Global assessment and mapping of ecological vulnerability to wildfires

Fátima Arrogante-Funes [1], Inmaculada Aguado [1] and Emilio Chuvieco [1]

[1] Environmental Remote Sensing Research Group, Department of Geography and Geology, Universidad de Alcalá, Colegios 2, 28801 Alcalá de Henares, Spain.

Correspondence to: Fátima Arrogante-Funes (fatima.arrogante@uah.es)

Abstract. Fire is a natural phenomenon that has played a critical role in transforming the environment and maintaining biodiversity at a global scale. However, the plants in some habitats have not developed strategies for recovery from fire or have not adapted to the changes taking place in their fire regimes. Maps showing ecological vulnerability to fires could contribute to environmental management policies in the face of global change scenarios. The main objective of this study is to assess and map ecological vulnerability to fires on a global scale. To this end, we created ecological value and post-fire regeneration delay indices on the basis of existing global databases. Two ecological value indices were identified: biological distinction and conservation status. For the post-fire regeneration delay index, various factors were taken into account, including the type of fire regime, the increase in the frequency and intensity of forest fires and the potential soil erosion they can cause. These indices were combined by means of a qualitative cross-tabulation to create a new index evaluating ecological vulnerability to fire. The results showed that global ecological value could be reduced by as much as 50% due to fire perturbation of poorly adapted ecosystems. The terrestrial biomes most affected are the tropical and subtropical moist broadleaf forest; tundra; mangroves; tropical and subtropical coniferous forests; and tropical and subtropical dry broadleaf forests.

1 Introduction

Fire is a natural phenomenon that has played an important role in the transformation of the environment and the maintenance of biodiversity on a global scale. It can have numerous positive and negative impacts. Most of the world's terrestrial habitats where fires occur depend on them for ecological sustainability. (Kirkman et al., 2001; Midgley & Bond, 2015). Fire can affect the distribution of habitats, carbon and nutrient fluxes, and the water-holding properties of soils (Bowman et al., 2009). In habitats that are adapted to and even dependent on fire exclusion policies, this can result in a decrease in biodiversity (Guyette et al., 2002). In addition, the absence of fire results in increases in fuel loads (Bond et al., 2005), which frequently augment the risk of catastrophic fires over time. Fire has also been and continues to be used by humans as a crucial tool for managing terrestrial ecosystems, producing cultural landscapes that also benefit ecological health (Caprio & Graber, 2000; Guyette et al., 2002).

On the other hand, there are some habitats, such as moist tropical forests, that have never adapted to fires. The introduction of fire by humans can lead to an irreparable loss of their structure and composition (Cochrane & Laurance, 2002). Even in fire-adapted areas such as the Mediterranean ecosystems, recent human and climate-related changes in fire regimes are having negative impacts on the functioning of ecosystems (Bajocco et al., 2011; Midgley & Bond, 2015). The increasing frequency and intensity of fires can have negative impacts on forest stands

37 and landscapes, human life, infrastructures and ecosystem services and wildlife; and can cause changes in
38 regeneration dynamics, hydrological regimes and air quality, among other environmental consequences on a global
39 scale (Alcasena et al., 2016; Barrio et al., 2011; Buhk et al., 2007; Díaz-Delgado et al., 2002; Flannigan et al.,
40 2009; Hobson & Schieck, 1999; Moreira et al., 2011; Scott & Van Wyk, 1990). As a result of this process of
41 change, wildfires have become one of the main environmental problems today at both global and local levels.

42 This means that fires must be included in global and regional assessments of vulnerability to global change
43 (Houghton et al., 2001; Lindner et al., 2010). Furthermore, fire risk assessment should be carried out spatially in
44 order to design and implement prevention strategies that enable the conservation of the ecological value of
45 ecosystems and landscapes. When fires happen, assessments of this kind can also be useful for implementing post-
46 fire strategies to bring about the recovery of pre-fire ecological values and cultural and socioeconomic assets
47 (Aretano et al., 2015; Chuvieco et al., 2010). In terms of natural hazards terminology, spatially measured fire risk
48 is a combination of 'danger' and 'vulnerability'. 'Danger' is defined as the probability of fire occurring in a given
49 place and time, while vulnerability refers to the potential damage that fire could cause to this place (Chuvieco et
50 al., 2007).

51 The concept of vulnerability has been studied and applied at different spatial scales and in a wide range of
52 disciplines, in both social and natural studies (Abson et al., 2012; Berry et al., 2006; Cinner et al., 2012; Cutter et
53 al., 2003; Moreno & Becken, 2009).

54 Vulnerability has many different definitions. For example, the definition proposed by the UNISDIR, (2009)is
55 based on the assumption that an ecosystem cannot cope with a disturbing event (earthquake, fire, flood, etc.) and
56 is therefore vulnerable to it. In order to assess where adaptation actions may be necessary and beneficial,
57 vulnerability assessment must analyse the factors that determine the potential for damage from exogenous threats,
58 as well as the endogenous adaptive capacity of the ecosystem (Preston et al., 2011).

59 An ideal assessment of ecological vulnerability must therefore take into account the biotic and abiotic aspects of
60 the environment (e.g. species richness, conservation status of the ecosystems), the relationship between them (e.g.
61 ecosystem functionality) (Ippolito et al., 2010), as well as any temporal and spatial pressures (e.g. landscape
62 fragmentation) (Williams & Kapustka, 2000). An integrated approach to vulnerability can therefore be achieved
63 by developing different indices that characterize the biodiversity and ecological quality of the environment and its
64 capacity to adapt and regenerate once a fire has been extinguished.

65 The integration and harmonization of spatial data of different origin and typology on a global scale in an index is
66 a challenge. Numerous integration techniques exist, such as multicriteria methods (El Gibari et al., 2019). But for
67 a global scale, the lack of a panel that is sufficiently representative of the world would lead to a biased result
68 (depending on the territory of which there was representation or not) (Borrero & Henao, 2017; Hämäläinen &
69 Alaja, 2008). For this reason, qualitative cross-tabulation seems like an integration tool that could be objective
70 enough when dealing with categorical data as proposed by numerous studies (Arrogante-Funes et al., 2021;
71 Martínez Vega et al., 2007).

72 Some attempts to assess vulnerability do not take all these elements into account (Turner et al., 2003). The study
73 by Duguy et al., (2012) characterized ecological vulnerability using the species richness measurement, at a local
74 scale, in Mediterranean forests. In research in southern Italy, also on a local scale, Aretano et al., (2015) proposed
75 an ecological sensitivity index covering unique habitats, susceptibility to fire and regeneration capacity, but did
76 not evaluate soil erosion after disturbance. At the regional level, Chuvieco et al., (2010) studied ecological

vulnerability in line with the degree of protection of the area, reviewing the different legal forms for the official protection of ecosystems, homogeneous landscape units and land uses. This approach focused more on landscape ecology than on species biodiversity, in which adaptation to fire is considered through the strategies developed by plants in response to fire through the dynamic global vegetation model called ORCHIDEE developed by (Krinner et al., (2005). In other research, such as the study by González, Kolehmainen, & Pukkala, (2007), the vulnerability of the ecosystem to fire was evaluated by a group of experts who were provided with images and data on forest metrics measured in the field, together with aerial photographs. Regional studies have been conducted to evaluate the effects of fire on soils and post-fire dynamics in ecosystems (Duguy & Vallejo, 2008; Giovannini & Lucchesi, 1997). The first global analysis of wildfire vulnerability was done by Chuvieco et al. (2014), who estimated the standing ecological value of ecosystems from biodiversity data, their state of conservation and the fragmentation of the landscape. The delay in the post-fire regeneration of vegetation was estimated by assessing their adaptation to fire and potential soil erosion. Adaptation to fire was analysed by comparing the real land cover with fire simulations based on the dynamic global vegetation model.

In this paper, we carry out a systematic assessment of ecological vulnerability to wildfires on a global scale using an index that combines the two main components of vulnerability, namely the ecological value of ecosystems and the delay in post-fire regeneration. The novelty of this approach lies in the characterization of structural biodiversity from the point of view of its exceptionality, while also assessing biodiversity in terms of ecosystem functionality. In addition, in this study, rather than approaching the post-fire regeneration of forests as part of a static, immutable system, as most previous researchers have done, we view these strategies within the dynamic context of changing fire regimes. This study will be carried out on a global scale so as to enable us to tackle the planetary ecosystem as a whole, unrestricted by governmental or geographic borders. In this way, this research could become an essential tool for decision-making about resource management and nature conservation across the globe.

2 Materials and methods

2.1 Conceptual framework

In order to develop the Ecological Vulnerability Index proposed in this study, our first task was to estimate the ecological value of the environment and its regeneration capacity after fire disturbance. To do so, we had to process the different input variables and devise a way to integrate them into the index (Table 1). In addition, the basic integration tool in the different indicators and index is the qualitative cross tabulation used in many spatial studies (Arrogante-Funes et al., 2021; Martínez Vega et al., 2007).

3

Table 1: Conceptual framework and diagram for the Ecological Vulnerability Index, and reference sources used in the input variables.

Factor	Input Variables	Source	Method
Taxonomic Rarity	Vertebrate and Vascular Plant Endemisms	Kier et al., (2009); World Wildlife Fund, (2006)	Endemism Ratio to Total Species
Species Richness	Number of Vertebrates and Vascular Plants	Kier et al., (2005); World Wildlife Fund, (2006)	Species Total Normalized by Area
Functional Diversity	Specific Leaf Area, Leaf Dry Matter Content, Leaf Nitrogen Content, Leaf Phosphorus Content	Moreno-Martínez et al., (2018)	Carbon, Nitrogen and Phosphorous Cycle Productivity
Unique Habitats	Global 200 Map	Olson & Dinerstein, (2002)	Percentage Unique Habitats in relation to the Total by Ecoregion
Unique Preservation Habitats	35 Priority Places Map, Red List of Threatened Species	Burgess et al., (2014); World Wildlife Fund, (2006)	Percentage of Protected Area, Number of Threatened Species
Intact Forest Landscapes Blocks	Intact Forest Landscapes Maps	Potapov et al., (2008)	Percentage of Intact Forest in relation to the Total Area by Ecoregion
Degree of Fragmentation	Fragmentation by Ecoregion	Hoekstra et al., (2010)	Homogeneity Percentage by Ecoregion
Degree of Protection	World Database on Protected Areas	IUCN & UNEP-WCMC, (2020)	Percentage of Protected Area in relation to the Total Area by Ecoregion
RUSLE	RUSLE Map	Borrelli et al., (2017)	Qualitative Ranges
Relationship fire-ecoregion	Map	Shlisky et al., (2007)	Qualitative Ranges
Degradation Natural condition of fire	Fire Condition Natural Degradation Map		

Hierarchy (left side of diagram):
Ecological Vulnerability Index → Ecological Indicator → {Biological distinctiveness (Taxonomic Rarity, Species Richness, Functional Diversity, Unique Habitats); Conservation Status (Unique Preservation Habitats, Intact Forest Landscapes Blocks, Degree of Fragmentation, Degree of Protection)}; Post-fire Regeneration Delay Indicator → {Potential Soil Erosion (RUSLE); Adaptation to Fire (Relationship fire-ecoregion, Degradation Natural condition of fire)}.

2.2 Spatial Unit

The spatial units used in this study were the terrestrial ecoregions proposed by the World Wildlife Fund (WWF), as corrected in 2017 (Olson et al., 2001). The terrestrial ecoregion concept refers to a land unit large enough to house a set of natural communities composed of different species, dynamics and similar environmental conditions. Thus, ecoregions are a good way to structure ecological and fire information on a global scale, since they are relatively homogeneous in terms of climate and vegetation (Pausas & Ribeiro, 2017). For this reason, ecoregions are considered a more suitable unit of reference on which to add spatial biological information, compared to other possible units such as grids.

The database is made up of 827 ecoregions distributed in 14 biomes. The ecoregions in which it is impossible for forest fires to occur were excluded, while other areas, such as Antarctica, were excluded due to lack of data. In this way, the final number of ecoregions was 660, having representation of all terrestrial biomes.

2.3 Burnable Area

It was necessary to define the burnable area in order to identify areas in which fires are unable to expand. Our assessment of Burnable Area was based on the global Land Cover (LC) dataset produced under the Climate Change Initiative (CCI) program of the European Space Agency (ESA) (www.esa-landcover-cci.org). The CCI-LC map was generated from MERIS-Envisat images acquired at 300 m between 2008 and 2012. The original product

4

133 includes 22 land covers, which were reclassified to burnable/unburnable covers and then resampled at a resolution
134 of 0.25 degrees.

