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POST-FIRE DYNAMICS OF GROWTH AND STRUCTURE IN MIXED CONIFER FORESTS OF NORTHERN MEXICO

DINÁMICA POST-INCENDIO DEL CRECIMIENTO Y ESTRUCTURA EN BOSQUES MEZCLADOS DE CONÍFERAS DEL NORTE DE MÉXICO

THESIS

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to obtain the degree of:

DOCTOR EN CIENCIAS EN RECURSOS NATURALES Y MEDIO AMBIENTE EN ZONAS ÁRIDAS

By:

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DEDICATION

A Dios, por permitirme culminar otra etapa de mi vida.

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BIOGRAPHICAL DATA

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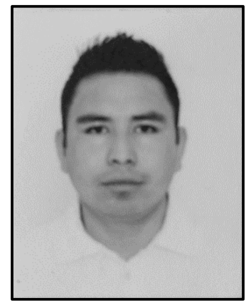

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GENERAL SUMMARY

POST-FIRE DYNAMICS OF GROWTH AND STRUCTURE IN MIXED CONIFER FORESTS OF NORTHERN MEXICO

Fire is one of the most important disturbances for fire-dependent ecosystems and drives the patterns of vegetation structure and composition. The objective of this study was to characterize the structural dynamics of a mixed conifer forests of the Sierra Madre Occidental (SMO) affected by fires. Vegetation was sampled at three fire severity levels: high, moderate, and low, and the unburned level was included as a control. Species richness (*S*), Shannon index (*H*), and the Evenness index (*E*) were calculated. The diameter-size class and height class of the trees were determined by applying the indices *H*, and *E* indexes and the coefficient of variation (*CV*). Differences in the indices calculated across fire severity levels were determined by analysis of variance (ANOVA) and Tukey's multiple comparison tests. Results showed no significant differences ($p \leq 0.05$) in species diversity indices among fire severity levels; however, diameter and height classes were lower in areas affected by high-severity fire. Increment cores and cross-sections of fire-scarred trees were also collected and analyzed using dendrochronological techniques to generate tree ages and descriptive statistics involved in fire history studies. Synchrony between fire history and tree establishment was determined, and climatic data values were correlated with the number of trees established per year. Forty-one fire events were reconstructed over the period 1855-2019. Overall, the mean fire interval (MFI) was 2.28 years and 12.17 years for large fires. The number of trees established per year was influenced by the prevailing dry conditions in September and October of the previous year and the wet conditions that occurred in December of the same year. These results constitute an example of forest response to fire severity and its historical behavior and may support further studies related to the influence of fire on other forest communities present in the SMO.

Keywords: fire severity, fire frequency, dendrochronology, fire scars, tree recruitment, species composition.

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RESUMEN GENERAL

DINÁMICA POST-INCENDIO DEL CRECIMIENTO Y ESTRUCTURA EN BOSQUES MEZCLADOS DE CONÍFERAS DEL NORTE DE MÉXICO

El fuego es una de las perturbaciones más importantes para los ecosistemas dependientes del fuego y determina los patrones de estructura y composición de la vegetación. El objetivo de este estudio fue caracterizar la dinámica estructural de un bosque mixto de coníferas de la Sierra Madre Occidental (SMO) afectado por incendios. La vegetación se muestreó en tres niveles de severidad del fuego: alto, moderado y bajo, y el nivel no quemado se incluyó como control. Se calculó la riqueza de especies (*S*), el índice de Shannon (*H*) y el índice de uniformidad (*E*). La clase diámetro-tamaño y la clase altura de los árboles se determinaron aplicando los índices *H*, *E* y el coeficiente de variación (CV). Las diferencias en los índices calculados a través de los niveles de severidad del fuego se determinaron mediante análisis de varianza (ANOVA) y pruebas de comparación múltiple de Tukey. Los resultados no mostraron diferencias significativas ($p \leq 0.05$) en los índices de diversidad de especies entre los niveles de severidad del fuego; sin embargo, las clases de diámetro y altura fueron menores en las áreas afectadas por incendios de alta severidad. También se recopilaron y analizaron núcleos incrementales y secciones transversales de árboles con cicatrices de incendios utilizando técnicas dendrocronológicas para generar edades de los árboles y estadísticas descriptivas involucradas en estudios de historia de incendios. Se determinó una sincronía entre el historial de incendios y el establecimiento de árboles, y los valores de los datos climáticos se correlacionaron con el número de árboles establecidos por año. Se reconstruyeron 41 eventos de incendio en el período 1855-2019. En general, el intervalo medio de incendios (MFI) fue de 2,28 años y de 12,17 años para incendios extensos. El número de árboles establecidos por año estuvo influenciado por las condiciones secas predominantes en septiembre y octubre del año anterior y las condiciones húmedas que se presentaron en diciembre del mismo año. Estos resultados constituyen un ejemplo de la respuesta del bosque a la severidad de los incendios y su comportamiento histórico y pueden apoyar estudios posteriores relacionados con la influencia del fuego en otras comunidades forestales presentes en la SMO.

Palabras clave: severidad del fuego, frecuencia de fuego, dendrocronología, cicatrices de incendios, reclutamiento de árboles, composición de especies.

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CHAPTER I: GENERAL INTRODUCTION

1.1. Introduction

Fire is an ecological process which plays a complex role in shaping many ecosystems around the world (Harper et al., 2018; Bowman et al., 2020). Therefore, the composition and structure of many ecosystems are attributed to their dominant fire regime (He et al., 2019). However, human-caused climate change and radical shifts in ecosystems have altered fire regimes, with increasingly devastating impacts on ecosystems, infrastructure, and even human health (Shuman et al., 2022).

In annual terms, vegetation fires burn an average of 400 to 500 million of hectares worldwide (Bowman et al., 2020), and based on climate change projections, those could increase (Stephens et al., 2013). In Mexico, an average of 7,077 fires occurs annually, affecting on average around 270,967 ha (Comisión Nacional Forestal [CONAFOR], 2023). The climate change projections also indicate that Mexico will become more arid with more recurrent droughts (Seager et al., 2009), which may increase the occurrence and severity of fires.

Mexico has the greatest diversity of pine species worldwide, most of which are adapted to fire (Rodríguez-Trejo, 2015). The Sierra Madre Occidental (SMO) has the greater diversity of pines, oaks, and arbutus associations in the world (González-Elizondo et al., 2012), and fire is considered to have played a determining role in this diversity (Heyerdahl and Alvarado, 2003).

The SMO is categorized as a critical forest fires region (Zúñiga-Vásquez et al., 2017; Zúñiga-Vásquez et al., 2019) and several studies have identified altered fire regimes at some sites (Fulé and Covington, 1999; Heyerdahl and Alvarado, 2003; Cerano-Paredes et al., 2019; Cerano-Paredes et al., 2022), condition that may impact future fires affecting vegetation structure and composition (Lafon et al., 2017).

In Mexico, great efforts are made to prevent, timely detect, and to suppress fires (Domínguez and Rodríguez-Trejo, 2008; Vega-Nieva et al., 2019; CONAFOR, 2023). However, the characterization of the fire regime at small scales as well as the generation of knowledge of the post-fire dynamics of an ecosystem through the interpretation of its current structure, allows the design of effective strategies for ecosystem management and restoration. It can also help predict forests responses to future changes in the fire regime (Johnstone and Chapin, 2006; Gómez-Sánchez et al., 2017).

1.2. Objectives and hypotheses

1.2.1. General objective

To characterize the structural dynamics of mixed conifer forests of the Sierra Madre Occidental (SMO) affected by fires to generate information to design effective management strategies and forecast ecosystem responses to future changes in the fire regime.

Specific objectives:

(1) To compare species diversity and forest structure in sites affected by different levels of fire severity nine years after a fire event.

(2) To reconstruct tree-ring-based fire history.

(3) To interpret the impacts of fire and weather on tree age structure.

1.2.2. Hypotheses

(1) Species diversity and structural diversity are more complex in areas affected by intermediate fire severity regimes.

(2) Tree recruitment dates in the current stand are attributed to a combined effect of past fires and prevailing climatic conditions after those fires.

1.3. Thesis structure

This thesis is structured in several chapters: Chapter 1 presents a general introduction, research objectives, and hypotheses. Chapter 2 presents a literature review that includes key concepts in the evaluation of the effect of fires on ecosystems and their structural response. Chapter 3 includes the first scientific article generated, entitled "*Effect of fire severity on the species diversity and structure of a temperate forest in northern Mexico*", which addresses the first specific objective and the first hypothesis, and attempts to answer the question of whether there is a relationship between vegetation structure and diversity and fire severity, and whether vegetation diversity and structure vary along a fire severity gradient. Chapter 4 includes the second scientific article entitled "*Impact of fire history on the structure of a temperate forest in northern Mexico*" in which the second and third specific objectives are addressed, as well as the second hypothesis. Finally, Chapter 5 includes the general conclusions of the thesis.

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CHAPTER II: THEORETICAL-CONCEPTUAL FRAMEWORK

2.1. Fire and combustion

A simple definition states that fire is the release of light and heat generated by the combustion of a material (Merrill and Alexander, 1987; Cochrane and Ryan, 2009). Combustion is the breaking and reforming of chemical bonds, where the total energy in the rearranged bonds is less than that of the original bonds. Thus, the energy change generated by these rearrangements is released in the form of light and heat (Cochrane and Ryan, 2009).

2.2. Forest fire definition

A forest fire is the decomposition of forest fuels into carbon dioxide, water vapor, minerals, and small amounts of numerous gases, in addition to the release of energy in the form of light and heat (Rodríguez-Trejo, 2015). In operational terms, a forest fire is understood as the unplanned spread of fire over forest vegetation, where combustible material is the element that determines its magnitude (CONAFOR, 2021).

2.3. The fire triangle

There are three elements necessary for a fire to occur: fuel, heat, and oxygen (Stein, 2010). When these three elements are present at the appropriate levels, combustion begins (Lafon et al., 2017). However, fuel is the easiest element to control (Stein, 2010). In terms of forest fires, fuel is plant material that can ignite and burn, these can be classified according to their dimensions (Chandler et al., 1983).

2.4. Fire regime

Fire regime can be described as the patterns of fires that characterize an area (Canadian Interagency Forest Fire Centre [CIFFC], 2003). A fire regime reflects the spatiotemporal variation of the fire triangle elements in a landscape (Lafon et al., 2017), it also integrates interactions between fires and vegetation, climate, and terrain conditions at the landscape scale (Jardel et al., 2014).

Some of the most important characteristics of a fire regime are fire extent, seasonality, frequency, intensity, and severity (Bowman et al., 2020; Franquesa et al., 2022).

- Fire extent refers to the size of a fire or the average size of fires experienced by the ecosystem over long periods of time (Cochrane and Ryan, 2009).

- Fire seasonality refers to the period of the year during which the conditions exist for fires to start and spread. The fire season is generally divided according to the seasonal flammability of fuel types (e.g., spring, summer) (CIFFC, 2003).

- Fire frequency is one of the most used characteristics to describe a fire regime and is defined as the number of fires occurring per unit time at a given site. It is also described as the interval between fires at a given site (Li, 2002).

  Ffire intensity refers to the energy released by the fire (kW/m) and can be estimated as the product of the linear propagation rate (m/s), the heat of combustion (kJ/kg) and the mass of fuel burned (Cochrane and Ryan, 2009).

- Fire severity refers to the ecological change that the fires generate in the ecosystems. The severity of the fire is characterized based on the loss or decomposition of organic matter, both on the surface and underground (Keeley, 2009).

Fire regimes are classified based on the characteristics of the fire or the effects it produces (Agee 1996). In that sense, Brown and Smith (2000) proposed a classification of fire regimes into four types:

- Understory fire regimes: fires are not lethal to the dominant vegetation and approximately 80% of the dominant vegetation survives.
- Stand replacement fire regime: fires kill the aerial parts of the dominant vegetation, modifying the aerial structure, and approximately 80% of the vegetation is consumed or dies.
- Mixed severity fire regime: there is selective mortality of trees according to the susceptibility to the fire of the different species.
- Non-fire regime: the occurrence of fire is scarce or null.

2.5. Severity and frequency of fire

Fire intensity and severity are operational and manageable measures and have been used to assess ecosystem responses to fire (Keeley, 2009). Fire intensity and severity appear to follow similar topographic patterns as fire frequency (Lafon et al., 2017), therefore, knowledge about historical fire frequency and severity can help guide silvicultural systems and fire suppression policies (Brookes et al., 2021).

2.6. Influence of fire on vegetation (diversity, structure, and regeneration)

Structure, diversity, and tree density are the main characteristics of forest stands (Gadow et al., 2007), which are modified by environmental and anthropogenic disturbances, resulting in forest change (Payette, 1992), therefore are key elements in assessing forest stability (Lähde et al., 1999).

Fire disturbance determines patterns of vegetation structure and composition in many forest ecosystems (Johnstone et al., 2004), as they affect tree mortality and recruitment (Kemp et al., 2019). In that sense, tree regeneration patterns can be interpreted because of the frequency and magnitude of disturbances. Therefore, the structure and composition of future forests will be mainly a consequence of current recruitment patterns (Rozas, 2003).

