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Effect of fire intensity on the earlier post-fire succession in *Pinus nigra* Arnold stands of Taşköprü, Kastamonu

Kastamonu-Taşköprü'deki karaçam (*Pinus nigra* Arnold) meşcerelerinde yangın şiddetinin erken dönem süksesyon dinamiklerine etkisi

Fatma Nur Yılmaz[1]
Ömer Küçük[2]
Kerim Güney[2]
Bülent Sağlam[1]

[1] Artvin Çoruh Üniversitesi, Artvin
[2] Kastamonu Üniversitesi, Kastamonu

Abstract

This study investigates the post-fire vegetation development in black pine (*Pinus nigra*) stands, which are widely distributed in Kastamonu and are under threat from forest fires. To this end, sampling was conducted in 2017 in the Taşköprü and Bayam Forest Management Directorates, where fires occurred in 2003 and 2012. Sample plots (25 × 50 m) were established using the minimum area method in two regions affected by surface fires (low intensity) and crown fires (high intensity). To observe the temporal effects of high-intensity fire, a sample area from a 2003 fire site was also analyzed and compared using the Shannon diversity index and the Chi-square test. Results showed significant differences in the family Pinaceae and the life form cryptophyte. Although black pine was observed to remain the dominant species after the surface fire, one- and two-year-old species belonging to the Rosaceae and Fabaceae families were recorded during the early stage of vegetation. In areas with reduced canopy closure, species such as *Cistus laurifolius*, *C. creticus,* and *Rubus* sp. were observed. Species composition in the crown fire site closely resembled that of surface fire areas with broken canopy. *Cistus laurifolius* was the dominant species in crown fire areas, and the highest species diversity was found in the 2003 crown fire site.

Keywords: Blackpine, fire ecology, forest fire, post-fire vegetation

Özet

Çalışmada; Kastamonu'da geniş yayılış gösteren ve orman yangınları tehdidi altında bulunan karaçam (*Pinus nigra*) meşcerelerinde yangın sonrası vejetasyon gelişimi incelenmiştir. Bu amaçla, 2003 ve 2012 yıllarında yangınların meydana geldiği Taşköprü ve Bayam Orman İşletme Müdürlüklerinde 2017 yılında örnekleme yapılmıştır. Yangın şiddetinin etkisini görmek maksadıyla örtü (düşük yangın şiddeti) ve tepe (yüksek yangın şiddeti) yangınına maruz kalmış iki bölgeden en küçük alan yöntemi ile deneme alanları (25×50 m) alınmıştır. Yüksek yangın şiddetinin vejetasyon üzerinde etkisinin zamansal değişimini belirlemek amacıyla 2003 yılında yangın geçiren sahadan alınan örnek alan, diğer örnek alanlarla Shannon çeşitlilik indeksi ve Ki-kare analizi ile karşılaştırılmış; örtme dereceleri hayat formu ve familya bazında incelenmiştir. Analiz sonucu en yüksek farklılığa sahip olan familya Pinaceae olurken hayat formu kriptofit olmuştur. Örtü yangını geçiren alanda tepe kapalılığına sahip bölgelerde örtüş derecesi açısından hakim tür karaçam olmasına rağmen biyoçeşitlilik açısından tek ve iki yıllık Rosaceae ve bazı Fabaceae familyasına ait taksonlar tespit edilmiştir. Tepe kapalılığının kırıldığı yerlerde *Cistus laurifolius, C. creticus* ve *Rubus* sp.'ye rastlanmıştır. Tepe yangını sahası deneme alanında rastlanan türler ise örtü yangını geçirmiş, ancak tepe kapalılığı olmayan kısımlar ile tür kompozisyonu bakımından büyük miktarda benzerlik göstermiştir. Tepe yangını alanında hakim tür *Cistus laurifolius*'tur. En fazla çeşitlilik ise 2003 tepe yangını sahasında olmuştur.

Anahtar kelimeler: Karaçam, orman yangını, yangın ekolojisi, yangın sonrası vejetasyon

Sorumlu yazar (*Corresponding author)*
Fatma Nur Yılmaz
fatmanuryilmaz@artvin.edu.tr

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Oğuzhan Sarıkaya
oguzhan.sarikaya@btu.edu.tr

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1. Introduction

Fires are one of the factors that affect ecosystem structure and dynamics in a wide variety of ways (Pausas, 2019; Dafis, 1987; Agee, 1993; Küçük, 2006). Fires are part of the natural regeneration cycle (Arslantürk, 2007), and their periodic occurrence has been reported to be beneficial for the development of pine species (Barbero and Quezel, 1980; Thanos and Marcou, 1991; Agee, 1998; Richardson and Rundel, 1998).

In fire-adapted ecosystems, a fire occurring in a forest community where species diversity begins to decline allows the system to renew itself and alters the stabilized species diversity (Küçük, 2006). Although dominant taxa in the ecosystem are generally adapted to fire, their continued existence depends on elements of the fire regime (Kruger, 1983; Thanos and Marcou, 1991; Tsitsoni, 1991; Pausas et al., 2003; Kazanis and Arianoutsou, 2004). The fire regime-comprising the intensty, frequency, type, season, and size of fires in a region- can change the species composition and vegetation development of forested areas (Weber and Flannigan, 1997; Kilgore, 1981). Understanding the effect of a fire on the ecosystem is only possible through a comprehensive understanding of fire regime elements (Küçük, 2006).

Fire frequency is expressed as the number of fires that occur at specific time intervals and plays an important role in determining the life history of a species. For a species to survive in an area, the age at which it reaches seed-setting maturity must be compatible with the fire frequency. Therefore, fire frequency can alter the floristic composition of an ecosystem by determining the species that constitute the vegetation structure of an area (Ryan, 2002). The size of the area affected by fire is one of the important ecological factors that influence the recolonization of the area by the species that make up the ecosystem. For example, if plants do not have the ability to regenerate from shoots or form metamorphic stems beneath the soil, or if seeds are not transported by wind or animals, some species may be unable to recolonize if the burned area is too large for effective seed dispersal. The season in which the fire occurs is also important, as it directly influences the moisture content of the combustible material.

