Forest Ecology and Management 509 (2022) 120083

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Post-fire ecological restoration in Latin American forest ecosystems: Insights and lessons from the last two decades

Pablo Souza-Alonso [a,b,c,*], Gustavo Saiz [b], Rafael A. García [c], Aníbal Pauchard [c,d], António Ferreira [e,f], Agustín Merino [a]

[a] University of Santiago de Compostela, Escuela Politécnica Superior, Department of Soil Science and Agricultural Chemistry, Lugo, Spain
[b] Facultad de Ciencias, Universidad Católica de la Santísima Concepción, Concepción, Chile
[c] Laboratorio de Invasiones Biológicas (LIB), Facultad de Ciencias Forestales, Universidad de Concepción, Concepción, Chile
[d] Institute of Ecology and Biodiversity (IEB), Santiago, Chile
[e] CERNAS—Study Center for Natural Resources, Environment and Society, Agricultural School-Escola Superior Agrária de Coimbra (ESAC), 3045-601 Coimbra, Portugal
[f] Polytechnic Institute of Coimbra (IPC), Agricultural School (ESAC), 3045-601 Coimbra, Portugal

ARTICLE INFO

Keywords:
Wildfires
Forest degradation
Postfire restoration
Natural/assisted restoration
Social participation
Socio-economic limitations

ABSTRACT

Wildfires are responsible for a substantial loss of forest ecosystem services globally and represent a major driving force of forest degradation across Latin American and the Caribbean (LAC). The detrimental effect of forest fires is particularly relevant in regions where fire has been historically absent or has rarely occurred. Nowadays, there is an increasing interest to promote and develop ecological restoration (ER) following fire. LAC constitutes a hotspot where work and interest in ER has been steadily increasing over the last decades, mainly due to the drive of a new generation of young and experienced ecologists and foresters.

Despite the increasing attention in post-fire restoration in the region, there is a dearth of initiatives compiling and organizing all the available information on this topic. This work aims to address such constraint, providing current information on post-fire ER in LAC forests. After a brief contextualization of environmental and social consequences of wildfires, we collect and discuss recent advances on restoring degraded forests. From the conifer Mexican ecosystems to the Southern Patagonian evergreen forests, we look back over the last two decades (2000–2020) mainly discussing experiences of success and failure, as well as limitations of implementing approaches based on passive/natural restoration or active/assisted restoration. Furthermore, we also explore other aspects of the restoration process, including those related to social participation and community engagement (e.g. education in restored areas), the use of fire regulation and management to reduce fire risks and increase ecosystem resilience, educational aspects and intermediate approaches as agroforestry and silviculture practices. In the last sections, we identify three major categories of specific constraints that condition ER, including environmental limitations (biotic and abiotic factors), technical/management factors and the socio-economic challenge of restoration. Finally, we briefly discuss future perspectives for ER in LAC.

1. Introduction

From the mixed conifer ecosystems in the highlands of Mexico to the Southern Patagonian evergreen stands, forests cover nearly a billion hectares across Latin America and the Caribbean (LAC), making up a significant share of Earth's aboveground biodiversity, providing essential functions and natural resources and playing an essential role on climate regulation at the regional and global scale. We are currently witnessing a progressive degradation of virtually all biomes on Earth that is mainly caused by human-driven factors (Bradshaw et al., 2021). Specifically, anthropogenic wildfires are responsible for a substantial loss on global forest areas. Acting alone or synergistically with deforestation, agricultural and livestock pressure, invasive species introduction, forestry plantations, habitat fragmentation, or climate change, wildfires represent a driving force of forest loss through LAC. It is also worth noting that, even in those environments where fire has naturally shaped the landscape, human intervention in the form of fire suppression may also have serious consequences on ecosystem services and their

* Corresponding author.
E-mail addresses: souzavigo@gmail.com, pablo.souza@usc.es (P. Souza-Alonso).

https://doi.org/10.1016/j.foreco.2022.120083
Received 3 September 2021; Received in revised form 23 January 2022; Accepted 2 February 2022
Available online 10 February 2022

own functioning.

Considering that causes, triggers, and their intricate interrelationship represent a complex field of study and that the variables are specific to each particular biome or region, our work will focus on post-fire stages and discuss the different strategies that have been applied for the different scenarios of postfire restoration. However, this effort requires a short overview of the historic and current scenario, and the problem of forest fires, together with the causes and consequences across different biomes in LAC. Then, we aim to collect and disentangle recent advances on restoring degraded forest ecosystems after wildfires in LAC following the precepts of ecological restoration (ER). We consider all aspects of natural or assisted restoration but also those aspects related to social participation and community engagement to provide a general vision of how the discipline has moved forward in different parts of the continent. To do so, we put special emphasis on those works published in the last two decades (2000–2020) since ER during this time has witnessed a significant increase in global attention. Finally, we identify some deficiencies that may compromise restoration outcomes and discuss them along with environmental, ecological, and socioeconomic constraints that may limit its widespread use.

2. Fires in LAC forest ecosystems

2.1. Forests in LAC

The complexity of forests goes beyond a mere population of trees growing on the land. Forests represent the habitat of a substantial portion of the terrestrial flora, fauna, and microorganisms, which make them key environments for biodiversity conservation. Forests are also critical in the regulation of global climate and provide a myriad of other essential functions, which include the provision of natural resources, regulation of the hydrological cycle, watershed protection, air quality, and recreational purposes (FAO and UNEP, 2020).

Coinciding with the vast biogeographic realm *Neotropic* defined by Olson et al. (2001), **Latin America and the Caribbean (LAC)** conform a vast territory in which forests make up about half of its total area (FAO, 2020). LAC forests cover nearly a billion hectares and stretch from the mixed conifer ecosystems in the highlands of Mexico to the Southern Patagonian evergreen forests, the most meridional forest ecosystem in the world. Altogether, LAC forests represent about a quarter of the world's forest area (FAO, 2020). Administratively, LAC includes mega-diverse countries with the greatest biological forest diversity on Earth and hosts 57% of the primary forests globally (FAO and UNEP, 2020). The *Neotropic* encompasses several biomes, which include humid tropical forests (evergreen broadleaf, semi-humid broadleaf), dry tropical forests (deciduous, semi-deciduous, and semi-deciduous transitional), and temperate forests (evergreen broadleaf, evergreen mixed, and deciduous), distributed into lowland, premontane and montane ranges (Eva et al., 2004). The carbon (C) stored in the biomass in LAC forests is over 100 gigatons, which makes these ecosystems highly significant in terms of their contribution to the global C cycle (UNEP-WCMC, 2016).

2.2. Recent trends in forest cover in LAC

Environmental conditions and soil properties play a fundamental role in the type, structure, and geographical distribution of forest vegetation (Holdridge, 1947; Saiz et al., 2012; Veenendaal et al., 2015). Since the onset of the Holocene, variations in forest cover in LAC have not just responded to climate-related forcing (Maksic et al., 2019), but to ever-increasing anthropogenic pressures that have led to massive landscape transformations in the region, particularly over the past few centuries (Armesto et al., 2010). Indeed, anthropogenic-related factors are greatly responsible for the broad variation in forest surface through the region, where the area currently covered with forest ranges from 9.5% in Uruguay to 96.5% in the French Guiana (Giri and Long, 2014).

Deforestation in LAC during the first decade of the 21st century was ~5.20 million ha, which is equivalent to an area slightly larger than the size of Spain. While this trend is still negative, the rate of forest loss in LAC has declined to ~2.6 million ha during the second decade of this century (FAO, 2020). Causes of deforestation are often multiple and complexly interlinked. These include agricultural expansion, particularly in the form of pasture and feed crops grown to meet the increasing global demand for meat, wood procurement, population growth, land tenure insecurity, and poor governance (Aide et al., 2013; Manners and Varela-Ortega, 2017). While deforestation has been the major force behind changes in woody cover in LAC, particularly in moist and dry forests as well as in savanna/shrubland ecosystems, there have also been regions that have experienced a recovery in woody vegetation as a result of reduced anthropogenic pressure, increased precipitation, and CO_2 fertilization (Aide et al., 2013).

Besides natural forest regeneration, land-use changes in the form of forest plantations also contribute to the total forest cover, accounting for about 2% of the total forest area in South America. However, 99% of these forest plantations in the region are fast-growing monocultures intensively managed for commercial purposes (FAO, 2020). While highly productive, the forestry species typically chosen are non-native species commonly associated with the occurrence of wildfires (Úbeda and Sarricolea, 2016).

2.3. Fires in LAC forests

2.3.1. Historical occurrence of fire in LAC forests

Fire has occurred on the Earth's surface ever since there has been a combination of biomass, atmospheric oxygen levels above the 16–19% threshold, episodically low moisture conditions, and an ignition source such as lighting (Scott et al., 2013). These environmental conditions have been met for millions of years and have certainly contributed to shaping all terrestrial landscapes. The fossil record shows that there have been very large fluctuations in fire occurrences throughout geological times (Scott et al., 2013).

Charcoal records from tropical South America show relatively low numbers of fire events during the Last Glacial Maximum (LGM; 18–24 cal ka BP) (Mayle et al., 2009). Besides promoting an overall reduction in biomass, the inherent cool conditions of this period would have also limited the occurrence and spread of fires. The archaeological record also shows an augmented presence of fire events as soon as humans populate any region, which in the case of tropical South America seem to date back to the late (terminal) Pleistocene period that followed the LGM (Arroyo-Kalin, 2012). This is well reflected in the overall increase in the abundance of pyrogenic C in numerous sedimentary records that reflects the increasing use of fire by humans (Bird et al., 2015). The onset and gradual increase of human populations in the LAC region led to the widespread use of fire for agricultural practices, but also to the establishment of settlements, procurement of timber, hunting, etc. Indeed, it was through these activities that humans achieved increasing control over fuel loads and landscape connectivity (Archibald et al., 2012). Therefore, human-induced fires have been a considerable force shaping the landscape in LAC, even in some of its most undisturbed ecosystems, the Amazonian forest (Mayle et al., 2009; Arroyo-Kalin, 2012). Furthermore, the sheer scale of land use changes occurring in LAC has effectively altered the burning frequency and severity of ecosystems where the presence of fire has been fairly uncommon (e.g., temperate *Araucaria* forest; González, 2005). Large fires have also played a significant ecological role in shaping even the wettest Southern LAC forests in the past centuries (Veblen et al., 2008). Charcoal material deposited during the Holocene indicates large spatial heterogeneity across the South American continent, but in general, it could be inferred a shift toward higher fire occurrences over time (Power et al., 2008). Nevertheless, as is the case in many other aspects in LAC, its broad biocultural diversity underlies large differential patterns in the way landscapes are managed throughout this vast region (Armesto et al., 2010).

2.3.2. Current scenario and future prospects of forest fires in LAC

Wildfires alone have been responsible for nearly one-fourth of the global forest loss reported for the 1990–2015 period (Curtis et al., 2018). Nearly 100 million ha of forest burned in the year 2015 alone, which is about 4% of the global forest area, affecting tropical ecosystems (2/3 of the total), particularly in Africa and South America (FAO, 2020). While these are highly concerning figures, up until very recently the burned area in LAC was not increasing as much as in other world regions (i.e. Southern Africa, Southeast Asia, or Australia) (Giglio et al., 2013). However, the calamitous Amazonian fires in 2019 (Lizundia-Loiola et al., 2020) are solid proof of low fire regions being transformed into fire-prone areas (Davidson et al., 2012; Jia et al., 2019). Three broad types of fires can be distinguished in the Amazon: deforestation fires (fire after clearing), fires in previously cleared areas, and fires affecting standing forests (Barlow et al., 2020). Spatially tied to human land use, deforestation, habitat fragmentation, and climate change act synergistically through the Amazon basin to increase fire risk (Cochrane and Barber, 2009).

Outside the tropical realm, recent trends in forest fires in LAC have not been much better. For instance, the number of wildfires in Chile has increased significantly in the past 30 years (Úbeda and Sarricolea, 2016). More recently, over 500,000 ha burned in the central regions of Chile in just over a month during the austral summer of 2017, in what was considered the biggest wildfires in the country's history. The proportion of forest cover affected in those mega-fires was 60%, including both native forests and commercial forest plantations (de la Barrera et al., 2018). In the northern tip of LAC, the heterogeneous landscape of Mexico also presents numerous fire events in forest ecosystems, particularly in mid-elevation coniferous forests (Zúñiga-Vásquez and Pompa-García, 2019).

2.3.3. Causes of forest fires in LAC

Within the current context of global environmental change, both human pressure (through land cover and land use) and global warming are playing a growing role in determining wildfire regimes with future climate variability expected to enhance the risk and severity of wildfires in many biomes (Liu and Wimberly, 2016; Jia et al., 2019). Indeed, changes in land cover, lightning activity, and meteorology contribute to increasing fire occurrence, projecting a dramatic scenario of higher fire frequency under the 2050 conditions (Huang et al, 2015). In addition to fire frequency, virtually all forests ecosystems have experienced a significant increase in the length of fire weather season (Jolly et al., 2015).

Wildfires may be considered the result of social, economic, and biophysical factors operating with feedbacks and interactions across spatial scales (Sorrensen, 2009). The combination of recurrent droughts, promotion of fire-prone vegetation (i.e. forestry plantations), and increasing anthropogenic land pressure promote the occurrence and spread of forest fires in temperate regions of Chile and Mexico (Úbeda and Sarricolea, 2016; Zúñiga-Vásquez and Pompa-García, 2019). Furthermore, tropical forests are also suffering from altered fire regimes, which pose a fundamental threat not only to the many environmental functions they serve but to their very existence (FAO 2020; FAO and UNEP, 2020). However, there is still great uncertainty about how climate change and increasing anthropogenic pressure will exactly affect fire regimes across the various forest biomes, and the likely enhanced detrimental impacts that these burns will cause on the different ecosystems (Krawchuk et al., 2009).

Besides climate-related factors, other factors that dramatically increase the occurrence of forest fires include land clearing for agricultural purposes, proximity to human settlements, land tenure disputes, and forestry practices that promote the accumulation and connectivity of high fuel loads. The fires raging across the Brazilian Amazon and surrounding areas accumulate the highest rates of fire events for the last decade in South America (Chen et al., 2013) and capture a substantial part of the world's attention. Here, fire represents a traditional tool for forest clearing and land preparation, with decisions about when and how to use it being taken locally by landowners. The use of fire to convert dry and humid forest areas for agriculture is concentrated in the 'arc of deforestation' along the southern and eastern edges of the Amazon in Brazil and Bolivia, increasing the likelihood of fire escape into standing forests (Chen et al., 2013), and allowing fire expansion into historically intact areas (Barlow et al. 2020). Serving as a direct footprint of land-use change, Cardil et al. (2020) recently evidenced a widespread sequence of fire events in tropical moist forests immediately after deforestation. Without the same media relevance, agricultural and livestock pressure have transformed the dry tropical forests of the Chaco, one of the last large contiguous areas of dry tropical forest in the world (Kernan et al. 2010), into one of the regions with the highest rate of deforestation linked to agriculture expansion (Hansen et al., 2013). On a much smaller scale, fire also represents a source of tension within communities with ongoing competing interests. Furthermore, intentional wildfires are occasionally related to conflicts between the local population and those interested in resource exploitation (Celentano et al., 2018).

The structure and type of woodland cover are also strong determinants affecting forest fire regimes regionally (Úbeda and Sarricolea, 2016). Currently, the expansion of tree cover in the central provinces of Chile is primarily the result of large monoculture plantations, mainly composed of flammable species that have been associated with fire severity and propagation (González et al., 2020), which have resulted in a dramatic increase in wildfire events (de la Barrera et al., 2018). Fires affecting understory forest vegetation are also common in tropical LAC and depend on multiple variables that strongly determine its flammability and ignition exposure. These include the size of the forest stand, proximity to the forest edge, and distance to charcoal pits, agricultural settlements, and cleared paths (Alencar et al., 2004).

3. Post-fire effects/consequences on forest ecosystems

It is important to distinguish between two main concepts routinely used in wildfire research: fire *intensity* and *severity*. Used as synonyms for a long time, there is a clear difference between both concepts with important consequences on post-fire assessment. While fire *intensity* describes the physical combustion process of energy release from organic matter, fire *severity* (or more properly, *burn severity*) describes how the burning process affects ecosystems, mainly based on soil organic matter (SOM) loss and aboveground OM conversion to ash (Keeley, 2009). Although both terms usually correlate in the field, many restoration studies refer to a wildfire *intensity* classification when dealing with fire consequences. For the purposes of this work -and restoration in general- fire consequences in terms of burn severity appears more relevant than intensity. Therefore, we will refer hereon to fire severity.

3.1. Soil conservation after wildfires

The characteristically high temperatures and heat transfer caused by biomass combustion during wildfires affect all levels of soil organization, including its structure, porosity, infiltration, thermal regime, water storage, pH, OM content, and nutrient availability (Neary et al., 1999, 2005; González-Pérez et al., 2004; Certini, 2005; Saiz et al., 2018). The impact of temperature varies with soil depth (temperature effects dissipates after a few centimeters) and with soil moisture (water acts as a buffer) (Ferreira et al., 2008). Soil temperatures experienced during biomass burning strongly determine the post-fire evolution of forest ecosystems. Post-fire impacts include the significant alteration of soil properties (key for soil conservation), the death of seeds and rhizomes, and potentially strong impacts on the composition of the soil microbiome. Furthermore, soil erosion rates may be high in burned forest ecosystems, particularly in mountainous ranges and areas with pronounced slopes. These losses are the result of soil structural changes, a decrease in infiltration capacity, water repellence, and the enhancement of detrimental hydrological processes.

Soil burn severity (SBS), defined as the degree of SOM loss, is dependent on the type of forest ecosystem. Forest typology poses a strong influence not only on fire categorization and behavior but also on soil moisture and thickness of the soil organic horizon. These factors determine the degree of perturbation in soil properties, thus greatly influencing soil conservation and the subsequent evolution of vegetation. However, despite the direct influence of soil properties on forest resilience and the large diversity and extent of LAC forest ecosystems, post-fire variations in soil physical and chemical properties have not been explored in much detail compared to fire-prone areas of Europe, Australia, or the USA (Meira-Castro et al., 2015, Muqaddas et al., 2015, Pingree and DeLuca, 2018).

Research work conducted in LAC shows that wildfire impacts on soils of temperate and subtropical forests with a moderately developed organic horizon are highly variable, as revealed by studies carried out in stands dominated by *Pinus* and *Quercus* (Capulin-Grande et al., 2018; Hernández Vallecillo et al., 2020), *Araucaria* (Santana et al., 2020a), or *Austrocedrus* and *Nothofagus* (Urretavizcaya, 2010; Urretavizcaya et al., 2018). A similar situation can be seen in fire-prone tropical forests (Quintero-Gradilla et al. 2020) or in the gallery forests of Cerrado (Gomes et al., 2018; Pivello et al., 2010). The high variability in fire temperature across ecosystems is strongly determined by a combination of season, fire behavior, and fuel load (Saiz et al., 2015). While in the dry season both moderate and high levels of SBS cause substantial negative impacts on soils, fires tend to be cooler during the rainy season and the beneficial fertilizing effects of ashes become more apparent.

In tropical rainforests, notwithstanding the massive clearance of land that occurred over the past few years, most of the fires are the consequence of the slash-and-burn agricultural practices widely employed across relatively small areas. Due to their typical wet conditions (abundant precipitation and high humidity both in biomass and soil), rainforest fires are usually of low intensity (Cochrane et al., 1999), and therefore changes in soil properties are low to moderate (Béliveau et al., 2015; Juárez-Orozco et al., 2017). Similarly, tropical mountain forests hosting peatlands show high soil humidity levels, which usually limits any strong perturbations in their soil physicochemical properties (Román-Cuesta et al., 2011), and edaphic microbiota (Torres Vargas et al., 2004).

Forest stands recently burned present harsher soil conditions (increased light intensity, soil temperature, and enhanced evaporation) for sustaining life compared to unaffected areas (Lippok et al., 2013). Once the forest is fragmented, the density and the average height of the leaf canopy is reduced, which favours rapid evaporation of soil surface water, increasing fire susceptibility, particularly if the vegetation is not fire-adapted (e.g. tropical rainforest, Ray et al., 2005). This progressive loss of forest cover promotes drought events at regional scale, which together with the increasing anthropogenic ignition sources, have caused widespread tree mortality and forest degradation across south-eastern Amazon rainforests (Brando et al., 2014). Humid tropical forests (evergreen rainforests) are resilient to initial disturbances, but if these are sustained, forest structure and nutrient dynamics may get significantly altered, potentially leading to long-term changes in vegetation composition (Davidson et al., 2012). The linkage between fire trends and the hydrological cycle is reflected in the recent Amazonian droughts that have been shown to fuel wildfires (Aragão et al., 2007, 2018). Also, it has been reported that widespread fires in *Nothofagus* forests depend on drought at monthly, seasonal, annual, and supra-annual time scales (Veblen et al., 2008).