On the other hand, fire is known to influence plant species diversity, and the effects can be grouped into some general patterns (He et al., 2019). However, there may be an optimal range of fire regime characteristics that support the greatest diversity, as indicated by the Intermediate Disturbance Hypothesis (IDH) (Conell et al., 1978). Thus, the absence of fire can reduce tree diversity in some ecosystems (Abreu et al., 2017).

Fire enhances biodiversity through evolutionary and ecological processes (He et al., 2019). Fires create a new habitat with more resources and less competition (Pausas and Keeley, 2019). On the other hand, to take advantage of this habitat, many plants have developed adaptive strategies for persistence under recurrent fires (Keeley et al., 2011).

In Mexico, fire favors an abundant natural regeneration, specifically in conifer ecosystems (Juárez-Martínez and Rodríguez-Trejo, 2003), and the structure and diversity of species in some forests affected by the fire are greater than in unburned areas (Cadena-Zamudio et al., 2022). Therefore, an understanding of the mechanisms and effects of disturbances on ecosystems is essential to interpret their current structure, design management strategies, and anticipate responses to future changes in the disturbance regime (Johnstone and Chapin, 2006).

2.7. Fire ecosystem responses

Ecosystem responses to fire and fire regimes over time are manifested in changes in species composition, changes in vegetation structure and habitat conditions, changes in landscape dynamics, and changes in water, carbon, and nutrient dynamics in the ecosystem (Jardel et al., 2014).

On the other hand, post-fire recovery depends on the interaction of various fire-related factors, post-fire environmental conditions, and the characteristics of the species that dominate landscapes before and after a fire (Hayes and Robeson, 2011).

Ecosystem recovery after a fire depends on tree regeneration, which in turn depends on seed production, seedling establishment, and survival (Johnstone et al. 2016). Fire severity and spatial heterogeneity of burning affect seed availability by modifying the distance at which seeds must disperse to reach a burned site (Haire and McGarigal, 2010). On the other hand, wildfires can also create conditions suitable for tree germination and survival (Tepley et al. 2013). Additionally, germination and survival require suitable post-fire weather conditions (Stevens-Rumann et al., 2018; Hankin et al., 2019).

In high severity burned areas, mineral seed beds are created and resources such as light, water, and nutrients are released that facilitate tree recruitment (York et al. 2003). However, high tree mortality may also contribute to lower seed availability (Kemp et al. 2016).

In that regard, Kemp et al. (2019) found that temperature and seed availability were the most important predictors of regeneration in some conifers. An Another factor that influences post-fire regeneration is the openings in the canopy created by the fire since competition is reduced, and resources increase (York et al. 2003). In Mexico, tree regeneration of both *Pinaceae* and *Fagaceae* is higher in forests with a dominant canopy of pines, in comparison with those dominated by an oak canopy (Alfaro-Reyna et al., 2019). In addition, mixed forests are characterized by being more productive, having greater stability and less risk of disturbances (Liu et al., 2022).

In Mexico, *Fagaceae* regeneration appears to be favored by warmer climates and presence of forest fires, relative to *Pinaceae*, suggesting that climate change may favor oak dominance (Alfaro-Reyna et al., 2019). Therefore, further studies linking fire histories to post-fire changes in forests are needed (Harley et al., 2018).

2.8. Assessment of the fire effects on forest ecosystems

Fire effects assessment is essential to (1) document the effects of fire, (2) assess damage, (3) assess a burn, and (4) assess the potential for future treatments.

However, the assessment of the effects of fire requires abundant funds, resources, and proven experience (Lutes et al., 2006).

Several methodologies have been proposed for the evaluation of fire effects on forest ecosystems, for example, Key and Benson (2004) proposed the Composite Burn Index (CBI), which visually evaluates the changes in vegetation and soil caused by fire through the following variables: amount of fuel burned, level of soil degradation and mortality in the vegetation stratum; resulting in the site at various levels of severity. They also indicate that remote sensing rates correlate with fire severity measured in the field, which facilitated assessments of fire effects on ecosystems (Key and Benson, 2006). Thus, De Santis and Chuvieco (2009) developed and evaluated a modification to the CBI called the Geometrically structured Composite Burn Index (GeoCBI), which considers the fraction of cover of the different plant strata and changes in the Leaf Area Index (LAI) of the canopy and subcanopy.

Those studies rely on pre- and post-fire satellite imagery to estimate the amount of fire-induced change; the most used metrics are delta normalized burn ratio (dNBR) (Key and Benson, 2006), relativized delta normalized burn rate (RdNBR) (Miller and Thode, 2007), and relativized burn rate (RBR) (Parks et al., 2014). These metrics generally have a high correspondence with field measures of fire severity (Parks et al., 2018). The feasibility of studies using satellite imagery and drones to assess changes in vegetation cover and composition following wildfire also has been evaluated recently (Martinez et al., 2021).

Guidelines have also been proposed to measure the immediate post-fire conditions in the soil (cover and loss of organic matter, color, and changes in its structure) and in the vegetation (percentage of sooting and calcination of the foliage of the tree and shrub strata) (Lutes et al., 2006; Parson et al., 2010). Other studies have considered different variables to estimate post-fire severity in the field. For example, percentage of tree basal area mortality (Welch et al., 2016), decrease in vegetation cover (Tessler et al., 2016), minimum terminal diameter size in thin branches of vegetation (Moreno and Oechel 1989), age-class distribution (Wagner, 1978), tree mortality and recruitment (Etchells et al., 2020).

On the other hand, dendrochronological techniques are suitable for reconstructing disturbances, forest structure and composition (Fulé and Covington, 1997). Reconstruction of past fire regimes linking fire history with population dynamics and climate effects on survival has proven to be a useful tool for understanding the fire effects on ecosystems (Swetnam and Baisan, 1996; Grissino-Mayer, 2001; Wang and Ying, 2009). Globally, studies have been conducted to reconstruct fire regimes, the influence of fire on ecosystem dynamics, and fire drivers operating at a range of temporal and spatial scales (Harley et al., 2018).

Similarly, tree ring dating has been widely used to determine the age of trees (Metsaranta, 2020), as a group of trees of the same age is the result of past disturbance (Duncan and Stewart, 1991; Clark, 1991). Fire is a disturbance that influences the age structure of the stand (Iniguez et al., 2016). Severe fires cause high tree mortality and when subsequent climatic conditions are adequate, there is an establishment of trees with a uniform age structure (Fulé and Laughlin, 2007). In contrast, uneven-aged forests are associated with a continuous regeneration pattern where disturbances are frequent and of low severity (Weatherspoon, 1996; Taylor, 2010; Harley et al., 2018; Sáenz-Ceja and Pérez-Salicrup, 2020).

Although most methodologies to assess the effects of fires focus mainly on vegetation, other studies have analyzed other aspects of the ecosystem as soil erosion (Fernandez et al., 2010), runoff and sediment production in fire-affected areas (Rubio et al., 1997), post-fire water quality (Basso et al., 2020), and hydrological balance (Venkatesh et al., 2020). CO_2 emissions from wildfires (Amiro et al., 2009), human health (Dvornik et al., 2018), properties of fire-affected soils (Verma and Jayakumar, 2012), and the influence of fire on wildlife (Van Lear and Harlow, 2002).

2.9. Pyrosilviculture

Understanding the relationships between silviculture and fire management can contribute to more effective management of forest ecosystems (Weatherspoon, 1996). In that sense, the concept of pyrosilviculture has been originated, which can be defined as the use of prescribed fire to achieve forests management objectives or as the alteration of non-fire silvicultural treatments to the incorporation of fire in the future (York et al., 2021).

The goals of pyrosilviculture are to create the conditions, so that the next fire that occurs is a prescribed fire (Levine et al. 2020). In addition, preparation treatments can occur decades before burning (York et al., 2022a). Pyrosilviculture can be applied independently of objectives, can be applied, and then adjust prescribed burning applications (York et al., 2021).

Prescribed fire is arguably the ideal silvicultural tool for creating conditions that most closely resemble a disturbance regime that has been disrupted by prolonged fire suppression. (York et al., 2022b). Pyrosilviculture expands the objectives of prescribed burning since it includes reducing stand density, greater forest heterogeneity and the selection of species and trees better adapted to disturbance regimes (North et al., 2021).

Given the economic, ecological, and social importance of forests in Mexico, and the fundamental role of fire in some of these forests, the need to develop sustainable management plans that include fire in accordance with the fire regime to which these forests have adapted has been suggested (Fulé and Covington, 1997; Cerano-Paredes et al., 2022). Specifically, it has been suggested that pyrosilviculture could be an interesting and a helpful tool for the management of fire-adapted forests (Cerano-Paredes et al., 2022). However, a previous step is the fire regime characterization and its influence on vegetation.

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Effect of Fire Severity on the Species Diversity and Structure of a Temperate Forest in Northern Mexico

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CHAPTER III: EFFECT OF FIRE SEVERITY ON THE SPECIES DIVERSITY AND STRUCTURE OF A TEMPERATE FOREST IN NORTHERN MEXICO

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Article

Effect of Fire Severity on the Species Diversity and Structure of a Temperate Forest in Northern Mexico

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Abstract: Forests experience disturbances, such as fire, that affect their functioning, structure, and species composition. The objective of this work was to compare the species diversity and forest structure at sites affected by different degrees of fire severity, 9 years after a forest fire event. We used the differenced Normalized Burn Ratio (dNBR) index. Vegetation was sampled in three severity levels: high (HS), moderate (MS), and low (LS) and included the unburned (U) level as control. In addition, we calculated the species richness (S), the Shannon index (H), and the Evenness Index (E). The structural diversity of tree diameters and heights was measured applying the indices H, E, and the coefficient of variation (CV). The differences in the indices calculated across the fire severity levels were determined through analyses of variance (ANOVA) and Tukey's multiple comparison tests. The results showed no significant differences ($p \leq 0.05$) in the species diversity indices between fire severity levels. The structural diversity of tree diameters and heights was lower at the HS level. dNBR was negatively related to structural diversity; thus, it is concluded that HS tends to reduce structural variability in terms of diameter, height, and age. These results provide a baseline to understand how fire can modify forest structure and species diversity.

Keywords: dNBR; structural diversity; intermediate disturbance hypothesis (IDH); forest response; tree age structure; fire disturbance

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1. Introduction

At the global level, forest ecosystems face increasingly frequent natural and human disturbances. Forest fires are one of the natural disturbances with the greatest impact on the dynamics of these ecosystems [1–3]. Specifically, the functioning of forests and the structure and composition of forest species are largely conditioned by the frequency and severity of fire [4,5]. Climate change projections indicate a trend toward higher severity and frequency of fires [6]. Given this situation, analyzing the response of ecosystems to fire is essential to support natural resource managers in anticipating the effects of future fires and determining better management practices [7,8].

Two approaches can be used to evaluate these responses: descriptive or process-based. The descriptive approach is used to measure the impacts of forest fires and generate a statistical description of the relationships between fire severity and ecosystem response. The process-based approach is based on controlled experimental conditions and studies the process including variables such as fire intensity measurements and ecosystem response variables [9].

In the descriptive approach, fire severity quantifies the ecological effects of a forest fire and the degree of change in ecosystem components [10]. Species diversity and forest structure are important features of a forest ecosystem, and the assessment of these features provides essential elements to analyze forest alteration and response [11].

Several studies have explored the response of ecosystems to fire. Barton and Poulos [12] found that fire and topography were drivers of plant diversity in a site in the southwestern United States. Berkey et al. [13] indicate that succession after a severe fire plays a central role in shaping forest structure. Other studies have identified a decrease in species richness as fire severity increases [14].

In Mexico, it has been found that as the severity of the fire increases, the trees tend to form dimensionally heterogeneous stands [15]. Moderate severity has also been found to favor the highest composition, structure, and diversity of forest ecosystems [16]. In parallel, it has been observed that a fire of moderate severity fosters the natural regeneration of trees and shrub species [17]. These results have been associated with the intermediate disturbance hypothesis, which states that moderate disturbance levels increase local species diversity [18].

The objective of this work was to compare the species diversity and forest structure at sites affected by different degrees of fire severity, 9 years after a forest fire event in a temperate mixed forest in the Sierra Madre Occidental (SMO). This study addresses the following questions: (1) Is there a relationship of vegetation structure and diversity with fire severity? (2) Do vegetation diversity and structure vary across the fire severity gradient? It is also hypothesized that species diversity and structural diversity are higher in areas affected by an intermediate or moderate fire severity.

2. Materials and Methods

2.1. Study Area

The study was carried out in the El Brillante ejido, located in the SMO mountain region within the Pueblo Nuevo municipality, southwest state of Durango, Mexico (23°37′12.97″–23°50′55.76″ N and 105°18′56.23″–105°31′9.84″ W) (Figure 1a). The average altitude is 2480 m a.s.l. and the local climates are semi-cold humid C(E)(m) and temperate subhumid C(w), both with summer rains [19]. The mean annual temperature ranges from 10 °C to 18 °C and the mean annual precipitation is 1000 mm [20]. The dominant soil types in the area are Luvisol and Regosol [20]. The predominant vegetation is mixed-conifer forest (mainly of the genus *Pinus*); in some cases, this vegetation is mixed with broadleaved trees (mainly *Quercus*) [21].