In order to better understand the ecological effects of fire, post-fire vegetation studies —which have been conducted in forest ecosystems in several regions of the Mediterranean Basin— should be emphasized (Agee, 1998; Ganatsas et al., 2012). In most cases, these studies have focused on fire-adapted tree species. However, not all species distributed in the Basin have developed fire adaptation mechanisms (Arnan et al., 2007). Coniferous species such as *Pinus nigra*, *P. sylvestris*, *Abies cephalonica*, and *Abies borisii-regis* have recently experienced intense forest fires (Retana et al., 2002). In Türkiye, there is a large body of literature on the post-fire vegetation dynamics of Turkish red pine (*P. brutia*), including studies on natural regeneration, seed bank dynamics, species diversity, and successional patterns (Boydak, 2004; Kavgacı & Tavşanoğlu, 2010; Tavşanoğlu & Gürkan, 2014; Tavşanoğlu, 2008a; Tavşanoğlu, 2008b; Kavgacı et al., 2010; Tavşanoğlu & Gürkan, 2009). These studies have provided valuable insights into the resilience of *P. brutia* forests under varying fire regimes. However, studies focusing specifically on black pine (*P. nigra*) are much more recent and still limited (Keleş and Kavgacı, 2025).

Post-fire vegetation studies are generally conducted to reveal changes in the fire regime, the effects of these changes, and how plant species adapt to them. This study gains importance due to the recent high risk of forest fires in the Western Black Sea Region and the lack of research on post-fire vegetation development in black pine forests, which have the second-largest distribution area after red pine forests in Türkiye. Black pine forests have wide natural and artificial distributions in both Türkiye and most parts of Europe (Mikulová et al., 2019; Čahojová et al., 2024).

In recent years, the need to maintain and conserve post-fire ecosystem services has necessitated the consideration of new approaches to post-fire regeneration and fire management (Roces-Díaz et al., 2020). The seeds at the surface in *P. nigra* does not provide a viable seed source for post-fire regeneration (Habrouk et al., 1999; Ordóñez et al., 2005; Martín-Alcón and Coll, 2016; Rodman et al., 2023), and it has been attributed to the inability of black pine seeds to tolerate high fire temperatures and their very limited dispersal distance (Touchan et al., 2012).

Since black pine sheds its seeds in spring, the seed bank is almost completely empty during spring and summer fires; however, autumn fires may coincide with newly shed seeds, which could slightly increase regeneration potential. Post-fire regeneration therefore relies largely on the survival of rare or non-existent local seed sources, as well as seeds dispersed from distant unburned areas or from sparse surviving individuals (Trabaud and Campant, 1991; Retana et al., 2002; Pausas et al., 2008). Nevertheless, seed distribution in the burned area is often insufficient to ensure recovery (Tavşanoğlu, 2008b; Retana et al., 2012). Studies have

reported that post-fire regeneration is nearly zero in black pine areas that have experienced high-intensity fires and complete canopy consumption (Christopoulou et al., 2014). Even though the main factor determining fire resistance is bark thickness (Tapias et al., 2004), individual survival after fire is also influenced by tree age, canopy height, self-pruning and root damage (Retana et al., 2002; Ordóñez et al., 2006; Fernandes et al., 2012; Keleş and Kavgacı, 2025).

P. nigra can be characterized as a fire-resistant species if the fire return interval exceeds 30 years (Leys et al., 2014). However, the high intensity and increasing frequency of crown fires in black pine forests across Europe- driven by global climate change- may threaten the species' survival and compromise planned conservation efforts (Ordóñez et al., 2005; Fernandes, et al., 2008; Kakouros and Chrysopolitou, 2010; Christopoulou et al., 2013; Moriondo et al., 2006; Giannakopoulos et al., 2009; Fyllas and Troumbis, 2009).

Given the critical role of seed availability, fire intensity, and fire frequency in shaping post-fire regeneration of *P. nigra* forests, there is a need for detailed assessments of vegetation responses under different fire regimes. Understanding these dynamics is crucial for developing effective conservation and management strategies in the face of increasing fire intensity and frequency driven by climate change. Therefore, the objectives of this study are:

- to determine the vegetation dynamics after fire in *P. nigra* forests,
- to assess the effects of different fire types on plant species diversity,
- to contribute to the regional literature and provide a basis for future studies on this species,
- and to evaluate the effect of fire on vegetation development over time.

2. Materials and Methods

2.1. Study area

The study area is located in the Western Black Sea Region of Türkiye, within the boundaries of the Bayam Forest District, which is affiliated with the Taşköprü Forest Enterprise Directorate under the Kastamonu Regional Directorate of Forestry (Figure 1). The area is situated approximately between 41°30′N and 34°00′E and lies within an elevation range of 800–1200 m. The region has a transitional climate between the Black Sea climate and continental influences; the annual mean temperature is 8–10 °C, and the mean annual precipitation is 600–800 mm, with rainfall mostly concentrated in spring and autumn (MGM, 2023). The dominant soil type is brown forest soils, with partially non-calcareous brown forest soils; the soils are moderately deep, well-drained, and have a sandy-loam texture (OGM, 2017).