Fire effects on soil properties depend on many factors such as the type and intensity of the burn, soil moisture conditions, soil type, and the nature of burned biomass (González-Pérez et al., 2004; Certini, 2005; Santín and Doerr, 2016). On the one hand, short-term increases in SOC and nutrient content have been reported in savannas and evergreen seasonal forests (Nardoto and Bustamante, 2003; Ivanauskas et al., 2003; Saiz et al., 2015), attributed to the partial combustion and degradation of litter and biomass. Although positive, the rapid increase in nutrient availability also makes them more prone to potential volatilization and export, diminishing soil nutrient stocks and ecosystem resilience to future disturbances. On the other hand, medium and long-term reduction in soil moisture and nutrient pools (mainly C, N, and P) has been observed following fire events in temperate (Alauzis et al., 2004; Nave et al., 2011) and tropical forests (Ivanauskas et al., 2003). In other cases, such as in tropical or subtropical dry forests (Cerrado, Caatinga), their low and sparse vegetation presumably prevent high values of SBS (Roscoe et al., 2000; Pivello et al., 2010).

Forest fires are responsible for a substantial loss of C in LAC (Van der Werf et al., 2017; Aragão et al., 2018) and can produce a vicious cycle, induced by positive feedbacks, where a high release of C and other greenhouse gases as a result of combustion and ash erosion may represent a significant increase in overall C emissions. These losses reduce environmental services provided by forest ecosystems, namely in their ability to act as net C sinks. Typically, the new equilibrium following severe and frequent fires occurs in the form of degraded ecosystems with much less capacity to retain C in both the soil and the aboveground vegetation. While a significant share of aboveground C biomass may get transferred to the atmosphere during combustion, fires also promote C shifts between terrestrial pools and alter the physicochemical nature of thermally-affected C compounds, which impact the quantity and quality of C stored in the soil (González-Pérez et al., 2004; Merino et al., 2014; Saiz et al., 2015). Besides the direct effect on SOC stocks, recurrent fire events also limit the capacity of soils to be restored. Zarin et al. (2005) analyzed 93 stands across the Amazonian region burned in the last 30 years and showed that stands that suffered $\geq$ 5 fire events showed a significant reduction (>50%) in C accumulation. These authors forecast a scenario of increasingly degraded fire-prone landscapes in case no intervention is implemented. The time needed for forest ecosystems to regain C losses following a fire event depends on their capacity to recover and re-establish vegetation structure (resilience). For instance, using estimations of losses and C stocks for the central region of the Andean Patagonia, Bertolin et al. (2015) projected an interval of 100–200 years to recover former C stocks in fire-affected forests.

Finally, the negative impacts caused by forest fires do get exacerbated when other detrimental activities take place shortly after. Recurrent fires followed by intense heavy cattle browsing have led to a new type of disturbance regime in the otherwise stable ecosystems like those of Northern Patagonia (Blackhall et al., 2008; Raffaele et al., 2011). In the mesic, high-mountain rangelands of Córdoba (Argentina), the common presence of livestock increased 50% soil loss after fire, driving this system into a rocky desert (Cingolani et al., 2013). The combined presence of fire and cattle has created a new scenario whereby variations in vegetation structure and composition pose a strong potential to hamper the stability of forest ecosystems (Blackhall et al., 2015).

3.2. Changes in the plant community

Intrinsic fuel properties of vegetation, namely moisture, ignitability, and the heat released during combustion, affect fire frequency, intensity, seasonality, and SBS (Mandle et al., 2011). Reciprocally, woodland and forest structure are largely controlled by fire history. In addition to the charred appearance, changes in vegetation structure and composition are the first evidence of recently burned ecosystems. In general, fire promotes the herbaceous layer by increasing plant density and species richness, thus creating a more divergent matrix of vegetation in terms of species composition, abundance, and secondary forest specialists (Ribeiro et al., 2010). Also, fire recurrence favours the presence of plant populations with small-sized stems (Cesca et al., 2014).

The revegetation rate will depend on the SBS and fire history. In this sense, there is an important distinction between fire-adapted (e.g. gallery forest in the Cerrado) and non-adapted ecosystems (e,g. tropical rainforest). In fire-adapted ecosystems (dry forest, Mediterranean forests) low severity fires maintain beneficial plant ecological attributes -fire adaptations- that evolved in accordance: e.g. the genus *Pinus* (Rodríguez-Trejo and Fulé, 2003), *Quercus* (Rodriguez-Trejo and Myers, 2010) or species from the Cerrado (Simon and Pennington, 2012) or the Chaco Serrano Forest (Torres and Renison, 2017). In addition, the availability of seeds is crucial to ensure revegetation. The capacity of the seed bank to resist fire severity also vary across LAC forests reflecting their degree of adaptation to fire events. These range from adapted seed banks in the Chaco forests (Jaureguiberry and Díaz, 2015; Lipoma et al., 2018), to intermediate in the Chilean matorral (Gómez-González et al., 2017), up to sensitive seedbanks in the Amazonian rainforest (Cochrane et al., 1999) or the Atlantic Forest (de Silva and Matos, 2006).

The current increase in wildfire incidence produces changes at the ecosystem level in regions where fire is occasional, or it has been historically absent (tropical, temperate forests). Here, plant communities are more vulnerable to postfire degradation and invasion by invasive alien plants (IAPs). In many cases, light and nutrient availability in burned areas create novel niches rapidly occupied by pioneer species, as IAPs (see also Section 5.1). When the fire frequency, intensity, or severity exceeds historical regimes, native vegetation tends to be displaced, since fire selectively excludes sensitive species (Hoffmann and Moreira, 2002; Brooks et al. 2004; Gomes et al., 2014; Urrutia-Estrada et al., 2018). This fact generates favourable conditions for the colonization and establishment of exotics (Zouhar et al., 2008), progressively reducing species diversity and promoting ecosystem homogenization (Libano and Felfili, 2006).

Even resilient forests are experiencing changes in their structure, composition, and dynamics as a result of increasing wildfire frequencies. Such is the case of *Nothofagus* forests in Chile, where environmental proxies spanning over 3,200 years reveal that the persistence of dense forest stands has been largely unaffected by lower frequency wildfires (Simi et al., 2017). Lack of adaptation to new fire regimes (natural or human-induced) or the occurrence of major disturbances (logging, windthrows, etc) favours the presence of transitional stages (second-growth forest) dominated by a mix of different species (González et al., 2015). Synergistic trends between economy, agriculture, forests, and climate in the Amazon Basin could lead to the replacement or severe degradation of a significant proportion of the closed-canopy forests, promoting a large-scale substitution by savanna-like vegetation (Nepstad et al., 2008). Considered as a transitional state in ecological succession or a degraded state from original evergreen sclerophyllous vegetation, more open savannas represented by early successional species (e.g. *Acacia caven*) dominate secondary forests in South America (Schulz et al., 2010). In extreme cases, fire-promoted *Acacia*-dominated savannas may be further threatened if fires become recurrent, leading to barren landscapes eventually (Van de Wouw et al., 2011).

3.3. Effects on soil fauna and terrestrial biota biodiversity

Forest fires affect soil biota directly through impacts derived from high temperatures, burning, or gases produced during combustion, as well as indirectly as a result of secondary post-fire changes (Cochrane et al., 1999; Neary et al., 2005). Although fires can eliminate every living organism above a temperature threshold, the topsoil microbial activity generally increases due to the rapid release of nutrients from SOM (Fuentes-Ramirez et al., 2018). Nevertheless, belowground changes on key ecosystem elements, e.g. mycorrhiza, will promote different plant growth responses (Allen et al., 2003) and therefore, community assemblage and succession during post-fire restoration.

While the type of disturbance affects the rate of change in the soil community, it is also important to consider the scale, intensity, and severity of such disturbance, as well as any potential vegetation replacement (Allen et al., 2005). Although aboveground fauna is not as physically constrained as plants or soil biota, they are affected by fire-induced changes. For instance, changes in plant species composition reduce large frugivores and other vertebrates, promoting a transition between primary forest specialists to species associated with second-growth forests and other disturbed habitats (Barlow and Peres, 2006). In non-adapted forest ecosystems of Southern Chile, fire reduces diversity and disrupt the assemblage of trophic networks of rodents (Zúñiga et al., 2021), carnivorous mammals (Zúñiga et al., 2020), or saproxylic beetles (Tello et al., 2020). Nevertheless, ecosystems with a long history of fire disturbance are also exposed to fire consequences. In the Serrano Forest in Argentina, high-severity fire regimes transformed original forests into dense grasslands which resulted in a severe reduction in the species present (70%), and causing structural changes in the avifauna community (Albanesi et al., 2014). Fire modified the community composition of fruit-feeding butterflies in a South-eastern Amazon Forest in Mato Grosso (Brazil) (de Andrade et al., 2017). Here, a single fire event altered the community composition, increasing the presence of drought-tolerant savanna specialists in contrast to forest specialists. In the Iberá Natural Reserve, a mosaic of habitats in Northern Argentina, high-intensity fires had a modest and short-lived negative effect on the abundance of ant species with some functional groups being affected (Calcaterra et al., 2014).

4. Soil protection and ecological restoration after wildfire

4.1. First steps in post-fire management

4.1.1. Preliminary damage assessment and local considerations

Following a wildfire, the early evaluation of SBS in soil and vegetation will condition restoration. Also, the estimation of erosion risk, threatened resources, and prioritization of action areas complement the set of immediate actions to be considered. Such needs will be further refined considering local information like land-use history, environmental context, restoration goals, and available resources, which should all be clarified before any restoration effort (Holl and Aide, 2011; Bannister et al., 2016).

The selection between active or passive/assisted restoration depends on the balance between ecological needs and available resources. Urgent intervention is often necessary in cases where of pronounced slopes, high SBSs, and/or imminent rains threaten with the occurrence of extremely high soil erosion rates. However, these interventions usually represent costly operations, which are rarely budgeted for. Economic investment in ER makes a difference in relatively short periods (Blignaut and Aronson, 2020), but each intervention relies on evaluating the necessity and feasibility. Moreover, the level of intervention is often questioned since ecosystems have the capacity for self-recovery (Chazdon and Guariguata, 2016). While passive restoration may be preferable in many instances for ecological, practical and economic reasons, it is mainly effective if soils have not been degraded, abiotic limitations are not severe, or where biotic support is available.

4.1.2. Urgent interventions required to protect soil against erosion

Soil degradation and high erosion rates following wildfires, particularly if SBS is high, require urgent measures to protect soil against an irreversible degradation that hampers ecosystem resilience and future restoration efforts (Vega et al., 2013a). SBS can be assessed *in situ* by visual indicators (Vega et al., 2013a), through remote sensing imagery (Holden et al., 2010), or by analyzing field-collected samples through elemental analysis coupled with isotope ratio mass spectrometry (EA-IRMS), pyrolysis-gas chromatography coupled to mass spectrometry (Py-GCMS), or nuclear magnetic resonance spectroscopy (NMR) (Santoiemma, 2018; De la Rosa et al., 2019).

Mulching, erosion barriers, soil scarification, slash spreading, or seeding exemplify urgent interventions to stabilize burned areas and prevent or reduce fire adverse effects (USDA Forest Service, 2012; Vega et al., 2013b; Ferreira et al., 2015). These urgent field interventions generally start by preventing soil erosion and promoting the accumulation of SOM (Alanís-Rodríguez et al., 2015). Emergency post-fire erosion control is often reported alone, without an overall strategy that implements complementary ecological schemes. Precisely because of its nature, emergency actions are not usually considered as part of restoration, which requires a planned long-term vision. Consequently, the scientific literature addressing the effects of rapid interventions on restoration is not abundant in LAC.

4.2. Situation is stabilized. Now what? The basics of ecological restoration

While in art disciplines restoration involves recapturing an object's aesthetic value, this concept has much broader implications in ecology. ER can be defined as the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed, and returning it to its historical trajectory (SER, 2004). In the last decades, we have witnessed large efforts promoting and developing ER as a permanent part of our conception of long-term sustainability (Aronson and Alexander, 2013). This has led to the declaration of 2021–2030 as the *Decade on Ecosystem Restoration* (United Nations, 2019), or to striking statements claiming that "the future of the planet depends on the maturation of ER" (Roberts et al., 2009).

From an intuitive point of view, ER represents the application of ecological principles to restoration. Simple to enunciate, complex to describe. ER aims to restore the structure, content and functioning of the degraded ecosystem, not necessarily to its previous state but to the fullest possible extent (Blignaut and Aronson, 2020). This is based on principles that must be constantly adjusted according to the level of disturbance and local conditions (Clewell and Aronson, 2013). On a global scale, it seems clear that restoration in different biomes demands different strategies (Clewell and Aronson, 2013). Nevertheless, recovering forests is not just about planting trees. Indeed, our capability depends on the understanding of local ecological processes, the analysis of the potential for natural recovery, and foreseeing the outcomes of different restoration options (Bannister, 2015). Then, based on available information and after the identification of SBS, restoration efforts could be theoretically implemented to return ecosystems to their normal successional paths (Fig. 1).

4.3. Ecological restoration in LAC

In the last decades, the growing interest in restoring degraded ecosystems -including those affected by fire- throughout LAC has significantly increased the number of publications (Fig. 2). Furthermore, this interest has led to the creation of restoration networks that promote research, exchange experiences, develop capacities and communicate advances on ER as REPARA (Red Mexicana para la Restauración Ambiental), REDCRE (Red Colombiana de Restauración Ecológica), Restauremos Chile (Red Chilena de Restauración Ecológica), REA (Red de Restauración Ecológica de la Argentina), SOBRADE (Sociedade Brasileira de Recuperación de áreas degradadas), or international societies as the SIACRE (Sociedad Ibero-Americana y del Caribe para la Restauración Ecológica) (Echeverría et al., 2015).

Recent progress and innovations in national ER projects have been compiled in Colombia (Murcia and Guariguata, 2014), Mexico (Calva-Soto and Pavón Hernández, 2018; Méndez-Toribio et al., 2018) or Chile (Smith-Ramírez et al., 2015). Although these works pointed out that efforts to protect natural values are increasing and ER is gaining attention, information concerning post-fire restoration is still scarce. In a recent study, Mexican managers recognized fire as a limitation for ER, but it was not identified as a disturbing agent in restoration projects (Méndez-Toribio et al., 2018).

4.3.1. The importance of assessing fire ecological restoration in LACs

The presence of contrasting socio-economic realities, land degradation scenarios, or the existence of highly valuable pristine or semi-pristine environments illustrate a different scenario for ER in LAC compared to Europe or North America, where most restoration activities are carried out (Armesto et al., 2007). The large area represented by LAC forests (1/4 of the world's forest area), the wide variety of biomes and forest types covered in the Neotropic realm, the large pool of C stored, the increasing perception of vulnerability to fires, and the need to preserve community resources and maintain or promote ecosystem resilience, justify the study of the current state of ER in LAC. As far as we know, works summarizing ER in this vast and highly diverse region are still absent. Therefore, we performed a literature search in SCOPUS, Web of Science (WOS), and Google Scholar using the terms *forest* and *fire* and *restoration*, and selected those articles based on the match between title and abstract information and the subject of the study. A significant proportion of information originated from 4 countries: Mexico, Brazil, Chile, and Argentina, which accounted for over 80% of the papers consulted.

We included works aimed at restoring (insofar as possible) the characteristics of natural ecosystems (SER, 2004, Clewell and Aronson, 2013; Gann et al., 2019), collecting information on ER *sensu stricto*, but also information on related aspects exploring the ecology of restoration. Some initiatives were focused on returning ecosystems to pre-disturbance conditions whereas others tried to restore specific

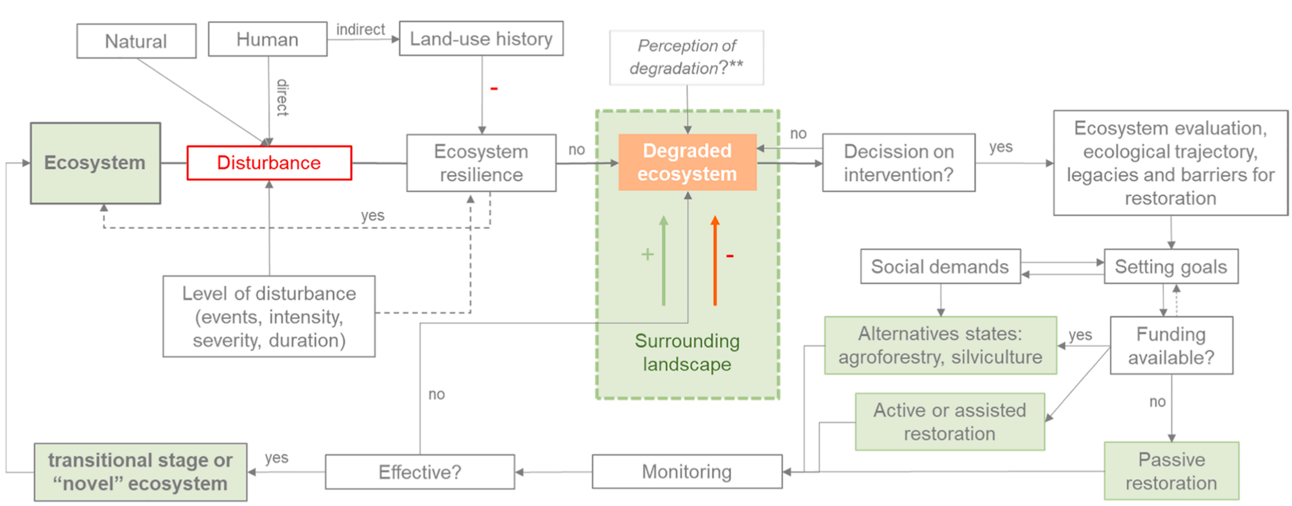

Fig. 1. Schematic diagram representing factors conditioning the implementation of ecosystem restoration. The identification of factors and processes are partially based on those identified by Holl and Aide (2011). **Perception refers to the uncertain stage of some ecosystems that can resemble an appearance of degradation Hobbs (2016).

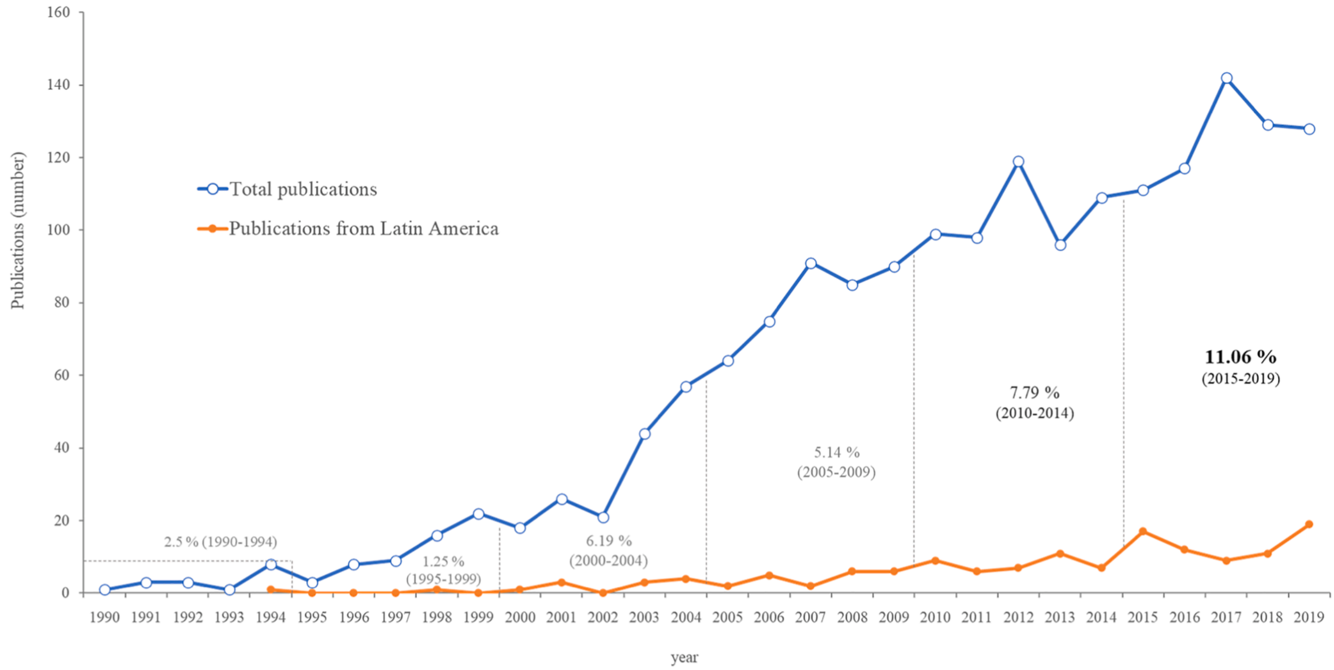

Fig. 2. Publications (n) on fire restoration based on the search in SCOPUS using the terms Fire AND forest AND restoration. The percentage of publications from LAC countries has progressively increased in the last years, reaching its maximum (11.06%) for the 2015–2020 period.

attributes or ecosystem services and thus, would classify better as *rehabilitation* (Table S1). Hence, single biophysical interventions as soil amendment or tree transplantation can be hardly considered ER. However, interventions often are limited by economic funding, political agenda or other aspects. As such, these actions can still be important pieces of ongoing processes or larger projects. Therefore, we included works that can be generally assigned to the family of restorative activities (Gann et al., 2019). Since it is important to separate restoration, afforestation, or reforestation programs, we focused on the restoration of native forest ecosystems excluding information on new stands or single-species plantations.