135 Ecoregions with burnable areas of ≤ 33% were removed from further analysis, as they would suffer only marginal
136 impacts of fire. This reduced the final number of ecoregions and terrestrial biomes used in our analysis to 647 and
137 14, respectively (Fig. A1).

2.4 Representativeness Criteria

139 The approach used to establish the ecological value of the different terrestrial ecoregions is based on the concept
140 of representativeness. In this way, each biome is guaranteed to have at least one priority ecoregion, so ensuring,
141 for example, that the ecoregions in the savanna forest biome can also be classified, in addition to the more
142 important moist tropical forests, which would otherwise dominate the list of values due to their high rates of species
143 richness and endemic species (endemisms). This approach is used in ecoregional evaluations that enable
144 comparison between studies (Burgess et al., 2006; Ricketts et al., 1999). The biological values were studied by
145 ecoregion within the biome to which they belong. Then, all the ecoregions with their respective biological values
146 were combined in a map at global level.

2.5 Ecological Indicator

148 To evaluate the ecological component relative to the ecoregions within each biome, two indicators were
149 qualitatively generated and integrated by cross-tabulation: i) Biological distinctiveness and ii) Conservation Status.
150 This approach enables us to characterize structural biodiversity from the point of view of its exceptionality, and in
151 terms of ecosystem functionality (Dinerstein et al., 1995; Ricketts et al., 1999).

2.5.1 Biological distinctiveness

153 Biological distinctiveness is more than just biodiversity at the species level, in that it also covers the diversity of
154 ecological functions and the processes that support structural biodiversity (Ricketts et al., 1999). Specifically, this
155 study is based on taxonomic rarity, species richness, functional diversity, and habitats with a unique evolution.
156 **Taxonomic Rarity and Species Richness.** The lists of species and endemisms (i.e. at least 75% of the taxon
157 occurs in the same place) by ecoregion for mammals, birds, reptiles and amphibians form a dataset that can be
158 gleaned from the literature, distribution databases, and fieldwork carried out by expert taxonomists (WWF, 2006).
159 Likewise, the data relating to diversity and vascular plant endemisms (Kier et al., 2005, 2009) have been used in
160 numerous ecological studies (Freudenberger et al., 2012; Poos, Walker, & Jackson, 2009).

161 To find out more about vertebrate species diversity, the total number was obtained by adding up all the vertebrate
162 species belonging to the same ecoregion. The data were then normalized according to land area (Eq. (1)):

$$SA = S/(A)^Z \qquad \textbf{(1)}$$

164 where SA is the number of species corrected by ecoregion, S the total number of species, A is the area in km^2 and
165 Z is the correction factor for continental mainland (value of 0.2) and islands (value 0.25) (Rosenzweig, 1995). As
166 numerous studies show (Burgess et al., 2006; Olson et al., 2001; Ricketts et al., 1999), the behaviour of this data
167 type is associated with the size of the territory, which is why in order to make them comparable we had to apply
168 this method of approximation to the species-area distribution curve. The same process was followed to obtain the

169 richness of vascular plant species, except that the data for the total number of species by ecoregion had already
170 been collected.

171 To assess the absolute taxonomic rarity for vertebrates and vascular plants, the endemism-richness ratio (Eq. (2))
172 was calculated. This estimates the number of species endemic to the ecoregion as a proportion of its species
173 richness:

174
$$R = (\Sigma E / \Sigma S) \quad \textbf{(2)}$$

175 where R is the percentage of endemisms, E the endemisms and S the species.

176 **Functional Diversity.** The continuous data about Specific Leaf Area (SLA), Leaf Dry Matter Content (LDMC),
177 Leaf Nitrogen Content (LNC) and Leaf Phosphorus Content (LPC) (g x g -1) was provided by Moreno-Martínez
178 et al., (2018) at 500m spatial resolution. It was used as a proxy of Carbon, Nitrogen and Phosphorus cycle
179 productivity.

180 To obtain the productivity of each cycle, an average figure by ecoregion was estimated. The productivity values
181 were then scaled in a monotonous linear manner increasing from 1 to 100, so as to enable us to compare
182 productivity between the different ecoregions. Finally, functional diversity was integrated as a sum of the
183 productivity values for the carbon, nitrogen and phosphorus cycle.

184 The environment is a holistic system, which means that loss of function affects the capacity of the ecosystem to
185 support not only itself, but also its neighbours (Pausas & Ribeiro, 2017). Ecoregions with high functional diversity
186 values are therefore considered more vulnerable to fires because they provide support for other ecosystems that
187 could also be damaged indirectly by fire in this way.

188 **Unique Habitats.** The Global 200 (G200) cartography (Olson & Dinerstein, 2002) shows the area in square
189 kilometres of habitats with unusual ecological and evolutionary phenomena by ecoregion, which make them
190 irreplaceable (Myers et al., 2010). In this way, 141 terrestrial ecoregions were identified. To assess the G200
191 cartography, we calculated the ratio between the area occupied by these habitats and the total area of the ecoregion.

192 **Integrating the Factors into the Biological Distinctiveness.** The above factors were integrated into the
193 Biological Distinctiveness using the criteria established by Burgess et al., (2006). First, the factors were scaled
194 between 1 and 100 through a linear function per biome. The taxonomic rarity scores were given the most weight
195 as they establish the qualitative ranges of the biodiversity through quartiles: Very High, High, Moderate and Low
196 (Table 2). In the case of endemic species, this is because if a fire occurred in one of these ecoregions, the entire
197 species would be wiped out. For the other factors, the first quartiles of species richness and of unique habitats and
198 scores of > 95 for functional diversity are taken into account when assigning these ecoregions to the exceptional
199 category (Table 2).

200
201
202
203
204
205
206
207
208 **Table 2: Summary of the criteria for assigning ecoregions within the biomes to the different categories.**

Categories	Endemisms	Species Richness	Functional Diversity	Unique Habitats
Very High	First quartile of total endemisms within the biome	First quartile of species richness within the biome	Ecoregions with more than 95% productivity	First quartile of unique habitats
High	Second quartile of total endemisms within the biome			
Moderate	Third quartile of total endemisms within the biome			
Low	Fourth quartile of total endemisms within the biome			

209

2.5.2 Conservation Status

The Conservation Status seeks to estimate the current and future capacity of an ecoregion to meet the following biodiversity conservation and quality objectives: maintain populations and communities of viable species, maintain ecological processes, and respond effectively to environmental changes over time. Specifically, this study is based on the preservation of unique habitats, the presence of landscapes that contain intact habitats, the degree of environmental fragmentation and the level of protection they enjoy.

Unique Habitats Preservation. The 35 Priority Places (35PP) cartography, proposed by the WWF, consolidates special conservation areas because they are an extensive and intact representation of unique ecosystems (Burgess et al., 2014). Of these, we maintained the 33 terrestrial ecoregions with a degree of protection and then estimated the ratio between the area occupied by these protected ecosystems and the total area of the ecoregion to which they belonged.

For its part, the Red List of Threatened Species (RL) provides data about the current situation of the biodiversity (WWF, 2006). We maintained the species on this list categorized as: "critically endangered", "endangered" and "vulnerable". These categories were selected because there are common criteria for the management and conservation of the habitats that host these species (Hilton-Taylor, 2000; Mace & Lande, 1991). We then calculated the total number of threatened species by ecoregion.

Both processed variables were scaled from 1 to 100 in an increasing monotonic linear manner and were added together to obtain the singular habitats preservation factor.

Intact Forest Landscapes Blocks. From an ecological point of view, old-growth forests are of great importance, albeit more structural than functional, in terms of their role in the conservation of most of terrestrial diversity, hosting indigenous populations and contributing enormously to the regulation of the global climate. Outside these blocks, for example in planted forests, characteristics such as the age of the plants or the composition of the stands could not be maintained in such an exceptional way. The Intact Forest Landscapes (IFL) cartography (Potapov et al., 2008) charts the location and extent of the forests and terrestrial ecosystems that remain unaltered by humans, with a minimum mappable unit of 500 km^2. The IFL area data was added to the corresponding ecoregions and the area occupied by these forests as a percentage of the total area of the ecoregion was calculated.

Degree of Fragmentation. Landscape fragmentation mapping by ecoregion is based on the method proposed by Hoekstra et al., (2010). It shows the degree of fragmentation as a percentage, with the highest percentages

238 corresponding to highly degraded or heterogeneous landscapes and the lowest to areas that are unfragmented or
239 homogeneous.
240 The degree of homogeneity was established by scaling the values for terrestrial ecoregions in a monotonic linear
241 manner reversing the original scale from 1-100 to 100-1. The more homogeneous compositions have higher
242 biodiversity ratios (Collinge, 1996), so making them more vulnerable to fire due to the ecological loss that this
243 would cause (Pausas et al., 2003).
244 **Degree of Protection.** Protected status, mainly in the form of national parks and reserves, plays an essential role
245 in conservation. These areas are mapped in the World Database on Protected Areas (WDPA), which was generated
246 as part of a project developed by the United Nations Environment Program (UNEP) and by the IUCN, administered
247 by the World Center Conservation Monitoring Committee (WCMC) and UNEP (IUCN & UNEP-WCMC, 2020).
248 In this study, we only considered the terrestrial protected areas classified under IUCN categories I-IV, as for these
249 categories there is reliable data, verified on the ground, and they are managed in a similar way, thus enabling us
250 to assume that they all have the same biodiversity conservation values. The area data for the WDPAs were added
251 to the corresponding ecoregions and we then calculated the area occupied by WDPAs as a percentage of the total
252 area of each ecoregion.
253 **Integrating the factors into the Conservation Status.** The weights (Table 3) for the different factors (i.e. unique
254 habitats, intact forest landscapes, degree of fragmentation and degree of protection) and the method for integrating
255 them into the Conservation Status were as proposed by Burgess et al., (2006) and by Ricketts et al., (1999). These
256 variables were multiplied by their weight (Table 3) and then added together to obtain the Conservation Status. In
257 this way, the scores that can be obtained by an ecoregion vary between a minimum of 1 and a maximum of 100
258 (Table 3). The variables awarded the most weight are those that indicate the quality of an ecosystem in terms of
259 its size and homogeneity. Then, the values were scaled according to this criterion and qualitative ranges were
260 generated using quartiles such as (Pereira et al., (2020)and Xing & Ree (2017), among others (Table 4).

Table 3: Values assigned on the basis of conservation status obtained from the G200 cartography

Factors	Weights
Unique Habitats Preservation	40
Intact Forest Landscapes	25
Degree of Fragmentation	20
Degree of Protection	15

Table 4: Criteria for assigning ecoregions within biomes to the different categories

Categories	Conservation Status
Very High	First quartile
High	Second quartile
Moderate	Third quartile
Low	Fourth quartile

265 **2.5.3 Integrating the Ecological Indicator**

266 The Biological Distinctiveness and Conservation Status were constructed using a qualitative cross-tabulation that
267 prioritized the most valuable elements, given that high biodiversity and quality values also imply high ecological
268 values in the environment (Ricketts et al., 1999) (Table 5).
269

Table 5: Criteria for assigning ecoregions within biomes to the different categories in the Ecological Indicator.

		Conservation Status			
		Very High	High	Moderate	Low
Biological Distinctiveness	Very High	Very High	Very High	High	Moderate
	High	Very High	Very High	High	Moderate
	Moderate	High	High	Moderate	Moderate
	Low	High	Moderate	Low	Low

2.6 Post-Fire Vegetation Regeneration Delay Indicator

The delay in the regeneration of vegetation after a fire is an indicator of the difficulties faced by the environment when recovering naturally from fire. It depends on the various strategies adopted by forest species that have adapted to fire and also on the physical state of the soil after the fire. This study provides a dynamic approach which includes an assessment of the alteration of the fire regime. Habitats that have not adapted to the change in fire regimes observed in recent decades will also be assessed.

2.6.1 Adaptation of the Vegetation to Fire Regimes

We used the two maps provided by Shlisky et al., (2007), which were generated in collaboration with WWF, the Nature Conservancy (TNC), the University of Berkeley and the IUCN. Firstly, in this database, the ecoregions were grouped into relationship between fire and ecoregion characterized by fire behaviour, plant strategies in response to fire, climatic variables and human use of fire as management tool. Secondly, the ecoregions were grouped together on the basis of the alteration of the natural state of fire regimes, measured in terms of frequency, severity, size and seasonality. The first grouping includes fire-dependent, sensitive and independent ecoregions, while the second classifies ecoregions according to intact, altered and highly altered respect the first grouping.