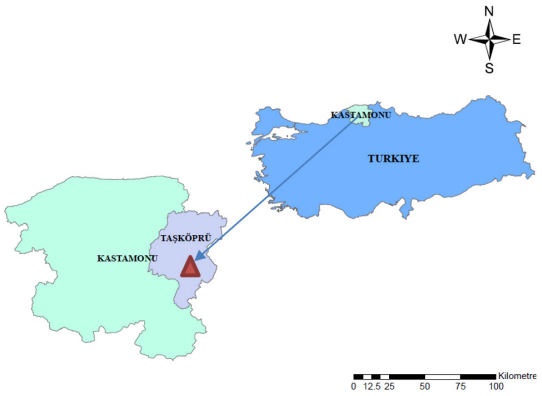

Figure 1. The study area
Şekil 1. Çalışma alanı

2.2. Study area description

The first sample plot, which was used to examine post-fire vegetation development, is located in the area affected by a 2012 surface fire (SF). This area has a slope of 45% and consists of a *P. nigra* stand at the pole timber stage with 70% canopy closure. The 2nd plot, affected by a crown fire (CF) in 2012, has a 25% slope and consists of a young and sapling-stage *P. nigra* stand with full canopy closure (71-100%). After the fire, the area was cleared and partially replanted by *P. nigra*. The current canopy closure is about 2-3%. The bedrock is metamorphic, and the soil types are sandy loam and sandy clay loam. The 3rd plot, affected by a crown fire in 2003, has a 30% slope and consists of a *P. nigra* stand at the sapling stage with moderate canopy closure (41-70%). Following the plantation after the fire, the current canopy closure is approximately 55%.

Adjacent to the burned areas is an unburned (control) plot dominated by a fully closed *P. nigra* stand in the pole timber stage. The slope in the control area is 20%. The stand has 1032 trees per ha, with average tree height ranging between 10 m and 12 m. The litter layer consists of needles, cones, bark, and branch fragments, with a thickness of 2-3 cm.

2.3. Identification of the sample areas

Sample plots were established in 2016 and 2017 using the minimum area method in areas affected by the 2003 and 2012 forest fires. Concept of the "minimum area" is important because it characteri-

zes the plant community-i.e., it includes all the taxa that make up the community. In vegetation studies, the appropriate sample plot size varies depending on the habitat type. According to the minimum area method, the suitable plot size for *P. nigra* is 1000 m² (Akman and Ketenoğlu, 1987). However, since there is only a single sampling point for each study area, the sampling area has been expanded to represent the entire fire and control zones, and to a level where no additional species are expected to be found.

To determine the temporal development of post-fire vegetation, to evaluate the effects of different fire types on vegetation recovery, and to compare post-fire vegetation dynamics in black pine forests, which are of great ecological importance in Türkiye, sample plots (Table 1) were selected from areas affected by both crown and surface fires.

Table 1. Fires' and plots'characteristics
Tablo 1. Yangınların ve deneme sahalarının özellikleri

Locality	Soil characteristics	Slope	Fire time	Fire type	Burned area (ha)	Time since last fire (years)	Fire intensity
Taskopru	Sand- Clay Loam	%30	August 2003	Crown fire (CF)	141.5	14	High
Taskopru	Sand- Clay Loam	%45	August 2012	Surface fire (SF)	30	5	Medium-low
Taskopru	Sand- Clay Loam	%25	August 2012	Crown fire (CF)	70	5	High
Taskopru	Sand- Clay Loam	%20	Control				

Accordingly, four sampling plots were established: one from the 2003 crown fire site, two from the 2012 fire site (one surface fire, one crown fire), and one from an unburned control area. Within each plot, all plant species were recorded and representative specimens were collected during the 2016 spring and 2017 summer vegetation seasons. Vegetation sampling was conducted using the Braun–Blanquet cover-abundance method, and for each plot, the cover-abundance values of all species were estimated according to the Braun–Blanquet scale (+, 1, 2, 3, 4, 5). These data were subsequently used to calculate species richness and Shannon–Wiener diversity indices. Particular attention was paid to collecting diagnostic plant parts such as cones, fruits, and flowers to ensure accurate identification. The collected specimens were pressed and preserved in the Bilgehan Bilgili Herbarium of the Faculty of Forestry at Kastamonu University for future reference. The life forms of the identified species were determined by examining the vegetative structures of the plants in the field as well as the samples brought to the laboratory. The life form classification used in the study is based on Raunkiaer's system, which classifies plants according to the position of renewal buds or apical shoots during the unfavorable (critical) season (Akman and Ketenoğlu, 1987).

Plant identification was carried out using the *Flora of Turkey*, and reference specimens from the Herbarium. The life spans and flowering periods of the identified species were determined using the Turkish Plants Data Service database (TUBIVES, 2017).

2.4. Statistical analysis

The species collected from these plots were classified according to Raunkiaer's life form system. Similarly, the collected species were grouped according to their families, and their periodic changes were analyzed based on the degree of family overlap. A Chi-square test of independence was conducted to determine whether there was a significant relationship between life forms and the periodic changes in plant families.

Since the data used for statistical analysis were categorical and ordinal, the nonparametric Chi-square test was deemed appropriate. The Chi-square test of independence was chosen as the most suitable method to assess whether there were differences among the data obtained from the sample plots. SPSS 19 (2010) was used to perform the analysis. The Chi-square test was also applied to identify which plant family showed the most significant variation. The family with the highest Chi-square value was considered to exhibit the greatest difference. Total Chi-square values (Formula 1) representing the degree of family overlap in each sample plot were calculated.

$$\boldsymbol{x^2} = \sum \frac{(\boldsymbol{O_i} - \boldsymbol{E_i})^2}{\boldsymbol{E_i}} \qquad (1)$$

Where O_i = Observed value, Ei = Expected value, X^2= Chi-square value, and ∑= Summation.