4.4. Natural regeneration to allow ecological restoration.

For many of us, it is not uncommon to contemplate a degraded landscape and think that something should be done to revert the situation. However, in the case of fire-affected landscapes, experiences show that passive restoration may be as effective as active restoration, provided that soils are not severely affected (low SBSs) and present low erosion risks (Holl and Aide, 2011; Crouzeilles et al., 2017). A recent *meta*-analysis including 133 studies focused on restoration of tropical forests suggested that the benefits of natural regeneration surpassed active restoration for different biodiversity groups and vegetation structure parameters (Crouzeilles et al., 2017). In fire-prone habitats, as in pine-oak forests of the National Park Cumbres de Monterrey (Mexico), passive recovery showed higher plant diversity than active restoration (Alanís-Rodríguez et al., 2008), demonstrating that natural regeneration can be rapid, obtaining similar pre-fire woody species composition after 60 years of plant succession (González-Tagle et al., 2008). Fragments of sclerophyllous forests dominated by *Nothofagus glauca* in central Chile present a great post-fire regeneration potential (Litton and Santelices, 2002; Promis et al. 2019). However, woody species recruitment in degraded areas where fires have been historically absent, as in tropical mountain rainforests, can be strongly affected by vegetation and land-use history (Palomeque et al., 2017).

The maintenance of forest remnants or *fire refugia* is critical to allow natural ecosystem recovery. Appearing as individuals or small forest patches, fire refugia promote re-population and ecosystem recovery. Fire refugia are left unburned or get less affected by fire severity (or frequency) than contiguous areas for several reasons, which include topography, isolation, or vegetation composition (Meddens et al., 2018). In the case of post-fire colonization of fire-sensitive trees, such as *Austrocedrus chilensis* that occur in Chilean and Argentinian temperate forests, the identification, protection, and maintenance of fire refugia is crucial and should be prioritized for conservation. In the first steps of colonization, light attenuation facilitates the survival and growth of shade-tolerant species, as *A. chilensis* seedlings (Urretavizcaya et al., 2017), allowing nearby tree recruitment and long-distance seed dispersal (Landesmann and Morales, 2018). The preservation of fire refugia is even more important in species with slow growth rates and low regeneration capacity as *Fitzroya cupressoides* (Lara et al., 2008).

On a small scale, fire refugia are also of great importance due to their capacity to improve microhabitat conditions. In a fragmented 50-year-old post-fire successional area in the Valdivian and North-Patagonian evergreen temperate forests, Albornoz et al. (2013) associated the increase in plant richness and the abundance of woody species to the size of remnant vegetation patches, suggesting higher tree regeneration compared to open areas. Vegetation patches were progressively expanding due to the modification of their immediate surroundings (e.g., reduced waterlogging) that facilitate plant recruitment. Additionally, the perch effect of large patches and higher trees enhanced bird-mediated seed rain. Here, both facilitation and the perch effect seem to act in combination to favor nucleation. Although fire refugia benefit seed dispersal contributing to passive restoration (Holl and Aide, 2011), forest recovery is especially difficult under harsh environmental conditions (i.e., hot and dry microclimates) or in areas with frequent fires. In tropical montane habitats of the Bolivian Andes (the *altiplano*), Lippok et al. (2013) propose an oriented selection favoring certain species, such as *Myrsine coriacea* (a bird-dispersed small-seeded forest species), that facilitate forest recovery due to their ability for long-distance dispersion.

Fire refugia are also fundamental to maintain genetic diversity since they act as reservoirs to preserve species variability. Although fires reduce the size of plant populations, Céspedes et al. (2003) found high levels of gene flow between separated populations of *Swietenia macrophylla* (big leaf mahogany) in successional post-fire sites in a moist transition dry forest in Santa Rosa National Park (Costa Rica). To protect the endemic and threatened *Austrocedrus* forests (IUCN, 2010), Souto et al. (2012) identified genetic patterns of diversity, inbreeding, and divergence, indicating that *Austrocedrus*-dominated dryland forests of northern Patagonia harbour genetic diversity and have the potential to be relatively resilient to climate disturbances. In any case, even if fire refugia are reduced to small patches or individuals, their preservation is critical since their reproductive capacity may be intact (Torres and Renison, 2017).

Remnant trees (alive or dead) can be maintained following a multipurpose vision that encompasses many aspects of restoration. Standing *Araucaria* or *Nothofagus* individuals after a large fire event in the Tolhuaca National Park (Araucanía, Chile), serve as propagule sources, shelter for native species establishment, barriers to protect biodiversity, or fauna refugia (González and Veblen, 2007). Besides the benefits of maintaining living trees, other non-visual aspects are of key importance. Remnant living trees serve as a source of propagule for fungal networks, facilitating the establishment and growth of non-pioneer plants. Bannister et al. (2020) argued that besides the increase of direct light and the absence of shelter, the removal of standing vegetation eliminates mycorrhiza, hindering nutrient uptake in an environment (Northern Patagonia) where most soil nutrients are present in an organic form, whose mineralization is greatly impaired by the naturally harsh environmental conditions of the region. Finally, it should be noted that the possibility of taking advantage of fire refugia in LAC is quite variable, and greatly relates to the scale and heterogeneity of the ecosystem. Also, contrasting land uses across regions limit the presence, and thus the relevance of fire refugia in ER. For instance, extensive agriculture areas replacing dry tropical forests in South-eastern Brazil have restricted access to fire refugia (limiting the potential for passive restoration) in contrast to patchy, small-scale forestry or agroforestry areas of Central America.

Topography is another key factor that needs to be considered in the restoration of heterogeneous habitats since environmental conditions change rapidly with altitude favoring the presence of a greater variety of ecosystems. Taking advantage of specific local conditions may benefit the natural regeneration of the Atlantic Forest, a large area of special conservation interest that contains high biodiversity and several endemic species (Myers et al., 2000). Here, pronounced slopes, aspect, low solar radiation, and proximity to forest fragments favour natural regeneration, whereas the proximity to urban areas, roads, or highways inhibits natural regeneration and increases the possibility of subsequent wildfires (dos Santos et al., 2019). In the Cerrado, environmental factors, soil properties, and land use shape a wide ($\sim 2 \times 10^6$ km^2) and complex mosaic that range from open grasslands to scleromorphic forests. Within this heterogeneity, stable formations from rocky outcrops serving as plant refugia (the *Cerrado rupestre*) are particularly resilient to fires in comparison with open areas (Gomes et al., 2014).

Elevation must be also considered in ER. Although it is generally assumed that high temperatures favor rapid plant growth, Lippok et al. (2013) indicated that forest recovery was facilitated with elevation. They observed that species density increased with altitude, suggesting a compensatory effect between the harsh microclimate conditions at deforested sites (warmer and drier) with the temperature drop observed with increasing elevation. These findings evidence that the relationship between species traits (growth rate, shade tolerance, water requirements) and elevation gradient (light irradiance, precipitation, temperature) is fundamental, and influences the capacity of forest recovery in mountain ecosystems.

4.4.1. Someone does the work for us. Attracting seed dispersers

The maintenance of fire refugia or standing burned woody vegetation favors passive restoration by providing structural components that attract frugivorous birds, therefore promoting seed dispersal during early post-fire regeneration (Cavallero et al., 2013). The multipurpose use of remnant *Araucaria* or *Nothofagus* trees mentioned above can also serve to attract flying visitors increasing bird-mediated seed rain (Albornoz et al., 2013). Flying vertebrates act as seed dispersers for a variety of fleshy-fruited broadleaf species from undisturbed cloud forest patches into burned *Pinus* areas located nearby or even to less connected patches in the Sierra de Manantlán Biosphere Reserve (Mexico), promoting cloud-forest regeneration (Rost et al., 2015). An imaginative approach consisted in the attraction of frugivorous bats from preserved forests to burned areas (Preciado-Benítez et al., 2015). Using tropical fruits as a reward, they increased frugivorous visits to degraded areas and consequently seed raining, serving as an initial restoration strategy that may be further complemented with more active interventions, such as the planting of successional tree species.

After experimental burns in the south-eastern Amazonia, Paolucci et al (2019) indicated that the lowland tapir (*Tapirus terrestris*) has the potential to collaborate in natural forest regeneration by dispersing a variety of seeds over long distances through disturbed forests. Probably attracted by the presence of more palatable leaves and higher temperatures in open areas, the number of propagules (seeds per ha/year) collected in tapir dungs from disturbed forest tripled those from undisturbed forest. Despite the potential capacity of herbivores as seed dispersers, the use of cattle is controversial since it is generally perceived as a limitation for restoration, and grazing control is commonly implemented to protect tree plantings and promote secondary forest recovery (Weaver and Schwagerl, 2008). Nevertheless, in subtropical dry forests, cattle can also eliminate grass competition and benefit tree recruitment (Braasch et al., 2017) or serve as dispersal agents that help tropical woody plant recovery (Miceli-Méndez et al., 2008).

4.5. Assisted / active restoration

While passive restoration may be adequate in some cases, fire severity together with ecological (seed bank or propagules depletion, SOM consumption, erosion rates) or environmental variables (changes in regional climate, flood, or drought risk) may limit the capacity of the ecosystem to naturally recover after fire. In these cases, passive restoration may be insufficient, and so active interventions are necessary (USDA Forest Service, 2012). Nevertheless, although passive and active restoration are generally distinguished, the reality resembles more a palette of colors where multiple shades are intermingled.

The use of remnant trees or nurse shrubs (passive restoration) as facilitating agents in active restoration activities is well documented (Castro et al. 2002; Gómez-Aparicio et al. 2004), also in LAC (Galindo et al. 2017). Nurse plants modify their surroundings (Fig. 3), augmenting light interference, increasing water availability, and diminishing grass competition, which is expected to benefit shade-tolerant species more than pioneer ones (Galindo et al., 2017; Blanco-García et al., 2011). The establishment of nurse shrubs can serve to prevent fire incidence in areas under ongoing restoration; in a fire occurred during a subtropical dry forest restoration (Sierra Bermeja, Puerto Rico), Santiago-García et al. (2008) registered lower immediate and delayed mortality of functional groups in areas hosting nurse trees compared to more open sites. This was probably due to differences in fuel load that resulted in contrasting fire intensities.

After severe fire events, Bannister et al. (2013, 2020) compared the responses and growth under different planting conditions of the vulnerable and fire-sensitive *Pilgerodendron uviferum* - the Worlds southernmost distributed conifer (Martinez, 1981; Bannister et al., 2012). Two years after planting, tree seedlings suffered water stress in open areas but tolerated humid conditions provided by nurse shrubs. Beneath the protective canopy *P. uviferum* seedlings showed lower mortality and higher shoot growth, foliar nutrients, and better photosynthetic performance than unsheltered seedlings (Bannister et al., 2013). After 4 years of growth protected *P. uviferum* seedlings reached higher lengths and showed a superior photosynthetic performance (higher F_v/F_m values) (Bannister et al., 2020). However, natural recovery might be insufficient to efficiently restore degraded areas, since *P. uviferum* recruitment is limited by seed availability (Bannister et al.,

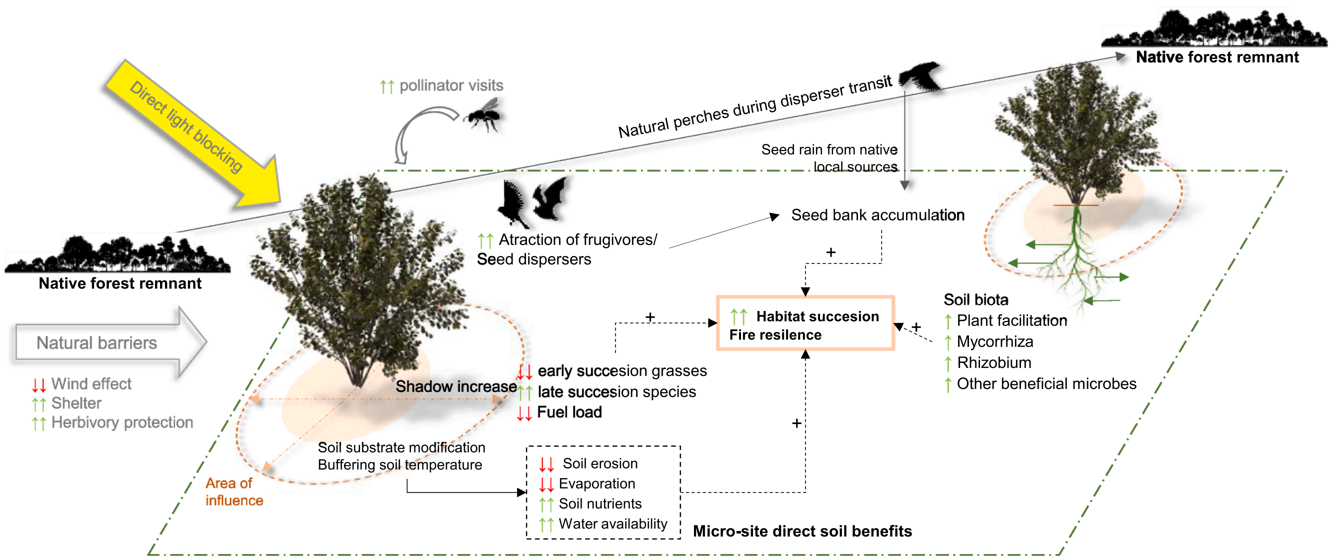

Fig. 3. Simplified representation of aboveground and belowground benefits of using nurse shrubs in ecological restoration. Dotted circles represent the area of shrub/tree influence (defined only for representation purposes). Continuous lines represent ecological processes. Dotted lines denote direct and indirect effects produced by nurse establishment.

2013). A mixed strategy based on the complementary use of remnant trees with the plantation of individuals in groups (*cluster* plantation) would be preferable to restore *P. uviferum* forests in Northern Patagonia (Bannister, 2015). Variations in water availability also conditioned seedling establishment and growth of *Austrocedrus chilensis, Nothofagus pumilio*, and *N. dombeyi* seedlings in fire-affected sites (Tercero-Bucardo et al., 2007).

In the case of the shade-tolerant *Abies religiosa* in the Monarch Butterfly Biosphere Reserve (Mexico), 5-year individuals significantly increased survival rates from 18% to 72% when planted contiguously to a native nurse species (Carbajal-Navarro et al. 2019). The successful implementation of native and fire-sensitive species such as *Nothofagus pumilio* or *Fitzroya cupressoides* has also been improved due to the amelioration of site conditions e.g., the direct application of organic amendments to create *fertile islands* (Varela et al., 2006, 2011) or from more elaborated strategies. Isolated individuals can be mixed with abiotic structures as shelters to ameliorate soil physicochemical conditions and plant performance (Urretavizcaya and Defossé, 2013), also protecting plants from grazing. After a fire event in the Torres del Paine National Park (Chile), active restoration by planting *N. pumilio* individuals was constantly influenced by strong biotic and abiotic constraints, and it was shown that plant survival depended on nucleated planting, protection against herbivores, and microsite facilitation (Vidal et al 2015; Valenzuela et al., 2016, 2018). Microsite facilitation using native shrubs or logs increased the survival rate and height of *N. pumilio* seedlings, even in comparison with plants protected with polyethylene shelters (Valenzuela et al., 2018). In temperate Argentinian forests (Subantarctic or Patagonic-Andean forests), Urretavizcaya et al. (2018) indicated that *N. pumilio* regeneration is highly constrained if protective measures are not implemented. Several aspects considering *Nothofagus* restoration projects (beyond post-fire restoration) across the Southern hemisphere have been recently reviewed (van Galen et al., 2021). In the Cerrado, Pellizzaro et al. (2017) selected a mixture of 75 species with different lifeforms (trees, shrubs, and grasses) in different proportions for a field experiment. After 2 years, >80% of the planted species were observed in areas formerly dominated by invasive grasses.

Although often considered as a degraded state, the ability of second-growth forests to provide ecosystem services is significant, but these fire-prone structures attract less attention -in ecological terms- than old-growth forests. Studying how different land-uses in the past affected the recovery of vegetation structure, richness, and species composition in a lowland Brazilian Atlantic Forest, Sansevero et al. (2017) indicated that low values of vegetation recovery in the secondary forests suggested a pattern of arrested succession related to the dominance of fire-resistant species. Thus, they recommended interventions like enrichment plantings, nucleation techniques, and assisted natural regeneration to accelerate forest restoration.

Ecologically-oriented proposals promoting forest transformation to ecosystems more similar to the natural range of structural and compositional variability represent better options for increasing forest resilience. In these cases, alternatives favoring the transition from secondary to old-growth forests exist but mainly depend on species dominance (González et al., 2015). Here, it is important to pay attention to what ecological attributes are selected to evaluate our knowledge of ecosystem recovery. For instance, some old-growth forest attributes (large trees) would be rapidly acquired in *Nothofagus*-dominated forests, whereas other attributes (high diversity) would be acquired quickly in mixed evergreen forests (González et al., 2015).

4.6. *Social aspects of restoration*

4.6.1. *The social challenge of postfire restoration actions in LAC*

A good understanding of the ecological and technical aspects of fire is necessary but social aspects of fire restoration -and restoration in general- are less explored. The social relevance of forests in LAC is paradigmatic and so are the actions aimed to restore them. This importance is perfectly exemplified in South America where a significant proportion of total forest area (>20% vs 7% rest of the world) is designated primarily for social services (FAO-Global Forest Resources Assessment, 2020). Consequently, most ER projects in LAC are framed in the social dimension (Ceccon and Martínez-Garza, 2016) and thus, developing initiatives to effectively address the complex nature of restoration should include socioeconomic aspects to fulfill stakeholder needs, integrating multidisciplinarity into the restoration experience, and consider the diversification of subjects and capabilities (Bloomfield et al., 2019). To this end, acquiring a deep knowledge of the socio-ecological context where forest fires occur is essential not only in terms of biological conservation or sustainable forestry operations but to design locally adapted, effective management practices (Jardel et al., 2006; Blignaut and Aronson, 2020). Cooperation and community engagement are fundamental since respect and appreciation for ecosystems are linked to the degree of participation in decision-making processes and restoration activities (Clewell and Aronson, 2013). At this point, communication becomes crucial, and it is important to create a profound narrative with society, one that articulates and ideally culminates in a collective vision based on local values (Blignaut and Aronson, 2020). This statement becomes even more powerful in the LAC context, since all along the territory different cultures and communities are closely tied to the land.

4.6.2. Restored areas as centers for education

In many cases, fires represent only a part of the community problem. In general, a variable and *ad hoc* mixture of human disturbances including browsing, ranching, agriculture, timber extraction, or hunting lead to different degrees of ecosystem degradation. Conservation is a key strategy in protected areas, and few initiatives illustrate this better than the Guanacaste Conservation Area (ACG) in north-western Costa Rica. This community developed a powerful tool for the conservation of natural resources in the long term: the development of *biocultural restoration* (Cruz and Segura, 2010). Spanning over 30 years, biocultural restoration focuses on the younger generations (ages from 9 to 12) as heirs of the ability to decide on future environmental issues. Based on immersive *in situ* bioliteracy, the goal of biocultural restoration is to change younger community attitudes toward natural resources: from the restoration of dry/rain forest occupied by pastures of invasive species, through fire suppression, to understand ecological concepts and different processes of forest succession.

Recovered areas can also serve as educative places. Restored *Fitzroya cupressoides* areas in temperate forests of Chile serve as an important source for education and diffusion of concepts and possibilities of ER (Lara et al., 2008). Also, as an intermediate step between ER and social participation experience, Carrasco et al. (2019) carried out a participatory approach to recover postfire soil health by using organic amendments combined with the small-scale implementation of native trees and also an educational program focused on involving students in restoration. Functioning also as an investment in fire prevention, the social participation and citizen involvement may not only reduce fire occurrence and therefore damage, but also the usually high costs associated with fire extinction operations (Monroe et al., 2016).