After reviewing the data base, 647 terrestrial ecoregions were maintained (repeated and confusing information was eliminated, as were ecoregions without data, covered with ice or rock). To estimate the adaptation of the ecoregions to fire regimes, the two factors (regimes and their alteration) were integrated through a qualitative cross-tabulation (Table 6).

The lowest values for Adaptation to Fire Regimes were for the independent and sensitive categories, while the highest were for the ecoregions that were well adapted to fire. In ecosystems that are well adapted to fire, it plays a fundamental role in the conservation of biodiversity. However, in poorly adapted ecosystems, fire can cause serious problems in the recovery and conservation of biodiversity because the plants do not have the necessary strategies to cope with and recover from it (Shlisky et al., 2007).

297 **Table 6: Criteria for assigning ecoregions to the different categories of adaptation to fire regimes**

	Natural Condition Fire		
Relationship between fire and ecoregion	**Very Degraded**	**Degraded**	**Intact**
Independent	Low	Low	Moderate
Sensitive	Low	Moderate	High
Dependent	Moderate	High	Very High

298

299 **2.6.2 Soil Erosion Potential**

300 Post-fire soil erosion can reduce the recovery capacity of the vegetation, and consequently of the ecosystem. The
301 expansion capacity of the roots depends on the quality of the soil, in terms for example of its texture. This is why,
302 after a fire, regeneration of the vegetation does not begin instantaneously. The soil must first recover its original
303 structure and composition and this takes time. The Global Soil Erosion map (Borrelli et al., 2017) was developed
304 using the Revised Universal Soil Loss Equation (RUSLE) with a spatial resolution of 250 m.

305 Potential soil losses were calculated in tons per pixel. The potential soil erosion per ecoregion (tn / ha) was
306 estimated by adding together all the soil losses and then dividing by the total area. The values were then
307 transformed into a categorical variable according to the criterion for soil erosion due to water, proposed by the
308 Food and Agriculture Organization of the United Nations (FAO) (FAO/UNEP/UNESCO, 1979) (Table 7), which
309 is also applicable to fire erosion processes (Chuvieco et al., 2014).

310

311 **Table 7: FAO criteria for assigning ecoregions to different categories of potential soil erosion.**

Categories	Values (tn/ha year)
Low	0 – 20
Moderate	20 – 50
High	50 – 200
Very High	> 200

312

313 **2.6.3 Integrating the Post-Fire Vegetation Regeneration Delay Indicator**

314 The two factors - Adaptation of Vegetation to Fire and Potential Soil Erosion - were combined by qualitative cross-
315 tabulation (prioritizing the most valuable element) to obtain the Post-Fire Regeneration Delay Indicator (Table 8).
316 This is an indicator of the time required for an ecosystem to regenerate naturally, i.e. for it to recover a structure
317 and composition similar to that that existed pre-fire. Therefore, the higher the delay values, the greater the
318 vulnerability to fire. This factor is the opposite of the post-fire regeneration capacity index calculated by other

319 authors in local studies (Baeza et al., 2007). Post-Fire Regeneration Delay values from High to Very High were
320 assigned to ecoregions with a Moderate or Low Adaptation to Fire and High Potential Soil Erosion values. The
321 lowest Regeneration Delay values corresponded to ecoregions that were well adapted to fire and had low soil
322 erosion potential.
323
324 Table 8: Criteria for assigning ecoregions to the different Post-Fire Vegetation Regeneration Delay categories.

		Potential Soil Erosion			
		Low	Moderate	High	Very High
Adaptation of Vegetation to Fire	Very High	Low	Low	Moderate	High
	High	Low	Low	Moderate	High
	Moderate	Moderate	Moderate	High	Very High
	Low	Moderate	High	Very High	Very High

325

2.7 Combining the Ecological Indicator and the Post-Fire Vegetation Regeneration Delay Indicator to form the Ecological Vulnerability to Fire Index

328 Once the different components of our Ecological Vulnerability to Fire Index had been obtained, they were
329 combined by means of a qualitative cross-tabulation in which the most valuable component was prioritized (Table
330 9). In other words, the potential ecological losses due to fires were estimated. The lower the Post-Fire Regeneration
331 Delay values, the lower the impacts of fire.
332

Table 9: Criteria for assigning ecoregions to the different Ecological Vulnerability Index categories.

		Post-Fire Vegetation Regeneration Delay Indicator			
		Low	Moderate	High	Very High
Ecological Indicator	Low	Low	Low	Moderate	High
	Moderate	Low	Moderate	Moderate	High
	High	Moderate	High	Very High	Very High
	Very High	High	High	Very High	Very High

334

335 2.8 Sensitivity Analyses: One at a time

The objective of a sensitivity analysis is to test the uncertainty of the result of a mathematical model due to the integration of numerical variables. The one-at-a-time (OAT) method is the most widely used in the literature and consists of analysing the effect of making small variations on one input while others remain fixed (Saltelli et al., 2000).

In this study, the variables that make up the Ecological Fire Vulnerability Index are of a categorical type and it is for this reason that a modification of the OAT method is proposed in order to be able to estimate the uncertainty of the product such as Gonzalez et al., (2015) and Clavijo et al., (2019) estimated in theirs studies. In the way of integrating said index through the Ecological and Post-fire Regeneration Delay indicators, the resulting label of ecological vulnerability obtained through the qualitative cross tabulation has been varied (Table 10). In this way we will be able to establish stable ecoregions (reference) and changing ecoregions (uncertainty).

Table 10: Criteria for assigning ecoregions to the different Ecological Vulnerability Index categories in order to test the OAT.

		Post-Fire Vegetation Regeneration Delay Indicator			
		Low	Moderate	High	Very High
Ecological Indicator	Low	Low	Moderate	Moderate	High
	Moderate	Moderate	Moderate	High	High
	High	Moderate	High	High	Very High
	Very High	High	High	Very High	Very High

The changes made correspond to: (i) the same category of label corresponds to the same resulting label, (ii) if two continuous categories face each other, the resulting label will be the one with the highest category and (iii) between two different categories the label of resulting vulnerability will be an intermediate category, prioritizing the highest when there are several in between.

3 Results

3.1 Ecological Indicator

Figure 1 shows the Ecological Value by ecoregion in terms of Biological Distinctiveness (Fig. A2) and Conservation Status (Fig. A3) indices. Ecoregions of increasing ecological value are shown in a range of tones from light green to dark green.
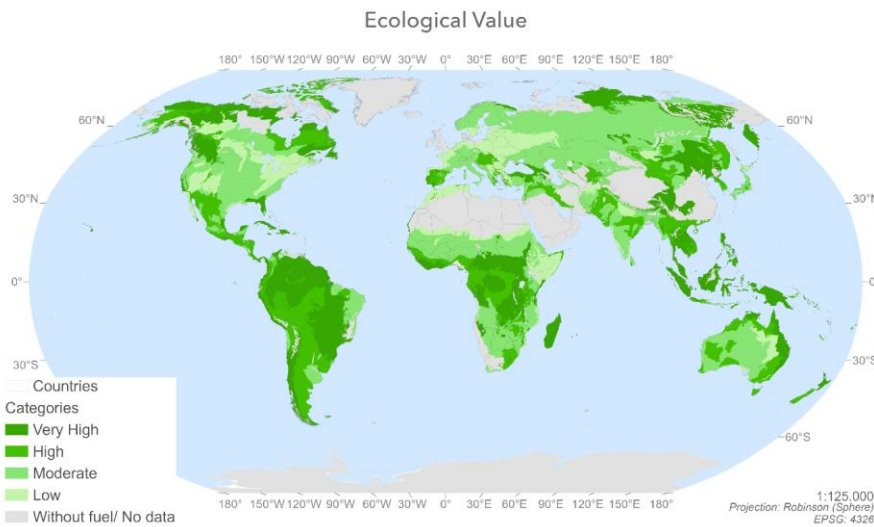
Ecological Value

Figure 1: Spatial distribution of Ecological Value by ecoregion (Ecological Indicator) calculated by combining the Biological Distinctiveness (by ecoregion evaluated within the biome to which it belongs) and the Conservation Status (by ecoregion).

There are 220 ecoregions with Very High Ecological values, 163 with High values, 206 with Moderate values and 59 with Low values. The ecoregions with the highest Ecological Values (Fig. 1) are located in temperate zones, such as British Columbia, forests in the US and European Mediterranean, China, New Zealand; and in the tropical and subtropical regions, for example the Amazon Basin, Sierra Leone, Cameroon, the Congo Basin, Zambia, Madagascar, New Guinea and northern Australia. Boreal areas, such as Canada and Russia, also show high ecological values.

3.2 Post-Fire Regeneration Delay Indicator

Figure 2 shows the Post-Fire Regeneration Delay value by ecoregion, in terms of Adaptation of Vegetation to Fire (produced by combining the plant strategies and fire-regime alteration factors) (Fig. A4) and Potential Soil Erosion (Fig. A5). The very high and high Delay values, highlighted in dark purple tones, are for areas with high Erosion and low Adaptation to Fire, while the moderate and low values, highlighted in lighter lilac tones, are associated with vegetation with very high and high Adaptation to Fire values and moderate or low Erosion values.

13
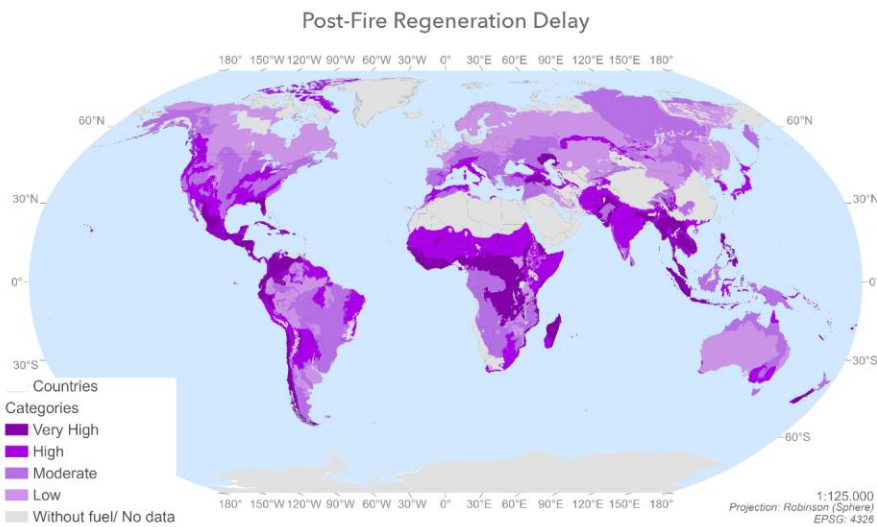
Figure 2: Spatial distribution of Post-fire Regeneration Delay Values by ecoregion calculated by combining the Adaptation to Fire and the Potential Soil Erosion values by ecoregion.

Of the 647 ecoregions evaluated, 154 had very high Post-fire Regeneration Delay values, 271 had high values, 157 had moderate values and 120 had low values. The least resilient zones (with low or moderate Adaptation to Fire and high or very high Potential Soil Erosion) belonged to temperate regions such as Florida, the Yucatan Peninsula, eastern United States, the forests of California, Chile and the Spanish Mediterranean and forests in the Caucasus, Himalayas and New Zealand; and in tropical and subtropical areas, for example in Colombia, Ecuador, the Congo Basin, Zambia, Tanzania, Madagascar, countries bordering the Tibet Autonomous Region, the Philippines, Bangladesh, India and New Zealand.

By contrast, the most resilient areas of the planet (very high or high Adaptation to Fire values and low or moderate Potential Soil Erosion) are in ~~the temperate broadleaf and mixed forests of northern Europe,~~ the boreal forests of Canada and Russia, Mediterranean forests, the woodlands and scrubs of southern Australia, and the temperate grasslands, savannas and shrublands of Euro-Asia.

3.3 Ecological Vulnerability to Fire Index

3.3.1 Spatial distribution

Figure 3 shows the Ecological Vulnerability to Fires values by ecoregion (from Ecological Vulnerability to Fire Index). These values were calculated by combining the delay in post-fire regeneration and the ecological indicator values. In other words, this map shows the intensity of potential damage and the capacity to regenerate after wildfires. The areas with the highest values are shown in dark red and correspond to those with significant Post-Fire Regeneration Delay values and high Ecological values. By contrast, the areas shown in lighter salmon tones correspond to ecoregions that are not particularly vulnerable to fire and would incur few potential ecological losses, since they have low Ecological and low Post-Fire Regeneration Delay values.