Forest fires in the study area are usually small and of low intensity, affecting mostly herbaceous and shrubby understory vegetation. They do not regularly exceed 100 ha; however, in years with adverse conditions, large fires of mixed severity may occur, which can cause tree mortality. This situation was present in 2012, when an atypical fire occurred, possibly influenced by the dominant dry conditions of the previous year [22]. According to the Comision Nacional Forestal (National Forestry Commission, CONAFOR), the agency that manages fires in Mexico, this fire was classified as mixed severity, affecting an area of near 1300 ha of pine-oak forest.

Through the "Forest Fire Hazard Prediction System of the National Forestry Commission (CONAFOR) of Mexico. Available online: http://forestales.ujed.mx/incendios2/# (accessed on 14 July, 2022)", historical records of fire in this area were taken into consideration for this study. Some scared trees affected by fire in this area were also considered in this analysis, indicating that the area of influence of the 2012 fire was not burned for at least 14 years before the analyzed fire event and 9 years after the occurrence of this fire.

We determined the fire severity gradient of the 2012 fire using the differenced Normalized Burn Ratio (dNBR) index (Figure 1b), which represents the change in the landscape caused by fire and allows the classification of fire severity into different levels, namely unchanged, low, moderate, and high [23]. The dNBR was calculated from Landsat satellite images on the Google Earth Engine cloud-based platform using the command sequence from Parks et al. [24]. This code uses average pre- and post-fire dNBR values ('composite mean') over a specified period. Fire severity levels were based on the "United States Geological Survey (USGS) thresholds. Available online: https://un-spider.org/fr/node/10959 (accessed on 14 July, 2022)", sorted into four severity levels: Unburned (U), Low Severity (LS), Moderate Severity (MS), and High Severity (HS).

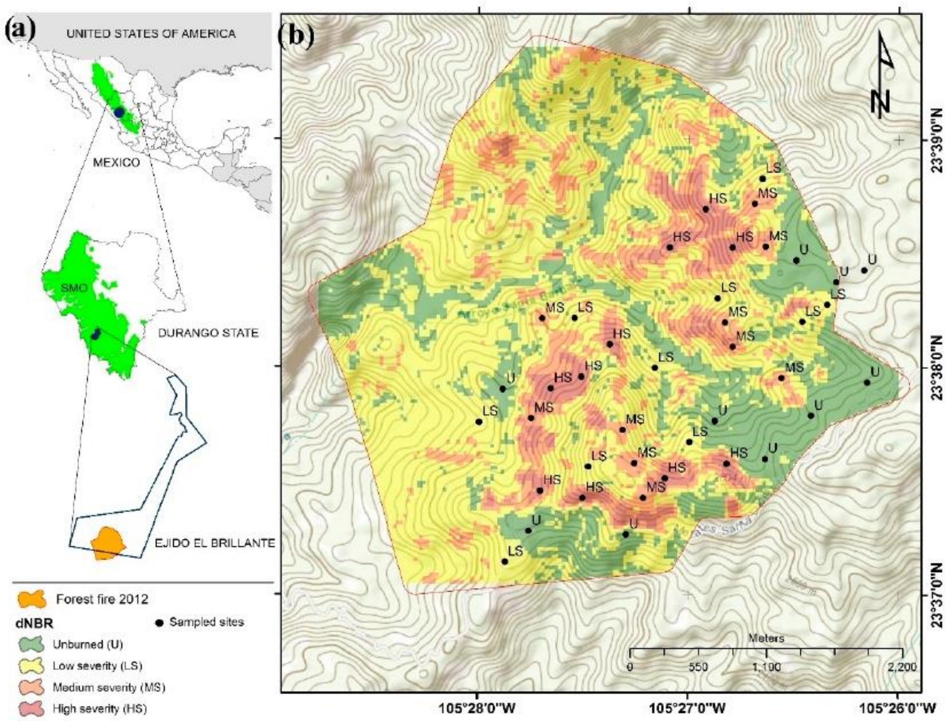

Figure 1. (**a**) Location of El Brillante ejido and the area burned in 2012 in the Sierra Madre Occidental (SMO); (**b**) fire severity levels calculated through the dNBR and distribution of sampled sites by severity level.

2.2. Data Collection

Based on the dNBR index, the most representative pixels of each severity level were located; then, 40 circular plots with a surface area of 0.10 ha were set following a targeted approach. The distribution consisted of 10 sampling sites by severity level, including the U category, which was used as a control or reference category. Tree vegetation sampling was carried out in June 2021. At each sampling site, all live and dead trees with diameter $\geq$ 5.0 cm were recorded; for each, we determined the condition (alive or dead), species, diameter at breast height (DBH in cm), overall height (m), and crown diameter (m). Additionally, using a Pressler increment borer, we obtained tree increment cores of each diameter class (5.0 cm classes) found in the site to determine age. Cores were extracted at a height closest to the ground; the number of rings was adjusted when the increment core did not contain the pith [25]. The regeneration by species was identified and counted the number of individuals per species, including height (cm) and diameter at ground level (cm).

2.3. Species Diversity and Structural Diversity

For each sampling site, we calculated the species richness (S), the species diversity through the Shannon index (H), and the Evenness Index (E). We analyzed the percent similarity in species composition between zones with different fire severity levels. This analysis was carried out by means of a Bray–Curtis dendrogram, which considers species abundances [26].

To measure structural diversity in terms of diameter and height, the indices H and E were applied. In this case, the number of species was replaced by the number of diameter or height-size classes, similar to the approached followed in other studies [11,27,28], which have reported relevant results. In addition, the coefficient of variation (CV) was included (Table 1). All the indices were calculated using the *BiodiversityR* package in R software (Version 4.1.0, Vienna, Austria) [29].

Table 1. Formulas for calculating species diversity and structural diversity indices.

	Index	Formula
Species diversity	Species richness	$S = NS$
	Shannon Index	$H = -\sum_{i=1}^{NS} n_i \times ln(n_i))$
	Evenness Index	$E = \frac{H}{ln(NS)}$
Structural diversity	Shannon Index	$H = -\sum_{j=1}^{NC} n_j \times ln\left(n_j\right)$
	Evenness Index	$E = \frac{H}{ln(NC)}$
	Coefficient of variation	$CV = \frac{\sqrt{\sigma_X^2}}{\overline{X}}$

S = Species richness, H = Shannon index, E = Evenness index, CV = Coefficient of variation, NS = total number of species at sampling site i. n_i = individual number of the i-th species. n_j = individual number of the j-th diameter or height class. NC = total number of diameter or height classes or categories, as appropriate (a size-class of 5.0 cm was used for diameter and 1.0 m for height). X = variable of interest (diameter or height).

2.4. Tree Age Structure

Increment cores were dated using conventional dendrochronological techniques [30]. The approximate age of each tree was estimated by counting the number of annual rings plus the estimated years taken for the tree to reach 1.3 m in height, adjusting for the missing years when the growth core did not reach the pith [31].

2.5. Statistical Analysis

For the statistical analysis, the species diversity and structural diversity indices calculated for each sampling site were tested for normality and homoscedasticity of variances. The differences in species richness, species diversity, and diameter and height structural diversity between fire severity levels were also assessed through one-way analyses of variance (ANOVA). When statistically significant differences were found at a 5% significance level, we conducted Tukey's multiple comparison tests.

We explored whether fire severity (value of dNBR) and some topographic factors, such as elevation (m), slope (degrees), and slope exposure (degrees), were associated with data on the species diversity and structural diversity indices using a Pearson's correlation analysis (r). Subsequently, those variables that showed a statistically significant correlation at a 5% significance level were included in a linear regression model to determine the relative influence of each factor on the diversity and structure indices. The explanation of the variation of the data in the different relationships explored was interpreted with the coefficient of determination (R^2) for each model. All the statistical analyses were performed in the R software [32].

3. Results

3.1. Species Diversity

The study recorded 2065 trees belonging to 18 species in five families: *Pinaceae*, *Fagaceae*, *Cupressaceae*, *Ericaceae*, and *Betulaceae*. For S, by fire severity level, 15 species were found in U areas, 17 in LS, 13 in MS, and 14 in HS (Table 2).

Table 2. Tree species found by fire severity level.

Unburned	Low Severity	Medium Sseverity	High Severity
Pinus cooperi	*Pinus cooperi*	*Pinus cooperi*	*Pinus cooperi*
Pinus durangensis	*Pinus durangensis*	*Pinus durangensis*	*Pinus durangensis*
Pinus leiophylla	*Pinus leiophylla*	*Pinus leiophylla*	*Pinus leiophylla*
Pinus teocote	*Pinus teocote*	*Pinus teocote*	*Pinus teocote*
Pinuis engelmannii	*Pinus engelmannii*	*Pinus strobiformis*	*Pinus engelmannii*
Pinus lumholtzi	*Pinus lumholtzi*	*Juniperus deppeana*	*Pinus strobiformis*
Pinus strobiformis	*Pinus strobiformis*	*Quercus sideroxyla*	*Pinus herrerae*
Juniperus deppeana	*Pinus herrerae*	*Quercus urbanii*	*Quercus sideroxyla*
Quercus sideroxyla	*Juniperus deppeana*	*Quercus crassifolia*	*Quercus crassifolia*
Quercus urbanii	*Quercus sideroxyla*	*Quercus fulva*	*Quercus fulva*
Quercus crassifolia	*Quercus urbanii*	*Quercus rugosa*	*Quercus rugosa*
Quercus rugosa	*Quercus crassifolia*	*Alnus* sp.	*Alnus* sp.
Alnus sp.	*Quercus fulva*	*Arbutus xalapensis*	*Arbutus xalapensis*
Arbutus xalapensis	*Quercus rugosa*		*Arbutus madrensis*
Arbutus madrensis	*Alnus* sp.		
	Arbutus xalapensis		
	Arbutus madrensis		

No differences in S were determined across fire severity levels ($F = 2.16$, $p = 0.110$) (Figure 2a). For species diversity, H (Figure 2b) and E (Figure 2c) values also did not show significant differences across fire severity levels at a 5% significance level (H: $F = 1.54$, $p = 0.222$; E: $F = 0.96$, $p = 0.421$).

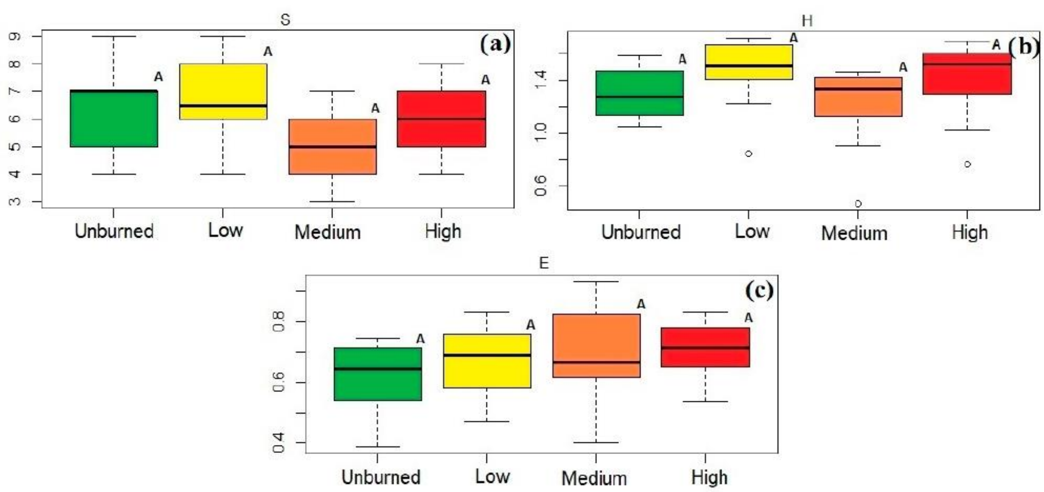

Figure 2. Tukey's multiple comparison test of (**a**) species richness (S), (**b**) Shannon Index (H), and (**c**) Evenness Index (E) by fire severity level. Different letters indicate significantly different means ($p < 0.05$); the same letters indicate non-significant differences.

The similarity analysis showed two groups: one, zones affected by LS and U fire, and two, zones affected by MS and HS fires, indicating similarity of species (Figure 3).

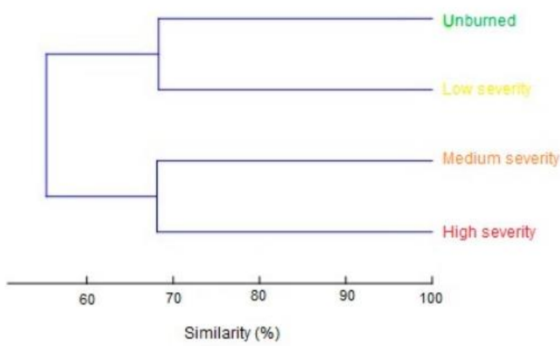

Figure 3. Dendrogram of species composition similarity at sites affected by different degrees of fire severity.

3.2. Diameter Structural Diversity

The number of diameter categories (*CATD*) in the HS level was lower than the number of *CATD* in the other severity levels ($F = 10.32$; $p = 0.001$). In the severity levels U, LS, and MS, the number of *CATD* was not significantly different (Figure 4a).