4.6.3. Restoration and fire regulation to reduce fire risks and increase ecosystem resilience

A fact that has been highlighted throughout these lines is the distinct responses of fire-sensitive and fire-prone ecosystems. Despite the differences in species composition, humidity, fuel types, fire behavior, community resilience, recovering times, etc. With some exceptions (Mistry et al., 2016), and up until recently, fire management has mainly followed the same recipe for all ecosystems: fire exclusion. But this trend has been changing for some years now. Some classical fire control measures are no longer perceived as beneficial, or at least they are not deemed as being applicable in all cases. Although inherent to some ecosystems, a large part of the society still perceives fire as a natural menace. Consequently, fire exclusion has been used as a preventive tool in many regions during the last century, largely following the US example. From the ER perspective, fire exclusion can be considered negative in functional terms, as it interferes with the long-term dynamics of fire-adapted habitats. Indeed, fire clearing helps to maintain different vegetation types and habitats in fire-prone ecosystems (Rodriguez-Trejo and Myers, 2010). However, the general perception has started to change, and forest fire regulation is moving from suppression to integrated management, incorporating ecological and social considerations (Jardel et al., 2006; Pérez-Salicrup et al., 2020; Martínez-Torres et al., 2018).

Fire management strategies must be consistent with habitat regeneration patterns but should also integrate the empirical knowledge of local populations (Pérez-Salicrup et al., 2020). Incorporating *traditional ecological knowledge* (TEK) or *local ecological knowledge* (LEK), like the use of prescribed fires into current management practices offers important ecological insights, but also key knowledge that can greatly help ER (Gann et al., 2019). Traditional fire management represents a good example of an approach that promotes the association between indigenous and non-indigenous institutions to share and implement an understanding of cultural burning practices (Mistry et al., 2016). In turn, these bottom-up fire management approaches also benefit from the exchange of information and improve cultural connections since its effectiveness relies on a socioecological system where knowledge, culture, and community livelihood are intimately interconnected with landscape management (Mistry et al., 2016). Notwithstanding the benefits, the diverse epistemologies of different TEK systems limits its implementation, and thus, it is necessary to validate their diversity into policy and management processes (Guerrero-Gatica et al., 2020).

La Sepultura Biosphere Reserve (Mexico) represents a highly diverse forest reserve, amalgamating different cloud, tropical and pine-oak forests, where communal lands and private property comprise 95% of the reserve. Here, local communities (ejidos) use prescribed fire. With restricted use of forest resources, community participation in fire management provides a basis for sustainable forestry and environmental services payments (ESPs), reciprocally promoting community welfare and fire protection (Huffman, 2010). Working together, communities and researchers generated an integrated a fire management plan combining the traditional use of fire to maintain both natural pine forest dynamics and local livelihoods. Thus, prescribed fires had a multipurpose objective: reduce hazardous fuels, remove vegetation to favor *Pinus* regeneration, improve grass forage quality, and train younger community members. Learnings from this project served to create guides to implement prescribed-fire projects, and most importantly, this work contributed to the approval of a National Strategy for Fire Protection and Fire Management. The extended thought and stereotypes of forest fires caused by rural communities are increasingly perceived as mistaken since many communities use fire wisely (Rodríguez-Trejo et al., 2011), e.g., La Sepultura farmers consider 40 variables when they use fire (Huffman 2010).

4.6.4. Agroforestry and silviculture practices in burned and fire-prone areas

Agroforestry and silviculture can be distinguished from restoration activities since these practices balance the recovery of ecosystem structure, content, and function with the provision of ecosystem services. Agroforestry or silviculture can be seen as a functional approach itself (Holl and Aide, 2011), but also as transitional stages between degraded and restored forests (Bannister et al., 2016). In both cases, these alternatives are based on the partial recovery, focused on enhancing ecosystem functioning with a socio-ecological perspective (Gann et al., 2019), often neglected in restoration attempts (Wortley et al., 2013). The design of silvicultural systems that integrate economic and ecological objectives requires a comprehensive vision of development patterns, including the role of disturbances, biological legacies, and their influence on species ecological responses (González et al., 2015). Rehabilitation using agroforestry practices may be more complex and difficult in areas subject to excessive and intense fire cycles, where secondary forests decrease their ability to recover original C stocks within average return intervals (Villa et al., 2015).

Agroforestry practices can be specially recommended when residual forests remain fragmented and isolated, as it occurs in Atlantic Forest patches in Brazil (Cullen et al., 2001). Here, diversified agroforestry belts would function as buffer zones protecting forest fragments, reducing the edge effect in the transition zone between forest and open areas. Enriching buffering zones with economically valuable trees and crop species represent both ecologically viable and socially acceptable options for conserving regional biodiversity (Cullen et al., 2001). Furthermore, restoration based on the balance between forest biodiversity and community well-being could be economically sustained through the development of biodiversity-based products (Nobre et al., 2016). Here, TEK and LEK are fundamental to improve species selection and thereby increasing the success of forest restoration efforts (Fremout et al., 2021). Participatory workshops that included farmers, local foresters, and agricultural professionals defined a list of native species for a mixed restoration experience in the Hidalgo State (Mexico) (Santana et al., 2011). Selected species, classified into *catalyst* (species that favor conditions for regeneration or succession) and *endangered* species (species with reduced populations, at risk of extinction or having protected status) were combined in a multi-scale and diversified approach creating separate systems (i) management of natural regeneration, (ii) forest plantations, (iii) enrichment plantings, (iv) stream restoration, and (v) agroforestry systems.

In this sense, it is worth noting that community willingness to restore its forest diversity may be associated with the benefits obtained from native plants (timber, food, ornamental, cultural). Thus, tropical forest communities may have a greater predisposition (due to the large diversity and species usage), while temperate forest communities (where timber resources are more important) might be more inclined to select species for monoculture plantations. However, there are also cases, like in the Chilean Mediterranean sclerophyllous woodlands that are dominated by native plants that provide scarce and local benefits (e.g. coal, firewood, honey), where rural communities encourage restoration even without public financial support (Smith-Ramírez et al., 2019).

Traditional agroforestry practices, such as those supported by the Lacandon community in Chiapas (Mexico), facilitate forest regeneration by helping overcome barriers to secondary succession and avoid arrested succession (Falkowski et al., 2020). Considered as a sustainable form of tropical forest restoration management, sustainable activities of the Lacandońs mainly consist of recurrent cycles of short milpa cultivation (3–5 years) followed by prescribed fire and habitat restoration. Restoration is mainly based on the maintenance of crucial tree species in combination with active planting after burning. Although fires are human-induced and controlled, fundamental stand variables like canopy cover, leaf litter or stand basal area, reached similar values to those observed in mature forests over a period of 10–20 years after burning.

5. Constraints to ecological restoration

When confronting the reality of a landscape affected by fire, it is important to elucidate to which extent restoration is possible, and particularly the meaningfulness of recovery strategies and actions set to reverse ecological degradation to avoid only focusing on those that are typically based on returning ecosystems to a formed, often idealized state. Factors such as the level of ecosystem degradation and resilience, the balance between conservation and human intervention, cost-analysis of intervention or potential alternative uses should be considered. Moreover, returning dynamic systems to historical pre-conditions may not lead to effective conservation strategies. A good example is exemplified in the rainforest-dry forest border of the Amazonia where anthropic deforestation, temperature increase, and predictive models suggest an upcoming transformation of vast rainforest areas (IPCC, 2014). Emerging post-fire ecosystems may require strategies that consider historical and novel landscapes that seek to enhance the adaptive capacity of ecosystems and their inhabitants, facilitating connectivity and management, with emphasis on functional integrity rather than species (Barnosky et al., 2017). In the last years, several studies pointed out similar necessities and limitations for ER (Smith-Ramírez et al., 2015; EFTEC et al. 2017; Fisher et al., 2019; Cortina-Segarra et al., 2021) (Fig. 4).

5.1. *Environmental limitations. Abiotic and biotic factors limiting restoration*

Besides the direct impacts caused by wildfires, ER will be also influenced by the progressive variation in local environmental conditions, including water regimes, temperature, insolation, etc. Ecological succession can transform degraded forest stands into secondary forests. It is particularly challenging to design restoration strategies, which strongly rely on the establishment of target vegetation in areas considered transition zones (such as moist forest-dry forest or open Cerrado-close Cerrado interfaces) that are characterized by heterogeneous and alternative states. The Cerrado represents a perfect example of a complex mosaic of habitats where species selection is a priority since the identification of the target ecosystem is difficult (Schmidt et al., 2019). Here, the use of conceptual models considering alternative states together with causes and constraints to transitions, known as state-and-

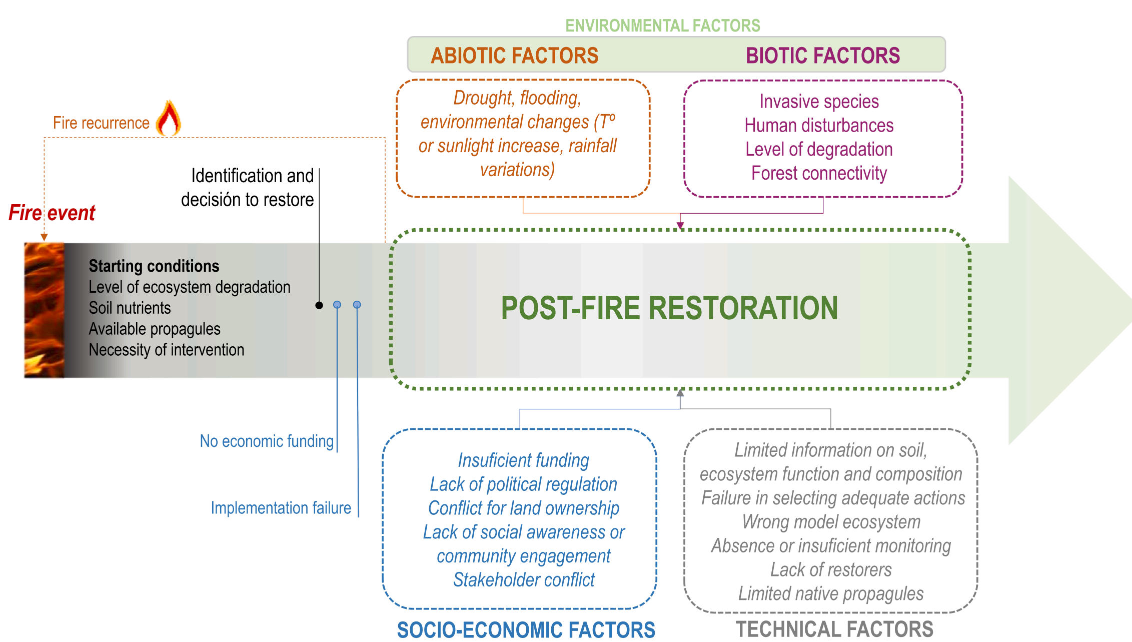

Fig. 4. Schematic representation of important aspects to evaluate when considering ER in fire-affected forest ecosystems: the degradation level, the necessity of intervention, and proper limitations of post-fire restoration process (including abiotic, biotic, socio-economic, and technical constraints). From left to right, the scheme represents a temporal line starting with the fire event and goes up to the restored ecosystem. This classification is dynamic, as some factors could be considered in different groups: thus, a *Pinus* invasion represents a biotic factor limiting restoration, but economic costs of clearing this invaded area could be otherwise considered as an economic constraint.

transition models (STMs) can be useful. Peinetti et al. (2018) defined different states and transitions for Caldenal forests (Argentina), each one with different recommendations (using decision trees) indicating that effective restoration would require a combination of different strategies, emphasizing the importance of adapted management rather than interventions.

When planning post-fire restoration, the level of necessary intervention needs to be carefully considered. The different aspects to ponder include the extent to which plant cover should be maintained, implemented or removed before restoration, and how to manage dead trees. The ecological outcome of interventions like salvage logging may in some cases yield unpredictable results since its effectiveness depends on variables that heavily interact with disturbance (wildfire) effects. Furthermore, notwithstanding some experiences reported in Australia, there is limited information on previous experiences in the Southern hemisphere (Leverkus et al., 2018). Logging or excessive site preparation after fire may undermine the potential for natural restoration by increasing soil erosion and compaction, increasing light incidence and surface soil temperature, as these may negatively impact the trajectory of natural regeneration or ecological processes (González and Veblen, 2007; Carbajal-Navarro et al. 2019). Therefore, preserving standing living trees for soil and seedling protection, as well as for the provision of complementary functions is highly recommended (Castro et al., 2011; Urretavizcaya and Defossé, 2019).

The presence of invasive alien plants (IAPs) represents a major cause for concern when considering ER. IAPs are capable of causing further ecosystem degradation (Brooks et al. 2004), to the extent that the magnitude and variety of their potential impacts leads to the consideration of IAPs as a major constraint for ER. A survey between project managers highlighted that > 20% of the projects identified the presence of invasive species as a limiting factor for restoration in Mexico (Méndez-Toribio et al., 2018). Their presence often alters the assemblage of natural communities generating positive feedbacks with fire regimes, perpetuating and increasing fire pressure on ecosystems already degraded (Brooks et al. 2004; Mandle et al., 2011). Post-fire increase of IAPs has been reported throughout different LAC biomes including tropical, Mediterranean, temperate, and cold ecosystems (García et al. 2015; Gallegos et al., 2015, 2016; Silvério et al., 2013; Gómez-González et al. 2017; Urrutia-Estrada et al. 2018; Paula and Labbé, 2019). Many dominant IAPs in LAC as *Bromus* spp, *Pinus* spp., *Cytisus* spp. *Ulex* spp. are highly flammable and thus capable of promoting fire spread (Speziale et al., 2014; García et al., 2015; Cóbar-Carranza et al., 2014; Altamirano et al., 2016; Paritsis et al., 2018) and increasing the probability of frequent wildfires (Taylor et al., 2017; Paritsis et al., 2018). Moreover, fire can directly favor the presence of IAPs, as their intrinsic and extrinsic fuel properties increase flammability and fire frequency in many ecosystems, altering fire regimes, and replacing native plants or animals (Franzese et al., 2017; Paritsis et al., 2018).

Fragments of well-preserved natural forests in Central Chile are more resistant to invasion, especially by *Pinus radiata* (Bustamante et al., 2003, Gómez et al., 2011). Forest degradation by fire increases the probability of invasion, hindering restoration and favoring the spread of future fires. Although wildfire origin is not necessarily related to the presence of IAPs, fire impact (spread, intensity, and severity) can be enhanced by its presence. Reciprocally, frequent fires benefit IAPs creating positive feedbacks which might lead to rapid and permanent ecosystem modifications (Brooks et al., 2004; Contreras et al., 2011).

In this sense, ER becomes essential to recover forest functioning, limit the expansion of IAPs and serve as a preventive measure against new fires. It is worth noting that fire (prescribed fire) is used to manage IAPs (Weidlich et al., 2020), e.g., for *Pinus* control (Nuñez et al., 2017) or to limit the escape and invasion of *Eucalyptus* (Toledo et al., 2020). However, prescribed fire in LAC should be carefully considered if IAPs originate from fire-prone areas or with species that can be considered fire promoters as *Pinus* (Holmes et al., 2000; Kremer et al., 2014), *Teline* or *Cytisus* (Alexander and D'Antonio, 2003; Pauchard et al., 2008), and *Ulex* (Johnson, 2001), due to their rapid post-fire response and the consequences on vegetation recovery.

5.2. Technical/management factors

Soil monitoring represents a pending task for ER in LAC. While priority is mainly placed on attaining a rapid vegetation recovery, the monitoring of soil physicochemical parameters during post-fire restoration is limited, and hence, our understanding of ER dynamics remains incomplete. In this sense, the lack of information regarding the initial state of degraded ecosystems or SBS hampers the interpretation of restoration outcomes; it becomes difficult to distinguish whether the success of restoration is due to the intervention or to a low level of soil degradation.

Soil communities are generally overlooked in ER, which explains the limited information available about its post-fire condition or recovery. This lack of information is counterproductive, as soil microbial communities are first receptors of fire effects, and also serve as facilitators for ER. Fire is responsible for large soil disturbances, but it is important to consider the scale, severity, and vegetation replacement to evaluate the potential of remaining soil communities for ER (e.g. mycorrhiza). Although plant diversity increases as well as mycorrhizal diversity does (van der Heijden et al., 1998), and positive results (in terms of plant growth, nutrient mobilization) have been reported (Allen et al., 2003, 2005; Scotti and Corrêa, 2004; Aguirre-Monroy et al., 2019), a regular, purposedly use of symbiotic organisms is limited in ER. In fact, despite the importance of ectomycorrhiza (EM) for tree establishment (e.g. species of *Nothofagus* in temperate forests), some authors pointed out the limited research evaluating EM inoculation for postfire restoration (Policelli et al., 2020) or restoration of degraded areas in general (van Galen et al., 2021). There are incipient efforts to characterize postfire soil communities in the case of *Araucaria araucana* (Chávez et al., 2020).

It is also necessary to emphasize that restoration -either active or assisted- depends on the supply of plant material or the *nursing capacity*. Restoration using appropriate species and genetic material adapted to local conditions that reproduce the target ecosystem is crucial (Gann et al., 2019). The poor quality and low supply of native species is a major bottleneck for natural forest restoration in Chile (probably extensive to other regions), highlighting the importance of providing sufficient (and adequate) plan material to fulfill restoration demands (Bannister et al., 2018; León-Lobos et al. 2020, Acevedo et al. 2021). Although some progress has been made, regions differ in their capacity to satisfy seedling demand for restoration. In Brazil, Moreira da Silva et al. (2017) indicated that regions or biomes where restoration has more tradition (e.g. Atlantic Forest) have higher nursing capacity, whereas the potential to provide plant material for ER in other relevant biomes (Amazon, Caatinga, Cerrado) is far more limited. Consequently, reduced seedling availability and the lack of adequate and adapted species may aggravate species mismatch between nursery-grown plants and the floristic composition of the target ecosystem. Exploring the capacity of Chilean nurseries, Acevedo et al. (2021) indicated a low production capacity, poor seedling quality, and inadequate training of nursery managers as fundamental barriers that delay restoration timelines and biodiversity goals. Far from pessimism and despite the evident limitations, these authors foresee the possibility of increasing forest restoration by generating a new economic activity for rural economies. Nevertheless, they call for governmental policies that incentivize nursery as an economic activity with the contribution of science-based information (e.g. developing cultivation standards for native species). Nevertheless, even if plant material is available, it should be used efficiently; some authors identify the necessity of trained professionals as a key limitation for ER in LAC (Smith-Ramírez et al., 2015).

5.3. The socio-economic challenge of restoration in LAC

The future of ER will be social or it will not be. Behind these words emerges one of the fundamental precepts of the Society for Ecological Restoration (SER) principles (SER, 2004; Gann et al., 2019): ER supports processes that improve human wellbeing and development at the individual, community, and regional level. In this sense, local communities generally identified fires as a major cause of forest degradation (Santana et al., 2011) but there is no unique formula to engage local needs and recovery demands. While technical solutions to overcome barriers for ER are abundant, the development of policy strategies to promote sustainable forest management and restoration is insufficient (Peinetti et al., 2019). In some cases, the disconnection between regulatory decision centers and the necessities of execution sites aggravates the situation (Sorrensen, 2009). Even when social inputs are required, the participation of local governments in ER programs is complex due to the general lack of long-term working plans (Santana et al., 2011), especially under administrations with low environmental sensitivity (Lizundia-Loiola, 2020)

Traditional communities have little power on the decision-making concerning the exploitation of their natural resources, and the legal framework imposes certain limitations on their uses (Pineda-López et al., 2015). The lack of mechanisms to generate a collaborative environment among stakeholders is part of the problem. Therefore, participatory approaches are fundamental to allow information exchange and to identify constraints for implementing decisions that would increase opportunities for further scientific research and cooperation (Santana et al., 2011). In many cases, stakeholders (communities, experts, government managers) have different perceptions of what the restoration priorities should be (Castillo et al., 2021). Also, ecological aspects of restoration eventually conflict with socioeconomic issues (necessities and interests of different stakeholders) at the local level, representing a barrier for ER in LAC, but also in other countries (Fisher et al. 2019). For instance, urgent restoration measures are often complemented with logging or salvage logging to recover economic losses or assisting ecosystem recovery. Logging is used in the postfire management of *Austrocedrus chilensis* due to its high-quality timber, but tree survival significantly improves (from 10% to 90% in naturally restored areas) if logging is excluded (Urretavizcaya and Defossé, 2019).

The lack of financial resources is a fundamental limitation for the implementation of ER projects. Compared to other countries, LAC governments typically allocate fewer resources to restoration since many countries in the region are financially constrained. Often, the mosaic of the economic reality across the region, close community-land ties, social fragmentation, or the necessity for regular incomes, place strict limits on planning and execution costs of ER projects. Here, performing specific actions such as robust cost-effective analyses of restoration (Moran et al., 2010) and the implementation of projects with limited intervention and a strong base on natural restoration (Armesto et al., 2007) is particularly relevant.