14

Ecological Vulnerability to Fire
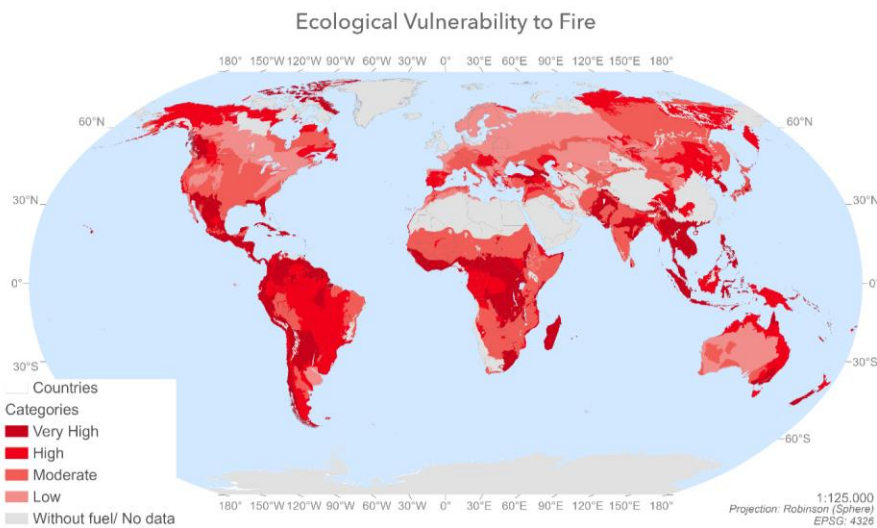
Figure 3: Spatial distribution of Ecological Vulnerability to Fire Index values calculated by combining the Post-Fire Regeneration Delay and the Ecological Indicators values by ecoregion.

Of the 647 ecoregions analysed, 246 had very high Vulnerability to Fire values, 155 had high values, 182 were moderately vulnerable and 77 had low values. The areas that would suffer the greatest ecological losses per biome in the event of fire are the temperate zones of British Columbia, the Himalayas, central China, California, Spain, South Africa, Florida, South Sudan, New Zealand, Mongolia, eastern Australia, Chile, Hungary, Romania, Croatia, Serbia, Italy and the Caucasus area; and tropical and subtropical areas such as Mexico, Central America, the Amazon Basin, Sierra Leone, Cameroon, Guinea, the Congo Basin, Paraguay, Argentina, Uruguay, Madagascar, Borneo, Sumatra, the Philippines, Namibia and northern Australia. The ecosystems of the Canadian and Russian boreal forests and the Bolivian and Chinese montane grasslands and shrublands are also vulnerable to fire.

3.3.2 Biome area assessment

Almost 50% of the ecoregions have either very high or high Ecological Vulnerability to Fire values (calculated by combining the Post-Fire Regeneration Delay and the Ecological indicators), while only 21% of ecoregions have low Ecological Vulnerability to Fire. This is due to an increase in the frequency and intensity of large Wildfires. The terrestrial biomes that contain most land in the very high and high Vulnerability categories as a proportion of their total area are: tropical and subtropical moist broadleaf forests, tundra, mangroves, tropical and subtropical coniferous forests, and tropical and subtropical dry broadleaf forests.

Within the very high Vulnerability to Fires category, the dominant terrestrial biomes are: tropical and subtropical moist forests, tropical and subtropical grasslands, savannas and shrublands, and xeric shrublands. By contrast, the least common biomes in this category are: wetlands, temperate grasslands, savannas and shrublands, and mangroves. Boreal forests do not have any areas with very high vulnerability values.

Of the 106,605,491 km^2 considered in this study (Table 11), the area classified as having very high vulnerability to fires consisted (from highest to lowest) of 7,611,385 km^2 of tropical and subtropical moist broadleaf forests,

15

5,905,304 km^2 of tropical and subtropical grasslands, savannas and shrublands, 1,980,099 km^2 of xeric shrublands, 1,593,959 km^2 of tropical and subtropical dry broadleaf forests, 1,300,302 km^2 of temperate broadleaf and mixed forests, 1,170,778 km^2 of temperate conifer forests, 1,053,305 km^2 of montane forests and shrublands, 556,032 km^2 of tundra, 524,545 km^2 of tropical and subtropical conifer forest, 172,422 km^2 of Mediterranean forests, woodlands and scrubs, 154,022 km^2 of mangroves, 87,651 km^2 of temperate grasslands, savannas and scrublands; and finally 25,131 km^2 of flooded grasslands and savannas.

By contrast, if we look at the different biomes (Table 11), the most vulnerable (from highest to lowest) are as follows: Tropical and subtropical moist coniferous forests with 75.07% of their area classified as very high vulnerability, mangroves with 59.61%, tropical and subtropical dry broadleaf forests with 53.08%, tropical and subtropical moist broadleaf forests with 41.82%, montane grasslands and shrublands with 33.83%, temperate conifer forests with 29.65%, tropical and subtropical grasslands, savannas and shrublands with 29.27%; xeric shrublands with 14.02%, tundra with 13.55%, temperate broadleaf and mixed forests with 12.22%, Mediterranean forests, woodlands and scrubs with 5.38%, flooded grasslands and savannas with 2.93% and, lastly, temperate grasslands, savannas and shrubs with 0.88%. None of the 'Boreal forests and taigas' biome falls within the very high vulnerability category, but around 20% of its area is classified as high vulnerability.

As regards the biomes with the lowest Vulnerability to Fire values as a proportion of their total area (Table 11), the temperate broadleaf and mixed forests stand out (44.85%) followed by boreal forests and taiga (41.37%), xeric shrublands (35.01%), and Mediterranean forests, woodlands and scrubs (31.85%). The mangroves biome is also worth highlighting in that its entire area is vulnerable to fire (Table 10).

465 **Table 11: Number of ecoregions, surface area and percentage of land ecologically vulnerable to fires.**

Biome	Percentage of total area studied by biome	Vulnerability Categories	Ecoregions	km²	Percentage per biome
Tropical & Subtropical Moist Broadleaf Forests	92.05	Very High	105	7611385	41.82
		High	55	8318171	45.70
		Moderate	20	1972358	10.84
		Low	3	300554	1.65
Tropical & Subtropical Dry Broadleaf Forests	99.77	Very High	28	1593959	53.08
		High	11	454328	15.13
		Moderate	9	929016	30.94
		Low	1	25432	0.85
Tropical & Subtropical Coniferous Forests	98.52	Very High	12	524545	75.07
		Moderate	2	174236	24.93
Temperate Broadleaf & Mixed Forests	82.96	Very High	17	1300302	12.22
		High	14	1600054	15.03
		Moderate	19	2970276	27.91
		Low	16	4773459	44.85
Temperate Conifer Forests	96.62	Very High	19	1170778	29.65
		High	4	558328	14.14
		Moderate	20	1369471	34.69
		Low	6	849432	21.52
Boreal Forests & Taiga	94.85	High	12	2753116	19.19
		Moderate	5	5659834	39.45
		Low	8	5935488	41.37
Tropical & Subtropical Grasslands, Savannas & Shrublands	100.00	Very High	14	5905304	29.27
		High	12	4217891	20.90
		Moderate	16	9362256	46.40
		Low	3	691856	3.43
Temperate Grasslands, Savannas & Shrublands	98.26	Very High	2	87651	0.88
		High	8	2631992	26.52
		Moderate	18	4622103	46.57
		Low	8	2584338	26.04
Flooded Grasslands & Savannas	78.70	Very High	2	25131	2.93
		High	4	425610	49.54
		Moderate	5	250872	29.20
		Low	3	157458	18.33
Montane Grasslands & Shrublands	60.01	Very High	16	1053305	33.83
		High	5	628994	20.20
		Moderate	14	1089028	34.98
		Low	2	341828	10.98
Tundra	35.20	Very High	2	556032	13.55
		High	11	2916345	71.09
		Moderate	3	385270	9.39
		Low	1	244865	5.97
Mediterranean Forests, Woodlands & Scrubs	99.47	Very High	3	172422	5.38
		High	5	624670	19.50
		Moderate	21	1385415	43.25
		Low	9	1020796	31.87
Xeric Shrublands	50.64	Very High	13	1980099	14.02
		High	8	882566	6.25
		Moderate	23	6314163	44.71
		Low	14	4944312	35.01
Mangroves	74.59	Very High	9	154022	59.61
		High	3	55773	21.58
		Moderate	4	48602	18.81
Total	78.85			106605491	

466 **3.4 Sensitivity analysis: OAT**

467 Table 11 shows the results of the sensitivity analysis called OAT carried out through the qualitative cross-
468 tabulation method between Ecological and Post-Fire Regeneration Delay Indicator in order to obtain the Ecological
469 Vulnerability to Fire Index. The categories of the Ecological Vulnerability to Fire Index that present the greatest
470 changes are: High, reaching higher numbers of ecoregions (+95) and Low, decreasing its number of ecoregions
471 considerably to 14 (-65). The number of stable ecoregion per category of Ecological Vulnerability to Fire (obtain
472 the same tag in the Ecological Vulnerability to Fire Index and then, in the OAT sensitivity method) that represent
473 ecoregion of reference are: 185 of Very High, 152, of High, 159 of Moderate and 14 of Low. The total of it reaching
474 510 ecoregion stables from the 647 ecoregion of this study (Fig. A6). Thus, the percentage of matches is 80.37%.
475

476 **Table 11: Accuracy of the model, number of ecoregions per category of Ecological Vulnerability to Fire**
477 **from the Index and Sensitivity method, and number of stable and net change of ecoregion between the Index**
478 **and Sensitivity method.**

Categories of Ecological Vulnerability	Number of ecoregion of the Ecological Vulnerability Index	Number of ecoregion of sensitivity of Ecological Vulnerability Index	Number of stable ecoregion per category of Ecological Vulnerability Index	Net change of ecoregion per category of Ecological Vulnerability Index
Very High	247	185	185	-62
High	194	289	152	95
Moderate	127	159	159	32
Low	79	14	14	-65
Total of ecoregions	647	647	520	-
Matches (%)		80,37		

479

480 **4 Discussion**

481 This study presents an index for assessing and mapping Ecological Vulnerability to Fire on a global scale on the
482 basis of Ecological Indicator and Post-Fire Regeneration Delay Indicator. Our results show that global ecological
483 value may be reduced by as much as 50% due to the perturbation by fire of ecosystems that are poorly adapted to
484 fire and have degraded fire regimes. The terrestrial biomes most affected are the tropical and subtropical moist
485 broadleaf forest, tundra, mangroves, tropical and subtropical coniferous forests, and tropical and subtropical dry
486 broadleaf forests. The most important determining factor is fire regime, in that current alterations to the fire regime
487 are causing areas that were previously considered to be relatively safe to now be classified as vulnerable to fire.
488 This study attempts to evaluate Ecological Vulnerability to Fire on a global scale. Although the databases used
489 were carefully examined before selection, the results are inevitably affected by the different spatial units, the lack
490 of information, the lack of updating and the uncertainty in the data for some ecoregions; and to a lesser extent, by
491 the way we combined the factors in the different indices.
492 In order to avoid problems with estimations of Species Richness, we used field data which measured this variable
493 exactly. In comparison with the use of remote sensing data, the study by Duro et al., 2007 shows that the Net

494 Primary Productivity (NPP) value overestimates biodiversity in areas covered by reforestations. This is because
495 forests made up of young trees or saplings, which fix more carbon than mature forests, are being overestimated.
496 In addition, the NPP biodiversity values are evaluated in terms of the number of different individuals and not in
497 terms of the number of different species, a fundamental indicator for establishing the biodiversity values of
498 particular environments (Nagendra & Rocchini, 2008).

499 As regards the ecosystem functionality variables, remote sensing data has the advantage of providing updated
500 information for the entire planet. Despite the extensive bibliographic review carried out as part of this research,
501 we were unable to find a concise way of combining these variables due to the fact that little research has been done
502 on the specific issue of ecosystem functionality. This is one of the first studies of ecological vulnerability to fire
503 that takes this issue into account, by integrating it into ecological value. This is of the utmost importance since fire
504 affects both the functioning of the ecosystem and its ability to maintain itself (Pausas & Ribeiro, 2017).