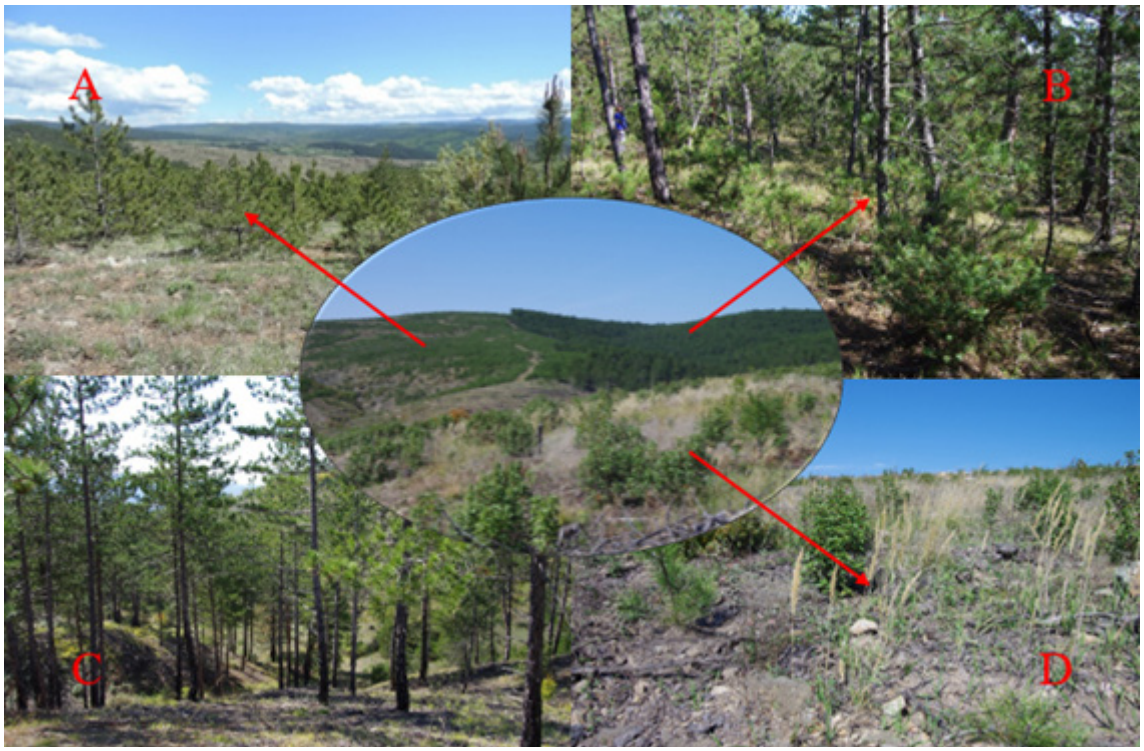

Figure 2. The sample areas: A) 2003 crown fire, B) Control, C) 2012 surface fire, D) 2012 crown fire (Photos by F.N. Yılmaz-Ö. Küçük)
Şekil 2. Örnek alanlar: A) 2003 Tepe yangını, B) Kontrol, C) 2012 Örtü yangını, D) 2012 Tepe yangını

The Shannon-Weiner diversity index was used to quantify species diversity as a numerical value, while the Beta diversity index was employed to statistically compare the degree of diversity among sample areas affected by fire in different years. Cluster analysis was also performed to group species based on their presence in the sample areas. Cluster analysis is used to group *N* individuals with *p* characteristics (variables) into distinct clusters with a homogeneous structure based on their similarities (Ingram and Margetis, 2010). The abundance-dominance value for each species used in the index was determined according to the DOMIN-KRAJINA scale. Diversity indices and cluster analysis were calculated using the PAST X.3.

3. Results

In the field survey conducted in the area affected by the surface fire (2012), wheatgrasses (Poaceae) were observed as the dominant family. Poaceae family members were found in the densely vegetated areas, while Cistus creticus and Rubus sp. were observed in zones where canopy closure was slightly disrupted. It was determined that some P.

nigra individuals in the sample area affected by the surface fire had dried up and died due to high temperatures, and were subsequently removed. The gaps formed by these losses are now densely covered by C. creticus.

In areas where black pine individuals managed to survive, canopy closure was measured at around 40-45%. During the field survey, small black pine saplings were found heterogeneously distributed throughout the area. This suggests that the site is capable of natural regeneration under normal conditions, albeit with difficulty.

When species were ranked according to their degree of coverage, Cistus laurifolius was the most dominant, followed by Rubus sp. and members of the Poaceae family. In addition to these dominant species, many associated species were also identified, although specific coverage rates were not provided. Some of these species include: Muscari sp., Salvia sp., Pilosella hoppeana, Viola sp., and Polygala anatolica. A small amount of dead litter was heterogeneously distributed throughout the burned area. For the same reason, humus was not measured quantitatively; however, field observations indicated that it was very limited. The fallen needles were also observed to have only recently begun forming a layer on the litter.

As observed during our fieldwork, the species diversity of the crown fire site (2012) is similar to that of the surface fire sample area, particularly in zones with little or no canopy closure. Although the species composition is nearly the same, there are differences in physical characteristics and coverage levels. For example, *Cistus* species are more dominant and taller (reaching 40-60 cm) in the crown fire area compared to the surface fire area.

Following the crown fire in 2003, plantation was carried out immediately. The canopy closure of the plantation area is approximately 50-55%. In addition to *P. nigra*, *Populus tremula*, *Juniperus macrocarpa,* and *C. laurifolius* were found to occupy the site. In the control area, due to a relatively dense canopy (~70%) and an insufficient and uneven light regime, the height and coverage of *Cistus* species are not as uniform as in the other sample plots. However, in areas where the light regime improves, especially along the stand edges, an increase in species diversity was observed.

To evaluate compositional patterns, we performed a floristic similarity analysis on the plot × species (presence–absence) matrix (n = 4 plots).

Hierarchical clustering based on Bray–Curtis dissimilarity (UPGMA) separated the plots into two main groups (Figure. 4); in the dendrogram, each leaf represents a plot, and the colors denote the k = 2 clusters. At the k = 2 cut, the plots are divided into two main groups: Group 1: 2012 crown fire and 2012 surface fire plots; these show high similarity and represent early successional communities shortly after fire disturbance. Group 2: 2003 crown fire and control plots; these are closer to each other, indicating more mature communities with developed canopy closure. These results confirm that both fire intensity and time since fire are the main drivers of floristic differentiation.