The problem of IAPs has also an evident economic dimension. In this case, the economic costs (staff, machinery) are generally high but it will depend on the species and the difficulty of its management, especially if the invaded area exceeds a certain size (Nuñez et al., 2017). In such cases, priority should be given to those areas with the highest intrinsic value.

The public origin of the invested funds may also represent another socio-economic limitation (inefficiency, distrust, corruption) for ER (Ceccon and Martínez-Garza, 2016). Culturally, ER strengthens communities, institutions, and interpersonal relationships by participation in a common pursuit (Clewell and Aronson, 2013), but this objective is hindered if land ownership is too concentrated or excessively fragmented. Strengthening community skills and tools for land management (e.g., creating communal native forests in abandoned lands) may simplify the allocation of subsidies while simultaneously improving the social perception of the use and proper management of public funds (Smith-Ramirez et al., 2019). Limitations derived from private land ownership are exemplified in ER actions conducted in Chile. In this country, post-fire restoration actions represent almost half (44%) of the total restoration projects reported. However, nearly 3/4 of registered post-fire restorations (73%) were carried out by private forestry companies (https://gis.mma.gob.cl/portal/home, accessed 22/05/2020). This case exemplifies the priority of restoring certain areas because of their high production and high economic return.

6. Future perspectives

The future of post-fire restoration, and restoration in general, is uncertain but essential information can be learned from recent restoration experiences (Ockendon et al., 2018). While most of those learnings may be applied in LAC forest ecosystems (Fig. 5), it is imperative to identify the main potential limitations for ER in such diverse and vast region.

Promoting collaborative actions: expertise sharing, co-financing, and co-development should be stimulated between regions to increase the effectiveness of policy and restoration practices. This aspect is particularly relevant in the LAC context, where south-south cooperation is fundamental (Gann et al., 2019). However, the reinforcement of ER also requires information to be more openly shared. Therefore, decisive efforts are needed to overcome political boundaries and share geospatial data to understand patterns and processes of ecological, hydrologic, and socio-cultural systems (Villarreal et al., 2019). In this sense, remote data acquisition represents a rapid, powerful, and reliable tool to estimate fire potential based on fuel characteristics (Pettinari et al., 2014), analyze fire occurrence and severity (Lizundia-Loiola et al., 2020), model forest dynamics and vegetation recovery (Cantarello et al., 2011; Santana et al., 2020b), or monitoring IAPs (Kattenborn et al., 2019). The acquisition of near-to-remote data using phenocams that complement satellite and field observations is also gaining attention to monitor plant phenology at high temporal resolutions (Alberton et al., 2017). This type of novel, low-cost technology also contribute to establish *e*-Science research bringing together classic researchers (e.g., those conducting field- or lab-based research) and computer scientists, while also allows for the incorporation of society into collaborative scientific networks (Alberton et al., 2017).

Celebrate and protect the uniqueness of LAC ecosystems: strengthening the science of restoration in LAC is fundamental (Armesto et al., 2007), but it is also critical that the region has its own, adapted narrative. This aspect is particularly relevant when considering that the inherent difficulty to quantify the long-term benefits of ER is mostly ignored, underestimated or explicitly discounted, while the measurable costs are typically accounted in full (Blignaut and Aronson, 2020). Restoration benefits are complex to evaluate since natural capital has been traditionally ignored but the value of ecosystem services is progressively increasing (Costanza et al., 2014; Kubiszewski et al., 2020). The development of innovative applications based on TEK and LEK, including Living Labs and business nests, appears to empower local populations and preserve ecosystems. Attaining sustainable solutions from traditional assets, as it occurs in the Cofre de Perote National Park (Mexico) can be used as a successful example. As it is the case in other protected areas elsewhere, productive activities in the NP are prohibited. Here, forest resource-management activities are limited to pruning *Abies religiosa* both to prevent forest fires and manufacture Christmas wreaths (Pineda-López et al., 2015). Branch logging has ecological and economic implications (i.e. while reducing the fuel load, it also complements familiar economy, promotes the park, increases the social attention and the possibility of attracting further funding).

It is necessary the establishment of a common regulatory framework that allow for the development of national policies that may promote and financially support ER at different scales (Smith-Ramírez et al., 2015). However, despite some recent advances in restoration policies, only 4 countries in LAC (Colombia, Brazil, Ecuador and Guatemala)

Fig. 5. Major topics to consider for ER in LAC (adapted from Ockeldon et al. 2018). Aspects related to the four topics above (in green) are available from the literature and previous experiences. While these four topics may be to some extent adapted to LAC, ER actions should focus emphasize the topics below (in yellow). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

have developed national restoration plans (Toribio et al., 2017). Restoration guidelines appear once the concern on environmental degradation struck deep into society and the lack of regulation is symptomatic of low societal awareness. This ultimately results in a systemic lack of funds to conduct ER activities across LAC.

Improving social responsibility and private participation in ER: In a geographical context where corporations exploiting natural resources are of outstanding relevance, it is imperative to increase demands for corporate social responsibility through different tools such as public-private partnerships, the payment for ecosystem services or compensation mechanisms for ER. Companies should understand that participating in environmental remediation processes have global benefits for all of us, and realize that these can also report individual benefits in terms of corporate image. Fiscal recovery policies, including restoration of C-rich habitats represent natural capital investments that offer high economic multipliers and represent positive impacts on climate (Hepburn et al., 2020). In Chile, CONAF (the National Forestry Corporation) offers economic incentives (subsidies) for native forest restoration or by activities that favour its regeneration (CONAF, 2015). While offering compensatory approaches might be of some interest in those cases where economic losses can be easily quantified, these actions are less tangible when it comes to assigning value to complex concepts such as biodiversity. Moreover, profit-driven approaches should be carefully adopted since they could lead towards a simplistic commoditization of environmental values.

Strengthen monitoring and the effectiveness of restoration: As mentioned above, the lack of funding (public or private) is one of the main limitations of ER. But in addition to the initial economic effort, the lack of funding translates into a lack of monitoring of the restoration progress. In many cases, restoration actions/programs are based on the plantation of propagules, but little concern is given on what happens to the future individuals, resulting in high mortality rates, and consequently, in the low efficiency of invested funds. To this end, more effective and precise schemes that maximize its efficiency are necessary. Also, the implementation of ER projects that are based on cost-effective analyses, reducing the level of intervention, and focus on natural restoration, become essential. If funds are limited, the efficiency in its management should be a priority.

Recognizing and promoting the real value of ecological restoration: restoration has been traditionally considered an aesthetic sink for economy, unable to supply added value and return to society, and also conflicting with job creation (Bezdek et al., 2008). However, reality is far from that misperception. Restoration represents an emerging niche for sustainable economic development, offering opportunities for local stakeholders, and capable of creating positive ecological and economic feedback loops (Gann et al., 2019). Activities related to ER differ from those associated with the economic sector, but it is gaining increasing attention within the green economy (BenDor et al., 2015), creating more jobs per million invested than other traditional activities (Jaeger et al., 2021), and with an economic output and employment rates that can be measured (Bezdek et al., 2008; BenDor et al., 2015).

Take advantage of the current positive context: Although limitations for ER are still considerable, a time of opportunities is coming and restoration possibilities in LACs are likely to multiply in the coming decades (Armesto et al., 2007). More than ever, society is increasingly conscious of environmental problems, demanding actions to reduce GHG emissions and to recover forest landscapes that increase natural sinks and C lost during fires (Bertolin et al., 2015). In this sense, LAC initiatives could benefit from ongoing environmental agendas. International calls such as the UN Convention on Biological Diversity (CBD, 2014) or the upcoming Decade on Ecosystem Restoration are forcing regulation, increasing project demand and generating funding opportunities. Although the legal framework was already existent, the EU has recently launched the EU Biodiversity Strategy for 2030 (EC, European Commission, 2020), especially focusing on the restoration of degraded ecosystems. National plans can be the basis for policy regulations and may be used to create financial instruments and regulatory mechanisms that include fiscal incentives such as taxes, permits and subsidies, donor or government grants, and the payment for environmental services (Méndez-Toribio et al., 2017). Environmental services payment (ESP) encompasses initiatives for sustainable management and the ecological recovery of publicly owned land and forests, involving land inhabitants whose economic activities and necessities are duly considered (Kerr et al., 2014).

Education and training as a pillar for restoration: proper implementation of ER is impossible without restorers and there is an urgent need for trained professionals and scientists that design and monitor effective restoration activities (Smith-Ramírez et al., 2015). Both environmental (restoration) and societal (employment) needs converge at this point, creating a niche that will engage social work and scientific research. In this line, educational experiences may be the basis for collaborative effort between multidisciplinary academic teams and stakeholders to identify common strategies promoting effective environmental protection measures adapted to the local socio-economic reality. Educational projects that consider student and stakeholder collaboration in restoration based on the Service-Learning methodology (S-L) (Souza-Alonso, *under elaboration*) represent a promising path to explore. Besides restoration targets, S-L projects also ensure a direct benefit (learning) to those involved. Based on its demonstrated potential, restoration activities based on S-L projects should be considered as viable options within the wider choice of ER approaches that can be adopted in the coming decades (Cortina-Segarra et al., 2021).

7. Conclusions

While ER is essential in post-fire recovery and provides us with tools to transform our society based on principles such as social justice, equity, and respect for environmental values, our biggest challenge is to detect and understand the human drivers of wildfires and their intricate relationships; under the umbrella of climate change, anthropogenic factors such as extensive cattle ranching, forestry plantations and logging, inadequate fire management, market pressure, land speculation, and fragile governance interact with wildfires occurrence and severity. In many cases, land-use changes still represent the major force behind these fires, and without recognition, interest, or economic disincentives to landowners, the situation will continue. Following different international initiatives, we encourage practitioners, researchers, and those interested in ER to join efforts in identifying the major barriers for post-fire restoration in LAC.

Unquestionably, there is still great uncertainty on the different aspects related to forest restoration after wildfires: the identification of post-fire soil severity, the selection of the most suitable intervention, the chronical lack of funds and appropriate regulation, the limited social awareness, the interaction between traditional and modern knowledge, the provision of land management tools that reconcile sustainable land use and human welfare, the extent to which climate change will impact environmental conditions, or the increasing presence of IAPs. The recognition of fire as a basic element of many ecosystems and the consequent distinction between fire-sensitive and fire-prone systems, becomes also fundamental to make appropriate decisions on restoration. These aspects and their interactions represent consistent limitations for ER, and nearly all of them are exemplified in the paradigmatic case of the Amazonian rainforest. However, we now have a better understanding of the causes and effects of wildfires, and a growing awareness on the importance of forest conservation and recovery.

Compared to other regions, LAC still has a vast area of natural forests with great potential for post-fire recovery. While different strategies have been adapted from regions where ER has a solid and stable trajectory, the variety and uniqueness of the different biomes and ecosystems in LAC demand unique restoration actions and should become a reference by itself. To successfully implement ER initiatives, each region must find specific formulas adapted to its reality based on interdisciplinary approaches that consider several dimensions (technical, strategic, social, economic, political, etc.). It is also important to highlight the relatively recent introduction of national regulations on ER, the increasing positive perception regarding the use of prescribed fire to manage fire-prone ecosystems, the concern on IAPs expansion or the creation of national and international networks to collaborate on ER initiatives.

Funding

PS was funded by the Regional Council of Education, University and Professional Training (Consellería de Educación, Universidade e Formación Profesional) from the Government of Galicia (Xunta de Galicia) through the Postdoctoral Plan "Axudas de apoio á etapa de formación posdoutoral nas universidades do Sistema universitario de Galicia", (Ref - ED481B-2019-088). GS was funded by the Fondo Nacional de Desarrollo Científico y Tecnológico (Fondecyt Regular REF-1191905), and the "Fondo Interno para la Adquisición de Equipamiento Científico de la Universidad Católica de la Santísima Concepción (FIAEC 2019). RG was funded by Fondecyt (REF - 11170516). RG, AP were funded by Grant ANID PIA/BASAL FB210006. AF was funded by CERNAS, Centro de Estudos de Recursos Naturais, Ambiente e Sociedade (FCT-UID00681-2020-2023).

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The authors are very grateful for the institutional and financial support of the different entities mentioned in the Funding section, without which this work would not have been possible. Also, authors sincerely thank the two anonymous reviewers for their helpful comments that significantly improve the final version of this work.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2022.120083.

References

Acevedo, M., Álvarez-Maldini, C., Dumroese, R.K., Bannister, J.R., Cartes, E., González, M., 2021. Native plant production in Chile. Is it possible to achieve restoration goals by 2035? Land. 10 (1), 71. https://doi.org/10.3390/land10010071.

Aguirre-Monroy, A.M., Santana-Martínez, J.C., Dussán, J., 2019. *Lysinibacillus sphaericus* as a Nutrient Enhancer during Fire-Impacted Soil Replantation. Appl. Environ. Soil Sci. 2019, 1–8.

Aide, T.M., Clark, M.L., Grau, H.R., López-Carr, D., Levy, M.A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Núñez, M.J., Muñiz, M., 2013. Deforestation and reforestation of Latin America and the Caribbean (2001–2010). Biotropica 45 (2), 262–271.

Alanís-Rodríguez, E., Jiménez-Pérez, J., Espinoza-Vizcarra, D., Jurado-Ybarra, E., Aguirre-Calderón, O.A., González-Tagle, M.A., 2008. Evaluación del estrato arbóreo en un área restaurada post-incendio en el Parque Ecológico Chipinque. México. Rev. Chapingo. 14, 113–118.

Alanís-Rodríguez, E., Valdecantos-Dema, A., Jiménez-Pérez, J., Rubio-Camacho, E.A., Yerena-Yamallel, J.I., González-Tagle, M.A., 2015. Post-fire ecological restoration of a mixed *Pinus-Quercus* forest in northeastern Mexico. Rev. Chapingo. 21, 157–170.

Alauzis, M.V., Mazzarino, M.J., Raffaele, E., Roselli, Lucía, 2004. Wildfires in NW Patagonia: long-term effects on a *Nothofagus* forest soil. Forest Ecol. Manag. 192 (2-3), 131–142.

Albanesi, S., Dardanelli, S., Bellis, L.M., 2014. Effects of fire disturbance on bird communities and species of mountain Serrano forest in central Argentina. J. Forest Res. 19 (1), 105–114.

Alberton, B., Torres, R.d.S., Cancian, L.F., Borges, B.D., Almeida, J., Mariano, G.C., Santos, J.D., Morellato, L.P.C., 2017. Introducing digital cameras to monitor plant phenology in the tropics: applications for conservation. Persp. Ecol. Conser. 15 (2), 82–90.

Albornoz, F.E., Gaxiola, A., Seaman, B.J., Pugnaire, F.I., Armesto, J.J., 2013. Nucleation-driven regeneration promotes post-fire recovery in a Chilean temperate forest. Plant Ecol. 214 (5), 765–776.

Alencar, A.A.C., Solórzano, L.A., Nepstad, D.C., 2004. Modeling forest understory fires in an Eastern Amazonian landscape. Ecol. Appl. 14 (sp4), 139–149.

Alexander, J.M., D'Antonio, C.M., 2003. Control methods for the removal of French and Scotch Broom tested in coastal California. Ecol. Rest. 21 (3), 191–198.

Allen, E.B., Allen, M.F., Egerton-Warburton, L., Corkidi, L., Gómez-Pompa, A., 2003. Impacts of early- and late-seral mycorrhizae during restoration in seasonal tropical forest. Mexico. Ecol. Appl. 13 (6), 1701–1717.

Allen, M.F., Allen, E.B., Gomez-Pompa, A., 2005. Effects of mycorrhizae and nontarget organisms on restoration of a seasonal tropical forest in Quintana Roo, Mexico: Factors limiting tree establishment. Rest. Ecol. 13 (2), 325–333.

Altamirano, A., Cely, J.P., Etter, A., Miranda, A., Fuentes-Ramirez, A., Acevedo, P., Salas, C., Vargas, R., 2016. The invasive species *Ulex europaeus* (Fabaceae) shows high dynamism in a fragmented landscape of south-central Chile. Environ. Monit. Assess. 188 (8) https://doi.org/10.1007/s10661-016-5498-6.

Aragão, L.E.O.C., Malhi, Y., Roman-Cuesta, R.M., Saatchi, S., Anderson, L.O., Shimabukuro, Y.E., 2007. Spatial patterns and fire response of recent Amazonian droughts. Geophys. Res. Lett. 34 (7) https://doi.org/10.1029/2006GL028946.

Aragão, L.E.O.C., Anderson, L.O., Fonseca, M.G., Rosan, T.M., Vedovato, L.B., Wagner, F. H., Silva, C.V.J., Silva Junior, C.H.L., Arai, E., Aguiar, A.P., Barlow, J., Berenguer, E., Deeter, M.N., Domingues, L.G., Gatti, L., Gloor, M., Malhi, Y., Marengo, J.A., Miller, J.B., Phillips, O.L., Saatchi, S., 2018. 21st Century drought-related fires