505 Our Ecological Vulnerability to Fire Index highlights those biomes considered most susceptible (tropical and
506 subtropical moist broadleaf forests, tundra, mangroves, tropical and subtropical coniferous forests, and tropical
507 and subtropical dry broadleaf forests) to suffering a decline in their ecological value. Two clusters can be observed.
508 The first consisted of mangroves and tropical and subtropical forests associated with tropical latitudes. These
509 regions obtained high or very high Ecological Vulnerability to Fire values due to the fact that they had the highest
510 ecological values and also had high regeneration delay values. Within the ecological value dimension of this index,
511 tropical latitudes show the highest values for both Biological Distinctiveness and Conservation Status due to the
512 fact that they host the highest ratios for biodiversity and endemisms, and have high ecosystem functionality values
513 and low levels of landscape degradation. They also have high levels of official protection. In addition, these areas
514 have the highest regeneration delay values due to the low adaptation capacity of the vegetation, the high current
515 alterations of the natural fire regime and the high potential soil erosion after fire disturbance. For this reason, if a
516 wildfire occurs in biomes such as mangroves, tropical and subtropical moist and broadleaf forests, and coniferous
517 forests, the ecological value of these biomes will almost certainly be heavily degraded due to the fact that most
518 areas within these biomes fall within the very high Ecological Vulnerability to Fire category of our index.

519 Second on this list of the biomes with the largest area with a high potential for degradation by fire is tundra, due
520 to the fact that it scores highly in both Ecological Value and Regeneration Delay, the two components of our
521 Vulnerability to Fire index. In terms of the first component, the intrinsic behaviour of the tundra biome explains
522 why it has similar ecological values to the biomes in the first cluster. However, the high levels of Regeneration
523 Delay have a different explanation. Within the Regeneration Delay Indicator, tundra has a fire regime in which the
524 vegetation is well adapted to fire due to the fact that, unlike the tropical and subtropical biomes, frequent fire
525 disturbance has been a constant feature of its development. In spite of this, tundra biomes have large areas in the
526 high or very high Vulnerability to Fire categories due to the fact that they score high values for potential soil
527 erosion and fire regime modification. As a result, pre-fire ecological values will be difficult to recover if the
528 wildfire occurs under a different regime than that to which the vegetation has adapted. This is why large swathes
529 of the tundra biome are classified within the very high Vulnerability to Fire category of our index.

530 In the end, both clusters meet the two requirements of our index for them to be considered highly vulnerable to
531 losing their pre-fire ecological values in the event of perturbation by fire: (i) high Ecological Indicator values and
532 (ii) high Regeneration Delay values. Within the Ecological Indicator, the factors which led the different ecoregions
533 to obtain high Ecological Indicator values are related to the ability of their ecosystems to host different kinds of

534 plants and wildlife (endemisms, functional and structural biodiversity) and the degree of official protection
535 afforded to them. For its part, the factor with the greatest impact on Regeneration Delay values is the alteration of
536 the fire regime, as this means that the strategies developed by the vegetation in response to fire are no longer fit
537 for purpose, and cannot help it recover the Ecological Indicator values existing prior to the fire. This is why
538 alteration of the fire regime is the most important factor and the most closely associated with human action in that
539 it is largely a consequence of human-induced global change. In this context, a determined shift towards more
540 sustainable lifestyles would reduce ecological vulnerability to fire.

541 In this sense, up to 50% of the terrestrial ecosystem analysed in this study is vulnerable to potential degradation of
542 its ecological value due to the changes taking place in fire regimes. This estimate coincides with the climate change
543 projections that indicate an increase in the frequency and intensity of large forest fires, recently dubbed
544 "megafires", as a result of longer, drier fire seasons (Stephens et al., 2013, Aponte et al., 2016). This increase, at
545 least in the medium term, will lead to new fire regimes and an increase in aridity in some regions as a consequence
546 of climate change (Flannigan et al., 2009). Terrestrial ecosystems will need to adapt not only to changes in mean
547 climatic variables, but also to greater variability with increased risk of extreme weather events, such as prolonged
548 droughts, storms, and floods (Lindner et al., 2010). As a result of this process of change, forest fires have become
549 one of the main environmental problems at a global scale today.

550 If we compare our evaluation of Ecological Vulnerability to Fire Index with the study carried out by Chuvieco et
551 al., (2014), substantial differences can be observed. Firstly, in our study the temperate conifer forests in the British
552 Columbia region had high vulnerability values compared to those estimated with their index. Lightning fires are
553 frequent in this area and the ecosystem has learnt to adapt to them. However, in our study, we included the
554 possibility of change in the fire regime, which indicates that these areas are in fact quite vulnerable to fire. Nitschke
555 & Innes (2013) found that due to climate change, fire regimes in boreal areas are changing in frequency and area.
556 If we look for example at the temperate broadleaf and mixed forests of Patagonia and the boreal forests of Alaska,
557 although both have adapted to fire to some extent, they also obtained high vulnerability to fire values, because of
558 the alteration in their fire regimes due to climate change, as indicated by Higuera et al., (2009) and Landesmann
559 et al., (2015).

560 If we turn our attention to the tropical and subtropical dry broadleaf forests of India, one of the greatest biodiversity
561 areas in the world, in the study by Chuvieco et al., (2014) they were considered to have low vulnerability to fire
562 because their plant communities had adapted to it. However, our study offers a different assessment, awarding
563 these parts of India higher Ecological Vulnerability to Fire values. This may be due to the fact that our model takes
564 into account a variable that characterizes the delay in post-fire regeneration as a result of changes in the fire regime.
565 In this sense, Kodandapani, Cochrane, & Sukumar (2008) refer to the fact that logging and forest fragmentation,
566 grazing and the collection of non-timber forest products are affecting the behaviour of fire in these forests.

567 In relation to the Amazon Basin, in this study the highest vulnerability to fire values only occur in the regions close
568 to the mouth. This may be due to the way in which the Species Richness variable is characterized. Species
569 Richness, adjusted in line with the size of the ecoregion, enables us to compare ecoregions of different sizes so as
570 to assess the ecological value fairly, rather than just comparing the raw data (Ricketts et al., 1999). It should be
571 noted that the areas near the coast, which have a more open plant canopy that allows sunlight to penetrate, have
572 managed to develop undergrowth vegetation that supports other forms of life (greater species richness understood
573 as diversity of species rather than abundance of species). In this case, it is important to realize that we are dealing

574 with tropical and subtropical moist broadleaf forests, which have not developed in the presence of fire. The
575 introduction of fire into these ecosystems could therefore result in significant losses in that plant species have
576 never developed post-fire regeneration strategies. This is why the small ecoregions at the mouth of the Amazon
577 suffer slightly greater losses due to fire, compared with the large central ecoregions (Cochrane & Laurance, 2002).
578 In addition, in the present study, the large temperate broadleaf and boreal forests of northern Europe and Russia
579 show less ecological vulnerability to fires than estimated by Chuvieco et al. (2014). This may be due to the fact
580 that our model, by following a representative criterion of estimating the ecological value within the biome, gives
581 higher species ratios to smaller regions, and less weight to the large ecoregions in northern Europe and Russia.
582 This is why, in our study, on a global scale, these ecoregions obtained a low vulnerability to fire value given that
583 to destroy all their ecological wealth, their entire immense area would have to be affected.
584 As for our index, despite the similarities and differences in the results with other studies, it has its own uncertainty
585 like all models. From the sensitivity analysis, it could be said that approximately 80% of the ecoregions evaluated
586 with the Ecological Vulnerability to Fire index would be considered robust. On the other hand, of the small changes
587 made, around 20% of the ecoregions would show uncertainty in the result of the index.
588 For example, some of them are located in Africa. Focusing on them, it is surprising to see Zambia and NE Angola
589 mapped with a very high Post-fire Regeneration Delay, especially considering how often they burn. Another
590 example would be that the most resilient areas on the planet (very high or high Fire Adaptation values and low or
591 moderate Potential Soil Erosion) are found in the temperate broadleaf and mixed forests of northern Europe when
592 fire is a rare event in these ecoregions and thus lack a history of fire-attuned evolution. Given the global scale, the
593 heterogeneity of the sources used and the extensive area that an ecoregion represents, sometimes the uncertainty
594 does not come from the integration method but from the prior uncertainty of the databases to be used (Richards &
595 Rowe, 1999). On the other hand, it should be noted that the use of the global scale gives us general information on
596 what is happening in order to detect points of controversy on which to proceed to a study at a local/regional scale
597 (Goodchild et al., 1993). Despite this, these uncertainties will be explored for future versions.
598 All integration methods, both quantitative and in our case qualitative (cross tabulation), show uncertainty in their
599 results, but as the literature points out, it is necessary to deal with it (Heuvelink, 1998; Heuvelink et al., 1989).
600 At various points in our study, we combined different factors to create an index. Although the model is based on
601 the bibliography, improvements such as multi-criteria evaluations involving expert participation could be applied
602 in the future in a bid to enrich the proposed approach in local/regional scales (Gómez-Delgado & Tarantola, 2006).
603 We could also apply machine-learning techniques to enable us to establish a more precise relationship between
604 the different factors (Semeraro et al., 2016). For all of the above, the resulting estimates should be interpreted as
605 an initial approximation.
606 Despite the aforementioned limitations, this study presents a robust, pragmatic and easily understood aggregation
607 methodology. The negative effects of fires can only be identified after the fire. This means that a model of
608 ecological vulnerability to fire cannot be correctly validated on a global scale as there is no representative sample
609 for doing so. However, at regional and local scales, there are studies that monitor post-fire ecological damage
610 (Gouveia et al., 2010). This is because the effects of fire can best be understood at these scales. As this methodology
611 can be replicated easily and the factors can be adapted to the model (to a greater or lesser extent depending on the
612 information available), the model could and indeed should be validated at these scales.

The ecological vulnerability model at a global scale is also very useful as it can help us to understand the global impacts that fires could have on ecosystems and on climate change. In addition, on a global scale, there are studies that focus on the early detection of places where fires may occur (based on climate data) (De Groot et al., 2006). If these studies were combined with our map, they could help prevent or mitigate ecological losses, as well as encourage the development of action plans in the event of fire, aimed at accelerating the regeneration of the ecosystem.

This model could also be used in the field of forest management to prioritize fire intervention areas in terms of ecological value, as proposed by Burgess et al. (2006) and Ricketts et al. (1999). If this vulnerability index were included in fire management plans, in the event of several fires breaking out at the same time, priority action could be directed at the most vulnerable area in order to protect its ecological value. Although in these cases, the protection of human lives is normally the first priority, future studies are expected to develop and integrate the idea of socioeconomic vulnerability into this ecological component of vulnerability. It would therefore seem more logical to develop policies, prevention and restoration plans in the most vulnerable areas in order to preserve them. Although this model for evaluating ecological vulnerability to fires on a global scale is an initial approximation, it allows us to identify which ecoregions of the different biomes are more likely to have their ecological value impaired by fire and why.

5 Conclusions

This paper makes an initial assessment of the spatial distribution of ecological vulnerability to fire on a global scale. The methodology we implemented enabled us to systematically integrate all the ecological components likely to be affected by forest fires. A novel aspect of this methodology is the way it integrates the variables in the biological distinction index, the characterization of functional diversity and the fact that it takes into account the impact of the alteration of the natural condition of the fire in post-fire regeneration delay. This index made it possible to identify the most susceptible biomes in terms of the loss of their ecological values, and it could be useful as a starting point for developing plans and strategies in response to global change scenarios.

