversity indices (diversity, richness and evenness) for the burned area were significantly higher than for the unburned area (Table 3).

Table 2 Frequency and mean percent cover of herbaceous species in burned and unburned areas, 37 years after fire

Species	Family	Frequency		Mean percent cover (%)		*p*
		UB	B	UB	B	
Asperula odorata (*Galium odoratum*) (L.) Scop	Rubiaceae	17	24	13.88	18.12	0.050[a]
Asplenium adianthum-nigrom L.	Aspleniaceae	0	1	0	0.50	–
Blechnum spicant (L.) Sm.	Blechnaceae	3	0	2.66	0	–
Bromus benekenii (Lange) Trimen	Gramineae (Poaceae)	0	24	0	20.81	–
Calamintha officinalis Moench	Labiatae (Lamiaceae)	16	16	2.28	4.78	0.028[a]
Campanula odontosepala Boiss.	Campanulaceae	3	4	2.66	0.87	0.381
Cardamine sp.	Cruciferae (Brassicaceae)	19	9	11.42	10.11	0.736
Carex sp.	Cyperaceae	15	27	3.96	11.37	0.000[a]
Epimedium pinnatum subsp. *colchicum* AGM	Podophyllaceae (Berberidaceae)	0	1	0	9.00	–
Euphorbia amygdaloides L.	Euphorbiaceae	21	21	15.33	22.66	0.050[a]
Fragaria vesca	Rosaceae	18	12	1.77	2.45	0.177
Geranium robertianum L.	Geraniceae	1	4	2.00	0.87	0.272
Hypericum androsaemum L.	Hypericaceae	14	14	1.71	3.67	0.046[a]
Lamium album	Lamiaceae	22	18	4.47	4.86	0.819
Mentha spicata L.	Labiatae (Lamiaceae)	15	13	2.30	4.53	0.013[a]
Mercurialis perennis L.	Euphorbiaceae	4	7	4.50	8.70	0.028[a]
Petasites hybridus(L.) G.Gaertn., B.Mey. & Scherb	Compositae (Asteraceae)	3	7	4.16	1.35	0.104
Phyllitis schlopendrium L.	Aspleniaceae	12	12	0.63	4.83	0.005[a]
Phytolacca decandra L.	Phytolaccaceae	1	5	0.50	6.00	0.046[a]
Platanthera bifolia	Orchidaceae	1	0	0.50	0	–
Polygonatum orientale Desf.	Liliaceae (Asparagaceae)	10	12	1.25	2.21	1.000
Polystichum woronowii Fomin	Aspidiaceae	28	28	10.21	15.39	0.021[a]
Primula heterochroma Stapf.	Primulaceae	16	11	5.37	15.18	0.020[a]
Pteridium aquilinum (L.) Kuhn	Hypolepidaceae	9	5	5.22	6.70	0.659
Pteris dentate Forssk.	Pteridaceae	0	1	0	0.50	–
Ranunculus brutius Ten.	Ranunculaceae	1	8	2.00	6.87	0.025[a]
Rubus fruticosus L.	Rosaceae	14	23	1.60	4.47	0.016[a]
Rumex sp.	Polygonaceae	0	2	0	2.00	–
Salvia glutinosa	Labiatae (Lamiaceae)	9	10	1.88	3.15	0.108
Sanicula europaea	Umbelliferae (Apiaceae)	2	1	2.00	2.00	1.000
Sedum stoloniferum S.G. Gmel.	Crassulaceae	8	11	7.87	6.31	0.614
Solanum kieseritzkii C.A. Mey.	Solanaceae	1	6	0.50	6.00	0.048[a]
Stellaria media (L.) Vill.	Caryophyllaceae	0	2	0	0.50	–
Tamus communis (*Dioscorea communis* (L.) Caddick & Wilkin)	Dioscoraceae	4	4	0.50	0.87	0.317
Vicia crocea B. Fedtsch	Papilionaceae (Fabaceae)	4	6	0.50	4.12	0.022[a]
Vincetoxicum scandens Sommier & Levier	Asclepiadaceae	11	7	1.45	2.57	0.305
Viola sylvestris Lam.	Violaceae	27	22	3.92	6.91	0.040[a]

Note: *p*-values < 0.05 indicate a significant difference in mean percent cover between B and UB.