counteract the decline of Amazon deforestation carbon emissions. Nature Comm. 9 (1) https://doi.org/10.1038/s41467-017-02771-y.
Archibald, S., Staver, A.C., Levin, S.A., 2012. Evolution of human-driven fire regimes in Africa. P. Natl. Acad. Sci. USA 109 (3), 847–852.
Armesto, J.J., Bautista, S., Del Val, E.k., Ferguson, B., García, X., Gaxiola, A., Godinez-Álvarez, H., Gann, G., López-Barrera, F., Manson, R., Núñez-Ávila, M., Ortiz-Arrona, C., Tognetti, P., Williams-Linera, G., 2007. Towards an ecological restoration network: reversing land degradation in Latin America. Front. Ecol. Environ. 5 (4), w1–w4.
Armesto, J.J., Manuschevich, D., Mora, A., Smith-Ramirez, C., Rozzi, R., Abarzúa, A.M., Marquet, P.A., 2010. From the Holocene to the Anthropocene: A historical framework for land cover change in southwestern South America in the past 15,000 years. Land Use Policy. 27 (2), 148–160.
Aronson, J., Alexander, S., 2013. Ecosystem restoration is now a global priority: Time to roll up our sleeves. Restor. Ecol. 21 (3), 293–296.
Arroyo-Kalin, M., 2012. Slash-burn-and-churn: Landscape history and crop cultivation in pre-Columbian Amazonia. Quatern. Int. 249, 4–18.
Bannister, J.R., 2015. Recuperar bosques no es solo plantar árboles: lecciones aprendidas luego de 7 años restaurando bosques de *Pilgerodendron uviferum* (D. Don) Florin en Chiloé. An. Inst Patagonia. 43 (1), 35–51.
Bannister, J.R., Donoso, P.J., Bauhus, J., 2012. Persistence of the slow growing conifer *Pilgerodendron uviferum* in old-growth and fire-disturbed southern bog forests. Ecosystems 15, 1158–1172.
Bannister, J.R., Coopman, R.E., Donoso, P.J., Bauhus, J., 2013. The importance of microtopography and nurse canopy for successful restoration planting of the slow-growing conifer *Pilgerodendron uviferum*. Forests. 4, 85–103.
Bannister, J.R., Donoso, P.J., Mujica, R., 2016. La silvicultura como herramienta para la restauración de bosques templados. Bosque. 37 (2), 229–235.
Bannister, J.R., Vargas-Gaete, R., Ovalle, J.F., Acevedo, M., Fuentes-Ramirez, A., Donoso, P.J., Promis, A., Smith-Ramírez, C., 2018. Major bottlenecks for the restoration of natural forests in Chile. Restor. Ecol. 26 (6), 1039–1044.
Bannister, J.R., Travieso, G., Galindo, N., Acevedo, M., Puettmann, K., Salas-Eljatib, C., 2020. Shrub influences on seedling performance when restoring the slow-growing conifer *Pilgerodendron uviferum* in southern bog forests. Restor. Ecol. 28 (2), 396–407.
Barlow, J., Peres, C.A., 2006. Effects of single and recurrent wildfires on fruit production and large vertebrate abundance in a central Amazonian forest. Biodivers. Conserv 15 (3), 985–1012.
Barlow, J., Berenguer, E., Carmenta, R., França, F., 2020. Clarifying Amazonia's burning crisis. Glob. Change Biol. 26 (2), 319–321.
Barnosky, A.D., Hadly, E.A., Gonzalez, P., Head, J., Polly, P.D., Lawing, A.M., Eronen, J. T., Ackerly, D.D., Alex, K., Biber, E., Blois, J., Brashares, J., Ceballos, G., Davis, E., Dietl, G.P., Dirzo, R., Doremus, H., Fortelius, M., Greene, H.W., Hellmann, J., Hickler, T., Jackson, S.T., Kemp, M., Koch, P.L., Kremen, C., Lindsey, E.L., Looy, C., Marshall, C.R., Mendenhall, C., Mulch, A., Mychajliw, A.M., Nowak, C., Ramakrishnan, U., Schnitzler, J., Das Shrestha, K., Solari, K., Stegner, L., Stegner, M. A., Stenseth, N.C., Wake, M.H., Zhang, Z., 2017. Merging paleobiology with conservation biology to guide the future of terrestrial ecosystems. Science 355 (6325), eaah4787. https://doi.org/10.1126/science:aah4787.
Béliveau, A., Davidson, R., Lucotte, M., Do canto lopes, L.OTÁVIO., Paquet, S., Vasseur, C., 2015. Early effects of slash-and-burn cultivation on soil physicochemical properties of small-scale farms in the Tapajós region, Brazilian Amazon. J. Agr. Sci. 153 (2), 205–221.
BenDor, T., Lester, T.W., Livengood, A., Davis, A., Yonavjak, L., Hernandez Montoya, A. R., 2015. Estimating the size and impact of the ecological restoration economy. PLoS ONE 10 (6), e0128339. https://doi.org/10.1371/journal.pone.0128339.
Bertolin, M.L., Urretavizcaya, M.F., Defossé, G.E., 2015. Fire emissions and carbon uptake in severely burned lenga beech (*Nothofagus pumilio*) forests of Patagonia. Argentina. Fire Ecol. 11 (1), 32–54.
Bezdek, R.H., Wendling, R.M., DiPerna, P., 2008. Environmental protection, the economy, and jobs: National and regional analyses. J. Environ. Manage. 86 (1), 63–79.
Bird, M.I., Wynn, J.G., Saiz, G., Wurster, C.M., McBeath, A., 2015. The pyrogenic carbon cycle. Annu. Rev. Earth Pl. Sci. 43 (1), 273–298.
Blackhall, M., Raffaele, E., Veblen, T.T., 2008. Cattle affect early post-fire regeneration in a *Nothofagus dombeyi–Austrocedrus chilensis* mixed forest in northern Patagonia. Argentina. Biol. Conser. 141 (9), 2251–2261.
Blackhall, M., Raffaele, E., Veblen, T.T., 2015. Combined effects of fire and cattle in shrublands and forests of northwest Patagonia. Ecología Austral. 25, 1–10.
Blanco-García, A., Sáenz-Romero, C., Martorell, C., Alvarado-Sosa, P., Lindig-Cisneros, R., 2011. Nurse-plant and mulching effects on three conifer species in a Mexican temperate forest. Ecol. Engin. 37 (6), 994–998.
Blignaut, J., Aronson, J., 2020. Developing a restoration narrative: A pathway towards system-wide healing and a restorative culture. Ecol. Econ. 168, 106483. https://doi.org/10.1016/j.ecolecon.2019.106483.
Bloomfield, G., Meli, P., Brancalion, P.H., Terris, E., Guariguata, M.R., Garen, E., 2019. Strategic insights for capacity development on forest landscape restoration: implications for addressing global commitments. Trop. Conserv. Sci. 12, 1–11.
Braasch, M., García-Barrios, L., Ramírez-Marcial, N., Huber-Sannwald, E., Cortina-Villar, S., 2017. Can cattle grazing substitute fire for maintaining appreciated pine savannas at the frontier of a montane forest biosphere-reserve? Agr. Ecosyst. Environ. 250, 59–71.
Bradshaw, C.J., Ehrlich, P.R., Beattie, A., Ceballos, G., Crist, E., Diamond, J., et al., 2021. Underestimating the challenges of avoiding a ghastly future. Front. Conserv. Sci. 1, 9.
Brando, P.M., Balch, J.K., Nepstad, D.C., Morton, D.C., Putz, F.E., Coe, M.T., Silverio, D., Macedo, M.N., Davidson, E.A., Nobrega, C.C., Alencar, A., Soares-Filho, B.S., 2014. Abrupt increases in Amazonian tree mortality due to drought-fire interactions. P. Natl. Acad. Sci USA 111 (17), 6347–6352.
Brooks, M.L., D'antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., DiTOMASO, J. M., Hobbs, R.J., Pellant, MIKE, Pyke, DAVID, 2004. Effects of invasive alien plants on fire regimes. Bioscience 54 (7), 677. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2.
Bustamante, R.O., Serey, I.A., Pickett, S.T.A., 2003. Forest fragmentation, plant regeneration and invasion processes across edges in central Chile. In: How landscapes change. Springer, Berlin, Heidelberg, pp. 145–160.
Calcaterra, L.A., Di Blanco, Y., Srur, M., Briano, J., 2014. Fire effect on ground-foraging ant assemblages in northeastern Argentina. J. Insect Conserv. 18 (3), 339–352.
Calva-Soto, K., Pavón Hernández, N.P., 2018. La restauración ecológica en México: una disciplina emergente en un país deteriorado. Madera y bosques. 24 (1) https://doi.org/10.21829/myb.2018.2411135.
Cantarello, E., Newton, A.C., Hill, R.A., Tejedor-Garavito, N., Williams-Linera, G., López-Barrera, F., Manson, R.H., Golicher, D.J., 2011. Simulating the potential for ecological restoration of dryland forests in Mexico under different disturbance regimes. Ecol. Model. 222 (5), 1112–1128.
Capulin-Grande, J., Suárez-Islas, A., Rodríguez-Laguna, R., Mateo-Sánchez, J.J., Razo-Zárate, R., Islas-Santillán, M., 2018. Influence of fire on soil and vegetation properties in two contrasting forest sites in Central México. Int. J. Agr. Nat. Res. 45 (2), 128–137.
Carbajal-Navarro, A., Navarro-Miranda, E., Blanco-García, A., Cruzado-Vargas, A.L., Gómez-Pineda, E., Zamora-Sánchez, C., Pineda-García, F., O'Neill, G., Gómez-Romero, M., Lindig-Cisneros, R., Johnsen, K.H., Lobit, P., Lopez-Toledo, L., Herrerías-Diego, Y., Sáenz-Romero, C., 2019. Ecological restoration of *Abies religiosa* forests using nurse plants and assisted migration in the Monarch Butterfly Biosphere Reserve, Mexico. Front. Ecol. Evol. 7 https://doi.org/10.3389/fevo.2019.00421.
Cardil, A., De-Miguel, S., Silva, C.A., Reich, P.B., Calkin, D., Brancalion, P.H., et al., 2020. Recent deforestation drove the spike in Amazonian fires. Environ. Res. Lett. 15, 121003.
Carrasco, B., González, P., Marín, C., Cobos, C., Valenzuela, J., Rojas-Alvarado, C., 2019. Recuperación de salud de suelos y su implicancia en el restablecimiento de bosque nativo incendiado en zonas del secano de la VI Región. 10.13140/RG.2.2.30690.68806.
Castillo, J.A., Smith-Ramírez, C., Claramunt, V., 2021. Differences in stakeholder perceptions about native forest: implications for developing a restoration program. Restor. Ecol. 29 (1) https://doi.org/10.1111/rec.v29.110.1111/rec.13293.
Castro, J., Zamora, R., Hodar, J.A., Gomez, J.M., 2002. Use of shrubs as nurse plants: a new technique for reforestation in Mediterranean mountains. Restor. Ecol. 10 (2), 297–305.
Castro, J., Allen, C.D., Molina-Morales, M., Marañón-Jiménez, S., Sánchez-Miranda, Á., Zamora, R., 2011. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. Restor. Ecol. 19 (4), 537–544.
Cavallero, L., Raffaele, E., Aizen, M.A., 2013. Birds as mediators of passive restoration during early post-fire recovery. Biol. Conserv. 158, 342–350.
CBD, Convention on Biological Diversity, 2014. Conference of the parties to the Convention on Biological Diversity. XII meeting, Korea. UNEP/CBD/COP/DEC/XII/19.
Ceccon, E., Martínez-Garza, C., 2016. Experiencias mexicanas en la restauración de los ecosistemas. Universidad Nacional Autónoma de México, Centro Regional de Investigaciones Multidisciplinarias; Universidad Autónoma del Estado de Morelos; Ciudad de México, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad.
Celentano, D., Miranda, M.V., Mendonça, E.N., Rousseau, G.X., Muniz, F.H., Loch, V.D. C., et al., 2018. Desmatamento, degradação e violencia no "Mosaico Gurupi", a região mais ameaçada da Amazônia. Estudos Avançados. 32, 315–339.
Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143 (1), 1–10.
Cesca, E.M., Villagra, P.E., Alvarez, J.A., 2014. From forest to shrubland: Structural responses to different fire histories in *Prosopis flexuosa* woodland from the Central Monte (Argentina). J. Arid Environ. 110, 1–7.
Céspedes, M., Gutierrez, M.V., Holbrook, N.M., J. Rocha, O., 2003. Restoration of genetic diversity in the dry forest tree Swietenia macrophylla (Meliaceae) after pasture abandonment in Costa Rica. Mol. Ecol. 12 (12), 3201–3212.
Chávez, D., Machuca, Á., Fuentes-Ramirez, A., Fernandez, N., Cornejo, P., 2020. Shifts in soil traits and arbuscular mycorrhizal symbiosis represent the conservation status of *Araucaria araucana* forests and the effects after fire events. Forest Ecol. Manag. 458, 117806. https://doi.org/10.1016/j.foreco.2019.117806.
Chazdon, R.L., Guariguata, M.R., 2016. Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. Biotropica 48 (6), 716–730.
Chen, Y., Morton, D.C., Jin, Y., Collatz, G.J., Kasibhatla, P.S., van der Werf, G.R., DeFries, R.S., Randerson, J.T., 2013. Long-term trends and interannual variability of forest, savanna and agricultural fires in South America. Carbon Manag. 4 (6), 617–638.
Cingolani, A.M., Vaieretti, M.V., Giorgis, M.A., La Torre, N., Whitworth-Hulse, J.I., Renison, D., 2013. Can livestock and fires convert the sub-tropical mountain rangelands of central Argentina into a rocky desert? Rangeland J. 35 (3), 285. https://doi.org/10.1071/RJ12095.
Clewell, A.F., Aronson, J., 2013. Ecological restoration: principles, values, and structure of an emerging profession. Island Press, Washington.
Cóbar-Carranza, A.J., García, R.A., Pauchard, A., Peña, E., 2014. Effect of *Pinus contorta* invasion on forest fuel properties and its potential implications on the fire regime of

Araucaria araucana and *Nothofagus antarctica* forests. Biol. Invasions 16 (11), 2273–2291.
Cochrane, M.A., Alencar, A., Schulze, M.D., Souza, C.M., Nepstad, D.C., Lefebvre, P., Davidson, E.A., 1999. Positive feedbacks in the fire dynamic of closed canopy tropical forests. Science 284 (5421), 1832–1835.
Cochrane, M.A., Barber, C.P., 2009. Climate change, human land use and future fires in the Amazon. Glob. Change Biol. 15 (3), 601–612.
CONAF, Corporación Nacional Forestal, 2015. Ley sobre recuperación del bosque nativo y fomento forestal y reglamentos. Editorial CONAF, Santiago, Chile.
Contreras, T., Figueroa, J., Abarca, L., Castro, S., 2011. Fire regimes and spread of plants naturalized in central Chile. Rev. Chil. Hist. Nat. 84, 307–323.
Cortina-Segarra, J., García-Sánchez, I., Grace, M., Andrés, P., Baker, S., Bullock, C., Decleer, K., Dicks, L.V., Fisher, J.L., Frouz, J., Klimkowska, A., Kyriazopoulos, A.P., Moreno-Mateos, D., Rodríguez-González, P.M., Sarkki, S., Ventocilla, J.L., 2021. Barriers to ecological restoration in Europe: expert perspectives. Restor. Ecol. 29 (4) https://doi.org/10.1111/rec.v29.410.1111/rec.13346.
Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Glob. Environ. Change. 26, 152–158.
Crouzeilles, R., Ferreira, M.S., Chazdon, R.L., Lindenmayer, D.B., Sansevero, J.B.B., Monteiro, L., Iribarrem, A., Latawiec, A.E., Strassburg, B.B.N., 2017. Ecological restoration success is higher for natural regeneration than for active restoration in tropical forests. Sci. Adv. 3 (11) https://doi.org/10.1126/sciadv.1701345.
Cruz, R.E., Segura, R.B., 2010. Developing the bioliteracy of school children for 24 years: a fundamental tool for ecological restoration and conservation in perpetuity of the Area de Conservación Guanacaste, Costa Rica. Ecol. Restor. 28, 193–198.
Cullen Jr, L., Schmink, M., Pádua, C.V., Morato, M.I.R., 2001. Agroforestry benefit zones: a tool for the conservation and management of Atlantic forest fragments, Sao Paulo. Brazil. Nat Areas J. 21, 346–356.
Curtis, P.G., Slay, C.M., Harris, N.L., Tyukavina, A., Hansen, M.C., 2018. Classifying drivers of global forest loss. Science 361 (6407), 1108–1111.
Davidson, E.A., de Araújo, A.C., Artaxo, P., Balch, J.K., Brown, I.F., C. Bustamante, M.M., Coe, M.T., DeFries, R.S., Keller, M., Longo, M., Munger, J.W., Schroeder, W., Soares-Filho, B.S., Souza, C.M., Wofsy, S.C., 2012. The Amazon basin in transition. Nature 481 (7381), 321–328.
de Andrade, R.B., Balch, J.K., Carreira, J.Y.O., Brando, P.M., Freitas, A.V.L., 2017. The impacts of recurrent fires on diversity of fruit-feeding butterflies in a south-eastern Amazon forest. J. Trop. Ecol. 33 (1), 22–32.
de la Barrera, F., Barraza, F., Favier, P., Ruiz, V., Quense, J., 2018. Megafires in Chile 2017: Monitoring multiscale environmental impacts of burned ecosystems. Sci. Total Environ. 637-638, 1526–1536.
de la Rosa, J.M., Merino, A., Jiménez Morillo, N.T., Jiménez-González, M.A., González-Pérez, J.A., González-Vila, F.J., et al., 2019. Unveiling the effects of fire on soil organic matter by spectroscopic and thermal degradation methods. In: Pereira, P., Mataix-Solera, J., Übeda, X., Rein, G., Cerdà, A. (Eds.), Fire Effects in soil properties. CSIRO Publishing, Australia, pp. 281–307.
Eva, H.D., Belward, A.S., De Miranda, E.E., Di Bella, C.M., Gond, V., Huber, O., Jones, S., Sgrenzaroli, M., Fritz, S., 2004. A land cover map of South America. Glob. Change Biol. 10 (5), 731–744.
dos Santos, J.F.C., Gleriani, J.M., Velloso, S.G.S., de Souza, G.S.A., do Amaral, C.H., Torres, F.T.P., Medeiros, N.D.G., dos Reis, M., 2019. Wildfires as a major challenge for natural regeneration in Atlantic Forest. Sci. Total Environ. 650, 809–821.
Echeverría, C., Smith-Ramírez, C., Aronson, J., Barrera-Cataño, J.I., 2015. Good news from Latin America and the Caribbean: national and international restoration networks are moving ahead. Restor. Ecol. 23 (1), 1–3.
EFTEC, ECNC, UAntwerp, CEEWEB, 2017. Promotion of ecosystem restoration in the context of the EU biodiversity strategy to 2020. Report to European Commission, DG Environment.
EC, European Commission, 2020. EU Biodiversity Strategy for 2030. Bringing nature back into our lives. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. COM (2020). https://ec.europa.eu/info/strategy/priorities-2019-2024/european-greendeal/actions-being-taken-eu/eu-biodiversity-strategy-2030_en.
Falkowski, T.B., Chankin, A., Diemont, S.A.W., 2020. Successional changes in vegetation and litter structure in traditional Lacandon Maya agroforests. Agroecol. Sust Food. 44 (6), 747–767.
FAO. 2020. Global Forest Resources Assessment 2020: Main report. Rome.
FAO, UNEP, 2020. The state of the World's Forests 2020. Forests, biodiversity and people. Rome.
Ferreira, A.J.D., Coelho, C.O.A., Ritsema, C.J., Boulet, A.K., Keizer, J.J., 2008. Soil and water degradation processes in burned areas: lessons learned from a nested approach. Catena 74 (3), 273–285.
Ferreira, A.J.D., Alegre, S.P., Coelho, C.O.A., Shakesby, R.A., Páscoa, F.M., Ferreira, C.S. S., Keizer, J.J., Ritsema, C., 2015. Strategies to prevent forest fires and techniques to reverse degradation processes in burned areas. Catena 128, 224–237.
Fisher, J.L., Cortina-Segarra, J., Grace, M., Moreno-Mateos, D., Rodríguez González, P. M., Baker, S., et al., 2019. What is hampering current restoration effectiveness? Report prepared by an EKLIPSE Expert Working Group. UK Centre for Ecology & Hydrology, Wallingford, United Kingdom.
Franzese, J., Urrutia, J., García, R.A., Taylor, K., Pauchard, A., 2017. Pine invasion impacts on plant diversity in Patagonia: invader size and invaded habitat matter. Biol. Invasions. 19 (3), 1015–1027.
Fremout, T., Gutiérrez-Miranda, C.E., Briers, S., Marcelo-Peña, J.L., Cueva-Ortiz, E., Linares-Palomino, R., La Torre-Cuadros, M.d.L.Á., Chang-Ruíz, J.C., Villegas-Gómez, T.L., Acosta-Flota, A.H., Plouvier, D., Atkinson, R., Charcape-Ravelo, M., Aguirre-Mendoza, Z., Muys, B., Thomas, E., 2021. The value of local ecological knowledge to guide tree species selection in tropical dry forest restoration. Restor. Ecol. 29 (4) https://doi.org/10.1111/rec.v29.410.1111/rec.13347.
Fuentes-Ramirez, A., Barrientos, M., Almonacid, L., Arriagada-Escamilla, C., Salas-Eljatib, C., 2018. Short-term response of soil microorganisms, nutrients and plant recovery in fire-affected *Araucaria araucana* forests. Appl Soil Ecol. 131, 99–106.
Galindo, V., Calle, Z., Chará, J., Armbrecht, I., 2017. Facilitation by pioneer shrubs for the ecological restoration of riparian forests in the Central Andes of Colombia. Restor Ecol. 25 (5), 731–737.
Gallegos, S.C., Beck, S.G., Hensen, I., Saavedra, F., Lippok, D., Schleuning, M., 2016. Factors limiting montane forest regeneration in bracken-dominated habitats in the tropics. Forest Ecol. Manag. 381, 168–176.
Gallegos, S.C., Hensen, I., Saavedra, F., Schleuning, M., 2015. Bracken fern facilitates tree seedling recruitment in tropical fire-degraded habitats. Forest Ecol. Manag. 337, 135–143.
Gann, G.D., McDonald, T., Walder, B., Aronson, J., Nelson, C.R., Jonson, J., 2019. International principles and standards for the practice of ecological restoration. Restor. Ecol. 27, S1–S46.
García, R.A., Engler, M.L., Peña, E., Pollnac, F.W., Pauchard, A., 2015. Fuel characteristics of the invasive shrub *Teline* monspessulana (L.) K. Koch. Int. J. Wildland Fire 24 (3), 372. https://doi.org/10.1071/WF13078.
Giglio, L., Randerson, J.T., van der Werf, G.R., 2013. Analysis of daily, monthly, and annual burned area using the fourth-generation global fire emissions database (GFED4). J. Geophys. Res-Biogeo. 118 (1), 317–328.
Giri, C., Long, J., 2014. Land cover characterization and mapping of South America for the year 2010 using Landsat 30 m satellite data. Remote Sens. 6 (10), 9494–9510.
Gomes, L., Maracahipes, L., Marimon, B.S., Reis, S.M., Elias, F., Maracahipes-Santos, L., Marimon-Junior, B.H., Lenza, E., 2014. Post-fire recovery of savanna vegetation from rocky outcrops. Flora. 209 (3-4), 201–208.
Gomes, L., Miranda, H.S., Bustamante, M.M.d.C., 2018. How can we advance the knowledge on the behavior and effects of fire in the Cerrado biome? Forest Ecol. Manag. 417, 281–290.
Gómez, P., Bustamante, R., San Martín, J., Hahn, S., 2011. Estructura poblacional de *Pinus radiata* D. Don en fragmentos de Bosque Maulino en Chile central. Gayana Bot. 68 (1), 97–101.
Gómez-Aparicio, L., Zamora, R., Gómez, J.M., Hódar, J.A., Castro, J., Baraza, E., 2004. Applying plant facilitation to forest restoration: a meta-analysis of the use of shrubs as nurse plants. Ecol Appl. 14 (4), 1128–1138.
Gómez-González, S., Paula, S., Cavieres, L.A., Pausas, J.G., Armas, C., 2017. Postfire responses of the woody flora of Central Chile: Insights from a germination experiment. PLoS ONE 12 (7), e0180661. https://doi.org/10.1371/journal.pone.0180661.
González, M.E., 2005. Fire history data as reference information in ecological restoration. Dendrochronologia. 22 (3), 149–154.
González, M.E., Veblen, T.T., 2007. Wildfire in *Araucaria araucana* forests and ecological considerations about salvage logging in areas recently burned. Rev. Chil. Hist. Nat. 80, 243–253.
González, M.E., Szejner, P., Donoso, P.J., Salas, C., 2015. Fuego, madereo y patrones de establecimiento de bosques secundarios en el centro-sur de Chile: implicaciones para su manejo y restauración. Ciencia e Investigación Agraria 42, 415–425.
González, M.E., Sapiains, R., Gómez-González, S., Garreaud, R., Miranda, A., Galleguillos, M., et al., 2020. Incendios forestales en Chile: causas, impactos y resiliencia. Universidad de Chile, Universidad de Concepción y Universidad Austral de Chile.
González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fire on soil organic matter-a review. Environ. Int. 30 (6), 855–870.
González-Tagle, M.A., Schwendenmann, L., Pérez, J.J., Schulz, R., 2008. Forest structure and woody plant species composition along a fire chronosequence in mixed pine–oak forest in the Sierra Madre Oriental. Northeast Mexico. Forest Ecol. Manag. 256 (1-2), 161–167.
Guerrero-Gatica, M., Mujica, M.I., Barceló, M., Vio-Garay, M.F., Gelcich, S., Armesto, J. J., 2020. Traditional and local knowledge in Chile: review of experiences and insights for management and sustainability. Sustainability. 12 (5), 1767. https://doi.org/10.3390/su12051767.
Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O., Townshend, J.R.G., 2013. High-resolution global maps of 21st-century forest cover change. Science 342 (6160), 850–853.
Hepburn, C., O'Callaghan, B., Stern, N., Stiglitz, J., Zenghelis, D., 2020. Will COVID-19 fiscal recovery packages accelerate or retard progress on climate change? Oxford Rev. Econ. Pol. 36, S359–S381.
Hernández Vallecillo, G.A., Gutiérrez Castorena, M., Barragán Maravilla, S.M., Ángeles Cervantes, E.R., Gutiérrez Castorena, E.V., Ortiz Solorio, C.A., 2020. La mineralogía en la estimación de las temperaturas de los incendios forestales y sus efectos inmediatos en Andosoles, Estado de México. Madera y bosques, 26.
Hobbs, R.J., 2016. Degraded or just different? Perceptions and value judgements in restoration decisions. Restor. Ecol. 24 (2), 153–158.
Hoffmann, W.A., Moreira, A.G., 2002. The role of fire in population dynamics of woody plants. In: Oliveira, O.S., Marquis, R.J. (Eds.), The Cerrados of Brazil. Columbia University Press, New York, USA, pp. 159–177.
Holden, Z.A., Morgan, P., Smith, A.M.S., Vierling, L., 2010. Beyond Landsat: a comparison of four satellite sensors for detecting burn severity in ponderosa pine forests of the Gila Wilderness, NM, USA. Int. J. Wildland Fire. 19 (4), 449. https://doi.org/10.1071/WF07106.
Holdridge, L.R., 1947. Determination of world plant formations. Science 105, 367–368.