At a global level, our results show that almost 50% of the world's land surface is vulnerable to a decline in its ecological value due to fire as a result of the current alteration of the fire regime. The terrestrial biomes with the highest degree of ecological vulnerability to fire were found in the tropical and subtropical moist broadleaf forests; tundra; mangroves; tropical and subtropical coniferous forests; and tropical and subtropical dry broadleaf forests. The greatest determining factor is the fire regime, a problem that is being exacerbated by current alterations, in that areas that were previously considered to be relatively safe now have much higher vulnerability values due to alterations in the natural condition of the fire, caused by global climate change.
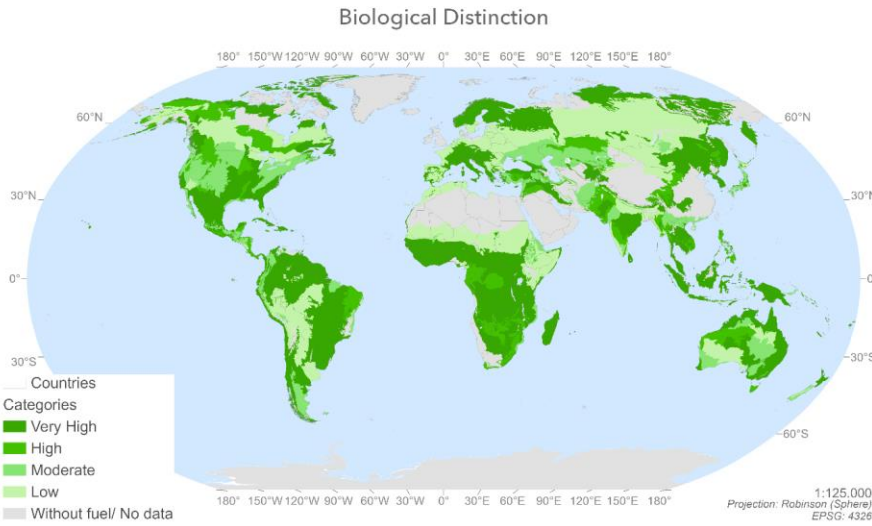
652

653

654

655

Appendix A: Maps of the study area, indicators and sensitivity method

In this section, we show the maps produced by the study area, indicators and sensitivity method (Fig A1-6).

658
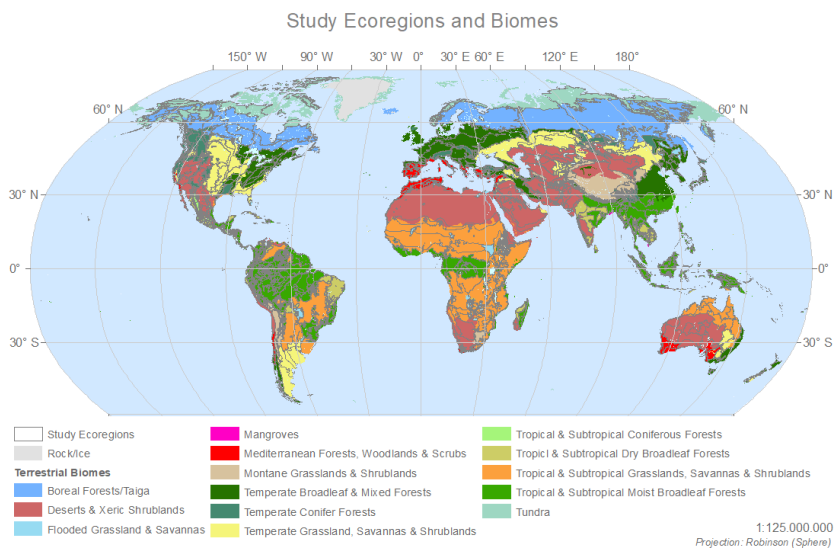
**Fig. A1: Terrestrial ecoregions within their respective biomes for this study. (Source: The authors).**

Biological Distinction

Fig. A2: Spatial distribution by ecoregion of the Ecosystem Biological Distinction Value prepared by combining the indices for Endemic Species, Species Richness, Functional Diversity and Unique Habitats by ecoregion evaluated within the biome to which they belong. (Source: The authors).
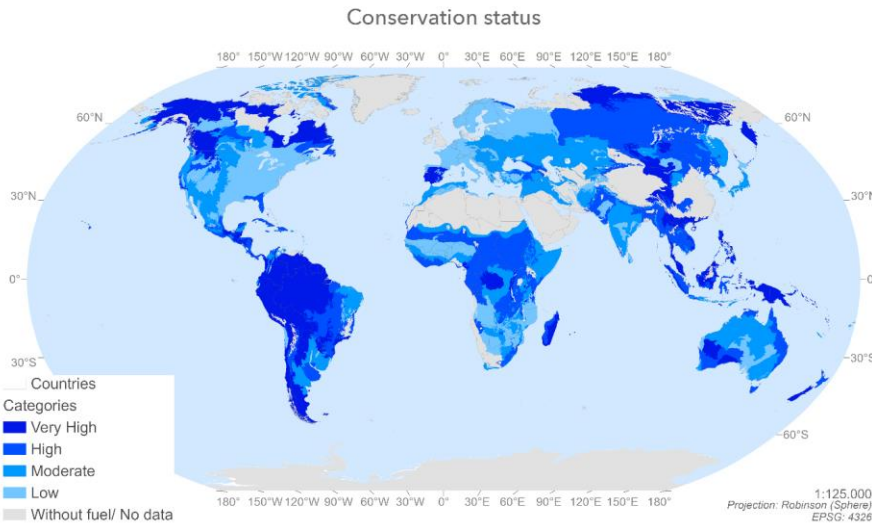
Conservation status

Fig. A3: Spatial distribution by ecoregion of the Ecosystem Conservation Status Value produced by combining the Indices for Unique Habitats Preservation, Intact Forest Landscapes, Degree of Fragmentation and Degree of Protection. (Source: the authors)
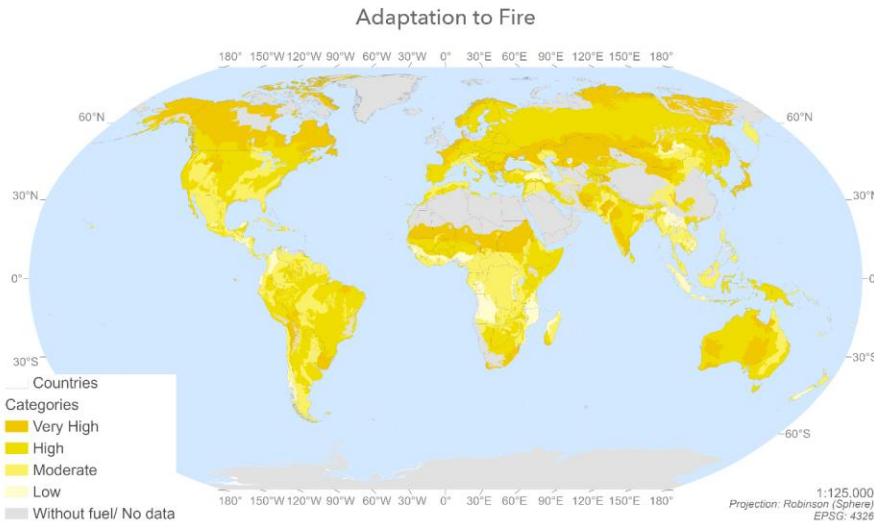
Adaptation to Fire

668

669 **Fig. A4: Spatial distribution by ecoregion of the Ecosystem Adaptation to Fire Value produced by**

670 **combining the Fire Regime and its degree of alteration. (Source: The authors).**

Potential Soil Erosion
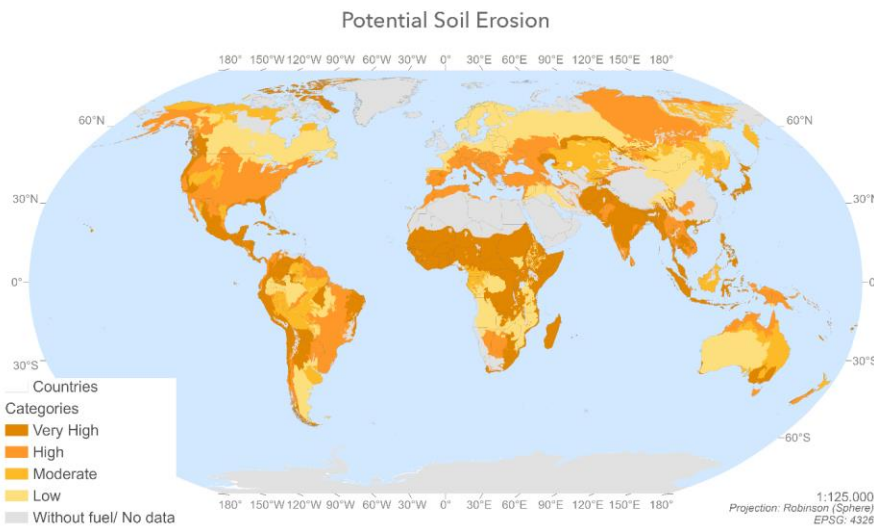
671

672 **Fig. A5: Spatial distribution of Potential Soil Erosion values by ecoregion resulting from the application of**

673 **the FAO criterion for water erosion. (Source: The authors).**

674

675

676

Changing ecoregions in the categories of Ecological Vulnerability to Fire Index
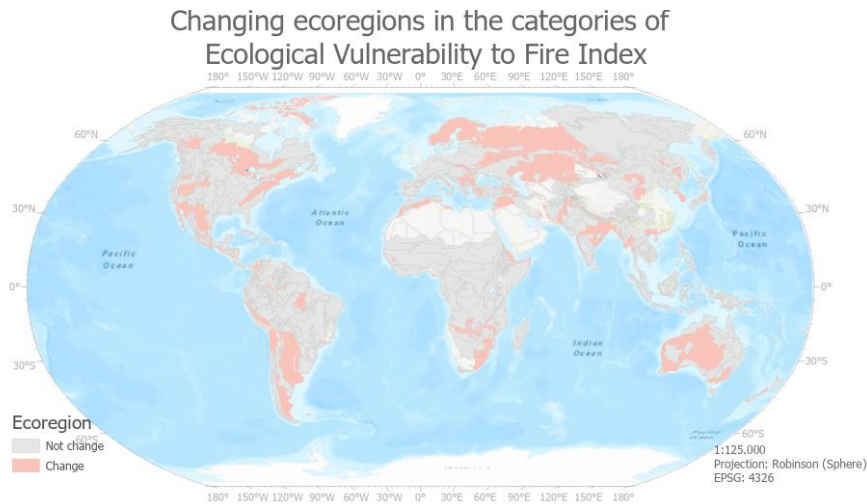
Fig. A6: Spatial distribution of changing ecoregions in the categories of Ecological Vulnerability to Fire Index resulted from the OAT analyses (sensitivity method). (Source: The authors).

Author contributions

Fátima Arrogante-Funes: Conceptualization, data curation, formal analysis, investigation, methodology, resources, software, validation, visualization, writing – original draft preparation, review & editing

Inmaculada Aguado: Conceptualization, funding acquisition, investigation, methodology, project administration, supervision, writing – review & editing.

Emilio Chuvieco: Conceptualization, funding acquisition, investigation, methodology, project administrator, resources, supervision, writing – review & editing.

References

Abson, D. J., Dougill, A. J., & Stringer, L. C. (2012). Using Principal Component Analysis for information-rich socio-ecological vulnerability mapping in Southern Africa. *Applied Geography*, *35*(1–2), 515–524. https://doi.org/10.1016/j.apgeog.2012.08.004

Alcasena, F. J., Salis, M., Nauslar, N. J., Aguinaga, A. E., & Vega-García, C. (2016). Quantifying economic losses from wildfires in black pine afforestations of northern Spain. *Forest Policy and Economics*, *73*, 153–167.

700   https://doi.org/10.1016/j.forpol.2016.09.005

701   Aretano, R., Semeraro, T., Petrosillo, I., De Marco, A., Pasimeni, M. R., & Zurlini, G. (2015). Mapping ecological

702   vulnerability to fire for effective conservation management of natural protected areas. *Ecological Modelling*,

703   *295*, 163–175. https://doi.org/10.1016/j.ecolmodel.2014.09.017

704   Arrogante-Funes, P., Bruzón, A. G., Arrogante-Funes, F., Ramos-Bernal, R. N., & Vázquez-Jiménez, R. (2021).

705   Integration of vulnerability and hazard factors for landslide risk assessment. *International Journal of*

706   *Environmental Research and Public Health*, *18*(22). https://doi.org/10.3390/ijerph182211987

707   Baeza, A. M. J., Valdecantos, A., Alloza, J. A., Vallejo, V. R., Centro, I., Ambientales, D. E., Darwin, C. C. R., &

708   Tecnoldgico, P. (2007). *Human Disturbance and Environmental Factors as Drivers of Long-Term Post-Fire*

709   *Regeneration Patterns in Mediterranean Forests Stable URL : http://www.jstor.org/stable/4499220*

710   *REFERENCES Linked references are available on JSTOR for this article : long-term . 18*(2), 243–252.