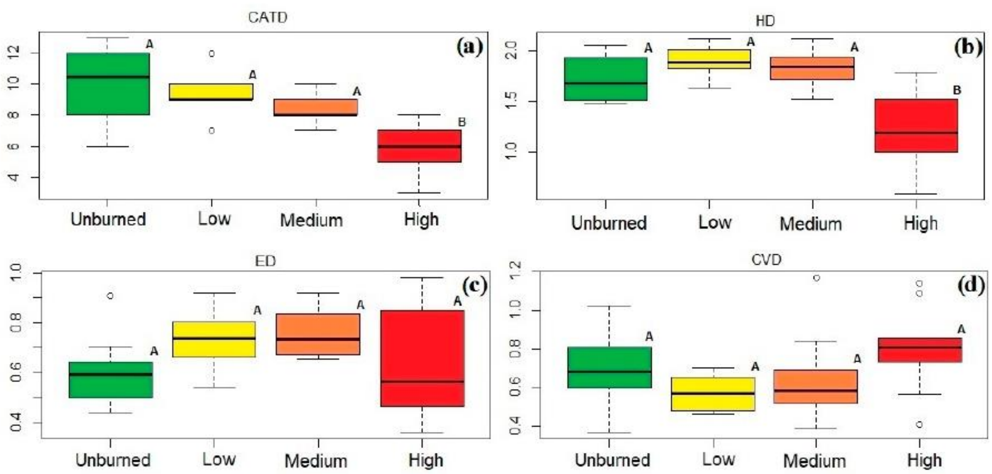

Figure 4. Tukey's multiple comparison test for (**a**) number of diameter categories (*CATD*), (**b**) Shannon index of diameter categories (*H-D*), (**c**) Evenness Index of diameter categories (*E-D*), and (**d**) the coefficient of variation for diameter categories (*CV-D*) by fire severity level. Different letters indicate that means were significantly different ($p \leq 0.05$); the same letters indicate non-significant differences.

Similarly, *H* values for diameter (*H-D*) showed significant differences across the severity levels ($F = 14.34$; $p = 0.001$). Particularly, *H-D* for the HS level was the only structural diversity value that showed significant differences compared to the *H-D* values of the other fire severity levels. The means of severity levels U, LS, and MS were not statistically different between them (Figure 4b).

The *E* index for diameter (*E-D*) (Figure 4c) and the *CV-D* (Figure 4d) were statistically equivalent across fire severity levels at a 5% significance level (*E-D*: $F = 2.80$, $p = 0.054$; *CV-D*: $F = 2.66$, $p = 0.063$).

3.3. Height Structural Diversity

The number of height categories (*CATH*) was statistically different across severity levels ($F = 3.49$; $p = 0.026$). The highest number of *CATH* was observed in the U level and the lowest in the HS level (Figure 5a). The Shannon index for height (*H-H*) in the HS level was the lowest relative to the other fire severity levels ($F = 21.54$; $p = 0.000$) (Figure 5b).

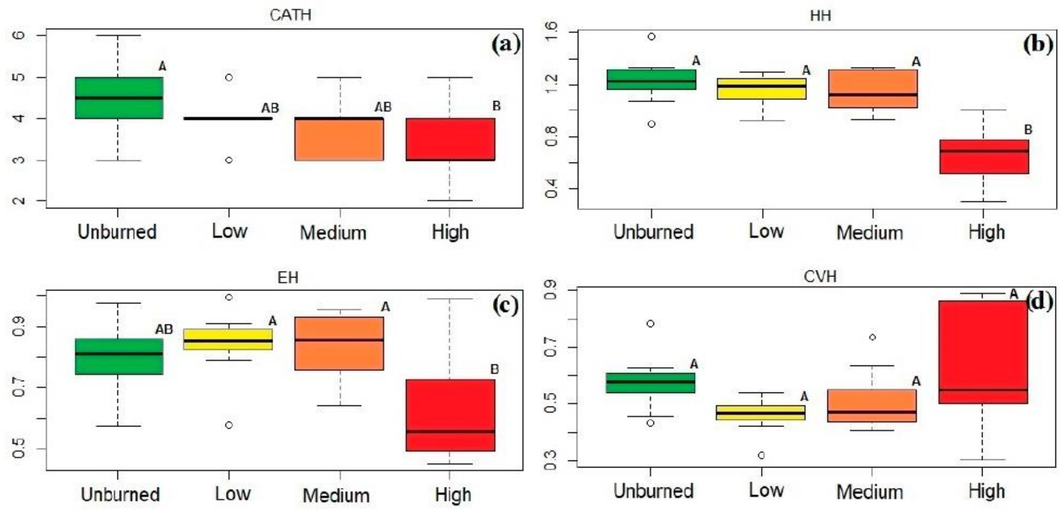

Figure 5. Tukey's multiple comparison test for (**a**) number of height categories (*CATH*), (**b**) Shannon index (*H-H*), (**c**) Evenness Index (*E-H*), and (**d**) coefficient of variation for height (*CV-H*) by fire severity level. Different letters indicate that means were significantly different ($p \leq 0.05$); the same letters indicate non-significant differences.

As regards the *E* index for height (*E-H*), statistically significant differences ($F = 5.09$; $p = 0.005$) were found across the fire severity levels. The *E-H* index values for U and HS were lower than those for the LS and MS levels. However, the *E-H* value for the HS fire level was even lower than the value for the U level (Figure 5c). The coefficient of variation for tree height (*CV-H*) was not statistically different ($F = 5.09$; $p = 0.005$) across fire severity levels (Figure 5d).

The Tukey's multiple comparison test only indicated at what fire severity level there is higher or lower structural diversity; however, it did not indicate tree size (height or diameter). A graphical representation of the number of individuals in each diameter class across severity levels showed that most individuals in sites corresponding to HS and U levels belong to the smallest diameter classes (i.e., 5.0, 10.0, 15.0, and 20 cm) (Figure 6a). The same trend was observed for tree height, that is, most individuals in sites subjected to HS and U fires belong to the smallest height classes (Figure 6b).

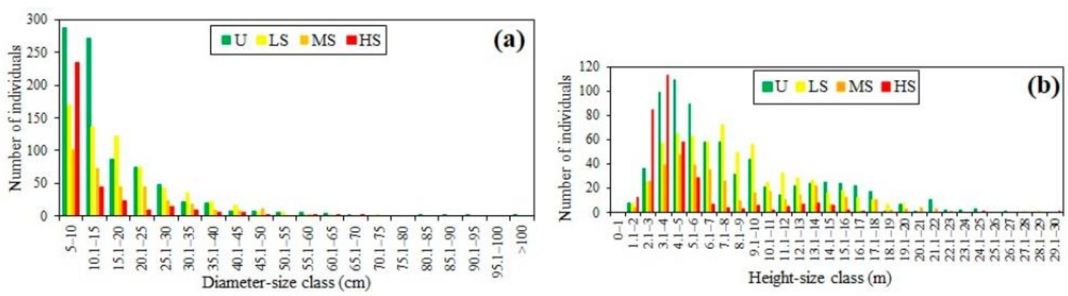

Figure 6. (**a**) Distribution of diameter-size classes across fire severity levels and (**b**) distribution of height classes in each fire severity level.

3.4. Tree Age Structure

In U areas, individuals of all age size-classes were found. Figure 7 displays the distribution of age classes by fire severity level. In areas affected by LS fire, the individuals found correspond to all age categories from 21 years of age (Figure 7b). In MS areas, individuals were in most age categories from 31 years (Figure 7c). In contrast, in HS areas, most individuals (50%) belong to the youngest age classes (5.0–10.0 years), and the representativeness of individuals in other age size-classes was very low or nil (Figure 7d). The number of age classes in areas affected by HS fire was statistically different from those in the other areas, which were not different from each other at a 5% significance level ($F = 4.60$; $p = 0.004$).

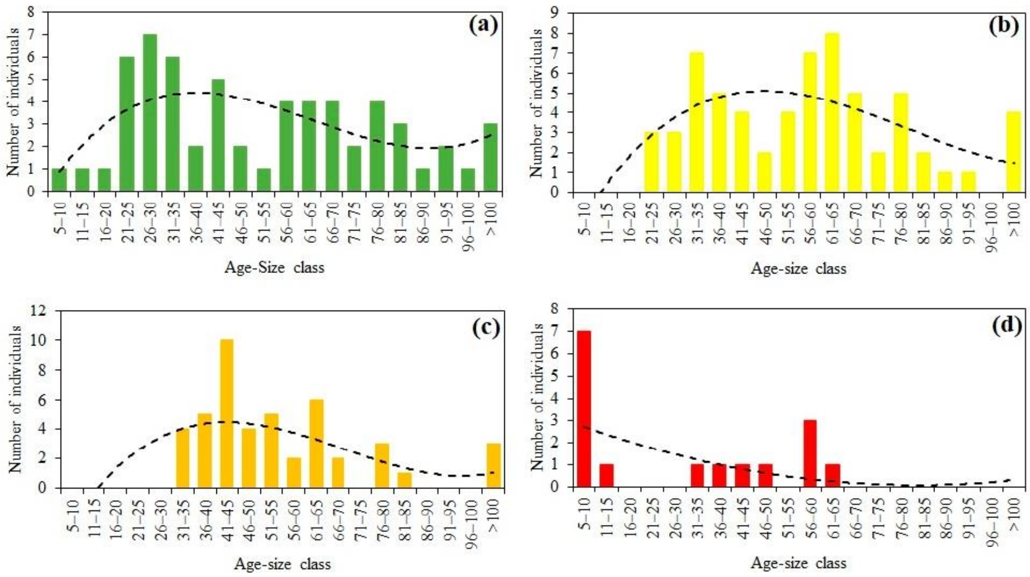

Figure 7. Tree age structure by severity levels (**a**) unburned areas, (**b**) areas affected by low fire severity, (**c**) areas affected by medium or moderate fire severity, and (**d**) areas affected by high fire severity.

3.5. Correlation and Regression Analysis

The relationships between topographic variables and diversity indices through r were statistically equal at a 5% significance level (Figures S1–S4). However, the relationship between the values of the structural diversity indices *CATD*, *H-D*, *CATH*, *H-H*, *E-H* and dNBR was statistically different at a 5% significance level. The linear regression analysis between dNBR and structural diversity indices shows that all relationships are negative and statistically different at a 5% significance level (Figure 8).

Species diversity and structural diversity indices of the regeneration stratum were not statistically different across fire severity categories. Additionally, an analysis of the species diversity and structural diversity of regeneration by sprouting showed that there were also no statistically significant differences across fire severity categories.

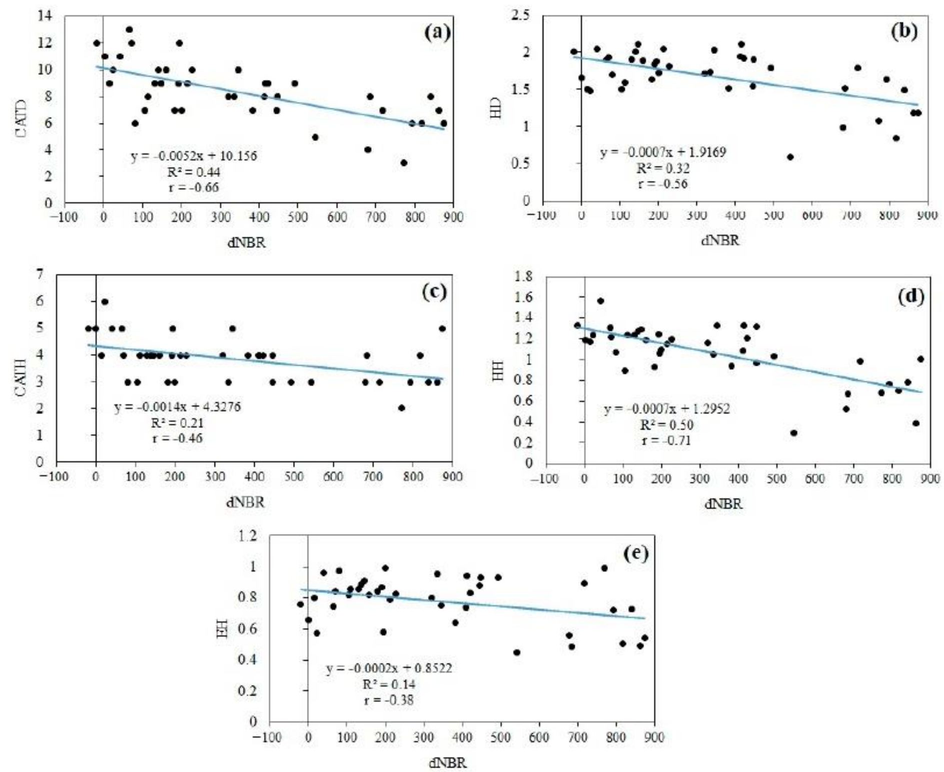

Figure 8. Scatter plots showing the linear association between (**a**) *CATD* and dNBR; (**b**) *H-D* and dNBR; (**c**) *CATH* and dNBR; (**d**) *H-H* and dNBR; and (**e**) *E-H* and dNBR.

4. Discussion

Species diversity and structural diversity in a mixed temperate forest affected by different levels of fire severity were analyzed using different indices and fire severity levels. Since understanding fire behavior and the drivers of variability in forest structure and species composition support forest conservation and fire management [33], these results are a baseline for the analysis of forest responses after almost one decade of disturbance.