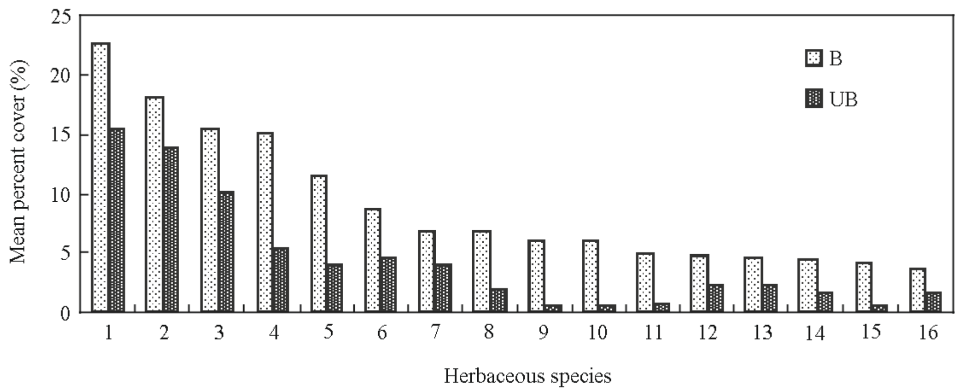

Fig. 1 Mean percent cover of species showing significant differences between B and UB areas. Each herbaceous species is denoted with the first three letters of the gender and the species whereas their full names are listed in Table 2. 1: *Euphorbia amygdaloides*; 2: *Asperula odorata*; 3: *Polystichum woronowii*; 4: *Primula heterochroma*; 5: *Carex* sp.; 6: *Mercurialis perennis*; 7: *Viola sylvestris*; 8: *Ranunculus brutius*; 9: *Solanum kieseritzkii*; 10: *Phytolacca decandra*; 11: *Phyllitis schlopendrium*; 12: *Calamintha officinalis*; 13: *Mentha spicata*; 14: *Rubus fruticosus*; 15: *Vicia crocea*; 16: *Hypericum androsaemum*.

4 Discussion

Percent herbaceous cover is significantly greater in the burned than in the unburned area (Table 1), a result reflected in the higher values of the biodiversity indices (Table 3). Other studies have demonstrated an increase in herbaceous species cover over time after fire (Nuzzo et al., 1996; Arthur et al., 1998; Elliott et al., 1999; Hutchinson and Sutherland, 2000; Kuddes-Fischer and Arthur, 2002; Elliott et al., 2009).

This response can be partly explained by the lower canopy cover in the burned area, which increases light availability for the understory. Accordingly, plant species richness in forest environments seems to be largely affected by the percent cover of the dominant canopy left after disturbance (Bazzaz, 1975; Rees and Juday, 2000). Indeed, the death of canopy trees can become a critical process in increasing resource availability (Ludwig et al., 2004) that could be related to the two-fold increase in herbaceous production 20 years post-fire, observed by Lowe et al. (1978) and to the increase in herbaceous species richness observed in a Missouri Ozark oak forest (Hartmann and Heumann, 2003).

Increases in herbaceous cover, richness and diversity can also be related to leaf litter depth. In the absence of fire, litter often inhibits germination by preventing seed contact with the soil, while canopy trees intercept light necessary to initiate germination (McConnell and Menges, 2002). Moreover, litter acts as a mechanical barrier against seed germination and seedling establishment (Hutchinson et al., 2005). Therefore, by consuming a considerable amount of organic matter (Loucks et al., 2008), fire can prevent litter accumulation and should promote seedling establishment (Vickery, 2002). The significantly thinner litter layer in the burned area of the present study (Table 1) thus suggests that it could be related to the higher herbaceous cover, diversity and richness. This is in accordance with Ffolliott et al. (2009), who attributed an increase in herbaceous production to litter reduction after fire and with Abrahamson and Harnett (1990) who observed higher species richness after fire due to a reduction in litter depth.