711   Bajocco, S., Salvati, L., & Ricotta, C. (2011). Land degradation versus fire: A spiral process? *Progress in Physical*

712   *Geography*, *35*(1), 3–18. https://doi.org/10.1177/0309133310380768

713   Barrio, M., Loureiro, M., & Chas, M. L. (2011). Aproximación a las pérdidas económicas ocasionadas a corto

714   plazo por los incendios forestales en Galicia en 2006. *Economía Agraria y Recursos Naturales*, *7*(14), 45.

715   https://doi.org/10.7201/earn.2007.14.03

716   Berry, P. M., Rounsevell, M. D. A., Harrison, P. A., & Audsley, E. (2006). Assessing the vulnerability of

717   agricultural land use and species to climate change and the role of policy in facilitating adaptation.

718   *Environmental Science and Policy*, *9*(2), 189–204. https://doi.org/10.1016/j.envsci.2005.11.004

719   Bond, W. J., Woodward, F. I., & Midgley, G. F. (2005). The global distribution of ecosystems in a world without

720   fire. *New Phytologist*, *165*(2), 525–538. https://doi.org/10.1111/j.1469-8137.2004.01252.x

721   Borrelli, P., Robinson, D. A., Fleischer, L. R., Lugato, E., Ballabio, C., Alewell, C., Meusburger, K., Modugno,

722   S., Schütt, B., Ferro, V., Bagarello, V., Oost, K. Van, Montanarella, L., & Panagos, P. (2017). An assessment

723   of the global impact of 21st century land use change on soil erosion. *Nature Communications*, *8*(1).

724   https://doi.org/10.1038/s41467-017-02142-7

725   Borrero, S., & Henao, F. (2017). Can managers be really objective? Bias in multicriteria decision analysis.

726   *Academy of Strategic Management Journal*, *16*(1), 244–260. https://www.abacademies.org/articles/can-

727   managers-be-really-objective-Bias-in-multicriteria-decision-analysis-1939-6104-16-1-114.pdf

728   Bowman, D. M. J. S., Balch, J. K., Artaxo, P., Bond, W. J., Carlson, J. M., Cochrane, M. A., D'Antonio, C. M.,

729   DeFries, R. S., Doyle, J. C., Harrison, S. P., Johnston, F. H., Keeley, J. E., Krawchuk, M. A., Kull, C. A.,

730   Marston, J. B., Moritz, M. A., Prentice, I. C., Roos, C. I., Scott, A. C., … Pyne, S. J. (2009). Fire in the earth

731   system. *Science*, *324*(5926), 481–484. https://doi.org/10.1126/science.1163886

732   Buhk, C., Meyn, A., & Jentsch, A. (2007). The challenge of plant regeneration after fire in the Mediterranean

733   Basin: Scientific gaps in our knowledge on plant strategies and evolution of traits. *Plant Ecology*, *192*(1),

734   1–19. https://doi.org/10.1007/s11258-006-9224-2

735   Burgess, N., de Lima, M. G., N., B., Blythe, S., Tompkins, I., Coad, L., & Leverington, F. (2014). Protected Area

736   Management Effectiveness in WWF Global Priority Places. *Cambridge: UNEP-WCMC*.

737   Burgess, N., Hales, J. D. A., Ricketts, T. H., & Dinerstein, E. (2006). Factoring species, non-species values and

738   threats into biodiversity prioritisation across the ecoregions of Africa and its islands. *Biological*

739   *Conservation*, *127*(4), 383–401. https://doi.org/10.1016/j.biocon.2005.08.018

27

740 Caprio, A. C., & Graber, D. M. (2000). *Returning fire to the mountains: can we successfully restore the ecological*
741 *role of pre-Euroamerican fire regimes to the Sierra Nevada* (D. Cole, S. McCool, & J. O'Loughlin (eds.)).
742 In Wilderness science in a time of change conference.

743 Chuvieco, E., Aguado, I., Yebra, M., Nieto, H., Martín, M. P., Vilar, L., Martínez, J., Padrón, D., Martín, S., &
744 Salas, J. (2007). Cartografía del peligro de incendios forestales mediante Teledetección y SIG. *Teledetección*
745 *- Hacia Un Mejor Entendimiento de La Dinámica Global y Regional*, *January*, 19–26.

746 Chuvieco, E., Aguado, I., Yebra, M., Nieto, H., Salas, J., Martín, M. P., Vilar, L., Martínez, J., Martín, S., Ibarra,
747 P., de la Riva, J., Baeza, J., Rodríguez, F., Molina, J. R., Herrera, M. A., & Zamora, R. (2010). Development
748 of a framework for fire risk assessment using remote sensing and geographic information system
749 technologies. *Ecological Modelling*, *221*(1), 46–58. https://doi.org/10.1016/j.ecolmodel.2008.11.017

750 Chuvieco, E., Martínez, S., Román, M. V., Hantson, S., & Pettinari, M. L. (2014). Integration of ecological and
751 socio-economic factors to assess global vulnerability to wildfire. *Global Ecology and Biogeography*, *23*(2),
752 245–258. https://doi.org/10.1111/geb.12095

753 Cinner, J. E., McClanahan, T. R., Graham, N. A. J., Daw, T. M., Maina, J., Stead, S. M., Wamukota, A., Brown,
754 K., & Bodin, O. (2012). Vulnerability of coastal communities to key impacts of climate change on coral reef
755 fisheries. *Global Environmental Change*, *22*(1), 12–20. https://doi.org/10.1016/j.gloenvcha.2011.09.018

756 Clavijo, A. U., Delgado, M. G., & González, P. B. (2019). Análisis de Sensibilidad aplicado a modelos de
757 crecimiento urbano basados en autómatas celulares de estructura irregular. *Cuadernos Geográficos*, *58*, 326–
758 348.

759 Cochrane, M. A., & Laurance, W. F. (2002). Fire as a large-scale edge effect in Amazonian forests. *Journal of*
760 *Tropical Ecology*, *18*(3), 311–325. https://doi.org/10.1017/S0266467402002237

761 Collinge, S. K. (1996). Ecological consequences of habitat fragmentation: Implications for landscape architecture
762 and planning. *Landscape and Urban Planning*, *36*(1), 59–77. https://doi.org/10.1016/S0169-
763 2046(96)00341-6

764 Cutter, S. L., Boruff, B. J., & Shirley, W. L. (2003). Social vulnerability to environmental hazards. *Social Science*
765 *Quarterly*, *84*(2), 242–261. https://doi.org/10.1111/1540-6237.8402002

766 De Groot, W. J. B., Goldammer, J. G., Keenan, T., Brady, M. A., Lynham, T. J., Justice, C. O., Csiszar, I. A., &
767 O'Loughlin, K. (2006). Developing a global early warning system for wildland fire. *V International*
768 *Conference on Forest Fire Research*. https://doi.org/10.1016/j.foreco.2006.08.025

769 Díaz-Delgado, R., Lloret, F., Pons, X., & Terradas, J. (2002). Satellite evidence of decreasing resilience in
770 mediterranean plant communities after recurrent wildfires. *Ecology*, *83*(8), 2293–2303.
771 https://doi.org/10.1890/0012-9658(2002)083[2293:SEODRI]2.0.CO;2

772 Dinerstein, E., Olson, D., Graham, D., Webster, A., Pimm, S., Bookbinder, M., & Ledec, G. (1995). *A*
773 *Conservation Assessment of the Terrestrial Ecoregions of Latin America and the Caribbean.* World Bank.

774 Duguy, B., Alloza, J. A., Baeza, M. J., De La Riva, J., Echeverría, M., Ibarra, P., Llovet, J., Cabello, F. P., Rovira,
775 P., & Vallejo, R. V. (2012). Modelling the ecological vulnerability to forest fires in mediterranean
776 ecosystems using geographic information technologies. *Environmental Management*, *50*(6), 1012–1026.
777 https://doi.org/10.1007/s00267-012-9933-3

778 Duguy, B., & Vallejo, V. R. (2008). Land-use and fire history effects on post-fire vegetation dynamics in eastern
779 Spain. *Journal of Vegetation Science*, *19*(1), 97–108.

780   Duro, D. C., Coops, N. C., Wulder, M. A., & Han, T. (2007). Development of a large area biodiversity monitoring
781   system driven by remote sensing. *Progress in Physical Geography*, *31*(3), 235–260.
782   https://doi.org/10.1177/0309133307079054

783   El Gibari, S., Gómez, T., & Ruiz, F. (2019). Building composite indicators using multicriteria methods: a review.
784   *Journal of Business Economics*, *89*(1), 1–24. https://doi.org/10.1007/s11573-018-0902-z

785   FAO/UNEP/UNESCO. (1979). A provisional methodology for soil degradation assessment. *Food and
786   Agricultural Organization of the United Nations, Rome.*

787   Flannigan, M. D., Krawchuk, M. A., De Groot, W. J., Wotton, B. M., & Gowman, L. M. (2009). Implications of
788   changing climate for global wildland fire. *International Journal of Wildland Fire*, *18*(5), 483–507.
789   https://doi.org/10.1071/WF08187

790   Freudenberger, L., Hobson, P. R., Schluck, M., & Ibisch, P. L. (2012). A global map of the functionality of
791   terrestrial ecosystems. *Ecological Complexity*, *12*, 13–22. https://doi.org/10.1016/j.ecocom.2012.08.002

792   Giovannini, G., & Lucchesi, S. (1997). Modifications induced in soil physico-chemical parameters by
793   experimental fires at different intensities. *Soil Science*, *162*(7), 479–486.

794   Gómez-Delgado, M., & Tarantola, S. (2006). GLOBAL sensitivity analysis, GIS and multi-criteria evaluation for
795   a sustainable planning of a hazardous waste disposal site in Spain. *International Journal of Geographical
796   Information Science*, *20*(4), 449–466. https://doi.org/10.1080/13658810600607709

797   González, J. R., Kolehmainen, O., & Pukkala, T. (2007). Using expert knowledge to model forest stand
798   vulnerability to fire. *Computers and Electronics in Agriculture*, *55*(2), 107–114.
799   https://doi.org/10.1016/j.compag.2006.12.005

800   Gonzalez, P. B., Aguilera-Benavente, F., & Gomez-Delgado, M. (2015). Partial validation of cellular automata
801   based model simulations of urban growth: An approach to assessing factor influence using spatial methods.
802   *Environmental Modelling & Software*, *69*. https://doi.org/10.1016/j.envsoft.2015.03.008

803   Goodchild, M. F., Parks, B. O., & Steyaert, L. T. (1993). Environmental Modelling with GIS. *Environmental
804   Modelling with GIS. Oxford University Press, New York,* 318–331.

805   Gouveia, C., DaCamara, C. C., & Trigo, R. M. (2010). Post-fire vegetation recovery in Portugal based on
806   spot/vegetation data. *Natural Hazards and Earth System Science*, *10*(4), 673–684.
807   https://doi.org/10.5194/nhess-10-673-2010

808   Guyette, R. P., Muzika, R. M., & Dey, D. C. (2002). Dynamics of an Anthropogenic Fire Regime. *Ecosystems*,
809   *6*(4), 326–335. https://doi.org/10.1007/s10021-002-0115-7

810   Hämäläinen, R. P., & Alaja, S. (2008). The threat of weighting biases in environmental decision analysis.
811   *Ecological Economics*, *68*(1–2), 556–569. https://doi.org/10.1016/j.ecolecon.2008.05.025

812   Heuvelink, G. B. (1998). Error propagation in environmental modelling with GIS. *CRC Press*.

813   Heuvelink, G. B., Burrough, P. A., & Stein, A. (1989). Propagation of errors in spatial modelling with GIS.
814   *International Journal of Geographical Information System*, *3(4)*, 303–322.

815   Higuera, P. E., Brubaker, L. B., Anderson, P. M., Hu, F. S., & Brown, T. A. (2009). Vegetation mediated the
816   impacts of postglacial climate change on fire regimes in the south-central Brooks Range, Alaska. *Ecological
817   Monographs*, *79*(2), 201–219. https://doi.org/10.1890/07-2019.1

818   Hilton-Taylor, C. (2000). *2000 IUCN Red List of Threatened Species*. IUCN, Gland, Switzerland and Cambridge,
819   UK.