Although work is currently ongoing in Mexico on the establishment of fire assessment and monitoring protocols, particularly as regards the assessment of fire severity [34], the assessment of long-term ecological impacts is highly relevant, and fire severity thresholds play a central role [35]. Although the aim of the present study was not to assess fire severity "per se", the preliminary analysis through dNBR was a very helpful tool for characterizing fire severity, particularly because the severity of fire is hard to characterize in the field 9 years after the event. In this sense, Flores-Rodriguez et al. [36] indicate that these indexes can be implemented when direct field assessment is challenging, such as in extensive, inaccessible areas, or when a rapid preliminary diagnosis is required.

4.1. Changes in Species Diversity

Regarding species diversity, it was found that the *S* values estimated for the study area were similar to the values reported for this region by Domínguez-Gómez et al. [37]. Additionally, the *H* index for each severity level is within the highest ranges reported for this region [37,38]. Studies at this respect, have found that fire severity modifies species diversity, mainly in the lower strata [12–14,39,40]. The results of this study show that the diversity of tree species did not differ across fire severity levels. This may indicate the adaptation of species to fire and their capacity to regenerate under the ecological conditions produced by fire [41].

4.2. Changes in Structural Diversity

In a forest ecosystem, diversity refers not only to species richness but also to the diversity of sizes of the tree species in it [11]. The assessment of diameters and heights through structural diversity indices allowed the characterization and comparison of the structure at sites affected by different fire severity levels. The most important findings of the present analysis are that *CATD* and *H-D* are lower in sites affected by HS than in sites under other fire severity levels, which do not differ from each other. In addition, regarding tree height, the *CATH*, *H-H*, and *E-H* indices were also lower in sites affected by high fire severity. An explanation of these results may be that the high fire severity is associated with higher mortality of the tallest trees, which in this case were more susceptible to high-intensity fires [42], which results in structures of similar size [43].

4.3. Tree Age Structure

The age structure of trees and the processes that shape this structural composition are also highly important for the proper management of forest ecosystems [25]. Age structure is commonly associated with disturbances, since age distribution is the result of tree mortality and the emergence of new cohorts after disturbances [44]. Fire is considered a key disturbance that influences forest age structure [25]. The results of the present study indicated that, similar to the structure of diameter and height, age structure was more uniform in areas affected by HS fire than in areas under other fire severity levels. The above is due to the fact that severe and infrequent fires trigger the replacement of the stand; later, suitable weather conditions favor a synchronous or pulsed establishment of trees with a uniform age structure [45]. In contrast, uneven-aged forests display a pattern of continuous regeneration that may be associated with frequent and slight disturbances such as shallow or low-intensity fires [46,47].

4.4. Implications of a Forest Homogeneous in Sizes and Ages

All our results suggest that the HS fire level did not affect species diversity. However, a lower structural diversity and tree age structure was found in areas affected by HS fires, resulting in homogeneous or even-aged forests. Although HS patches are smaller than patches subjected to the other fire severity levels, the former may produce adverse effects. For example, an even-aged forest is more susceptible to disturbance than an uneven-aged forest. In addition, a reduction in structural diversity may lead to a reduction in productivity, given the positive relationship between these two variables [27,28]. The homogenization of forest structure and species composition can also translate into more extensive and severe fires [43]. Additionally, trees affected by fire, given their stressed condition and loss of vigor, are more susceptible to pests such as bark beetles of the genus *Dendroctonus* [48].

4.5. Species Diversity-dNBR Relationship

This study also explored the relationship of species diversity and structural diversity indices with topographic variables and dNBR. Unlike other studies that have reported a relationship between species diversity and some biophysical variables [40], no relationship was observed between species diversity and topographic variables in the present study. With regard to the species diversity–fire severity relationship, no relationship was found either. However, the results may vary for different strata of the ecosystem, as reported in various studies. For example, Brodie et al. [49] found that the species richness in the understory increased in parallel with increasing fire severity. For their part, González-de Vega et al. [50] and Flores-Rodríguez et al. [17] indicate that moderate fire severity may foster natural regeneration. However, the relationship between alpha diversity and fire severity in higher vegetation strata is unclear [12]. The inclusion of other variables (i.e., species functional traits), including the adaptation of species to fire, regrowth capacity, and germination, among other physiological characteristics, may add clarity to this relationship [14].

4.6. Structural Diversity–dNBR Relationship

As regards structural diversity, a negative relationship was found between the structural diversity of diameter and height with dNBR. However, structural diversity showed no relationship with topographic variables. Although fire severity is known to strongly affect forest structure [43], biophysical variables also play a central role [13] since the fire severity gradient can be parallel to the altitude-related temperature and humidity gradients [43]. In this case, fire severity appears to be largely related to relief slope.

Finally, an aspect worth noting is the relationship observed between the structural diversity of diameter and height with dNBR since the forest structure variables were measured in 2021, while the dNBR was obtained from Landsat 7 ETM+ images captured in 2012. This fact supports our results on how the severity of fire triggered a change in forest structure in the study area, which has prevailed almost a decade after the disturbance. For example, Berkey et al. [13] suggest that the time elapsed after a fire event is also essential in determining forest structure. Therefore, future studies should explore the long-term effects of fire severity, considering the interannual variability of climate and the years after the fire, which will determine the potential influence of these variables on forest structure. In addition, most of the conifer and broadleaved species of the region are present in our study plots; this favors that the findings of this study could be applied in great part to the mixed-conifer ecosystems present in northern Mexico. This information could also provide knowledge to improve or to develop management strategies for future fires affecting this ecosystem.

5. Conclusions

Fire severity was not related to species diversity, but showed a negative linear relationship with structural diversity and did not influence the variation in species diversity. These results provide answers to our research questions. Structural diversity was different only in areas affected by high fire severity, a finding that differs from the research hypothesis. In this way, a fire of high severity tends to homogenize the forest, making it more prone to disturbance, including fire. These results represent an approximation to the response of forest to fire severity and establish a baseline on how fire shapes forest species diversity and structure in the study region. These findings can inform the development of similar studies in Sierra Madre Occidental. These results also provide valuable information for proper fire management in the study area, where preventive actions to avoid high-severity fires, such as prescribed burning to reduce litter in steep-slope areas, can contribute to the stability of forest communities.

Supplementary Materials: The supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/f13071121/s1, Figure S1: Scatter plots showing the association between tree diversity indices, structural diversity, and dNBR; Figure S2: Scatter plots showing the association between tree diversity indices, structural diversity, and slope (degrees); Figure S3: Scatter plots showing the association between tree diversity indices, structural diversity, and elevation (m); Figure S4: Scatter plots showing the association between tree diversity indices, structural diversity, and aspect (degrees).

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CHAPTER IV: IMPACT OF FIRE HISTORY ON THE STRUCTURE OF A TEMPERATE FOREST IN NORTHERN MEXICO

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Article

Impact of Fire History on the Structure of a Temperate Forest in Northern Mexico

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Abstract: Understanding the ecological role of fire in forests is essential for proper management and conservation programs. The objectives of this study were: (1) to reconstruct the history of fires in a temperate forest in Sierra Madre Occidental; and (2) to interpret the impacts of fire and climate on forest structure. Sixty tree cross-sections with fire scars were analyzed, and descriptive statistics of fire history were generated. Additionally, growth cores were analyzed, and the ages of trees of different diameter categories were calculated. The synchrony between fire history and tree establishment was determined, and precipitation and Palmer Drought Severity Index (PDSI) values were correlated with the number of trees established per year. The presence of 137 fire scars was determined, which allowed the reconstruction of 41 fire events over the period 1855–2019; however, only the period 1940–2015 was used to compare tree recruitment, as tree establishment was detected in this period. The mean fire interval (MFI) was 2.28 years in general, and 12.17 years for extensive fires. As regards vegetation, a continuous recruitment pattern was observed, typical of a frequent low-intensity fire regime, although peak regeneration occurred after extensive fires. The correlation analysis showed that the number of trees established per year was influenced by the wet conditions that occurred in December of the previous year and the dry conditions in September and October of the previous year. This finding demonstrates the historical influence of fire and climate on the structure of the current stand in the study area. Therefore, the present study highlights the importance of including fire in forest management programs, considering the natural fire regime to which the species in this ecosystem are already adapted.

Keywords: age structure; fire regimes; fire scars; tree recruitment; fire reconstruction

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1. Introduction

Fire has played a major role in the evolution of global ecosystems [1] and is a key ecological process in conifer forests in the Western North American Mountain Range (WNAMR), where the environment is influenced by multiple factors, such as high rainfall variability and a 4- to 6-month annual drought, which favor frequent fires [2,3].

The Sierra Madre Occidental (SMO) of Mexico, belonging to the WNAMR, supports a mosaic of diverse ecosystems of great environmental and economic importance [4], reflected in a complex mixture of fire regimes [2]. These ecosystems include mixed pine–oak forests, which harbor a high floristic diversity [5]. Historically, surface fires were frequent in these forests, which may have contributed to the maintenance of their biodiversity [6].

Wood production is one of the main economic factors in SMO, highlighting the need to develop sustainable forest management plans based on the understanding of natural fire regimes [7]. Therefore, the analysis of fire regimes is highly important for ensuring the integrity of forest ecosystems [8].

The reconstruction of fire history using tree rings to determine spatial and temporal patterns of previous and contemporary fires (dendropyrochronology) allows understanding of the fire regime [9,10]. Knowing the history of fires in a certain area allows for understanding forest dynamics and its effect on vegetation structure [11].

In this sense, the presence of groups of trees of the same age has been used as compelling evidence of the occurrence of past disturbances [12], since age distribution results from tree mortality and the emergence of new cohorts after disturbance [13]. Among ecosystem disturbances, fire is considered a key disturbance that influences forest age structure [14]. In this way, the combination of fire history and tree age can help us to understand the patterns of tree age structure and their underlying processes, which are essential to sustainable ecosystem management [15].

In Mexico, several studies have reported the influence of fire history and climate on tree age structure in the SMO [6,16–19]. Since the effects of fire vary according to the natural fire regime under which the species evolved [20], it is relevant to study the fire history at small spatial scales to determine the disturbance regime in a particular area [21].

Based on the above, the objectives of this study were (1) to reconstruct the history of fires in a temperate forest in SMO; and (2) to interpret the impacts of fire and climate on tree age structure. We hypothesized that dates of tree recruitment in the current stand are attributed to a combined effect of past fires and prevailing climatic conditions following those fires. The results of this work contribute information about the effects of fire on the dynamics of temperate forest ecosystems in SMO.

2. Materials and Methods

2.1. Study Area

The study was carried out in the ejido El Brillante, located in the southwestern region of the state of Durango, Mexico (extreme coordinates 23°37′12.9″–23°50′55.76″ N, and 105°18′56.23″–105°31′9.84″ W) (Figure 1a), in the mountainous region of the Sierra Madre Occidental (SMO) (Figure 1b). The average elevation of the study area was 2480 m above sea level, and the local climates are semi-cold humid C(E)(m) and temperate subhumid C(w), both with summer rains [22]. The mean annual temperature ranges from 10 °C to 18 °C, and the mean annual rainfall is 1000 mm [23]. The main soil types in the area are morphologically classified as Luvisol and Regosol [23]. The dominant vegetation was a mixed conifer forest, mainly *Pinus* sp., occasionally mixed with broad-leaved vegetation (mainly *Quercus* sp. [24]). The sampling site included an area of approximately 100 ha (Figure 1c).

2.2. Data Collection

Following a selective sampling strategy, we selected the oldest trees with specific characteristics, including evidence of fire scars and number of scars [25]. Subsequently, we obtained cross-sections of both living and dead trees (including fallen logs, standing logs, and stumps (Figure 2a–d) with fire scars) (Table 1) [26].

Table 1. Number and percentage of fire-scarred cross-sections collected.

Collected Samples	Dated Samples	Live	Dead	Stumps	Species
60	58	19	11	28	Pd, Pt, Ax
%	96.67	31.67	18.33	46.67	

Pd: Pinus durangensis Martínez, Pt: Pinus teocote Schiede ex Schltdl, Ax: Arbutus xalapensis Kunth.

Additionally, using a Pressler borer, increment cores were obtained from trees of different diameter classes (5.0 cm classes) to estimate their age. Cores were extracted at a height as close to the ground as possible to obtain the inner growth [15].