The consumption of leaf litter by fire also releases nutrients that are then incorporated into the mineral soil, thereby altering soil chemistry and likely affecting plant productivity, particularly on nutrient-poor sites (Gilliam and Christensen, 1986). The reduction of litter thickness by fire releases immobilized nutrients that could be rapidly assimilated by plant species, resulting in increases in their biomass (Brockway and Lewis, 1997). For example, Boerner et al. (2003) observed an increase in soil calcium availability after fire. Similarly, the germination of soil seed bank species can be stimulated after fire because of the in-

Table 3 Means (± SE) of biodiversity indices in burned and unburned areas, 37 years after fire. For each biodiversity index, p-values < 0.05 indicate significant differences between B and UB areas.

Index	UB	B	p
Margalef richness	2.27 ± 0.11	3.14 ± 0.12	0.000[a]
Smith-Wilson evenness	0.307 ± 0.02	0.391 ± 0.02	0.000[a]
Shannon-Wiener diversity	2.62 ± 0.07	3.11 ± 0.06	0.000[a]

creased availability of nitrates (Auchmoody, 1979).

Comparisons of biodiversity indices also show that fire increased species evenness (Table 3). This response suggests that fire could control the abundance of some dominant species, allowing for a more even distribution among all species, which is consistent with the observations of Brockway and Lewis (1997) and Akinsoji (1988).

Five species that were present in the burned area were absent from the unburned area (Table 2). Of these five species, one is potentially problematic. *Bromus benekenii* is an invasive species, which could strongly modify the community composition of the herbaceous layer in the burned area. This graminoid was absent prior to fire, but its percent cover exceeded that of all other species 37 years after fire. This species seems to be well-adapted to the new conditions that prevail after fire. In our study, a large proportion of the burned area was covered by *Bromus*, which could pose a threat to biodiversity. Therefore, to preserve diversity, silvicultural prescriptions such as herbicide applications should be used to decrease the abundance of this species. This situation is similar to the case of *B. tectorum* in the western United States. *Bromus tectorum* is a common exotic species, which increases its abundance after burning (Veblen, 2003; Franklin et al., 2006; Keeley, 2006). Another troublesome species is *Rubus fruticosus*, which exhibited a significant increase after fire. Because of its strong competitive capacity, this species could be a threat to other species and should be decreased by silvicultural operations. The increasing establishment of invasive species after fire can have a negative impact on native herbaceous communities because they can reduce plant biodiversity through inhibition of growth and development of other species.

In addition, six species were present at the same frequency in burned and unburned areas but their percent cover differed significantly (Table 2). Three of these species, *Polystichum woronowii*, *Euphorbia amygdaloides* and *Phyllitis schlopendrium*, considerably increased their percent cover after fire. In fact, these three species show that they can adapt to their local fire regime after burning, rapidly utilizing available resources and, therefore, increasing their percentage cover. Such adaptations include physical protection against heat, increased growth after a fire event and production of flammable materials that encourage fire and may eliminate competition. The response of these species to disturbance also shows that fire can stimulate the seedbank of this species. Smoke, charred wood and heat can stimulate the germination of seeds (Keeley and Fotheringham, 1998).

These results indicate that fire caused changes in species composition of the herbaceous layer, which could lead to changes in patterns of species dominance. This is in agreement with results obtained three decades after burning in Colorado (Coop et al., 2010), where an increase in richness and diversity of plant species was observed, together with changes in structure, composition and domination pattern of plant communities.

5 Conclusions

The present study surveyed long-term impacts of fire on herbaceous species diversity and we found that fire changed the composition and dominance patterns of the herbaceous community. Overall, the percent cover and diversity of the herbaceous layer increased significantly after fire, likely because of increasing light availability and decreasing litter depth. These results suggest that forest could recover 37 years after burning. Therefore, implementation of silvicultural practices are not required from a plant biodiversity point of view, with the exception of several invasive species, which might necessitate a certain control if their abundance will be maintained in the future.

The results of this study also suggest that prescribed burning can be considered as an interesting silvicultural tool if the main objective of forest managers is to maintain or even increase plant biodiversity at the landscape scale. However, because forest areas affected by fire seem to increase over time in Guilan Province, further studies are necessary to understand the deepening effects of fire on the diversity of herbaceous and woody species and on forest structures.

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