820 Hobson, K. A., & Schieck, J. (1999). Changes in bird communities in boreal mixedwood forest: Harvest and
821   wildfire effects over 30 years. *Ecological Applications*, *9*(3), 849–863. https://doi.org/10.1890/1051-
822   0761(1999)009[0849:CIBCIB]2.0.CO;2

823 Hoekstra, J. M., Molnar, J. L., Jennings, M., Revenga, C., Spalding, M. D., Boucher, T. M., Robertson, J. C.,
824   Heibel, T. J., & Ellison, K. (2010). The Atlas of global conservation: changes, challenges, and opportunities
825   to make a difference. In *Choice Reviews Online* (Vol. 48, Issue 01). J. L. Molnar. Berkeley: University of
826   California Press. https://doi.org/10.5860/choice.48-0280

827 Houghton, J. T., Ding, Y. D. J. G., Griggs, D. J., Noguer, M., van der Linden, P. J. Dai, X., & Johnson, C. A.
828   (2001). *Climate change 2001: the scientific basis*. The Press Syndicate of the University of Cambridge.

829 Ippolito, A., Sala, S., Faber, J. H., & Vighi, M. (2010). Ecological vulnerability analysis: A river basin case study.
830   *Science of the Total Environment*, *408*(18), 3880–3890. https://doi.org/10.1016/j.scitotenv.2009.10.002

831 IUCN, & UNEP-WCMC. (2020). *The World Database on Protected Areas (WDPA)*. Ambridge (UK): UNEP
832   World Conservation Monitoring Centre.

833 Kier, G., Kreft, H., Tien, M. L., Jetz, W., Ibisch, P. L., Nowicki, C., Mutke, J., & Barthlott, W. (2009). A global
834   assessment of endemism and species richness across island and mainland regions. *Proceedings of the
835   National Academy of Sciences of the United States of America*, *106*(23), 9322–9327.
836   https://doi.org/10.1073/pnas.0810306106

837 Kier, G., Mutke, J., Dinerstein, E., Ricketts, T. H., Küper, W., Kreft, H., & Barthlott, W. (2005). Global patterns
838   of plant diversity and floristic knowledge. *Journal of Biogeography*, *32*(7), 1107–1116.
839   https://doi.org/10.1111/j.1365-2699.2005.01272.x

840 Kirkman, L. K., Mitchell, R. J., Helton, R. C., & Drew, M. B. (2001). Productivity and species richness across an
841   environmental gradient in a fire-dependent ecosystem. *American Journal of Botany*, *88*(11), 2119–2128.
842   https://doi.org/10.2307/3558437

843 Kodandapani, N., Cochrane, M. A., & Sukumar, R. (2008). A comparative analysis of spatial, temporal, and
844   ecological characteristics of forest fires in seasonally dry tropical ecosystems in the Western Ghats, India.
845   *Forest Ecology and Management*, *256*(4), 607–617. https://doi.org/10.1016/j.foreco.2008.05.006

846 Krinner, G., Viovy, N., de Noblet-Ducoudré, N., Ogée, J., Polcher, J., Friedlingstein, P., Ciais, P., Sitch, S., &
847   Prentice, I. C. (2005). A dynamic global vegetation model for studies of the coupled atmosphere-biosphere
848   system. *Global Biogeochemical Cycles*, *19*(1), 1–33. https://doi.org/10.1029/2003GB002199

849 Landesmann, J. B., Gowda, J. H., Garibaldi, L. A., & Kitzberger, T. (2015). Survival, growth and vulnerability to
850   drought in fire refuges: implications for the persistence of a fire-sensitive conifer in northern Patagonia.
851   *Oecologia*, *179*(4), 1111–1122. https://doi.org/10.1007/s00442-015-3431-2

852 Lindner, M., Maroschek, M., Netherer, S., Kremer, A., Barbati, A., Garcia-Gonzalo, J., Seidl, R., Delzon, S.,
853   Corona, P., Kolström, M., Lexer, M. J., & Marchetti, M. (2010). Climate change impacts, adaptive capacity,
854   and vulnerability of European forest ecosystems. *Forest Ecology and Management*, *259*(4), 698–709.
855   https://doi.org/10.1016/j.foreco.2009.09.023

856 Mace, G. M., & Lande, R. (1991). *Assessing extinction threats: towards a reassessment of IUCN endangered
857   species categories.* Conservation Biology 5.

858 Martínez Vega, J., Romero Calcerrada, R., & Echavarría, P. (2007). Valoración paisajística y ecológica de la
859   Comunidad de Madrid: su integración en un índice sintético de riesgo de incendios forestales. *Revista de

860   *Teledetección*, *28*(April 2016), 43–60.

861   Midgley, G. F., & Bond, W. J. (2015). Future of African terrestrial biodiversity and ecosystems under
862   anthropogenic climate change. *Nature Climate Change*, *5*(9), 823–829.
863   https://doi.org/10.1038/nclimate2753

864   Moreira, F., Viedma, O., Arianoutsou, M., Curt, T., Koutsias, N., Rigolot, E., Barbati, A., Corona, P., Vaz, P.,
865   Xanthopoulos, G., Mouillot, F., & Bilgili, E. (2011). Landscape - wildfire interactions in southern Europe:
866   Implications for landscape management. *Journal of Environmental Management*, *92*(10), 2389–2402.
867   https://doi.org/10.1016/j.jenvman.2011.06.028

868   Moreno-Martínez, Á., Camps-Valls, G., Kattge, J., Robinson, N., Reichstein, M., van Bodegom, P., Kramer, K.,
869   Cornelissen, J. H. C., Reich, P., Bahn, M., Niinemets, Ü., Peñuelas, J., Craine, J. M., Cerabolini, B. E. L.,
870   Minden, V., Laughlin, D. C., Sack, L., Allred, B., Baraloto, C., … Running, S. W. (2018). A methodology
871   to derive global maps of leaf traits using remote sensing and climate data. *Remote Sensing of Environment*,
872   *218*(September), 69–88. https://doi.org/10.1016/j.rse.2018.09.006

873   Moreno, A., & Becken, S. (2009). A climate change vulnerability assessment methodology for coastal tourism.
874   *Journal of Sustainable Tourism*, *17*(4), 473–488. https://doi.org/10.1080/09669580802651681

875   Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A., & Kent, J. (2010). Biodiversity hotspots for
876   conservation priorities. *Nature*, *468*(7326), 895. https://doi.org/10.1038/468895a

877   Nagendra, H., & Rocchini, D. (2008). High resolution satellite imagery for tropical biodiversity studies: The devil
878   is in the detail. *Biodiversity and Conservation*, *17*(14), 3431–3442. https://doi.org/10.1007/s10531-008-
879   9479-0

880   Nitschke, C. R., & Innes, J. L. (2013). Potential effect of climate change on observed fire regimes in the Cordilleran
881   forests of South-Central Interior, British Columbia. *Climatic Change*, *116*(3–4), 579–591.
882   https://doi.org/10.1007/s10584-012-0522-5

883   Olson, D. M., & Dinerstein, E. (2002). The Global 200: Priority Ecoregions for Global Conservation. *Annals of*
884   *the Missouri Botanical Garden*, *89*(2), 199. https://doi.org/10.2307/3298564

885   Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C.,
886   D'amico, J. A., Itoua, I., Strand, H. E., Morrison, J. C., Loucks, C. J., Allnutt, T. F., Ricketts, T. H., Kura,
887   Y., Lamoreux, J. F., Wettengel, W. W., Hedao, P., & Kassem, K. R. (2001). Terrestrial Ecoregions of the
888   World: A New Map of Life on Earth. *BioScience*, *51*(11), 933. https://doi.org/10.1641/0006-
889   3568(2001)051[0933:teotwa]2.0.co;2

890   Pausas, J. G., Carreras, J., Ferré, A., & Font, X. (2003). Coarse-scale plant species richness in relation to
891   environmental heterogeneity. *Journal of Vegetation Science*, *14*(5), 661–668.
892   https://doi.org/10.1111/j.1654-1103.2003.tb02198.x

893   Pausas, J. G., & Ribeiro, E. (2017). Fire and plant diversity at the global scale. *Global Ecology and Biogeography*,
894   *26*(8), 889–897. https://doi.org/10.1111/geb.12596

895   Pereira, H. M., Rosa, I. M. D., Martins, I. S., & Al., E. (2020). Supplementary Materials for Global trends in
896   biodiversity and ecosystem services from 1900 to 2050. *Science*, *1*, 1–5.

897   Poos, M. S., Walker, S. C., & Jackson, D. A. (2009). Functional-diversity indices can be driven by methodological
898   choices and species richness. *Ecology*, *90*(2), 341–347. https://doi.org/10.1890/08-1638.1

899   Potapov, P., Yaroshenko, A., Turubanova, S., Dubinin, M., Laestadius, L., Thies, C., Aksenov, D., Egorov, A.,

900   Yesipova, Y., Glushkov, I., Karpachevskiy, M., Kostikova, A., Manisha, A., Tsybikova, E., & Zhuravleva,
901   I. (2008). Mapping the world's intact forest landscapes by remote sensing. *Ecology and Society*, *13*(2).
902   https://doi.org/10.5751/ES-02670-130251

903   Preston, B. L., Yuen, E. J., & Westaway, R. M. (2011). Putting vulnerability to climate change on the map: A
904   review   of   approaches,   benefits,   and   risks.   *Sustainability   Science*,   *6*(2),   177–202.
905   https://doi.org/10.1007/s11625-011-0129-1

906   Richards, D., & Rowe, W. D. (1999). Decision-making with heterogeneous sources of information. *Risk Analysis*,
907   *19(1)*, 69–81.

908   Ricketts, T.H., Dinerstein, E., Olson, D., Loucks, C. J., Eichbaum, W., DellaSala, D., Kavanagh, K., Hedao, P.,
909   Hurley, P. T., Carney, K. M., Abell, R., & Walters, S. (1999). *Terrestrial Ecoregions of North America: A
910   Conservation Assessment.* (b). Island Press.

911   Ricketts, Taylor H, Dinerstein, E., Olson, D. M., & Loucks, C. (1999). Who's Where in North Patterns of species
912   richness and the utility of indicator taxa. *Bioscience*, *May*.

913   Rosenzweig, M. L. (1995). *Species Diversity in Space and Time*. Cambridge University Press, Cambridge, UK.

914   Saltelli, A., Chan, K., & Scott, E. . (2000). Sensitivity Analysis. *Chichester, John Wiley & Sons, LTD*.

915   Scott, D. F., & Van Wyk, D. B. (1990). The effects of wildfire on soil wettability and hydrological behaviour of
916   an   afforested   catchment.   *Journal   of   Hydrology*,   *121*(1–4),   239–256.   https://doi.org/10.1016/0022-
917   1694(90)90234-O

918   Semeraro, T., Mastroleo, G., Aretano, R., Facchinetti, G., Zurlini, G., & Petrosillo, I. (2016). GIS Fuzzy Expert
919   System for the assessment of ecosystems vulnerability to fire in managing Mediterranean natural protected
920   areas. *Journal of Environmental Management*, *168*, 94–103. https://doi.org/10.1016/j.jenvman.2015.11.053

921   Shlisky, A., Waugh, J., Gonzalez, P., Gonzalez, M., Manta, M., Santoso, H., Alvarado, E., Ainuddin, A.,
922   Rodríguez-trejo, D. A., Swaty, R., Schmidt, D., Kaufmann, M., Myers, R., Alencar, A., Kearns, F., Johnson,
923   D., Smith, J., & Zollner, D. (2007). Fire, ecosystems and people : threats and strategies for global biodiversity
924   conservation.   *The   Nature   Conservancy   Global   Fire   Initiative   Technical   Report*,   17.
925   http://mrcc.isws.illinois.edu/living_wx/wildfires/fire_ecosystems_and_people.pdf

926   Turner, B. L., Kasperson, R. E., Matsone, P. A., McCarthy, J. J., Corell, R. W., Christensene, L., Eckley, N.,
927   Kasperson, J. X., Luers, A., Martello, M. L., Polsky, C., Pulsipher, A., & Schiller, A. (2003). A framework
928   for vulnerability analysis in sustainability science. *Proceedings of the National Academy of Sciences of the
929   United States of America*, *100*(14), 8074–8079. https://doi.org/10.1073/pnas.1231335100

930   UNISDIR, U. (2009). Making Disaster Risk Reduction. *Sensitive: Policy and Practical Guidelines*.

931   Williams, L. R. R., & Kapustka, L. A. (2000). Ecosystem vulnerability: A complex interface with technical
932   components.   *Environmental   Toxicology   and   Chemistry*,   *19*(4   II),   1055–1058.
933   https://doi.org/10.1002/etc.5620190435

934   World Wildlife Fund. (2006). *WildFinder: Online database of species distributions*.

935

936