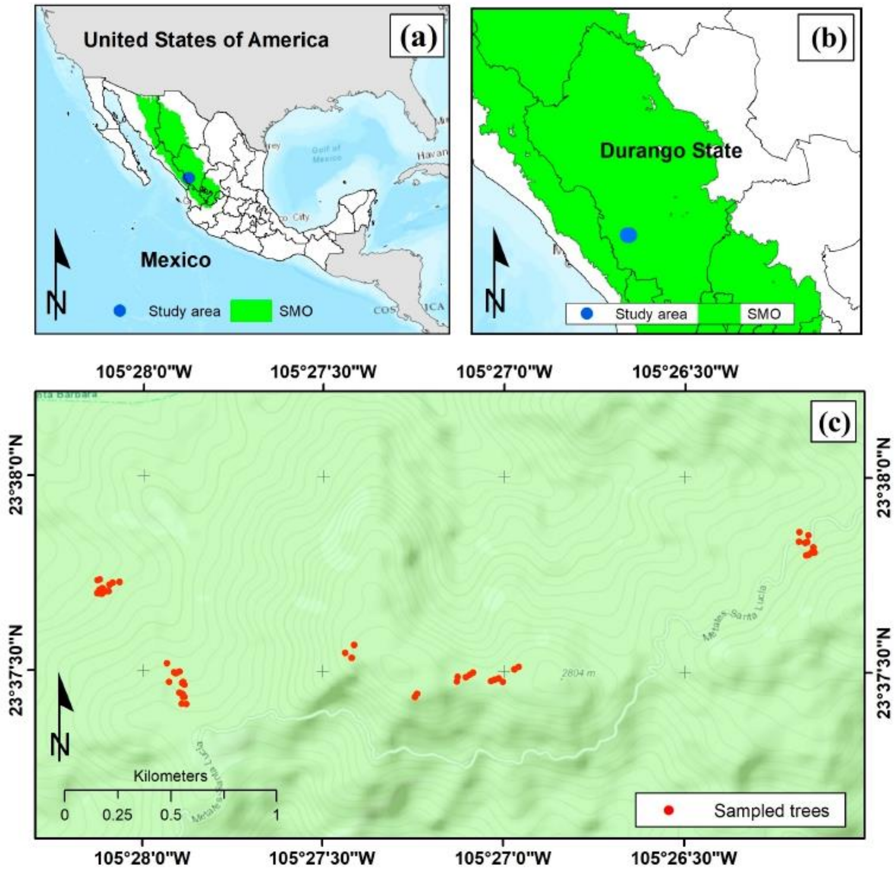

Figure 1. (**a**) location of the study area, (**b**) location of the study area in the context of the SMO, and (**c**) geographic distribution of collected tree cross-sections.

2.3. Fire Reconstruction

Samples with fire scars were analyzed using standard dendrochronology techniques to determine the exact calendar year of each fire scar [27]. In all samples, total ring-width was measured with a precision of 0.001 mm using the Velmex measurement system [28], and the dating quality was validated with the COFECHA program [29].

2.4. Seasonality of Fires

The season of fire occurrence was estimated based on the relative position of each fire scar within the annual ring (Figure 3a): Early Earlywood "EE" (Figure 3c), Middle Earlywood "ME" (Figure 3d), Late Earlywood "LE", Latewood "L", and Dormancy "D" (Figure 3b) [30,31]. The location of fire scars within the annual ring was grouped into two categories for further analysis: (1) spring (EE + D); and (2) summer (ME + LE + L) [9,17].

2.5. Determination of Tree Age

Increment cores were also dated using standard dendrochronology techniques [27]. The approximate age of each tree was estimated by counting the number of rings and adjusting for missing years when the growth core did not reach the pith [32]. The quality of dating was validated with the COFECHA program [29].

2.6. Fire Interval Analysis

Fire history data were explored using the FHAES software [33]. Two periods of fire history were analyzed: (1) full fire history (from the oldest to the most recent scar); and (2) from the establishment of the oldest tree to the establishment of the youngest tree.

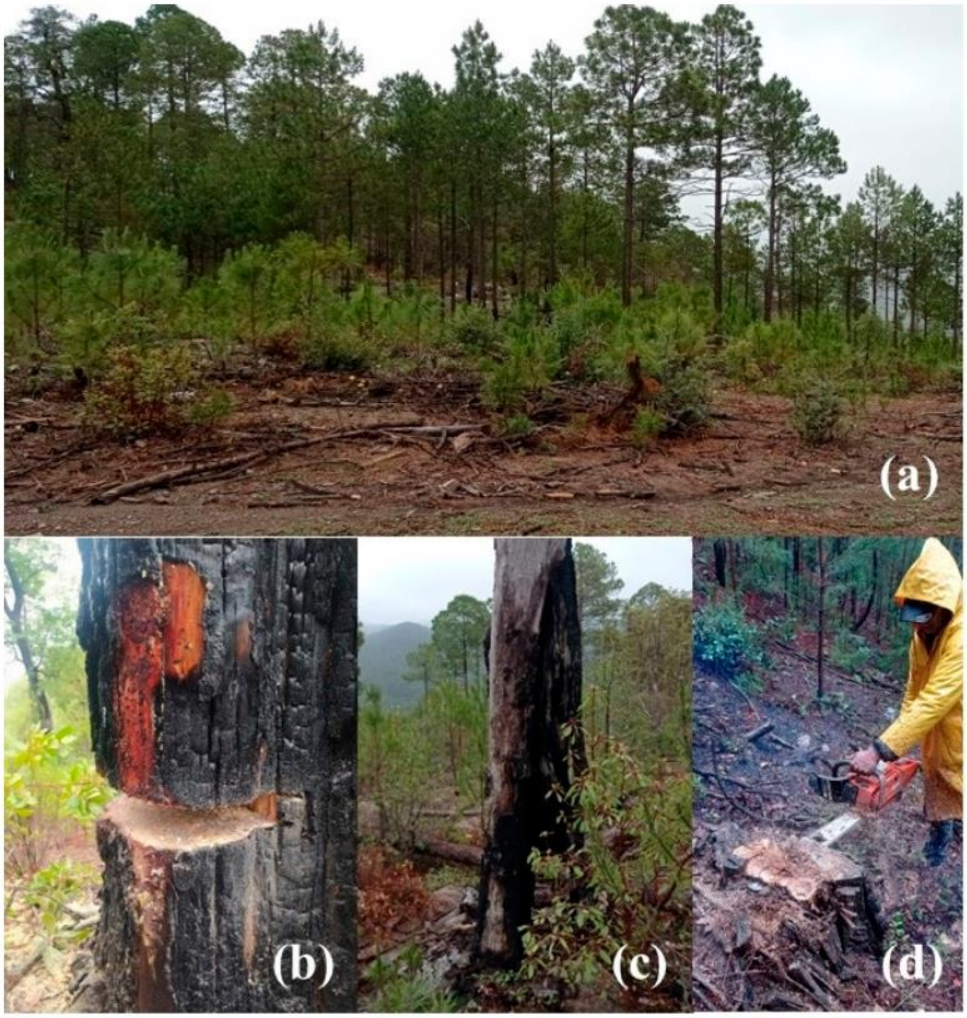

Figure 2. (**a**) Example of dominant ecological conditions in the study area. Fire-scarred samples were collected from a combination of (**b**) standing live trees, (**c**) standing logs, and (**d**) stumps.

For each period, fire history descriptive statistics were obtained, including mean fire interval (MFI), maximum and minimum fire intervals, and the Weibull median probability interval (WMPI) [9]. For each metric, three fire scar filters were considered: (1) all years with fire scars; (2) years in which 10% or more samples recorded a fire scar; and (3) years in which 25% or more samples recorded a fire scar. The 25% filter represents the years when fires were most extensive [34].

2.7. Climate–Fire Relationship

The relationship between climate variability and fires in the period 1940–2015 was determined by a superposed epoch analysis (SEA) using the FHAES software [33]. Three climatic proxies were used:

(1) The number of rainy days in the December–February period [35];
(2) Instrumental values of Palmer's Drought Severity Index (PDSI) for May [36]; and
(3) El Niño 3 Sea Surface Temperature (SST) index values for June-August (NOAA, https://psl.noaa.gov/data/climateindices/list/; accessed on 2 December 2022).

All values were calculated during the year of the fire, as well as five years before and two years after the year of the fire. The statistical significance of the SEA was evaluated by calculating the 95% confidence interval derived from 1000 bootstrap simulations.

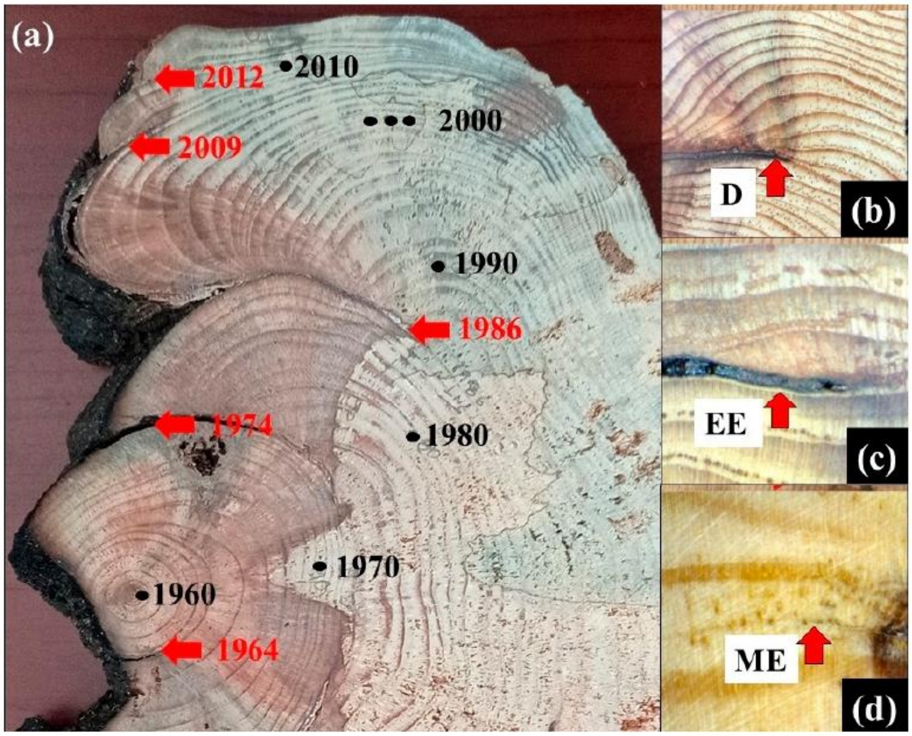

Figure 3. (**a**) An example of a fire-scarred sample that has been cross-dated using dendrochronological techniques. The specimen had five fire scars. Black numbers represent the calendar year of the sample. The numbers in red represent the calendar year when the wood was scarred. Examples of fire scars are: (**b**) the dormancy "D", (**c**) early-earlywood "EE", and (**d**) middle-earlywood "ME" within the annual tree-ring (This is adapted from [25]).

2.8. Relationship of Tree Age Structure with Forest Fires and Climate

The establishment of trees over time was graphically compared with the history of fires, particularly with extensive fires (fire scars are present in at least 25% of samples). Finally, to explore the influence of climate on post-fire recruitment, the number of trees established each year was correlated with rainfall data and PDSI values [36].

3. Results

3.1. Fire History

The 58 samples analyzed yielded a total of 137 fire scars. The oldest and newest rings corresponded to the years 1838 and 2020. The oldest scar was recorded in 1855 and the most recent in 2019, representing a reconstruction of the frequency of fires over the past 165 years. The period when scars were recorded in at least 25% of the samples ranged from 1855 to 2012 (Figure 4).

3.2. Fire Intervals

In the analysis of fire intervals for the period 1855–2019, and based on all years marked by fires, we found an MFI of 4.02 years and a WMPI of 2.4 years (Table 2). Based on the 10% filter, the MFI was 7.86 years, with a WMPI of 5.27 years. More extensive fires that produced scars in 25% of the samples occurred at intervals of less than 20 years (MFI and WMPI of 15 and 12.43 years, respectively). For the period 1940–2015, and based on all years marked by fires, fires occurred more frequently, with an MFI of 2.28 years, and a WMPI of 2.09 years. Considering fires that left scars in 10% of the samples, the MFI was 4.56 years, and the WMPI was 3.87 years. For fires that produced scars in 25% of the samples, we found an MFI of 12.17 years, and a WMPI of 11.21 years (Table 2).

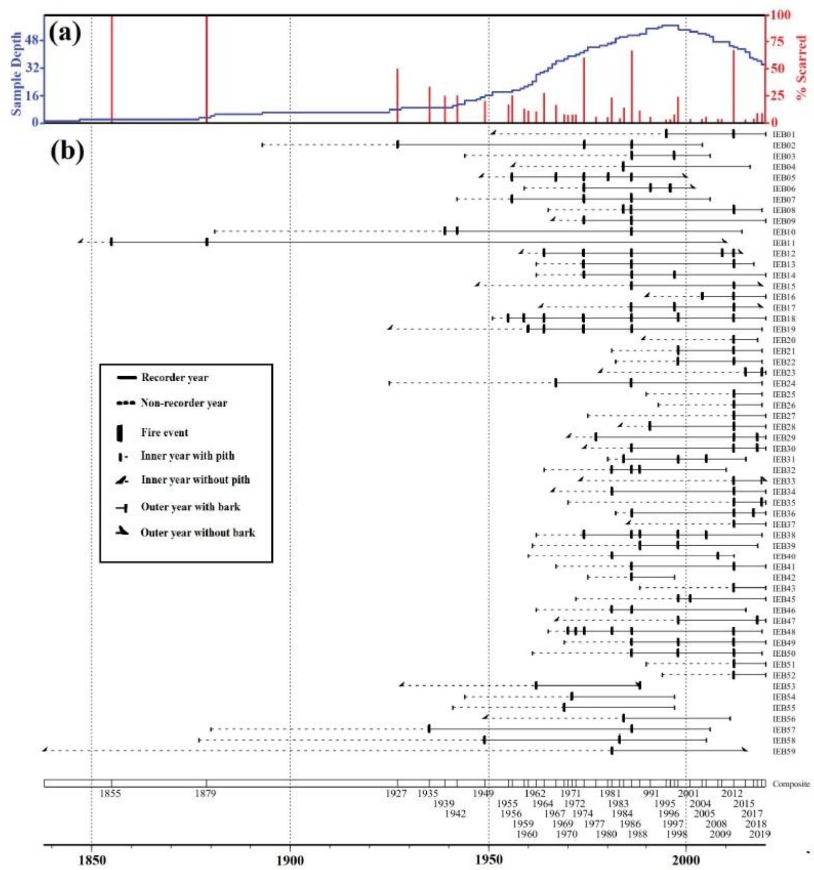

Figure 4. Timeline of fire history in the study area, (**a**) number of trees with recorded fire scars (blue line) and percentage of trees that recorded a fire per year (red vertical lines), (**b**) Individual timeline of a tree with fire scars represented by horizontal lines. Black vertical dashes represent fire scars recorded by that tree.