Complementary to this floristic analysis, we summarize the Raunkiaer life-form co-occurrence by plot in Table 2. In 2012, the high dominance of Phanerophytes in all areas except for the crown fire site is attributed to the prevalence of *P. nigra*, the dominant species in those regions. However, in the 2012 crown fire area, the high Phanerophyte cover is mainly due to the presence of numerous herbaceous species classified under this life form.

According to the Chi-square test of independence, a statistically significant difference was found ($\chi^2 = 50.495$; df = 12; $p = 0.000$; $p < 0.05$). This indicates that the degree of overlap among life forms significantly varies depending on the year and the fire regime. Chi-square values obtained from the sample areas were analyzed, and it was determined that the life form showing the greatest variation based on fire type and year was the Cryptophyte ($\chi^2 = 24.83$), referring to plants with regeneration buds or shoot apices located in the soil or water layer. Cryptophytes play a crucial role in the successional process, as their ability to rapidly recolonize gaps after fire allows them to dominate in the early stages of post-fire recovery. This finding is consistent with the observation that the highest cover values for cryptophytes were recorded in the 2003 crown fire site, suggesting that this group becomes increasingly established over time. In contrast, the life form exhibiting the least variation was the Chamaephyte ($\chi^2 = 0.61$). These plants possess renewal buds or shoot apices located up to approximately 40 cm above the soil surface, making them relatively less affected by the fire regime. Their distribution remained relatively stable across all sample areas. This indicates that chamaephytes have a more fire-resistant survival strategy, maintaining their presence regardless of fire intensity or time since fire. These results highlight that, in post-fire management strategies, cryptophytes may serve as indicators of ecosystem recovery, whereas chamaephytes represent a key component of ecosystem continuity.

Table 2. The distribution of life forms
Tablo 2. Yaşam formlarının dağılımı

Raunkiaer's life forms	Sample Area			
	2012 SFA	2012 CFA	2003 CFA	Control
Phanerophyte	6	9	12	11
Chamaephyte	4	5	4	3
Cryptophyte	4	4	15	12
Hemicryptophyte	16	21	24	22
Geophyte	7	8	8	8
Therophyte	4	25	17	2

SFA: surface fire area, CFA: crown fire area

As part of the post-fire species regeneration study, a total of 117 species belonging to 37 families were identified. In the fifth year following the fire, the families Cistaceae, Fagaceae, and Rosaceae were the most prominent in the surface fire area. In the crown fire area (fifth year), Cistaceae ranked first,

followed by Rosaceae. In the fifteenth year, the dominant families in the crown fire area were Cupressaceae, Salicaceae, and Cistaceae, respectively.

The results of the Chi-square test of independence showed that the degree of overlap among families varied significantly depending on fire type and time ($\chi^2 = 176.385$; df = 21; $p < 0.05$). According

to the analysis, the Pinaceae family exhibited the highest difference ($\chi^2 = 59.49$). This finding reflects the sensitivity of black pine (*Pinus nigra*) to fire type and time since fire, as well as the influence of post-fire canopy development and reforestation practices. In particular, the reforestation carried out in the 2003 crown fire site increased stand closure, which in turn raised the cover value of Pinaceae. In contrast, the Cupressaceae family showed the lowest difference ($\chi^2 = 4.70$), indicating that species such as *Juniperus macrocarpa* were relatively less affected by fire type and time, maintaining similar abundance across different sites. This stability can be attributed to the high fire tolerance of Cupressaceae species and their ability to regenerate both from seed and vegetatively. These results suggest that, in post-fire ecosystem management, the regeneration dynamics of Pinaceae should be carefully considered, while Cupressaceae can serve as an indicator of ecosystem continuity.

According to the results of the Shannon-Wiener diversity index, the highest species diversity was observed in the 2003 crown fire area (H'= 4.223), while the lowest was recorded in the 2012 ground (cover) fire area (H'= 3.554). The 2012 crown fire area had a diversity index value of H'= 4.133, and the control site had H'= 3.877.

Pairwise beta dissimilarities were calculated using the Bray–Curtis and Jaccard indices. The highest similarity was obtained between sites 2012 SFA and 2003 CFA (Bray–Curtis = 0.333; Jaccard = 0.532). The greatest dissimilarity was observed between sites 2012 SFA and CONTROL according to Bray–Curtis (0.538) and between sites 2012 SFA and 2012 CFA according to Jaccard (0.709). These findings indicate that species composition differs markedly among the sites, although sites 2012 SFA and 2003 CFA share a relatively more similar species composition.

Table 3a. Species recorded in the sample plots and their presence
Tablo 3a. Örnek Alanlarda bulunan türler ve bulunma durumları