Holl, K.D., Aide, T.M., 2011. When and where to actively restore ecosystems? Forest Ecol. Manag. 261 (10), 1558–1563.
Holmes, P.M., Richardson, D.M., Wilgen, B.W., Gelderblom, CAROLINE, 2000. Recovery of South African fynbos vegetation following alien woody plant clearing and fire: implications for restoration. Austral Ecol. 25 (6), 631–639.
Huang, Y., Wu, S., Kaplan, J.O., 2015. Sensitivity of global wildfire occurrences to various factors in the context of global change. Atm. Environ. 121, 86–92.
Huffman, M.R., 2010. Community-based fire management at La Sepultura Biosphere Reserve, Chiapas, Mexico. PhD Dissertation. Colorado State University, Fort Collins, USA.
IPCC, 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. In: Field, C.B., Barros, V.R., Dokken, D.J., Mach, K.J., Mastrandrea, M.D., Bilir, T.E., et al. (Eds), Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Iucn, 2010. IUCN Red List of Threatened Species. Version 2010, 2. http://www.iucnredlist.org.
Ivanauskas, N.M., Monteiro, R., Rodrigues, R.R., 2003. Alterations following a fire in a forest community of Alto Rio Xingu. Forest Ecol Manag. 184 (1-3), 239–250.
Jaeger, J., G. Walls, E. Clarke, J.C. Altamirano, A. Harsono, H. Mountford, S., 2021. The green jobs advantage: How climate-friendly investments are better job creators. Working Paper. Washington, DC: World Resources Institute.
Jardel, E.J., Ramírez, V.R., Castillo, N.F., García, R.S., Balcázar-Medina, O.E., Chacón-Mathieu, J.C., Morfín, J.E., 2006. Manejo del fuego y restauración de bosques en la Reserva de la Biosfera Sierra de Manantlán, México. In: Flores-Garnica, J.G., Rodríguez-Trejo, D.A. (Eds.), Incendios Forestales. Mundi Prensa-CONAFOR, México D.F. y Madrid, pp. 214–242.
Jaureguiberry, P., Díaz, S., 2015. Post-burning regeneration of the Chaco seasonally dry forest: germination response of dominant species to experimental heat shock. Oecologia 177 (3), 689–699.
Jia, G., Shevliakova, E., Artaxo, P., De Noblet-Ducoudré, N., Houghton, R., House, J., et al., 2019. Land–climate interactions. In: Shukla, P.R., Skea, J., Calvo Buendia, E., Masson-Delmotte, V., Pörtner, H.-O., Roberts, D.C., et al. (Eds.), Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems. Intergovernmental Panel on Climate Change, pp. 131–247.
Johnson, P.N., 2001. Vegetation recovery after fire on a southern New Zealand peatland. New Zeal. J. Bot. 39 (2), 251–267.
Jolly, W.M., Cochrane, M.A., Freeborn, P.H., Holden, Z.A., Brown, T.J., Williamson, G.J., Bowman, D.M., 2015. Climate-induced variations in global wildfire danger from 1979 to 2013. Nature Comm. 6, 1–11.
Juárez-Orozco, S.M., Siebe, C., Fernández y Fernández, D., 2017. Causes and effects of forest fires in tropical rainforests: a bibliometric approach. Trop. Conserv. Sci. 10, 1940082917737207.
Kattenborn, T., Lopatin, J., Förster, M., Braun, A.C., Fassnacht, F.E., 2019. UAV data as alternative to field sampling to map woody invasive species based on combined Sentinel-1 and Sentinel-2 data. Remote Sens. Environ. 227, 61–73.
Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. Int. J. Wildland Fire 18 (1), 116. https://doi.org/10.1071/WF07049.
Kernan, B.S., Cordero, W., Macedo Sienra, A.M., 2010. Report on Biodiversity and Tropical Forests in Paraguay. USAID Tropical Forests and Biodiversity Assessment, Washington, D.C.
Kerr, J., Vardhan, M., Jindal, R., 2014. Incentives, conditionality and collective action in payment for environmental services. Int. J. Comm. 8.
Krawchuk, M.A., Moritz, M.A., Parisien, M.-A., Van Dorn, J., Hayhoe, K., Chave, J., 2009. Global pyrogeography: the current and future distribution of wildfire. PLoS ONE 4 (4), e5102. https://doi.org/10.1371/journal.pone.0005102.
Kremer, N.J., Halpern, C.B., Antos, J.A., 2014. Conifer reinvasion of montane meadows following experimental tree removal and prescribed burning. Forest Ecol. Manag. 319, 128–137.
Kubiszewski, I., Costanza, R., Anderson, S., Sutton, P., 2020. The future value of ecosystem services: Global scenarios and national implications. In: Ninan, K.N. (Ed.), Environmental Assessments. Edward Elgar Publishing, UK, pp. 81–108.
Landesmann, J.B., Morales, J.M., 2018. The importance of fire refugia in the recolonization of a fire-sensitive conifer in northern Patagonia. Plant Ecol. 219 (4), 455–466.
Lara, A., Echeverría, C., Thiers, O., Huss, E., Escobar, B., Tripp, K., 2008. Restauración ecológica de coníferas longevas: el caso del alerce (*Fitzroya cupressoides*) en el sur de Chile. Restauración de bosques en América Latina 39–56.
León-Lobos, P., Bustamante-Sánchez, M.A., Nelson, C.R., Alarcón, D., Hasbún, R., Way, M., 2020. Lack of adequate seed supply is a major bottleneck for effective ecosystem restoration in Chile: friendly amendment to Bannister et al. (2018). Restor Ecol. 28, 277–281.
Leverkus, A.B., Lindenmayer, D.B., Thorn, S., Gustafsson, L., 2018. Salvage logging in the world's forests: Interactions between natural disturbance and logging need recognition. Glob. Ecol. Biogeo. 27 (10), 1140–1154.
Libano, A.M., Felfili, J.M., 2006. Mudanças temporais na composição florística e na diversidade de um cerrado sensu stricto do Brasil Central em um período de 18 anos (1985–2003). Acta Botanica Brasilica 20 (4), 927–936.
Lipoma, M.L., Funes, G., Díaz, S., 2018. Fire effects on the soil seed bank and post-fire resilience of a semi-arid shrubland in central Argentina. Austral Ecol. 43 (1), 46–55.
Lippok, D., Beck, S.G., Renison, D., Gallegos, S.C., Saavedra, F.V., Hensen, I., Schleuning, M., 2013. Forest recovery of areas deforested by fire increases with elevation in the tropical Andes. Forest Ecol. Manag 295, 69–76.
Litton, C.M., Santelices, R., 2002. Early post-fire succession in a *Nothofagus glauca* forest in the Coastal Cordillera of south-central Chile. Int. J. Wildland Fire 11 (2), 115. https://doi.org/10.1071/WF02039.
Liu, Z., Wimberly, M.C., 2016. Direct and indirect effects of climate change on projected future fire regimes in the western United States. Sci. Total Environ. 542, 65–75.
Lizundia-Loiola, J., Pettinari, M.L., Chuvieco, E., 2020. Temporal anomalies in burned area trends: satellite estimations of the Amazonian 2019 fire crisis. Remote Sens. 12 (1), 151. https://doi.org/10.3390/rs12010151.
Maksic, J., Shimizu, M.H., de Oliveira, G.S., Venancio, I.M., Cardoso, M., Ferreira, F.A., 2019. Simulation of the Holocene climate over South America and impacts on the vegetation. The Holocene. 29 (2), 287–299.
Mandle, L., Bufford, J.L., Schmidt, I.B., Daehler, C.C., 2011. Woody exotic plant invasions and fire: reciprocal impacts and consequences for native ecosystems. Biol. Invasions. 13 (8), 1815–1827.
Manners, R., Varela-Ortega, C., 2017. Analysing Latin American and Caribbean forest vulnerability from socio-economic factors. J. Integr. Environ. Sci. 14 (1), 109–130.
Martinez, O., 1981. Flora y fitosociología de un relicto de *Pilgerodendron uvifera* (D. Don) Florin en el fundo San Pablo de Tregua (Valdivia-Chile). Bosque. 4, 3–11.
Martínez-Torres, H.L., Pérez-Salicrup, D.R., Castillo, A., Ramírez, M.I., 2018. Fire management in a natural protected area: what do key local actors say? Human Ecol. 46 (4), 515–528.
Mayle, F.E., Burn, M.J., Power, M., Urrego, D.H., 2009. Vegetation and fire at the Last Glacial Maximum in tropical South America. In: Vimeux, F., Sylvestre, F., Khodri, M. (Eds.), Past climate variability in South America and surrounding regions. Springer, Dordrecht, pp. 89–112.
Meddens, A.J.H., Kolden, C.A., Lutz, J.A., Smith, A.M.S., Cansler, C.A., Abatzoglou, J.T., Meigs, G.W., Downing, W.M., Krawchuk, M.A., 2018. Fire refugia: what are they, and why do they matter for global change? Bioscience. https://doi.org/10.1093/biosci/biy103.
Meira-Castro, A., Shakesby, R.A., Espinha Marques, J., Doerr, S.H., Meixedo, J.P., Teixeira, J., Chaminé, H.I., 2015. Effects of prescribed fire on surface soil in a *Pinus pinaster* plantation, northern Portugal. Environ. Earth Sci. 73 (6), 3011–3018.
Méndez-Toribio, M., Martínez-Garza, C., Ceccon, E., Guariguata, M.R., 2018. La restauración de ecosistemas terrestres en México. Estado actual, necesidades y oportunidades. Documentos Ocasionales 185. Bogor, Indonesia: CIFOR.
Merino, A., Ferreiro, A., Salgado, J., Fontúrbel, M.T., Barros, N., Fernández, C., Vega, J. A., 2014. Use of thermal analysis and solid-state ^{13}C CP-MAS NMR spectroscopy to diagnose organic matter quality in relation to burn severity in Atlantic soils. Geoderma 226-227, 376–386.
Miceli-Méndez, C.L., Ferguson, B.G., Ramírez-Marcial, N., 2008. Seed dispersal by cattle: Natural history and applications to Neotropical forest restoration and agroforestry. In: Myster, R.W. (Ed.), Post-Agricultural succession in the Neotropics. Springer, New York, NY, pp. 165–191.
Mistry, J., Bilbao, B.A., Berardi, A., 2016. Community owned solutions for fire management in tropical ecosystems: case studies from Indigenous communities of South America. Philos. Tr. Roy. Soc. B. 371 (1696), 20150174. https://doi.org/10.1098/rstb.2015.0174.
Monroe, M.C., Ballard, H.L., Oxarart, A., Sturtevant, V.E., Jakes, P.J., Evans, E.R., 2016. Agencies, educators, communities and wildfire: partnerships to enhance environmental education for youth. Environ. Educ. Res. 22 (8), 1098–1114.
Moran, D., Laycock, H., White, P.C.L., 2010. The role of cost-effectiveness analysis in conservation decision-making. Biol. Conserv. 143 (4), 826–827.
Moreira da Silva, A.P., Schweizer, D., Rodrigues Marques, H., Cordeiro Teixeira, A.M., Nascente dos Santos, T.V.M., Sambuichi, R.H.R., Badari, C.G., Gaudare, U., Brancalion, P.H.S., 2017. Can current native tree seedling production and infrastructure meet an increasing forest restoration demand in Brazil? Restor. Ecol. 25 (4), 509–515.
Muqaddas, B., Zhou, X., Lewis, T., Wild, C., Chen, C., 2015. Long-term frequent prescribed fire decreases surface soil carbon and nitrogen pools in a wet sclerophyll forest of Southeast Queensland. Australia. Sci. Total Environ. 536, 39–47.
Murcia, C., Guariguata, M.R., 2014. La restauración ecológica en Colombia: Tendencias, necesidades y oportunidades. Documentos Ocasionales 107. Bogor, Indonesia: CIFOR.
Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403 (6772), 853–858.
Nardoto, G.B., Bustamante, M.M.d.C., 2003. Effects of fire on soil nitrogen dynamics and microbial biomass in savannas of Central Brazil. Pesqui. Agropecu. Bras. 38 (8), 955–962.
Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S., 2011. Fire effects on temperate forest soil C and N storage. Ecol. Appl. 21 (4), 1189–1201.
Neary, D.G., Klopatek, C.C., DeBano, L.F., Ffolliott, P.F., 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecol. Manag. 122 (1-2), 51–71.
Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland fire in ecosystems: effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-vol. 4. Ogden, UT: US Department of Agriculture, Forest Service, Rocky Mountain Research Station.
Nepstad, D.C., Stickler, C.M., Filho, B., Merry, F., 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. Phil. Tr. Roy. Soc. B. 363 (1498), 1737–1746.
Nobre, C.A., Sampaio, G., Borma, L.S., Castilla-Rubio, J.C., Silva, J.S., Cardoso, M., 2016. Land-use and climate change risks in the Amazon and the need of a novel sustainable development paradigm. P. Natl. Acad. Sci. 113 (39), 10759–10768.
Nuñez, M.A., Chiuffo, M.C., Torres, A., Paul, T., Dimarco, R.D., Raal, P., Policelli, N., Moyano, J., García, R.A., van Wilgen, B.W., Pauchard, A., Richardson, D.M., 2017. Ecology and management of invasive Pinaceae around the world: progress and challenges. Biol. Invasions. 19 (11), 3099–3120.

Ockendon, N., Thomas, D.H.L., Cortina, J., Adams, W.M., Aykroyd, T., Barov, B., Boitani, L., Bonn, A., Branquinho, C., Brombacher, M., Burrell, C., Carver, S., Crick, H.Q.P., Duguy, B., Everett, S., Fokkens, B., Fuller, R.J., Gibbons, D.W., Gokhelashvili, R., Griffin, C.y., Halley, D.J., Hotham, P., Hughes, F.M.R., Karamanlidis, A.A., McOwen, C.J., Miles, L., Mitchell, R., Rands, M.R.W., Roberts, J., Sandom, C.J., Spencer, J.W., ten Broeke, E., Tew, E.R., Thomas, C.D., Timoshyna, A., Unsworth, R.K.F., Warrington, S., Sutherland, W.J., 2018. One hundred priority questions for landscape restoration in Europe. Biol. Conserv. 221, 198–208.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial Ecoregions of the World: A new map of life on Earth. A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. Bioscience 51 (11), 933. https://doi.org/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2.

Palomeque, X., Günter, S., Siddons, D., Hildebrandt, P., Stimm, B., Aguirre, N., Arias, R., Weber, M., 2017. Natural or assisted succession as approach of forest recovery on abandoned lands with different land use history in the Andes of Southern Ecuador. New Forest. 48 (5), 643–662.

Paolucci, L.N., Pereira, R.L., Rattis, L., Silvério, D.V., Marques, N.C.S., Macedo, M.N., Brando, P.M., 2019. Lowland tapirs facilitate seed dispersal in degraded Amazonian forests. Biotropica 51 (2), 245–252.

Paritsis, J., Landesmann, J., Kitzberger, T., Tiribelli, F., Sasal, Y., Quintero, C., Dimarco, R., Barrios-García, M., Iglesias, A., Diez, J., Sarasola, M., Nuñez, M., 2018. Pine plantations and invasion alter fuel structure and potential fire behavior in a Patagonian forest-steppe ecotone. Forests 9 (3), 117. https://doi.org/10.3390/f9030117.

Pauchard, A., García, R.A., Peña, E., González, C., Cavieres, L.A., Bustamante, R.O., 2008. Positive feedbacks between plant invasions and fire regimes: *Teline monspessulana* (L.) K. Koch (Fabaceae) in central Chile. Biol. Invasions. 10 (4), 547–553.

Paula, S., Labbé, D.L., 2019. Post-fire invasion in Torres del Paine Biosphere Reserve: the role of seed tolerance to heat. Int. J. Wildland Fire. 28 (2), 160. https://doi.org/10.1071/WF18124.

Peinetti, H.R., Bestelmeyer, B.T., Chirino, C.C., Kin, A.G., Frank Buss, M.E., 2019. Generalized and specific state-and-transition models to guide management and restoration of Caldenal Forests. Rangeland Ecol. Manag. 72 (2), 230–236.

Pellizzaro, K.F., Cordeiro, A.O.O., Alves, M., Motta, C.P., Rezende, G.M., Silva, R.R.P., Ribeiro, J.F., Sampaio, A.B., Vieira, D.L.M., Schmidt, I.B., 2017. Cerrado restoration by direct seeding: field establishment and initial growth of 75 trees, shrubs and grass species. Braz. J. Bot. 40 (3), 681–693.

Pérez-Salicrup, D.R., Garduño-Mendoza, E., Martínez-Torres, H. L. y Del Río Pesado, G. 2020. Plan Integral del Manejo del Fuego en la Reserva de la Biósfera Mariposa Monarca: acción e investigación participativa y adaptable. FMCN, CONANP, Alternare, A.C., IIES –UNAM. 71 pp.

Pettinari, M.L., Ottmar, R.D., Prichard, S.J., Andreu, A.G., Chuvieco, E., 2014. Development and mapping of fuel characteristics and associated fire potentials for South America. Int. J. Wildland Fire. 23 (5), 643. https://doi.org/10.1071/WF12137.

Pineda-López, M.D.R., Sánchez-Velásquez, L.R., Ventura, Y.P., Fernández, P.G., Binnqüist, C.L., Rojo-Alboreca, A., 2015. The role of women in the forest conservation in a Mexican National Park: pruning firs for the manufacture of Christmas wreaths. Human Ecol. 43 (3), 493–501.

Pingree, M.R.A., DeLuca, T.H., 2018. The influence of fire history on soil nutrients and vegetation cover in mixed-severity fire regime forests of the eastern Olympic Peninsula, Washington, USA. Forest Ecol. Manag. 422, 95–107.