Table 2. Descriptive statistics of fire intervals.

Period of Analysis	Fire Interval Filter	Number of Intervals	MFI	Min	Max	WMPI
1855–2019	All scars	41	4.02	1	48	2.4
	≥10%	21	7.86	1	48	5.27
	≥25%	11	15	3	48	12.43
1940–2015	All scars	32	2.28	1	7	2.09
	≥10%	16	4.56	1	14	3.87
	≥25%	6	12.17	3	26	11.21

MFI: Mean fire interval; Min: Minimum frequency interval; Max: Maximum frequency interval; WMPI: Weibull median probability interval.

3.3. Seasonality of Fires

Seasonality was determined for 100% of fire scars, of which 38.0% were recorded at early earlywood (EE), and 39.4% were recorded at mid-earlywood (ME). As regards the timing of fire scars, 2.2% corresponded to late earlywood (LE), 4.4% to latewood (L), and 16.1% to dormancy (D). Based on the above, 54.01% of fires were classified as spring events and 45.99% as summer events (Table 3).

Table 3. Seasonality of fire scars.

Scars	Determined Seasonality	Undetermined Seasonality	D	EE	ME	LE	L	Spring Fires	Summer Fires
Number	137	0	22	52	54	3	6	74	63
Percentage (%)	100	0	16.1	38	39.4	2.2	4.4	54.01	45.99

EE: Early Earlywood; ME: Middle Earlywood; LE: Late Earlywood; L: Latewood; D: Dormancy.

3.4. Climate–Fire Relationship

The Superposed Epoch Analysis indicated that the frequency of fires in the period 1940–2015 was influenced by the climatic conditions recorded during the year of fire and those of years prior to each fire. Particularly, fires were influenced by the number of rainy days from December to February two years before the fire occurrence (Figure 5a). Regarding drought, fire occurrence was influenced by the drought conditions in May of the year of occurrence and the conditions within two years prior to the occurrence of fires (Figure 5b). As regards large-scale atmospheric circulation phenomena, we found that the climatic conditions indicated by the El NIÑO 3 index SST in June, July, and August, within 3, 4, and 5 years prior to fires, influenced their occurrence (Figure 5c).

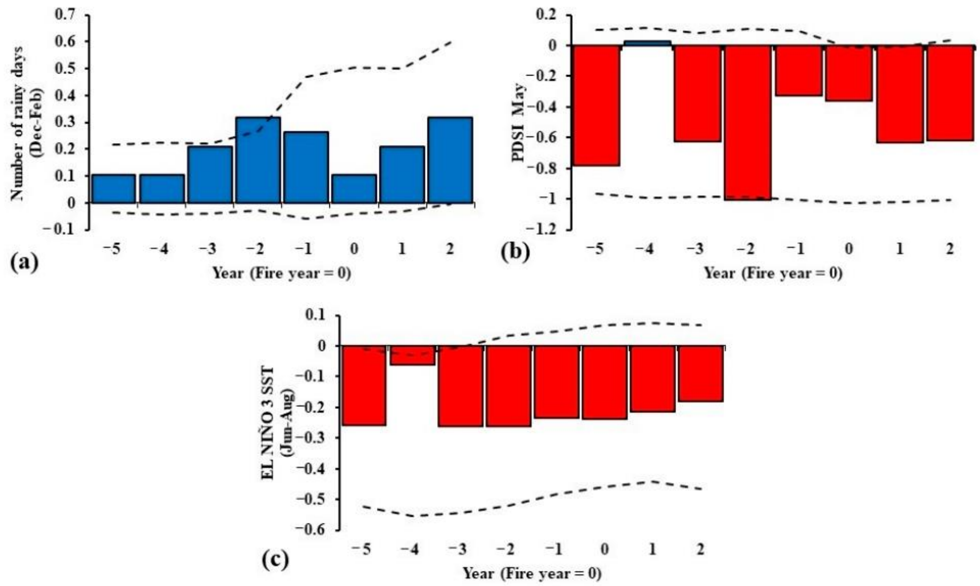

Figure 5. Superposed Epoch Analysis (SEA). Relationship between fire occurrence and (**a**) the number of rainy days in the months of December, January, and February, (**b**) PDSI (for the month of May) and (**c**) the NIÑO 3 SST index averaged over the months of June, July, and August. On the *X*-axis, the year in which the fire occurred is year zero, with climatic conditions on the *Y*-axis, which include conditions five years prior to the fire (negative values on the *X*-axis) and two years after the fire (positive values on the *X*-axis). The dashed lines represent a 95% confidence interval derived from 1000 Bootstrap simulations. The blue and red vertical bars represent wet and dry conditions respectively.

The most extensive fires (recorded in at least 25% of the samples) occurred in years with negative and near-zero positive PDSI values in the study area. In addition, drought conditions also prevailed over much of northern Mexico in most of the years of these fires (Figure 6).

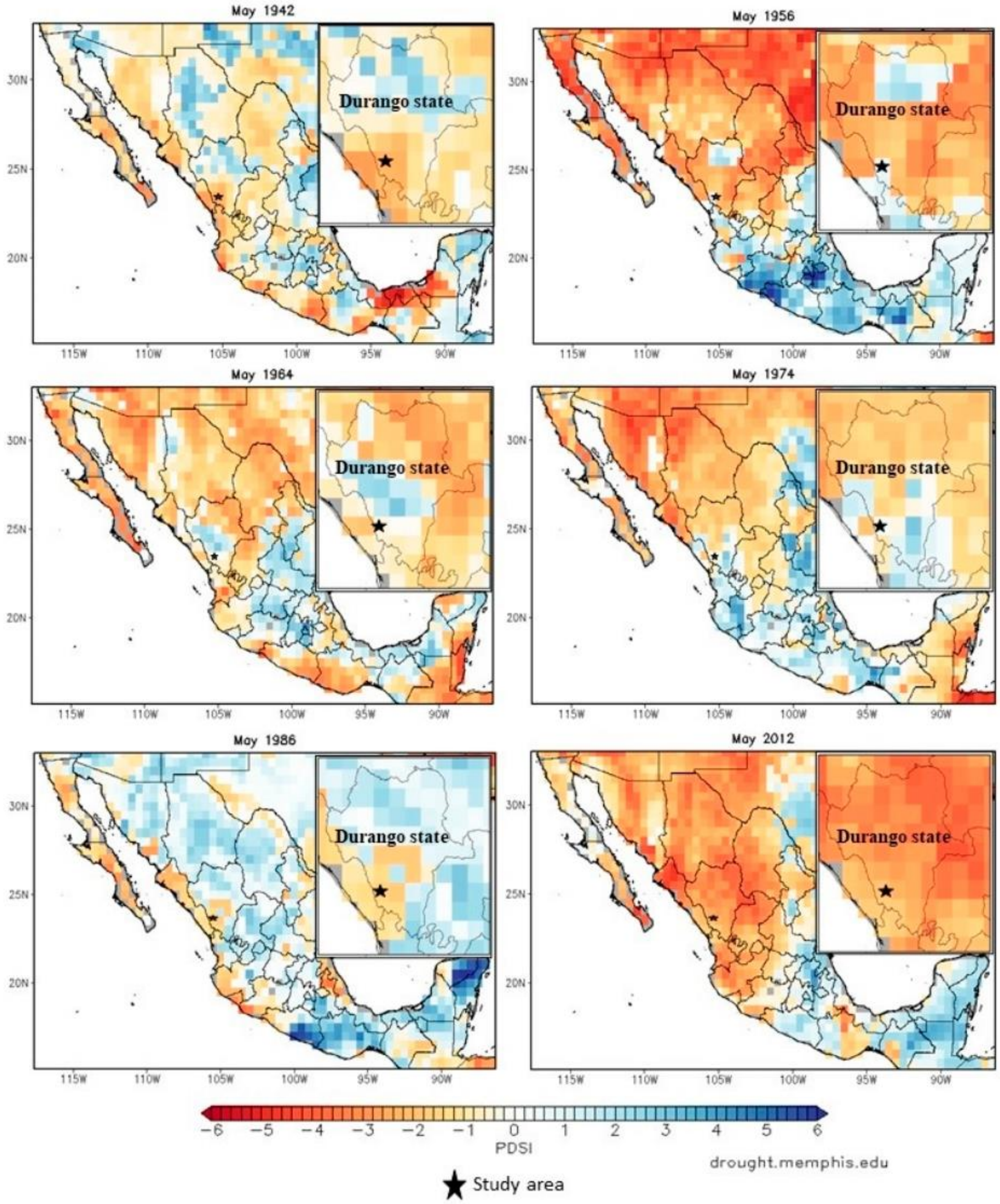

Figure 6. Drought conditions in Mexico during years of extensive fire occurrence in the study area [36].

3.5. Relationship of Tree Age Structure with Forest Fires and Climate

The tree species analyzed were *Pinus durangensis* Martínez, *Pinus leiophylla* Schiede ex. Schlechtedal & Chamissosubsp, *Pinus teocote* Schiede ex Schltdl, and *Pinus lumholtzii* Robins & Ferns. The first year of tree establishment, according to the analyzed samples, was 1940, while the last recorded year was 2015. The tree ages indicated that there were regeneration pulses after the occurrence of extensive fires (Figure 7). For the 1942, 1956, 1964, 1974, 1986, and 2012 fires, peak regeneration pulses occurred 5, 1, 5, 9, 2, and 1 years after the fire, respectively. To note, in all cases, tree regeneration occurred over at least 5 continuous years after the fire, showing a continuous regeneration pattern.

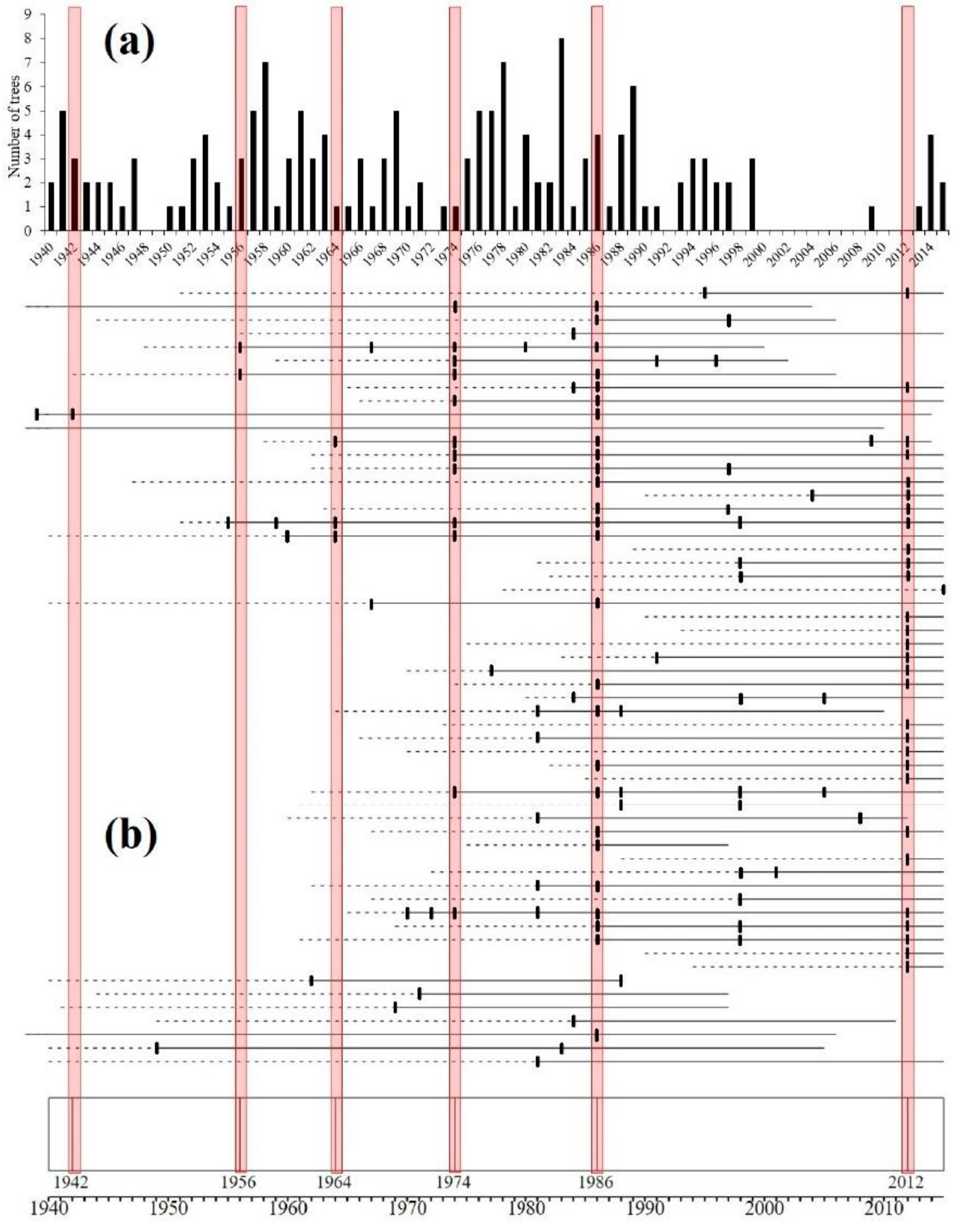

Figure 7. Graphical relationship between (**a**) tree establishment and (**b**) frequency of forest fires. Individual timeline of a tree with fire scars is represented by horizontal lines. Black vertical dashes represent fire scars recorded by that tree. The red lines represent the year of occurrence of extensive fires (scars marked in at least 25% of the samples).