			Cover: (%); (+): Present; (.) Absent; (Pl): Plantation			
Species	Familys	Life forms	Control	SF. 2012	CF. 2003	SF. 2012
Pinus nigra	Pinaceae	Phanerophyte	65	45	50 (Pl)	2 (Pl)
Crataegus monogyna	Rosaceae	Phanerophyte	+	.	+	.
Juniperus macrocarpa	Cupressaceae	Phanerophyte	5	.	10	5
Pyrus elaeagnifolia	Rosaceae	Phanerophyte	+	.	+	.
Populus tremula	Salicaceae	Phanerophyte	.	5	15	15
Quercus pubescens	Fagaceae	Phanerophyte	10	5	+	+
Cornus mas	Cornaceae	Phanerophyte	+	+	+	+
Genista lydia subsp. *Lydia*	Fabaceae	Phanerophyte	+	+	+	+
Pyracantha coccinea	Rosaceae	Phanerophyte	+	.	+	.
Rubus hirtus	Rosaceae	Phanerophyte	+	10	+	25
Rosa canina	Rosaceae	Phanerophyte	+	.	+	+
Ligustrum vulgare	Oleaceae	Phanerophyte	+	.	+	+
Chamaecytisus pygmaeus	Fabaceae	Chamaephyte	+	+	+	+
Berberis crataegina	Berberidaceae	Chamaephyte	+	.	+	+
Cotoneaster nummularius	Rosaceae	Chamaephyte	+	.	.	+
Galium verum	Rubiaceae	Chamaephyte	.	+	.	+
Globularia trichosantha	Globulariaceae	Chamaephyte	.	+	.	+
Sabulina juniperina (syn. *Minuartia juniperina*)	Caryophyllaceae	Chamaephyte	.	+	+	.
Pilosella hoppeana	Asteraceae	Chamaephyte	.	+	+	.
Dactylis glomerata	Poaceae	Cryptophyte	+	.	+	.
Daphne pontica	Thymelaeaceae	Cryptophyte	+	.	.	+
Cistus laurifolius	Cistaceae	Cryptophyte	5	20	10	35
Cistus creticus	Cistaceae	Cryptophyte	+	5	+	+
Coronilla varia subsp. *varia*	Fabaceae	Cryptophyte	.	.	+	.
Tussilago farfara	Asteraceae	Cryptophyte	+	.	+	.
Teucrium chamaedrys	Lamiaceae	Cryptophyte	.	.	+	.

Table 3b. Species recorded in the sample plots and their presence
Tablo 3b. Örnek Alanlarda bulunan türler ve bulunma durumları

Briza media	Poaceae	Cryptophyte	+	+	+	+
Galium albüm	Rubiaceae	Cryptophyte	+	.	+	.
Galium rotundifolium	Rubiaceae	Cryptophyte	+	.	+	.
Pentanema ensifolium (syn. *Inula ensifolia*)	Asteraceae	Cryptophyte	.	.	+	.
Poa bulbosa	Poaceae	Cryptophyte	+	+	+	.
Trifolium medium	Fabaceae	Cryptophyte	.	.	+	.
Trifolium pannonicum	Fabaceae	Cryptophyte	+	.	+	.
Veronica chamaedrys	Scrophulariaceae	Cryptophyte	+	.	+	.
Veronica orientalis	Scrophulariaceae	Cryptophyte	+	.	+	.
Thymus longicaulis	Lamiaceae	Hemicryptophyte	.	.	.	+
Filipendula vulgaris	Rosaceae	Hemicryptophyte	.	+	.	+
Euphorbia amygdaloides	Euphorbiaceae	Hemicryptophyte	.	.	.	+
Euphorbia nicaeensis	Euphorbiaceae	Hemicryptophyte	.	.	.	+
Euphorbia seguieriana	Euphorbiaceae	Hemicryptophyte	+	.	.	+
Euphorbia myrsinites	Euphorbiaceae	Hemicryptophyte	+	.	+	+
Helianthemum nummularium	Cistaceae	Hemicryptophyte	.	.	+	+
Hypericum perforatum	Guttiferae	Hemicryptophyte	.	+	+	+
Hypericum montbretii	Guttiferae	Hemicryptophyte	.	+	+	+
Potentilla recta	Rosaceae	Hemicryptophyte	+	.	+	.
Fragaria vesca	Rosaceae	Hemicryptophyte	+	.	+	.
Alyssum sp.	Brassicaceae	Hemicryptophyte	.	+	.	+
Anchusa leptophylla	Boraginaceae	Hemicryptophyte	.	+	.	.
Anthyllis vulneraria	Fabaceae	Hemicryptophyte	.	.	+	.
Brachypodium sylvaticum	Poaceae	Hemicryptophyte	+	.	+	+
Bunium microcarpum	Apiaceae	Hemicryptophyte	.	.	.	+
Campanula lyrata	Campanulaceae	Hemicryptophyte	+	.	.	.
Carex actuca	Cyperaceae	Hemicryptophyte	.	.	.	+
*Centaurea triumfetti*i	Asteraceae	Hemicryptophyte	.	.	.	+
Clinopodium vulgare	Lamiaceae	Hemicryptophyte	+	.	.	.
Cruciata taurica	Rubiaceae	Hemicryptophyte	.	+	.	+
Lotus graecus (syn. *Dorycnium graecum*)	Fabaceae	Hemicryptophyte	+	+	.	.
Echium vulgare	Boraginaceae	Hemicryptophyte	+	+	+	+
Epilobium parviflorum	Onagraceae	Hemicryptophyte	+	.	.	.
Koeleria macrantha	Poaceae	Hemicryptophyte	.	+	+	.
Lathyrus laxiflorus	Fabaceae	Hemicryptophyte	+	.	+	.
Leontodon hispidus	Asteraceae	Hemicryptophyte	+	.	+	.
Luzula forsteri	Juncaceae	Hemicryptophyte	+	+	.	.
Myosotis sylvatica	Boraginaceae	Hemicryptophyte	+	.	+	.
Pimpinella tragium	Apiaceae	Hemicryptophyte	.	.	.	+
Poa nemoralis	Poaceae	Hemicryptophyte	.	+	+	.
Polygala anatolica	Polygalaceae	Hemicryptophyte	.	+	+	+
Polygala pruinosa	Polygalaceae	Hemicryptophyte	.	.	+	+
Polygala supina	Polygalaceae	Hemicryptophyte	.	.	+	.
Primula vulgaris	Primulaceae	Hemicryptophyte	+	.	+	+
Prunella sp.	Lamiaceae	Hemicryptophyte	.	.	+	.
Prunella vulgaris	Lamiaceae	Hemicryptophyte	+	.	+	.
Salvia sclarea	Lamiaceae	Hemicryptophyte	.	+	.	.
Salvia tomentosa	Lamiaceae	Hemicryptophyte	+	+	.	.
Salvia verticillata	Lamiaceae	Hemicryptophyte	+	.	.	.
Sanguisorba minor	Rosaceae	Hemicryptophyte	+	.	+	.
Silene latifolia (syn. *Silene alba*)	Caryophyllaceae	Hemicryptophyte	.	+	+	+