Pivello, V.R., Oliveras, I., Miranda, H.S., Haridasan, M., Sato, M.N., Meirelles, S.T., 2010. Effect of fires on soil nutrient availability in an open savanna in Central Brazil. Plant Soil. 337 (1-2), 111–123.

Policelli, N., Horton, T.R., Hudon, A.T., Patterson, T., Bhatnagar, J.M., 2020. Back to roots: The role of ectomycorrhizal fungi in boreal and temperate forest restoration. Front. Forest. Glob. Change. 3, 97.

Power, M.J., Marlon, J., Ortiz, N., Bartlein, P.J., Harrison, S.P., Mayle, F.E., 2008. Changes in fire regimes since the Last Glacial Maximum: an assessment based on a global synthesis and analysis of charcoal data. Clim. Dynam. 30, 887–907.

Preciado-Benítez, O., Gómez y Gómez, B., Navarrete-Gutiérrez, D.A., Horváth, A., 2015. The use of commercial fruits as attraction agents may increase the seed dispersal by bats to degraded areas in Southern Mexico. Trop. Conserv. Sci. 8 (2), 301–317.

Promis, A., Olivares, S., Acuña, S., Cruz, G., 2019. Respuesta temprana de la regeneración de plantas leñosas después del incendio forestal denominado "Las Máquinas" en la Región del Maule. Chile. Gayana Bot. 76, 257–262.

Quintero-Gradilla, S.D., Martínez-Yrizar, A., García-Oliva, F., Cuevas-Guzmán, R., Jardel-Peláez, J.E., 2020. Post-fire recovery of ecosystem carbon pools in a tropical mixed pine-hardwood forest. Forest Syst. 29, 1–13.

Raffaele, E., Veblen, T.T., Blackhall, M., Tercero-Bucardo, N., 2011. Synergistic influences of introduced herbivores and fire on vegetation change in northern Patagonia. Argentina. J. Veg. Sci. 22, 59–71.

Ray, D., Nepstad, D., Moutinho, P., 2005. Micrometeorological and canopy controls of flammability in mature and disturbed forests in an east-central Amazon landscape. Ecol. Appl. 15, 1664–1678.

Ribeiro, M.B.N., Bruna, E.M., Mantovani, W., 2010. Influence of post-clearing treatment on the recovery of herbaceous plant communities in Amazonian secondary forests. Restor. Ecol. 18, 50–58.

Roberts, L., Stone, R., Sugden, A., 2009. The rise of restoration ecology. Science 555.

Rodríguez-Trejo, D.A., Fulé, P.Z., 2003. Fire ecology of Mexican pines and a fire management proposal. Int. J. Wildland Fire. 12 (1), 23. https://doi.org/10.1071/WF02040.

Rodriguez-Trejo, D.A., Myers, R.L., 2010. Using oak characteristics to guide fire regime restoration in Mexican pine-oak and oak forests. Ecol. Restor. 28 (3), 304–323.

Rodríguez-Trejo, D.A., Martínez-Hernández, P.A., Ortiz-Contla, H., Chavarría-Sánchez, M.R., Hernández-Santiago, F., 2011. The present status of fire ecology, traditional use of fire, and fire management in Mexico and Central America. Fire Ecol. 7 (1), 40–56.

Román-Cuesta, R.M., Salinas, N., Asbjornsen, H., Oliveras, I., Huaman, V., Gutiérrez, Y., Puelles, L., Kala, J., Yabar, D., Rojas, M., Astete, R., Jordán, D.Y., Silman, M., Mosandl, R., Weber, M., Stimm, B., Günter, S., Knoke, T., Malhi, Y., 2011. Implications of fires on carbon budgets in Andean cloud montane forest: The importance of peat soils and tree resprouting. Forest Ecol. Manag. 261 (11), 1987–1997.

Roscoe, R., Buurman, P., Velthorst, E.J., Pereira, J.A.A., 2000. Effects of fire on soil organic matter in a "cerrado sensu-stricto" from Southeast Brazil as revealed by changes in δ13C. Geoderma 95 (1-2), 141–160.

Rost, J., Jardel–Peláez, E.J., Bas, J.M., Pons, P., Loera, J., Vargas-Jaramillo, S., Santana, E., 2015. The role of frugivorous birds and bats in the colonization of cloud forest plant species in burned areas in western Mexico. Anim. Biodiv. Conserv. 38 (2), 175–182.

Saiz, G., Bird, M.I., Domingues, T., Schrodt, F., Schwarz, M., Feldpausch, T.R., Veenendaal, E., Djagbletey, G., Hien, F., Compaore, H., Diallo, A., Lloyd, J., 2012. Variation in soil carbon stocks and their determinants across a precipitation gradient in West Africa. Glob. Change Biol. 18 (5), 1670–1683.

Saiz, G., Wynn, J.G., Wurster, C.M., Goodrick, I., Nelson, P.N., Bird, M.I., 2015. Pyrogenic carbon from tropical savanna burning: production and stable isotope composition. Biogeosciences. 12 (6), 1849–1863.

Saiz, G., Goodrick, I., Wurster, C., Nelson, P.N., Wynn, J., Bird, M., 2018. Preferential production and transport of grass-derived pyrogenic carbon in NE-Australian savanna ecosystems. Front. Earth Sci. 5 https://doi.org/10.3389/feart.2017.00115.

Sansevero, J.B.B., Prieto, P.V., Sánchez-Tapia, A., Braga, J.M.A., Rodrigues, P.J.F.P., 2017. Past land-use and ecological resilience in a lowland Brazilian Atlantic Forest: implications for passive restoration. New Forest. 48 (5), 573–586.

Santana, M.R.A., Montagnini, F., Suárez, A., Palácios-Mendoza, C., Razo-Zárate, R., Mohedano-Caballero, L., 2011. Community perceptions of the degradation and restoration of forest ecosystems in Southeastern Hidalgo state, Mexico. In: Finney, C. (Ed.), Montagnini F. America. Nova Science Publishers, Restoring degraded landscapes with native species in Latin, pp. 159–172.

Santana, L.D., Ribeiro, J.H.C., van den Berg, E., Carvalho, F.A., 2020a. Impact on soil and tree community of a threatened subtropical phytophysiognomy after a forest fire. Folia Geobot. 55 (2), 81–93.

Santana, N.C., Júnior, O.A.d.C., Gomes, R.A.T., Fontes Guimarães, R., 2020b. Comparison of post-fire patterns in brazilian savanna and tropical forest from remote sensing time series. ISPRS Int. J. Geo-Inf. 9 (11), 659. https://doi.org/10.3390/ijgi9110659.

Santiago-García, R.J., Colón, S.M., Sollins, P., Van Bloem, S.J., 2008. The role of nurse trees in mitigating fire effects on tropical dry forest restoration: a case study. Ambio 37 (7), 604–608.

Santín, C., Doerr, S.H., 2016. Fire effects on soils: the human dimension. Philos. Trans. R. Soc. B. 371 (1696), 20150171. https://doi.org/10.1098/rstb.2015.0171.

Santoiemma, G., 2018. Recent methodologies for studying the soil organic matter. Applied Soil Ecol. 123, 546–550.

Schmidt, I.B., Ferreira, M.C., Sampaio, A.B., Walter, B.M.T., Vieira, D.L.M., Holl, K.D., 2019. Tailoring restoration interventions to the grassland-savanna-forest complex in central Brazil. Restor. Ecol. 27 (5), 942–948.

Schulz, J.J., Cayuela, L., Echeverria, C., Salas, J., Rey Benayas, J.M., 2010. Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008). Appl. Geogr. 30 (3), 436–447.

Scott, A.C., Bowman, D.M., Bond, W.J., Pyne, S.J., Alexander, M.E., 2013. Fire on earth: an introduction. Wiley & Sons.

Scotti, M.R., Corrêa, E.J.A., 2004. Growth and litter decomposition of woody species inoculated with rhizobia and arbuscular mycorrhizal fungi in Semiarid Brazil. Annals Forest Sci. 61 (1), 87–95.

SER, 2004. Society for Ecological Restoration (SER), Science and Policy Working Group. The SER International Primer on Ecological Restoration. www.ser.org/ Resources/ Primer/Eng., accessed on 11 April 2020.

de Silva, Ú.S.R.d., Matos, D.M.d.S., 2006. The invasion of *Pteridium aquilinum* and the impoverishment of the seed bank in fire prone areas of Brazilian Atlantic Forest. Biodivers Conserv. 15 (9), 3035–3043.

Silvério, D.V., Brando, P.M., Balch, J.K., Putz, F.E., Nepstad, D.C., Oliveira-Santos, C., Bustamante, M.M.C., 2013. Testing the Amazon savannization hypothesis: fire effects on invasion of a neotropical forest by native Cerrado and exotic pasture grasses. Phil. Trans. R. Soc. B. 368 (1619), 20120427. https://doi.org/10.1098/rstb.2012.0427.

Simi, E., Moreno, P.I., Villa-Martínez, R., Vilanova, I., de Pol-Holz, R., 2017. Climate change and resilience of deciduous *Nothofagus* forests in central-east Chilean Patagonia over the last 3200 years. J Quater Sci. 32, 845–856.

Simon, M.F., Pennington, T., 2012. Evidence for adaptation to fire regimes in the tropical savannas of the Brazilian Cerrado. Int. J. Plant Sci. 173 (6), 711–723.

Smith-Ramírez, C., González, M.E., Echeverría, C., Lara, A., 2015. Estado actual de la restauración ecológica en Chile, perspectivas y desafíos: Current state of ecological restoration in Chile: Perspectives and challenges. An. Inst. Patagonia 43 (1), 11–21.

Smith-Ramírez, C., Castillo, J., Armesto, J.J., 2019. Willingness of rural communities to reforest with native tree species in central Chile. Restor Ecol. 27 (6), 1401–1408.

Sorrensen, C., 2009. Potential hazards of land policy: Conservation, rural development and fire use in the Brazilian Amazon. Land Use Pol. 26 (3), 782–791.
Souto, C.P., Heinemann, K., Kitzberger, T., Newton, A.C., Premoli, A.C., 2012. Genetic diversity and structure in *Austrocedrus chilensis* populations: Implications for dryland forest restoration. Restor. Ecol. 20 (5), 568–575.
Speziale, K.L., Lambertucci, S.A., Ezcurra, C., Novak, S., 2014. *Bromus tectorum* invasion in South America: Patagonia under threat? Weed Res. 54 (1), 70–77.
Taylor, K.T., Maxwell, B.D., McWethy, D.B., Pauchard, A., Nuñez, M.A., Whitlock, C., 2017. *Pinus contorta* invasions increase wildfire fuel loads and may create a positive feedback with fire. Ecology 98 (3), 678–687.
Tello, F., González, M.E., Valdivia, N., Torres, F., Lara, A., García-López, A., 2020. Diversity loss and changes in saproxylic beetle assemblages following a high-severity fire in *Araucaria-Nothofagus* forests. J. Insect Conserv. 24 (3), 585–601.
Tercero-bucardo, NORLAN, Kitzberger, THOMAS, Veblen, T.T., Raffaele, ESTELA, 2007. A field experiment on climatic and herbivore impacts on post-fire tree regeneration in north-western Patagonia. J Ecol. 95 (4), 771–779.
Toledo, F.H., McIntosh, T., Knothe, C., Aubrey, D.P., 2020. *Eucalyptus* are unlikely to escape plantations and invade surrounding forests managed with prescribed fire in southeastern US. Forests 11 (6), 694. https://doi.org/10.3390/f11060694.
Méndez-Toribio, M., Martínez-Garza, C., Cecconc, E., Guariguata, M.R., 2017. Planes actuales de restauración ecológica en Latinoamérica: Avances y omisiones. Rev. Cienc. Amb. 51 (2), 1. https://doi.org/10.15359/rca.51-2.1.
Torres Vargas, D.L., Quiroz Guerra, R., Juscamaita Morales, J., 2004. Efecto de una quema controlada sobre la población microbiana en suelos con pasturas en la SAIS Tupac Amaru-Junín. Perú. Ecol. Apl. 3 (1-2), 139. https://doi.org/10.21704/rea.v3i1-2.283.
Torres, R.C., Renison, D., 2017. Human-induced vegetation changes did not affect tree progeny performance in a seasonally dry forest of central Argentina. J. Arid Environ. 147, 125–132.
Úbeda, X., Sarricolea, P., 2016. Wildfires in Chile: A review. Glob. Planet. Change 146, 152–161.
UNEP-WCMC, 2016. The State of Biodiversity in Latin America and the Caribbean. Cambridge, UK.
United Nations, 2019. https://www.unenvironment.org/news-and-stories/press-release/new-un-decade-ecosystem-restoration-offers-unparalleled-opportunity. (Accessed on 11 April 2020).
Urretavizcaya, M.F., 2010. Propiedades del suelo en bosques quemados de *Austrocedrus chilensis* en Patagonia. Argentina. Bosque. 31, 140–149.
Urretavizcaya, M.F., Defossé, G.E., 2013. Effects of nurse shrubs and tree shelters on the survival and growth of two *Austrocedrus chilensis* seedling types in a forest restoration trial in semiarid Patagonia. Argentina. Ann. Forest Sci. 70 (1), 21–30.
Urretavizcaya, M.F., Defossé, G.E., 2019. Restoration of burned and post-fire logged *Austrocedrus chilensis* stands in Patagonia: effects of competition and environmental conditions on seedling survival and growth. Int. J. Wildland Fire. 28 (5), 365. https://doi.org/10.1071/WF18154.
Urretavizcaya, M.F., Gonda, H.E., Defossé, G.E., 2017. Effects of post-fire plant cover in the performance of two cordilleran cypress (*Austrocedrus chilensis*) seedling stocktypes planted in burned forests of northeastern Patagonia. Argentina. Environ Manag. 59 (3), 419–430.
Urretavizcaya, M.F., Peri, P.L., Monelos, L., Arriola, H., Oyharçabal, M.F., Contardi, L., Muñoz, M., Sepúlveda, E., Defosse, G., 2018. Condiciones de suelo y vegetación en tres bosques quemados de *Nothofagus pumilio* en Argentina y experiencias para su restauración activa. Ecol. Austral. 28 (2), 383–399.
Urrutia-Estrada, J., Fuentes-Ramirez, A., Hauenstein, E., 2018. Differences in floristic composition of *Araucaria-Nothofagus* forests affected by mixed levels of fire severity. Gayana Bot. 75, 625–638.
USDA Forest Service, 2012. Forest Service Manual 2523 – Emergency Stabilization – Burned Area Emergency Response (BAER). USDA Forest Service, Washington DC.
Valenzuela, P., Arellano, E.C., Burger, J.A., Becerra, P., 2016. Using facilitation microsites as a restoration tool for conversion of degraded grasslands to *Nothofagus* forests in Southern Patagonia. Ecol. Eng. 95, 580–587.
Valenzuela, P., Arellano, E.C., Burger, J., Oliet, J.A., Perez, M.F., 2018. Soil conditions and sheltering techniques improve active restoration of degraded *Nothofagus pumilio* forest in Southern Patagonia. Forest Ecol. Manag. 424, 28–38.
Van de Wouw, P., Echeverría, C., Rey-Benayas, J.M., Holmgren, M., 2011. Persistent *Acacia* savannas replace Mediterranean sclerophyllous forests in South America. Forest Ecol. Manag. 262 (6), 1100–1108.
van der Heijden, M.G.A., Klironomos, J.N., Ursic, M., Moutoglis, P., Streitwolf-Engel, R., Boller, T., Wiemken, A., Sanders, I.R., 1998. Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. Nature 396 (6706), 69–72.
van der Werf, G.R., Randerson, J.T., Giglio, L., van Leeuwen, T.T., Chen, Y., Rogers, B.M., Mu, M., van Marle, M.J.E., Morton, D.C., Collatz, G.J., Yokelson, R.J., Kasibhatla, P. S., 2017. Global fire emissions estimates during 1997–2016. Earth Syst. Sci. Data. 9 (2), 697–720.
van Galen, L.G., Lord, J.M., Orlovich, D.A., Larcombe, M.J., 2021. Restoration of southern hemisphere beech (*Nothofagaceae*) forests: a meta-analysis. Restor Ecol 29 (3). https://doi.org/10.1111/rec.v29.310.1111/rec.13333.
Varela, S.A., Gobbi, M.E., Laos, F., 2006. Seed bank of a burned *Nothofagus pumilio* forest: effect of biosolids compost application. Ecol. Austral. 16, 63–78.
Varela, S.A., Gobbi, M.E., Laos, F., 2011. Can biosolids compost improve, in the short term, native vegetation and soils fertility in burned *Nothofagus pumilio* forest in Patagonia, Argentina? Bosque. 32 (3), 267–278.
Veblen, T.T., Kitzberger, T., Raffaele, E., Mermoz, M., González, M.E., Sibold, J.S., Holz, A., 2008. The historical range of variability of fires in the Andean-Patagonian *Nothofagus* forest region. Int. J. Wildland Fire. 17 (6), 724. https://doi.org/10.1071/WF07152.
Veenendaal, E.M., Torello-Raventos, M., Feldpausch, T.R., Domingues, T.F., Gerard, F., Schrodt, F., 2015. Structural, physiognomic and above-ground biomass variation in savanna–forest transition zones on three continents–how different are co-occurring savanna and forest formations? Biogeosciences. 12, 2927–2951.
Vega, J.A., Fontúrbel, T., Merino, A., Fernández, C., Ferreiro, A., Jiménez, E., 2013a. Testing the ability of visual indicators of soil burn severity to reflect changes in soil chemical and microbial properties in pine forests and shrubland. Plant Soil. 369 (1-2), 73–91.
Vega, J.A., Fontúrbel, T., Fernández, C., Arellano Díaz, A., Díaz Raviña, M., Carballas, T., 2013b. Acciones urgentes contra la erosión en áreas forestales quemadas. Universidad de Santiago de Compostela, Guía para su planificación en Galicia.
Vidal, O.J., Aguayo, M., Niculcar, R., Bahamonde, N., Radic, S., San Martín, C., Kusch, A., Latorre, J., Félez, J., 2015. Plantas invasoras en el Parque Nacional Torres del Paine (Magallanes, Chile): estado del arte, distribución post-fuego e implicancias en restauración ecológica. An. Inst. Patagonia. 43 (1), 75–96.
Villa, P.M., Martins, S.V., Delgado Monsanto, L., de Oliveira Neto, S.N., Mota Cancio, N., 2015. La agroforestería como estrategia para la recuperación y conservación de reservas de carbono en bosques de la Amazonía. Bosque 36 (3), 347–356.
Villarreal, M.L., Haire, S.L., Iniguez, J.M., Montaño, C.C., Poitras, T.B., 2019. Distant neighbors: recent wildfire patterns of the Madrean Sky Islands of southwestern United States and northwestern Mexico. Fire Ecol. 15, 1–20.
Weaver, P.L., Schwagerl, J.J., 2008. Secondary forest succession and tree planting at the Laguna Cartagena and Cabo Rojo Wildlife Refuges in southwestern Puerto Rico. Ambio 37 (7), 598–603.
Weidlich, E.W.A., Flórido, F.G., Sorrini, T.B., Brancalion, P.H.S., Peralta, G., 2020. Controlling invasive plant species in ecological restoration: A global review. J. Appl. Ecol. 57 (9), 1806–1817.
Wortley, L., Hero, J.-M., Howes, M., 2013. Evaluating ecological restoration success: a review of the literature. Restor. Ecol. 21 (5), 537–543.
Zarin, D.J., Davidson, E.A., Brondizio, E., Vieira, I.CG., Sá, T., Feldpausch, T., Schuur, E. AG., Mesquita, R., Moran, E., Delamonica, P., Ducey, M.J., Hurtt, G.C., Salimon, C., Denich, M., 2005. Legacy of fire slows carbon accumulation in Amazonian forest regrowth. Front. Ecol. Environ. 3 (7), 365–369.
Zouhar, K., Smith, J.K., Sutherland, S., Brooks, M.L., 2008. Wildland fire in ecosystems. Fire and nonnative invasive plants. Gen. Tech. Rep. RMRS-GTR-42-vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
Zúñiga-Vásquez, J.M., Pompa-García, M., 2019. The occurrence of forest fires in Mexico presents an altitudinal tendency: a geospatial analysis. Nat. Hazards. 96 (1), 213–224.
Zúñiga, A.H., Rau, J.R., Fuenzalida, V., Fuentes-Ramírez, A., 2020. Temporal changes in the diet of two sympatric carnivorous mammals in a protected area of south-central Chile affected by a mixed-severity forest fire. Anim. Biodiv. Conserv. 43, 177–186.
Zúñiga, A.H., Rau, J.R., Jaksic, F.M., Vergara, P.M., Encina-Montoya, F., Fuentes-Ramírez, A., 2021. Rodent assemblage composition as indicator of fire severity in a protected area of south-central Chile. Austral Ecol. 46 (2), 249–260.