Regarding the relationship between tree establishment and the dominant climatic conditions, a statistically significant ($p \leq 0.05$) positive relationship was found between the number of trees established per year and precipitation in December of the previous year. A negative relationship was also found between the number of trees established per year and PDSI values in September and October of the previous year (Figure 8).

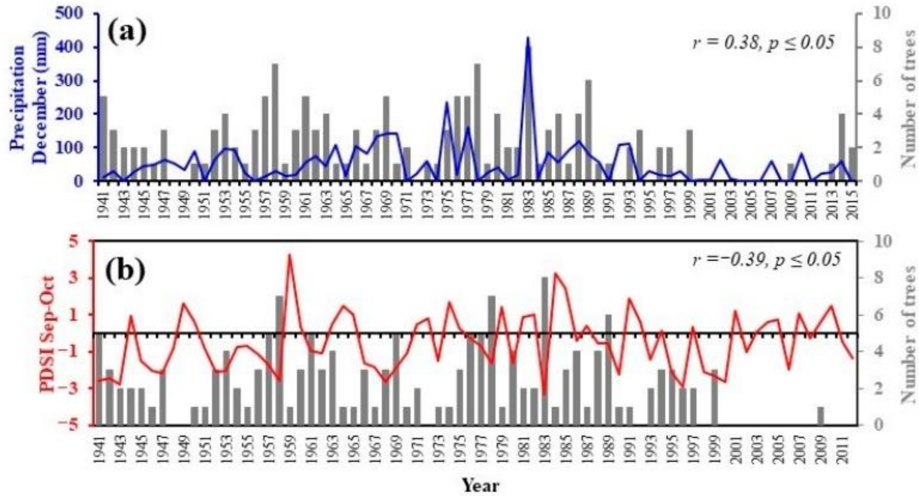

Figure 8. Graphical relationship between tree establishment and climate: (**a**) precipitation in December of the previous year and (**b**) PDSI values from September–October of the previous year.

4. Discussion

In this study, we reconstructed the fire history of a temperate forest located in the SMO and analyzed the influence of fire and environmental conditions on the recruitment rates of dominant forest species. Knowing the fire regime and its changes and effects on vegetation allows forest managers to develop plans to ensure the stability of these forests [7]. However, the management plans for many forests still lack quantitative information on the intensity, frequency, and seasonality of fires [21]. The findings reported herein will help to understand the fire regime and the historical influence of fire on vegetation, particularly in the study area. Given that the diversity of environmental conditions characterizing the SMO have favored a mixture of fire regimes, our results may not be representative of the entire ecological conditions of this mountain range. However, our results contribute to improve the network of site-specific fire history reconstructions for the North American tree-ring fire scar network [37], and to improve our knowledge of the influence of fires and climatic conditions in forest succession after the fire event.

4.1. Fire History

We reconstructed 165 years of fire history; however, the sample size was limited in the first 60 years, since almost 80% of the samples were no more than 70 years old. According to Sáenz-Ceja et al. [38], this is due to rapid re-population and intensive logging; therefore, the number of fire-scarred trees is limited, which also limits the reconstruction of extensive fire chronologies. Another aspect to highlight is that the area has a history of intensive logging, which seems to impact fire dynamics and tree regeneration [39].

4.2. Frequency of Fires

From the fire history analyzed, at least the past 70 years have been characterized by a regime of frequent and low-intensity fires; however, this does not preclude a similar frequency of fires in previous years. For example, a more extensive history of frequent forest fires has been found in areas near our study site [6], a pattern that may resemble the one in our study area, since climatic and vegetation conditions are similar over a large section of this region of the Sierra Madre Occidental.

Regarding the MFI, similarities were found with other studies conducted in the SMO. For instance, when considering all fires over the period 1855–2019, the MFI was four years, similar to the one reported by Fulé and Covington [16]. The MFI for the period 1940–2015 was 2.28 years, and is consistent with the MFI reported by Cerano-Paredes et al. [40] for the headwater of the Nazas River basin in the SMO. This is evidence that a higher frequency of fires has occurred in the study area in recent decades, in contrast with other SMO areas where a reduction in the frequency of fires after 1950 has been detected [6,7,17].

With regard to extensive fires (scars in at least 25% of the samples) and considering the periods 1855–2019 and 1940–2015, MFI values were 15 and 12 years, respectively. These results are similar to those of Rosales-Mata [41], who reports an MFI for extensive fires of almost 12 years in a zone adjacent to our study site. Similarly, Cerano-Paredes et al. [42] reported an MFI of nearly 14 years for a site in the Cerro El Mohinora Reserve, Chihuahua, within the SMO, and Sáens-Ceja and Pérez-Salicrup [43] found a similar MFI for the Monarch Butterfly Biosphere Reserve, located along the Trans-Mexican Volcanic Axes in the state of Michoacan, Mexico. This fire behavior can be attributed to the scarcity of combustible materials in the forest floor, as a result of frequent fires, relatively short periods with no fires, and a mosaic of forest patches that burned at different times [16].

4.3. Fire Seasonality

The evaluation of fire seasonality using the position of the fire scar within the ring has proven to be an accurate method for reconstructing the timing of historical fires [44]. Based on this method, most of the scars analyzed (54.01%) suggest that fires occurred mostly in the spring. In Mexico, the peak of the fire season in most regions of the country occurs during April, May, and June (spring) [45], before the summer rains and vegetation greening [46]. For the SMO, this information has been extensively documented by several studies, which support our results [7,40,42].

4.4. Climate-Fire Relationship

It is well known that climate and weather play a key role in forest fire dynamics [47]. In northern Mexico, a close relationship has been found between the occurrence of fires and climate [48]. Our results indicate that the occurrence of fires in the period 1940–2015 was influenced by both the climatic conditions in the year of occurrence and the climatic conditions in previous years. Specifically, the occurrence of fires was related to the number of rainy days in the December—February seasons of the two years prior to the event. In this sense, it has been mentioned that the presence of one to two wet years leads to the formation of a fine vegetation cover, which subsequently serves as combustible for fires in the following dry year [17].

A relationship was also found between the occurrence of fires and drought indicated by the PDSI, both in the year of occurrence and within two years before the fire. Unlike other regions of the country, the SMO requires more time (at least two years) for combustible materials to build up in amounts sufficient to sustain a fire. This can occur under conditions of low water availability in the soil and with the presence of combustible materials contributing to trigger the fire [49].

With regard to large-scale circulation phenomena, it is known that, in Mexico, the El Niño/Southern Oscillation (ENSO) modifies precipitation patterns in both winter and summer [50], which in turn has been related to the occurrence of forest fires. For example, in the southern United States and northern Mexico, warm events (El Niño) are associated with a lower incidence of fires, while cold events (La Niña) tend to favor a higher number of fires [51]. Likewise, our findings indicate that the occurrence of forest fires was influenced by dry conditions fostered by the El Niño phenomenon (negative El Nino SST Index values) in June—August of 3, 4, and 5 years prior to the fire.

Another aspect to consider is the occurrence of extensive fires, since they are commonly linked to dry years, caused by global and regional climate forcing mechanisms such as ENSO [40,42,51]. In the present study, the most extensive fires (i.e., fire scars in at least 25% of the samples) occurred under the dry conditions that dominated in northern Mexico

(Figure 4) [36]. In addition, the years of fire occurrence coincided with dry conditions present in the previous year and associated with ENSO.

4.5. Relationship of Tree Age Structure with Forest Fires and Climate

Fire may act as a disturbance that replaces the stand, or as a frequent disturbance of lower intensity that promotes continuous regeneration [52]. The unequal tree age structure found in the present study reflected a continuous regeneration pattern associated with a frequent, low-severity, low-intensity fire regime [14,20]. However, recruitment peaks were also observed after extensive fires. Although these fires are usually of high-intensity but infrequent, they cause considerable tree mortality, leading to the replacement of the stand and, subsequently, when weather conditions are adequate, they favor a synchronous or pulsed establishment of trees with a uniform or synchronous age structure [11,53].

The post-fire succession trajectories of vegetation depend on multiple factors [54,55]. Although seed availability is a major driver of tree regeneration after a fire [56], inter-annual climate variability in post-fire years may be a key factor to tree regeneration [57–59]. Our results showed that the dry conditions in September and October of the previous year, together with the December rains of the previous year, promote tree regeneration. This is because, although forest fires can occasionally create adequate conditions for seed germination and tree survival [60], adequate post-fire climate conditions are required [57], such as energy and humidity [61].

4.6. Management Considerations

The fire history characterized in the present study revealed a frequent low-intensity fire regime, which has contributed to the current forest structure. However, continuous recruitment is frequently impossible without major disturbance events [21]. This reflects the importance of both frequent low-intensity fires and infrequent high-intensity fires in the study area.

Since 1980, the roles of fire in several forest ecosystems and the role of humans in modifying fire regimes have been demonstrated in Mexico [62]. In this sense, it has been mentioned that for SMO forests with a fire regime such as the one observed in the present study, the exclusion of fires over long periods of time should be avoided to prevent the build-up of combustible materials that promote larger and more severe forest fires [40].

Climate change projections indicate a trend toward higher severity and frequency of fires [63]. In this sense, the application of novel fire management practices, such as prescribed fire, represents a suitable option in the face of such projections [7,11,62], since proper use of fire at appropriate intervals can serve to restore and maintain healthy forests [15].

On the other hand, a decline in tree regeneration and changes in the ecosystem after fires, especially high-severity fires, have also been foreseen [64]. In this regard, it has been suggested that the establishment or planting of trees in areas recovering from recent disturbances may help to delay the projected impacts of climate change [65].

In Mexico, the General Law on Sustainable Forest Development states that: "*...in the event of a fire, the legitimate owners of forest land are obliged to carry out the restoration of the affected area within two years maximum*" [66]. However, the roles of natural and managed regeneration in promoting forest recovery are little known, highlighting the need to strategically target economic, personnel, and time resources to effective reforestation [67]. It is our view that, after a fire, trees should be planted in an optimal time, considering the key factors for the regeneration and survival of the species, for example, those related to the nature of the fire, the post-fire environmental conditions, and the life-history characteristics of the dominant tree species before and after the fire [54,68], which could take a longer time for implementation.

5. Conclusions

This study reconstructed the fire history over the past 165 years, during which 41 fire events occurred. This fire history is associated with a regime of frequent low-severity fires and infrequent extensive fires, which together have shaped the structure of the current stand. The frequency of fires was driven by local and regional climatic conditions. The establishment of tree species was influenced by the frequency of fires and climate conditions in the years after each fire.

These findings contribute to the knowledge of the forest fire regime and its historical influence on vegetation, which will set the basis for better planning of fire management in the study area and the conduct of further studies addressing this topic in the region.

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CHAPTER V: GENERAL CONCLUSIONS

Fire drives forest structure patterns in the study region. Fire severity did not show relationship with species diversity but indicated a negative relationship with structural diversity, being different only in areas affected by high severity fire, which tended to homogenize forest structure, making it more prone to disturbances, including fire.

The characterization of forty-one fires that have occurred in the last 165 years in the study area is associated with a regime of frequent and low-severity fire regime and infrequent presence of severe fires, which acting together have shaped the current stand structure. In this study, fire frequency was driven by local and regional climatic conditions where the establishment of tree species was influenced by the fire frequency and dominant climatic conditions present in the years following each fire.

These results represent an approximation of the forest response to fire severity and the historical influence of fire. These findings may support the development of similar studies in the Sierra Madre Occidental. In addition, these findings may be useful to for proper fire management in the study area, where preventive actions to avoid high-severity fires, such as prescribed burns to reduce fuels can contribute to the stability of the dominant forest communities. As in other studies, it is proposed that pyrosilviculture can be an interesting and helpful tool for forest management to be implemented, but further studies could be carried out to evaluate the application of fire as a silvicultural tool in the study area.