Table 3c. Species recorded in the sample plots and their presence
Tablo 3c. Örnek Alanlarda bulunan türler ve bulunma durumları

Atocion compactum (syn. *Silene compacta*)	Caryophyllaceae	Hemicryptophyte	.	.	+	.
Silene italica	Caryophyllaceae	Hemicryptophyte	+	.	.	.
Vicia cracca	Fabaceae	Hemicryptophyte	.	.	+	.
Viola sieheana	Violaceae	Hemicryptophyte	+	+	.	.
Digitalis ferruginea	Scrophulariaceae	Hemicryptophyte	+	+	.	.
Orchis mascula	Orchidaceae	Geophyte	+	.	+	+
Cephalanthera rubra	Orchidaceae	Geophyte	+	+	+	+
Scilla bifolia	Liliaceae	Geophyte	+	+	+	+
Crocus ancyrensis	Iridaceae	Geophyte	+	+	+	+
Cynoglossum montanum	Boraginaceae	Therophyte	.	+	+	+
Ornithogalum umbellatum	Liliaceae	Geophyte	+	+	+	+
Gagea villosa	Liliaceae	Geophyte	+	+	+	+
Muscari neglectum	Liliaceae	Geophyte	+	+	+	+
Fumaria officinalis	Papaveraceae	Geophyte	+	+	+	+
Bromus japonicus	Poaceae	Therophyte	.	+	.	+
Euphorbia stricta	Euphorbiaceae	Therophyte	.	.	+	+
Lactuca serriola	Asteraceae	Therophyte	.	.	+	+
Lamium purpureum	Lamiaceae	Therophyte	.	.	+	+
Lapsana communis	Asteraceae	Therophyte	.	.	+	+
Linum usitatissimum	Linaceae	Therophyte	.	+	+	+
Rhinanthus major	Scrophulariaceae	Therophyte	.	.	+	+
Mutarda arvensis (syn. *Sinapis arvensis*)	Brassicaceae	Therophyte	.	.	.	+
Noccaea perfoliata (syn. *Thlaspi perfoliatum*)	Brassicaceae	Therophyte	.	.	.	+
Tragopogon coloratus	Asteraceae	Therophyte	.	.	.	+
Senecio vernalis	Asteraceae	Therophyte	.	+	+	+
Euphorbia falcata	Euphorbiaceae	Therophyte	+	.	+	+
Geranium robertianum	Geraniaceae	Therophyte	+	.	.	+
Erigeron canadensis (syn. *Conyza canadensis*)	Asteraceae	Therophyte	.	+	+	+
Crepis foetida	Asteraceae	Therophyte	.	.	+	+
Stellaria media	Caryophyllaceae	Therophyte	.	.	+	+
Holosteum umbellatum	Caryophyllaceae	Therophyte	.	.	.	+
Chenopodium albüm	Chenopodiaceae	Therophyte	.	.	.	+
Erodium cicutarium	Geraniaceae	Therophyte	.	.	+	+
Scandix pecten-veneris	Apiaceae	Therophyte	.	.	+	+
Draba verna (syn. *Erophila verna*)	Brassicaceae	Therophyte	.	.	+	+
Orlaya daucoides	Apiaceae	Therophyte	.	.	.	+
Centaurium erythraea	Gentianaceae	Therophyte	.	.	+	+
Aethionema arabicum	Brassicaceae	Therophyte	.	.	.	+
Lysimachia arvensis (syn. *Anagallis arvensis*)	Primulaceae	Therophyte	.	.	+	+
Bellis sylvestris	Asteraceae	Therophyte	.	.	.	+
Total			58	41	80	72

4. Discussion and Conclusion

Studies on the post-fire regeneration of black pine and the identification of substrate species have shown that the species can survive following low-intensity fires, whereas regeneration does not occur after high-intensity fires (Keleş and Kavgacı, 2025; Mendez-Cartin et al., 2025; Retana et al., 2002; Ordonez and Retana, 2004; Ordonez et al., 2005; 2006; Christopoulou et al., 2014). Similar findings supporting previous studies were observed at the end of the fifth year in the present study. Specifically, no black pine individuals originating from seed were found in the areas affected by

crown fires, whereas seed-originated individuals were identified in the areas affected by ground fires. Therefore, in regions where black pine stands have been exposed to crown fires, plantation emerges as a necessary and prioritized strategy.

In the study of Radanova (2014) on black pine stands affected by fires in different years, it was observed that the Asteraceae and Lamiaceae families were dominant during the 3-5 years following the fire, while in the subsequent period (5-10 years), the Rosaceae and Liliaceae, along with Lamiaceae, became dominant. Furthermore, Hemicryptophytes were initially the dominant life form, followed by Phanerophytes. Although the dominant life form remained unchanged in the later years, its relative proportions shifted. Hemicryptophytes decreased while Phanerophytes increased. In the present study conducted, succession analysis covered the period between 4 and 15 years after the fire. Phanerophytes were found to be the dominant life form throughout the first 15 years post-fire, which aligns with the findings of Radanova (2014). In 2012, the high proportion of Phanerophytes observed in all sample areas except the site affected by a crown fire was attributed to the presence of black pine. In contrast, the abundance of herbaceous Phanerophytes in the crown fire site was linked to the proliferation of such species in that area.

When post-fire vegetation development was evaluated at the family level over time, Cistaceae was found to be the dominant family between 4 and 10 years post-fire, often accompanied by Rosaceae. Although individuals from the Liliaceae and Lamiaceae families were present in considerable numbers, they were not considered dominant due to their low degree of overlap. While the results of Radanova (2014) were consistent in terms of life form dominance, there were differences in dominant families. In this study, Cistaceae dominated during the 4-10-year post-fire period, whereas Rosaceae, Liliaceae, and Lamiaceae were more dominant in Radanova's findings. The dominance of Cistaceae in the current study may be attributed to the favorable growing conditions present in the study area.

Núñez et al. (2008), in a study conducted in a Scots pine (*Pinus sylvestris*) stand, reported that species capable of regenerating through vegetative shoots were the first to colonize post-fire areas. Similarly, their findings indicated that perennial herbaceous species were the earliest to reestablish, whereas annual herbaceous species exhibited a slower regeneration rate. This pattern was attributed to the ability of perennial species to survive fire events both through their seeds and resilient underground structures, highlighting an adaptive regeneration strategy in response to fire disturbance. In our study, species such as *Quercus* sp., *Cistus sp*., and *Rubus sp*., were also identified in both surface- and crown-fire-affected sites, exhibiting similar resprouting and early colonization strategies. Furthermore, several studies conducted in Türkiye, particularly in *Pinus brutia* forests, have reported comparable post-fire regeneration patterns, emphasizing the dominance of resprouting species and early establishment of shrub taxa (Tavşanoğlu & Gürkan, 2009; Kavgacı et al., 2010). Our results also partially correspond to the early successional stages described by Arianoutsou and Ne'eman (2000) for *P. halepensis* forests, particularly in terms of the early dominance of grasses and shrubs. Together, these findings support that shrub and resprouting species play a key role in the early stages of post-fire succession in black pine forests.

According to the results of the Shannon diversity index, the highest species diversity was observed in the area affected by the 2003 crown fire. This high diversity is attributed to the recolonization of herbaceous species in the gaps formed by the decline of *Cistus* spp. Similar findings were reported by Arianoutsou and Ne'eman (2000). In the current study, conducted 15 years after the fire (corresponding to Stage 4), the Cistaceae family was found to have the lowest representation in the 2003 crown fire area among the three fire-affected sample sites. The rarity of this family in both the 15-year post-fire area and the control site, as well as its decreasing trend over time following fire disturbance, was attributed to canopy closure. Comparable patterns have been documented in the literature (Arianoutsou and Ne'eman, 2000). Furthermore, among the three fire-affected sample areas, the Cupressaceae family exhibited the highest cover abundance in the 2003 crown fire area.

As a result of the Chi-square test of independence, the greatest variation observed in the Pinaceae family is attributed to the fact that the plantation within the 2012 crown fire sample area has not yet formed a closed canopy. Once black pine individuals establish a denser canopy, the observed variation is expected to diminish due to reduced light availability in the understory. Conversely, since the sample areas had relatively similar numbers of individuals belonging to the Cupressaceae family, this family exhibited the least difference in the Chi-square test results.

Additionally, 39 species recorded in the sample area affected by the 2012 crown fire were not detected in the sample area subjected to a surface (cover) fire. This difference is explained by the

loss of ground vegetation in the crown fire area, in contrast to the cover fire area, where many trees survived. The continued presence of tree cover in the latter likely inhibited the establishment of new species due to canopy closure. Thus, the type of fire appears to influence species diversity. In crown fire areas, the absence of light competition facilitates colonization by new species, whereas in areas affected by cover fire, such opportunities are more limited.

Moreover, fire intensity also affects the degree of cover exhibited by plant families and life forms. In other words, within the scope of this study, it can be concluded that fire type has a significant influence on the abundance and persistence of both plant species and life forms. This is likely due to the breakdown of canopy closure as fire intensity increases.

A total of nine species were found exclusively in the fire-affected areas but were absent in the control area. This pattern reflects the natural process of ecological succession, wherein some species colonize immediately after fire and later disappear from the area. Conversely, species found in the control area but not in the fire-affected areas may serve as indicators of late-successional species expected to recolonize over time. In addition, 17 species were found to be common across all sites. These species are considered to be fire-resilient, possessing regeneration strategies that enable them to persist through all successional stages and function as primary species within the regeneration cycle.

During the vegetation survey, it was observed that in black pine stands affected by crown fire, no young individuals were regenerating from seed. However, in the sample area subjected to cover fire, seed-originated regeneration was evident. These findings are consistent with those reported by Christopoulou et al. (2014). Furthermore, the black pine stands established in the crown fire area exhibit a variable floristic composition, shaped by the post-fire arrival of new species and the persistence of pre-existing ones.

In conclusion, the findings obtained in this study confirm the ecological processes predicted both by Keleş and Kavgacı (2025) and by other literature. Black pine can demonstrate resilience to surface fires; however, its natural regeneration fails following crown fires. Therefore, the sustainability of black pine forests is directly linked to the management of the fire regime. It is of great importance to keep the fire regime under control in these forests, ensuring that fires remain at the surface fire level as much as possible, while preventive silvicultural measures are implemented to avoid crown fires. In this context, dense black pine stands should be supported with thinning and silvicultural treatments to encourage optimal cone and seed production. In areas affected by crown fires, where natural regeneration of black pine does not occur or remains extremely limited, plantation and ecological restoration practices must be prioritized. Furthermore, long-term and periodic monitoring is necessary to better understand post-fire successional processes in black pine forests and to improve management strategies. Overall, when field findings and the literature are evaluated together, it is concluded that the future persistence of black pine forests can only be ensured through effective fire management, silvicultural practices, and restoration strategies.

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Authors' contributions

Conceptualization and study design: Ö. KÜÇÜK; Data collection and processing: Ö. KÜÇÜK, K. GÜNEY, and F.N. YILMAZ; Data analysis and interpretation: Ö. KÜÇÜK, B. SAĞLAM, K. GÜNEY, and F.N. YILMAZ; Literature review: Ö. KÜÇÜK and F.N. YILMAZ; Writing-original draft: Ö. KÜÇÜK and F.N. YILMAZ; Writing-review and editing: Ö. KÜÇÜK, B. SAĞLAM, K. GÜNEY, and F.N. YILMAZ; Supervision: Ö. KÜÇÜK.

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